

Freshwaters of New Zealand

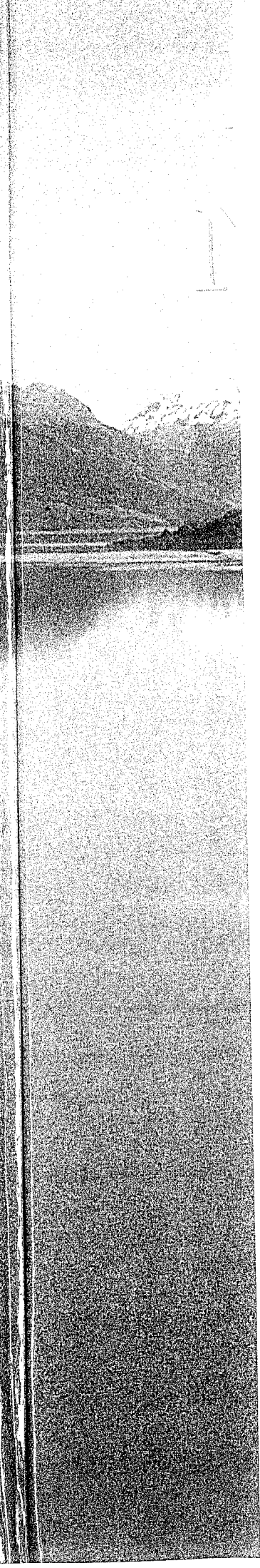
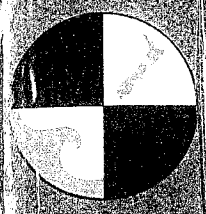


EDITED BY
NEW ZEALAND ENTOMOLOGICAL SOCIETY
AND
NEW ZEALAND LIMNOLOGICAL SOCIETY
IN CO-OPERATION WITH THE
DEPARTMENT OF LAND AND FORESTRY

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Freshwaters of New Zealand

HARDING
MOSLEY
PEARSON
SORRELL



**Freshwaters of
New Zealand**

Front cover: Lake Wakatipu (Paul Mosley)

Back cover: Purakaunui Falls (Jon Harding)

Freshwaters of New Zealand

2004

Edited for the

NEW ZEALAND HYDROLOGICAL SOCIETY AND
NEW ZEALAND LIMNOLOGICAL SOCIETY

by

JON HARDING, PAUL MOSLEY
CHARLES PEARSON AND BRIAN SORRELL

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ISBN 0-476-00708-9

Published by:

NEW ZEALAND HYDROLOGICAL SOCIETY INC.
NEW ZEALAND LIMNOLOGICAL SOCIETY INC.

Copy editing and layout by:

EILEEN McSAVENY
JO DICKSON

Printed by:

THE CAXTON PRESS, CHRISTCHURCH, NEW ZEALAND

Freshwaters of New Zealand

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ACKNOWLEDGEMENTS

Freshwaters of New Zealand has been a major exercise to which many people have contributed. Above all, we acknowledge the contributions of the eighty authors in bringing together this 764 page “state of the art” book on freshwater science and management. Many of the authors accepted the additional task of providing peer reviews of other chapters, a task shared by other colleagues too numerous to name. For this essential contribution to ensuring the accuracy of the book, we express our appreciation. Several people provided particular assistance to the authors of individual chapters; we thank Amanda Baldwin, Paul Broady, Peter Carran, Paul Champion, John Clayton, Erica Dalziell, Chris Daughney, Karen Denyer, Peter Kingsbury, Frank McGuire, Alistair McKerchar, Mark Meleason, Rhonda Mitchell, Alan Nicholson, Phil Novis, Leanne O’Brien, Tony Phipps, Bob Reid, Peter Russell, Eugenie Sage, Ude Shankar, Aleksey Sidorchuk, Joanna Sim, Ramon Strong, Judy van Rossem, Brendon Ward, John Waugh, Mike Winterbourn and John Youngson. We also record our thanks to Jo Dickson and Eileen McSaveney for the huge amount of effort they have put into producing a harmonious, clear and attractive publication, and to Jenny White for her assistance with the Indexes.

All these contributors have taken on a task that has added to their workloads or eaten into their spare time; we believe that the end result testifies to their hard work and commitment to making current knowledge about New Zealand’s freshwaters readily available to a wide audience.

Finally, we acknowledge the support and resources provided by the authors’ employers – tertiary educational institutions, crown research institutes, government departments, consulting companies. Many funding agencies have supported the science described in the book. We especially recognise the New Zealand Foundation for Research, Science and Technology, and the authors of Chapter 30 express their thanks to the Government of the Basque Country for support.

CITATION

We suggest that this publication be referenced as follows:

Harding J.S.; Mosley, M.P.; Pearson, C.P.; and Sorrell, B.K. (editors), 2004. *Freshwaters of New Zealand*. New Zealand Hydrological Society Inc. and New Zealand Limnological Society Inc., Christchurch, New Zealand. 764 pp. ISBN 0-476-00708-9.

FOREWORD

In his Millennium Report¹ United Nations Secretary-General Kofi Annan made the case to confront the growing fresh water crisis and build a new ethic in global stewardship. He said: “To arrest the unsustainable exploitation of water resources, we require water management strategies at national levels and local levels. They should include pricing structures that promote both equity and efficiency. We need a ‘blue revolution’ in agriculture that focuses on increased productivity per unit of water—‘more crop per drop’—together with far better watershed and flood plain management... but none of this will happen without public awareness and mobilisation campaigns, to bring home to people the extent and causes of current and impending water crises.”

In July 2002 Maurice Strong, former Secretary-General of the 1992 Earth Summit, addressing a US Senate Environmental Treaty Implementation Review, said: “Most of the changes we must make are in our economic life. The systems of taxes, subsidies, regulations and policies through which governments motivate the behaviour of individuals and corporations continue to incent unsustainable behaviours.”

These two quotes take us to the heart of water sustainability challenges in New Zealand. We must acknowledge (and largely do) that our fresh water resources are limited in many areas. We must purposefully build our waters knowledge and maintain our research capacities. We must have national and local strategies and processes for environmentally sustainable allocation and effective management. We must value and price water resources to reflect ecological, economic and cultural roles plus growing scarcity. And, finally, we must acknowledge the inevitable tensions in our tax frameworks, legislation and policies—tensions which are drivers of unsustainable activities and systems.

Freshwaters in New Zealand makes a timely contribution to our current knowledge of freshwaters in Aotearoa and, most importantly, puts a spotlight on the many deficiencies in our current knowledge ‘pool’. Of course there are always opportunities to learn more, to push the boundaries of knowledge. However in the case of our waters knowledge, there are some big gaps that can equate to big risks for our biotic economy, our unique ecologies, our cultural heritage, and our predominantly urban citizens. It is very hard to sustain such a vital resource without sufficient knowledge of the resource and the effects our demands have on it in terms of quantity and quality. Conversely, the better the understanding of a core resource such as fresh water, the greater the likelihood its true value will be widely acknowledged and a shared vision for its management developed.

Given the growing concerns about water quality in Lake Taupo and the Waikato River, water allocation on the Canterbury plains and in the Waitaki valley, and water availability on the Kapiti coast, improved understanding is urgently needed on many fronts. Some of those fronts are beyond the realms of the physical sciences, being deeply social, cultural, economic and political matters. These other ‘sciences’ are crucial, and it’s heartening to see some of them addressed in this book. In the context established by Annan and Strong, this interlinking of research into the physical, social, cultural, economic and political realms is essential to the sustainable management of our most precious resource—fresh water.

I congratulate the editors and the many authors on their great contribution to fresh water enlightenment in New Zealand—well done!

J. Morgan Williams

Parliamentary Commissioner for the Environment

¹ United Nations 2000

Glossary

- Abiotic** non-living, usually referring to physical or chemical conditions.
- Ablation** processes that remove snow or ice from a glacier or snow field, e.g., melting, sublimation, evaporation, or calving.
- Abstraction** the removal of water from any natural water source.
- Advection** process of transfer of air-mass properties by the horizontal movement of air
- Aerobic** condition of water or soil that contains dissolved oxygen sufficient to support aerobic bacteria.
- Aeolian** influenced by wind. Aeolian lakes form in depressions created by wind action.
- Algal bloom** a dense proliferation of algae, often due to eutrophication.
- Alkalinity** amount of cations balanced by weak acids, expressed as milliequivalents of neutralized hydrogen ions per litre of water.
- Allelopathy** adverse influence exerted by plants over competitors by the production of a chemical inhibitor.
- Allochthonous** energy or material created outside the ecosystem. Commonly used to refer to organic matter produced from photosynthesis in the catchment rather than within the water body (see **Autochthonous**).
- Alluvium** clay, silt, sand, gravel or other detrital material deposited by water.
- Amixis** absence of water circulation in a lake. Amictic lakes are permanently stratified and never mix.
- Amphibious macrophyte** an aquatic plant that can tolerate or that responds to wetting and drying of the habitat.
- Amphidromous** fish that migrate from freshwater to salt water, but not for spawning.
- Anadromous** fish that migrate from salt water to freshwater to spawn.
- Anaerobic** condition of water or soil in which the dissolved oxygen is too low to support aerobic bacteria.
- Angiosperms** Plants that produce true flowers, and seeds enclosed in a fruit. There are two major divisions of angiosperms: monocotyledons (grasses, sedges and their allies) and dicotyledons (broad-leaved plants).
- Annual exceedance probability (AEP)** the probability of exceedance of a given flood flow in any given year.
- Annual non-exceedance probability** is the cumulative probability distribution function $F(x)$.
- Annual maximum (minimum) series** extreme-value series with largest (smallest) annual values.
- Annual runoff** (*synonym*, **annual flow**) total volume of water that flows during a year, usually referring to outflow from a drainage basin.
- Anoxic** without oxygen.
- Antecedent precipitation index** weighted summation of past daily precipitation amounts, used as an index of soil moisture.
- Antecedent soil moisture** soil moisture state in a catchment at the start of a rain storm.
- Aquifer** porous sediments and rocks that can store and yield groundwater.
- Arheic drainage basin** a drainage basin where there is not enough runoff for a drainage pattern to develop.
- Artesian aquifer** aquifer in which the piezometric surface lies above the ground surface.
- Assemblage** a group of species found together in an area or habitat, but not necessarily interacting (see **community**)
- Aufwuchs** a film of algae and small animals that encrust the surfaces of submerged objects such as rocks and plant stems (see also **biofilm**, **epilithon**, **epiphyton** and **periphyton**).
- Autotroph** primary producer – an organism that uses sunlight (photosynthesis) and inorganic chemicals to make its own nutritive substances (e.g., plants). Some autotrophs use chemical energy sources (chemisynthesis). Primary production is synonymous with autotrophy.
- Autochthonous** created within the ecosystem. Refers usually to organic matter produced from photosynthesis within the water body (see. **Allochthonous**).
- Average recurrence interval (ARI)** synonymous to return period; defined as the reciprocal of the annual exceedance probability for floods, and reciprocal of the annual non-exceedance probability for low flows.
- Bankfull** the point at which a stream overflows its banks.
- Base flow** stream discharge level during long periods when no precipitation has occurred.
- Bed load** in a streambed, the sediment load that is too heavy to be suspended, so it moves by rolling and sliding.
- Bedrock-controlled** refers to stream channels in which the channel shape is governed largely by bedrock; common in upland areas. In contrast, the shape of the mainly alluvial channels of lowland rivers is controlled by streamflow.
- Benthic** bottom-dwelling. Usually refers to organisms living on the substrate. This assemblage is collectively known as benthos.
- Biochemical oxygen demand (BOD)** index of water pollution, which represents the content of biochemically degradable substances in the water.

- Biodiversity** collective term for all the taxa of plants, animals and microorganisms in an area. Sometimes extended to refer to diversity ranging in scale from that of genes to that of habitats and ecosystems.
- Biofilm** living layer of microorganisms on aquatic substrates (see **Aufwuchs**).
- Biogenic** derived from biological activity.
- Biomanipulation** the artificial alteration of trophic groups such as herbivores, predators or autotrophs to control the abundance of unwanted species.
- Biomass** the amount of living material (either plants or animals) found in a particular environment. Recorded in grams dry weight per unit area, or as the total weight present in a water body.
- Biota** all the organisms of an ecosystem (usually the fauna and flora).
- Biotic** living components of an ecosystem.
- Blue-green algae** members of the Cyanobacteria (or Cyanophyta), characterised by blue-green pigmentation and a lack of cellular organisation.
- Bog** a wetland that derives all of its water from rainfall, with no input from groundwater or surface flow, and accumulates peat in the soil.
- Braided river** a river in which the flow passes through a number of smaller interlaced channels.
- Caldera** a volcanic crater formed by the collapse of the roof of a magma chamber after much of the molten rock has erupted from it; can be occupied by a caldera lake.
- Catadromous** organisms that migrate from freshwater to salt water to spawn.
- Catchment** the drainage basin, synonymous with watershed.
- Cation** a positively charged particle (e.g., Na^+).
- Channel capacity** the volume of river discharge that can be contained within the river banks. When channel capacity is exceeded, over-bank flooding occurs across the floodplain.
- Chemocline** a density gradient formed by a difference in salt concentrations (see **meromictic**). Equivalent to the redoxcline.
- Cirque** a depression formed by glacial erosion, can be filled with water to form a cirque lake.
- Climate** synthesis of weather conditions in a given area, characterised by statistics for long-term meteorological conditions (mean values, variances, probabilities of extreme values).
- Coefficient of variation** a statistical measure of variability derived by dividing the standard deviation by the mean (abbreviated CV).
- Collector** an organism that feeds on fine organic matter either by filter-feeding (collector-filterer) or obtaining material from the substrate (collector-gatherer).
- Compensation depth** the water depth at which the light intensity is such that oxygen production by photosynthesis exactly balances oxygen consumption in respiration. The term is sometimes used in relation to particular plant species, such as rooted aquatics, and sometimes to communities, for example planktonic communities.
- Competence** the largest particle that can be moved by a stream as bedload.
- Conductivity** or specific conductance, an indirect measure of charged particles (electrolytes) in water.
- Cone of depression** depression of the piezometric groundwater surface which defines the area of influence of a well.
- Confidence interval** interval which gives an estimated range of values (calculated from a given set of sample data) that has a specified probability of containing the parameter being estimated.
- Confidence limits** values which form the lower and upper limits to the confidence interval.
- Confined aquifer** aquifer overlain and underlain by an impervious formation.
- Conjunctive use** combined use of surface water and groundwater.
- Consumptive use** activities in which the use of water results in a loss of the original water supplied. Also the quantity of water transpired by vegetation, used directly in the building of plant tissue, and evaporated from the vegetated area.
- Continuity equation** equation describing the conservation of mass of flowing matter (e.g., water).
- Convective precipitation** precipitation caused by upward motion of the atmosphere.
- Conveyance** physical characteristic of an open channel flowing bankfull, taking account of cross-sectional area, roughness, and the square root of slope, that gives the corresponding channel discharge.
- Cosmopolitan** widely distributed. Usually used to refer to a species that is widespread throughout a region, continent or the world (see **endemic**).
- CPOM** coarse particulate organic matter (>1 mm).
- Critical depth** depth of water flowing in an open channel under conditions of critical flow.
- Critical erosion velocity** the lowest current velocity at which a particle resting on the streambed will move.
- Critical flow** flow condition at which the discharge is a maximum for a given specific energy, or at which the specific energy is a minimum for a given discharge.
- Cross-section** section of a watercourse at right angles to the main (average) direction of flow.
- Cryptobiosis** (= "hidden life"), the ability to survive loss of almost all body water and revive upon re-wetting, exemplified by some chironomid midge larvae.

- Cumec** a measure of discharge, abbreviated from cubic metres per second (see **discharge**).
- Cyclomorphosis** seasonal variation in body shape in some animals (e.g., rotifers and cladocerans), apparently in response to predation pressure.
- Cyclonic precipitation** rain, hail or snow caused by the activity of an atmospheric depression.
- Darcy's law** expression of the proportionality of the specific discharge of water flowing through a porous medium to the hydraulic gradient under laminar flow.
- Decomposers** organism that consume dead organisms.
- Degree-day** the difference between the mean temperature of a given day and a reference temperature (usually 0°C). For a given period, algebraic sum of the degree-days of the different days of the period.
- Denitrification** microbial production of N₂ and N₂O from nitrites and nitrates, usually under anaerobic conditions.
- Depression storage** volume of water which is required to fill small depressions to their overflow levels.
- Design flood** flood hydrograph or instantaneous peak discharge adopted for the design of a hydraulic structure or river control work.
- Detention storage** that part of precipitation which is stored temporarily en route to or in the stream system, during or shortly after rainfall.
- Detritivore** an animal that feeds on dead organic matter (see **detritus**).
- Detritus** particles of fine dead organic matter suspended in water or on the benthos.
- Diadromous** usually of fish, migrating between freshwater and saltwater.
- Diatom** member of the algal division Bacillariophyta, characterised by a single cell, with a wall of two siliceous valves. Found in virtually all waters, fresh to saline, and in diverse habitats as phytoplankton in the benthos, and attached to stones and larger plants.
- Diffuse source** a non-point source, usually refers to pollution that comes from a wide area.
- Diffusivity (of an aquifer)** coefficient of transmissivity of an aquifer divided by its storage coefficient.
- Dimictic** two circulation periods annually. Dimictic lakes mix twice a year (spring and autumn). They are common in temperature regions of the Northern Hemisphere, but rare in New Zealand (see **monomictic**).
- Dinoflagellate** member of the algal division Dinophyta. This group of single-celled algae is common in fresh waters, and the cells, surrounded by a rigid, sculptured wall, are motile, with two flagella.
- Direct flow** flow of water entering stream channels promptly, including surface flow and interflow.
- Discharge** volume of water transported per unit time. Frequently calculated from a cross sectional area of a channel (m²) multiplied by current velocity (m/s) to give discharge (m³/s). (See **cumec**).
- Dissolved load** material <0.45 µm transported by river water.
- DOC** dissolved organic carbon. Dissolved carbon < 0.45 µm.
- DOM** dissolved organic matter. Dissolved matter < 0.45 µm.
- Double mass curve** plot of successive accumulated values of one variable against the contemporaneous accumulated values of another variable.
- Drainage density** total channel segment length, accumulated for all stream orders within a drainage basin, divided by the area.
- Drawdown** fall in water level, often due to abstraction.
- Drift** various forms of downstream movement of invertebrates and fish, often with distinct peaks at dusk and dawn.
- Drought** prolonged absence or marked deficiency of precipitation.
- Drought index** a computed value related to some of the cumulative effects of a prolonged or abnormal water deficiency. An index of hydrological drought corresponds to levels below the mean in watercourses or water bodies. An index of agricultural drought relates to the cumulative effect of either an absolute or abnormal transpiration deficit.
- Duration curve** graph representing the time during which the value of a given parameter, e.g. water level, is equalled or exceeded, regardless of continuity of time.
- Dynamic viscosity** a measure of the force per unit area required to maintain a difference in velocity between two parallel layers.
- Dystrophic** waters that might be expected, on the basis of nutrients, to contain high concentrations of phytoplankton, but instead have heavily stained waters of low productivity.
- Ecoregion** a geographic area with distinct climatic and geological conditions, and assemblages of communities.
- Ecosystem** the combination of a community (biota) and its abiotic environment. Ecosystems are characterised by ecological processes such as the flow of energy and nutrients through food webs.
- Ecosystem engineer** an organism that physically modifies its environment, frequently resulting in changes that influence other organisms.
- Ecotone** region lying between two ecosystems, often sharing some ecological features of both. The riparian zone represents an ecotone between wetlands and their catchments.

- Ectogenic** derived from “outside” events or processes, usually in reference to meromixis.
- Emergent macrophyte** a plant growing or protruding above the water surface (e.g., sedges, reeds) as opposed to floating or submerged plants.
- Endemic** restricted in distribution. Usually used to refer to species of plants or animals found only in one region and nowhere else in the world (see **cosmopolitan**).
- Energy gradient** decrease or loss of total head along a channel per unit distance.
- Environmental flows** releases of water, periods of drying, or river flows allocated for the maintenance of aquatic and riparian ecosystems.
- Ephemeral** containing water only after unpredictable rain.
- Epilimnion** the zone within a water body above the thermocline that is well mixed and therefore of uniform temperature – the surface mixed layer.
- Epilithon** biofilm occurring on rocks.
- Epiphyton** biofilm occurring on plants. Epiphytes can be plants or animals that grow attached to plants, using them as support rather than as a direct source of organic matter.
- Equilibrium line of glacier** line or zone on a glacier where a year’s ablation balances a year’s accumulation.
- Euphotic zone** the upper section of a water mass penetrated by light of sufficient intensity and of suitable wavelength to promote photosynthesis by aquatic plants. For convenience, this zone has been defined as the water down to a depth where 99% of the light during the period of maximum illumination has been absorbed. Sometimes also called the photic zone.
- Eutrophication** nutrient enrichment of a waterbody, usually leading to growth and proliferation of large masses of plant material (either phytoplankton, macrophytes, or both).
- Evaporation** transformation of water to a vapour at a free surface at a temperature below boiling point.
- Evapotranspiration** the combined loss of water by evaporation from a free water surface and the soil surface, and water evaporating from the leaves of plants (transpiration).
- Extreme value distribution** probability distribution of the largest (smallest) observations in a sample. Commonly used extreme value distributions are Gumbel and Weibull. Types I, II, and III, encapsulated in the Generalised Extreme Value distribution.
- Falling limb** part of a hydrograph in which the discharge is decreasing from a peak.
- Fen** a wetland receiving significant water inputs from both rainfall and groundwater, and with a peaty, waterlogged, substratum, in which the water-table rarely rises above the surface.
- Fetch** area in which ocean, lake and reservoir waves are generated by the wind. The length of the fetch area is measured in the direction of the wind.
- Field capacity** amount of water held in a soil after water has drained away by gravity.
- Filamentous alga** an alga made up of chains of cells joined end to end.
- Firn** snow that has become granular and dense, due to melting and refreezing, or sublimation and crystallization.
- Firn line** boundary at a glacier’s surface that separates the zone of accumulation from the zone of ablation.
- Flash flood** flood of short duration with a relatively high peak discharge.
- Flood** rise in water level to exceed channel capacity, followed by recession. The water pulse is illustrated on a hydrograph as the rising limb, peak, and falling limb over time.
- Flood forecasting** estimation, for a specified point on a stream, of the stage, discharge, time of occurrence, and duration of a flood resulting from precipitation and/or snowmelt.
- Flood frequency** number of times a flood above a given discharge or stage is likely to occur over a given number of years.
- Flood pulse concept (FPC)** a model emphasising the importance of floods in lowland river ecosystems. In contrast to the longitudinal emphasis of the river continuum concept, the FPC highlights the role of lateral transfer between the river and its floodplain.
- Floodplain** temporarily inundated lateral river flats, usually of lowland rivers.
- Flood routing** technique used to compute the movement and change of shape of a flood wave moving through a river reach or a reservoir.
- Flood wave** rise in streamflow to a maximum crest, and its subsequent recession, caused by a period of precipitation, snow melt, dam failure, or power plant release.
- Flotation load** material such as logs, sticks and leaves carried by a stream at the water surface.
- Flow-duration curve** curve showing the percentage of time during which the flow of a stream is equal to or greater than given amounts, regardless of chronological order.
- Fluvial** referring to running waters.
- Forecast** definite statement or statistical estimate of the occurrence of a future event.
- Fossil water** groundwater arbitrarily considered non-renewable because its renewal rates exceed 100 years.
- FPOM** fine particulate organic matter (0.45 μm –1 mm). See **DOM** and **CPOM**.

- Frequency analysis** procedure for interpreting a past record of hydrological events, in terms of future probabilities of occurrence.
- Frequency curve** curve relating the possible values of a variate, e.g., the value of a specified hydrological event, to the frequency of its occurrence.
- Frequency distribution** specification of the way in which the frequencies of members of a population are distributed, according to the values of the variates which they exhibit.
- Fresh water** naturally occurring water having a low concentration of dissolved salts (< 1000 ppm)
- Froude number** dimensionless number expressing the ratio of inertia forces to gravity forces.
- Functional feeding group** categories assigned to aquatic invertebrates to describe their main feeding strategy, e.g., shredder, grazer, predator.
- Glacier** accumulation of ice of atmospheric origin, generally moving slowly on land over a long period. The New Zealand Glacier Inventory defines glaciers as ice formations > 1 ha and having existed for more than 20 years.
- Grazers** organisms that feed on algae scraped from surfaces.
- Green algae** members of the algal division Chlorophyta, characterised by green chloroplasts. May be unicellular (e.g., desmids), filamentous (e.g., *Spirogyra*), colonial (e.g., *Volvox*) or macrophytic (e.g., *Chara*).
- Groundwater storage** quantity of water in the saturated zone of an aquifer.
- Hard water** water with high concentrations of Ca^{2+} and Mg^{2+} , which cause dissolved compounds to precipitate readily.
- Head loss** decrease of total head, expressed in units of height, due to energy dissipation.
- Heavy metals** metals with high atomic weights (> 50, e.g., Pb, Zn). Usually, even at low concentrations, these metals are toxic to most plant and animal life.
- Heterotroph** an organism that obtains energy by using matter made by autotrophs, e.g., a consumer.
- Hjulstrom curve** curve describing the relationships between sediment size, stream velocity, erosion and sedimentation.
- Holomixis** complete mixing. In holomictic lakes, the seasonal circulation of water causes mixing of the water column down to the bottom (see **meromixis**).
- Homothermous** a water body which is the same temperature throughout and therefore of uniform density. This lack of stratification indicates recent mixing and that the water can be readily mixed by the wind.
- Humidity** a measure of the amount of moisture in the atmosphere.
- Hydraulic conductivity** property of a saturated porous medium which determines the relationship, called Darcy's Law, between the specific discharge and the hydraulic gradient causing it.
- Hydraulic gradient** the slope of the water surface in an open channel; measure of the decrease in head per unit distance in the direction of flow through a porous medium.
- Hydraulic jump** sudden transition of water level in an open channel from a depth less than critical depth to a depth greater than critical depth.
- Hydraulic radius** ratio of the wetted cross-sectional area to the wetted perimeter of a water course.
- Hydrogen bond** the connection between a positively-charged hydrogen atom and a negatively-charged atom of another molecule. Hydrogen bonds confer many of water's special properties.
- Hydrograph** plot of discharge or water level (stage) against time. Depending on the time scale, this shows the shape of the flood pulse in response to storms (days) or shows the flow regime of a river (decades).
- Hydrological cycle** the circulation of water between ocean, atmosphere, and land via evaporation, precipitation, and flow through and over the land.
- Hydrological drought** period of abnormally dry weather sufficiently prolonged to give rise to a shortage of water, as evidenced by below-normal streamflow and lake levels and/or the depletion of soil moisture and a lowering of groundwater levels.
- Hydrology** the study of the mechanics of surface and subsurface water and flow. Sometimes used loosely to describe the water regime.
- Hydrometric station** station at which data on water in rivers, lakes or reservoirs are obtained.
- Hydroperiod** the seasonal pattern of when, where and to what extent water is present in a wetland. The components of hydroperiod are the timing, duration, frequency, extent and depth, and variability of water presence. Also termed "water regime".
- Hydrophilic** attracted to water. Opposite to "hydrophobic".
- Hydrophyte** a plant growing in water – they may be submerged, floating or emergent (see **macrophyte**).
- Hyetograph** graph displaying the intensity of precipitation versus time.
- Hygroscopicity** capacity of a porous medium to absorb atmospheric moisture.
- Hypolimnion** the water mass below the thermocline in a stratified lake. Often this water is poorly illuminated and chemically distinctive.

Hyporheic zone the wetted interstitial zone below and alongside rivers; inhabited by many organisms specialised for a subsurface existence (termed the "hyporheos").

Hysteresis variability of the stage-discharge relationship at a gauging station subject to variable water surface slope where, for the same gauge height, the discharge on the rising stage is different from that on the falling stage.

Impervious (synonym **Impermeable**) having a texture that does not permit water to move through it perceptibly under static pressure ordinarily found in subsurface water.

Infiltration flow of water through the soil surface into a porous medium.

Infiltration capacity maximum rate at which water can be absorbed by a given soil per unit area under given conditions.

Infiltration coefficient ratio of infiltration to rainfall.

Instream use use of water within the channel (see **off-stream use**).

Interbasin transfer artificial movement of water via channels or pipes from one catchment to another.

Interception process by which precipitation is caught and held by vegetation, then may be lost by evaporation without reaching the ground.

Interflow that portion of the precipitation which has not passed down to the water table, but is discharged from the area as subsurface flow into watercourses.

Internal loading storage or entry of nutrients or pollutants into a wetland from its sediment.

Interstitial water water contained in the spaces between the benthic substrate.

Ions chemicals dissolved in water dissociate into positive and negative particles called ions. For example, each molecule of table salt, sodium chloride (NaCl), dissociates in solution to positively charged sodium ions (Na⁺) and negatively charged chloride ions (Cl⁻).

Isohyet line joining the points where the amount of precipitation in a given period is the same.

Karst limestone and dolomite areas that possess a distinctive topography created by underground solution and the diversion of surface water to underground routes.

Kinematic viscosity a measure of the interference between adjacent layers of fluid.

Labile readily available for breakdown, when used in reference to organic matter, in contrast to refractory organic matter. Most labile dissolved organic matter comprises low molecular weight compounds such as simple sugars.

Lacunae spaces in the stems, leaves, rhizomes and roots of aquatic plants, which allow gas to be transported at a much faster rate than could be achieved by diffusion in water. Important for the transport of oxygen and carbon dioxide.

Lag time time from centre of mass of rainfall to centre of mass of runoff, or to the peak of runoff.

Laminar flow smooth flow without turbulence or mixing.

Larvae the juvenile life stage of an invertebrate.

Leaching removal of salts from upper soil layers by relatively salt-free water.

Lentic referring to standing water.

Liebig's Law of Minimum the rate of a process is limited by the rate of the slowest component's supply.

Limnetic zone open water in a lentic wetland.

Limnology the science of the physical, chemical and biological processes that take place in inland waters.

Limnoplankton plankton typical of standing waters.

Lithology the science of the nature and composition of stones and rocks.

Litter uppermost layer of organic debris, composed of freshly fallen or slightly decomposed organic materials.

Littoral zone edge or shore region where the water is shallow enough for continuous mixing.

Live storage volume of a lake or reservoir between the maximum and minimum operating levels.

Lotic referring to running waters.

Lysimeter vessel containing local soil placed with its top flush with the ground surface for the study of infiltration, runoff, evapotranspiration etc.

Macroinvertebrates invertebrates, functionally defined as > 500 µm. Their body length usually exceeds 1 mm. A term of convenience without fundamental, taxonomic significance.

Macronutrients nutrients required in relatively large amounts (e.g., nitrogen, phosphorus).

Macrophytes large water plants, represented in freshwaters by submerged, floating and emergent plants. A term of convenience to distinguish them from smaller plants (such as phytoplankton) that must be viewed with a microscope.

Marl calcareous sediments (mostly CaCO₃), usually in the littoral zone.

Marsh wetland with fluctuating water levels and an inorganic substratum, i.e., one that is sandy rather than composed of peat. Characteristic of estuaries, where salt marshes may be dominated by plants such as sea rush (*Juncus kraussii*) and oioi (*Apodasmia similis*).

Mass wasting gravity-driven bulk movement of sediments into stream channels (e.g., landslides, soil creep, bank slumping).

- Meander** one curved portion of a sinuous or winding water course.
- Mean annual flood** arithmetic mean of all the individual annual flood peaks in a period of record, synonymous with “index flood” for regional flood estimation.
- Mean monthly discharge** arithmetic mean of all the individual monthly mean discharges for a named month in a period of record.
- Meiofauna** small invertebrates, > 50 but < 500 μm in size.
- Meromictic lakes** permanently stratified water bodies with a layer of dense saline water below a layer of less dense, relatively fresh, water.
- Meromixis** partial circulation, the lower, denser layer (monimolimnion) below the chemocline never mixing with the upper, usually fresher layer (mixolimnion).
- Mesotrophic** an intermediate condition, naturally occurring or due to nutrient enrichment, between eutrophic and oligotrophic.
- Metalimnion** the region of water between the surface mixed layer (the epilimnion), and the lower layer (the hypolimnion) of a stratified water body.
- Microbial loop** the role bacteria play as a link between detritus and higher consumers in aquatic food webs.
- Micronutrients** nutrients required in relatively small amounts (e.g., iron, manganese, molybdenum).
- Minimum annual flow** least value of discharge in a water course during a hydrological year.
- Mixolimnion** the upper layer in a meromictic lake that occasionally mixes.
- Moment** the mean value of a power of a variate (“product” moments). For a distribution of a variate, moments may be taken about the origin or about a given fixed value; if the latter is the mean, the moments are central moments. “L-moments” are alternative moments which avoid raising variates to powers, “linear” moments.
- Monimolimnion** the permanently stagnant layer in a meromictic lake below the chemocline.
- Monomictic** mixing only once a year. The water in monomictic lakes usually stratifies into layers in summer and begins mixing in autumn. Most of the larger lakes in New Zealand are monomictic.
- Motile** moving. Usually used in relation to microscopic plants or animals that, in contrast to non-motile forms, can move actively through the use of flagellae (long whip-like structures) or cilia (small, beating hairs).
- Mucilage** a glutinous substance (usually carbohydrate) secreted by some organisms.
- Muskingum method** approximate hydrological method of flood routing, based on the continuity equation for a reach and a storage equation expressing the linear dependence of the water volume in the reach on the weighted inflow and outflow discharges.
- Nanoplankton** tiny plankton, ranging from 2–20 μm .
- Nekton** strongly swimming, highly mobile organisms within the water column.
- Net plankton** plankton retained by a net with mesh of 50 to 53 μm .
- Net radiation** difference between downward and upward radiation; net flux of all radiations.
- Nitrification** the process of oxidising NH_3 to NO_2 , and NO_2 to NO_3 .
- Nitrogenase** the enzyme that reduces N_2 to NH_2 .
- Non-structural flood mitigation** system for reduction of the effects of floods using non-structural means, e.g., land-use planning, advance warning systems, flood insurance.
- Non-uniform flow** flow in which the velocity vector is not constant along every streamline.
- Non-vascular plants** plants without defined tubular transport systems; includes algae, mosses, liverworts and fungi.
- NTU** nephelometric turbidity unit, a measure of turbidity based on light transmission through water.
- Nutrient** substance, element or compound necessary for the growth and development of plants and animals.
- Nutrient spiralling** the longitudinal cycling and transport of nutrients down a stream or river.
- Off-stream use** water abstracted for use elsewhere (e.g., irrigation).
- Oligomixis** rare or occasional water circulation in lakes.
- Oligotrophic water** water that is low in nutrients, hence with a relatively low organic productivity. Opposite to **eutrophic**.
- Omnivore** animal that eats both plants and animal matter.
- Orographic precipitation** caused by the ascent of moist air over geographic barriers (e.g., mountains).
- Overland flow** downslope surface movement of runoff other than in defined channels, usually with high erosion potential.
- Oxic** oxygenated.
- Palaeolimnology** the study of the history of water bodies.
- PAR** see **photosynthetically active radiation**.
- Partial duration series** series of values of independent events such as floods, occurring above a base value selected, without regard to the number within a given period. Synonymous with “peaks over a threshold” series.
- Peak flow** maximum instantaneous flow or discharge of a given hydrograph.
- Pelagic** open-water zones, limnetic. Also used to mean actively swimming.
- Perched lake** lake sealed from the groundwater by impervious sediments such as organically bonded material.

- Percolation** flow of a liquid through an unsaturated porous medium under the action of gravity.
- Perennial stream** stream which flows continuously throughout the year.
- Periphyton** the biota attached to submerged surfaces (see **Aufwuchs**).
- Persistence (hydrological)** tendency of a hydrological process or time-series to recur or continue.
- pH** The negative logarithm to base 10 of the hydrogen ion concentration. Acidic solutions have a pH <7, basic solutions have a pH >7. The ocean has a pH of about 8.3, and an acid bog a pH of about 3.5. The scale is logarithmic, not linear, as hydrogen ion concentrations differ over a very wide range in different solutions.
- Photic zone** see **euphotic zone**.
- Photosynthesis** process whereby plants use solar energy, water and carbon dioxide to generate organic molecules. Some bacteria carry out photosynthesis in which hydrogen sulphide instead of water is used as the source of hydrogen, and elemental sulphur is released instead of oxygen.
- Photosynthetically active radiation (PAR)** light of wavelengths that can be used for photosynthesis (350-700 nm).
- Phreatic water** groundwater occurring in the zone of saturation and having a water table.
- Phytoplankton** photosynthetic plankton (algae and Cyanobacteria).
- Picoplankton** plankton ranging from 0.2 to 2 μm .
- Piezometric head** elevation to which water will rise in a piezometer connected to a point in an aquifer, or the sum of the elevation and the pressure head in a liquid.
- Piezometric surface** surface joining points which are at an elevation equal to the piezometric head in a given aquifer.
- Piscivores** predators that consume fish
- Plankton** community of tiny organisms, plants (phytoplankton) and animals (zooplankton), freely moving in the open water.
- Point source** a localised site of production, usually referring to pollution (see **diffuse source**).
- Polymictic** lakes in which the water column undergoes frequent periods of stratification and re-mixing. Typical of small, shallow lakes, especially in coastal regions.
- POM** particulate organic matter (>0.45 μm).
- Porosity** ratio of the volume of the interstices in a given sample of a porous medium to the gross volume, inclusive of voids.
- Porous medium** permeable medium that contains connected interstices and may be considered as a continuous medium with respect to its hydraulic properties.
- Potable** suitable for drinking by humans.
- Potential evapotranspiration** maximum quantity of water capable of being evaporated, in a given climate, from continuous vegetation covering the whole ground and well supplied with water.
- Precipitation** liquid or solid water from the condensation of water vapour falling from clouds or deposited from the air on the ground.
- Predators** organisms that consume other organisms.
- Pristine** completely unaltered by humans; in a natural state.
- Probability distribution** distribution giving the probability of a value of a variate as a function of the variate.
- Probable maximum flood (PMF)** estimated greatest flood that may be expected, taking into account all pertinent factors of location, meteorology, hydrology and terrain.
- Probable maximum precipitation (PMP)** estimated amount of precipitation that is the statistical and/or physical upper limit for a given duration over a particular basin.
- Producers** general term given to organisms that fix carbon into organic matter, either by photosynthesis (primary producers) or by consuming other organisms (secondary producers).
- Profundal zone** bottom of a water body lying below the euphotic and littoral zone.
- Prostrate** extend along the ground, referring to a growth form of plants.
- Psammon** the community of microscopic animals and plants which live attached to sand grains or in the spaces between them. It is found at the edges of water bodies.
- Pseudo-faeces** material (such as small aggregations of sand grains) that has been processed in the mouth of an animal, the food material removed, and has been rejected without having passed through the alimentary tract.
- Rainfall excess** that part of rainfall that contributes directly to surface runoff
- Rainfall intensity** rate at which rainfall occurs, expressed in units of depth per unit of time.
- Rapid** reach of a stream where fast, white water occurs.
- Rating curve** curve showing the relation between stage and discharge of a stream at a hydrometric station.
- Rational method** formula expressing the estimated peak rate of storm runoff as the product of the catchment area, a peak rate of rainfall, and a runoff coefficient.
- Reach** length of open channel between two defined cross-sections.
- Recession** period of decreasing discharge as indicated by the falling limb of a hydrograph, starting from the peak.

- Recharge** process by which water is added from outside to the zone of saturation of an aquifer, either directly into a formation, or indirectly by way of another formation.
- Recharge area** area where surface water from rain, irrigation or streams infiltrates the groundwater or soil water.
- Recurrence interval** see **return period**, **average recurrence interval** and **annual exceedance probability**.
- Redox potential** the ability of a solution or soil to oxidise or reduce; usually measured with platinum electrodes.
- Reduction** chemical process leading to loss of oxygen or increase of electrons by a compound. Opposite of oxidation.
- Reflectivity** ratio of the energy reflected by a given surface to that incident on it.
- Refraction** the change in direction and velocity of light waves passing from one medium to another.
- Regression analysis** statistical method developed to investigate the interdependence or relationship between two or more measurable variates.
- Relative humidity** at a given pressure and temperature, the percentage ratio of the mole fraction of the water vapour to the mole fraction that the air would contain if it were saturated with respect to water at the same pressure and temperature.
- Representative basin** basin in which hydrological stations are installed to make simultaneous hydrometric and meteorological measurements that represent a broad area or region.
- Residence time** period during which water or a substance remains in a component part of the hydrological cycle.
- Respiration** the process by which the energy of organic material is made available to drive energy-consuming reactions in the cell (for example, the formation of cell walls, proteins, cell movements and division).
- Resurgence** reappearance above ground, at the end of its underground course, of a surface water flow that disappeared underground.
- Retention** that part of the precipitation falling on a drainage area that does not escape as surface streamflow during a given period.
- Retention curve** graph representing the suction pressure versus the moisture or water content in an unsaturated soil.
- Return flow** any flow which returns to a stream channel or to the groundwater after use.
- Return period** long-term average interval of time or number of years within which an event will be equalled or exceeded. Applied to floods, it is defined as the reciprocal of the annual exceedance probability (AEP), and for low flows it is the reciprocal of the annual non-exceedance probability. A flood with a 50 year return period (or 50 year average recurrence interval, the "1 in 50 year flood") is not one that occurs every 50 years but one with a 2% chance of occurring in any year (2% AEP).
- Reynolds number** dimensionless number expressing the ratio of inertia forces to viscous forces.
- Rhizoids** the simple organs of attachment produced by algae and mosses, consisting usually of single unbranched cells but sometimes of branched cells or cells arranged end-to-end.
- Rhizomes** horizontal plant stems that run under the soil surface, bearing roots and erect shoots and leaves, and containing food reserves.
- Riffle** shallow rapids, where the water surface is broken into waves by totally or partially submerged obstructions.
- Riparian zone** the terrestrial zone immediately adjacent to and interacting with a water body. The zone may be inundated by water from time to time. It protects banks against erosion, is a habitat for animals associated with the water, and contributes organic material to the water.
- Rising limb** part of a hydrograph in which the discharge is increasing towards a peak.
- River continuum concept (RCC)** an ecosystem-based model predicting the response of biota along the physical gradient of a river.
- River regulation** modifications to the flow regime, channel shape or immediate floodplain of a river to control it for human needs.
- River training** engineering river works built in order to direct the flow, lead it into a prescribed channel, or increase the water depth for navigation.
- Runoff** water that flows across the land surface and does not soak into the ground. Also, that part of precipitation that appears as streamflow.
- Salinity** a measure of the total amount of dissolved salts in water, usually expressed as parts per thousand. The spectrum of salts in inland waters may deviate greatly from that of sea water, and so the term "total dissolved salts" is sometimes used to take account of this difference.
- Salt-water intrusion** phenomenon occurring when a body of salt water invades a body of fresh water. It can occur in either surface or groundwater bodies.
- Saturated zone** part of the water-bearing material in which all voids are filled with water.
- Saturation vapour pressure** maximum possible partial pressure of water vapour in the air at a given temperature.
- Scour** erosive activity, in particular pronounced local erosion, by water in streams.
- Scraper** an organism that feeds by scouring algae from surfaces, e.g., snails.

- Secchi depth** the depth to which a circular disc with black and white quadrants (Secchi disc) is visible when lowered into water to estimate water clarity. Generally used in lentic systems only.
- Sedentary** scarcely mobile (see **sessile**).
- Sediment** material usually inorganic transported by water. In watercourses, sediment is the alluvial material carried in suspension or as bed load.
- Sedimentation** process of settling and deposition by gravity of suspended matter in water.
- Sediment discharge** discharge of sediments of a stream at a given cross-section.
- Sediment yield** total sediment outflow from a drainage basin or past a given location in a specified period of time.
- Seepage velocity** discharge of seepage water through an unsaturated porous medium, per unit area of void space, perpendicular to the direction of flow. It is the average actual velocity of an unsaturated flow.
- Seiche** an oscillation of water that travels as a wave to return to equilibrium after disturbance, usually by wind
- Sessile** fixed, immobile (see **sedentary**).
- Seston** suspended organic particles
- Shredder** organisms that feed on coarse particulate organic matter, macerating leaves, twigs and bark, providing fine particulate organic matter to the ecosystem.
- Sinuosity** a measure of the extent of meandering of a river channel, calculated as channel distance divided by down-valley distance.
- Slope-area method** method of estimating discharge by application of open-channel hydraulic formulae. The most commonly used is the Manning equation.
- Snow course** line laid out and permanently marked, along which snow is sampled and/or its depth is measured.
- Snow line** Climatic snow line is lowest altitude of a continuous layer of snow in summer on the highest mountains. Orographic snow line is lowest altitude at which, in summer, snow persists in the form of isolated sheets.
- Snow melt** transformation of snow into liquid water; water from melting snow.
- Snow pack** field of naturally accumulated snow that ordinarily melts slowly and yields water during the early summer months.
- Soil drainage** movement of free water contained within a soil under the effect of gravity.
- Soil moisture** moisture contained in the portion of the soil that is above the water table, including water vapour present in the soil pores.
- Soil moisture deficit** difference between the field capacity of a soil and the actual soil moisture.
- Soil moisture tension** negative pressure (suction) to which water must be subjected in order to be in hydraulic equilibrium, through a porous permeable wall or membrane, with the water of the soil.
- Soil water** water suspended in the uppermost belt of soil, or in the zone of aeration near the ground surface, that can be discharged into the atmosphere by evapotranspiration.
- Spate** a small flood, usually scouring upland streams and acting as a disturbance.
- Specific discharge** discharge per unit area of a drainage basin; for groundwater, ratio of discharge per unit area measured at right angles to the direction of flow.
- Specific energy** sum of the elevation of the free surface above the bed and the velocity head based on the mean velocity at that section.
- Specific yield** ratio of the volume of water that can be drained by gravity from an initially saturated porous medium to the total volume of the medium.
- Sporangia** a specialised plant structure in which spores are formed and from which they are released. The structure is found throughout the plant kingdom, including in fungi, algae, mosses and ferns.
- Sporeling** a young plant produced from a spore, for example in the algae.
- Stage** river height, usually as measured at a river gauging station.
- Stage-discharge relation** relation between stage and discharge at a river cross-section, expressed as a curve, table, or equation.
- Standing crop** the total quantity (usually expressed as weight) of a species or group of species per unit surface area or in a whole ecosystem.
- Stationary process** a stochastic process is said to be strictly stationary if its probability distribution is completely independent of time. It is said to be stationary in the wide sense if the mean and variance are independent of time.
- Stem flow** process whereby rainfall intercepted by vegetation flows down the trunks of trees or stems of bushes to reach the ground.
- Stochastic process** a process which produces variates, $x(s)$, where s assumes consecutive values in a certain range, t . In most cases, s and t refer to time (viz. "time series"), but they may also refer to space.
- Stoichiometry** the ratio of elements to each other.
- Storage** impounding of water in surface or underground reservoirs for future use; the volume of water stored.
- Storage coefficient** volume of water an aquifer releases from or takes into storage per unit surface area of the aquifer per unit change of head.
- Storm transposition** transposition of data for an observed storm (depth-area-duration pattern) from the basin area of its occurrence to a project basin or area within the same region of meteorological homogeneity.

- Stratification** separation of different layers of water because of differing densities, especially in lakes. The density differences may be due to temperature or salinity, or both. Stratification may be only transient, but it can be seasonal or more permanent. It has profound implications for biological, chemical and physical processes in water.
- Stream order** a measure of stream or river size defined by the degree of branching in a drainage system, e.g., a first-order stream has no tributaries, while a second-order stream has at least two first-order tributaries.
- Structural flood mitigation** structural system for reducing the effects of floods using physical solutions, e.g., stopbanks, reservoirs, flood-proofing etc.
- Subcritical flow** flow with a mean velocity less than the critical velocity.
- Sublimation** change in phase directly from a solid to a gaseous state or a gaseous state to a solid, without going through an intermediate liquid phase.
- Submerged macrophyte** a water plant with foliage entirely submerged; its reproductive organs may be aerial, floating or submerged.
- Subsurface flow** any flow below the surface of the ground that may contribute to interflow, base flow, or deep percolation.
- Supersaturation** condition in which dissolved gases exceed 100% saturation at a given temperature, salinity and pressure.
- Surface detention** that part of the rain which remains on the ground surface during rain and either runs off or infiltrates after the rain ends, not including depression storage.
- Surface mixed layer** See **epilimnion**.
- Suspended sediment load** fine material entrained by currents and transported down streams in the water column; includes washload, drifting organisms, and particulate organic matter.
- Swamp** used colloquially to refer to any wetland characterised by emergent vegetation. More technically, the term is applied to a wetland in which most of the water is derived from groundwater or streams, the water-table is usually above the soil surface, and there is emergent vegetation on a peat or mineral substrate. In contrast, see **bog**, **fen** and **marsh**.
- Synthetic (unit) hydrograph** hydrograph developed on the basis of estimation of coefficients related to various physical features of a catchment.
- Taxon** the lowest level of identification of an organism. Frequently cited instead of "species" where an organism cannot be confidently identified to species level.
- Tectonic lake** lake in a depression formed by large-scale movement of the Earth's crust.
- Tensiometer** instrument that consists of a porous cup inserted in soil and connected by a water-filled tube to a manometer, which makes it possible to measure capillary tension or suction.
- Thalweg** the line of greatest depth along a stream bed.
- Thermal spring** groundwater-fed stream with water temperature above the mean annual air temperature.
- Thermocline** the steep temperature gradient between two of the layers in a stratified lake. Typical of temperate-climate lakes during the summer. The sharpness of the temperature gradient is determined by the depth of wind mixing.
- Throughfall** part of rainfall that reaches the ground directly through the vegetative canopy, through inter-plant spaces in the canopy, and as drip from the vegetation.
- Throughflow** the downslope movement of water through the soil.
- Time of concentration** period of time required for storm runoff to flow to the outlet from the point of a drainage basin having the longest travel time.
- Top-down effects** biological regulation of lower trophic levels by higher organisms in the foodweb.
- Total dissolved solids (TDS)** the sum of dissolved salts and organic residues.
- Total head** sum of the elevation of the free surface above a horizontal datum, at a section, and the velocity head based on the mean velocity at that section.
- Transmissivity** rate at which water is transferred through a unit width of an aquifer under a unit hydraulic gradient. It is expressed as the product of the hydraulic conductivity and the thickness of the saturated portion of the aquifer.
- Transmittance** the ratio of transmitted radiant energy to incident radiant energy, or the fraction of light that passes through a medium. Varies with wavelength, e.g., blue light penetrates water better than the longer wavelengths of red light.
- Transpiration** the loss of water by evaporation from the leaves of plants. The water is lost through small pores (stomata), which close at night and under conditions of water stress. The transpired water is derived from the walls of cells in the leaves, and is replaced by water taken up through the roots of the plant.
- Trap efficiency** ability of a reservoir to trap and retain sediment, expressed as the percent of sediment yield that is retained in the reservoir.
- Travel time** time elapsed for the passage of a water parcel, e.g. a flood wave, between a given point and another point downstream.
- Trophic cascade** sequence of changes in abundance of prey and predators or plants and herbivores due to biological interactions.

- Turbidity** the reduction in passage of light through water that is brought about by suspended material. Light may also be reduced by the presence of dissolved material, and the term turbidity is sometimes incorrectly used to describe this as well.
- Turbulence** water flow that is rough and mixing in a complex fashion, common in riffles in upland streams. See **laminar**.
- Turnover** action of turning over a water body, whether or not stratified, due to a change in ambient temperature and/or wind.
- Unconfined aquifer** aquifer containing unconfined groundwater, that is, having a water table and an unsaturated zone.
- Underflow** movement of water through a pervious stratum under the bed of a water course.
- Uniform flow** flow in which the velocity vector is constant along every streamline.
- Unit hydrograph** a hydrograph of direct runoff resulting from a unit amount of effective rainfall generated uniformly over a watershed during a specified duration.
- Vadose water** any water that occurs in the unsaturated zone.
- Variate** quantity which may take any of the values of a specified set with a specified relative frequency or probability.
- Varied flow** flow occurring in water courses having a variable cross-section or slope.
- Velocity head** theoretical height to which a liquid particle could be elevated by its kinetic energy.
- Vertical absorption coefficient** the fraction of incident radiant energy absorbed per unit mass or thickness of water due to dissolved substances and suspended particles.
- Vertical attenuation coefficient** the rate of decrease of light per unit distance in the water column, caused by both absorption (see **vertical absorption coefficient**) and scattering of light.
- Viscosity** property of a fluid to resist shearing in the presence of velocity gradients.
- Void ratio** ratio of the volume of voids in a given porous medium sample to the volume of solid particles.
- Wash load** fine sediments (e.g., clay and silt) carried in suspension in river water (see **suspended sediment load**).
- Water balance** inventory of water based on the principle that during a certain time interval the total water gain to a given catchment area of water body must equal the total water loss plus the net change in storage.
- Water deficit** cumulative difference between potential evapotranspiration and precipitation during periods when the precipitation is the smaller of the two.
- Water loss** In a water balance, it is the sum of water lost from a given land area during a specific time by transpiration, building of plant tissue, evaporation, and interception. In irrigation, it is the losses from seepage and evaporation from canals and ditches, water wasted from the canal system, excess water drained from the land surfaces, and water lost to deep percolation.
- Water quality** the suitability or condition of water for a given use (e.g., human drinking water, stock water, etc.).
- Water regime** see **hydroperiod**.
- Water resources** water available, or capable of being made available, for use in sufficient quantity and quality at a location and over a period of time appropriate for a given demand.
- Watershed** the boundary of a catchment. Some texts use the term watershed interchangeably with catchment.
- Water column** general term to describe the water overlying the bed of a water body.
- Water table** the interface between groundwater and the soil substrate or air.
- Well capacity** maximum rate at which a well will yield water under a stipulated set of conditions, such as a given drawdown.
- Wetland** permanently or intermittently wet areas, shallow water or land/water margins that support a natural ecosystem of plants and animals that are adapted to living in wet conditions. They may be saline, freshwater, or brackish.
- Wetted perimeter** length of the wetted contact between a stream of flowing water and its containing channel, measured in a direction normal to the flow.
- Wetting front** air/water interface in the process of water infiltrating into the soil.
- Wilting point** moisture content of the soil at which the leaves of plants growing in that soil become permanently wilted.
- Yield** quantity of water that can be collected for a given use from surface or groundwater sources in a basin in a given time interval. Also, the quantity of water derived from a unit area of a drainage basin in a given time interval.
- Zooplankton** the animal component of plankton. These animals may be grazers on phytoplankton (especially diatoms), detritivores, which feed on organic particles and bacteria, or predators on other zooplankton. Like the phytoplankton, they are vital components of food webs in large lakes and oceans.

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Chapter 46

Water and human health

David Slaney and Philip Weinstein

INTRODUCTION

Freshwater is fundamental for human survival. Hunter-gatherer societies in the modern era provide some insight into the importance of water to our early ancestors, whose foraging patterns, migration routes, cultural evolution, health and very existence depended on the availability of freshwater. Water availability was also critical to the development of the earliest settlements in the Old World and to agricultural development. It is no accident that the first significant concentrations of cities, towns and villages all arose in major river valleys: the Nile, the Tigris and Euphrates, and the Indus. To the Maori, water was essential to both physical and spiritual wellbeing (Chapter 44), and its importance is reflected in the frequency of *wai* in Maori place-names. Even in industrialised societies, population health is directly dependent on safe water supplies, as is perhaps best illustrated when things go wrong. Public health practitioners who work in disaster management operate on the basis of providing, firstly and foremost, water, closely followed by food, shelter and sanitation.

In the developed world, we tend to take the availability of freshwater for granted, and worry more about the potential health effects of water with chemical and microbiological contaminants. Increasingly though, the very availability of water is becoming a problem, as is noticeable in every household when drought necessitates the imposition of water restrictions for watering gardens, as has for example been the case on the Kapiti Coast, New Zealand, since 1996.

Such is the importance of access to freshwater, that in November 2002, water for the first time was explicitly recognised as a fundamental human right, when the United Nations Committee on Economic, Social and Cultural Rights adopted General Comment No.15 (ECOSOC 2002) to the International Covenant on Economic, Social and Cultural Rights. Although not

legally binding, the 146 States that ratified the Covenant aim to progressively ensure that everyone has access to safe and secure drinking water, equitably without discrimination. Presently, 1.1 billion people lack access to an improved water supply and 2.4 billion to improved sanitation (UNESCO 2003). Both inadequate water supplies and sanitation are the underlying cause and outcome of the cycles of poverty and ill-health.

Within the next 20 years, the amount of water available per capita worldwide is expected to drop by a third (UNESCO 2003). This will be driven by continued anthropogenic disruptions of the global ecosystem (e.g., deforestation, urbanisation), resulting in the degradation and loss of freshwater. The increasing pressures on freshwaters and the expected reduction in availability has led water to be called “the next oil”, where scarcity and demand will fuel conflict and increase the global burden of disease. In 2000, waterborne diseases (e.g., giardiasis) and diseases caused by lack of water, poor personal hygiene and lack of proper human waste disposal (e.g., trachoma) killed 2.2 million people and affected more than 2 billion, the majority being children under the age of 5 (UNESCO 2003).

In trying to understand the relationship between water, health, and sustainable ecosystems, we must consider both the security of freshwater supplies and the quality of those supplies. The pressure-state-response model provides a useful tool for examining this relationship—we will discuss pressures on and the state of drinking water supplies in relation to human health, and possible responses to inadequate availability or quality of freshwater for human consumption.

As health researchers, we depend upon contributions from various disciplines in both the sciences and humanities to describe the pressures on and state of drinking water supplies, as is well illustrated by a diversity of other chapters in this book. The contribution of population health research comes to the fore in devising

appropriate responses, because such responses will ultimately depend on the level of health risk from drinking water that the population is prepared to tolerate. To quantify the health risk associated with the compromised availability or quality of drinking water is therefore a high research priority.

PRESSURES ON FRESHWATER SUPPLIES

The main pressure on the world's freshwaters arises, not surprisingly, through an imbalance between supply and demand. We consider freshwater to be a renewable resource, but it is also a finite resource. If a balance is to be maintained between its use and its renewal via the hydrological cycle, then the demand for freshwater must not exceed rates of supply. This is an obvious statement, however in many parts of the world this is not occurring. Although our "blue planet", as seen from space, is predominantly covered with water, less than 1% of this water is available for drinking: 97% of Earth's water is salt water, and more than two-thirds of the remainder is locked up in the polar icecaps (Mackenzie 1998). In addition, there are large disparities between global population densities and water availability. Asia is worst off, having the lowest volume of freshwater to population ratio, compared to Australia and Oceania with the highest ratio (Table 46.1). This situation will deteriorate as the world population grows. By 2025, at least 3.5 billion people, or 48% of the world's population, are expected to face water scarcity (Revenga *et al.* 2000).

The available reserves of freshwater are therefore limited, and importantly, not growing. The population, on the other hand, is growing and, seen simplistically, consuming an ever-increasing proportion of an ecologically limited resource. The imbalance arises from the technological advances that have accompanied our cultural evolution: whereas animal populations are generally limited by the availability of resources, the size of the human population and our water consumption rose sharply following the development of agriculture and urbanisation some 10,000

years ago. Water was used for irrigation, thus providing, for the first time in human history, a reliable source of food that permitted urban settlement and major population growth. This was the first epidemiological transition, and diseases of starvation and warfare were replaced by infectious diseases and specific nutrient deficiencies. We then became industrialised only some 200 years ago, and water consumption again rose because of industry needs and further population growth. Water use became distributed roughly as it is today, with 70% going to irrigated agriculture, 20% to industry, and 10% to domestic use. This second epidemiological transition was associated with another shift in the disease burden as a result of exposure to environmental hazards and toxins, and a more sedentary lifestyle and over-consumption. The global population has now swelled to 6 billion (10 billion by 2050; UNESCO 2003), all growing food and therefore further increasing water use. Between 1900 and 1995, water use worldwide increased six-fold, more than twice the rate of population growth (Revenga *et al.* 2000). Ironically, as we head into the 21st century, it is predicted that water availability will be one of the key factors that will limit development.

The increasing pressures on freshwater ecosystems, both via demand for water and ecosystem disruption by humans, has resulted in a loss of available water and a reduction in its quality, both for human consumption and in a wider ecological sense for "healthy" ecosystem function. The risk now is that water consumption will keep rising at a rate that is not ecologically sustainable, or, in other words, that it will be drawn off more rapidly than it can be replenished. Because biological systems are often not linear in their responses to such pressures, positive feedback loops can result where they have not been anticipated. In Mexico City, for example, groundwater has been drawn off so rapidly that the city has subsided several meters in the last decade (Lanz 1995). The result is a cracked and leaky water distribution system, leading to more rapid draw-off and further subsidence, and to the loss of a full one third of this valuable resource to leakage. Another example of a non-linear response is afforded by the advent of arsenic in drinking water wells in Bangladesh—when water tables were lowered through excess draw-off, some minerals became exposed to atmospheric oxygen, were oxidised, and released arsenic into the remaining groundwater (Frisbie *et al.* 2002).

In New Zealand, as overseas, demand for water for all purposes is increasing. Water is valued by New Zealanders for many reasons (Chapter 42), including economic (for irrigation and industry), environmental (maintaining ecosystems that rely on streams and groundwater), health (for water supply and safe swimming), cultural (mahinga kai and mauri), and recreation (for fishing, boating and

Table 46.1 Global population and freshwater availability.

Continent	Freshwater	Population	Water to
	volume		
	%	%	ratio
Asia	36	60	0.60
Europe	8	13	0.62
Africa	11	13	0.85
North and Central America	15	8	1.88
South America	26	6	4.33
Australia and Oceania	5	<1	5

Data from UNESCO (2003)

canoeing). New Zealand's low population and limited industrial base means that the current pressures on freshwater ecosystems are less severe than those in many other industrialised countries. However, the increasing demand for water (in terms of both higher quantity and quality) has brought about considerable public debate on how water is used, and how much is available for different purposes. Among the greatest impacts in New Zealand are hydroelectric power generation (Chapter 37), agriculture, and horticulture (Chapters 33 and 34). Water supply and demand factors will be of increasing prominence in agricultural investment decisions over the next 10 years and beyond. Other chapters have detailed discussions of individual pressures on freshwater ecosystems in New Zealand.

The pressures on freshwaters are not only an important economic issue, but they also have considerable public health implications, as in the example of arsenic contamination of groundwater in Bangladesh. As pressure builds upon the world's freshwater and other ecosystems, some researchers have argued that we have begun a third epidemiological transition—where the disease burden results from global ecosystem disruptions. The research literature on the health effects of climate change, the emergence of infectious diseases, and the disruption of social systems is extensive and tends to support this suggestion (e.g., Grifo and Rosenthal 1997; Aron and Patz 2001; Martens and McMichael 2002). The health impact

of the third epidemiological transition is likely to be felt hardest where resources, especially water, are most limited.

Human activities such as agriculture, industry and mining, and their associated inputs (run-off or byproducts) into freshwater ecosystems can directly affect human health (Table 46.2). Industrial developments frequently modify water catchments, and often result in the reduction in water quality or the complete loss of the water resource. Industries can affect water chemistry via SO₂ and NO₂ emissions, leading to water acidification, and via the addition of nutrients, toxic metals, organochlorine toxins (e.g., DDT) and organohalogens (e.g., PCBs). Additional health problems caused by human disruptions of ecosystems include waterborne diseases, e.g., *Cryptosporidium* outbreaks, increased *Giardia* prevalence, and possible waterborne transmission of *Campylobacter* (see next section).

An important and highly charged issue in New Zealand is the debate concerning dairy farming and the degradation of New Zealand's waterways and groundwater aquifers. This is a concern not only in New Zealand but globally—increasing stock levels, poor management practices and the clearing of riparian vegetation for further grazing allows high volumes of farm effluent, excess nutrients and chemicals to enter waterways, contributing to microbiological contamination and excessive nutrient levels. For example, in May 2000, *Escherichia coli* O157:H7 and *Campylobacter jejuni* contaminated the drinking water supply in Walkerton, Ontario, Canada (Anonymous 2000).

Table 46.2 Human activities and associated inputs into freshwater ecosystems with human health risks.

Activity	Inputs	Health Risks
Agriculture, Horticulture	Sediments Fertilisers Pesticides and other toxic chemicals and metals Animal faeces	Immune and endocrine disruption
Industry	Nutrients Toxic chemicals and metals Oils	Retarded physical and cognitive development Blue baby syndrome Fetal malformation/death
Mining	Sediments Toxic chemicals and metals	Nervous system and reproductive dysfunction Behavioural changes Cancers
Urbanisation, Infrastructure development	Sediments Pesticides and other toxic chemicals and metals Oils Sewage	Waterbourn disease
Recreation	Oils Fuel Toxic chemicals	

Six people died and over 2,000 people were ill as a result. Investigation of the outbreak traced the source of the disease to a contaminated well supplying the municipal water system. Evidence from molecular subtyping suggested that the pathogens originated from cattle manure on an adjacent farm. In New Zealand, the most pathogen-contaminated waters (e.g., with *Campylobacter*) are usually in intensively farmed areas, and waters where lots of birds live (McBride *et al.* 2002).

In addition, demands on water supplies and anthropogenic environmental change can have indirect health effects, by limiting the availability of safe drinking water. Not only are communities forced to access unsafe drinking water, but water scarcity also increases the risk of disease from lack of sanitation. This will be exacerbated by climate change. Although precipitation will probably increase from latitudes 30°N and 30°S, many tropical and subtropical regions are likely to receive lower and more erratic rainfall. It is estimated that climate change will account for approximately 20% of the increase in global water scarcity over the next 20 years (UNESCO 2003).

STATE OF FRESHWATER SUPPLIES FOR HUMAN USE

The definition of water quality is not objective and depends on the desired use of the water. Hence, water quality is defined as the physical, chemical, and biological characteristics of water necessary to sustain water uses (United Nations Economic Commission for Europe 1995). For example, water of poor quality for drinking could contain dissolved and suspended constituents above a level that affects human health, but conversely the water may be acceptable for industrial purposes. This definition is a purely anthropocentric one, and the one used in this chapter.

Worldwide water quality conditions have degraded in almost all regions with intensive agriculture and large urban and industrial areas (Revenga *et al.* 2000). However, global water quality data are scarce and tends to be limited to chemical rather than biological measurements, and usually limited to developed countries. This trend is changing, given the high global incidence of waterborne disease, to a greater emphasis on collecting microbiological data. Over the past century, chemical toxins and nutrients have increased in rivers in both industrialised and developing countries, with reductions in some pollutants such as metals and organic toxicants only in the last 30 years (Malmqvist and Rundle 2002). Pollution continues to be a major problem for many of the large freshwater lakes and wetlands around the world (Beeton 2002; Brinson and Malvarez 2002; Junk 2002). Algal blooms and eutrophication are being reported more frequently,

and of 82 major river basins, North America, Europe and Africa had the highest organic matter concentrations from 1976 to 1990 (Revenga *et al.* 2000). Nitrate pollution of groundwaters is getting worse in northern China, India and Europe (Revenga *et al.* 2000).

In New Zealand, both surface and groundwaters have declined in quality, and the natural character of many waterways has also been lost (Ministry for the Environment 1997). As in other parts of the world, agricultural land use has had a significant impact on freshwater quality and quantity, particularly in lowland streams. The decline in quality in these streams and the increased demand for freshwater are two of the most significant environmental issues facing New Zealand. Other chapters in this book discuss the state of New Zealand's freshwaters beyond that used for drinking and contact recreation.

The pressures on freshwater ecosystems and their resulting degradation have considerable public health implications. In the 19th Century the main health problems arose from faecal and organic pollution from untreated human wastewater. Today much of this contamination has been eliminated in industrialised countries, however, this is still a problem in rapidly industrialising countries, such as China, India, Mexico and Brazil (Shiklomanov 1997), and in most tropical countries (except Australia), where their wastewater treatment systems are in a precarious state (Junk 2002). In addition, the disposal of untreated effluent, e.g., marine and portable toilet waste, directly into the environment still occurs in industrial countries. Lest we forget, 3.5 billion people in mainly developing countries still lack improved sanitation or access to a secure supply of safe drinking water—a lethal combination for the transmission of waterborne diseases.

Microbial water quality

It was disappointing, but not surprising, that the latest cholera pandemic swept the globe only last decade. The re-emergence of this severe form of gastrointestinal disease occurred across the Western Hemisphere in the early 1990s (Carmichael 1997; see Response section for details of the cholera pandemic; and Okun (1996) for an historical overview of waterborne diseases). Thus, despite the best efforts of John Snow and his followers (see Response section), the microbial state of drinking water, from a global perspective, is not good. Waterborne diseases continue to be a major cause of mortality and morbidity across the world. In 2000, waterborne and water-related diseases killed 2.2 million people (most of these children) and affected more than 2 billion people (UNESCO 2003). It is likely that the reported numbers, although high, highly underestimate the real incidence of waterborne diseases. The large burden of disease is a direct result of water scarcity and poor water quality.

In faeces there are numerous classes of pathogens that cause infections, including bacteria (enteric and aquatic), enteric protozoa, and enteric viruses, which survive readily in the water environment and are strongly resistant to most disinfectants. Few data are available on the incidence of endemic waterborne disease in the developed world, however, it appears that there are now an enlarged variety of waterborne pathogens, many with a low infectious dose, and having moderate to high resistance to disinfectants. In addition, we are seeing the occurrence of waterborne disease without a faecal reservoir. These new pathogens include environmental bacteria that can survive and proliferate in water distribution and plumbing systems. For example, *Legionella* and *Mycobacterium avium* complex (MAC) are environmental pathogens that have found an ecological niche in drinking and hot water supplies. *Mycobacterium avium* complex frequently causes disseminated infections in AIDS patients—drinking water has been suggested as a source of infection, and in some cases the relationship has been proven (Leclerc *et al.* 2002).

On a global level, New Zealand has good quality drinking water, with the most common health problems associated with drinking and recreational freshwaters arising from microbiological contamination of the source waters (as is common internationally). But there is room for improvement, especially for rural supplies. In 2002 New Zealand had 2,060 water treatment plants and 2,138 distribution zones, servicing approximately 87.5% of the population. Of these, water supplies to 78% of the people complied with the *E. coli* criteria of the 2000 Drinking

Water Standards for New Zealand (DWSNZ; Ministry of Health 2000), and 80% complied with the *Cryptosporidium* criteria (Ministry of Health 2003). Thus, approximately 752,000 (22%) of people were supplied with drinking water that failed to comply bacteriologically with the criteria. In many places in New Zealand it is risky to drink untreated stream or lake water or use these locations for swimming and water sports. Microbial surveys of a number of swimming areas across New Zealand in 1994 revealed that over half the inland swimming areas were not fit for swimming, but suitable only for livestock watering (Ministry for the Environment 1997).

The largest outbreak of waterborne disease in New Zealand, affecting some 3,500 people, occurred in 1984 in Queenstown (Thorstensen 1985). In the absence of epidemiological evidence, it is believed that the public water supply was contaminated by a sewer overflow, which discharged sewage into a creek within 200 metres of the water supply intake. Given the source of the contamination, it is likely that a range of microbial pathogens were involved. By far the most important bacterial pathogen of concern in New Zealand is *Campylobacter*, which is the most common source of waterborne (and foodborne) illness. Since 1980, the number of notified cases of campylobacteriosis per year has steadily increased to 12,489 cases in 2002 (Fig. 46.1; Sneyd and Baker 2003). Campylobacteriosis has a complex ecology, and it is not clear to what degree waterborne sources contribute to levels of the disease in New Zealand (Eyles *et al.* 2003; Hearnden *et al.* 2003). The main aquatic sources are river waters,

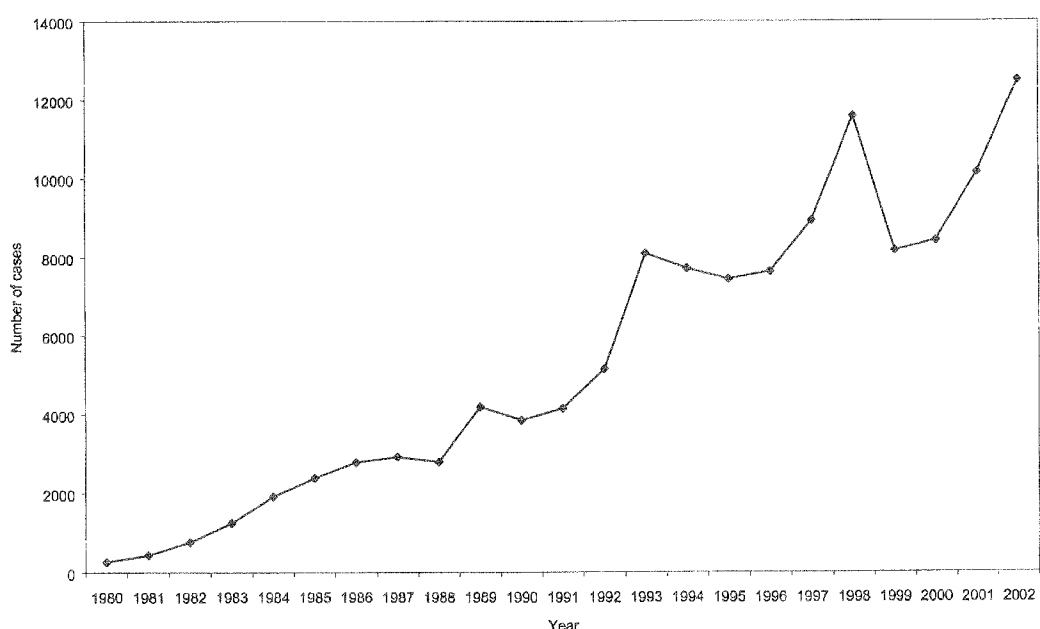


Figure 46.1 Number of notified campylobacteriosis cases per year from 1980 to 2002. Data from Sneyd and Baker (2003).

particularly in catchments containing sheep, where human contact via recreational use is a public health concern. McBride *et al.* (2002) estimated that 4% of notified campylobacteriosis in New Zealand could be attributable to water contact recreation. Low levels of *Campylobacter* can also be found in treated drinking water (Savill *et al.* 2001). Although the consumption of undercooked chicken has been identified as one of the main risk factors for campylobacteriosis in New Zealand, a number of outbreaks have been attributable to contaminated water sources (e.g., Brieseman 1987; Stehr-Green *et al.* 1991; Bohmer 1997).

The infection dose of protozoan and viral agents is lower than bacteria, in the range of one to ten infectious units or oocysts (Leclerc *et al.* 2002). One of the most common protozoan agents that cause gastrointestinal disease in humans is *Cryptosporidium*. Oocysts of this protozoan have been identified in human faecal specimens from more than 50 countries on six continents. In developed areas such as Europe and USA the prevalence of infection ranges from 1% to 4.5%, while in developing countries it is significantly higher, up to 20% (Chin 2000). One of the modes of transmission is via water, and outbreaks have been associated with drinking water and recreational water contact, including rivers and lakes. Oocysts are highly resistant to the chemical disinfectants used to purify drinking water, and advanced filtration systems are required to remove them. In 1993, an estimated 403,000 residents of the greater Milwaukee, Wisconsin, USA (population, approximately 1.61 million) became ill, and some 100 died, when an ineffective filtration process led to the inadequate removal of *Cryptosporidium* oocysts in one of two municipal water treatment plants (Mac Kenzie 1994). The documentation on the Milwaukee outbreak highlighted the need for better drinking water treatment, and the quality of public water systems have improved significantly since 1993 in the United States and internationally, but there is still need for improvement. For example, ultra-violet disinfection is becoming more popular in New Zealand, as in Europe, with the Ministry of Health recently proposing to establish a standard for the use of UV for disinfection of drinking water in New Zealand.

A second common protozoan agent, which is distributed worldwide and has a high burden of disease, is *Giardia*. The prevalence of infection for this protozoan ranges from 1% to 30% in different parts of the world, with the highest levels occurring in countries with poor sanitation (Chin 2000). This enteric disease is similar to cryptosporidiosis but is milder and treatable, generally self-limiting and less resistant to chemical disinfectants, such as chlorine. Over the last 30 years, giardiasis has become the most common cause of human waterborne disease in

the United States. It is associated with drinking water from unfiltered surface water sources or shallow wells, and recreational contact in bodies of freshwater. In addition to *Giardia* and *Cryptosporidium*, some species of the genera *Cyclospora* and *Isospora*, and of the family Microsporidia are emerging as opportunistic pathogens and may have waterborne routes of transmission (Leclerc *et al.* 2002).

In New Zealand, protozoa such as *Giardia* and *Cryptosporidium* are increasingly being highlighted as causing ill health through drinking contaminated water. Giardiasis is the most commonly notified waterborne disease in New Zealand, and the rates for this disease and for cryptosporidiosis are high compared to other developed countries (Table 46.3). Although giardiasis appears to have a relatively stable incidence rate in New Zealand, under-reporting is a problem in *Giardia* disease surveillance. This is in contrast to the incidence rate for cryptosporidiosis, which has increased significantly since it was made notifiable in 1996. Surveys of New Zealand's freshwaters indicate that both pathogens are widespread in the environment (e.g., Brown *et al.* 1992, 1998). Cases of giardiasis have been reported in users of national and forest parks, but the highest rates occur in urban regions. It is possible the former cases are a result of drinking contaminated water, but there is no epidemiological evidence to support this. In urban areas giardiasis is likely to include a large proportion of faecal-oral cases, but a number of cases have been linked to unfiltered or untreated water supplies (Fraser and Cooke 1991; Mitchell *et al.* 1993; Hoque *et al.* 2002). Again, in contrast to giardiasis, cryptosporidiosis tends to occur at higher rates in rural areas, probably due to animal contact, however waterborne transmission via water supplies or swimming pools are risk factors for the disease (Duncanson *et al.* 2000; Sneyd and Baker 2003). In terms of relating water catchment and treatment quality directly to disease outcomes in New Zealand, Duncanson *et al.* (2000) provide one of the few studies that consists of more than a risk assessment. By

Table 46.3 Rates of notified giardiasis and cryptosporidiosis per 100, 000 population in selected countries in 2001.

Country	Giardiasis	Cryptosporidiosis
New Zealand ¹	42.9	32.3
United States ²	9.5 (in 1997)	1.3
England and Wales ³	6.9	7.1
Northern Ireland ³	0.9	21.3
Belgium ⁴	13.1	6.0

¹ Sneyd and Baker (2003)

² www.cdc.gov/epo/dphsi/annsum/index.htm, Furness *et al.* 2000 – giardiasis is not a nationally notifiable disease in USA

³ www.hpa.org.uk/infections/topics_az/topics.asp

⁴ www.iph.fgov.be/epidemiol

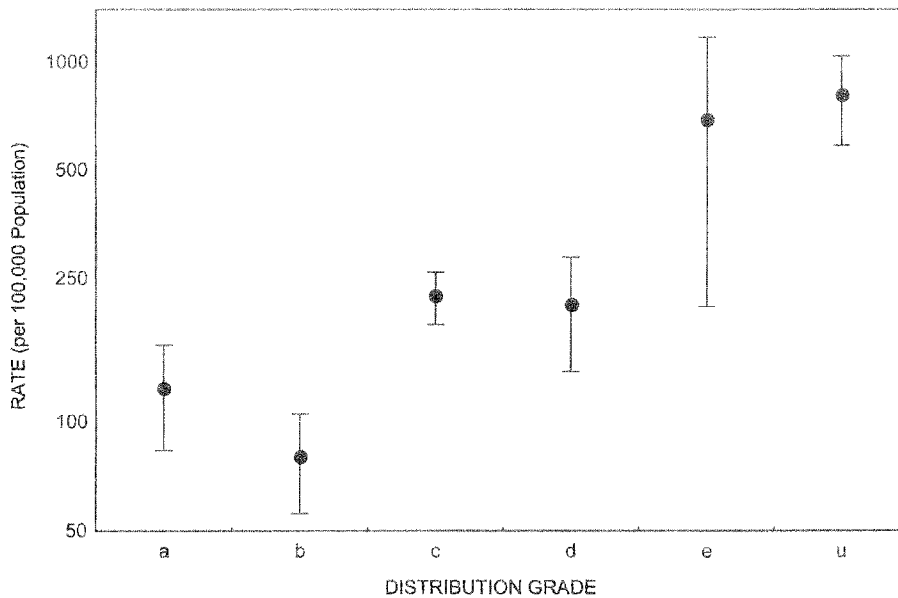


Figure 46.2 Rates of notified cryptosporidiosis per 100,000 population in New Zealand June 1996 to August 1998 by distribution zone component of the public health grading of community drinking water supply. a = very satisfactory, b = satisfactory, c = marginal, d = unsatisfactory, e = very unsatisfactory and u = ungraded. Log scale on Y axis. Error bars indicate standard error. Modified from Duncanson *et al.* (2000).

using the Ministry of Health water quality gradings, they demonstrated a direct correlation between water quality and the incidence of cryptosporidiosis (Fig. 46.2). Preliminary results by Ball *et al.* (2003), based on directly recording disease incidence in primary school children, indicate that poor quality drinking water supplies are a significant cause of gastrointestinal disease in New Zealand.

A further group of pathogens that are responsible for numerous cases of gastroenteritis worldwide are viruses, specifically Norwalk-like viruses (NLVs). In 2002 these viruses were reclassified into a new genus *Norovirus* in the *Caliciviridae* family. Molecular detection methods indicate that Norwalk-like viruses are the major culprits for food and waterborne nonbacterial gastroenteritis. In the United States, it is estimated that more than 60% of the population have antibodies to Norwalk-like viruses by their fifties (Chin 2000), while in developing countries antibodies are acquired at a much earlier age. Cases of gastroenteritis from Norwalk-like viruses most often occur in outbreaks rather than sporadically. For example, an outbreak of gastroenteritis due to a Norwalk-like virus occurred in a Swedish ski resort during February-March 2002, affecting approximately 500 people (Carrique-Mas *et al.* 2003). Epidemiological investigations indicated that one of the communal water systems was a significant risk factor, however, microbiological findings were inconclusive. It was not until a month later when a crack in a sewage pipe 10 meters from the well was discovered,

that an intervention was put into place.

Other viruses that are frequently transmitted via contaminated water are Hepatitis A (HAV) and Hepatitis E (HEV). Hepatitis A occurs worldwide, and is sporadic and epidemic, with a tendency to cyclic recurrences. In developing countries, adults are usually immune and epidemics of Hepatitis A are uncommon (Chin 2000). Ironically, improved sanitation has resulted in individuals lacking immunity, and the frequency of outbreaks is increasing. In contrast, Hepatitis E has a more limited distribution, mostly confined to tropical and subtropical areas, primarily in areas with inadequate sanitation. However, recently it is becoming an problem in countries where it was not traditionally endemic, such as in Europe (Worm

et al. 2002). Outbreaks of Hepatitis A and Hepatitis E typically follow heavy rains, when water sources become contaminated by sewage, or during dry periods when viruses are concentrated in contaminated water sources. Between December 1992 and April 1993, 3,682 individuals were affected by a major epidemic of waterborne hepatitis E in the city of Saharanpur, Uttar Pradesh, India (Singh *et al.* 1998). The source of the contamination was traced back to a leakage in the municipal water supply pipes, which passed through sewerage holes.

Although Norwalk-like viruses, Hepatitis A and Hepatitis E outbreaks have been linked to waterborne modes of transmission overseas, in New Zealand it appears that outbreaks have a predominantly foodborne origin (Sneyd and Baker 2003). However, gastroenteritis cases resulting from viruses are likely to be under-reported in New Zealand, and water supplies are not monitored for viruses.

Chemical water quality

As with the microbial state of drinking water, the chemical quality of drinking water on a global level is poor, particularly in developed and rapidly industrialising countries. In the 1970s, the United States Environmental Protection Agency found hundreds of organic chemicals in drinking water sources, many of which were believed to be carcinogenic and teratogenic (Okun 1996). Epidemiological studies in New Orleans at this time

revealed higher levels of cancer in individuals using the treated water supply versus those using untreated groundwater (Talbot and Harris 1974). The New Zealand Ministry of Health is currently aware of this problem, and it is to be addressed by the current Health Drinking Water Amendment Bill. This led to the passage of the Safe Drinking Water Act in the United States in 1974. At the same time on the other side of the world, Rook (1974) showed that the common chemical used for water treatment, chlorine, created disinfection by-products (DBPs) that were carcinogenic in rodents. To date, epidemiological data indicate potential developmental, reproductive, or carcinogenic health effects in humans exposed to disinfection by-products (Malcolm *et al.* 1999; Anderson *et al.* 2002). The data so far are inconclusive, and there is need for further research (see Research section). A further controversial topic is the occurrence of endocrine-disrupting chemicals (EDCs) in aquatic environments, particularly freshwater systems used for human drinking water. Pollutants that contain endocrine-disrupting chemicals include pesticides, dioxins, excreted drugs, alkylphenols, and furans, which enter the environment directly via agricultural or industrial activities or from treated sewage effluent. Although endocrine-disrupting chemicals currently occur in low concentrations, they may have significant health effects on aquatic organisms and humans (Melnick *et al.* 2002). Other chemicals are also of concern: there are a growing number of instances of groundwater contaminated by metal ions from natural and anthropogenic sources around the world, as illustrated by the almost inconceivably vast scale of arsenic poisoning on the Indian subcontinent (Frisbie *et al.* 2002).

Chemical contamination of New Zealand's freshwaters has occurred over a number of years as a result of both human activities and natural processes. Most chemicals mentioned in the DWSNZ (Ministry of Health 2000) specify levels likely to cause adverse effects only after prolonged periods of exposure, and monitoring suggests chemicals in drinking water supplies do not contribute a major health risk. That aside, there are cases where chemical pollution of freshwaters have occurred and may be detrimental to human health. For example, base-metal mining in Te Aroha, North Island, has resulted in concentrations of arsenic, cadmium, lead and zinc above the levels recommended for drinking, in the Tui and Tunakohoia streams (Sabti *et al.* 2000). In the Eastern Bay of Plenty, pulp mill effluent discharged into the Tarawera River for over 35 years has resulted in the accumulation of a variety of contaminants, including fatty and resin acids, and chlorophenolic compounds (Wilkins *et al.* 1996). Reports to the Ministry of Health in the 1990s highlighted that a significant number of water supplies, either in waters

before or after treatment, contained various levels of halogenated hydrocarbons, halophenols, polycyclic aromatic hydrocarbons and pesticides, and exceeded the standard guideline values. Natural sources of contamination of water supplies in New Zealand include elevated concentrations of mercury and arsenic from geothermal activity (some sources of arsenic have been linked to geologically very early geothermal activity), but this is limited to a small percentage of the population, mainly rural communities. Malcolm *et al.* (1999) estimated that for both cancer and birth defects the relative risk associated with exposure to disinfection by-products is about 1.5 and the population attributable risk per cent, for cancers and birth defects in New Zealand, is about 25%. This is equal to 139 preventable cancer deaths in 1995 and 94 preventable birth defects in 1996.

Thus, there are two reasons why the foregoing discussion cannot be dismissed as irrelevant to developed world countries such as New Zealand. Firstly, if we accept the earlier description of the third epidemiological transition brought about by ecosystem disruption, then it is clear that we must think at a global scale. Freshwater ecosystem disruptions at this scale will affect both developed and developing countries, much as will be the case for the health effects of climate change (Woodward *et al.* 1998). On one hand, waterborne outbreaks of bacterial origin (particularly typhoid fever) have declined dramatically since the 1900s in the developing world, but on the other hand we have had increased rates of waterborne disease in the developed world. Secondly, consider the emergence of *Giardia* and *Cryptosporidium* as a significant waterborne pathogen in communities served by state-of-the-art water treatment plants. These examples suggest that even the best "western standard" drinking water supplies can pose a health risk, and that the sound state of New Zealand drinking water is not one that we can keep taking for granted.

RESPONSE TO INADEQUATE SUPPLIES AND QUALITY OF DRINKING WATER

The re-emergence of cholera in the Western Hemisphere in the early 1990s was unwelcome news to all sectors of society, except, arguably, the media. This severe form of gastrointestinal disease results from infection with the potentially food or waterborne bacterium *Vibrio cholerae*, and the finger was rapidly and probably accurately pointed at the ecosystem disruptions that might have resulted in this public health catastrophe (including the recent upsurge in global travel and trade). Importantly though, this was indeed a re-emergence of a familiar disease that had first emerged some 200 years earlier: the first cholera pandemic originated in India in 1817 and spread

across Asia to Europe in the following decades (Carmichael 1997).

Although global travel and trade probably also contributed to the spread of this first cholera pandemic, it is unlikely at that time to have accounted for the emergence of the disease in the first place. A more likely scenario is a complex ecological and social disruption in the form of urbanisation. In earlier centuries, most of India was still sparsely settled, and semi-nomadic subsistence farmers formed part of a sustainable and cholera-free rural ecosystem (Watts 1997). Under the influence of British rule, these peoples were gradually concentrated into villages, many of which were in direct contact with environments that we now recognise as cholera reservoirs—the zooplankton of brackish water estuaries, such as the Ganges delta. In addition, such population concentrations were at greater risk of exposure to faecally contaminated food and water, creating opportunities for the evolution of a new microbial ecology with dire consequences for human health.

This pandemic occurred in an era before the development of the germ theory of disease, and medical intervention in the modern sense was therefore extremely limited. People who contracted the infection suffered severe gastrointestinal upset with vomiting and diarrhoea, resulting in the rapid loss of both salts and fluid. With a significant proportion of the population already weakened by famine, mortality often exceeded 50%, and tens of millions died (Carmichael 1997). The only effective medical response, seen from our perspective, was palliative care, and the more fortunate victims survived.

Responses at the individual level

By the time the latest cholera pandemic reached Peru in 1991, all of this had changed. Over a million cases had been recorded in South America by 1994, but mortality was kept below 1% (Chin 2000). Rehydration therapy was readily available, effective antibiotics (tetracyclines) had been developed, and tertiary medical centres were able to deal with the majority of complications. Medical interventions thus proved their worth, and there is no question that making western medical services widely available is one very effective response to the emergence of infectious diseases. Many readers may now be thinking about “public health” responses rather than “medical” responses and, inevitably, about the work of John Snow.

Responses at the population level

John Snow was a physician in 19th century London. He was Queen Victoria’s physician, but is most famous for his study of the 1853–4 cholera epidemic. This study was arguably the most significant milestone in the development of modern epidemiology, and demonstrated for the first

time (1) that cholera was waterborne, and (2) that an epidemic could be curtailed by public health intervention. His almost legendary map of London demonstrates cases clustered around Broad Street, where the water being drawn by the pump had become faecally contaminated (for a good map reproduction and historical account see Stolley and Lasky 1995). Although he was operating before the germ theory of disease, John Snow realised that the disease was waterborne, and had the pump handle removed to abort the epidemic. The provision of clean drinking water and adequate sanitation is now the mainstay of public health interventions to combat cholera and other waterborne diseases, and to some extent has the capacity to reverse the damage done by ecological disruptions in rural India over 200 years ago. In the current day and age this includes the installation of water treatment plants, which filter and chlorinate water destined for human consumption.

Since the 1960s New Zealand’s drinking water supplies were considered suspect. In the early 1990s, a quarter of the supplies surveyed failed the microbial standards set at the time, and a number of communities were advised to boil their water. The government responded via the Ministry of Health undertaking a programme to improve water quality management. The programme included the review of management procedures and legislation relating to drinking water supplies, with the publication of Guidelines for Drinking Water Quality Management, 1995 DWSNZ, and the Register of Community Drinking Water Supplies in New Zealand (see www.moh.govt.nz/water). The latter is updated each year by accessing water suppliers, based on the water quality standards criteria. For each zone (i.e. each area receiving similar water), a two-letter grading is designated, such as Aa, Cb, Ed, etc. The capital letter (A1, A, B, C, D or E) represents the grade of the water coming into the zone (i.e. source quality and treatment) while the lower-case letter (a, b, c, d or e) indicates the quality of the water received at your gate. Typically, if one tends to be high (A or B), so will be the other (a or b), but any combination is possible. The 1995 DWSNZ, which details the maximum concentrations of chemical, radiological and microbiological contaminants acceptable for public health in drinking water, has been replaced by the 2000 standards and is currently being amended for new standards in 2004. Over the years the Ministry of Health programme has resulted in a significant improvement in the quality of water supplies and the information available to the public on their drinking water. But we must not become complacent—although there has been an increase in the number of people who have access to safe drinking water, a number of water suppliers are still catching-up with the new standards. One possible response could be to make standards mandatory (see McKay and

Moeller 2002). We direct the reader to Chapter 45 for a detailed discussion of the New Zealand's government directed water quality management strategies and policies in response to water quality and supply problems.

Response at the community level

We saw earlier that a population weakened by famine was more vulnerable to cholera, and it is a reasonable assumption that any socio-economically deprived community will be more vulnerable to the health effects of ecosystem disruptions (Woodward *et al.* 1998). John Snow himself observed that, in addition to the quality of the water supply, there was an association between cholera and "poverty, and the crowding and want of cleanliness which always attend it" (cited in Cliff and Haggett 1988, p.7). His "want of cleanliness" preempts the importance of health education in combating cholera, in so far as good toilet hygiene and hand washing are essential elements that we in the industrialised world usually take for granted. Thus a complex set of actions at the community level operate synergistically to combat cholera or the risk of cholera and other waterborne diseases, and include interventions in nutrition, hygiene, and crowding, and addressing socio-economic deprivation generally.

In New Zealand, other responses at the community level include government and industrial initiatives to conserve water (see Chapter 45). None of our water uses were originally designed for efficiency, as water has generally been treated as an endless resource. Much emphasis has therefore been placed recently on improving this efficiency, and in Perth we have seen the introduction of Government subsidies for water-efficient showers, washing machines and other domestic water uses. Remember, however, that domestic use accounts for much lower water consumption than does agriculture, so the most significant water savings could arguably be made by improving the efficiency of agricultural irrigation. Low-pressure drip-irrigation, for example, can decrease the use of water by over half, and has the additional advantage of not creating a mosquito breeding hazard. It is likely that the year-round availability of standing water in irrigation channels and flooded areas has resulted in the establishment of an endemic focus of Murray Valley encephalitis in the Kununurra area of the Ord River irrigation scheme, Western Australia. Similar increases in potential vector-borne disease risk are likely to have been created in over-irrigated areas of New Zealand, which could harbour introduced mosquitoes. This is yet a further example of ecosystem disruption leading to flow-on health effects. Additional responses at the community level include the introduction of market mechanisms such as pricing to promote water efficiency, or a more draconian one (although often needed), is the introduction of water restrictions, as we have seen in Australia and New Zealand

in recent years. In Australia this has resulted in fines being issued to residents breaching the restrictions.

Response at the ecosystem level

Beyond the community lies the ecosystem that sustains that community. A healthy ecosystem provides services that enable the community to remain healthy and continue to exist on a sustainable basis. In this case, the provision of clean water from less disturbed or modified catchments avoids the health hazards of drawing water from, say, the contaminated River Thames in the 19th century. More recently, water treatment plants have of course facilitated the provision of microbiologically safe drinking water. However, there are recent warning signs that more emphasis is needed on maintaining ecosystem services. In the case of providing good quality water, these warning signs come in the form of: outbreaks of cryptosporidiosis from treated water supplies, the health risks associated with chlorination by-products, and the insecurity of the supply itself in cities like Mexico City, and Perth, Australia. Intensifying farming, in New Zealand lowland areas, is thought to contribute the majority of the contaminants found in rivers. Responses at the ecosystem level to address disruption to freshwater ecosystems include the re-forestation of water catchments, the planting of riparian vegetation along river banks (Weinstein *et al.* 2000; Vant 2001), and the construction of wetlands (Stott *et al.* 2001). Such responses have been shown to decrease the volume and speed of runoff during heavy rains, and increase the natural self-purification of water via soil filtration of contaminated stock excrement (including filtering off of protozoan oocysts), thus, improving the quality and security of drinking water supplies downstream. In New Zealand, the Ministry for the Environment has produced a guide to the sustainable management of riparian margins and for the protection and management of catchments e.g., the Ministry for the Environment's catchment grading, to help Council staff and farmers to improve their skills in managing their land (Ministry for the Environment 2001). The Ministry has also worked with Regional Councils to run a series of workshops training people in waterway management. However, there are presently no controls on catchment land uses, such as dairying or irrigation, with respect to drinking water.

Another response that would operate at the ecosystem level would be the limitation of population growth and urban development to levels that were sustainable with existing resources. Rome grew into the world's first city of one million people only by bringing water from surrounding areas through an architecturally magnificent series of aqueducts, thereby enlarging, like any city, its "ecological foot print". Although these aqueducts remain awesome feats of engineering, even 2000 years later, they

also represent the tendency of urban populations to outstrip the local natural resources essential to their own survival (Lanz 1995). This tendency has been exacerbated by the Judeo-Christian belief in man having dominion over all the earth, to the point where we now appear to have reached the third epidemiological transition (see Pressure section). It is tempting to suggest that city planners should heed sustainability considerations and put a cap on the population size, based on the limited availability of water (or other resource).

Certainly, the re-emergence of waterborne diseases as a significant health risk in developed countries has highlighted the importance of source water quality, and the protection and management of water catchments. An integrated catchment approach is required, one that incorporates all stakeholders within the catchment, community groups, government and local authorities to provide sustainable management of the ecosystem, i.e., ecosystem integrity (Parkes and Panelli 2001; UNESCO 2003). Thus, such ecological analyses and integrated responses can result not only in ecosystem integrity but also provide public health benefits, with a consequent reduction in the frequency with which pathogens contaminate water. A good example of this in New Zealand is the Motueka Integrated Catchment Management research programme launched in 2000 (<http://icm.landcareresearch.co.nz>). This is a collaborative effort involving Landcare Research, Cawthron Institute, Tasman District Council, University of Otago, GNS, Forest Research, Lincoln Environmental, and NIWA. This “ridge tops to the sea” perspective takes a holistic approach to large-scale regional environmental problems, particularly those concerning water. In the health industry, this is also related to the “water cycle”, in terms of taking into account upstream and downstream effects, water intakes and discharge points, and the recycling of lower grade water for appropriate uses such as watering gardens.

At a global level, we have recently seen water explicitly recognised as a fundamental human right (ECOSOC 2002). Although it is not legally binding, the 146 countries that signed the agreement will now be compelled to progressively ensure that everyone has access to safe and secure drinking water. It requires governments to adopt national strategies and plans of action to “move expeditiously and effectively towards the full realisation of the right to water”. The proof will be in the pudding, if it will ensure equitable access to water and reduce global levels of waterborne disease.

The example of cholera has provided the opportunity to explore possible medical and public health responses to ecosystem disruptions at a variety of levels. To devise an optimal response (or set of responses) to any particular ecosystem disruption, a detailed understanding of that ecosystem is needed, and it is natural, therefore, to draw

on the science of ecology. Ecology is the study of the distribution and abundance of organisms, and the interactions that determine their distribution and abundance. We can look at emergent infectious disease research as the study of the distribution and abundance of pathogens, and the interactions that determine their distribution and abundance. This is in effect ecology, and in devising our medical and public health responses, we can follow tried and true ecological principles (Weinstein 1997). Processes and interactions are usefully categorised as those affecting individuals (a simple organism), populations (many individuals of the same species), communities (a set of interacting populations of different organisms), and ecosystems (all interacting organisms and their environments) (Townsend *et al.* 2000). In discussing responses to the emergence of cholera and other waterborne disease, these same categories have been followed, moving from individual health care, to population health interventions, to community-based programmes, to the management of freshwater ecosystems. This is a useful and generalisable framework in which to consider the possible range of medical and public health responses to address the health effects of ecosystem disruptions.

RESEARCH NEEDS

As we highlighted in our introduction, the contribution of population health research comes to the fore in devising appropriate responses, because such responses will ultimately depend on the level of health risk from human use of freshwaters that the population is prepared to tolerate. As health planners need to allocate financial resources to provide a given set of population health interventions, the selection of an appropriate response or set of responses is unlikely to be made on a scientific basis alone. Ultimately, cost-benefit decisions about investment in the provision of safe drinking water will depend on the disease burden that the community is prepared to tolerate. Currently, the public are largely unaware of water-related health risks, however once they are made aware of them and the costs of reducing them, they may accept higher risks rather than paying for improvements.

In New Zealand, and in most communities in developed countries, we have a zero tolerance for waterborne cholera outbreaks. However, we accept relatively low risks of possible cancers and birth defects resulting from disinfection by-products in our drinking water (Malcolm *et al.* 1999; see State section). Malcolm *et al.* (1999) estimated that in New Zealand up to a quarter of all bladder, colon and rectal cancers, and birth defects, may be preventable by eliminating our exposure to disinfection by-products, although such estimates must be treated with caution because of uncertainties in exposure assessment.

Importantly, the health effects of disinfection by-products must be weighed against the cost of their reduction and not against the potential waterborne disease prevented by disinfection. However, further research is needed on the occurrence of disinfection by-products and their health consequences for a properly informed risk assessment and cost-benefit analysis.

With increasing public awareness of water quality problems, there has been an increased demand for higher quality water supplies. This is mirrored by the increasing global consumption of bottled water, the introduction of grading systems and classes of water for different purposes, and water re-use. This raises the dilemma of increasing quality versus the problem of economically achieving this quality. As water of good quality becomes scarcer, there will be increasing pressure to lower drinking water standards, and thereby possibly also increasing levels of disease. Evidence of this is already apparent in Europe, where EUREAU (European Association of Waterworks) has campaigned for the abolition of the “zero” standard for levels of pesticides in drinking water (Lanz 1995). The contention is that there is no scientific evidence that low-level exposure has any health effect—while this might be true at present, an informed decision clearly cannot be made without specific laboratory and population health research. If pressures to reduce drinking water standards lead to poor decisions about investment in the provision of safe drinking water, then the cost of the resultant increase in disease and community outrage may exceed the cost of retaining more conservative drinking water standards.

The relationship between microbial water quality and disease levels also needs direct quantification, rather than depending exclusively on risk assessment models that may not take account of unique aspects of New Zealand’s aquatic ecosystems. Duncanson *et al.* (2000) and Ball *et al.* (2003) have responded to this need, but we do not yet know enough to fully inform our approach to water management in New Zealand. The integrated catchment approach to water management, although appealing, requires considerable co-operation between stakeholders and must be based on sound research. “Sustainable management” is certainly a current buzz word, but there is considerable research yet to be done. This research will flourish with a cross-disciplinary approach incorporating fields such as environmental health, ecology and health, zoology, geography, and commerce (e.g., Parkes *et al.* 2003).

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Chapter 45

Managing water-related risks

David Painter

RISK MANAGEMENT

We all take risks every day—driving a car, using an appliance attached to mains electricity, leaving a wallet or purse in an unlocked room. Some people are more averse to risk than others, but we know what “risk” means. “Risk” also has a related, more precise, meaning as a technical term, combining the likelihood of something happening with the magnitude of the effect if it does. An earthquake of Richter magnitude 8 or more on the Alpine Fault of New Zealand’s South Island has a relatively low probability of occurrence (large quakes last occurred there about 1717, 1620, 1450 and 1100 AD), but the magnitude of the effect on the Southern Alps, alpine rivers, Christchurch city, other population centres and regional infrastructure such as roads and power supplies, could be very severe (Stirling *et al.* 2001). The “risk” is high, even though such events occur infrequently. On the other hand, the occurrence of landslips on to roads and highways after heavy rain is quite common in New Zealand. Usually the efforts of local contractors with loaders, excavators and trucks can restore traffic flow within a few hours or days. The magnitude of the effects is low, but they have a high probability of occurrence, again leading to quite a high “risk”.

Risk is not something physical; it is not synonymous with hazard. Risk is something social and cultural (Rees 2002). Managing water-related risks is done for social reasons such as human safety, and for economic reasons such as avoiding unwelcome and unnecessary costs. Risk and cost are related, but not simply. Some costs associated with high-probability risks clearly will occur at some time. Other costs, much higher perhaps, might or might not occur, depending whether a particular event takes place. Managing water-related risks to suit one sector of society, say farmers, might increase the risk for another sector, recreational fishers perhaps.

“Risk management” has, in the last few decades, also

taken on a quite precise meaning as a set of principles and methods that are now being applied to the management of risks of many kinds: financial; physical (hazard); military and health, as just a few examples (Keey 2000). It provides a way of managing diverse risks in a consistent manner. Water-related risks in New Zealand have been managed since Maori voyagers arrived here many centuries ago—well before “risk management” was defined. The management of water-related risks was one of the preoccupations of the settlers and colonists from the Northern Hemisphere from the mid-nineteenth century onward.

This chapter concentrates on the management of water-related risks in New Zealand recently and today, early in the twenty-first century. It is a period during which “risk management” principles and techniques are just beginning to be applied in resource and hazard management, including water resources and water hazards (Griffiths and Ross 1997). This is exemplified by the passing of the Civil Defence and Emergency Management Act (2002), which is firmly based on risk management principles.

For convenience, managing water-related risks is discussed here under headings related to categories of perceived risk. It is important to understand, however, that managing water-related risks is likely to be most successful if water resources are considered as systems, themselves embedded in physical and societal systems, and if the risks are considered in an integrated way. First described are risks associated with living things in water, and water in occupied places. Then the risks of excess and insufficient water are contrasted. Water risks in relation to energy, land movement and contamination are followed by the special risks from water as ice, snow and steam. Inevitably there are overlaps which result in some repetition in this linear exposition. And a lack of data on many risks ensures that the treatment is largely descriptive.

INTEGRATED WATER RESOURCE MANAGEMENT

To adequately discuss the management of water-related risks we must consider the management of water resources in more general terms. Groups of people in New Zealand society have diverse and sometimes conflicting expectations for water use, and managing water resources involves water-related risks to one group or another. The way in which water resources have been managed until the present has been somewhat piecemeal, with different agencies and groups being responsible for different aspects of water resource management.

Internationally, it has been suggested that there is an imminent risk of a “water crisis” (e.g., World Water Council 2000). According to the Stockholm-based Global Water Partnership: “The water crisis is mainly a crisis of governance. The present threat to water security lies in the failure of societies to respond to the challenge of reconciling the various needs for and uses of water.” (Global Water Partnership 2000). The response to this crisis, advocated by both the World Water Council and the Global Water Partnership, involves adopting the concept of “Integrated Water Resource Management”.

New Zealand has a relative abundance of fresh water (Painter 1990). Nevertheless, the water resources are managed in ways that are being labelled “unsustainable” and “fragmented” when used elsewhere. That in itself constitutes a risk to be managed. An important risk involving regional water resources is that future flexibility in their management as an integrated system could be prejudicially affected by prior resource allocation. There has been criticism of the Resource Management Act (1991)—that it encourages single-project, and first-in, first-served water allocation. Regional Councils have found it difficult to discharge their responsibilities under the Act towards sustainable husbandry of a region’s water resources. There has been debate whether this is a deficiency of the Act, or of the Councils. It seems that an Act whose purpose is to “promote the sustainable management of natural and physical resources” should be able to be interpreted in a way which does that in an integrated and systematic manner. But so far, integrated risk management of water resource systems has failed to be either integrated, or applied to whole systems in New Zealand (Painter 2000).

PEOPLE, ANIMALS AND PLANTS IN WATER

People

Drowning ranks third, after road accidents and accidental falls, as a cause of accidental death in New Zealand. Although annual deaths from drowning show a decreasing trend since 1985 (Fig. 45.1) there are still more than twice as many per 100,000 people as in the USA, Canada or Australia, and four times as many as in the UK (Water Safety New Zealand web site). This has been an unfortunate aspect of New Zealand’s history—drowning in early settlement times was called “the New Zealand death”, or simply “settlers’ disease”.

Water Safety New Zealand Incorporated is the organisation charged by the New Zealand Government with “ensuring that all New Zealanders are safe in and around water—at home, in public pools, at the beach, in lakes, in rivers or out at sea” (Water Safety New Zealand web site). Water Safety New Zealand’s philosophy is achieving water safety through education, training and actively promoting water safety awareness. Their main tools for influencing water safety, thus managing the risk, are web site, video, and printed and public display media, together with television advertising. They have three specialised web sites (accessible from the main web site) devoted to water safety in boating, swimming pools and rivers.

There is special national legislation (Fencing of Swimming Pools Act 1987) governing fencing of privately-

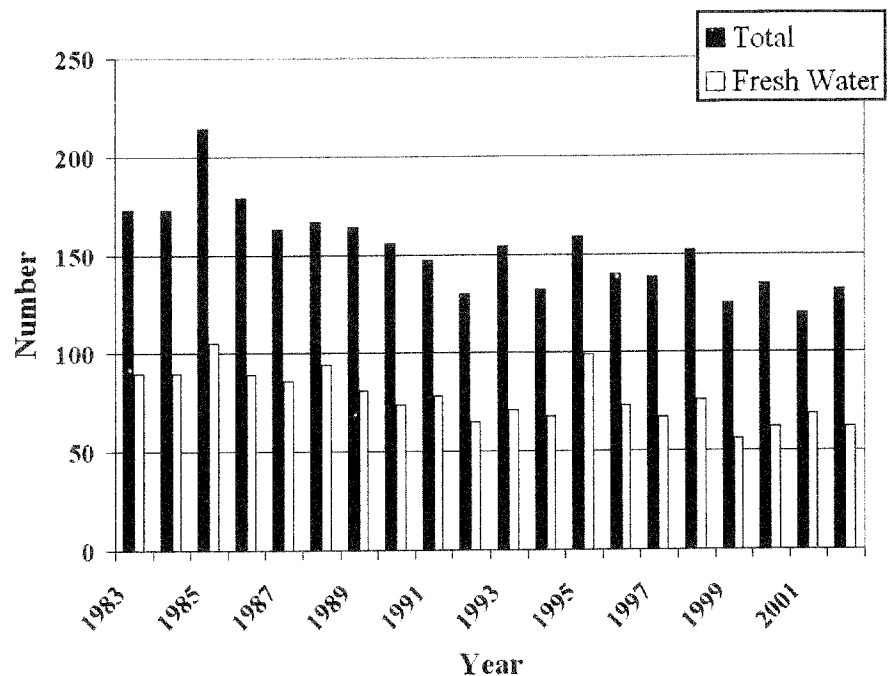


Figure 45.1 Drownings annually in New Zealand 1983-2002, total, and fresh water

owned swimming pools. This was enacted specifically to promote the safety of young children, particularly toddlers. The risk of small children drowning is intended to be greatly reduced by preventing them from having access to the pool area, unless they are with someone responsible, who is capable of opening a gate in the fence, which has to be an approved structure.

About a third of all drownings have occurred in rivers, but fewer than half of the people involved intended to be in the river. About 28% were in vehicles that crashed into rivers, 22% fell in from the banks and 2% were fishing (1983-2002 statistics, Water Safety New Zealand website). Many roads follow valley bottoms beside streams and rivers. Roadside reflectors and crash barriers, especially on bridge approaches and where roads run parallel to a stream or river, are intended to assist in keeping vehicles on the road and out of the water.

About 24% of those who drowned in rivers in the same period were engaged in swimming, canoeing or other boating, rafting or tubing, or diving or jumping in deliberately. A special category is those, such as trampers and walkers, who entered the river only to cross to the other side (6% of those who drowned). Eighty percent of those who drowned while crossing entered the river when it was in flood. Special courses available from alpine and tramping clubs provide instruction on crossing rivers safely.

Animals

It is common for farm livestock in New Zealand to graze freely up to the banks of unfenced streams and rivers. These sometimes form the boundaries between paddocks or properties. The natural or modified floodplain areas, and sometimes constructed floodways, can be grazed at all times other than during floods. Therein lies a risk to be managed. Advance notice of flooding is needed to provide time for farmers to move their livestock out of areas expected to be flooded. Some regional councils assist farmers in managing the risk to livestock by providing flood warnings on radio, web sites, or dedicated telephone answering services.

A very different type of risk is that associated with growth in the populations of unwanted fish species. These can overwhelm native and endemic species in a certain habitat, as well as having unwanted effects on vegetation and recreation. An example is the spread in the Waikato River and its tributaries of exotic koi carp, an ornamental strain of the common carp (*Cyprinus carpio*) native to Asia and Europe. Management of the risks posed by escape and proliferation of koi carp is the responsibility of the Department of Conservation. Management of their presence in ponds and lakes is primarily the responsibility of regional councils, such as Environment Waikato (see their web site).

Plants

There are risks associated with unwanted growth of plant species. The 1200-hectare Lake Omapere in Northland is a taonga (treasure) to the tangata whenua, Ngapuhi. The lake was used as a source of water for the Kaikohe region in the 1970s and 1980s. In December 1985 there was an outbreak of stomach disorders among residents whose drinking water came from the lake. There had been such prolific growth of the exotic *Egeria densa*, a larger and leafier version of the oxygen weed used in domestic aquariums, that the whole lake bed had been covered, with disastrous effects on most of the flora and fauna in the lake (Champion and Baldwin 2003). The stomach disorders were probably caused by an algal bloom, the result of the total collapse and decay of the weed beds.

Management of the weed problem has been the responsibility of the Northland Regional Council and Far North District Council. They enlisted the National Institute of Water and Atmospheric Research (NIWA) to provide scientific advice. Ironically, given the comments on risks from growth of unwanted fish species in the previous section, one successful weed management tool has been the introduction of 20,000 weed-eating grass carp (*Ctenopharyngodon idella*) in January 2002. Spraying the weed with herbicide was another part of the control strategy, but was not needed because the vegetation collapsed unaided, presumably as the climax of a “boom and bust” cycle. The use of carp was strongly opposed by the Royal Forest and Bird Protection Society. Among other risks they pointed out was the threat of extinction of the fern ally *Isoetes* aff. *kirkii*, a rare small native plant which is classified as being critically endangered and taxonomically indeterminate (unpublished media release by Eric Pyle of the Royal Forest and Bird Protection Society 2001). This species had probably disappeared from the lake prior to carp introduction, as a consequence of recolonisation of Lake Omapere by *Egeria densa* in the late 1990s and its subsequent collapse in 2000/2001. Any risk of the weed’s significant return is likely to be managed by using grass carp. Further restoration of water quality will be by “riparian management and other nutrient reduction strategies, and possibly the restoration of native submerged vegetation” (Champion and Baldwin 2003).

Genetically modified organisms

There would be a water-related risk requiring management if a genetically modified organism that survived or multiplied in water, or was spread by transport in water, were to be released in New Zealand. No such event has been recorded to date (2003). A moratorium on applications to introduce new genetically modified organisms into New Zealand was applied by Parliament from November 2001 to October 2003. The deliberate

introduction or development of organisms new to New Zealand is managed under the Hazardous Substances and New Organisms Act (1996). Inadvertent introduction is managed under the Biosecurity Act (1993).

WATER IN OCCUPIED PLACES

Some history

Flooding in occupied areas is the most costly natural disaster in New Zealand. The total average annual cost was estimated as approximately NZ\$125 million in 1986 (Ericksen 1986), a figure confirmed as “unlikely to have diminished”, in 1997 (Ministry for Environment 1997). The Insurance Council of New Zealand lists (on their web site) flood costs to the insurance industry of over NZ\$400 million from 1968–2001, an average over NZ\$12 million per year. This excludes contributions from the Earthquake Commission, which also contributes in some circumstances of land flooding. The direct cost to the central government of intervention in civil defence flooding emergencies is approximately NZ\$15 million annually (Ministry Of Civil Defence and Emergency Management 2002). Over 70% of all emergency declarations since 1963 have been flood-related.

Many New Zealand towns and cities are sited on the banks of rivers and streams. This is a result of the relative ease of access during settlement, the need for a reliable water supply and good access to crossing places. This siting provided access to water for municipal use, both as supply and to dilute liquid wastes including sewage. Insufficient water in occupied places will be considered below under “Urban Water Supply”.

But first the problem of too much water in occupied places will be considered. New Zealand rivers are rarely a constant-flow single channel which remains confined between two clearly-defined natural banks. Rather, during floods they regularly overflow onto their natural floodplain alongside the “baseflow” channel, or would do so but for river control works (q.v., below). So it is hardly surprising that river control and drainage were preoccupations from early settlement times. The earliest river boards or trusts started in 1868 in Marlborough and Hawkes Bay (Acheson 1968). There were both a Rivers Board Act and a Land Drainage Act in 1908 “to consolidate certain enactments of the General Assembly relating to the constitution of River Boards and the construction of river works”, and “relating to the drainage of land”. Later the Soil Conservation and Rivers Control Act (1941) was an important Act “to make provision for the conservation of soil resources and for the prevention of damage by erosion, and to make better provision with respect to the protection of property from damage by floods”. It was complemented,

and partly superseded, by the Water and Soil Conservation Act (1967), now repealed. The primary relevant Act now is the Resource Management Act (1991) and there are relevant provisions in the Building Act (1991).

There are two contrasting approaches to managing the risk that rivers may flood areas occupied by people and their property—to confine the river or to confine the people. These approaches are illustrated in the next two sections.

River control works

Riverbanks have commonly been occupied long before the natural habits of the river have been fully understood. “River control” was then attempted as risk management. River control schemes, as implemented from the 1940s until the 1970s, involved a whole package of central and local government financial grants and loans, and ratings for the local community share, with considerable experimentation, innovative engineering design and extensive physical works (e.g., see Acheson 1968).

River control includes built structures such as stopbanks, groynes (Fig. 45.2) and weirs, channel realignments and replacement channels, riprap and placed large rocks, and live streambank plantings such as willow trees (Fig. 45.3). There is an important paradox within floodplain management, including river control works—it could be called “the risk of managing risk”. River control works include engineered structures such as the stopbanks visible in Figure 45.4 (called levees or dykes in other parts of the world). These are designed to contain floods within the river, up to a “design flood” level. The implication, clear to engineers and hydrologists, is that floods appreciably greater than the design flood will not be contained.



Figure 45.2 Stub rock groynes and pole planting upstream of a highway bridge, Hurunui River, North Canterbury.

Photo: River Engineering Section, Environment Canterbury

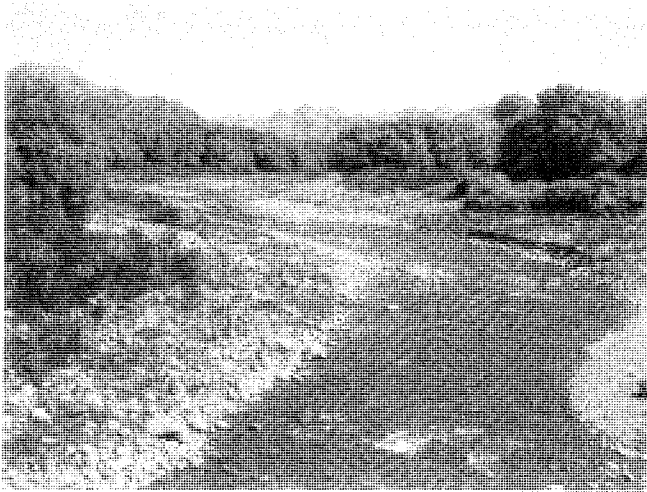


Figure 45.3 Bankside willows and channel shaping, Chatterton River, North Canterbury.

Photo: River Engineering Section, Environment Canterbury



Figure 45.4 Stopbanks and berm floodways form part of the river control works where the Waipa River passes through Otorohanga, King Country, North Island.

Photo: David Painter

This is not always clear to would-be occupiers of the floodplain, including owners of residential dwellings, business owners and even elected councillors. It is quite well documented in New Zealand and internationally that the value of the assets at risk quite often increases in line with the expenditure that has occurred on flood protection (Ericksen 1986; Painter 2000).

The Lower Waikato and Waipa Flood Control Scheme was constructed between 1961 and 1982 (a small part of it in Otorohanga township is illustrated in Figure 45.4). It was put to an interesting test in 1992. A landslide dammed a tributary upstream (Tunawaea Stream) in August 1991. Prior to the dam's failure in July 1992, it was perceived that there was a risk that a rainfall-induced dam-break flood in the tributary could coincide with a flood in the Waipa River caused by the same storm, and that the river

control works protecting Otorohanga could be overcome. The risk was managed by the regional council, local authority and others, using a variety of methods.

The state of the dam, the water level and the rate of leakage through the dam were monitored. All dwellings and farm buildings potentially affected by a flood wave were identified. Bridge waterways and other obstructions to flow were inspected. Computational hydraulic modelling was carried out to predict likely discharges and water levels (Webby and Jennings 1994). The overall prediction was that a flood wave would be contained within the river, in particular where it passes through Otorohanga. There was regular communication with the Otorohanga townsfolk, and more particularly with people living alongside the river in rural areas upstream of Otorohanga but downstream of the Tunawaea tributary junction. When the dam did fail, in July 1992, the predictions of safe passage of the flood were borne out.

Floodplain management

The second approach to managing the risk of rivers flooding areas occupied by people is to confine the people to areas away from where the river might go during a flood. The essential elements of this approach are mapping, planning and zoning. For example, areas can be mapped which might be suitable for recreational use, such as a golf course, but where residential building is not permitted. Then it is up to the regional and local authorities to publicise, legislate for, and enforce the rules. A degree of regulation is involved, by central, regional and local government, which has implications for citizens' views on personal freedom. A more recent development in New Zealand is floodplain flow modelling (Connell *et al.* 2001), to indicate where floodwaters can be expected to go, and their depth, in a flood of a given magnitude.

Floodplain management has become more common in New Zealand since the 1970s. In particular, it has been encouraged by the approach to integrated resource management exemplified by the Resource Management Act (1991). Floodplain management plans result from a process which involves a "flood hazard assessment", based on existing knowledge and relevant research, development of options by an authority and its community, consideration of flood control (structural) and non-structural (e.g., zoning) options, and a benefit-to-cost comparison for a predicted level of flood risk.

The two apparently mutually exclusive approaches to floodplain management outlined above are a simplification both of what actually occurs and what might ideally occur. Morrison (1995) has provided a rigorous and comprehensive doctoral thesis that presents, *inter alia*, methods to allow co-evolution of flexible floodplain management and the floodplain itself. But no current

authorities have come close to implementing his recommendations.

On some occasions river control works have been built, but when the risks of floodplain occupation have been later evaluated, they have been proven to be greater than earlier believed, or perhaps greater than experts and authorities consider compatible with human safety. The town of Franz Josef in Westland is adjacent to the Waiho River. The Callery River tributary to the Waiho has been considered to pose a considerable risk to Franz Josef from a possible natural landslide dam formation and dambreak flood (Davies and Scott 1997). Even without this occurring, the Waiho River sediment and floodwaters have already caused expensive problems of river control and forced the closure of businesses. Appropriate management of the risk could involve relocation of part or all of the town (Davies 2002, OptimX 2002).

The management of risk for occupied lakeshores involves problems similar to those of floodplains of rivers and streams. On long time scales, river and lake overflows onto their floodplains have positive effects, including the development of fertile soils. In the medium term, risk management is needed when a lake level is raised slowly and deliberately, as when the lake has been modified for use as a hydro-electric reservoir. Short term, ongoing risk management is needed, similar to that for riverside communities, when lake levels fluctuate according to the vagaries of catchment rainfall. When the lake is a tourist destination as important as Lake Wakatipu, and the town has significant commercial property below recurring lake levels, as Queenstown has, the risk to be managed can be very high (Fig. 45.5). Managing water-related risks to property—well illustrated by the Queenstown example— involves financial tools such as insurance. Property values



Figure 45.5 The central business district of Queenstown, with Lake Wakatipu in flood in November 1999.

Photo: Southland Times

might also signal risk and the potential cost of flooding to intending purchasers.

Flood forecasting and flood warnings

Flood warnings were briefly mentioned in the section on “Animals”. The term “flood prediction” is usually applied to the “likelihood of something happening”, i.e., in a probabilistic sense. Phrases like “the one in a hundred year flood”, or preferably, “the flood of 1% annual exceedance probability”, are used in talking about flood prediction. “Flood forecasting” is a term usually applied to the real-time estimation of flood discharge or level in the immediate future at a particular site. This can be based on information such as the immediate past and present levels at the site, levels at sites upstream up to the present time, rain that has fallen in the catchment up to the present and the forecast rainfall, and experience of previous floods in the catchment. Flood forecasting is usually aided by telemetered hydrometric information, and sometimes by computer modelling.

Flood forecasting and warning has been increasingly common practice in New Zealand in the last few decades. It has been beneficially used to forecast floods on the Wairau River in Marlborough, South Island (Rae and Wadsworth 1990), and on many other rivers in both North and South Islands (Ibbitt et al. 2001; McKerchar *et al.* 1997). Channel routing models project discharge information measured upstream to a downstream location of interest. Rainfall-runoff models do a similar projection, but based on upstream measured catchment rainfall. In more recent years, forecasting has used conceptual computer modeling, with real-time measured rainfall and discharge, e.g., for the Clutha River, South Island (Ibbitt and Woods 2003).

Davies and Hall (1992) have suggested that risk management for alpine disasters (which could include flash floods in occupied places) is not accomplished well by simply combining “the likelihood of something happening” with the “magnitude of the effect if it does” in the form of a “design event” and a “benefit : cost analysis”. They propose that smaller events should be controlled by structural means, and all events larger than some chosen size always managed by warning and evacuation. They provide an interesting example of a two-stage warning system put in place for the Blandswold holiday home settlement on the Kowai River near Little Mt Peel in South Canterbury, New Zealand. A Stage-1 alert begins whenever antecedent 15-day rainfall exceeds 70 mm; residents are informed that a “state of readiness” exists. Synoptic weather monitoring begins, to allow forecasting 12 hours in advance of any intense rain on the catchment. Stage 2 is invoked if such an event seems likely; an evacuation with ample time and in daylight then occurs.

Urban water supply

The human body is 60% to 75% water. Adults typically lose 2–3 litres of water per day and should consume 0.9–1.2 litres directly as non-alcoholic, non-caffeinated fluids daily. Most of us know that severe dehydration can be fatal. Less-known health risks from chronic slight dehydration include urinary stone disease; breast, colon and urinary tract cancer; childhood and adolescent obesity; mitral valve prolapse; salivary gland malfunction; and overall poor health in elderly people (Kleiner 1999). Accordingly, public water supply is a basic service, provided or regulated, in New Zealand as in other well-governed countries.

In addition to direct human consumption, water is needed in communities for hygiene, commerce, industry, gardening and recreation, fire-fighting and a host of other reasons. It is the responsibility of local government to manage the provision of water in cities and towns and to manage the risk of failure of the supply. Prior to the water supply crisis that afflicted Auckland cities in 1994, the management by local authorities of water supply risk was haphazard. The “drought management plan” produced during the crisis (by Watercare Services Ltd) may well have been the first such plan produced in New Zealand. More generally, asset management for municipal water supply is now moving to a risk management approach (Bull 2000). Together with stormwater and wastewater disposal, local authorities in New Zealand have about NZ\$7500 million invested in water infrastructure. They spend about NZ\$600 million on operational costs each year and will need to spend about NZ\$5000 million upgrading this water infrastructure in the period 2000–2020 (Parliamentary Commissioner for the Environment 2000).

Integrated risk management (Painter 2000) requires that water supply authorities consider possible hazards related to water supply catchments, both surface water and groundwater, and the distribution systems downstream of water reservoirs and treatment plants.

EXCESS WATER

Some aspects of the occurrence of excess water, as extreme rainfall and river floods, have been covered in previous chapters (see especially Chapters 2 and 10) and in the preceding sections here on river control works and floodplain management. Such occurrence is the “likelihood of something happening” referred to in this chapter’s introduction as the first half in the definition of “risk”. To complete a consideration of risk, it is necessary to combine the “likelihood of something happening” with “the magnitude of the effect if it does”. Many of the high-magnitude effects occur when heavy rain raises the levels

of rivers and lakes, affecting occupied floodplains and lakeshores. Others relate particularly to flooding in urban areas, impacts on transport routes, and the safety of dams and reservoirs.

Stormwater and drainage

Urban areas of New Zealand have a considerable infrastructure to manage the risk that extreme rainfall will lead to stormwater in excess of what can be locally absorbed, ponded for subsequent evaporation, or put to good use on site. Both capital and operational aspects are involved in managing stormwater risk in urban environments. There is a need to deal with stormwater that has been contaminated by sewage or hazardous wastes, especially where urban authorities still have combined pipelines for sewage and stormwater. Health hazards arise from the discharge of such stormwater, especially to beaches used for swimming or shellfish gathering. These hazards are usually managed by public notices on-site, and through warnings to the public via print, radio and television media.

Surface and sub-surface drainage of farmland (Bowler 1980) is a well-known practice used as a form of risk management. Similar principles (but at a much greater cost per unit area) apply to managing the risk of excess water on sports grounds and urban recreation areas. The drainage and turf system installed in 2003 at Auckland’s Eden Park (for rugby union and cricket games) cost about NZ\$4 million for less than a hectare of turf (New Zealand Herald 2003). Farm drainage would typically cost about 0.1% of that.

Road and rail bridges

High discharges and water levels in rivers can undermine and wash away the banks and stopbanks of rivers and any structures within or crossing them. Bridges are usually designed to have a waterway area beneath them that is capable of passing a flood of some particular likelihood of occurrence. For example, a flood discharge with an annual exceedance probability of 2% might be used for major highway bridges. That implies that floods larger than the design flood might exceed this capability, thus risking damage to the bridge and danger for traffic. There have been fatalities and numerous injury accidents in New Zealand as a result of bridges being damaged by floods: the accidents have occurred either while road or rail traffic has been in transit, or before warning signs and barriers could be put in place. Figure 45.6 provides a recent example; fortunately the train driver escaped from the cab. The risk to vulnerable bridges could be managed by fitting monitored warning devices, but this is not current practice in New Zealand.

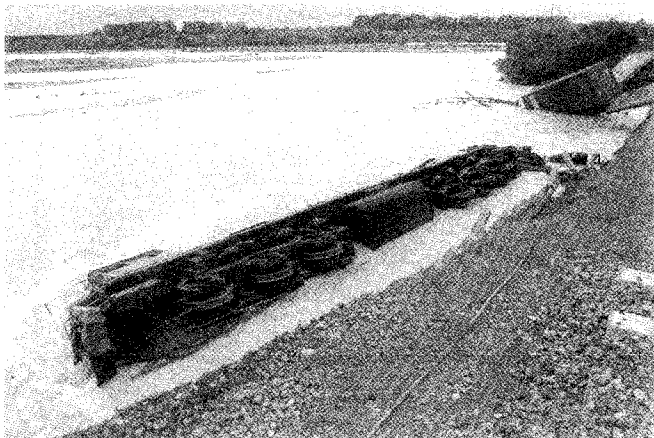


Figure 45.6 Scour failure of the southern approach to the Rangitata River Railway Bridge, 4 January, 2002.

Photo: The Press, Christchurch

Dams and reservoirs

Building dams in rivers with a substantial water catchment upstream poses a high risk for a short time during particular stages of the construction. This risk usually is managed by providing for diversion of the inflow, but a trade-off is often required between the cost of diversion structures and the magnitude of flood flow that can be catered for. A large flood (about 5% annual exceedance probability) occurred on the Rangataiki River, Bay of Plenty, North Island in July 1998, during strengthening of the Matahina dam earth embankment. A specific risk management plan had been implemented for the duration of the project and it was called in to play during the flood event (Keane 1999). The flood volumes were eventually contained and spilled without overtopping the dam, but only just.

A less fortunate outcome resulted from a storm in the catchment of the Opuha Dam, South Canterbury, in February 1997. A diversion culvert was in place while the main embankment was being built. The main spillway was 80% complete, but not usable. When the flood flows threatened to destroy the partly completed embankment, the main contractor “cut a channel in one side of the dam to avoid the water spilling over it. However, the force of the flood water through the channel caused roughly one-third of the compacted fill material that had already been placed in the dam to be eroded and washed downstream—causing damage to roads, fencing and river protection works” (Office of the Auditor General 2001).

Reservoirs can fulfill an important role in attenuating flood peaks. Flood detention dams are built specifically for this purpose, usually as part of urban stormwater management. Other reservoirs, such as hydro-electric storage reservoirs and outflow-controlled lakes, can have designated “buffer storage”, whereby the water level is

lowered at certain times of the year in expectation of the need to accept flood inflows and provide reduced outflows. For example, control gates on Lake Taupo are used to partially manage flood risks on the Waikato River (Barnett *et al.* 2000). But in a contrasting example, sediment deposition in Lake Roxburgh since it was formed in 1956 eventually caused flooding upstream in Alexandra, particularly in 1994, 1995 and 1999. The risk had been known, but both the rate of deposition and the flood magnitude frequencies in the Clutha River were underestimated by the designers of Roxburgh dam. A flood remediation project, involving both structural works and changes in land use, was commissioned in 2002. The NZ\$22 million cost included a NZ\$6 million contribution from the hydro-electricity generating company involved (Contact Energy Ltd).

The spillway capacities of dams and reservoirs are continually assessed in the light of improved meteorological and hydrological information on flood magnitudes, and changed standards of safety. Changed information in the 1999 edition of the influential Australian Rainfall and Runoff (Institution of Engineers Australia 1998) and changed guidelines issued in 2000 (Australian National Committee on Large Dams 2000) have led to much activity and expense in increasing spillway capacities on many dams in Australia. Revised guidelines have also been issued in New Zealand (New Zealand Society on Large Dams 2000), but have not resulted in the same level of spillway upgrading.

Managing the risk of damage from flood discharges on a river that has a series of dams in place has been outlined for the Waikato River (Barnett *et al.* 2000). There are risks to river users, riparian land owners and residents, dam operators, public utility operators (e.g., water treatment plants) and electricity users. Management of these risks requires, among other things, understanding of the detailed interactions between catchment rainfall and tributary inflows, the effects of travel time for releases from one dam reaching the next, the automated and human control systems, and the human dimension of the times, places and purposes of people using the river.

INSUFFICIENT WATER

Lower-than-expected rainfall over an extended period of time (“meteorological drought”) leads to streamflow, lake and reservoir levels, soil moisture and groundwater aquifer levels all being lower than expected (“hydrological drought”). Risks arise for ecosystem health, primary industry, energy and extraction industries, and the general public when the period of little or no rainfall becomes extreme (see Chapter 10).

The occurrence of an extended period of little or no

rainfall is the “likelihood of something happening” referred to in this chapter’s introduction as the first half in the definition of “risk”. The second half is “the magnitude of the effect if it does”. In the case of insufficient water, it is not the absence of rainfall itself that gives rise to risk, but the cumulative effect on streamflow and other water levels. Unlike the effects of excess water, which are often most severe in occupied places, and occur over hours, days or weeks, the effects of insufficient water are often most severe in rural or wild areas and occur over months or years.

Agricultural drought

“Agricultural drought” is said to occur when soil moisture levels fail to meet the water requirements of crops over an extended period. The financial risks associated with agricultural drought in New Zealand are very high. Droughts in 1997–98 and 2001–02 are thought each to have cost New Zealand several hundred million dollars (McKerchar 2003). Just in the Canterbury region, the estimated total net impact of the 1997–99 drought on the Canterbury economy was NZ\$281 million—over 2% of the total annual Gross Domestic Product for the region (Ministry of Agriculture and Forestry 2000). Individual farmers are engaged in risk management during most of their work, but especially so prior to and during drought. Reserved feed, irrigation, stock water and irrigation water supply reservoirs, access to off-farm grazing, drought-resistant crop varieties, cultivation practices to counter wind erosion of soil—managing water-related risks in farming could occupy a book on its own (e.g., see Ministry of Agriculture and Forestry 1998). Another episode of drought occurred in summer and autumn of 2003, with record low streamflows in much of the west and south of the North and South Islands (McKerchar 2003). One lesson from international experience is that more emphasis in risk management should be placed on preparedness and mitigation, complementary to response measures implemented once a drought has taken hold (Wilhelmi and Wilhite 2002).

One unusual financial tool for risk management available to New Zealand farmers is an Adverse Event Income Equalisation Scheme, administered by the Inland Revenue Department. This allows unforeseen income generated when, for example, drought forces farmers to reduce livestock numbers, to be placed in a tax-free account for use later, when re-stocking after the drought.

Irrigation water supplies

Irrigation can be a risk management tool for farmers. But farmers do not use irrigation solely for drought. As an integral part of farm management, irrigation enables dramatic productivity increases by removing a major limitation on crop growth, in a similar manner to fertilizer

counteracting low soil fertility. With irrigation, there is then a risk transfer. A new risk arises from the likelihood of restrictions in the supply of irrigation water, and the consequences to dependent crops and livestock. On a regional basis, the use of water for irrigation by farmers can translate into depletion of surface and groundwater resources. This might well add to an existing hydrological drought, with mutual effects among recreational water users, hydro-electric energy generators, recreational water users and aquatic ecosystems.

The risks posed by restricted irrigation water supplies have to be managed by individual farmers. The restrictions may be physical, as supplies dry up, or legal, as minimum flow or water level conditions on consents to take water are invoked, restricting extraction. At the level of an irrigation scheme or company, the managers might be able to manage risk by using alternative supplies or by price variations to users. In New Zealand, at a regional level, the regional councils and unitary authorities under the Resource Management Act (1991) must balance the “downstream” risks to irrigators with the “upstream” risks to other users and the environment, as water supplies become depleted. The risk of insufficient water for irrigation is clearly just part of a more general risk of insufficient water for all uses, including ecological and aesthetic values, in a region.

Groundwater

Low water levels in groundwater aquifers can pose significant risk, due in part to the relatively slow rate of recharge in some aquifers. Contamination of groundwater is also likely to be high-risk, due both to the often very high quality of groundwater (e.g., suitable for drinking), and because decontamination can be slow and difficult once contamination has occurred. Christchurch city in the South Island enjoys the rare benefit of a municipal water supply from aquifers in which the water is of such high quality that it requires no treatment at all before entering distribution pipelines. Recharge of the aquifers underlying the city occurs from rainfall on the Canterbury Plains inland from the city, and from leakage of river water, particularly from the Waimakariri River. Thus contamination risks to the Christchurch city supply relate back to the likelihood of contamination of the river water, or to water passing through the soil and underlying strata in particular areas of the Canterbury Plains.

This example from a particular location illustrates two general points about groundwater risk management that are applicable elsewhere. First, the “source” and “extraction” locations for a groundwater aquifer can be separated by distances greater than those of farm properties, local authority boundaries or even regional authority boundaries. Secondly, the groundwater being

extracted may have entered the aquifer months, years or even millennia before. The Moutere aquifer near Nelson, South Island, has been found to contain “high quality glacial-age water more than 20,000 years old” (Stewart 2003). Managing risks of groundwater depletion and contamination can therefore involve time and space scales much greater than those considered for surface water risk management.

Wetlands

Chronically insufficient water destroys wetlands. When drained, wetlands in New Zealand are potentially “productive” land for agricultural or urban development. In the 165 years since the Treaty of Waitangi, about 85% of New Zealand’s wetland area has been “developed” in this way (Parliamentary Commissioner for the Environment 2002). The value of the remaining wetlands, to wildlife, species biodiversity and for aesthetic reasons as landscape elements, has been recognised in the last few decades. Managing risks from wetland degradation or disappearance involves securing their status, ensuring a water supply of appropriate quantity and quality, and educating the public about their value. Recently, natural wetlands have been joined in New Zealand by wetlands built for wastewater treatment.

Reservoirs and dams

Surface water storage in ponds, reservoirs and lakes can be very important in managing mismatches in supply and demand. Water in storage in reservoirs, including groundwater aquifers, is thus an important aspect of coping with low supplies. The water supply reservoirs in the Hunua and Waitakere Ranges that supply Auckland are a good example of several such systems in New Zealand. Reservoirs built for one purpose may provide additional benefits, such as flood mitigation. Often the operation of multi-purpose reservoirs involves compromises among objectives and thus involves risk management. Otherwise the risks associated with reservoirs, including drowning, hosting unwanted species and having poor water quality, are no different from those of pools, ponds and lakes.

WATER AND ENERGY

The most obvious connection between water and energy occurs when the potential energy of water at a high altitude is harnessed by passing it through turbines as it moves to a lower altitude—hydro-electricity generation. Managing the water-related risks of generating electricity from a dominantly hydro system (about 60%–70% of generation in 2000 in New Zealand) is a topic worthy of its own book, especially because there have been many changes in the way the national electricity system has been structured and



Figure 45.7 Lake Pukaki with Aoraki Mt Cook in the background. The lake has the largest single hydro storage capacity in New Zealand and feeds water to Ohau A and downstream power stations.

Photo: David Painter

managed in recent decades. There are potentially many lessons, and examples both good and bad, for other sectors of the economy engaged in managing water-related risks. Only very brief comment on the main points is given here.

The most significant water and energy-related risks in New Zealand in recent decades have been the financial, health and safety, and political risks associated with reduced electricity supplies caused by low hydro lake inflows, especially in the lakes of the Upper Waitaki catchment (Fig. 45.7). These risks have been demonstrated when they caused well-publicised and well-documented “crises” in 1992 and 2001 (e.g., Electricity Shortage Review Committee 1992).

Prior to 1992 it was intended that the then-integrated national grid would cope with a “one-in-twenty dry year” without power blackouts anywhere in New Zealand. The Electricity Shortage Review Committee (1992) concluded that the risks from this policy were too high. They recommended that a “one-in-sixty dry year” policy should be adopted. Such a policy was set aside when a wholesale electricity market was introduced in New Zealand in October 1996 and the former Electricity Corporation of New Zealand, a state-owned enterprise, was split into three generators in April 1999, to operate alongside private and public companies offering electrical energy to the market. It was assumed that security of supply would be ensured by price-response mechanisms.

That security of supply has not been ensured, and this has been the cause of much debate by the electrical generation industry, electricity users both major and individual, government, media and other commentators. In 2003, the government announced its intention to fund a dry-year reserve generation plant. This plant will not be

used in “normal” hydro reservoir inflow years but is intended to secure the 1992 recommended “one-in-sixty dry year” security level. It is an extreme form of supply assurance, whose worth it will be interesting to evaluate after a few years of non-operation.

Energy is also generated in New Zealand from water in the form of geothermal steam (about 4% in 2000). Thermal generation of electricity using coal, natural gas, landfill gas or various petroleum fractions also requires water, as steam for the turbines and as cooling water. Some of the water-related risks are significant. For example, there is a risk to the ecology of the Waikato River from the temperature rises caused by the Huntly Power Station (1000 MW) cooling water extraction and return. This has to be managed by the operator and is monitored by the regional council as part of conditions on relevant resource consents.

WATER AND LAND MOVEMENT

McConchie (1992) presents a complete chapter on water and land movement, including some of the risks and costs. An example of land movement giving rise to a water-related risk has already been mentioned—the landslide dam in the Tunawaea Stream. Landslide dams give rise to water-related risks because they impound water behind a “structure” whose “design” bears no relationship to the rate of arrival and total volume of water coming from its catchment. It is therefore likely to fail at some time that is difficult to determine, causing a dam-break flood downstream. This kind of “flash flood” can be very dangerous in its immediate effects. Such a natural dam can also alter the sediment flow from the catchment in ways that have unwanted effects both before and long after the dam fails.

A dam was formed in the Poerua River valley in Westland, when a landslide fell from Mt Adams on 6 October 1999. About 10–15 million cubic metres of fractured rock and loose debris fell 1800 m from near the summit (Hancox *et al.* 1999). This dam failed in the first heavy rainfall, sending floodwater and sediment over forest and farmland down the valley. There is likely to be an ongoing risk of sediment aggradation affecting the remaining farmland and a highway bridge. The risks posed to Franz Josef township by the Waiho River, including the risk of a dam-break flood from its tributary, the Callery River, have already been mentioned.

Submarine land movement is one of the well-known causes of ocean tsunami (e.g., Goldsmith *et al.* 1999). Not so often noted is the risk of such a surface wave caused by a landslide into an inland water body such as a lake or large reservoir. A case study has been carried out for Lake Wakatipu and Queenstown in South Island, New Zealand

(see Ruddenklau 1999, unpublished, cited in Painter 2000). Queenstown could look as it does in Figure 45.5 about 5 minutes after a landslide entered the lake from one of the unstable slopes identified.

There is a perceived risk to the Clyde Dam, and Clyde township, in Otago, New Zealand, from the Cairnmuir Slip and other potential mass movements alongside Lake Dunstan. Landslide stabilisation work includes over 14 kilometres of tunnels dug into the hillsides to prevent water build-up and buttresses of compacted rock and gravel placed to stabilise the toes of possible slips. There are about 3,500 measuring and monitoring instruments installed around the lake shore (McConchie 1992).

The Golden Cross gold and silver mine near Waihi, Bay of Plenty, North Island, has a tailings dam and silt pond inadvertently constructed on a pre-historic landslide. The risk became obvious in 1996 when cracks appeared near the dam, and measurements showed the dam was moving downhill several millimetres per day during wet weather. About NZ\$30 million was spent in a three-year stabilisation programme that included a 760-m-long underground drainage tunnel, and pumping removes about 12,000 cubic metres of water each week (Geological and Nuclear Sciences 1998).

Water removal from aquifers can result in land subsidence. There are risks both from long-term extraction for water supply and from short-term dewatering of building or other construction sites. A scoria and basalt quarry operator in Auckland was pumping 5000 cubic metres per day of groundwater to dewater the site in the late 1990s. Land subsidence between 3 and 17 mm was reported in a zone up to 1.8 km from the quarry. The zone included “some of Auckland’s most expensive suburbs” (Gardiner 2002).

In some parts of New Zealand, particularly areas with limestone karst geology (see Chapter 31), the subterranean action of water dissolving rock can lead to subsidence. The Maori name of the Waitomo Caves area in King Country, North Island, means “water entering the caves by long shafts”.

WATER QUALITY

Drinking water

There are many risks related to water being of a quality unsuitable for drinking, cooking, swimming, bathing, agricultural and other industrial use. “Pollution of our rivers, lakes and beaches has featured as the top environmental concern in many surveys of New Zealanders’ opinions over the past five years, so there is a clear message here for central and local government and the farming community.” (Ministry for Environment 2001).

A detailed report on the chemical quality of drinking

water in New Zealand (Davies *et al.* 2001) discusses the health risks of more than 100 compounds found in water. The most significant are corrosion-derived metals (e.g., antimony, cadmium, copper, lead and nickel), arsenic, disinfection by-products (mainly chlorine and other halogen derivatives) and nitrate. Fluoride is also found in drinking water in some parts of New Zealand. It is there because it is added by supply authorities to lower the incidence of tooth decay in young children. Management of that risk (tooth decay) is seen as introducing another risk (poisoning) by those who regard fluoride primarily as a toxin, even in the low doses employed. Managing the risks posed by these compounds can be as simple as ensuring people run a tap briefly before taking water to drink, to flush out the corrosion metals (reducing copper concentrations, however is best managed by using different materials for drinking water pipes). Or the risk management can be complex, such as dealing with the political, social, scientific and economic aspects of managing nitrate contamination in groundwater aquifers whose catchments include intensive agriculture, such as horticultural cropping or dairy farming.

The microbial quality of water for human contact, both internal and external, presents potential health risks. Microbial pathogens in drinking water can cause gastrointestinal illnesses, fevers, diarrhoea and dehydration. The Ministry of Health publishes annual reviews of the microbial quality of drinking water (e.g., Ball 2002). It is the responsibility of the Ministry of Health to monitor the quality of water supplied, and to manage any risks to health. There are, for example, published standards for drinking water (Ministry of Health 2000). These list the maximum concentrations of chemical, radiological and microbiological contaminants acceptable for public health in drinking water and the sampling procedures that should be used to monitor compliance.

Surface water and groundwater

Managing the water quality risks of natural surface water in streams, rivers and lakes does not imply that all water should ideally have no chemical or microbial content, only water molecules! Indeed such a state would be decidedly hazardous for dependent ecological systems. Although the Resource Management Act (1991) makes provision for national environmental standards, such as those for water quality, there are none yet in place in New Zealand. The Ministry for the Environment endorses guidelines (Australian and New Zealand Environment and Conservation Council 2000), which are used by regional councils and other agencies.

Managing the water quality risks of groundwater in aquifers, particularly, must take account of the two general points already made about time and space scales. There

has been considerable effort in the last few decades to determine water quality in groundwater aquifers in many parts of New Zealand (e.g., Roberts *et al.* 1996). There is a close, and circular, relationship in New Zealand between agriculture and water quality, both surface water and ground water. Intensive agriculture poses risks because of its potential and actual negative effects on urban and recreational water users as a result of lowered water quality. Low water quality poses risks as a result of its potential and actual negative effects on agriculture as a result of animal and plant health, commodity prices and market acceptance—a tarnishing of the “pure New Zealand” image.

Many industrial and semi-industrial activities are very important in managing risks related to water quality. Effluent application to land or discharge to water, pollutant disposal and spills of many kinds, mining activities in or near water, farming and forestry alongside water, are just some. Most activities of this kind are intended to be avoided, mitigated or remedied under provisions of the Resource Management Act (1991), administered by regional councils and unitary local authorities.

ICE, SNOW, HIGH-TEMPERATURE WATER AND STEAM

Most of this chapter on managing water-related risks in New Zealand deals with liquid water at ambient temperatures. Expanding the scope to include water at very low or very high temperatures, or water in its solid and gaseous forms, both modifies the risks already considered and adds new risks. In drowning, death occurs by suffocation. For a person falling in to a pool of boiling water in a geothermal area, death will occur as a result of scalding instead. A person trapped under snow as a result of an avalanche may die from hypothermia instead of suffocation.

Excess water as snow in occupied places can have some effects similar to floods. Deep snow on roads can close them. There are also new risks, such as structural damage to buildings from the weight of snow accumulating on roofs. Insufficient water falling as snow in the catchments of the southern South Island hydro lakes can limit the spring melt flows, and contribute to water and hydro-electric energy supply problems. It is also a risk to New Zealand's winter tourist industry if there is insufficient snow on the ski fields.

Hail has serious effects on horticultural crops, particularly some of high-value such as cherries and grapes, but also pip and stone fruit. And water changing state from liquid to solid brings its own risks: accidents due to ice on road surfaces, burst pipes in buildings, burst cells in frost-tender plants and crops. Paradoxically, spraying water

continuously on to frosted crops, to keep tender parts at zero degrees Celsius as the water freezes, can protect them from damage. It can also bring a new risk, of structural damage due to the weight of ice.

Outbursts of geothermal steam and high-temperature water are a risk to human safety, as has been demonstrated quite frequently in the city of Rotorua, North Island. On the other hand, using geothermal steam for electricity generation or domestic heating poses risks for the tourism industry and tangata whenua, as formerly active geysers diminish or cease and land subsidence may occur.

Managing these and other risks related to water at very high or low temperatures, and to solid and gaseous water, can involve modifications to approaches considered earlier. But some risks, such as those arising from snow avalanches and in alpine recreation, are peculiar to this field, and their management is outside the scope of this chapter.

ECONOMIC ASPECTS

It has been emphasised here that risk is a societal construct, even when the hazards involved are distinctly physical. Human safety and social welfare involve more than financial and economic factors, but they are important aspects of risk and its management. Obtaining resources for managing risks, particularly as risk avoidance, prevention or transfer, and mitigation of effects, rather than as risk response and the recovery necessary after disasters, is often primarily an economic task. This has not been the particular emphasis in this chapter, but its importance is acknowledged.

LEGISLATION AND RESPONSIBILITIES

Many organisations play a part in managing water-related risks, especially agencies of central, regional and local government in New Zealand. Likewise, risks are the subject of various acts of parliament and regulations. Neither cover all aspects of managing water-related risks, but those mentioned are thought to play a major role in relation to particular risks, and provide a starting point or place of contact for further information.

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Chapter 44

Management and conservation of natural waters

Chris Richmond, Victoria Froude, Andrew Fenemor and Bob Zuur

INTRODUCTION

Earlier chapters of this book describe the hydrological cycle, and physical and ecological processes in running waters, lakes, wetlands and subsurface waters. Later chapters describe a variety of resource management issues relating to freshwaters. The resolution of these issues requires the effective use of appropriate scientific information, with legal and planning processes to clarify management objectives and resolve conflicts between potentially competing uses, including environmental protection.

The purpose of this chapter is to introduce New Zealand's system of planning for and managing inland natural waters. This includes government strategy, legislation and planning processes relating to the protection of water bodies and their indigenous biota, water quality, water allocation, the management of introduced species, and catchment management.

This chapter addresses natural waters. Natural waters are those vested in the Crown by the 1967 Water and Soil Conservation Act, the property rights and Crown interests in which were "saved" by s.354 of the Resource Management Act (RMA). Inland natural waters include "freshwaters", geothermally influenced waters, and freshwaters within estuaries within the Coastal Marine Area, as defined in the RMA.

WHO DOES WHAT?

Other management chapters have described a variety of issues affecting New Zealand freshwaters. Sometimes these issues are difficult to solve because there may be a complex web of agencies, responsibilities and processes, unclear accountabilities or conflicting objectives. Table 44.1 is a summary of the agencies that are concerned with New Zealand freshwaters: it lists the agencies, relevant legislation, and summarises key freshwater functions, powers and responsibilities.

NEW ZEALAND LEGISLATION

A considerable amount of legislation affects freshwaters: the key statutes are the Resource Management Act (RMA) and the Conservation Act and its regulations, although, as Table 44.1 shows, there are several other relevant statutes and regulations.

The processes followed under the RMA may result in court cases on both planning documents and resource consent applications. There is a hierarchy of courts, with the Environment Court being the usual first court. Decisions from this court may be appealed on matters of law to the High Court. Its decisions may be appealed to the Court of Appeal. The associated decisions are known as case law. Some decisions under the RMA (e.g., notification decisions) and other statutes are taken directly to the High Court. Offences are usually addressed initially in the District Court.

The Resource Management Act 1991

Purpose and Principles – Part II of the Act

The Resource Management Act is the key piece of legislation for water management. Part II of the Act contains the purpose and principles. It is the context against which all functions, powers and duties specified in the RMA are exercised. Any use of discretionary powers under the Act is required to address Part II.

The purpose of the Act (section 5) is "to promote the sustainable management of natural and physical resources". Sustainable management is defined in section 5(2) as: "...managing the use, development, and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic and cultural well-being and for their health and safety while:

- (a) Sustaining the potential of natural and physical resources (excluding minerals) to meet the reasonably foreseeable needs of future generations; and

- (b) Safeguarding the life supporting capacity of the water, soil and ecosystems; and
- (c) Avoiding, remedying, or mitigating any adverse effects of activities on the environment.”

A number of guidelines can be drawn from the case law on section 5. These include:

- The definition of sustainable management should be read and applied as a whole.
- There should be an overall assessment as to whether a proposal promotes sustainable management while comparing conflicting considerations.
- Some adverse effects may be acceptable, such as where a proposal is of national significance and mitigation reduces the adverse effects sufficiently.
- The philosophy of enabling people and communities to provide for their well-being is subject to sections 5(2)(a)-(c) being met and sections 6-8 being addressed.

Section 6 requires all those exercising powers and functions under the Act to recognise and provide for specified matters of national importance. The key matters for freshwaters are:

- (a) “The preservation of the natural character of the coastal environment (including the coastal marine area), wetlands, and lakes and rivers and their margins, and the protection of them from inappropriate subdivision, use and development:
- (b) The protection of outstanding natural features and landscapes from inappropriate subdivision, use and development:
- (c) The protection of areas of significant indigenous vegetation and significant habitats of indigenous fauna:
- (d) The maintenance and enhancement of public access to and along the coastal marine area, lakes and rivers:
- (e) The relationship of Maori and their culture and traditions with their ancestral lands, water, sites, waahi tapu, and other taonga.”

There is considerable case law on section 6(a), although most of this relates to the coastal environment. Much of this case law has been summarised by Richmond and Froude (1998) and Maplesden (2000). Some key concepts relevant for freshwater include:

- “Natural character” refers exclusively to features derived from nature. They may be in an original pristine state or be culturally influenced (e.g., *Harrison vs Tasman District Council* (1994) W42/93).
- Natural character is assessed as a continuum from more pristine natural environments to ones that are a composite of nature and activities such as forestry and farming (e.g., *Director-General of Conservation vs Marlborough District Council* (1997) W89/97)
- Natural character includes ecological processes (e.g., *Gill and Others vs Rotorua District Council and Another*

(1993) W29/93). This decision relates to an application for a development on the shores of Lake Tarawera and is one of few freshwater decisions on this section of the Act.

- The “potential naturalness” of the site in terms of its ecological resilience and restoration can be important (e.g., *Steffan John Browning vs Marlborough District Council* (1997) W20/97).
- The context of the wider environment is important.
- Unmodified environment free from built elements retains the highest degree of natural character and therefore has the highest priority for protection and preservation (e.g., *Lowe and Another vs Auckland Regional Council* 1994)

Section 7 requires those exercising powers and functions under the Act to have particular regard to (as it relates to freshwaters):

- (a) “Kaitiakitanga:
 - (aa) The ethic of stewardship:
- (b) The efficient use and development of natural and physical resources:
- (c) The maintenance and enhancement of amenity values:
- (d) Intrinsic values of ecosystems:
- (f) Maintenance and enhancement of quality of the environment:
- (g) Any finite characteristics of natural and physical resources:
- (h) Protection of the habitats of trout and salmon.”

Section 8 requires all those exercising powers and functions under the Act to take into account the principles of the Treaty of Waitangi.

Resource Management Act planning processes

There is a hierarchy of planning documents prepared under the Resource Management Act, cascading down from National Policy Statements, to National Environmental Standards, to Regional Policy Statements, then Regional Plans and District Plans. While the initial stages of document preparation can be relatively informal, the later stages follow a formal prescribed process that includes provisions for appeals on council decisions about proposed plans and regional policy statements. Each document is required to “not be inconsistent with” documents further up the hierarchy.

In the case of national policy statements a local authority is required to change its planning documents to remove any inconsistency or conflict with the national policy statement. As at 1 December 2003 there was only one national policy statement—the mandatory New Zealand Coastal Policy Statement. The New Zealand Biodiversity Strategy (Department of Conservation and Ministry for

the Environment 2000) Actions (1.1.d and 2.1.g) include government's intention at the time to prepare a national policy statement on biodiversity. Because this national policy statement is intended to subsume and replace the 1986 New Zealand Wetland Management Policy, it would be likely to contain formal policies for freshwater ecosystems and species.

While regional policy statements and regional coastal plans are mandatory, other regional plans are not. Accordingly, regional councils have been handling their Resource Management Act planning functions (see Table 44.1) in quite different ways. Some councils have developed a combined plan covering a range of regional or unitary council functions (e.g., Environment Waikato; Tasman District Council); others are preparing a number of plans addressing different functions (e.g., Greater Wellington Regional Council). Many are also pursuing non-regulatory approaches. Some of these approaches are described in more detail under "National and regional perspectives on freshwater issues". In many cases regional planning documents for Council freshwater functions (either directly through water plans or indirectly through land or soil plans) are still being developed or being finalised through the formal processes (including appeals).

Section 14 of the RMA provides that no person may take, dam, divert or use natural waters unless this is authorised by a resource consent or a rule in a regional plan. The only situations where a water permit is not required are where the water is taken for fire fighting, or for stockwater or individual domestic uses if there is no adverse effect caused by the take, or (in Otago) where it is part of a mining privilege. Similarly, s.15 provides (*inter alia*) that no person may discharge contaminants or water into water, or onto land in circumstances where that contaminant may enter water, unless this is authorised by a resource consent or a rule in a regional plan.

The RMA provides minimum standards for water quality, such that consents may not be granted and plans may not permit activities where certain standards are breached after reasonable mixing (ss. 70 and 107). These standards are:

- The production of any conspicuous oil or grease films, scums or foams, or floatable or suspended materials;
- Any conspicuous change in colour or visual clarity;
- Any emission of objectionable odour;
- The rendering of fresh water unsuitable for consumption by farm animals;
- Any significant adverse effects on aquatic life.

The Third Schedule of the RMA provides for eleven different classes of water, protecting a variety of uses. Where a regional council provides for the protection of these uses in a regional plan, less stringent standards may not be adopted. Few plans have adopted the classes in the Third Schedule.

A particular challenge for water management is controlling the impacts of land use on water quality and quantity (as provided for in s. 30(1)(c)). This is related to the difficulty of measuring impacts or enforcing any rules relating to diffuse discharges from land uses (e.g., leaching fertilizer, urine or dung, and sediment runoff into water bodies). The management of land use is investigated in a case study later in this chapter.

Mandatory district plans are the primary planning documents under the Act for controlling land use and subdivision. These are prepared by territorial and unitary authorities. They are not intended to be the primary mechanisms for controlling land uses that affect freshwaters, but the reality is that their provisions can have a significant effect. Many permitted activities, such urban subdivision and existing rights for damaging activities (e.g. pastoral agriculture on unsuitable land) have a major impact on water bodies.

The Resource Management Act includes provisions for setting aside esplanade reserves or strips alongside lakes and rivers on subdivision and road stopping. The purposes of these esplanade reserves and strips include maintaining or enhancing natural functioning, water quality or aquatic habitats of the adjacent water body, and to enable public access alongside water bodies (s.230).

District and regional plans use rules to specify types of activities and their associated standards. In the district planning jurisdiction, the legal presumption is that you can carry out land-use activities unless they are explicitly proscribed in the plan. In the regional planning jurisdiction, activities such as water use or discharges require regional or unitary council consent unless described as permitted activities in a regional plan. Permitted activities are those allowed in a plan without a resource consent. Prohibited activities are those that are expressly prohibited in a plan. Other activities require a resource consent from the district or regional council. The Resource Management Act sets out clear procedures for this, which vary depending on the type of activity and the standards specified in the appropriate planning document.

Water conservation orders

The main mechanism under the Resource Management Act to protect the natural or cultural heritage values of lakes, rivers or other water bodies is the Water Conservation Order. Water conservation orders are addressed under Part IX of the Act, outside of the sustainable management purpose of the rest of the Act. Section 199(1) states:

- “Notwithstanding anything in Part II, the purpose of a water conservation order is to recognise and sustain:
- (a) Outstanding amenity or intrinsic values which are afforded by waters in their natural state;

Table 44.1 Agencies and bodies involved in freshwater management

Agency	Legislation	Key relevant functions, powers and responsibilities
Ministry for the Environment	Resource Management Act 1991	Administers Resource Management Act. Prepares national policy statements (except NZ Coastal Policy) and national environmental standards.
Regional councils (including unitary authorities)	Resource Management Act 1991	Functions include: <ul style="list-style-type: none"> • Establishing objectives, policies, methods for the integrated management of the resources of the region including biological diversity [s30(1)(a) and (ga)] • Controlling use of land for the purposes of: soil conservation; maintenance and enhancement of water quality; maintenance water quantity; maintenance and enhancement of ecosystems in water bodies [s30(1)(c)] • Controlling the taking, use, damming and diversion of water and the control of the quantity, level and flow of water in any water body [s30(1)(e)] • Controlling discharges [s30(1)(f)] • Controlling planting in a bed of a river for soil conservation, water quality maintenance and enhancement and water quantity maintenance [s30(1)(g)] Regional councils prepare mandatory regional policy statements and optional regional plans
Regional councils (Act administered by Department of Internal Affairs)	Land Drainage Act 1908	Powers to construct and maintain drains and watercourses including deepening and diverting existing watercourses; grazing or widening the defences against water; making any new watercourses for drainage (s17)
Regional councils (Act administered by Ministry for the Environment)	Soil Conservation and Rivers Control Act 1941	Function to minimise damage caused by flooding and erosion (s126) Powers to construct and maintain works to control water flow to and from watercourses, prevent rivers from breaking their banks and promote soil conservation (s126) Powers to widen, deepen, straighten or divert any watercourses; make new watercourses and divert water from watercourses; and erect any new defence against water (s133)
Territorial authorities (including unitary authorities)	Resource Management Act 1991	Functions include <ul style="list-style-type: none"> • Establishing objectives, policies, methods for the integrated management of the use, development or protection of land and associated resources of the district (s31(a)) • Controlling any actual or potential effects of the use, development or protection of land, including for the maintenance of indigenous biological diversity (s31(b)) Territorial authorities prepare mandatory district plans
Territorial authorities	Local Government Act 2002	Powers include provision of drainage works; and treating and dealing with sewage
Ministry of Health	Health Act 1956	The Minister may require any local authority to provide sanitary works.
Parliamentary Commissioner for the Environment	Environment Act 1986	May review governance framework for environmental management, investigate the effectiveness of environmental planning and management, enquire into and report on matters having a significant effect on the environment.

Agency	Legislation	Key relevant functions, powers and responsibilities
Department of Conservation	Conservation Act 1987	<p>Functions (s.6ab and 6a) and responsibilities [s.(53(3))] include:</p> <ul style="list-style-type: none"> • Preserving all indigenous freshwater fisheries (6ab), • Protecting recreational freshwater fisheries • Protecting freshwater fish habitats (s6ab) • Managing all land and other resources held under the Act for conservation purposes (s6(a)) <p>Powers and responsibilities include:</p> <ul style="list-style-type: none"> • Advocating the conservation of freshwater fisheries and other aquatic life • Protecting habitats by purchase and other means • Controlling any introduced species causing damage to any indigenous species or habitat • Managing the Taupo fishery <p>The Department prepares statements of General Policy, conservation management strategies (including national park management plans) and conservation management plans (including those for areas managed under the Reserves Act), freshwater fishery management plans.</p>
Department of Conservation	Reserves Act 1997 & National Parks Act 1980	Acquiring and protecting habitats within protected areas to ensure the survival of indigenous species (Reserves Act s3)
Department of Conservation	Wildlife Act 1953	Administers the Act which safeguards protected species of wildlife, including waterbirds, some invertebrates and amphibia, throughout New Zealand. It also enables establishment of Wildlife Refuges over water bodies to control some activities affecting wildlife
Department of Conservation	Freshwater Fisheries Regulations 1983	Administers the Regulations which include provisions for: securing, restoring (and preventing as appropriate) migratory access for freshwater fish; establishing Faunistic Reserves for aquatic life
New Zealand Fish and Game Council	Conservation Act 1987	Nationally represents the interests of anglers and hunters and co-ordinates the management of sports fish and game (s26B)
Regional Fish and Game	Conservation Act 1987	Functions are to locally manage, maintain and enhance sports fish and game resource and habitats in their region (s.26Q)
Ministry of Fisheries	Fisheries Act 1996	Freshwater fisheries with a significant commercial component (e.g., eels, mullet, flounder, but not whitebait) are managed under this legislation.
Guardians of Lakes Manapouri, Monowai and Te Anau	Conservation Act 1987	Functions include: Making recommendations to the Minister about the effects of the operations of various power schemes on these lakes and their shorelines and the inflowing and outflowing rivers.
Guardians of Lake Wanaka	Lake Wanaka Preservation Act 1973	Functions include making recommendations to the Minister of Conservation on matters affecting the preservation of the lake and its water levels and water quality

- (b) Where waters are no longer sustained in their natural state, the amenity or intrinsic values of those waters which in themselves warrant protection because they are considered outstanding.”

Under section 199 (2) a water conservation order may provide for:

- (a) “The preservation as far as possible in its natural state of any water body that is considered to be outstanding;
- (b) The protection of those characteristics which any water body has or contributes to, and which are considered to be outstanding:
- (i) As a habitat for terrestrial or aquatic organisms;
 - (ii) As a fishery;
 - (iii) For its wild, scenic, or other natural characteristics;
 - (iv) For scientific and ecological values;
 - (v) For recreational, historic, spiritual, or cultural purposes;
- (c) The protection of characteristics which any water body has or contributes to, and which are considered to be of outstanding significance in accordance with tikanga Maori.”

A water conservation order protects or preserves outstanding values by imposing limits on quantity, quality, rate of flow or level of water body, level ranges, flow rates, maximum abstraction and maximum contamination allowed and temperature ranges. A regional council must manage the water body within the bounds set by the water conservation order.

The water conservation order process has often been lengthy and sometimes expensive in terms of evidential requirements. This has limited the number of water bodies for which a water conservation order application has been made. Most water conservation orders apply to only part of the river system or water body, as case law from the Buller hearing established that the values and protection mechanisms must be established for those parts of the system found to be outstanding, not necessarily for whole systems. As at March 2004, twelve water conservation orders had been gazetted (see www.legislation.govt.nz under Regulations, then National Water Conservation (place) Orders and Water Conservation (place) Orders) and one had been formally recommended to the Minister for the Environment following Environment Court hearings. A case study on the Motueka River and its Water Conservation Order, recommended in 2003 by the Environment Court for gazettal, is included later in this chapter.

Conservation Act 1987

Controls on transfers and releases of aquatic life

Controls on the transfer or release of freshwater fish and other aquatic life are contained primarily in the

Conservation Act 1987 and its Freshwater Fisheries Regulations 1983. Under s.26ZM the prior approval of the Minister of Conservation is required for all introductions of a freshwater species into waters where it does not already occur, and for all releases into waters within lands administered by the Department of Conservation. For movement of aquatic life between sites where the species already occurs, and between the islands of New Zealand, the prior consent of the Minister of Fisheries is required, primarily to address the risks of transferring diseases. Liberations into lakes or rivers also require the consent of the local Fish & Game Council under Regulation 59 of the Freshwater Fisheries Regulations.

Controls on discharges of contaminants to water (or land leaching to water)

While the primary controls on the discharge of contaminants to water, or land leaching to water, are currently in the Resource Management Act 1991, such controls were traditionally in the fisheries legislation. The Crown has retained its power to provide some controls over discharges to water, or land leaching to water, that cause detrimental effects on freshwater fish and their habitats in waters of any tenure. These offences are defined in s.39(4-7) of the Conservation Act, together with acceptable defences under the Resource Management Act. Similar offences for discharges affecting wildlife and its habitat are retained in Regulation 43A of the Wildlife Regulations 1955.

Marginal strips and riparian protection

Riparian protection provisions are also contained in Part IV of the Conservation Act, through the requirement to reserve from sale a marginal strip 20 metres wide upon any disposal of land of the Crown adjoining a stream over 3 metres wide or a lake over 8 hectares. The purposes of marginal strips (s.24C) include the maintenance of water quality; the protection of aquatic life and riparian zone natural values; as well as providing for public access and recreation.

Freshwater Fisheries Regulations 1983

These regulations under the Conservation Act provide controls on several types of activities that affect freshwater environments, as well as controlling fishing itself.

Fish passage

Part V of the regulations contains the primary legislative code for the protection of fish passage, and addresses those situations where the Resource Management Act processes have failed to secure, restore or, where necessary, prevent access for freshwater fish. The primary emphasis is on new

dam and diversion structures (regulation 43). These require the approval of the Director General of Conservation, who may provide for a dispensation from the requirement to provide free passage for fish, or may require installation of a fish pass.

Noxious animals and plants

Freshwater fish species gazetted as “noxious” are controlled under regulation 65, which sets out restrictions on taking, possessing and sale of noxious fish, including species that may affect water quality and habitats. As at 2003, the species gazetted include seven not yet established in New Zealand and two that are still geographically contained—rudd and koi carp. The Chief Technical Officer Conservation has declared (as at March 2004) a number of freshwater species to be “unwanted organisms” under the Biosecurity Act 1993, including two fish, 13 amphibia and 24 waterweeds. Other agencies have similarly made declarations for mosquitoes and various crustaceans.

Faunistic Reserves

Faunistic Reserves may be established by gazettal under regulation 68 across any water body of any tenure to protect the freshwater fauna present from any taking or killing and from any introductions of aquatic plant species. The power was introduced at a time of unconstrained liberations of introduced fish and weed species, but has been used to protect only four lakes to date.

NATIONAL POLICY

There are two main overarching national policy documents affecting freshwater: the Sustainable Development for New Zealand Programme of Action, and the New Zealand Biodiversity Strategy.

Sustainable Development for New Zealand - Programme of Action

This programme (Department of Prime Minister and Cabinet 2003) devotes one of four themes to freshwater, for which it seeks to achieve the following outcomes:

- Freshwater is allocated and used in a sustainable, efficient and equitable way.
- Freshwater quality is maintained to meet all appropriate needs.
- Water bodies with nationally significant natural, social or cultural heritage values are protected.

As at December 2003 the programme of action is still being developed, with the anticipation that new legislation to improve governance, additional national policy and standards, together with implementation projects, will emerge from the programme.

New Zealand Biodiversity Strategy

This document (Department of Conservation and Ministry for the Environment 2000) has the following objectives for freshwater ecosystems:

- “Ensure that management mechanisms, including mechanisms under the Resource Management Act and protected area statutes, adequately provide for the protection of freshwater biodiversity from adverse effects of activities on land and in water.”
- “Protect a full range of remaining natural freshwater ecosystems and habitats to conserve indigenous freshwater biodiversity, using a range of appropriate mechanisms.”
- “Prevent, control and manage plant and animal pests that pose a threat to indigenous freshwater biodiversity.”
- “Restore areas of degraded or scarce natural freshwater habitats and ecosystems that are priorities for indigenous biodiversity.”
- “Enhance population numbers and ranges of indigenous freshwater species threatened with extinction and prevent additional species and ecological communities from becoming threatened.”
- “Ensure that harvest of indigenous and introduced freshwater species and associated activities do not have an adverse effect on indigenous freshwater biodiversity.”

General Policies under the Conservation and National Parks Acts

These are statutory policy documents that provide overarching strategic policy for the implementation by the Department of Conservation of the legislation covered by each General Policy. They will provide guidance for the ongoing reviews of each of the Conservancy-based formal Conservation Management Strategies and Conservation Management Plans. Each General Policy contains policies for the management of freshwater ecosystems, habitats and species, as well as biosecurity. As at March 2004, public submissions had been made on the formally notified drafts and were being analysed.

Transitional policies

There are several older national policies still in force, including the New Zealand Wetlands Policy 1986 and the New Zealand Geothermal Resources Policy 1986. The wetlands policy is administered by the Department of Conservation and seeks the adequate protection of wetlands, a national inventory of wetlands, and improved public awareness of wetland values. The parts relating to wetland protection also address re-establishment of wetlands for scenic purposes and fish habitat, and the management of habitats for native flora and fauna (especially migratory species) and the maintenance and enhancement of important hydrological functions.

The geothermal policy includes policies that seek the protection of specified geothermal fields, as well as, more generally, fields with surface features of scenic and scientific value. It is currently administered by the Ministry of Economic Development.

The New Zealand Biodiversity Strategy actions include the intention to review both of these policies and incorporate the relevant parts into a National Policy Statement on Biodiversity under the RMA.

FRESHWATER MANAGEMENT – ISSUES AND EXAMPLES OF GOOD PRACTICE

Water allocation

Surface water bodies, such as rivers, lakes and wetlands, support both “instream” and “out-of-stream” values and uses. Instream values include the ecological, recreational, landscape and cultural values of the water body. Out-of-stream uses include the abstraction, damming and diversion of water for domestic and community supply, stock water, power generation and agriculture and industry. Many of these out-of-stream uses are necessary for economic, social, and community health and safety reasons. However, these activities can also affect flow regimes, and as a consequence have adverse effects on instream values. This can lead to a conflict between protecting instream values and maintaining human use values, particularly during low flows or droughts.

Low flows can reduce the quality and quantity of aquatic habitat, reducing water quality through decreased dilution capacity and increased temperature. Low flows diminish the natural character, amenity, aesthetic and landscape values of a stream, affecting recreational use and cultural and spiritual values. Low flows reduce the amount of water available to meet the needs of existing users, and future users through lower rates of groundwater recharge. The extent of these adverse effects is dependent on the characteristics of a particular surface water body, the values associated with it, and the amount of water taken, dammed or diverted.

Water allocation is addressed through Regional Planning processes and implemented through the issuing of water permits to take, dam, divert or use natural waters under section 14 of the RMA. As at early 2000, all regional and unitary councils were preparing regional plans that include management of water quantity and allocation, but there were only five operative specific water management plans (Lincoln Environmental 2000).

For planning, most councils have adopted an approach to water allocation that first establishes the values and uses of each water resource, then sets limits—such as allocation limits, or minimum flows or groundwater levels—to the amount of water that can be extracted to avoid adverse

effects to those values and uses. The remaining water is allocated among applicants for water permits on a “first in, first served” basis. Minimum flows have been set for many rivers, but if extraction is a large proportion of low to medium flows, this can lead to “flat-lining” of the hydrograph, algal proliferations, sediment deposition and decline in aquatic habitat. In this situation, minimum flow regimes that incorporate flushing flows (for example, from hydroelectric dams) increasingly are being required by regional councils. Many councils are still at the early stages of setting allocation limits, and this is especially so for groundwater resources.

The RMA provides for the protection of the life-supporting capacity of the water. Other uses or values may also depend on there being enough water in the river or stream. For some water bodies, there may be conflicts between the needs of instream uses, such as canoeing, rafting and other boating, swimming, landscape and amenity values and hydroelectric uses, and those that take the water for agricultural, industrial or urban uses. The Ministry for the Environment has prepared “Flow Guidelines for Instream Values” (Ministry for the Environment 1998) to assist Councils in establishing minimum flows and allocation limits to protect instream values. The procedures recommended all depend upon first negotiating and establishing “instream management objectives”, which reflect the priorities and extent to which the community wishes to secure those values. For many Councils it has proven challenging to reach agreement on clear and measurable objectives. New guidance on developing and drafting objectives for policy statements and plans was commissioned by the Ministry for the Environment (Willis 2003). Other water management guidelines are listed in the bibliography. A Note on Best Practice for Water Allocation Planning is posted on the Best Practice index of the Quality Planning Website (www.qualityplanning.org.nz).

Regional councils commonly use a package of measures to manage water allocation (Fenemor and Robb 2001). These measures can include allocation limits for the resource, rationing restrictions during periods of low flow, and limits on the water allocated to specific types of use that tailor the allocation to the proven need for the water.

Once the allocation limit has been established, the Council then needs to ensure that allocations among abstractive users are equitable. However, apart from the “first in, first served” presumption, the RMA at present provides little guidance or tools for councils to ensure equitable allocation. Section 7 of the RMA requires consideration of the efficiency of resource use. Councils use a number of methods, including rules specifying how water is allocated, requiring water meters on takes, and encouraging transfers of water permits to higher valued

uses. Education and advocacy are used to encourage users to employ best management practices for efficient water use and cleaner production, and subsidies or incentives may be provided for soil moisture monitoring equipment or improved irrigation technology.

Where different water resources are connected, it is important that they be managed together. For example, excessive groundwater pumping from an alluvial aquifer could compromise the uses and values of rivers connected to that aquifer. At a larger scale, this illustrates the importance of integrated catchment management (e.g., Fenemor 2002). At the catchment scale, water allocation objectives may need to account for potential effects of changes in land use, riverbed levels and climate, as the allocatable volume of water may change over time in response to these factors. Actions such as the protection of riparian margins of rivers can mitigate the increased water temperature or loss of aquatic habitat resulting from water extraction. Likewise, the construction of water augmentation schemes, such as dams releasing water during low flow periods, can enhance instream values and mitigate the effects of extraction.

Case Study I: Managing fully allocated water resources – Tasman District

Problem

When the environmental limits for a water body are being regularly met or exceeded because of water extraction, that resource can be regarded as being fully or over-allocated. This may lead to stream or river flows below the levels required to sustain life, and it may mean that abstractive users are unable to access their allocated volume of water, potentially leading to significant environmental and economic costs. To avoid this situation, management objectives for that water body must be identified, together with the setting, monitoring and enforcing of suitable environmental triggers or allocation limits. This case study describes how such limits can be set and managed, using the rivers and aquifers of the Waimea and Moutere catchments in Tasman District as examples (see Figure 44.1).

The water resources available in summer from the three major alluvial aquifers and associated rivers in the Waimea catchment—excluding water from dams—became fully allocated in 1996. Water is taken from some 1100 wells

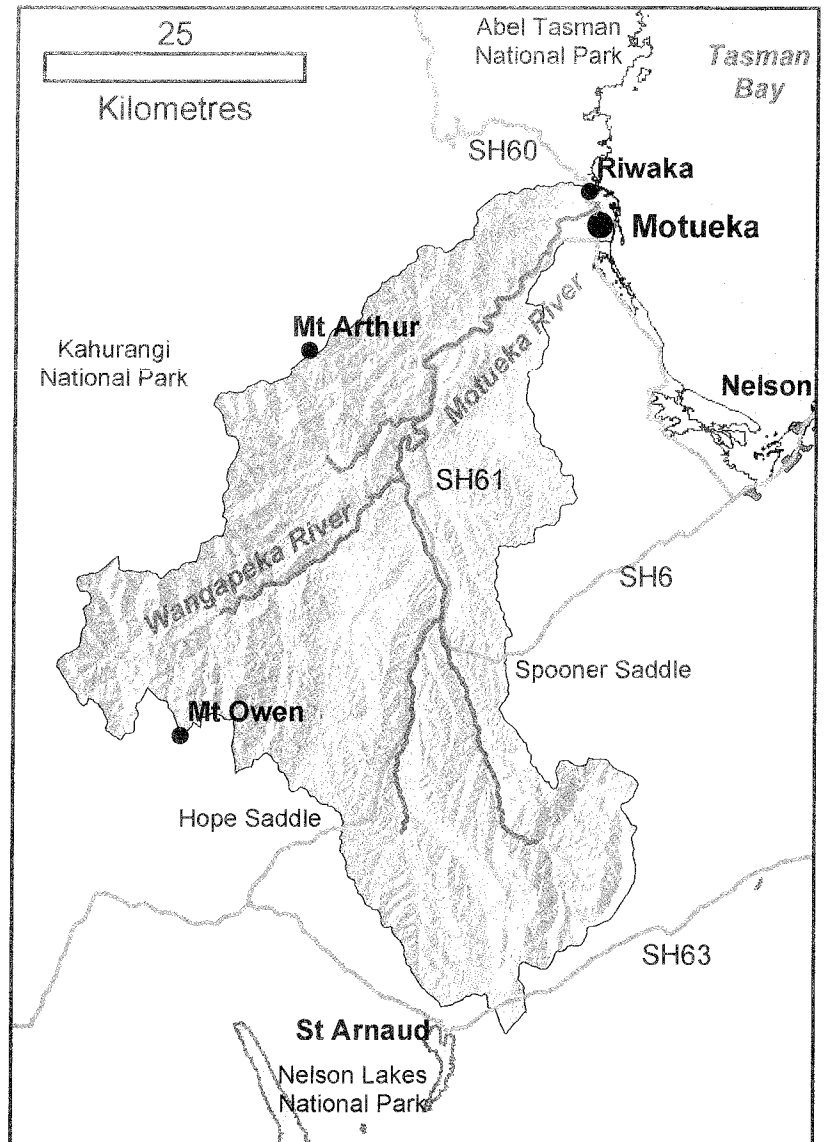


Figure 44.1 Motueka Catchment, Nelson

and stream pumps for irrigation (82%), urban supply (13%) and industrial supply (5%).

In contrast, the Moutere catchment north of the Waimea comprises clay-bound Tertiary gravels with mainly intermittent or ephemeral streams that go dry most summers. Here artesian aquifers up to 500 m deep, with low yields of high quality water, were discovered in the 1980s. The water is used mainly for orchard irrigation and domestic use. In 2001, water allocations comprised 58% from surface streams, 24% from deep aquifers and 18% from dams. The surface water resources, excluding dams, are fully allocated, as is the western portion of the deep aquifers.

Environmental impacts

To set allocation limits, the constraints or adverse effects of water abstraction need to be defined. In the Waimea Basin, there were several main concerns about water abstraction. Low flows in the Waimea and Wai-iti rivers,

caused by groundwater pumpage, could adversely affect aquatic life, recreation and the cultural values of the rivers. Over-pumping could cause potential seawater intrusion into unconfined and deeper confined aquifers, forcing the closure of coastal bores. Excessive falls in groundwater levels in wells due to pumping have already reduced some well yields in the Wai-iti Valley to make them unusable during droughts (“over-allocation”).

In the Moutere Valley, a major concern was the loss of even a minimal flow to support permitted stock and domestic water uses, and allow eels to survive droughts. Seawater could intrude the deep Moutere aquifers if groundwater levels are drawn down below sea level. There was also a need to halt the continued decline in deep aquifer water levels (“aquifer mining”) caused by summer pumping.

Management response

These water resources are managed through water management policies and rules contained in the regional water management chapter of Tasman District Council’s Tasman Resource Management Plan (Tasman District Council 2001). For the Waimea Basin, these policies and rules are the fourth-generation refinement of a water management plan first developed in 1981 (Nelson Catchment Board 1981). For the Moutere catchments, the policies and rules update those first developed in the Moutere Water Management Plan (Tasman District Council 1992).

Allocation limits and water rationing provisions were determined in each case using computer or water-balance modelling of the hydrology and groundwater resources in each catchment (Dicker *et al.* 1992; Fenemor 1988; Hong 2000; Thomas 2001)

Tasman District Council bases its water allocation limits on a security of supply standard such that a 35% reduction in water availability can be expected on average during a 10-year drought. This 35% reduction may be caused either by physical loss of water or through Council imposing rationing when a trigger flow or groundwater level is reached.

In the Waimea catchment, the plan sets a target minimum flow in the Waimea River of 500 l/sec, an increase over the 225 l/sec in the original water management plan (Nelson Catchment Board 1981). The catchment is divided into nine management zones for which individual allocation limits are specified. A three-tier rationing regime is specified as a condition on each water permit, triggered, depending on location, by either a river low flow being reached, seawater intrusion being detected in a sentinel well, or groundwater falling below a specified level.

In the Wai-iti water management zone of the Waimea

catchment, the combined surface water and groundwater have been over-allocated since 1983. The Wai-iti Water User Committee has been reluctant for Tasman District Council to reduce allocations by the 70% required to reach sustainable levels, and has instead promoted a community dam, which would release stored water into the Wai-iti River during summer. The project is to be funded by an annual rate on each m³/week allocated in all water permits held in the Wai-iti zone. Released water will be pumped from groundwater or from the river down the catchment by authorised users, whose take will be metered to ensure compliance.

Because of full allocation in parts of the deep Moutere aquifers, the Council has defined a Moutere Aquifer Recharge Area. Here new forestry is permitted on no more than 20% of any land title. This was the outcome of an Environment Court case (*Wratten vs Tasman District Council*), which recognised that tall vegetation can intercept up to 50% of the annual rainfall in the Moutere area. The aim is to protect groundwater recharge to sustain the groundwater resource already allocated to existing users down-gradient.

Water quality

New Zealand water quality management historically focused on point source discharges, especially from factories and sewerage systems. In the 1950s, many New Zealand waters, such as the Waikato River, were grossly polluted, prompting the development of the Waters Pollution Act in 1953 and its Regulations in 1963. While the Water and Soil Conservation Act 1967 recognised the link between land use and water quality, the efforts of Catchment Boards (precursors of regional councils) were focused on cleaning up point source discharges. Significant improvements in effluent treatment resulted in dramatic increases in water quality in many rivers and streams. However, the intensity and extent of agricultural and urban land use steadily increased in the 1980s and 1990s, putting increasing pressures on our water resources.

While the RMA specifically provides for the control of land use to protect water quality (s.30(1)(c)), the diffuse nature of some sources of pollution make them difficult to control. The following case studies focus on recent innovative solutions.

A recent overview and review of the science and management associated with the effects of land use on water quality was commissioned by the Ministry for the Environment as a new chapter on Land-water Interactions for the Lake Managers Handbook (Elliot and Sorrell 2002). Reviews have recently been completed of the impacts of agricultural land use on water quality for the New Zealand Fish and Game Council by NIWA, water quality impacts on rivers by the Ministry for the

Environment (in press) and the impacts of land use on lakes water quality by NIWA (Rutherford *et al.* in press). Earlier chapters in this book also discuss water quality trends.

Case Study II: Rotorua Lakes water quality

Problem

Lowland lakes throughout New Zealand have been significantly degraded as a result of intensified land use, and also by sewage discharges. Several of the eleven major lakes around Rotorua have suffered water quality problems for decades, and the problems for many appear to be getting worse. Severe cyanobacterial blooms on lakes Rotorua, Rotoiti, Rotoehu and Okaro have received considerable media attention since 1966, when Lake Rotorua was first described by an American expert as an “unflushed toilet”.

Concern increased when Lake Rotoiti, one of the major trout fishing and recreational lakes in the North Island, was entirely closed for swimming for the first time during the 2002/03 summer. The fundamental problem is excess nutrients (nitrogen and phosphorus) stimulating algal growth, cyanobacteria blooms and weed growth. Over the last few years, the depth to which adequate dissolved oxygen remains in Lake Rotoiti has been severely reduced. Toxic hydrogen sulphide now accumulates to the extent that the lake is at risk of becoming anoxic during autumn mixing, killing most of the fish and shellfish. This state was reached in Okaro during the early 1970s, with koura eliminated and finfish decimated.

These problems have occurred in spite of a significant investment by local and central government. Rotorua is the largest settlement on a lake in this country. For many years the sewage from Rotorua city was discharged into Lake Rotorua, which is connected to Lake Rotoiti by the Ohau Channel. In the late 1980s, Rotorua’s sewerage system was upgraded and the effluent discharged to land, and the Kaituna catchment control scheme was implemented to reduce diffuse inputs through extensive riparian planting. Since then the nutrients in Lake Rotorua water have reduced significantly, but the sediments in both lakes still contain high levels. The algal bloom in Lake Rotoiti is fed by nutrients from a range of sources, including Rotorua outflows, the lake sediments, farm run-

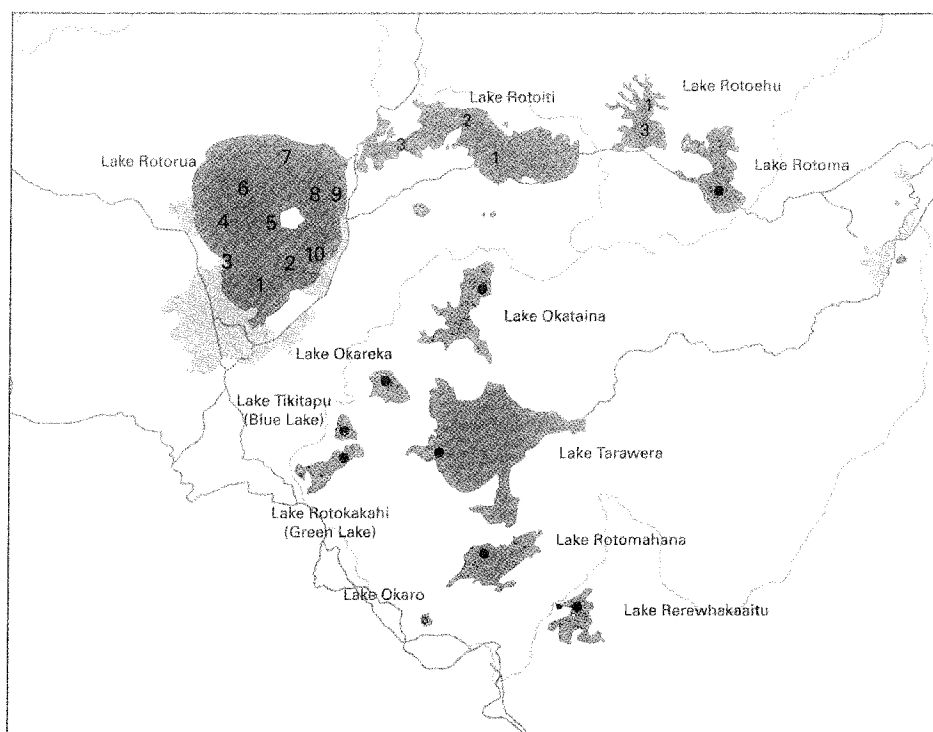


Figure 44.2 Major lakes in the Rotorua district (numbers and dots represent Environment BOP sampling sites).

off, septic tanks (most of the settlements around the lakes are not yet connected to a sewerage system), and groundwater (which can take some decades to reach the lakes).

Management response

Environment Bay of Plenty is responsible for water quality in eleven lakes in the Rotorua area, of which five are below the target quality set in the Council’s Proposed Regional Water and Land Plan. The main lakes and sampling sites are shown in Figure 44.2. A series of lake action plans is being developed to reduce nutrient flows in the most cost-effective manner (www.ebop.govt.nz). This is likely to involve various combinations of forestry on pastoral land, riparian retirement, wetland establishment, and sewage reticulation. Lake Okareka is a pilot for this work; lakes Rotorua and Rotoiti represent more complex problems with many questions still to be answered.

To help reduce the nutrients going into the lakes, Environment Bay of Plenty has included “Rule 11” in its Proposed Plan. This rule would limit any new development or farm intensification that might increase the nutrients flowing into the five most degraded lakes. Environment Bay of Plenty is also investigating engineering solutions, such as diverting water from the Ohau Channel directly down the Kaituna River to reduce the flow of nutrients from Lake Rotorua into Lake Rotoiti, water treatment options such as aeration, and chemical solutions such as activated clay treatment to reduce the release of nutrients from the sediment. Environment Bay

of Plenty has endowed a Chair in Lakes Water Quality and Management at the University of Waikato to carry out research on the Rotorua lakes.

Case Study III: Lake Taupo water quality

Problem

With an area of 622 square kilometres and a mean depth of 95 metres, Lake Taupo is our largest lake and undoubtedly one of New Zealand's greatest natural treasures. Its clear, blue waters attract tourists from all over the world and the lake is renowned for its trout fishing. The lake is a taonga of the Ngati Tuwharetoa. However, the lake is threatened by increasing agricultural and urban development in its catchment. Monitoring and modelling by Environment Waikato and NIWA show that increasing nitrogen inflows are feeding higher concentrations of planktonic algae, turning the waters greener and more turbid. The increasing nutrients are also causing increases in growth of slimes and some weeds in the littoral zone, and increases in the frequency and extent of cyanobacterial (blue-green algal) blooms in parts of the lake.

The currently declining quality of the lake water reflects land-use changes that began in the 1950s. The current inflows of groundwater are affected by the land-use activities of 30 to 50 years ago, as it takes that long (on average) for groundwater to transport pasture and excreta-derived nitrogen to the lake. Even maintaining the current land-use patterns, there would still be significant additional degradation over the next 30 years. The natural annual influx of nitrogen to the lake is estimated at 650 tonnes per year. The current nitrogen influx however is 1100 tonnes per year, and the likely nitrogen influx from existing land uses, when it arrives in the lake, could be as high as 1300 to 1500 tonnes per year.

There are significant Treaty of Waitangi implications for any management response. A "do-nothing" decision will fail to protect the lake, a taonga of Tuwharetoa. However any regional rules limiting activities that generate nitrogen leaching may constrain the options for Maori owners to develop and further profit from their land.

Minimising impacts

Environment Waikato sought feedback from the Taupo community about management objectives and desired water quality outcomes. There were three primary scenarios—restore Taupo to a natural state, maintain the present state, or allow current trends to continue. The community chose the middle option—preventing water quality from falling below the present condition in the long term, while acknowledging that it would inevitably decline over the next 30 years before the effects of changes in land use allowed it to recover to its condition in the year 2000.

To prevent further long-term degradation, manageable nitrogen inputs need to be reduced by 20%. About 90% of this reduction needs to come from converting farms to low-nitrogen land uses such as forestry or retiring the land and letting it revert to bush. This could be achieved through re-establishing native bush or by plantation forestry on about 13,500 hectares of pastoral land, although several other land-use permutations are possible. Also required are additional reductions in nutrient discharges from lakeshore settlements, including the Turangi sewage treatment plant, the Taupo stormwater system and from many settlements serviced by septic tanks.

Management response

Early concerns about the potential effect of land development on the lake and stream condition were addressed in major land retirement and reservation projects. The Lake Taupo Catchment Control Scheme and the Taupo Reserves Scheme commenced in the 1970s. All remaining Crown indigenous land and regenerating forests were eventually reserved from development in 1987, with over 100,000 ha of public land set aside in the catchment for conservation and recreation. While this was clearly necessary to safeguard the lake, there was some doubt about whether it was sufficient to prevent further degradation, especially from sediment. Recent trends in water quality have proved that it was not enough.

Since 1999, Environment Waikato, Taupo District Council, Tuwharetoa and central government (Ministry for the Environment) have developed a partnership to protect the lake. Most farmers in the Lake Taupo catchment are represented by Taupo Lake Care, which has been very active in all discussions. As at December 2003, details were being negotiated, but were likely to involve:

- rules in a regional plan to cap nitrogen losses from farm and forest land at current levels;
- an incentive fund to shift some farm land to low nitrogen uses, in order to achieve the 20% reduction in nitrogen inflows in the long term;
- septic tank improvements, sewerage reticulation and urban sewage treatment upgrades;
- research and extension to provide more nitrogen retention options for those landowners that continue farming; and
- further local and central Government political initiatives.

Environment Waikato announced in November 2003 that the estimated public cost of this programme would exceed \$83 million. A summary is contained in Petch, *et al.* (in press).

Case Study IV Effects of dairying on freshwaters

Problem

Dairying is an important industry in New Zealand, involving 3.9 million dairy cows (number of cows in milk in the 2002/03 season) and using 11% (1.76 million ha) of the country's total agricultural land. Dairying produces 20% of New Zealand's total export income (\$5.9 billion in the year to March 2003).

However, like most intensive land uses, dairying affects water quality and water environments. The potential effects of dairying include the altering of stream flow regimes due to intensive drainage and irrigation, and stream channelisation and habitat loss. Conversion to pasture can cause increased stream temperatures due to riparian shade removal. Stock access to stream margins can result in increased bacterial contamination, and high nutrient inputs can lead to eutrophication.

The increasing demand for irrigation, in areas like Canterbury, can result in associated contaminant leaching to groundwater and streams. The intensification of existing dairy farms and the expansion of dairying into new regions have increased the importance of addressing their impact on water environments.

Minimising impacts

There are a number of ways the impacts of dairying can be minimised. Streams can be fenced off to prevent direct stock access. Dairy shed effluent can be effectively treated and disposed of, especially through irrigation onto land. Riparian margins can be managed to control nutrient, sediment, and microbe runoff, as well as providing important stream habitats. The use of fertiliser, effluent and other sources of nutrients can also be effectively managed.

Management response

Many farmers are already adopting these techniques to reduce the effects of dairying on streams and lakes. Some regional councils are adopting regional plans and other activities, and dairy companies and rural extension agencies are providing information to farmers.

A "Dairying and Clean Streams Accord" was signed in May 2003 by the Minister for the Environment, Minister of Agriculture, Chairman of Fonterra, and the Chairman of the regional councils' Regional Affairs Committee. It focuses on reducing the impact of dairying on the quality of New Zealand streams, rivers, lakes, ground water and wetlands, and specifies priorities and performance targets. The minimum targets are to be achieved on a nationally aggregated level, which allows some regional flexibility. The targets are:

- Cattle are to be excluded from 50% of streams, rivers and lakes by 2007 (90% by 2012).

- Farm races are to include bridges or culverts where stock regularly cross a watercourse (50% by 2007, 90% by 2012).
- All farm dairy effluent discharges are to comply with resource consents and regional plans immediately.
- All dairy farms are to have in place systems to manage nutrient inputs and outputs by 2007.
- Existing regionally significant or important wetlands, as defined by regional councils, are to be fenced and their natural water regimes protected (50% by 2005, 90% by 2007).
- Fonterra and regional councils will develop regional action plans for the main dairying regions to implement this Accord by June 2004.

The targets have been criticised by the Fish and Game Council, however, as being insufficiently ambitious. Federated Farmers considered that these matters should be addressed by regional plans under the Resource Management Act, rather than through an industry initiative.

The Accord is supported by:

- Fonterra's Assessment Protocol, through which Fonterra will assess the status of its suppliers;
- *Market Focused*, an environmental management system that will help farmers improve their environmental performance. This was part-funded through Ministry for the Environment's Sustainable Management Fund;
- Regional Action Plans prepared by Fonterra and regional councils. These Plans will identify local commitments by regional councils and Fonterra, and will describe clear, regional, time-bound targets for the priority actions; and
- a monitoring programme coordinated by the Ministry for the Environment that will report overall progress towards the Accord targets.

Integrated catchment management

Successful water management is about managing the whole catchment in an integrated way to minimize downstream impacts. Integration requires collaboration among the different scientific disciplines, a common purpose for environmental management from scientists, decision-makers in councils, land users in the catchment, and interest groups, and integration of environmental management with social and economic concerns (Fenemor 2002).

Integrated catchment management isn't new to New Zealand. The establishment of catchment boards in the 1940s was visionary, providing the foundation for today's catchment-based approach to land and water management. However, as the case studies in this chapter show, for sustainable environmental management, and to live up to our reputation as "Pure New Zealand" and "clean and

green”, we must do better at integrated catchment management for freshwater protection, not only in our rural catchments but in urban areas as well.

Case Study V: The Motueka River

Problem

The Motueka River in Tasman District (Fig. 44.1) is fed in part from Kahurangi National Park, and it borders Nelson Lakes and Abel Tasman National Parks. The catchment has many outstanding characteristics (as determined by the Special Tribunal hearing the water conservation order application):

- the wild and scenic character of streams within the Department of Conservation estate;
- karst cave systems from Mt Owen to Mt Arthur and in the Riwaka River catchment;
- a brown trout fishery on the Motueka River from the Wangapeka River down to the Motueka Plains.

The Motueka and Wangapeka rivers west of Nelson are regarded as being among New Zealand’s best brown trout fisheries, attracting anglers from around the world. However, trout numbers have declined significantly since 1985. A number of reasons have been given for the decline—increasing water extraction during low flows, the impacts of major floods, excessive sediment affecting fish habitat, and increased angling pressure.

This case study illustrates the value of mediation and negotiation for resolving complex water management planning issues. It also demonstrates the importance of hydrological and other environmental data at the catchment scale for formulating water management policy—in the Motueka catchment, this is now one focus of a nationally funded research programme in Integrated Catchment Management (Basher 2003).

Management response

To protect the fishery, the predecessor organisation to Fish and Game New Zealand applied in 1987 for a Water Conservation Order over the entire Motueka catchment, seeking to limit water extraction, and protect water quality and aquatic habitat.

To protect the nationally important trout fishery, the Tribunal set water quality limits, prohibited all dams greater than 3 metres high, and limiting water allocations to those limits specified in the Tasman District Council’s informal Motueka and Riwaka Plains Water Management Plan (Tasman District Council 1989). Eighteen parties appealed, and resolution of these appeals was delayed pending legal decisions in the equally long process for deciding the Buller River Water Conservation Order.

The key management issue was a low flow management regime that would protect the fishery, while accom-

modating future water demand in the Motueka catchment. Water quality limits, especially limits on sediment discharge affecting aquatic habitats, were a second issue. The water allocation issues were resolved during 1999–2002, through discussions between the Tasman District Council as regulator, Fish and Game as applicant, the Department of Conservation, and an *ad hoc* Land Owners Water Action Group convened by Federated Farmers. A rewritten draft Water Conservation Order was then presented to the Environment Court at a hearing in July 2002 with a “Statement of Agreed Facts” and affidavits supporting the draft. In its decision six months later, the Court recommended that the Minister formally gazette the Order.

Negotiated water allocation limits removed the originally proposed minimum flow regime, replacing it with a limit of 12% of the flow extracted in the Motueka mainstem, and 6% in the Wangapeka River, as measured by the actual residual flow. Minimum flows were established in three contributing catchments to protect trout spawning from May to October inclusive. The Motueka River Water Conservation Order, following a 14-year process, was approved by the Environment Court in January 2003 (Decision W7/2003) and recommended for gazettal by the Minister for the Environment.

Among the criticisms of Water Conservation Orders have been the costs of the extended legal processes for the 13 already heard. The Motueka case study demonstrated for the first time for a Water Conservation Order that mediation could achieve a robust, agreed outcome without the need for prolonged hearings of appeals. The most contentious issue requiring negotiation was the setting of limits on water extracted from the Motueka River and its tributaries.

The “shared suffering” approach agreed for the Motueka and Wangapeka aligns with similar approaches in Tasman District Council water management planning. From the perspective of instream values, it maintains variability in low flows, and provides a sufficient flow buffer for the fishery and instream habitat to recover after severe droughts. From the perspective of the water users, the agreed percentage limits accommodate projected water demand, mainly for irrigation. It recognises that the water requirements of different crops vary through the irrigation season, and actual water use is less than allocations. Groundwater extractions distant from the river will have a less than 1:1 effect on flows. The agreement allows for a 10-year security of supply standard.

The agreement recognised that the place for setting detailed water allocation and rationing rules is in the Tasman District Council’s resource management plan, not in a Water Conservation Order, which once gazetted could be difficult to alter.

Case Study VII: Urban catchment management

Problem

Urban catchments represent only 3% of New Zealand's total land area. However, 85% of us live in urban centres, and city folk value and interact more extensively with urban environments than with other environments (Enviro Ventures *et al.* 2002).

Urban catchments typically exhibit some of the most intensive land use in New Zealand and so it is not surprising that environmental impacts are among the greatest (see Figure 44.3). The main concern of past urban managers has been to move flood waters out of urban and industrial areas as quickly as possible. Urbanisation of catchments invariably involves the clearance and re-contouring of land and the creation of impervious surfaces. This can have many adverse effects—increased flooding and erosion, reduced low flows in urban streams, the loss of stream habitat by channelling, and water contaminated with suspended sediment, pathogens, and toxic substances.

Streams, wetlands, estuaries and lakes within urban catchments are highly susceptible to the effects of urbanisation. The effects of urbanisation are most pronounced and visible in Auckland due to the city's size. Studies of the shallow estuaries surrounding the Auckland isthmus show toxic levels of contaminants there and predict that these levels will continue to increase steadily. Urban streams have been channelled, and in a number of cases piped, and typically have low ecological and aesthetic values. Some bathing beaches that receive stormwater are often unsuitable for swimming after rainfalls due to cross connections or overflows from sewers.

Minimising impacts

The impacts of urbanisation on water resources can be reduced (Enviro-Ventures *et al.* 2002).

Sources of contaminants need to be identified and controlled, including manipulating the bioavailability and toxicology of contaminants and treating stormwater. Hydrological changes (permeability, flows, erosion) and morphological changes to streams can be managed, and streams and their riparian areas can be restored. Urban streams need integrated catchment management; tools must be developed to help managers predict the effects of development, set priorities and make long-term predictions of impacts.

Management response

Historically, the management of the effects of urbanisation has been rather piecemeal, with the development of subdivisions, and stormwater and sewage reticulation and disposal addressed largely independently.

Auckland urban catchments are subject to the most detailed management in New Zealand. The Auckland Regional Council requires that all stormwater discharges within the urban centre have resource consents, and applications for consents must be supported by an integrated catchment management plan. These plans require a description of management responses to ensure Council water quality criteria are met. Environmental targets, and a recently developed monitoring and assessment protocol, are intended to determine the effectiveness of catchment management and to trigger responses. Auckland Regional Council has produced

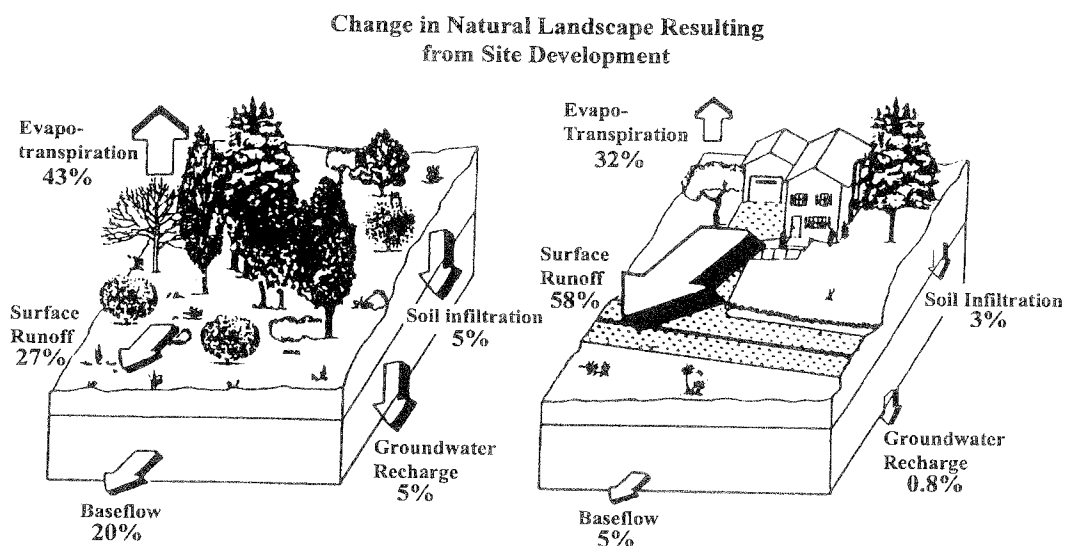


Figure 44.3 Impacts of site development (Enviro-Ventures *et al.* 2002)

technical guides to assist the management of stormwater and to control erosion and sediment.

Environment Canterbury and the Christchurch City Council have focussed on urban stream protection and restoration programmes. The Rotorua District Council has retrofitted sophisticated treatment systems for much of the flow of stormwater from the central business district before it is discharged into Lake Rotorua. The Wellington Regional Council has undertaken regional mapping of the effects of stormwater.

There is confusion in some areas over apparently conflicting responsibilities of territorial authorities under the Local Government Act and regional councils under the RMA. Many councils have adopted management frameworks that allow them to develop locally appropriate solutions.

DECLINING BIODIVERSITY VALUES

Case study VIII: New Zealand Biodiversity Strategy and Action Plan

Problem

The state of New Zealand's freshwater biodiversity is summarised in the New Zealand Biodiversity Strategy (Department of Conservation and Ministry for the Environment 2000), which points out a number of concerns.

Few lowland river systems have any protection. Many are ecologically degraded through biological invasions, reduced water quality, channelisation, sedimentation and the removal of floodplain connections. Only a couple of complete river systems still lie within unmodified catchments and remain free of alien species.

Many of New Zealand's 700 shallow lakes are degraded by nutrient enrichment, causing eutrophication—some to the extent that they are unable to support fish or rooted plants. Most lake fish communities are dominated by alien species and invasive alien plant species are extensive in most lake systems.

Swamps, bogs and marshes now cover only 1000 km², representing a loss of over 90% of New Zealand's original wetlands. There is considerable variation, ranging from a loss of 63% in Southland to a 99% loss in the Bay of Plenty region. Of the remaining wetlands, many are degraded by weed invasion, stock, modifications to the hydrological regime, and barriers to fish passage.

Many freshwater species now have a very restricted distribution because of habitat loss and modification, alien species and harvesting pressures.

Management response

The New Zealand Government prepared a biodiversity strategy and action plan in 2000, which contains a suite of

actions intended to reduce the decline in the extent and condition of freshwater habitat; and improve the status of threatened freshwater species. The Department of Conservation is preparing a Freshwater Natural Heritage Strategy to assist prioritisation of its management actions in implementation of the Biodiversity Strategy. As at December 2003 waters of national importance are being identified as part of Central Government's Sustainable Development Programme of Action for Freshwater.

Invasions from alien biota

Earlier chapters have described how alien species have had a detrimental impact on freshwater ecosystems, especially those in lowland environments. In most lowland lakes, plant and fish communities are now dominated by alien species. Few freshwater systems are free from introduced fish species, with the distribution of pest fish species such as gambusia, rudd, perch, and carp continuing to increase.

Case study IX: Spread of invasive weed species, especially hornwort, in New Zealand lakes

The problem

Submerged alien plants, especially those from the Hydrocharitaceae ("oxygen weeds") and the Ceratophyllaceae families, have been highly successful in invading New Zealand lakes. These plants exclude and replace native plants and change the habitat for other species, thereby reducing biodiversity. In addition, as they are tall and reach the surface of the water, they adversely affect recreation and amenity values and can interfere with hydro-electric generation and water abstraction. Today, few lakes in lowland environments are free from invasive alien plants.

Hornwort (*Ceratophyllum demersum*) is a relatively new invader from the Ceratophyllaceae. It is now ranked as one of the highest-risk aquatic weed species in New Zealand (Champion and Clayton 2001). It tends to completely displace native vegetation and often most other invasive species as well. The methodology for LakeSPI (a simple method for monitoring lake ecological condition) assigns hornwort the highest score for "invasive species impact" (Clayton *et al.* 2002).

When hornwort first began to spread through North Island lakes, it was assumed it would behave like other invasive weed species. In particular it was assumed that hornwort would be subject to the depth limitations of other invasive species and that the native characean meadows would at least survive in depths below that occupied by the invasive species. This would not, of course, apply in shallow lakes or lakes where poor water clarity restricts the lower depth of plant growth for invasive species. Recent work has, however, shown that hornwort

is able to completely displace native vegetation to depths well below those previously affected by other invasive weed species (Clayton *et al.* 2002). This has been observed in the deep clear water North Island Lakes Taupo and Tarawera.

Hornwort has now spread throughout much of the North Island. The first incursion into the South Island was recently recorded in waters in the Motueka area where coarse fish have been illegally introduced. Implementation of an eradication plan for the Motueka incursion began in 2003.

The overlapping and ambiguous responsibilities of the several agencies involved in different aspects of biosecurity and waterbody management, combined with a lack of regular surveillance and early detection of invasions, has meant that hornwort has largely spread throughout the North Island without constraint.

Management response

The response of management agencies to the spread of invasive weeds, especially hornwort, has been minimal, apart from limited monitoring and education activities. There has been minimal surveillance in recent years and little remedial action except to control the extent of major weed beds where they become stranded in storms on public amenity areas or where they directly affect economic activities such as land drainage and hydro-electric power generation.

One notable exception is Lake Waikaremoana in Te Urewera National Park. Here the Department of Conservation has demonstrated the benefits of regular surveillance for invasive weed species. The Department detected the arrival of *Lagarosiphon major* in Rosie Bay in 2000. At the time, the only alien invasive weed species present was the less aggressive *Elodea canadensis*. Responding to clear policies in the management plan, the Department, in association with NIWA, undertook an eradication programme. As at November 2003 the access to Rosie Bay is still closed and regular checks to remove the last fragments of *Lagarosiphon* are continuing.

ENVIRONMENTAL REPORTING

Section 35 of the Resource Management Act requires every local authority to gather information, undertake or commission research as is necessary for the local authority to carry out its functions under the Act. Every local authority is required to monitor the environment in its area, as well as the efficiency and effectiveness of the policies, rules and other methods in its policy statements and plans. The results of this monitoring are to be made available to the public at intervals of not more than five years. Most regional councils are publishing State of the

Environment reports for their regions on a regular basis. Aggregation of these to provide national-level reporting for freshwater is very limited at present.

The Ministry for the Environment is currently developing standard indicators, methods and reporting frameworks for national environmental performance and state-of-environment reporting. This will include indicators for freshwater quantity, quality and biodiversity.

WATER MANAGEMENT IN NEW ZEALAND: AN EVALUATION

This book has described the many challenges of water management in New Zealand. An earlier evaluation of the effectiveness of the new Resource Management Act framework and policies in managing freshwater resources was undertaken by Memon (1997). The evaluation below builds on and supplements the earlier review.

Successes

Large contaminated discharges to freshwater have generally been well controlled under the Resource Management Act and one of its predecessors, the 1967 Water and Soil Conservation Act. As the terms of permits issued under the earlier legislation expire, regional councils are using the Resource Management Act to tighten conditions and reduce the impact of these discharges. Many regional councils are also starting to use a variety of regulatory and non-regulatory methods to reduce the impact of diffuse sources of nutrients and other contaminants on water bodies. The effectiveness and adequacy of these has not yet been fully assessed.

Thirteen regional councils have operative or proposed water allocation plans. Most of these specify minimum flows, at least for important rivers, but fewer specify allocation limits. Several councils have developed innovative allocation solutions, as described in some of the case studies. Funding to ensure the effective implementation of regional plans is variable. Adverse environmental consequences of water allocation have, to date, generally been limited to a few, relatively restricted regions. Few applications to take water have been declined or significantly constrained.

The significant opportunities for public input into council decisions on plans and many consent applications have provided a well informed basis for decisions on medium to large projects. However, some community groups have expressed concerns about their limited ability to participate in decision-making, especially in relation to non-notified (minor) consents. Businesses have complained about excessive costs imposed as a result of consultation requirements.

There have been significant advances in the procedural and technical aspects of water management. The precise

requirements of crops are better understood and irrigation systems have been developed to apply the right quantity of water at the right time. Wastewater treatment continues to improve and, increasingly, liquid wastes are discharged onto land, through which nutrients become a resource, not a waste.

Environmental monitoring and research has increased our understanding of water resources and human impacts. In particular, our understanding of microbiological and trace chemical contaminants in surface waters and our modelling of groundwater has improved.

The Water Conservation Order process has been used to provide secure protection for a dozen large lake or river systems with outstanding values. Although the legalistic procedures are lengthy and expensive, the processes are intended to enable all views to be strongly contested and the outcome therefore robust.

Outstanding issues

Because of New Zealand's relatively low population and small industrial base, the distribution of adverse effects from water abstraction and point source discharges has been relatively restricted. Where human or farm animal population densities are high, however, such as in the vicinity of major cities or on developed flood plains, the physical, chemical, ecological, and aesthetic condition of our waters is compromised, sometimes even by international standards. Intensive agriculture has degraded the quality of many lowland water bodies—some are not even suitable for stock watering. Others have regular algal blooms and contact with the water is hazardous to human health.

The land-use planning functions of both territorial authorities and regional councils affect downstream water quality and quantity, and aquatic habitats. Unclear accountabilities, existing rights to continue current land use, together with the often long lags between changes in land use and their impact on water quality, means that the condition of many water bodies continues to deteriorate, for example lakes Rotorua, Rotoiti, Okaro, Rotoehu, and the lakes of the lower Waikato basin. Lake Horowhenua has, however, a rare example of a large-scale lake margin retirement and planting programme, driven largely by iwi and the community.

Few lakes now contain biological communities dominated by indigenous species. A lack of co-ordinated public agency planning and management has seen a wide variety of invasive plant and fish species becoming established in increasing numbers of lakes. An exception to this is the Department of Conservation eradication programme for *Lagarosiphon* oxygen weed in Lake Waikaremoana and the associated ongoing monitoring.

Only a few larger water bodies have received the degree of secure protection that can be conferred by water

conservation order status; smaller water bodies generally can only be protected through regional plans. This means that many water bodies, especially in the lowland areas, are vulnerable to cumulative development pressures.

The amount of land under irrigation is projected by the Ministry of Agriculture and Forestry to rise by about 40% from 2003 to 2013. While negative environmental and economic implications of water allocation have been limited to particular areas and water bodies, this will not continue, as the "easy water" (shallow groundwater and run-of-river) will soon all be allocated. Alternative allocation mechanisms and water harvesting (e.g., collection of flood peaks) will soon be needed if communities choose to meet the continued growth in demand for irrigation water.

The existing Resource Management Act regulatory framework assesses applications on the basis of "first in, first served". This means that water vested in the Crown is allocated to the first applicants to meet the "effects-based" tests in the Act, rather than to the uses that would provide the greatest benefit for society. This can result in a "Gold Rush" mentality for filing applications and a risk of over-allocation. While the RMA provides for transferable water permits, few councils have provisions in their water allocation plans to facilitate the transfer of permits between water users.

The RMA is structured on the basis that day-to-day water management would be the responsibility of regional councils, with formal guidance provided by the Minister for the Environment. However, to date (December 2003) no formal policy directives or standards regarding water management have been issued.

Water management plans are not mandatory under the RMA. The absence of such a plan for the Waitaki River catchment contributed to the central government call-in during late 2003 of approximately 300 applications from a variety of applicants (for hydro-electricity development and irrigation purposes) and the development of special legislation to plan for water allocation for the river.

Even though the RMA and other planning legislation related to water management are strongly dependent on plans specifying outcomes agreed by communities and setting measurable objectives (e.g., water quality objectives in Schedule 3, and Instream Management Objectives in the Instream Flow Guidelines), progress has been slow and the degree of specificity often limited or ambiguous. This can have the effect of transferring inevitable trade-off decisions from the planning process to the less appropriate resource consenting process.

The importance of freshwater, and its efficient and equitable management, to New Zealand's economy and heritage protection has been recognised by the RMA, including the challenging issues of water allocation failure,

the effects on water quality of land-use intensification, and the protection of water bodies of national importance within government's Sustainable Development for New Zealand: Programme of Action. This has identified the need to clarify the national interests in freshwater, provide national direction, improve governance frameworks and provide new technical tools and economic instruments to managers. It is anticipated that this programme and the various delivery projects will continue into the future.

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Chapter 43

Analysis of instream values

Ian Jowett and Paul Mosley

INTRODUCTION

There is increasing interest in understanding stream and river environments and protecting them from harmful human impact. This has been driven, in part, by the ecological and visual effects of past flow abstraction, regulation and canalisation of streams and rivers. Rivers have been modified for hydropower, irrigation, flood control and protection, water supply schemes and reclamation of wetlands for farming. The changes have often led to a loss of physical habitat diversity, and through that to a decrease in biological diversity, sometimes even to a total elimination of some of the biota.

Rivers have many and diverse values, both intrinsic and for aquatic organisms and people. The Resource Management Act (RM Act) has, since 1991, defined a framework of “sustainable management” within which a river’s values can be investigated, compared, and managed (Chapter 42). Section 5 of the Act implies that rivers (which are just one of the natural and physical resources which the Act addresses) have value to people and communities in terms of “their social, economic, and cultural well-being and for their health and safety”. Sections 6 and 7 list specific attributes that could be (and in some cases specifically are) described as “values”—natural character, outstanding natural features, amenity values, intrinsic values of ecosystems, habitat of trout and salmon, and the values placed on water by Maori. Clearly, a river’s values are not necessarily expressed in terms of money, or usefulness to people (Chapter 42).

The management of instream flow is a complex process, usually involving a combination of technical, public, and legal considerations. To be effective, instream flow managers should consider the present status of the river and its ecosystem, and then, in consultation with public and institutional organisations, set goals and objectives before establishing appropriate flow requirements.

Conventionally, we recognise instream values and uses—those that are associated with water that is flowing in a channel, and out-of-stream values and uses—those that are enjoyed when the water is taken from the channel for use elsewhere. The “social, economic and cultural” values referred to in Section 5 may be either instream or out-of-stream values. For example, water used for irrigation has out-of-stream value, whereas the water used by a commercial rafting enterprise has instream value. The attributes listed in Sections 6 and 7 of the Act generally are instream values.

The principle that water is an economic good for which there are alternative uses (including non-use) has become widely accepted internationally. It implies that the competing values of water should be compared objectively and quantitatively, so that water can be managed to best achieve the community’s aspirations. The benefits of using water for purposes such as irrigation, hydropower generation, and industrial or urban water supply can be expressed in monetary terms. For example, using many years of data and research on farm production, it is possible to demonstrate the monetary value of a certain quantity of water for agriculture, and for the rural economy as a whole. The benefits of allocating water for the instream values listed in Sections 6 and 7 of the RM Act generally cannot be expressed in monetary terms, although methods such as contingent valuation have been used occasionally in New Zealand to quantify the value of a water resource for non-commercial recreation. This tends to be detrimental to instream values, because “money talks”. Nevertheless, considerable recent progress has been made in quantifying the requirements of instream uses in terms of hydrologic and hydraulic parameters, and this is placing the appraisal of instream values on an increasingly strong footing (Chapter 8). This chapter summarises the requirements of instream uses and the principal methods used in New

Zealand to determine their relationship with flow, and describes some cases where biological responses to flow changes have been recorded. The two volumes of *Flow guidelines for instream values* (Ministry for the Environment 1998) provide a very much more comprehensive coverage than is possible here.

INSTREAM VALUES AND THEIR REQUIREMENTS

The instream values listed in Part II of the RM Act may be grouped into:

- Ecological or intrinsic values
- Landscape, scenic and natural characteristics of the river
- Amenity values—recreational angling and fishing
- Amenity values—boating and other recreational activities undertaken in, on or near the river
- Maori values

There are, of course, overlaps and linkages among these values. They differ in the extent to which they are influenced by variations in flow regime. “Flow-related values” change in a discernible way as flow changes. For example, the value of a particular river for rafting is likely to decline as flow declines, as the water becomes shallower (causing more frequent grounding), slower (requiring more effort for paddling) and narrower (making navigation more difficult). At the other end of the scale, increasing flow also may make the river increasingly dangerous for rafting, and there will be a range of flows that is preferred or optimal for the sport. “Flow-enhanced values” are those that do not depend on the availability of a particular flow, but nevertheless are enhanced by the presence of flowing water. For example, many New Zealanders visit river banks for family picnics and informal recreation. The sight and sound of flowing water enhances the experience, and the water provides additional recreational opportunities such as paddling and bathing, but is not absolutely essential. “Flow-independent values” are those values of a river that change to a minor or no extent as the flow changes. For example, the visual attractiveness of a river may be determined by attributes of the scene that are unrelated to water flow, such as the type of riparian vegetation or the presence of alpine vistas. Changes in flow do not affect these aspects of river scenery (except that on a long time scale, a change in flow regime may lead to a modification of riparian vegetation).

Holistic flow assessment

For long-term solutions to river flow management, we need to take a holistic view of the river system, including its geology, fluvial morphology, sediment transport, riparian conditions, biological habitat and interactions, and water quality.

The instream flow incremental methodology (IFIM; Stalnaker *et al.* 1995) is an example of an interdisciplinary framework that can be used in a holistic way to determine an appropriate flow regime by considering the effects of flow changes on instream values, such as river morphology, physical habitat, water temperature, water quality, and on sediment processes, such as flushing flows and channel maintenance flows (Fig. 43.1). Its use requires a high degree of knowledge about seasonal and life-stage requirements of biota and inter-relationships of the various instream values or uses.

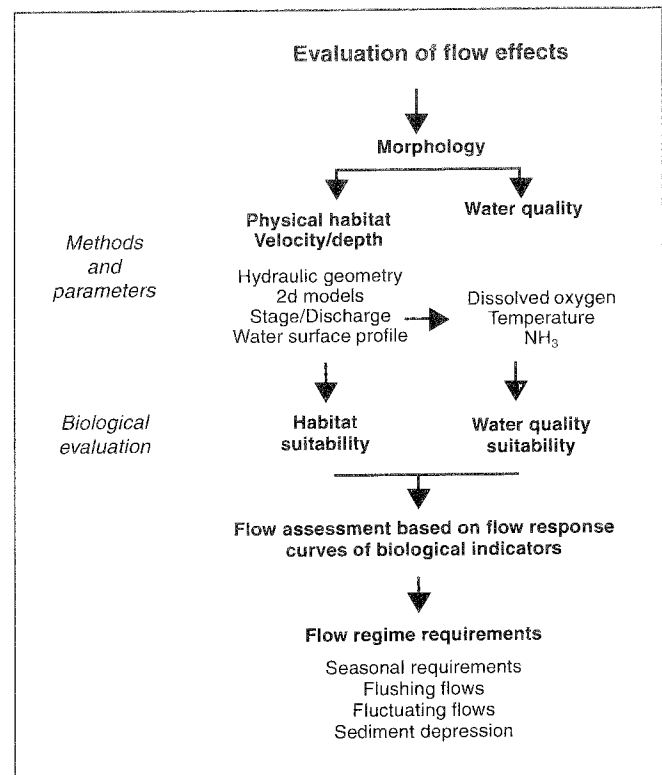


Figure 43.1 An Instream Flow Incremental Methodology (IFIM) framework for the consideration of flow requirements.

Other flow assessment frameworks are more closely aligned with the “natural flow paradigm” (Poff *et al.* 1997). The “range of variability approach”, and the associated “indicators of hydrologic alteration”, allow an appropriate range of variation, usually one standard deviation, in a set of 32 hydrologic parameters derived from the “natural” flow record (Richter *et al.* 1997). The implicit assumption in this method is that the natural flow regime has intrinsic values or important ecological functions that will be maintained by retaining the key elements of the natural flow regime. Arthington *et al.* (1992) described a “holistic method” that considers not only the magnitude of low flows, but also the timing, duration and frequency of high flows. This concept was extended to the “building block

methodology”, which “is essentially a prescriptive approach, designed to construct a flow regime for maintaining a river in a predetermined condition” (King *et al.* 2000). It is based on the concept that some flows within the complete hydrological regime are more important than others for the maintenance of the riverine ecosystem, and that these flows can be identified, and described in terms of their magnitude, duration, timing, and frequency.

In concept, the building block methodology is similar to the incremental methodology in aiming to maintain a prescribed condition, based on a high degree of knowledge about flow requirements of the various aspects of the ecosystem. However, identification of flow requirements in the building block methodology is based more on the “natural flow paradigm” than on an understanding of physical and biological relationships. A basic assumption of the building block methodology, and the major point of departure from the incremental methodology, is that biota associated with a river can cope with naturally occurring low flows that occur often and may be reliant on higher flow conditions, and that flows that are not characteristic of the river will constitute an atypical disturbance to the ecosystem and could fundamentally change its character (King *et al.* 2000).

Significant elements of the flow regime

Attention often has focussed on determining the low-flow conditions required to maintain particular instream values, because at this time there is the greatest competition for the limited amount of water that is available, and the river ecosystem is most under stress. However, a number of aspects of a river’s flow regime may influence its ability to maintain particular instream values.

Large floods, of the order of the mean annual flood and greater, are responsible for the overall form of an alluvial river channel. They are known as channel maintenance flows and also influence the nature of the river corridor—the floodplain surface, vegetation cover, and need for river control measures such as willow planting and groynes. Hence, large floods have a significant influence on the natural character of a river (RM Act Section 6(a)), on the presence of areas of significant indigenous vegetation and significant habitats of indigenous fauna (RM Act Section 6(c)), and on the amenity, intrinsic and heritage values of the river corridor (RM Act Section 7(c), (d), (e)). Large floods also are a major cause of disturbance to the river ecosystem, with potentially significant impacts—at least for a time—on life-supporting capacity, as aquatic biota are displaced and their habitats temporarily destroyed. Large floods during October to December can be particularly disruptive for bird species, such as the wrybill plover, that nest on river beds, although such birds may

re-nest once or twice, if it is not too late in the season (Hughey 1985). Similarly, floods that occur during incubation, emergence or early fry stages (August–November) can severely affect a river fishery in subsequent years, by reducing recruitment to the population (Hayes 1995).

Smaller floods and freshes, with a frequency of a few times each year, are contained within the channel, and therefore have a more restricted effect than large floods. Nevertheless, they are able to mobilise sediment on at least some areas of the river bed, remove periphyton and other aquatic vegetation, and assist juvenile salmonids on their passage to the sea. They generally “flush” and “refresh” the river bed by removing silt and algal coatings, and inhibit vegetation from colonising the riverbed gravels that are not covered by flowing water and, in terms of flow requirements, they are known as flushing flows. As with large floods, the effects of freshes can be both positive and negative—the effect of “flushing” and “refreshing” the river on the one hand, and the effect of disturbing and disrupting parts of the ecosystem on the other. An index of these ecologically significant freshes is the statistic *three times the median flow* (Clausen and Biggs 1997). Of particular significance may be the timing and spacing of freshes, particularly from the point of view of flushing the river and opening river mouths to allow passage for diadromous fish species. The time required for aquatic biota to re-establish after disturbance by a fresh depends on the life cycle of the species. Macroinvertebrates tend to re-colonise streams within weeks, whereas trout may take years to re-establish.

Flow recessions, the period during which flow is declining after a flood or fresh, can be of particular significance for amenity values. For much of the time, flows in small to medium-sized rivers are less than desirable for recreational boating, and may restrict the availability of spots suitable for angling. During a flood recession, flows are higher than usual for a few days, enhancing recreational opportunities. For example, the preferred flow for salmon angling in the Rangitata River is between 70 and 110 m³/s, and for rafting and kayaking is between 50 and 100 m³/s. This range of flows tends to be experienced during recessions, and its maintenance is regarded by many river users as crucial to sustaining the recreational value of the river.

Low flows are particularly important because they are the times at which there is greatest competition for water, the total wetted area of aquatic habitat is least, and the aquatic ecosystem is likely to be under greatest stress. On the other hand, stable low flows can be periods of high biological productivity, which permit recolonisation of the riverbed by macroinvertebrates and fish after a flood, and re-establishment of aquatic vegetation. Analysis of instream

habitat requirements and general observations suggest that, surprisingly, low flows can support good native fish and benthic invertebrate populations in small streams, that higher flows are required for juvenile salmonids, and that adult trout and salmon are most numerous in larger rivers and have the highest flow requirements of all the fish species in New Zealand.

Rivers have a distinctive annual flow regime, in which flows vary through the year in response to the annual distribution of rainfall, evapotranspiration, and snowmelt (Chapter 7). This regime is an element of the natural character of a river (Chapter 8), and in some cases may be sufficiently distinctive that its maintenance is included as an instream management objective. For example, when the instream values of the Rangitata River were considered, in response to an application for a Water Conservation Order, the glacial regime of the river was noted as a feature of its natural and scientific value. The “Shotover backflow”, a regularly occurring feature of the flow regime of the Kawarau River, in which large floods entering the Kawarau from the Shotover cause water to flow *upstream* into Lake Wakatipu, was specifically protected by the Water Conservation Order for the Kawarau.

The way in which flow varies almost continuously in a river has received much attention in designing flow management regimes. Many people consider that flow variations are an essential element of the regime that should be maintained, and that long periods of constant flow (“flat-lining”), which could result from adherence to a minimum flow, should be avoided. A common management approach has been to establish “flow sharing”, to prevent abstraction from rapidly bringing residual river flows down to a particular minimum flow. There is no particular theoretical justification for any particular level of sharing, although 1:1 sharing between instream and out-of-stream uses is widely regarded as inherently equitable. However, flow sharing limits the amount of time that flows are at minimum values, and so effectively ameliorates the biological effect of the minimum flow. There is as yet no evidence, in New Zealand’s temperate hydro-climatic regime, that a slow recession is ecologically preferred to a period of stable flow, and it is questionable whether, in practice, most people can discern the difference between a slow recession and a constant low flow.

Determining the river flows required to maintain particular instream values may present significant challenges, particularly if there are several values that have different—or even opposite—requirements. Depending on specific proposals for use of the river—e.g., damming, large-scale run-of-river abstraction, minor abstractions—it may be necessary to develop what might be called a “designer flow regime”, that considers the need to maintain floods, freshes, low flows, and aspects of flow variability

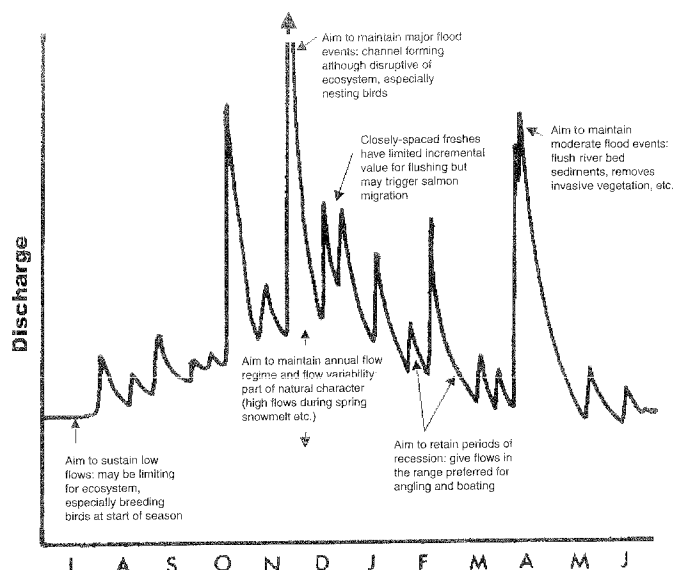


Figure 43.2 A conceptual “designer flow regime” for the Rangitata River, showing the particular aspects of instream flows that might need to be considered and sustained.

(Fig. 43.2). This, of course, means that the manager must have a clear idea of the outcomes that are desired, with regard to instream values.

Management objectives and levels of maintenance

A basic principle established in the *Flow guidelines* is that instream values and their requirements must be identified and appraised within the context of definite Instream Management Objectives (Chapter 48; Fig. 43.3). Resource management objectives have been defined by regional councils in their various regional policy statements, and an increasing number of councils are developing more specific objectives in regional or catchment water resources management plans. Where objectives have been developed consultatively to reflect community aspirations, they can be accorded appropriate weights, even though they might not be expressed in monetary terms. These objectives provide a reference point from which council officials, special tribunals, or the Environment Court can compare the merits of alternative uses of a given body of water, and in particular the extent to which instream values must be provided for.

An important choice is the level at which instream values are to be maintained. In areas that are short of water, resource managers have tended to allocate as much water as possible to purposes whose benefits can readily be expressed in monetary terms, and to set instream flow conditions that just maintained instream values at a minimal level. At the other end of the scale, Part IX (Water Conservation Orders) of the RM Act aims to recognise and sustain outstanding amenity and intrinsic values, and

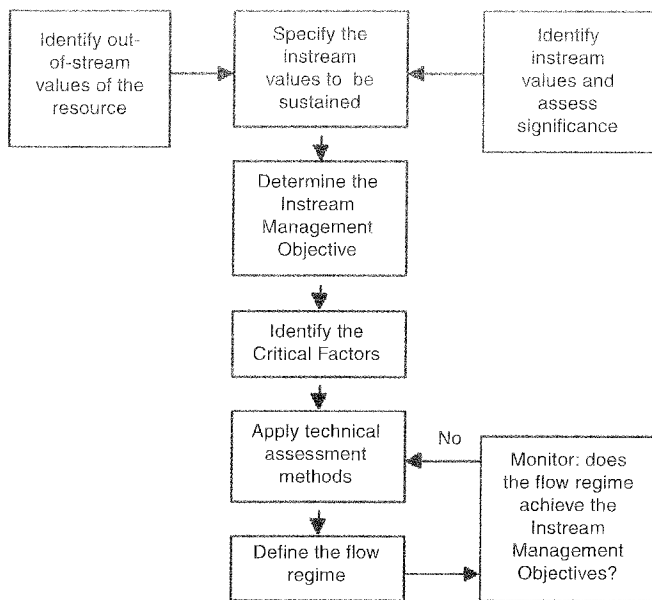


Figure 43.3 The process of setting objectives for management of instream flow regimes (from Ministry for the Environment 1998).

it seems reasonable to suppose that such values would be sustained as far as possible in an outstanding condition. In between, the *Flow guidelines* suggest that the level of maintenance should reflect the merits of instream values in a particular river—such as the quality of a recreational fishery, the biological diversity of a stream ecosystem, the conservation status of a breeding bird population on a river bed, the proximity to a large population centre of a kayaking river, or the availability of alternatives or means of mitigation. Determining the appropriate level of maintenance requires consultation with all interests likely to be affected by a particular plan or proposal.

Ecological and intrinsic values

An aquatic ecosystem includes a number of different types of biota that all must be considered if the ecosystem is to be sustained. The need to safeguard the “life-supporting capacity” of an ecosystem is included in Section 5 of the RM Act. This draws attention to the importance of the food chain in a river—aquatic and bordering terrestrial vegetation, insects and aquatic invertebrates, fish (native and introduced), riverbed birds, introduced mammalian predators and people. Even though most attention might focus, for example, on an endangered bird species or a trout fishery, the need to maintain the food chain and safeguard life-supporting capacity cannot be neglected.

There is a growing body of scientific information on the flow-related needs of aquatic biota, much of which is reported in the *New Zealand Journal of Marine and Freshwater Research* and summarised in other chapters of

this book. However, the ecological and intrinsic values in a particular river will need to be carefully researched. For example, the presence of areas of significant indigenous vegetation, significant wildlife habitats, or particular endangered species—whether plant, insect, invertebrate, fish or bird—may be known only to local naturalists, conservation staff, or scientists. Consultation with the Department of Conservation, Fish and Game New Zealand, NIWA, societies such as the NZ Ornithological Society, the regional council and local residents will be necessary.

The diverse ecological and instream values of a river have a variety of flow-related requirements, and these also may differ from river to river. The basic requirement might be expressed as a sufficient area of habitat for each life stage and activity of the species or species assemblages present, with appropriate:

- hydraulic attributes (water depth, water velocity, water turbulence),
- substrate (sediment-size distribution, mobility, packing, openness of interstitial spaces),
- cover (aquatic vegetation, overhanging banks and vegetation, woody debris, large bed sediment, deep),
- water quality (temperature, turbidity, dissolved oxygen concentration, dissolved constituents, nutrients), and
- food supply (in situ, drift).

A number of approaches have been developed, and are discussed more fully in a later section, that appraise the ability of aquatic habitat to support species and species assemblages (Jowett 1997). Much effort has been directed towards the aquatic habitat required for fish, especially sport fish, but the same methods can be applied to aquatic vegetation (Chapters 14, 15), macroinvertebrates (Chapter 16), and wetland birds (Chapter 18)—so long as their requirements are known. The Instream Flow Incremental Methodology (IFIM) is the most sophisticated and comprehensive method for analysing instream flow needs; as its name implies, it appraises the way in which available instream habitat varies with incremental changes in flow. The IFIM draws on observations of the frequency with which a given life stage or activity of a species is associated with particular combinations of habitat characteristics (water depth, substrate, cover types etc.), to infer the suitability of, or species’ preference for, particular values of those characteristics (Fig. 43.4). The procedure enables calculation of an index called Weighted Usable Area, which is a measure of the aggregate amount of habitat (usually expressed in square metres per metre of channel length) that is suitable for a particular species (see “Instream habitat modelling” below).

For bird, plant and insect species that inhabit or use the emergent parts of a riverbed, habitat requirements may be somewhat different to those conventionally dealt with by

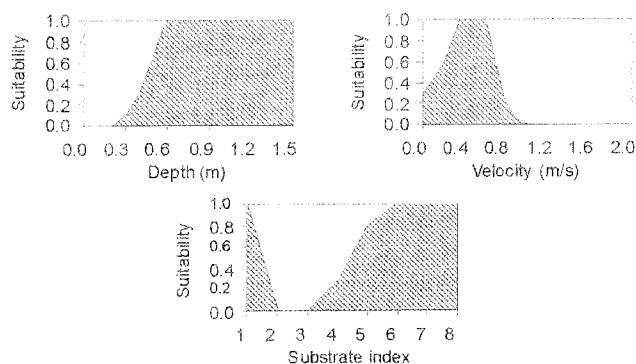


Figure 43.4 Habitat suitability curves for adult brown trout (adapted from Hayes and Jowett 1994).

instream flow methodologies, but they can be thought of in the same way. For example, the wrybill plover is an endangered wetland bird that nests and breeds on gravel river beds. Its habitat requirements for feeding include sufficient areas of water that have the depth, velocity and substrate conditions suitable for production of the macroinvertebrates on which it feeds. In addition, a critical habitat requirement is for extensive areas of unvegetated emergent gravel on which to nest. Colonisation of gravel surfaces by vegetation reduces the area available, and increases the risk of predation by animals that can more easily approach nests without being seen. The requirement for wrybill feeding habitat can be analysed using IFIM, as done for example by Duncan and Hicks (2001). Nesting habitat is not directly related to hydraulic conditions at the time of nesting (except when a flood destroys a nest), but depends on the periodic occurrence of floods that remove colonising vegetation, so that an instream flow requirement is to maintain the frequency and duration of such floods.

For some species, one of the instream flow needs that might need to be considered is the *absence* of some form of limitation or disturbance. The long-term viability and health of some species and ecosystems tends to be limited by disturbance events—floods and periods of low or zero flow—that reduce, perhaps only for a short time, the availability of and access to habitat (Chapter 13). This is not to say that a complete absence of floods and low flows is necessarily desirable. Clausen and Biggs (1997) concluded that invertebrate abundance and species richness appear to be highest with a moderate frequency of streambed disturbance, and Sagar (1983) concluded that frequent freshes increased rates of recolonisation after a major flood in the Rakaia River, by increasing the rate of drift.

A particular case of a disturbance event is river mouth closure, which can prevent passage of migratory fish during periods of low flow. Mouth closure is of particular concern

to salmon anglers in a number of South Island east coast rivers, because it reduces angling opportunities and may adversely affect spawning and recruitment to the population in subsequent years. It also may affect a number of native fish species, many of which spend part of their life cycle at sea and move to or from fresh water at certain times of year (see McDowall 1995 for details). Observations by local residents and anglers can often provide information on the flow and sea conditions under which a particular river mouth is likely to close or re-open; there is no theoretical way of predicting these events. When a river mouth is closed, fish may wait off-shore until it is re-opened by a flood or fresh, and then enter the river. In the Waipara River, floods in late winter and spring open the river mouth for only short periods of time, but this is sufficient to allow large numbers of native fish to migrate into the river (Jowett 2001). This particular instream flow need may therefore be met by both maintaining minimum flows above the level needed to maintain a surface flow over the bar, and/or by maintaining the frequency and ability of freshes to re-open a closed river mouth.

Water quality is one of the factors that should be included in a comprehensive IFIM analysis, although it may have a limiting rather than an incremental effect. Suren *et al.* (2003) found, for two Canterbury rivers, that nutrient enrichment can have an incremental impact on periphyton and macroinvertebrate communities—and therefore on instream fishery and amenity values—with higher biomass and an increase in “undesirable” filamentous green algae in the enriched river. Suren *et al.* suggested that degree of enrichment should be considered in assessing instream flow needs, and that an enriched river may require higher minimum flows than an unenriched river. Similarly, turbidity and suspended sediment concentration may have an incremental limiting effect on aquatic biota, both directly by affecting feeding and other behaviour of aquatic animals, and by sediment infiltrating the interstices of the streambed substrates and reducing their suitability as habitat for invertebrates and small fish (Jowett and Boustead 2001; Rowe *et al.* 2000).

Water temperatures are, to an extent, flow-related, and high temperatures may be limiting for aquatic animals in New Zealand rivers, particularly salmonid fish. Several species of native fish and invertebrates may tolerate temperatures in excess of 30°C, while, for the introduced salmonids, temperatures much above 20°C are approaching lethal levels, and they prefer markedly cooler water (Collier *et al.* 1995, Appendix 1).

Flow-related factors are not the only ones that influence instream ecological and intrinsic values. As mentioned earlier, the viability of populations of riverbed birds is very much affected by mammalian predation, as well as disturbance by people, livestock and vehicles, and

management of land use and vegetation in the river corridor. Year-to-year variations in the quality of a salmon fishery cannot readily be related to hydrology, and may be more influenced by sea conditions during the marine phase of the salmon life cycle.

Landscape, scenic and natural characteristics

Flowing water is an important element of the landscape along a river, and an analysis of instream flow needs should take account of landscape values. The discussion of hydraulic geometry in Chapter 8 showed how the appearance of a channel, considered in terms of the width, depth and velocity of the water body, varies with discharge, and that different types of channel respond in different ways to changing discharge (see also *Flow guidelines for instream values* - Ministry for the Environment, volume B, p. 139-144). Indeed, the nature of a channel is itself controlled, in the end, by the hydrologic regime—the sequence of flows—that it experiences, particularly the floods that are able to mobilise sediment and are responsible for reshaping the channel. However, to conclude that the physical appearance of a channel is related to the flows that it carries begs the question of how this relationship affects the visual or landscape value of the river. One might expect that fast, turbulent flow is more visually attractive than slow-moving, calm water, and a common view seems to be that, as discharge declines, a river loses the sense of movement and energy that it has at higher flows. On the other hand, Mosley's (1989) work revealed that members of the public value a placid stream like the Heathcote River, flowing quietly through the centre of Christchurch, as highly as the Arahura River, a fast-flowing, turbulent river flowing through a mountain valley in Westland. In other words, the relationship between landscape value and flow is very subjective, and not easy to define. To consider it in an appraisal of instream flow needs requires input from a landscape architect and—perhaps even more importantly—consultation with members of the interested public.

The *Flow guidelines* identify many aspects of landscape, scenic, and associated natural values, but few of them are actually flow-related. Mosley's (1989) investigation of public preference for river scenery indicated that attributes of the water body visible in a particular scene explain, in a statistical sense, only a small part of peoples' preferences, and most explanation is provided by aspects of the landscape around the river, such as bank vegetation and the presence of vistas. The water-related attributes were water colour, which is flow-related in that turbidity increases with flow but is strongly determined by conditions upstream in the catchment, and the relative area of water surface and bare gravel. This latter attribute is determined to a considerable extent by the overall form of

the river, which is controlled principally by large floods. It is also flow-related, because the area of water relative to gravel decreases as flow decreases, but the rate of change is small. It is questionable whether most non-specialists can perceive differences, or will respond adversely to them, unless they are so marked as to change the nature of the river (e.g., reduce a braided river to a single thread channel, or a meandering river to a discontinuous sequence of pools).

Nevertheless, procedures have been developed to appraise the effect on the riverscape of changes in flow. One that has been applied to several major rivers is to obtain the judgements of members of the public or of professional landscape architects on the landscape value of representative or particularly important reaches of river at a range of flows. This can be done by arranging repeat visits of the panel members, showing them photographs taken at a range of flows, or showing them computer or artist's simulations of how the appearance of the river would change with flow. An example of this approach is its application to the Waiiau River (Southland), as part of the assessment of the environmental effects of hydropower generation (see Ministry for the Environment 1998, p. 132-133). It should be re-emphasised that any relationship between flow and landscape value is subjectively assessed, so that the validity of the procedure relies either on the expertise and experience of professionals or on the representativeness of a sufficiently large group of laypersons (preferably 20 or more).

Amenity values – recreational angling and fishing

The value of a river as a recreational fishery is, of course, determined to a considerable extent by the ability of the river to support the particular fish species of interest, and the food chain on which it is based. However, the activity of fishing itself may have specific requirements that may be even more restrictive than those of the fish themselves. Some of the flow-related requirements for angling can be expressed in terms of depth, velocity and substrate measured in favoured angling locations, and analysed using IFIM procedures in the same way as for habitat suitability. Thus, for example, Jowett *et al.* (1996) used angling guides to identify rainbow trout "lies" in the Tongariro River, measured the hydraulic characteristics of the lies, and then used habitat modelling to predict that flows of 18–28 m³/s provided maximum rainbow trout angling habitat in the lower river.

The choice of a suitable angling site generally depends on rather more than simply considering depth, velocity and substrate, however. Practical matters, such as whether there is easy access to the riverbank or sufficient space for casting, are of importance, and are not directly flow-related. More importantly, anglers select locations which

experience tells them are likely to contain fish, such as deep pools along the foot of a river cliff, or eddies just downstream from a riffle, where the fish can wait in slack water for food to be carried past in the main current. For example, drift-feeding brown trout in three New Zealand rivers were found in depths of 0.67–0.86 m and mean column velocities of 0.38–0.48 m/s, and were usually associated with large substrate elements adjacent to areas with higher water velocity (Hayes and Jowett 1994). The location, number, and nature of such favoured sites can change with discharge, but not necessarily in a way that allows accurate prediction of trout locations. In most cases, pools change relatively little with flow, because their dimensions are largely controlled by the depth of water over the riffle at the downstream end, and this may change only a few centimetres as discharge changes.

Where angling is a particularly valued activity, field investigations at several flows may show the relationship between the availability of angling sites and flow. For example, Mark Webb (unpublished study by Central South Island Region, Fish & Game New Zealand) has used the results of a field study of angling “lies” by a panel of five expert anglers, at flows ranging between 60 and 130 m³/s, to support recommendations for instream flow needs in the Rangitata River. The two reaches were fully braided and partially braided, and the relationships between the number and area of angling lies and the flow in the reach were different in several respects (Table 43.1). Earlier work in this river drew on the opinions of expert anglers, and revealed that preferred conditions for angling are very dependent on the skills and expectations of individual anglers, as well as on many other factors that are unrelated to flow, such as cloudiness and other weather conditions that influence water temperature, fish feeding activity, and thus angling success (Davis *et al.* 1987). However, the transferability of angling habitat suitability from one river to another is doubtful. In the Tongariro River, field observations suggested that when trout were disturbed by

anglers, they select locations that are as far from anglers as possible, but not in water velocities that exceed their sustained swimming ability (Jowett *et al.* 1996). Thus, angling habitat in an undisturbed river is likely to be different to that in a high-use river. In general, a skilled and experienced angler is able to be successful over a wider range of flow conditions because of a greater knowledge of fish response and location in different flows, to the extent that flows that are unfishable for one angler may present a satisfying challenge to another.

Amenity values – boating and other recreational activities

The *New Zealand Recreational River Survey* (Egarr and Egarr 1981) provides an excellent starting point for appraising the recreational value of a particular river, but it is now more than twenty years old, and it does not specifically address the flow conditions required to sustain each activity. Changing hydraulic conditions strongly influence the suitability of a water body for many recreational activities, particularly those described as “water contact” activities and boating. Because people have a wide range of skills in their chosen activity and varying perceptions of the river environment, it is difficult to determine the hydraulic conditions that are optimal, minimal or maximal for a particular recreational activity. For example, a wide range of flows is cited by experienced kayakers as suitable for different activities and levels of skill (Table 43.2); experienced kayakers may themselves value flow variability, to provide different conditions and features on repeat trips to a particular stretch of water. The difficulty is compounded because there has been relatively little research into the instream flow needs for recreational activities in New Zealand, and we are still very much considering flow needs river by river.

Nevertheless, the conditions required for most water-based recreational activities have been tabulated (Mosley 1983; Ministry for the Environment 1998), and provide a starting point for consultation with users (Table 43.3). People are, in general, occasional and discretionary users of a river—they can be more adaptable to alternative flow regimes than the resident aquatic biota. Other approaches to managing a river can sustain recreational uses, in addition to flow regime management, such as (Ministry for the Environment 1998, p. A140):

- * providing periodic controlled flow releases at particular high-use times (summer weekends), for sporting events such as kayak slaloms (e.g., Mangahao River),

Table 43.1 Results of an expert evaluation of angling lies, Rangitata River (data from Central South Island Region, Fish and Game New Zealand).

	Fully braided reach		Semi-braided reach	
	7 June	13 July	6 June	12 July
Flow in reach (m ³ /s)	74	27	82	31
Total angling lies	32	24	14	19
Total area of lies (m ²)	13,700	6,900	7,100	11,400
No. of excellent lies	7	9	4	6
Area of excellent lies (m ²)	4,500	3,300	3,300	6,100
No. of good lies	14	9	5	6
Area of good lies (m ²)	5,200	2,500	2,200	3,100
Angling area (m ² /m)	2.28	1.15	1.41	2.29

Table 43.2 Required river flows for several types of recreational boating in the Rangitata River between Klondyke and Arundel (estimates provided by NZ Recreational Canoeing Association).

Activity	Suggested river flows (cumecs)					
	Minimum		Optimum		Maximum	
	Gauge ^a	River ^b	Gauge ^a	River ^b	Gauge ^a	River ^b
Sit On Kayaking	75	45-50	100	70-75	150	120-125
Kayak Instruction (Beginner)	60	30-35	75	45-50	90	65-70
Kayak Instruction (Intermediate)	75	45-50	110	70-75	150	120-125
Rafting	85	55-60	130	95-100	280	250

^a Gauge flow is at the Klondyke recorder, just upstream of the Rangitata Diversion Race intake.

^b River flows are the estimated actual residual flow in the lower Rangitata River, assuming 25-30 m³/s have been removed by the Rangitata Diversion Race.

Table 43.3 Instream conditions required to sustain recreational activities (modified from Mosley (1983) and Ministry for the Environment (1998)).

Activity	Water surface width (m), depth (m), velocity (m/s) requirements			Preferred sediment requirements	Preferred other requirements
	Minimum	Maximum	Preferred		
Paddling/wading (including angling while wading)	W – D – V –	W – D: 1.2 V: 1.8	W – D: 0.4-0.6 V: <0.5	Sand and gravel preferred. Algal or silt coating undesirable. No debris, broken glass, etc.	Bacteriological and toxicant water quality standards to be met. Water temperature 15-25°C (<19°C for salmonid angling) DxV product <1.0. Bottom visible. Easy access and sloping beach desirable. For angling, also fish habitat requirements.
Swimming	W: 5.0 D: 0.8 V –	W – D – V: 1.0	W: >10.0 D: 1.5 V: <0.3	As for paddling/wading.	As for paddling/wading. Length of channel useable >50 m. For diving from bank, D>2.0 m.
Tubing/drift diving	W: 5.0 D: 0.3 V –	W – D – V –	W – D: 0.8-1.5 V: 1.0-2.0	As for paddling/wading. For "white water" form of sport, as for rafting/canoeing.	No hazards – overhanging/submerged trees, etc. Bacteriological and toxicant water quality standards to be met. Water temperature 10-25°C. Bottom visible. Access at top and bottom of reach to be travelled. Class II or III on international scale (I or II for drift diving). Obstacles can be portaged. Slots between rocks >1.0 m.
White water rafting/canoeing	W: 7.5 D: 0.2 V –	W – D – V: 4.5	W: >20.0 D: 0.8-1.5 V: 1.0-3.0	Presence of large boulders and bedrock outcrops to provide interest. Sediment on riffles of gravel size and not angular to minimise wear and tear.	As for tubing/drift diving, except Class II to IV on international scale. Slots between rocks >2.0 m.
Tramping (riverbed routes)	W – D – V –	W – D: 1.2 V: 1.8	W – D – V –	Gravel bed desirable for easy travel. Algal or silt coating undesirable. Stable boulders, rock outcrops and small waterfalls desirable for interesting travel.	DxV product <1.0 on skewed gravel shoals for easy crossing, or footbridges available. River does not impinge on bluffs, to minimise need for river crossings. Floodplain or terrace surfaces present for easy travel. Water temperature >10°C. Bottom visible.
Angling (bank)	W – D – V –	W – D: bankfull V –	As for fish habitat preference	As for fish habitat preferences. No snags on stream bed.	As for fish habitat preferences, and Easy access to and along bank. Stable (non-caving) bank.
Boating (non-powered) rowing, flat water canoeing	W: 7.5 (20 rowing) D: 0.5 V –	W – D – V: 1.5	W: >20.0 D: 0.6-1.5 V: <0.5	Sand bed preferable. No snags on stream bed.	No snags in stream. Easy access to river. No hazards – weirs, etc.
Sailing	W: 30.0 D: 0.8 V –	W – D – V: 0.5	W: >60.0 D: >1.5 V: 0.0	As for boating (non-powered).	As for boating (non-powered).
Flat water power boating, water skiing	W: 30.0 D: 1.5 V –	W – D – V: 4.5	W: >90.0 D: >3.0 V: <1.5	As for boating (non-powered).	As for boating (non-powered).
Jetboating	W: 5.0 D: 0.1 V –	W – D – V: 4.5	W: >5.0 D: >0.6 V: <4.5	As for white water rafting.	Easy access to river. Minimum depth over riffles >0.2 m. No hazards – weirs, submerged piles, overhanging trees. Bottom visible.
Camping (for water supply and bathing/washing)	W: 0.5 D: 0.1 V –	W – D – V –	W – D – V –	As for paddling/wading.	As for paddling/wading.

- providing alternative activities on the same waterway (e.g., providing various boating and rowing facilities on Lake Ruataniwha, as a substitute for kayaking on the former Ohau River), and
- actively managing the river channel to permit recreation (e.g., digging swimming holes on the Ashley River near the town of Rangiora.)

Whether or not such approaches are acceptable to particular groups within the recreational river user community is obviously debatable, and there is a need for consultation when planning for water-based recreation.

Maori values

The RM Act (Sections 6(e), 7(a) and 8) makes specific reference to the need to consider Maori values in managing natural resources (Chapter 44). With regard to identifying instream flow regimes that sustain attributes of a river that are of particular interest to Maori, we should consider in particular mauri, mahinga kai and waahi tapu in the vicinity of the river:

- mauri: the “life force” or spiritual presence of a river, closely related to its life-supporting capacity, aesthetic quality, and degree of naturalness,
- mahinga kai: the aquatic resources, food and material, harvested by iwi, including fish, birds, flax and other vegetation, and
- waahi tapu: places or features that have particular significance to Maori people.

The relationship between these attributes and flow regime is not necessarily straightforward, so that consultation with the iwi, hapu and whanau that have an interest in a particular river is essential.

The concept of mauri is inherently integrative, so that many attributes of a river in addition to its flow must be considered. A study of the relationship between mauri and the flow regime of the Waipara River, for example, showed that evaluation by representatives of local Maori of the river’s mauri was influenced by factors such as the presence of birdlife, litter and rubbish, the naturalness of riparian vegetation, and structures along the river—factors that are related to the naturalness and apparent health of the river environment (Larking 2002). However, mauri was also perceived to change with flow, principally in response to the sound and movement of the water—indicative of a “living” and dynamic river, and the clarity and colour of the water—indicative of the river’s health, cleanliness and degree of human disturbance. In the Waipara study, mauri was perceived to vary seasonally, and to be poor or worse during low flows in summer. The summer flows that were observed were above the lowest flows to which the river falls, and Larking suggested that maintenance of mauri may require flows above the minimum flows necessary to maintain the aquatic ecosystem. The data from this study

suggest that the mean monthly 7-day low flow could provide a rule of thumb for the flow needed to sustain the attributes of the water—movement, sound, turbulence, etc.—that are essential components of mauri.

The instream flow regime (both water quantity and quality) required to sustain mahinga kai is likely to be closely related to that required to sustain ecological values and life-supporting capacity, and instream flow assessment procedures will be applicable. In fact, a cultural stream health index, based on Ngai Tahu perspectives about stream health in hill country rain-fed rivers, was significantly correlated with stream health as measured by the macroinvertebrate community index (MCI) and the stream health monitoring and assessment kit (SHMAK) score (Tipa and Teirney 2003). However, the species in a particular location may have distinctive habitat requirements and will require specific attention. For example, an important source of weaving materials may be dependent on the health of riparian wetlands and side streams, rather than on maintenance of flows in the main channel itself. The possibility should not be overlooked that mahinga kai have been degraded or lost as a result of past water resource or river management, and that suitable management of the flow regime, perhaps in conjunction with other remedial work, could re-establish them.

Waahi tapu can include a variety of human and natural features—burial sites, particular landscape features, pa sites, etc.—and the principal concern with regard to instream flow is whether they are affected in any way by management of the flow regime. Changes in instream flow could adversely affect a particular site, for instance by isolating it from flowing water for long periods and thereby changing its character, but it is perhaps more likely that inundation by a dam, erosion of the site as a result of altered flood flows, or modification by river control works could have an effect. Because waahi tapu are special features whose location often is known to only a few people, it is essential that Maori are consulted with regard to their existence and relationship to water resources management.

The RM Act (Section 7(a)) requires that particular regard shall be given to kaitiakitanga. This implies that, in assessing instream flow needs, Maori should be fully engaged in defining the values that exist, determining their relationship with flow regime, and setting flows that sustain those values. Chapter 13 of the *Flow Guidelines* provides a fuller outline of the process.

INSTREAM FLOW METHODS

A large number of methods have been used to determine flow requirements and new methods continue to be suggested; only a few are discussed here. The methods used to develop an appropriate minimum flow or flow regime

will depend on the case being considered and can vary from a quick rule-of-thumb assessment to detailed studies over a number of years. Even though some methods have been applied for more than thirty years, there is no universally accepted method for all rivers and streams, and there are very few cases studies of biological response to flow changes that can be used to judge the success or failure of different methods. Traditionally, instream flow methods have been used to define a minimum flow, below which no human influences should take place. However, the current trend is away from methods that set one "minimum flow" towards methods that consider the flow regime, with some degree of flow variability, needed to maintain the natural morphology and ecosystem.

Instream flow methods can be conveniently divided into three types: historic flow, hydraulic and habitat methods. The methods were described by Jowett (1997) and are summarised in the following sections.

Historic flow methods

These methods are based on historical flow records and are the simplest and easiest to apply. Stalnaker *et al.* (1995) describe this type of method as "standard setting", because they are generally rule-of-thumb methods that are used to set minimum flows. The minimum flow might be a proportion of the average flow, a percentile from the flow duration curve, or an annual minimum with a given exceedance probability. For example, a minimum could be specified as 80% of the mean annual low flow. The percentage is often referred to as the "level of maintenance".

The aim of historic flow methods is to maintain the flow within the historic flow range, or to avoid the flow regime deviating substantially from the natural flow regime. The underlying assumption is that the ecosystem has adjusted to the flow regime and that a reduction in flow will cause a reduction in the biological state (abundance, diversity, etc.) proportional to the reduction in flow; or in other words, that the biological response is proportional to flow (Fig. 43.5). It is usually also believed that the natural ecosystem will be only slightly affected as long as the changes in flow are limited and the stream maintains its natural character. It is implicitly assumed that the ecological state cannot improve by changing the natural flow regime.

The most well known historic flow method is the Tennant (1976) method, also known as the Montana method, which specifies that 10% of the average flow is the lower limit for aquatic life and 30% of the average flow provides a satisfactory stream environment. The Tennant method was based on hydraulic data from eleven U.S. streams (including streams in Montana) and an

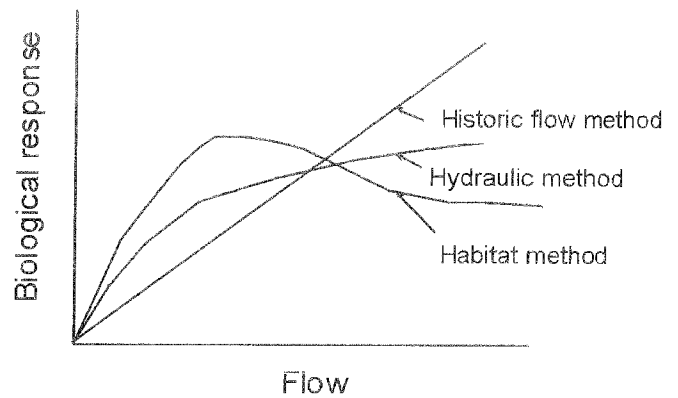


Figure 43.5 Hypothetical relationships between assumed biological responses to flow for the historic flow, hydraulic and habitat methods. The biological response is assumed to be proportional to the flow, the wetted perimeter or width, and the weighted usable area, for the historic flow method, the hydraulic method, and the habitat method, respectively.

assessment of the depths and velocities needed for sustaining the aquatic life. At 10% of average flow, he found that the average depth was 0.3 m and velocity 0.25 m/s, and considered these to be lower limits for aquatic life. He found that 30% of average flow or higher provided average depths of 0.45–0.6 m and velocities of 0.45–0.6 m/s and considered these to be in the good to optimum range for aquatic organisms. In New Zealand, Fraser (1978) suggested that the Tennant method could be extended to incorporate seasonal variation by specifying monthly minimum flows as a percentage of monthly mean flows.

Historic flows can also be used to define an "ecologically acceptable flow regime". Arthington *et al.* (1992) describe a "holistic method" that considers not only the magnitude of low flows, but also the timing, duration and frequency of high flows. Such a flow regime would not only sustain biota during extreme droughts, but would also provide high flows and the flow variability needed to maintain the diversity of the ecosystem. The building block method (King *et al.* 2000) is a similar approach. The range of variability approach, and the associated indicators of hydrologic alteration, identify an appropriate range of variation, usually one standard deviation, in a set of 32 hydrologic parameters derived from the "natural" flow record (Richter *et al.* 1997). The holistic, building block, and range of variability methods are conservative and maintain the ecosystem by retaining the key elements of the natural flow regime. These are "low risk" approaches aimed at maintaining an ecosystem in its existing state and preclude the possibility that a riverine ecosystem can be enhanced by other than a natural flow regime.

Hydraulic geometry and channel mapping methods

Hydraulic methods are more time consuming, in that they are based on measurements of hydraulic data (wetted perimeter, width, depth or velocity) from one or several cross-sections in the stream. The aim of hydraulic methods is to maximise food production by keeping adequate amounts of the food-producing area below water. Because the streambed is considered to be the most important area for food production (algae and invertebrates), it is usually the wetted perimeter or the width that is used as the hydraulic parameter.

The variation of the hydraulic parameter with flow can be found from carrying out measurements at different flows, or from calculations based on rating curves or Manning's equation. The graph of the hydraulic parameter versus flow (Fig. 43.5) is used for prescribing recommended flows, or to specify a minimum flow. The minimum flow can be defined as the flow in which the hydraulic parameter has dropped to a certain percentage of its value at mean flow, or the flow at which the hydraulic parameter starts to decline sharply towards zero (the curve's "inflection point" or, more correctly, a breakpoint). If the wetted perimeter or width is used, the inflection point is usually the point at which the water covers just the channel base. However, wetting of the channel base might not be enough to fulfil the requirements of depth and velocity for some species.

Gippel and Stewardson (1998) suggest an objective method for defining a breakpoint in wetted perimeter (P)/flow (Q) relationships that could be very useful for maintaining consistency in flow assessments between rivers. They suggested the breakpoint could be selected either as the point of maximum curvature or the point where the slope (dP/dQ) was 1, after first normalising the wetted perimeter and flow by dividing by their respective values at an index flow, such as the median flow.

Habitat methods

Habitat methods are the most advanced and are an extension of the hydraulic methods. Their great strength is that they quantify the loss of habitat caused by changes in the natural flow regime, which helps the evaluation of alternative flow proposals. According to a review by the Environment Agency in the UK on river flow objectives, "Internationally, an IFIM-type approach is considered the most defensible method in existence" (Dunbar *et al.* 1998). The Freshwater Research Institute of the University of Cape Town in South Africa states that "IFIM is currently considered to be the most sophisticated, and scientifically and legally defensible methodology available for quantitatively assessing the instream flow requirements of rivers" (Tharme 1996). A review of flow assessment

methods in the book "Instream flows for riverine resource stewardship" (Annear *et al.* 2002) described IFIM as the "most appropriate for relative comparisons of habitat potential from among several alternative flow management proposals" and as "the method of choice when a stream is subject to significant regulation and the resource management objective is to protect the existing healthy instream resources by prescribing conditions necessary for no net loss of physical habitat".

Computer models for the evaluation of physical habitat, water temperature, and sediment processes have been developed (e.g., PHABSIM—Milhous *et al.* 1989; RHYHABSIM—Jowett 1989). More recently, individual-based fish models (Railsback and Dixon 2003) and models based on energetic concepts (Hayes *et al.* 2003) have been developed to the stage where they could be used for flow assessment. Although increasingly complex hydraulic models from the one-dimensional (e.g., RHYHABSIM), to the two-dimensional (e.g., River2D: www.river2d.ualberta.ca) to the three-dimensional (e.g., SSIIM: www.bygg.ntnu.no/~nilsol/ssiimwin) can be used to model the physical environment, our knowledge of the relationships between complex flow patterns and biological requirements is very limited. However, in any modelling, the quality of the results will depend on the quality of the field work and calibration. This is especially true of two- and three-dimensional models, where the accuracy of the topographic model has a major effect on the accuracy of depth and velocity predictions. Two-dimensional models are likely to predict changes in velocity distribution more accurately than one-dimensional models, although in both cases predicted depths and velocities will be incorrect if water surface levels are not modelled accurately. In gravel-bed rivers, the accuracy of velocity prediction using a two-dimensional model (Duncan and Hicks 2001) and a one-dimensional model (Mosley and Jowett 1998) were similar. In the Ashley River, Mosley and Jowett (1998) predicted depths within ± 0.03 m and velocities with an average absolute error of about 0.15 m/s at flows ranging from 14.4 m³/s to 0.083 m³/s. Duncan and Hicks (2001) predicted depths and velocities in the Rangitata River with average absolute errors of 0.063 m and 0.18 m/s, respectively.

The aim of habitat-based methods is to maintain, or even improve, the physical habitat for instream values, or to avoid limitations of physical habitat. They require detailed hydraulic data, as well as knowledge of the ecosystem and the physical requirements of stream biota. The basic premise of habitat methods is that if there is no suitable physical habitat for the given species, then they cannot exist. However, if there is physical habitat available for a given species, then that species may or may not be present, depending on other factors not directly related to

flow. In other words, habitat methods can be used to set the "outer envelope" of suitable living conditions for the target biota.

Biological information is supplied in terms of habitat suitability curves for a particular species and life stage (e.g., adult brown trout, Fig. 43.4). A suitability value is a quantification of how well suited a given depth, velocity or substrate is for the particular species and life stage. The result of an instream habitat analysis is strongly influenced by the habitat criteria that are used. If these criteria specify deep water and high velocity requirements, maximum habitat will be provided by a relatively high flow. Conversely, if the habitat requirements specify shallow water and low velocities, maximum habitat will be provided by a relatively low flow and habitat will decrease as the flow increases. The suitability curves in Figure 43.4 were developed for New Zealand adult brown trout (Hayes and Jowett 1994) and specify higher depth and velocities than curves for adult brown trout developed in the U.S. (Raleigh *et al.* 1984). Whether this difference is due to different sizes of fish has not been clarified. However, it is clearly important to use suitability curves that are appropriate to the river and were developed for the same size and life stage of fish to which they are applied.

Habitat criteria have more influence on flow assessments than any other aspect of the analysis. Failure to use appropriate criteria can result in inappropriate flow assessments. Therefore, habitat criteria need to consider all life stages, including suitability criteria for the production of food for those life stages, so a good understanding of the species' life cycles and food requirements is needed.

The analysis can be separated into a hydraulic component and a habitat component. The hydraulic analysis (Figure 43.1 lists some of the available techniques) predicts velocity and depth for a given flow for each point, represented as a cell in a grid covering the stream area under consideration. In addition, information on bed substrate and other relevant factors such as shade, weeds and temperature, can be recorded for each cell.

The habitat analysis starts by choosing a particular species and life stage, and a particular flow. For each cell in the grid, velocity, depth, substrate, and possibly other parameters (e.g., shade) at the given flow are converted into suitability values, one for each parameter. These suitability values can then be combined (usually they are multiplied) and multiplied by the cell area to give an area of usable habitat (also called weighted usable area). Finally, all the usable habitat cell areas can be summed to give a total habitat area (total weighted usable area) for the reach at the given flow. Although weighted usable area is often interpreted as the area of usable habitat, it only truly represents the area of suitable habitat when binary habitat

suitability curves are used (i.e., habitat variables are either suitable (1) or unsuitable (0)). This whole procedure is then repeated for other flows until eventually the final outcome has been produced: a graph of usable habitat area versus flow for the given species. This graph has a typical shape, as shown in Figure 43.6, with a rising part, a maximum and a decline. The decline occurs when the velocity and/or depth exceed those preferred by the given species and life stage. Thus, in large rivers, the curve may predict that physical habitat will be at a maximum at flows less than those naturally occurring. Thus, in contrast to the historic flow method, the habitat method does not automatically assume that the natural flow regime is optimal for all aquatic species in a river.

The relationship between habitat and flow (Fig. 43.6) can be used to define a preferred flow range, a minimum flow, or a preferred maximum flow. As with hydraulic methods, the minimum flow can be defined as the inflection point or as the flow at which the habitat has dropped to a certain percentage of its value at mean or median flow. It can also be defined as the flow that has the lowest acceptable minimum amount of habitat in absolute terms. If minimum flows are at or above the habitat maximum for a particular species or instream use, the area of habitat available to that species will be less than maximum for most of the time. Often this does not matter, because the rate of change in habitat with flow is less at high flow than at low flow (Fig. 43.6) and the difference between maximum habitat and the amount of habitat at a high flow is relatively small. For example, most New Zealand native fish are found in shallow water along the edges of large rivers (Jowett and Richardson 1995) and there is usually some edge habitat available over a large range of flows. However, if maximum habitat for all species

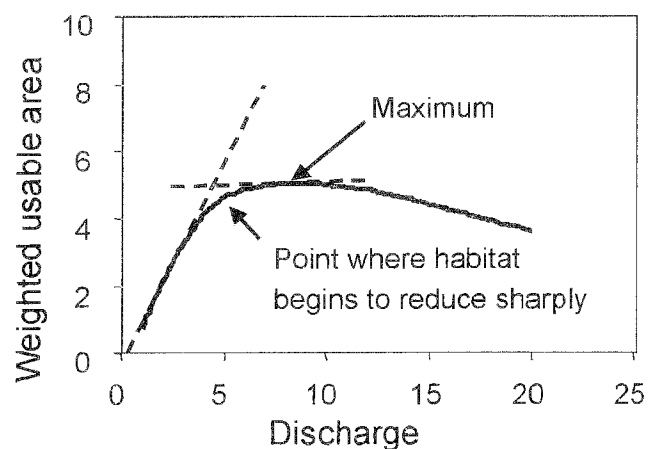


Figure 43.6 Selection of minimum flow at the point where habitat begins to decline sharply with decreasing flow for a hypothetical river.

and instream uses is less than the minimum flow, it suggests that a reduction in flow might enhance those values.

Habitat suitability curves have been developed for common fish and invertebrate species, threatened species (e.g., blue duck; Collier and Wakelin 1995), and recreational activities (Mosley 1983). When many fish species and life stages are present in a river, there are usually conflicting flow requirements. For example, young trout are found in water with low velocities, and adult trout are found in deep water with higher velocities. If the river has a large natural variation, with pools, runs and riffles, some of the different requirements may be provided for. Still, even in these rivers, and especially in rivers with small habitat variation, one species may benefit greatly from a reduction in depth and velocity, whereas habitat for another species will be reduced. If a river is to provide both rearing and adult trout habitat, there must be a compromise. One such compromise is to vary flows with the seasonal life stage requirements of spawning, rearing, and adult habitat, with the optimum flow gradually increasing as the fish grow and their food and velocity requirements increase. Biological flow requirements may be less in winter than summer because metabolic rates and food requirements reduce with water temperature. If flow requirements of individual species are different, a solution may be found by choosing one with intermediate requirements (Jowett and Richardson 1995) or to define flow requirements for aquatic communities.

Because little is known about the effect of short-term flow fluctuations on habitat, it is difficult to integrate this into habitat models. However, the potential habitat loss for macrophytes and benthic invertebrates, and thus fish food production, caused by flow fluctuations can be modelled using habitat methods. Because macrophytes and most benthic invertebrates are immobile, the locations within the river that remain as suitable physical habitat under fluctuating flows can be identified (Fig. 43.7). This analysis assumes that macrophyte habitat is suitable at a location only when it is suitable at that location during all flows that persist over the life of the organism. As the flow changes from the highest flow to lowest, some suitable locations at high flow become unsuitable at lower flows, while other locations remain suitable habitat, so that the area of suitable habitat over the full range of the fluctuation reduces as the magnitude of fluctuation increases.

Habitat methods and water quality models can be integrated, although usually the results of hydraulic models are transferred into water quality models. For example, water temperature models use water depth and velocity to model how water temperature varies with distance downstream. The integration of stream geometry and

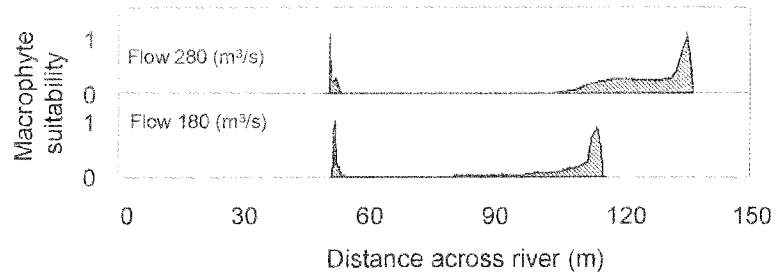


Figure 43.7 Habitat suitability for macrophytes (shown as hatched areas) across one cross-section in the Waikato River for flows of 280 m³/s and 180 m³/s. Note the loss of macrophyte habitat on the far left and far right of the cross-section as the flow reduces from 280 m³/s to 180 m³/s.

water temperature, dissolved oxygen and ammonia models has been implemented in the decision support system WAIORA (Jowett 1999).

Regional methods

Tennant's (1976) method is a good example of a regional method that combines the best features of historic flow methods and habitat methods, resulting in a biologically defensible method of minimum flow assessment. Regional methods could be developed for regions that are hydrologically and morphologically similar, with criteria that apply to trout, native fish, stream insects, or periphyton. By analysing habitat variation with flow for rivers within a region or even using expert opinion, it is possible to determine the level of flow as a proportion of median or mean annual low flow that maintains adequate or optimum conditions for various "target" communities. Variation in levels of maintenance could be achieved by assessing requirements for optimum habitat and minimum habitat, as in the Tennant method. Application of the method would involve selecting an appropriate target community and level of maintenance for the river in question and then applying a formula based on the proportion of natural flow, either recorded or estimated.

Environment Bay of Plenty established regional methods for assessments of minimum flows that "sustain aquatic life" in hydrologically homogeneous areas (Wilding 2002). They used a consistent method of determining minimum flow requirements in each stream with flow standards of 15%, 85% or 100% habitat retention, depending on the perceived value of the species. Their approach was as follows.

1. Identify a preferred flow for each fish species present in the river. This is either the flow where habitat is greatest, or, if maximum habitat occurs above the stream's median flow, the mean annual low flow is taken as the preferred flow.

2. Adjust the level of maintenance according to ecosystem significance. The minimum area of available habitat of the least "valued" species is 15% of that offered by the preferred flow, while the minimum area of habitat for the most "valued" species is 100% of that available at the preferred flow.
3. Examine flows that provide the minimum area of habitat for each species present: the highest is chosen as the minimum flow for the stream.
4. Compare minimum flow assessments to the 5-year 7-day low flows of each stream and derive a general relationship between minimum flow and 5-year 7-day low flow.

The benefit of regional methods over historic flow methods is that they can have explicit environmental goals, making water management more transparent. Thus, regional methods can be established as biologically defensible, and discussion and consultation can focus on whether the "target" and standards of maintenance are appropriate.

The rationale for habitat-based regional methods is primarily that of habitat methods. For a region, it is possible to develop formulae that predict when hydraulic conditions are optimum or become limiting for a range of aquatic species. For instance, most native fish are small stream species. Few are found in swift, deep water. In contrast, adult trout are rarely found in water less than about 0.4 m deep. Stream insects are most abundant in shallow swift habitats.

It is also possible to generalise velocity and depth criteria as levels of protection within a region. For instance, average velocities of less than 0.1 m/s might be considered poor, 0.1–0.3 m/s adequate, and 0.3–0.5 m/s optimum. Similarly average depths greater than 0.15 m might be considered suitable for native fish, and depths greater than 0.4 m suitable for adult trout.

These methods are potentially useful in that they combine the best features of habitat and flow methods and are likely to result in flow assessments that provide life-sustaining flows whilst retaining some degree of the river's "character".

Lamouroux and Capra (2002) demonstrated, using data from 58 streams in France, that the shape of habitat/flow relationships for three trout life stages and five other fish species could be generalised with reasonable accuracy using a dimensionless hydraulic variable, discharge per unit width. More recently, application of the same analysis to data from over 100 New Zealand streams also showed consistent relationships between habitat and discharge per unit width. Such relationships should facilitate more cost-effective habitat studies in many streams, at basin or larger scales, and enhance the biological validation of habitat model predictions.

CASE STUDIES OF BIOLOGICAL RESPONSE TO FLOW CHANGE

Although methods of assessing flow requirements continue to be developed and debated, there are very few studies that examine how well modified flow regimes have achieved their desired outcomes. Armour and Taylor (1991) surveyed 35 U.S. Fish and Wildlife field offices that had been involved in 616 IFIM applications, of which 6 had follow-up monitoring, the results of which were not reported. The survey found that opinions on IFIM were divided, with 40% considering the method technically too simplistic, 41% considering it too complex to apply, and 9% considering it not acceptable or biased. However, with any flow assessment method, the critical test is whether it is successful in achieving the desired outcome. In the U.S. survey, half rated success as higher than neutral, while one-third rated it lower.

Minimum flow requirements in the following cases were usually made on the point of inflection determined as shown in Figure 43.6. A horizontal line is drawn through the maximum and another line is extended through the low-flow section of the curve. Where the two lines meet is the point where habitat begins to reduce sharply with flow.

The first study of instream habitat carried out in New Zealand was in the Tekapo River (Jowett 1982), where diversion of flow for the Waitaki Power Development had reduced the flow in the river from about 80 m³/s to 12 m³/s. The habitat analysis showed that a flow of about 12 m³/s provided maximum trout spawning and food-producing habitat for that river. Prior to diversion, the Tekapo River was hardly recognised as an angling river and is not mentioned in angling surveys of that time (Allan and Cunningham 1957). Now it is one of the most popular rivers in the region, with angler use of 2400 days in 1994/95 and 4900 days in 2001/02 (Table 41.1), and high densities of brown and rainbow trout (Fig. 43.8). The diversion of turbid Lake Tekapo water increased water clarity in the river, and increased the variability of flow relative to mean flow because the source of water is unregulated tributary streams. Naturally occurring flows in the Tekapo River were probably too swift to support a good trout fishery, but the diverted turbid water now flows at a lower gradient through the Tekapo Canal, which is now the sixteenth most popular angling "river" in the country (Table 41.1).

Practically all of the natural flow of about 450 m³/s of the Waiau River in Southland was diverted through the Manapouri Power Station between 1977 and 1995. Instream habitat analysis (Jowett 1993) indicated that a flow of 12 m³/s or greater would provide excellent brown trout habitat, and a minimum flow regime of 12 m³/s in winter and 16 m³/s in summer was consequently implemented. Fish numbers increased (Fig. 43.9) and the

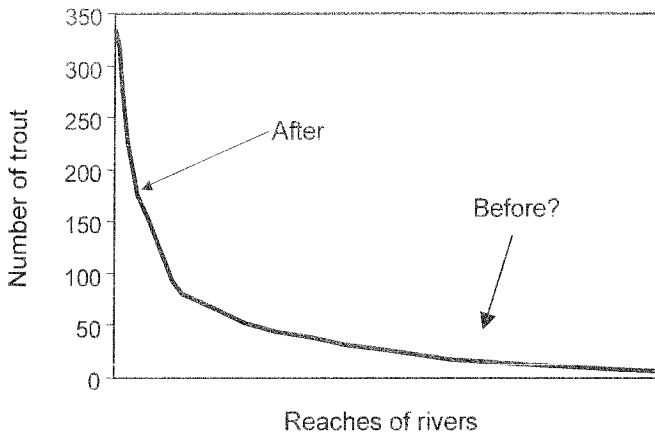


Figure 43.8 Total numbers of brown and rainbow trout (> 20 cm) per km in the Tekapo River before and after a reduction in flow from about 80 m³/s to about 12 m³/s, compared to data (arranged in descending order of trout numbers along the x axis) from 300+ river reaches.

fishery in this river is now regarded as excellent, with good numbers of trout and anglers, and high catch rates. In the 1994/95 fishing season, just before the implementation of the minimum flow, angler usage was 7700 days and this increased to 14600 angler-days in 2001/02 (Table 41.1), against the national trend of an overall decline in river fishing. The Waiau is now the eighth most popular trout fishing river in New Zealand, as it was before diversion (Allan and Cunningham 1957), and the present trout densities in the Waiau River rank it as one of the top rivers in New Zealand (Fig. 43.10).

The Tongariro Power Development diverts the natural flow of the Moawhango River, about 9.6 m³/s, to the Tongariro River, leaving practically no flow in the Moawhango River below the dam. As part of the Tongariro Power Development consent process, a minimum residual flow was suggested to re-establish benthic invertebrate communities below the dam (Jowett and Biggs 2000). In 2002, after about a year with a minimum residual flow in the Moawhango River, the composition of the invertebrate community had changed considerably, with the proportion of clean-water caddisflies doubling from 23% in 1997 to 44% in 2002, and the proportion of the less desirable elmid beetle decreasing from about 25% to 9% (Fig. 43.11). In particular, the proportion of the invertebrate community composed of mayflies + stoneflies + caddisflies (%EPT: a measure of the relative abundance of “healthy” invertebrates) increased from 37% to 57% with the residual flow, to the extent that it is now similar to the 60% EPT composition in the river upstream of the dam. In all cases, the relative abundance of “desirable” taxa increased with the increase in flow, and in most cases the relative abundance of “undesirable” taxa decreased.

The only case where predictions of trout abundance based on the quality of habitat were not successful is in the

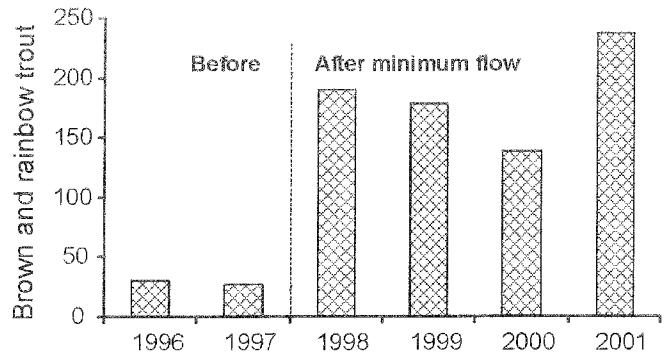


Figure 43.9 Density of brown and rainbow trout (> 20 cm) in the Waiau River per km before and after the implementation of a minimum flow regime of 12 m³/s in winter and 16 m³/s in summer.

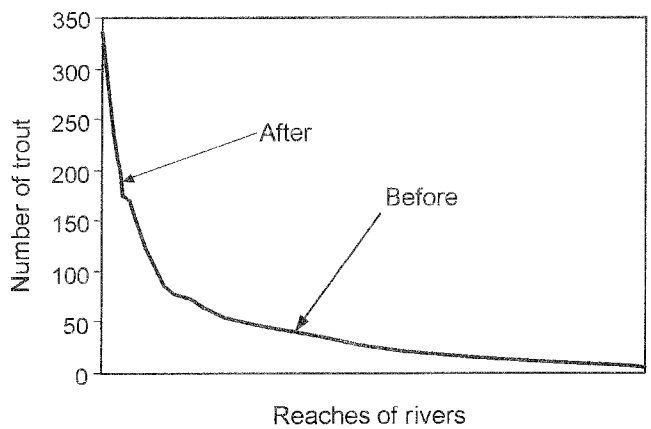


Figure 43.10 Number of brown and rainbow trout per km in the Waiau River before and after the implementation of a minimum flow, compared to data (arranged in descending order of trout numbers along the x axis) from 300+ river reaches.

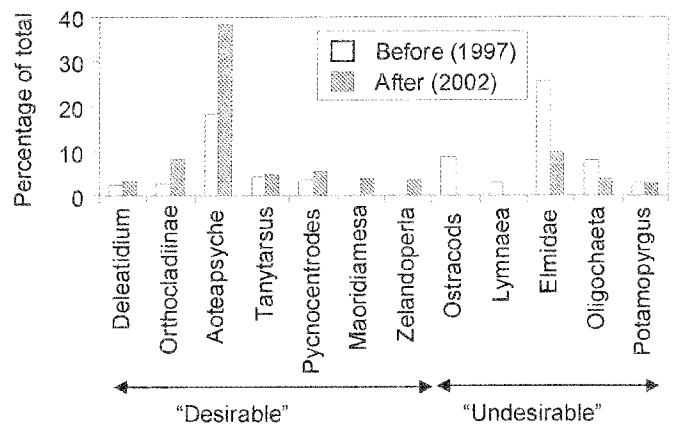


Figure 43.11 Benthic invertebrate species composition in Moawhango River before the minimum flow was implemented (open) and after (hatched) showing how the dominance of “desirable” invertebrate species (left) increased and “undesirable” invertebrate species (right) decreased as a result of a flow increase.

Ohau River in the South Island. The mean flow in the Ohau River was $80 \text{ m}^3/\text{s}$ prior to diversion in 1979, when less than $1 \text{ m}^3/\text{s}$ was left in the river. An instream habitat survey (James *et al.* 1992) showed that a flow of $10 \text{ m}^3/\text{s}$ would provide excellent trout habitat and this flow has been released at the lake outlet since 1994. Although the river now provides what is regarded as excellent angling water and trout habitat, trout numbers and angler usage have remained low, with 636 anglers-days in 1994/95 (Unwin and Brown 1998) and 500 angler-days in 2001/02. There is no mention of angling use in the Ohau river in an early angling diary scheme (Allan and Cunningham 1957), but in terms of crop, Graynoth and Skrzynski (1973) ranked it fifth of seven angling rivers in the Waitaki Valley district. The Ohau River begins at Lake Ohau, with a low dam and structure to release water, and flows for a short distance before it enters the artificially created Lake Ruataniwha. The lack of trout in the river may be related to poor food production because of glacial silt deposits on the substrate and lack of flow variation, and/or problems with recruitment and fish passage between the lake and river, or simply a preference for the lake environments.

An instream habitat study of the Waipara River, a small gravel-bed river north of Christchurch, was used to recommend a minimum flow of 120 L/s for native fish (Jowett 1994). This was based on consideration of habitat for common bullies, a species with habitat preferences that were intermediate between the fast-water species

(torrentfish and bluegill bullies) and the edge-dwelling species (upland bullies), as suggested by Jowett and Richardson (1995). Subsequently, fish populations were measured for three years. These surveys showed that the effect of low flows on fish populations increased with the magnitude and duration of low flow. In the first summer (1998/99 mean flow 1190 L/s), daily mean flows were less than 120 L/s for 31% of the time and fell to 32 L/s, with a substantial decline in abundance of three of the four common native fish species in the river. The following summer (1999/2000 mean flow 1243 L/s), daily mean flows were less than 120 L/s for 10% of the time and fell to 69 L/s, with little change in native fish abundance (Fig. 43.12). In the third year, flows were less than 120 L/s for 61% of the time and fell to 47 L/s, with fish numbers reducing for two of the four common species. Thus, minimum flows substantially less than half the recommended minimum flow did not have any detrimental effect on fish numbers, and the minimum flow recommendations for these native fish species may have been unnecessarily high.

Comparison and applicability of methods

Two schools of thought on flow assessment methods have gained prominence over the last 10 years. One is deterministic—practitioners believe that they can determine flow requirements from the physical characteristics of the river and a knowledge of habitat requirements, energy requirements, and sediment trans-

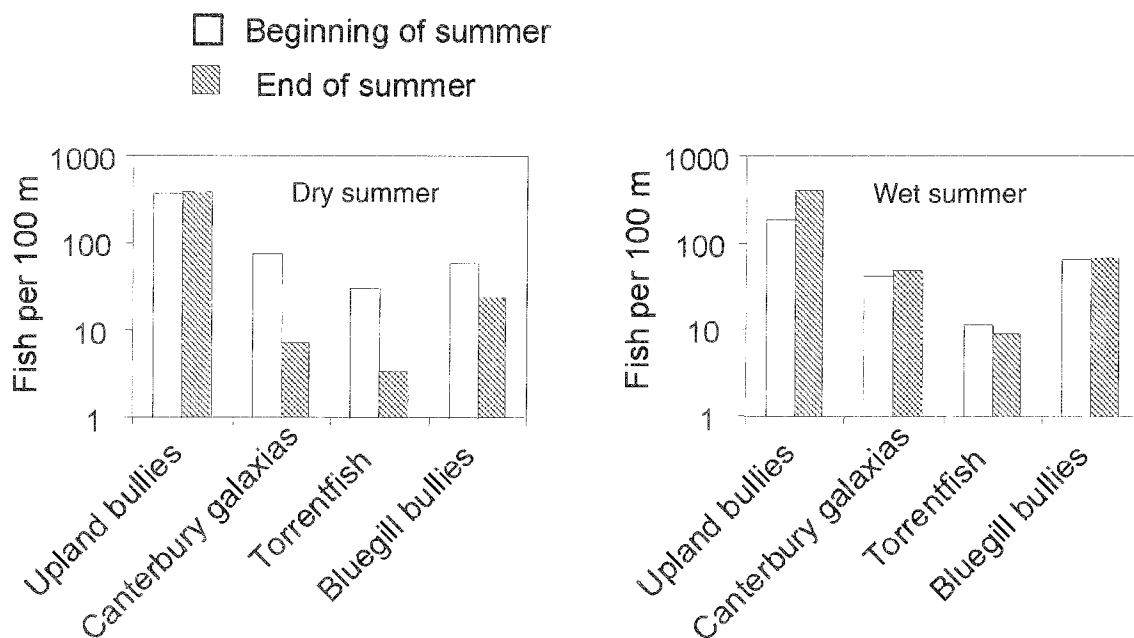


Figure 43.12 Native fish abundance in the Waipara River at the beginning and end of a dry summer (left) and wet summer (right).

port and morphological processes. The alternative approach is based on the “natural flow paradigm”, where the aim is to preserve the main characteristics of the natural flow regime along with the environmental “status quo” in terms of natural character and aquatic communities. Similarly, rule-of-thumb minimum flow assessments that maintain a commonly occurring minimum flow and do not significantly alter other aspects of the flow regime will also maintain all the natural characteristics. It is a relatively simple task to select a flow regime that mimics the natural flow regime and thus maintains the natural character and aquatic community. It is much more difficult to recommend a flow regime (the “designer” river approach) that maintains or enhances some natural aspects of the river, yet allows maximum use of water for “non-natural” purposes.

The idea of “adaptive management” has been proposed (Castleberry *et al.* 1996) when managers are faced with uncertainties and a basic philosophical conflicts between methods. This process involves trying a flow regime to see whether it works—if it doesn't you change the regime and try again. To work, adaptive management needs well designed monitoring systems, and a procedure that leads to an effective revision of the flow regime. U.S. experiences with adaptive management have been disappointing. In practice, it has been very expensive and interest has been difficult to maintain after a year or so. This is not surprising. Over the past 50 years or so there have been countless opportunities to examine the effectiveness of instream flow recommendations and to learn from experience. However, this has not happened, presumably because of practical difficulties in monitoring aquatic populations, and difficulties in associating population changes with flow regime changes, given the variability of natural flows and the non-flow-related factors that also influence aquatic populations, as well as lack of interest once a flow issue is settled.

In New Zealand cases where biological outcomes are known, flow assessments based on instream habitat have generally proved successful in achieving predicted biological responses. The case studies cited above show that good trout fisheries and benthic invertebrate communities can be supported by flow regimes that are markedly different to the natural flow regimes of the rivers. Obviously, a change to the flow regime of a river will change some aspects of that river's character and ecology. Application of “natural flow paradigm” methods will certainly preserve character and aquatic biota, but will limit the availability of water for other uses.

It is worth re-emphasising the need to set explicit objectives for management of river and stream flow regimes that will realise the community's aspirations for both instream and out-of-stream uses of water, and achieve the purposes of the Resource Management Act.

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Chapter 42

Values and uses of water

Christina Robb and John Bright

INTRODUCTION

New Zealand's freshwater systems provide a variety of environmental, social, cultural and economic benefits. Some of these arise from ecological and intrinsic characteristics of freshwater that exist whether or not there are people around to appreciate those characteristics. Some values require that the quality or quantity of water be modified in some way. In between are a set of values that are derived from the use of freshwater systems in their natural state. This chapter is about the set of values derived from human use of freshwater systems.

Throughout the chapter we use the following definitions:

- Instream values and uses—those values and uses that require the water in its natural state (includes values associated with groundwater systems).
- Development values and uses—those uses that require the modification of water flows by the damming, diversion or taking of water, and the economic, social, cultural and environmental values that arise from that use.
- Abstractive values and uses—those that require the removal of water from a freshwater system. Abstractive use covers water taken for drinking, urban and community water supplies, industrial uses and irrigation, but not water used for generating electricity. Abstractive values and uses are a subset of development values and uses.

Decision-makers must determine which combinations of values provide the greatest net benefit to the community, while taking into account those that can be complementary or competing. Such decisions are not easy, and are made more difficult where there is a lack of good information on how values are affected by modifying the quantity and quality of freshwater. Information on the values and uses of water, both now and in the future, is an essential element of water management, and will increase in importance as the demand for water increases.

This chapter provides an overview of the values derived from use of New Zealand's freshwater systems. It includes information on the amount of water needed for each value or use and, where possible, a breakdown by region or by type of water resource. We have assessed the likely increase in demands for water, and identified potential conflicts likely to arise if increasing demands are not managed. A final section looks at approaches to comparing values and uses by assigning direct and indirect monetary values.

OVERVIEW OF VALUES AND USES OF FRESHWATER

Ecosystem services

Ecosystems are complex combinations of natural assets—soil, plants, animals, air and water—that provide many “services” from which humans benefit (CSIRO 2003). Some examples of these services include:

- the provision of clean water,
- the maintenance of liveable climates and unpolluted atmospheres,
- the pollination of crops and native vegetation,
- the fulfilment of cultural, spiritual and intellectual needs, and
- the provision of options for the future, for example, though the maintenance of biodiversity.

The concept of ecosystem services has arisen from concerns that the fundamental importance of these natural services in sustaining human communities and economies is not adequately recognised by government policies or individual actions. Ecosystem services can be unknowingly lost or degraded because the complexity and interconnectedness of ecosystems makes them difficult to understand or predict, and virtually impossible to replicate. Recent work on ecosystem services seeks to ensure these important values are recognised in economic evaluations

of development and restoration proposals (Murtough *et al.* 2002).

Ecosystem services are associated with all freshwater systems. Much of this book examines surface water systems, however, it is important to recognise the integrated nature of all water flows in catchments, including overland, subsoil and groundwater flows. Groundwater provides flow to and from lakes, wetlands, springs and seeps. Where rivers flow on alluvial floodplains, flows from groundwater provide the summer base flow that is critical for sustaining ecosystem services in rivers during periods of low flow. The importance of the interconnectedness of surface and groundwater systems is increasingly being recognised in water management (White *et al.* 2001a), and the biodiversity of New Zealand's groundwater systems is only just being explored (see chapter 32).

One ecosystem service provided by freshwater is the assimilation of waste, including contaminants in stormwater, wastewater, and point-source and non-point-source discharges (Ministry for the Environment 1997). The capacity of our soils, rivers and groundwater systems to treat, dilute and transport contaminants must be carefully considered when discharging wastes onto land and into waterways. Similarly, the intensification of agriculture and horticulture has increased the release of sediments, nutrients and microbial contaminants to freshwater systems. The land's vegetation cover is one of the key determinants of water quality in a catchment. In turn, declining water quality reduces the value of freshwater and limits its potential uses.

Recreation

New Zealand's thousands of rivers and lakes are extensively used for recreation by both New Zealand residents and international visitors. Groundwater is an essential element of caving, glow-worm habitat, and activities associated with springs and geothermal activity. New Zealanders participate in a huge variety of recreational activities. Some activities, such as kayaking, rafting and angling, are directly associated with water. For other activities, such as tramping, picnics and walking, water is the backdrop for enjoying natural environments. These experiences with nature can range from remote wilderness adventures to wandering along an urban stream.

Egarr and Egarr (1981) carried out the only comprehensive survey of the recreational values of New Zealand's rivers. Since then there has been no national survey of the types and extent of freshwater recreation in New Zealand (APR Consultants 2003). The Freshwater Angler Survey undertaken by Fish and Game is the only regularly undertaken survey of recreational use of New Zealand's rivers—that survey provides the numbers of anglers visiting each river over a season.

For those activities directly associated with water, fishing is by far the most frequent activity. In the 2002/3 season 132,000 fishing licences were issued. Most outdoor recreation in New Zealand takes place close to centres of population (Booth and Peebles 1995). The greatest fishing effort occurs in lowland rivers, while back country and headwater fisheries account for less than 10% of national angling (NIWA 2003a).

Since the survey of Egarr and Egarr (1981), the types of activities undertaken on New Zealand's lakes and rivers have changed very little (APR Consultants 2003), but the popularity and extent of freshwater recreation has increased markedly. There have been some new variations within activities, for example white-water sledging and kite boarding. Technical advances have had a direct influence on recreation activities (e.g., a change in the materials used to build kayaks) or have indirectly influenced recreation activities (e.g., higher performance helicopters and the availability of global positioning systems allow access to a wider range of remote rivers) (APR Consultants 2003). These changes have increased the accessibility of rivers to recreational users, and are likely to have increased the range of rivers supporting recreation, particularly those in more remote locations.

Recreational values are not solely related to the numbers of people, and require a measure of the quality of the experience. For some activities, the uniqueness of an experience or its remoteness is one of its most valued characteristics. The recreational values obtained from freshwater systems are not just a function of water quantity and quality. Access to sites is critical, as is the landscape and vistas experienced and the ability of our rivers to support recreational fish species.

Tourism

Tourism is a very important component of the New Zealand economy, contributing directly and indirectly almost 10% of New Zealand's Gross Domestic Product. International visitor spending in 2001 was \$6.1 billion dollars. For the same year, it is estimated that New Zealanders spent \$6.9 billion on day and overnight trips within New Zealand. Domestic tourism and international visitor arrivals have grown constantly for the last four decades. Water-related domestic tourism, and increasingly international tourism, are very closely linked to the recreational values of fresh-water systems.

The role that New Zealand's freshwater systems play in supporting both domestic and international tourism is difficult to distinguish from that of other attractions, but freshwater is a critical element of nature, scenic and wilderness tourist experiences. Annual surveys of visitors report the number of participants in various activities, including swimming, fishing, kayaking/canoeing, and jet

boating, and using hot pools and visiting geothermal attractions. For example, in the year ended December 2002 around 24% of our 1.8 million international visitors visited geothermal sites, 10% participated in jet-boating, 2% fished in lakes, 2% fished in rivers, and 1% went river kayaking (Tourism Research Council 2003a).

The domestic tourism survey does not discriminate between activities such as swimming, fishing or kayaking occur in coastal waters or in freshwater. In 2001 New Zealanders made an estimated 1.6 million overnight trips and 39 million day trips (Tourism Research Council 2003b). The 2001 domestic tourism survey estimates 2% of daytrips and 6% of overnight trips made by New Zealanders involved fishing, while 11% of daytrips and 17.5% of overnight trips were for sightseeing and scenery.

Maori values

Maori place a very high value on water, regarding water as a taonga left by ancestors to provide and sustain life. Freshwater formed a fundamental part of traditional ways of life and remains extremely important to Maori today. From a practical perspective, water bodies are a source of sustenance, mahinga kai (cultural and customary food and resources), irrigation, cooking (geothermal water), recreation, navigation and cleansing (Parliament Commissioner for the Environment 2000). Water has a fundamental role in many ritual and healing processes. Specific water bodies may be tapu, or sacred, and are reserved for ceremonial use, or a special event or activity.

An essential part of Maori belief is that each water body has its own individual mauri or vital essence. A water body with a healthy mauri will sustain healthy ecosystems, support a range of cultural practices and reinforce the cultural identity of the people (Ministry for the Environment 1997). Two other important concepts are kaitiakitanga and ki uta ki tai. Kaitiakitanga is the obligation or responsibility to protect, guard and be custodians of Maori interests, resources and taonga, including the mauri of water bodies. Ki uta ki tai (mountains to the sea) reflects a whole catchment approach to water management (Tipa and Tierney 2003). The protection of mauri has become one of the principle issues for contemporary resource management, because Maori are increasingly concerned with the integrity of waterways on which their survival and cultural identity depends (Tipa and Tierney 2003).

The desire to exercise kaitiakitanga has led Maori to seek an effective role in the institutions that govern water management. Recent examples include a new provision in the Local Government Act 2002 that "councils must consider ways in which they can foster the development of Maori capacity to contribute to decision making processes", and the Statutory Acknowledgements in the

Ngai Tahu Claims Settlement Act 1988, which acknowledge the traditional associations of Ngai Tahu with specific locations, many of which are catchments and rivers.

Spiritual, cultural and historic values

New Zealand has an incredible variety and number of freshwater systems. There are unlikely to be many New Zealanders who do not identify with a river or lake that features in their childhood play, holidays, adventures, family histories and/or whakapapa. Rivers, lakes, springs and geothermal systems influence how people identify with and have pride in where they live. By far the majority of our cities and towns are located on the banks or near the mouth of rivers and lakes (Young and Foster 1986). Many of our places names include "wai" or "water".

"The ancient maps of this land are shimmering with resonance. In contemplating a river's flow or its meander, in listening to its murmur, one may come to hear the heartbeat of the land. In this way we glimpse what all our ancestors knew: that rivers may soothe, heal and regenerate. We most surely need them today for our wise use of electricity, for necessary irrigation, and maybe even for their gold and for carefully controlled waste disposal. Most vitally we require the wisdom to recognise that, as with all living things of this planet, their well-being is our own."

D. Young in the introduction to *Faces of the river—New Zealand's living water* (Young and Foster 1986)

The importance of natural freshwater systems to an individual, family, iwi or national identity cannot be easily described, yet must be taken into account in management decisions.

"Some witnesses spoke of the immeasurable health benefits afforded from the pleasure and relaxation of just being there—the intangible, unseen things about the river and the environment". "... braided rivers are important to the landscape and cultural identity of the people of Canterbury—these rivers have influenced the way in which Canterbury people express themselves and their sense of place in literature, music and art."

Special Tribunal on Rangitata Water Conservation Order 2002

Domestic water

Access to clean drinking water is a basic human necessity. The Resource Management Act (1991) allows an individual to take water for drinking as of right, provided the taking or use does not, or is not likely to, have an adverse effect on the environment. The average daily domestic use in New Zealand ranges from 180 to 300 litres per person per day, compared with 270 litres in Australia and 380 litres in England (Parliamentary Commissioner for the Environment 2000). The amount used per household increases significantly with garden

irrigation. For example, domestic use in Christchurch is around 800 litres per person per day in a very dry summer.

It is estimated that 85% of the population is supplied from registered community drinking water supplies. Private communal supplies provide water for 4% of the population and 11% have individual supplies from rainwater tanks or bores (Parliamentary Commissioner for the Environment 2000). Twenty-six percent of New Zealand's population drink groundwater, 48% drink water taken from rivers, 25% drink water from two or more sources and 1% drink water from rainwater tanks (Davies 2001).

The Ministry of Health maintains a register of the community supplies and their compliance with the Drinking Water Standards for New Zealand (Ministry of Health 2000a). In 1999, 87% of the population was supplied with water that complied with the microbial guideline values for faecal coliform, 79% with *gardia* guidelines and 76% with *cryptosporidium* (Ministry of Health 2000b).

Stock water

Just as for human drinking water, clean water for an individual's animals is provided as of right under the Resource Management Act (1991). The amount needed varies with stock types, management practices and climate. Table 42.1 presents the suggested water allowances for the design of stock water systems.

In many parts of the country, stock water is often carried in large open races. The amount of water actually consumed by stock can be as low as 3% of the water abstracted (Morgan *et al.* 2002). The merits of these races and their efficiency are strongly debated. Replacing races with pipes could reduce abstraction from rivers or free up allocation for other uses. The stock water races, however, have significant wildlife, recreational and aesthetic characteristics which, because the races have been around for many decades, are highly valued by local communities.

Table 42.1 Peak daily water consumption by stock type
Harrington (1980).

Stock type	Design peak daily water consumption (litres/day per animal)
Friesian dairy cattle – lactating	70
Friesian dairy cattle – non-lactating	45
Angus breeding cow – lactating	65
Angus breeding cow – non-lactating	40
Jersey dairy cow – lactating	55
Jersey dairy cow – non-lactating	35
Rangeland cattle – non-lactating	70
Breeding ewes	3

Industrial water

In most regions the industrial use of water is the smallest component of total water use, but it often has the highest economic value per unit of water used. Although the amounts of water required by industry may not be large, processing and manufacturing businesses must have access to reliable supplies of clean water. While businesses requiring high water quality can treat the water on site, a guaranteed reliability of water supply is critically important to them.

Forty percent of industrial water use is supplied through urban water supply systems, and the rest via individual takes (Parliamentary Commissioner for the Environment 2000). Industrial water use is not large and it is difficult to assess demands, as industries vary considerably in the quantity of water they require and the extent to which water is re-used.

Hydro-electricity

Hydro-electricity provides 70% of New Zealand's electricity generation capacity (Ministry of Economic Development 2003), two thirds of which is in the South Island. The generation capacity is located on a small number of large rivers and lakes (see chapter 37). Collectively, the catchments of the Waikato, Clutha, Waitaki and Waiau rivers provide 83% of New Zealand's hydro-electricity generation capacity.

New Zealand's commitments under the National Energy Efficiency and Conservation Strategy (Energy Efficiency and Conservation Association and Ministry for the Environment 2001) include a commitment to increase the supply of renewable energy by 30 Petajoules in 2012 (approximately a 30% increase). Hydro-electricity is likely to play a major part in meeting that target.

Irrigation

Irrigation is the largest abstractive use of water, accounting for 77% of water allocated to abstractive uses (Robb 2000). The area of land irrigated in New Zealand has increased from 270,000 hectares in 1985 to more than 470,000 hectares in 2002 (Statistics New Zealand 2003). Irrigation is common through the length and breadth of the country and in all regions. Irrigation water is used to support viticulture, horticulture, dairying, sheep, beef, and cropping. Of the land developed for irrigation, dairy pasture accounts for 31% of irrigation, other pasture 34%, arable land 22%, horticulture 11%, and viticulture 1% (Robb 2000).

Irrigation affects the economic viability of rural land, and therefore the socio-economic well-being of rural areas and regions. Irrigation reduces production risks and, more importantly, allows land to be used for purposes that would

not otherwise be viable in the local climate. The net benefit of irrigation in terms of the at-the-farm-gate value of production in Canterbury during the 1997-99 drought years was estimated at \$365 million (Ford and Butcher 2000). This gives some indication of the risk reduction value of irrigation.

Figure 42.1 illustrates how the amount of land irrigated area in New Zealand has grown steadily from beginnings in Central Otago in the early 1900s. The growth from the mid-1960s has been particularly rapid, and largely unaffected by the withdrawal of government involvement in irrigation in the mid-1980s (Table 42.2). The largest areas of irrigated land are in Canterbury and Otago, as shown in Table 42.2.

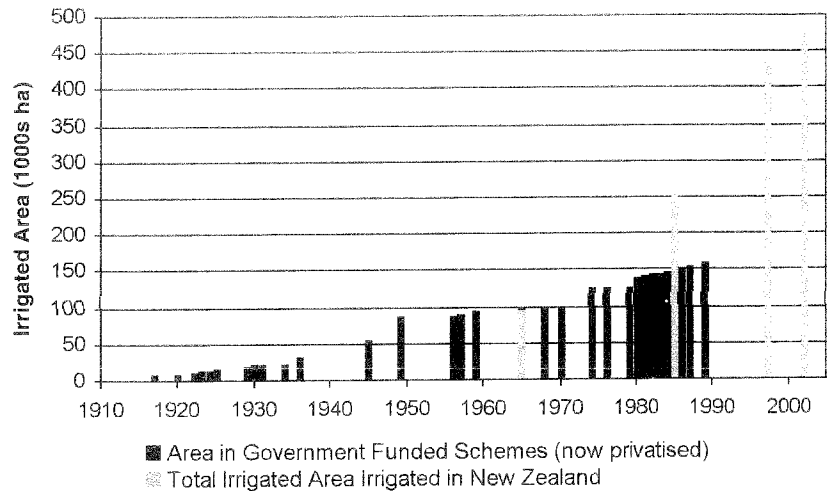


Figure 42.1 Irrigated area in New Zealand 1920 to 2002. Based on data reported in Robb (2000) and URS (2000).

Commercial freshwater fisheries

New Zealand’s freshwater supports commercial fishing for eel and whitebait. Commercial eel fishing peaked in the 1970s at 1400 tonnes per annum (Ministry for the Environment 1997). Whitebait are juvenile galaxiids caught as they move from the sea into rivers and streams. Most whitebait are inanga (*Galaxias maculatus*), although on the West Coast of the South Island whitebait species included three threatened species—koaro, short-jawed kokopu and giant kokopu (Ministry for the Environment 1997).

INFORMATION ON THE AMOUNT OF WATER USED

Instream values and uses

It is difficult to quantify how much water is “used” to sustain values and provide for instream uses in rivers and groundwater systems. Damming, diversion and abstraction can significantly alter flow patterns, and not just by removing water from a freshwater system. The modifications required to dam or abstract water raise a suite of other potential effects on values such as fish

Table 42.2 Land under an irrigation system during the year ended 30 June 2002 (Statistics New Zealand 2003).

Irrigation in the North Island		Irrigation in the South Island	
Region	Irrigated Area (hectares)	Region	Irrigated Area (hectares)
Northland	7,041	Tasman	10,030
Auckland	6,266	Marlborough	20,188
Waikato	12,652	West Coast	2,462
Bay of Plenty	8,839	Canterbury	287,168
Gisborne	1,325	Otago	68,869
Hawkes Bay	18,138	Southland	4,075
Taranaki	2,941		
Manawatu-Wanganui	7,967		
Wellington	9,550		
Total North Island	74,718	Total South Island	393,013

passage, temperature changes, sediment and gravel flows, water quality and bed disturbance.

In very simple terms, any water that is not abstracted or dammed is used to provide and sustain instream values. The Instream Flow Guidelines (Ministry for the Environment 1998) outline methods for assessing the flow requirements for various values and activities. These guidelines describe how much water should remain in a river to meet environmental baselines, but it should be acknowledged that instream values begin to be affected at even smaller levels of abstraction, and there is a judgement required as to the acceptable amount of change.

Nearly all rivers from which water is taken have a minimum flow, set to prevent abstraction or damming from further lowering river flows during times when flows are naturally low. As environmental flow methods have evolved, there has been a move away from a "minimum flow" concept to a "flow regime" concept, which recognises the importance of maintaining flow variability (Schofield *et al.* 2003).

It is becoming increasingly common for regional councils to "cap" or "limit" the amount of water that can be allocated from a river or groundwater system. That approach recognises that sustaining instream or groundwater-dependent values needs more than a minimum flow. It recognises the importance of flushing flows, maintaining flows between ground and surface water systems, small freshes, and ensuring that a river is not held at a constant flow for too long. A limit also provides an assurance that water will not continue to be allocated and that the

cumulative effects of multiple abstractions cannot increase indefinitely. In Australia the setting of caps or limits has triggered a far greater emphasis on efficiency, water storage and high-value uses of water.

Development values and uses

Development users have not routinely been required to measure and report water use. As a result, information on the amount of water used is not widely reported. The Ministry for the Environment (1997) used studies carried out in the early 1990s to report the volumes of water used in New Zealand. Since then, the total use has not been reported. However, the average annual flow through hydro-electricity stations from 1995 to 2001 has been estimated as 5,000 m³/s (NIWA 2003b). That figure counts the same water multiple times if it passes through more than one power station. Table 42.3 contains our reassessment of the volumes of water used for development uses, based on updated data on population, stock numbers and irrigated area.

Hydro-electricity generation and abstraction of water for drinking, crop and livestock production, industrial production, and normal household activities is estimated at almost 164,000 million cubic metres per year, equivalent to 115,000 litres per person per day. When the amount used for hydroelectricity is subtracted, the total still exceeds 2,160 litres per person per day. Of this total, 160 litres are used by each person in the home and at the workplace.

Regional council and unitary authority databases on resource consents provide another means to assess the

Table 42.3 Estimated yearly use of water

Type of use	Estimated yearly use in millions of cubic metres (as reported in Ministry for the Environment 1997)	Estimated yearly use in millions of cubic metres (re-estimated using 2003 data)	Source
Electricity generation	100,000	160,000	NIWA 2003b
Irrigation	1,100	2,300	Morgan <i>et al.</i> 2003 (average irrigation depths); Statistics New Zealand 2002 (irrigated area in 2002)
Livestock consumption	350	270	Harrington 1980 (standard stock requirements); Statistics New Zealand 2002 (stock numbers in 2002)
Industry consumption	260	260	No information to update from MfE 1997
Household consumption	210	230	Estimates of per capita use (MfE 1997), Statistics New Zealand 2002 (2001 population)

Table 42.4 Water allocated to abstractive uses by region and type of use (Robb 2000).

Region	Water allocated (m ³ /s)	% of regional total for each use		
		% of allocation for irrigation	% of allocation for industrial use	% of allocation for public water supply
Northland	7.9	46%	13%	41%
Auckland	8.1	26%	9%	65%
Waikato	10.3	32%	42%	26%
Bay of Plenty	8.9	40%	19%	41%
Gisborne	1.4	86%	<1%	14%
Hawke's Bay	16.8	68%	11%	21%
Taranaki	3.4	13%	41%	46%
Manawatu-Wanganui	5.2	39%	20%	41%
Wellington	9.8	40%	4%	56%
Tasman	6.7	87%	6%	7%
Marlborough	8.2	79%	12%	9%
Canterbury	249.8	84%	3%	13%
Otago	90.0	85%	10%	5%
Southland	2.4	21%	34%	45%
TOTAL New Zealand	428.9	77%	7%	16%

Table 42.5 Contribution to total water allocated and hydro-electric capacity by water resource type

Source of water	Contribution to national total from each water resource type	
	% of water allocated for abstraction	% of hydro-electric generation capacity
Low elevation rivers	11%	1%
Hill-fed rivers	35%	5%
Mountain-fed rivers	2%	7%
Glacial-fed rivers	13%	1%
Lake-fed rivers	9%	86%
Groundwater	30%	

relative amounts of water required for each abstractive use. Table 42.4 provides a breakdown, by region, of water allocated to classes of abstractive uses. The actual flow used for abstractive purposes will be lower than the allocated amount, because many allocations are made on the basis of peak requirements in a dry year, or for public water supplies based on the projected future population.

Another way to examine the information in Table 42.4 is to assess the types of water resources that support the various development uses (Table 42.5). The information gives some indication of the importance of each type of water resource for development uses and of the social, economic, cultural and environmental values associated with those uses.

FUTURE INCREASE IN DEMAND FOR WATER AND ITS VALUES

Instream values

As demand for more water abstraction increases, there will be an increase in concern about any loss or decline in instream values. The quality of the New Zealand environment is receiving increasing public attention—this trend may result in the revision of water allocation policies to improve protection of instream values. A recent survey (Hughey *et al.* 2002) on public perceptions on the state of water and water management found that most respondents thought that there was a high to moderate availability of water. This was no different to an earlier survey in 2000. More people (38%) identified farming as a cause of damage to water in 2002 than in 2000 (24%). Whether this relates to quantity or quality, or both, was not asked. The survey identified strong support for increased spending on water management.

Recreation and tourism

The major increase in demand for freshwater-based recreation will be driven by population increase, growth in tourism and increased leisure time. New Zealand's population is predicted to increase to a peak of 4.81 million in 2046 and then to slowly decline. The population in 2021 is predicted to be 16% greater than 2001, with a greater than 20% increase in Auckland (36%) and Bay of Plenty (25%).

New Zealand has experienced an almost constant growth in international visitor arrivals for the last four decades. Overseas visitor arrivals in to the year ended May 2003 represented a 5.2% increase over the May 2002 year, with an increasing proportion of independent travellers. International visitors nights are expected to increase on average by 6.5% per annum. Domestic nights are expected to increase 0.5% per annum, with day trips increasing by 3.1% per annum (Tourism Research Council New Zealand 2003c). The proportion of international visitors classed as “free and independent” is also predicted to increase. These people can be expected to undertake similar activities to local recreationists, which means an increase in international visitors involved in activities such as angling, canoeing and rafting at more remote locations. The experience of clean, healthy rivers in natural landscapes is very important for the overseas tourist experience, and pressure to maintain natural landscapes is expected to increase.

Hydro-electricity

New Zealand's energy sector has experienced a period of significant change and reform over the past decade and is projected to alter significantly over the period to 2020. A baseline scenario, based on a 2.5% per annum growth in GDP, predicts a 1.2% annual growth in demand for electricity (Ministry of Economic Development 2003). Compared to a prediction made in 2000 of a 1.8% growth in electricity demand, the more recent figures assumes additional improvements in energy efficiency as a result of the National Energy Efficiency and Conservation Strategy (Energy Efficiency Conservation Association and Ministry for the Environment 2001).

The composition of energy demand and supply is projected to change as, amongst other factors, the demand for energy grows, the Maui gas field declines, new technologies for the production, delivery and use of energy become available, and the emphasis on greenhouse gas emissions increases (Ministry of Economic Development 2003). It is projected that hydro-electricity will provide around 52% of the total capacity by 2020, compared to 65% of capacity in 2000. Those figures require 890 MW of new hydro-generation by 2020—a 27% increase in existing hydro-electricity generation capacity.

Irrigation

Farming in a drought-prone climate is now less viable than 30 years ago, because of tighter controls on product quality and decreasing commodity prices. It is very likely that physical and economic incentives for future irrigation expansion will continue to strengthen the demand for irrigation water. Increasingly, irrigation demand will need to be met from storage, because the available surface water

resources that are reliable are almost fully developed. We estimate that about 20% of the land that could technically be irrigated, if sufficient water were available, is currently irrigated.

The area of land that could be developed for irrigation varies significantly, depending on assumptions about future water management, the attitudes of existing and potential users, and access to capital within the farming sector. Scenarios developed by Morgan *et al.* (2003) showed major differences depending on environmental, institutional and capital constraints. For example, stricter water quality requirements (especially related to nitrates) would severely limit any further increases in irrigated area (Morgan *et al.* 2003). However, removing institutional and capital impediments would greatly increase the amount of irrigation development. The potential by region is illustrated in Table 42.6 for two of the scenarios.

Comparison of the possible increase in irrigated area under each of the scenarios in Table 42.6 indicates that irrigation development is very likely to be constrained (less than halved) by water supply in about seven regions—Auckland, Northland, Bay of Plenty, Gisborne, Tasman, Canterbury and Otago. Here the supply for irrigation can be limited by several factors—water resource limitations, limitations on development capital, and institutional or planning impediments. For Auckland and Tasman, increases in irrigated area are likely to be further constrained by competition for land, with urban fringe and rural lifestyle development.

In areas where irrigation expansion is limited by the lack of a reliable water supply, water storage may be necessary for further expansion, which may raise environmental, capital, institutional and planning issues. An increased focus on water allocation rules in these areas is highly likely.

Stock water

Stock water consumption is likely to increase in areas undergoing a change in land use from dryland pastoral farming to intensive irrigated pastoral farming. Meeting that additional demand may not require an increase in water abstracted. There is considerable potential for any increase in stock water consumption to be met through improvements in efficiency.

Industrial and drinking water

The demand for industrial and drinking water is driven both by population increases and by increasing tourist numbers. Increases in the processing of agricultural and horticultural products as a result of land use intensification will also drive up industrial demand. An allowance for future growth is often included in allocations of water to urban and community water supplies. Subsequently, the allocation to these uses may not increase at the same rate

Table 42.6 Predicted increase in irrigated area by region (Statistics NZ 2003, Morgan *et al.* 2003).

	Land under an irrigation system during the year ended 30 June 2002 (ha)	Possible increase in area under irrigation in 2010, given current trends and constraints (% change)	Possible increase in area under irrigation in 2010, without water supply constraints (% change)
Northland	7,041	70	240
Auckland	6,266	20	65
Waikato	12,652	45	70
Bay of Plenty	8,839	50	120
Gisborne	1,325	60	275
Hawkes Bay	18,138	45	70
Taranaki	2,941	0	0
Manawatu-Wanganui	7,967	100	130
Wellington/Wairarapa	9,550	70	130
Tasman/Nelson	10,030	20	75
Marlborough	20,188	70	110
West Coast	2,462	0	0
Canterbury	287,168	30	60
Otago	68,869	10	115
Southland	4,075	120	130
TOTAL	467,731	30	80

as the population. While drinking and industrial demands are expected to remain a small part of overall demand, they will continue to be the most highly valued development uses.

An aspect of urban water management that is receiving increasing attention is the integrated management of stormwater, wastewater and water supplies. Using rainfall as a source of water also reduces stormwater generation. Similarly, re-using water reduces both wastewater volumes and the need for water supply.

The potential for increased efficiency to reduce demand

Increased demands for water for development do not necessarily have to be met by an increased supply—they can be achieved by improved efficiency and the use of alternatives. Urban water use in Auckland is an example of the advantages of increased efficiency in water use. The population in the Auckland City area increased by 19.3% in the ten years between 1991 and 2001; at the same time the total water supplied decreased by 8%. Average per capita use has fallen 20% since 1990 (Parliamentary Commissioner for the Environment 2000).

During the 1994 drought, long-term requirements for an additional, more secure water source for Auckland resulted in proposals to pipe water from the Waikato River. The new Waikato Pipeline is expected to cost \$155 million

when completed and running at full capacity, and will supply 10% of the region's need. Increased efficiency did not prevent the need for more water to Auckland. However, the 20% reduction in per capita water use since 1990 is estimated to have saved over \$300 million in water supply costs. This figure does not take into account the environmental effects that would inevitably have resulted from the use of more water.

The success in Auckland is likely due to practices for managing demand such as leak prevention and a pricing regime introduced by the water delivery authority, Metrowater. Total water consumption was significantly reduced when charges for wastewater, on the basis of water supplied to a property, were introduced. The reduction was more significant than the reduction that accompanied the introduction of water meters.

The charges used in Auckland and other urban centres are not a charge for the water itself—they are to cover the provision of infrastructure, delivery costs and maintenance. The charges are made by an entity other than the final users of water. Many resource consents to take water, particularly for irrigation, are to supply an individual farm or industry. In that case the delivery and infrastructure costs are met directly by the person or company who is the water user. There is not the same opportunity to make use of water charges as an incentive for efficiency.

POTENTIAL CONFLICTS

Irrigation, because it is widespread and continues to grow, will increasingly come into conflict with instream interests and New Zealander's expectations of high environmental quality in all river types, and in groundwater-dependent ecosystems. Compounding the conflict is the intensification of land use that accompanies irrigation, particularly livestock intensification. That conflict is mostly like to occur in catchments where the land use is already agricultural. Small streams in lowland areas are particularly vulnerable to pressure from pastoral use of the adjacent land, in combination with abstraction. These smaller streams are not well represented in the flow monitoring network or in planning processes, which tend to focus either on major catchments or aquifers, or on region-wide concerns.

The storage infrastructure that will be required to meet long-term water demand is likely to compete with spiritual, cultural, recreational and ecosystem needs in hill and mountain rivers, and in major river systems that are currently free of dams.

As the two major development uses of water, hydro-electricity and irrigation are likely to increasingly compete for access to water. In some cases, the demand by both can be met from a single development, but there may be conflict when storage is involved. On a national level, three regions have significant potential for new irrigation and hydro-electricity that will be significant nationally (more than 5% of the total increase predicted nationally). These regions are Canterbury, Otago and Hawkes Bay; of which Canterbury provides at least four times as much as the next region.

COMPARING MONETARY VALUES

Balancing among different values is resolved through consultation and decision-making processes specified by the Resource Management Act. Application of that Act has to date concentrated on providing for environmental values, rather than the assessment of economic and social tradeoffs. Therefore values as a suite have not been routinely reported or analysed. One approach to comparison is describing the values in monetary terms, an approach that has been applied in some New Zealand examples.

Some of the values and uses of water may be quantified in monetary terms, particularly the values arising from development uses. Other values, particularly the instream values, are difficult, and more controversial, to assign a dollar value. There is a fundamental problem in placing a monetary value on all values because of the difficulty of inter-connected values and because people have different perceptions of the value of the "services" they receive from

water. On the other hand, valuing ecosystem services does raise the profile of those services as an important part of the economy. For example, Paterson and Cole (1998) estimated that the Waikato River ecosystem provides services to the value of about \$9.4 billion in 1997. An estimated 64% of the value is from water-based ecosystems. This compares to a \$9.9 billion Gross Domestic Product of the Waikato region in that year.

Non-market valuation techniques

Non-market valuation aims to provide a monetary measure of value for goods, services, activities, and experiences that are not themselves exchanged via markets, nor produce goods which are sold on a market. Non-market techniques apply to measuring use values, such as recreation, but also to more esoteric values such as existence value. Existence value is the benefit gained just by knowing something exists. Typically the approach measures the quantity of money people are willing to pay to obtain some desired outcome. Values derived can be used to help allocate resources and to determine the basis for compensation or mitigation.

Techniques of evaluation include indirect and direct methods. For example, indirect methods can be used to estimate the value of a river view by comparing the value of a riverside property with a parcel of land that is similar in every other way but has no view of the river. Direct methods include contingent valuation, travel costs and choice modelling. Choice modelling aims to determine acceptable trade-offs between environmental enhancements and alternatives, e.g., local jobs in agriculture versus preserving a wetland. As an example of a direct technique, the value of recreational fishing is assessed by estimating people's willingness to pay for fishing licences, equipment, and the cost of travel.

Water has been the topic of around 25 of the 85 non-market-valuation studies undertaken in New Zealand, highlighting the importance placed on water (Kerr 2002). Four non-market valuation studies of freshwater angling have estimated the value per angler day—on the Rakaia River (\$45 per day), the Greenstone and Caples Rivers (\$70), Tongariro River (\$60) and Rangitata River (\$70). With some of these rivers having over 30,000 angler days, the magnitude of angler benefits on a single river can exceed \$2 million per year (Kerr 2002).

Examples of comparative value across multiple uses

In New Zealand there are very few examples of assessing the comparative value of water. The values obtained are very dependent on the method uses to assign monetary value, and assumptions made about the provision of

alternatives. Different valuation methods can yield different rankings of value. How water should be valued in comparative units is a topic that has not received wide attention to date.

Example 1—Valuation of the Waimea Plains groundwater system

The economic value of groundwater abstracted from the Waimea Plains groundwater system (Tasman District) was calculated to be approximately \$250 million (Table 42.7). This value is analogous to the capital value of an asset. An estimate of how much the public (represented by householders in the locality) would be willing to pay for the environment benefits arising from a 20% reduction in abstraction was estimated at \$183 per household per year. Aggregated over 6,300 households, it provides an estimate of \$1.2 million per year as the *in situ* value of the last 20% of the groundwater allocated.

Table 42.7 Economic value of the Waimea Plains groundwater system (White *et al.* 2001b).

	Value
Irrigation	\$38 to \$42 million
Commercial Industrial	\$173 million
Bulk water supply	\$33 million
TOTAL	\$244 to \$268 million

Example 2—Manawatu-Wanganui

The economic value of water in the Manawatu-Wanganui region (Table 42.8) was calculated for development uses (e.g., agriculture, hydropower, commercial use) and *in situ* uses (e.g., hunting, fishing, jetboating). Development uses of water in the region are estimated to total approximately \$2.4 billion and *in situ* costs/values are estimated at \$283 million.

Table 42.8 Economic value of water in the Manawatu-Wanganui region (White and Sharp 2002).

Development uses	Value (\$ million)	Allocation m ³ per day	\$ per m ³ of allocation ¹
Agriculture	145	315,636	460
Human water supply	303	99,316	3051
Commercial uses	1231	Not calculated	Not calculated
Hydro-power	473	3,387,000	140
Waste disposal	261		
Fishing	28		
Hunting	53		
Boating	53		
Swimming/picnics	12		
Existence value	137		

* assumes only 40% of the allocation is used, the remainder is assumed to have no value

¹ values are not an annual value but the total capital value of the water as if it were an asset

The examples above assess the total value of all the water currently allocated. The dollars per unit of water allocated represent average values, equivalent to a capitalised value of water. In some allocation decisions it is not the average value that is important but the value of the next unit of water to be allocated (the marginal value). The value of water will vary considerably from year to year. Assessment of values therefore requires information on the incremental costs and benefits of abstracting, damming or leaving water instream, and detailed modelling of the interconnectedness of the economy with the hydrology of catchments and the activities of water users.

Research on the value of Waimea Plains groundwater combined a groundwater and surface water flow model, a groundwater quality model and an economic model to predict the net economic benefit of allocating five times as much water as was currently allocated (White 2001). At that increased allocation, the Waimea River would cease to flow over the summer. It was estimated that bores supplying a community water supply and irrigation water to around 600 hectares of land would suffer from salt water intrusion. The increased economic benefits of irrigation (\$3.3 million) were outweighed by an economic loss of \$4.4 million due to poorer environmental quality, reduced land prices and the costs of moving a community water supply.

Indicative figures on the value of water

- New Zealand's clean green image has been estimated as worth \$241 to \$569 million per annum in revenue from the dairy sector and \$530 to \$938 million in the tourism sector (Ministry for the Environment 2001).
- New Zealand's tourism industry is worth \$13 billion per annum.
- The value of recreational fishing in the Rangitata River is estimated at over \$2 million (Kerr 2002).

- Local authority water infrastructure is valued at approximately \$7.5 billion, with around \$600 million spent on operational costs each year (Parliamentary Commissioner for the Environment 2000). It has been estimated that around \$5 billion is needed by 2020 to upgrade water, wastewater and stormwater infrastructure.
- It will cost \$155 million to build and fully commission the Waitako pipeline to Auckland to supply 150 litres per second (Watercare Services 2003).
- The Waimea Plains groundwater system has 4 m³/s allocated to development uses and this water is estimated to provide \$250 million of economic value (White *et al.* 2001b). Given that 4 m³/s is 1% of the total water allocated in New Zealand (excluding hydro-electricity), then the economic value (capital) of water abstracted in New Zealand is estimated at around \$25 billion.
- A wholesale price of electricity of 6.3 cents per kilowatt hour in 2005 (Ministry of Economic Development 2003) would mean that the 24,000 gigawatt hours generated in 2002 was worth over \$1.4 billion. That figure would be considerably higher (over twice as high) if it were based on the price consumers pay for the electricity.
- The net benefit of irrigation in terms of the at-the-farm-gate value of production in Canterbury during the 1997-99 drought years was estimated at \$365 million (Ford and Butcher 2000).
- The benefits of irrigating 84,000 hectares of the Canterbury Plains are a \$153 million increase in annual net output at the farm gate, and an increase in on-farm employment of 477 full-time equivalent positions, with an increase in total regional employment of 2630 full-time equivalents (URS 2002).
- From 1951 to 1984, New Zealand spent over \$1 billion (in 1984 dollars) in flood protection works and repairing flood damage (Waugh *et al.* 1997).
- Irrigation is estimated to add a total of \$807 million dollars per year in at-farm-gate value to New Zealand's agricultural and horticultural enterprises (Robb 2000):

Horticulture	\$444 million
Viticulture	\$ 32 million
Dairy	\$ 213 million
Arable crops	\$ 70 million
Other pasture	\$ 48 million

CONCLUSION

New Zealand's freshwater is used for many purposes—for ecosystem services, recreation, tourism, drinking water, household activities, stock water, irrigation, industry and

hydro-electricity generation. Our waters are valued as an element of spiritual beliefs and experiences, and cultural identity and history. Water is always given a high value, whether it is the intangible value of listening to someone speak about their feelings for a river or a market valuation of water. For some uses water is irreplaceable, but for other uses there are alternatives, for example using wind as a source of electricity. For some values, it is easy to describe how much water is needed, and to predict the amount and location of increases in demand. Much is known about some values, very little about others—the task of managing the balance of these values is therefore complex, and information on the values and uses of water both now and into the future is critical for wise water management.

Demand for water in New Zealand continues to increase, leading to an increased potential for conflict, but also increased opportunity for innovative solutions. Informed choices require understanding values, how they change over time and how they relate to changes in water quantity and quality. Where our understanding is incomplete the challenge is to act in a precautionary way that still enables us to learn how freshwater systems respond to human-induced change, but does not do irreversible damage to the systems or the values they sustain.

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Chapter 41

Sport fishery management

Neil Deans, Martin Unwin and Maurice Rodway

INTRODUCTION

Sports fish (formerly acclimatised fish) are defined in the first schedule of the Freshwater Fisheries Regulations 1983. They include brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*), Chinook (or quinnat¹) salmon (*O. tshawytscha*), and coarse fish such as perch (*Perca fluviatilis*) and tench (*Tinca tinca*). Under the Conservation Act 1987 they are managed “in the recreational interests of anglers” by regional Councils of Fish & Game New Zealand (FGNZ). The trout fishery in the Taupo region is run by the Department of Conservation (DoC), acting as a Fish & Game Council for the purposes of the Conservation Act.

The former Acclimatisation Societies promoted and carried out the introduction of most sports fish species under statute, for recreational angling. Fishing for these species has been regulated since before their initial introduction. Salmon have been farmed since the early 1980s, but trout remain a wholly recreational species, and cannot be farmed or harvested commercially. The Minister of Conservation gazettes Annual Angler’s Notices on advice of FGNZ. These regulations specify, for each species and (usually) any particular water, the length and timing of the season, the daily bag limit, and any restrictions on the methods used to catch fish.

Whitebait (*Galaxias* spp.) are managed by DoC as a recreational fishery. Although not strictly a sports fish, a brief section on the whitebait fishery is included in this chapter.

HISTORY OF SPORTS FISHERIES

When European settlers arrived in New Zealand, they considered the country’s animal fauna to be relatively

depauperate (except for birds). With the possible exception of eels (*Anguilla* sp.) and grayling (*Prototroctes oxyrinchus*), no species was considered suitable for fishing by the European immigrants (see McDowall 1994a for a more detailed history). Acclimatisation movements considered the country to be a prime candidate for stocking with exotic species for both sport and sustenance. Many of our waters were particularly suitable for stocking with freshwater fish.

The obvious candidate for stocking was the brown trout: their introduction was first provided for by legislation in 1867 and they were introduced via Tasmania by several Acclimatisation Societies. Hatcheries were developed throughout the country, with trout being liberated into virtually every accessible water. Brown trout were immediately successful, forming largely self-sustaining populations. Today they are by far the most widely distributed introduced species. Only in Northland are brown trout populations limited, probably due to higher winter water temperatures, which do not favour ova development (Jowett 1990; Elliott 1994). Rainbow trout have generally proven better suited to lakes or their associated rivers, and are the basis of the internationally recognised fisheries in areas such as Lake Taupo, the Rotorua region, Otago and Southland. The Rangitikei, Pelorus and Whitcombe Rivers, however, also have rainbow trout populations, even though they have no significant lakes in their catchments. New Zealand trout are recognised internationally, not just because of their relatively widespread distribution, but also for their rapid growth and large size. This latter feature remains part of the reason why New Zealand has been regarded so highly as an angler’s mecca. It has also led to research into why our trout grow large while feeding on drifting inverte-

¹ The terms “Chinook” and “quinnat” are synonymous. Both are of North American origin, but “quinnat” has long since fallen into disuse everywhere except New Zealand. This chapter uses “Chinook” throughout.

brates, and subsequently research on the development of energetics models for trout (Hayes *et al.* 2000).

Chinook salmon were introduced by the Government as a commercial venture in the 1900s, in an effort to provide for a marine fishery and for sustenance. The original liberations, using stock imported from the Sacramento River in California (McDowall 1994b), were confined to the Waitaki River system, but by 1915 the species had become established along the east coast of the South Island as far north as the Waimakariri River. These runs quickly became self-sustaining, and hatchery releases into the Waitaki River ceased shortly thereafter. This success is unusual in a global context—New Zealand is the only country where Chinook have established self-sustaining anadromous populations outside their native range.

While rainbow trout, brown trout and Chinook salmon adapted extremely well to New Zealand freshwater environments, and are the main species for the angler today, other introductions were less successful. These include several species that became established, but with very restricted distributions, such as lake trout (*Salvelinus namaycush*), brook char (*Salvelinus fontinalis*), Atlantic salmon (*Salmo salar*), and sockeye salmon (*O. nerka*). Others failed to establish at all—e.g., whitefish (*Coregonus* sp.) and cutthroat trout (*O. clarkii*). They are mostly of curiosity value as sports fisheries today, and at least one species (sockeye salmon) is close to extinction. Perch and tench were introduced to small parts of the country early in settlement, but were largely ignored and seldom targeted until recently. Coarse angling now has a small but dedicated following in some regions, such as Waikato, Manawatu, and around Christchurch.

The management of sports fisheries has been reviewed several times since the advent of the Acclimatisation Societies. Management had always been shared somewhat uneasily, with the central government managing sports fisheries and game birds in some areas, and the societies managing the remainder. A review in the late 1980s resulted in the establishment in 1990 of 12 regional FGNZ Councils, and one national Council, in all areas except for the Taupo trout fishery. Taupo alone now is managed under DoC, acting as a FGNZ Council, by agreement between the Crown and Ngati Tuwharetoa. This management regime differs from other FGNZ Councils, since the Department is advised by a committee appointed by the Minister and remains directly responsible to the Minister; it is not governed by elected licenceholder representatives, as is the case with FGNZ.

All FGNZ Councils are established and constrained by very specific legislation under the Conservation and Wildlife Acts to “manage, maintain and enhance the sports fish and gamebird resource in the recreational interests of

anglers and hunters”. Fish & Game Councils are responsible to the Minister of Conservation in carrying out their functions, but have independence from central government as Crown entities. The twelve regional Councils are elected from angling and hunting licenceholders, to manage sports fish and game birds. Funding comes entirely from the sale of fishing and hunting licences, so it is a “user pays, user says” system. The national Council comprises one member from each regional council and is responsible for national advocacy and coordination through national policy. In day-to-day management, FGNZ has a role that is similar to that of DoC in respect of the latter’s freshwater fisheries responsibilities, but they are responsible for different species, namely the sports fish.

FISHERY MANAGEMENT

Management of sports fisheries involves a variety of tasks. Salmonids are arguably the most studied fish in the world, so most of the basic biology and physiology of the species is reasonably well known (Behnke 2002; Elliott 1994). Fisheries managers need to understand the breeding cycle, location of spawning areas of wild stocks and production of juvenile fish. Factors affecting fish production, the survival rates of fish to breeding age, and the causes of mortality (including harvest) are also necessary information for managing fisheries. In addition to the biology of the species, their habitat requirements need to be well understood. These include the effects of changes in water quality and the quantity of water (river flows or lake levels) fish require; the effects of habitat alterations, such as the effects of damming on fish passage or of snag removal on adult fish habitats. The final part of the equation is the angler. Harvest rates, angler success, satisfaction and expectations are all factors in sports fisheries management (Pollock *et al.* 1994). FGNZ receives recommendations on the regulations each year from professional staff, but it is also influenced by political considerations, including the views of angling groups. Most sports fisheries are managed conservatively, as there is no economic imperative to allow fish harvesting to levels close to the sustainable limit. Despite this, there has been a general tendency across the country towards a liberalisation of fishing methods, but a reduction in the daily bag limits. Essentially fisheries management is an art based upon an inexact science.

Trout populations in New Zealand rivers are measured by varying methods, including drift diving (Teirney and Jowett 1990); mark recapture (Young and Hayes 2002); electric fishing (in smaller rivers only), and radio-tracking (e.g., Dedual and Jowett 1999). Fishery regulations include restrictions on daily bag limits, on fishing methods (all sports fishing must be with rod and running line, for

example, and live bait can be used only in some areas), and on fishing season. At present there are no restrictions upon the numbers of anglers able to fish in a particular location, although, to maintain the quality of the fishing experience, this is currently being considered for some back-country fisheries, where overuse is a perceived problem.

RIVER TROUT FISHERIES

Trout fishing in rivers is arguably the icon of sports fishing in New Zealand. As New Zealand is well endowed with rivers that are inherently suitable to support trout fisheries, river trout fishing is undertaken throughout almost all the country. There are many large, well-stocked and famous fishing rivers such as the Mataura in Southland, Motueka in Nelson, Rangitikei in the Manawatu and Tukituki in Hawke's Bay. A second category includes several hundred smaller lowland rivers, which collectively provide a large proportion of the fishable water and accessible fishing for most anglers. Back-country rivers such as the Greenstone in Otago, Rough on the West Coast or Ruakituri on the East Coast provide a third group. These three groups of rivers can be separated, although the differences between them are not absolute.

Most suitable medium to large rivers developed self-sustaining brown trout populations soon after the initial liberations. It was many years, however, before hatchery liberations for most rivers ceased, as they were considered essential both to establish and sustain trout fisheries. This view was challenged by New Zealand Marine Department scientists such as K. Radway Allen and Derisley Hobbs, who undertook internationally recognised investigations into river trout fisheries in the Horokiwi River near Wellington and other locations (Hobbs 1948; Allen 1951). It is now generally accepted that New Zealand's abundance of freestone gravel-bottom rivers provides adequate tributary or mainstem spawning for almost all river systems that support trout fisheries, with the exception of a few rivers in limestone areas (such as the Maraetotara in Hawke's Bay), which may require periodic restocking.

Large trout fishing rivers

The larger rivers support intensive fishing (Table 41.1). By comparison with salmon and lake fisheries, most river trout fisheries provide comparatively extensive fishing opportunities. Rates of angler use vary from an average of about 0.6 anglers/day/km of river over the whole year in a long river such as the Mataura or about 12-13 anglers/day/km in a smaller, more intensively fished river such as the Ngongotaha Stream, which flows into Lake Rotorua, or an average of 3.1 anglers/day/km in the Tongariro River.

Angler preferences and catch rates were initially

surveyed by having anglers keep diaries (e.g., Allen and Cunningham 1957). Angler diaries from different acclimatisation districts were summarized in a series of Fisheries Research Division reports during the 1970s (e.g., Graynoth 1974). The first national angling survey, a mail survey undertaken during the late 1970s (Teirney and Richardson 1992), provided much qualitative information on angler preferences and was particularly useful for identifying and comparing the rivers considered most significant for angling nationally. Later surveys (e.g., Unwin and Brown 1998), based on random samples of anglers contacted by telephone, provide more reliable quantitative data on resource usage, although they do not cover overseas anglers, whose usage patterns and preferences remain poorly known.

Smaller lowland rivers

Smaller lowland rivers support a significant amount of fishing in total, but this effort is widely dispersed, as so many rivers have fishable populations of trout. While these rivers are often the ordinary "close to home" fishing for many anglers, land use and other changes have altered the fishing opportunities in many rivers. Analysis of angler perceptions provides regionally consistent views of the causes of change over a considerable time period (Jellyman *et al.* 2003). Water resource changes, such as the alteration of river form due to flood control and drainage; changes to river flows due to water abstraction for hydroelectric power generation and irrigation; changed land use such as afforestation; and reductions in water quality through pollution from both point sources and diffuse sources such as agriculture (Shearer and Hayes 2003), all have altered and continue to alter sports and other fisheries habitat. The trends for increased demands on finite water resources for irrigation, for hydroelectric power generation, and to support intensification in land use, is likely to continue, as these developments have all made significant contributions to New Zealand's economic growth in recent years. The challenge to sustainably manage water resources has affected river trout fisheries disproportionately, due to the relatively high water quality and quantity requirements of trout.

Determining the flow requirements of trout and other fish, to assist in resource management of water resources, has been assisted by the development of instream flow models (e.g., Jowett 1992). These models have recently been further extended to include the ecosystem as a whole, particularly the effects of flow and other instream variation on energy production, and on invertebrate production and uptake by trout (e.g., Hayes *et al.* 2000).

Despite these advances, relating instream habitat to quantifiable physical parameters remains a challenge. For example, a recent re-examination of the Horokiwi Stream,

Table 41.1 Estimated annual usage (thousands of angler-days, ± 1 standard error) for the 40 most heavily-used river and lake fisheries over the 2001/02 fishing season. Usage estimates for the same waters in 1994/95 are also shown. Figures for the Taupo Conservancy (for the 1994/95 season) were supplied by DoC. Significant salmon fisheries are denoted by an asterisk.

River	1994/95	2001/02	Lake	1994/95	2001/02
Mataura River	51.4 \pm 3.3	52.9 \pm 4.1	Lake Taupo	125.5	
Waimakariri River *	58.4 \pm 7.1	48.9 \pm 4.3	Lake Tarawera	38.4 \pm 4.0	43.5 \pm 2.9
Clutha River	26.3 \pm 3.2	36.6 \pm 4.2	Lake Rotoiti (North I.)	43.4 \pm 3.4	43.1 \pm 3.1
Tongariro River		28.5	Lake Rotorua	40.2 \pm 4.4	32.6 \pm 2.6
Waitaki River *	34.5 \pm 3.1	26.6 \pm 2.7	Lake Hawea	18.8 \pm 2.3	28.2 \pm 3.7
Rakaia River *	34.6 \pm 3.8	21.5 \pm 2.0	Lake Wanaka	25.5 \pm 2.4	25.3 \pm 2.3
Oreti River	27.2 \pm 2.3	20.6 \pm 2.2	Lake Benmore	12.8 \pm 1.5	21.9 \pm 1.7
Taieri River	11.5 \pm 1.3	19.0 \pm 2.8	Lake Dunstan	22.3 \pm 1.8	19.9 \pm 2.9
Ngongotaha Stream	8.8 \pm 2.7	17.8 \pm 2.9	Lake Waikaremoana	20.6 \pm 2.2	19.7 \pm 2.1
Tukituki River	14.0 \pm 0.4	17.1 \pm 1.5	Lake Wakatipu	21.4 \pm 2.2	17.7 \pm 1.9
Waiau River (Southland)	7.7 \pm 0.8	14.6 \pm 1.6	Lake Aniwhenua	11.3 \pm 1.6	12.3 \pm 3.8
Manawatu River	12.0 \pm 1.4	13.7 \pm 1.3	Lake Te Anau	10.3 \pm 1.2	12.2 \pm 1.9
Opihi River	18.5 \pm 1.7	13.4 \pm 1.7	Lake Aviemore	8.8 \pm 1.3	11.6 \pm 1.5
Rangitaiki River	5.7 \pm 1.3	13.3 \pm 3.5	Lake Rotoma	6.6 \pm 1.3	10.1 \pm 1.3
Rangitata River *	36.0 \pm 2.6	12.7 \pm 1.9	Lake Arapuni	7.3 \pm 0.9	9.9 \pm 1.0
Tauranga-Taupo River		12.0	Lake Alexandrina	4.5 \pm 0.7	9.5 \pm 1.4
Wairau River	8.5 \pm 0.8	8.3 \pm 0.9	Lake Brunner	4.2 \pm 0.6	9.3 \pm 0.9
Hurunui River	17.1 \pm 3.3	8.2 \pm 1.0	Lake Coleridge	7.1 \pm 1.3	9.2 \pm 0.9
Tekapo Canal	0.9 \pm 0.2	7.7 \pm 0.9	Lake Tekapo	3.0 \pm 0.8	8.7 \pm 1.0
Waitahanui River		7.5	Lake Rerewhakaaitu	9.4 \pm 1.7	8.4 \pm 1.3
Waikato River	11.0 \pm 1.8	7.1 \pm 1.4	Lake Okataina	5.8 \pm 0.9	7.1 \pm 0.9
Mohaka River	3.8 \pm 0.2	7.0 \pm 0.8	Lake Monowai	4.0 \pm 0.6	6.3 \pm 1.1
Waikaia River	6.8 \pm 1.0	6.9 \pm 1.2	Lake Manapouri	5.5 \pm 0.9	5.9 \pm 0.9
Ruamahanga River	7.4 \pm 0.9	6.8 \pm 0.9	Lake Ohau	1.5 \pm 0.4	4.8 \pm 0.7
Aparima River	11.3 \pm 1.4	6.7 \pm 1.0	Lake Mahinerangi	4.1 \pm 0.7	4.7 \pm 1.1
Tutaekuri River	7.1 \pm 0.2	6.7 \pm 0.8	Logan Burn Reservoir	1.1 \pm 0.3	4.3 \pm 0.9
Motueka River	10.1 \pm 1.3	6.3 \pm 0.7	Lake Okareka	3.4 \pm 0.8	3.7 \pm 1.2
Ngaruroro River	3.8 \pm 0.2	6.2 \pm 0.8	Lake Waitaki	5.2 \pm 1.2	3.5 \pm 0.9
Hutt River	20.0 \pm 2.0	6.2 \pm 0.8	Lake Onslow	2.7 \pm 0.5	3.4 \pm 0.6
Grey River	3.4 \pm 0.6	6.0 \pm 0.7	Poolburn Reservoir	2.3 \pm 0.5	2.8 \pm 0.6
Pomahaka River	6.8 \pm 1.2	6.0 \pm 1.4	North Mavora Lake	1.4 \pm 0.3	2.8 \pm 0.6
Rangitikei River	5.7 \pm 0.7	5.9 \pm 0.7	Lake Opuha		2.7 \pm 0.4
Manuherikia River	3.6 \pm 0.8	5.6 \pm 2.1	Lake Heron	2.6 \pm 0.7	2.6 \pm 0.9
Ashburton River	4.2 \pm 0.8	5.5 \pm 1.1	Flaxy Lake	1.5 \pm 0.4	2.4 \pm 0.7
Ohau Canal	1.1 \pm 0.6	5.4 \pm 2.1	Manorburn Reservoir	0.5 \pm 0.1	2.4 \pm 0.5
Hawea River	1.9 \pm 0.5	5.0 \pm 1.3	Lake Tutira	3.1 \pm 0.2	2.3 \pm 0.4
Tekapo River	2.4 \pm 0.5	4.9 \pm 0.7	Lake Rotoroa	1.0 \pm 0.2	2.3 \pm 0.5
Utuhina Stream	2.3 \pm 1.4	4.3 \pm 1.3	Lake Karapiro	4.8 \pm 0.7	2.3 \pm 0.5
Buller River	5.1 \pm 0.7	4.3 \pm 0.5	Lake Pearson	1.7 \pm 0.6	2.3 \pm 0.4
Tarawera River	5.0 \pm 1.2	4.2 \pm 0.9	Lake Rotoehu	2.3 \pm 0.6	2.2 \pm 0.8

50 years after K. R. Allen's pioneering research on the trout fishery (Allen 1951), showed that trout populations were only 1% of their former size, but no specific cause could be found for this decline (Jellyman *et al.* 2000).

Back-country rivers

The standard picture of New Zealand trout fishing promoted overseas is of an angler fishing some pristine back-country river. Usually surrounded by native forest or tussock grassland in hilly or mountainous country, with steeply descending sparkling clear waters, these rivers produce large trout by international standards. Access to these rivers is improving with more use of 4WD vehicles and helicopters, while interest in fishing them continues to increase.

Management of these fisheries is largely related to angler use and satisfaction. While these fisheries can be affected by natural events such as floods, their catchments are usually in lands that are either protected or farmed extensively, and have thus experienced few anthropogenic changes. Jellyman and Graynoth (1994) provide a good background summary of back-country fisheries.

Many of these rivers have very low usage, being fished by only a few anglers each year. On some rivers, however, increasing pressure has altered perceptions by different groups of anglers of appropriate carrying capacities. As harvest is now minimal, with high proportions of fish caught being released (Walrond and Hayes 1999), there are few traditional options available to the fishery manager to manage the fishery using bag limits, season lengths, and restrictions on fishing methods. In some areas these rivers are for fly fishing only, but as most anglers in such waters use fly fishing techniques this has little effect upon fishing pressure. In Otago and Nelson, the increasing proportions of overseas users have raised concerns from locals, especially as these fisheries are very sensitive to overuse. In a study looking specifically at whether encounters between anglers were rising above acceptable thresholds, Nelson anglers were found to have both lower encounter rates and lower tolerances than Otago back-country anglers (Walrond 2002). On some Otago back-country rivers, however, encounter rates are close to exceeding those preferred or tolerated by most anglers. A comparison between the "catchability" of brown trout in one accessible and one inaccessible back-country river in Nelson confirmed that "naïve" fish were more likely to be caught quickly, with less sophisticated trout flies, than those from the more frequently fished river (Young and Hayes 2002). This study also showed trout were able to learn to avoid catching the more basic flies, and became less visible and harder to catch over a period of three days, confirming that angler pressure can affect the catchability of trout. Thus, every other angler fishing these waters becomes a potential competitor for

the fishing opportunity. If angling thresholds are exceeded, the perceived quality of fishing declines. Given the attraction and value of these back-country fisheries, some Councils are contemplating regulations to restrict the number of angling days in particularly sensitive or overused waters. This is controversial, however, as rationing can affect different user groups in different ways.

LAKE FISHERIES

Like all moist temperate areas with an active tectonic history, New Zealand has large numbers of lakes, many of which provide high quality sports fishing (Table 1). The most important fisheries are found in the central North Island, with Lakes Taupo, Rotorua, Rotoiti, Okataina and Tarawera being the largest. In the South Island, the largest of the southern lakes (e.g., Wanaka, Hawea, Wakatipu, Te Anau, and Manapouri) are also important. Hydroelectric reservoirs such as Benmore, Dunstan, and those on the Waikato River, also attract large numbers of anglers each year. There are 60 lakes which receive approximately 1000 angler visits or more per year, and another 78 which receive at least 100 visits each per year, with an estimated annual total of 700,000 visits in 2001/02. The most popular fishery is Lake Taupo (Maclean *et al.* 2002). Of the more than 200 lakes identified in a recent survey (Unwin and Image 2003), 23% have been created principally for hydro-power generation, and a further 24% (including many of the largest), are subject to some degree of artificial control.

Lake fisheries provide a mixture of rainbow and brown trout fishing, although Chinook salmon are found in a few South Island lakes. With regard to trout ecology, the best studied lakes are Coleridge (Graynoth 1999) in the South Island, and Taupo and the Rotorua lakes (e.g., Dedual 1996) in the North Island. Most lakes rely on their inflowing tributaries for recruitment of stocks, and no artificial stocking is required. In most lakes, rainbow trout fry tend to move out of the spawning tributaries and into the lake soon after emergence, although Lake Taupo—where many juveniles remain in the tributaries for up to 18 months—is a notable exception. In Lake Coleridge rainbow trout and Chinook salmon moved into the littoral zone in spring and summer, whereas juvenile brown trout stayed in the tributaries for 1–2 years (Graynoth 1999). A similar pattern has been recorded in Lake Alexandrina (Hayes 1988a; Hayes 1995), with rainbows that stayed in the stream for longest exhibiting the highest survival. However, because most fry (91%) moved to the lake on or very soon after emergence, over half (55%) of the adult rainbow trout that survived to maturity began their lake residence as early migrant fry. Lake environments appear to be less hospitable to early migrants, but it is thought that complex habitats that provide shelter and food

were needed to enhance the survival of these very small (ca. 20 mm) fish (Hayes 1988a). This has implications for managing fluctuations in the levels of artificially controlled lakes, as large changes in lake levels, especially when the fry are small, are likely to be disadvantageous to them.

Where spawning areas are limited, competition between species can occur. Rainbow trout can have an adverse effect on brown trout, since the rainbow trout spawn later, and their redds may be superimposed on brown trout redds (Hayes 1987). However brown trout fry hatch earlier than rainbows—the brown trout fry thus have a size advantage over the rainbow fry when they emerge and are dominant in aggressive encounters (Hayes 1988b). Rainbow trout are more easily caught than brown trout, especially by trolling—a popular form of sports fishing on lakes—so it is important for managers to ensure one species is not adversely affecting the other to an extent that the production of the fishery is compromised.

A high proportion of lake fishing occurs in Taupo and the Rotorua lakes (Table 1). Rainbow trout dominate numerically. Lake Taupo rainbow trout are sustained by natural spawning in the many inflowing streams. Juveniles remaining in the streams reach about 100–200 mm before migrating, and contribute most to the adult population in the lake, with smaller migrants having a high mortality (Maclean *et al.* 2002). Mature adults spawn at three years of age, at a mean length of about 540 mm. Less than 30% survive to spawn more than once, to contribute to the lake fishery in successive years. Bag limits are an important regulatory control on angler harvest, which has been estimated to be as much as 50% of total annual production, a level considered too great to sustain the fishery (Maclean *et al.* 2002).

Environmental factors affecting the production of the Lake Taupo fishery include flooding at times of peak fry emergence. When combined with volcanic eruptions, which result in large quantities of ash being discharged to the Tongariro River in particular, such floods can cause significant fry mortalities. Blocking of access to spawning grounds through log jams in spawning tributaries or poorly designed culverts, and hydropower generation requirements are also threats. Increases in nitrate concentrations in the lake, following recent intensification of agriculture, will lead to further long-term changes in lake productivity. Smelt are the main food source for trout, so changes in nutrient status will change their productivity, and hence affect the trout population. A better understanding of the pathways by which agricultural nutrients such as nitrate and phosphorus pass from soils to watercourses is required to ensure that nutrient enrichment remains within acceptable levels. The recent popularity of lake margins for residential development also threatens the pristine state of many New Zealand lakes, with increases in discharges

of contaminated stormwater and sewage.

In contrast to Lake Taupo, several lake fisheries in the Rotorua area are reliant on regular releases of juvenile hatchery-reared trout. Lake Rotorua has tributaries (e.g., Ngongotaha Stream) that provide adequate natural recruitment, but other large lakes do not. Lake Tarawera, for example, has only small inflowing streams, but a high angler harvest. Release numbers and timing vary and are monitored closely to optimise angler satisfaction, which is also measured annually.

Lakes created in association with hydro-power development have also provided opportunities for sports fishing (Table 1), although these are at the expense of river fisheries. Lake Dunstan on the Clutha River was created with the construction of the Clyde dam. One arm of the lake, into which the Clutha River flows, provides a productive fishery, although the lake weed *Lagarosiphon major* must be controlled to ensure it does not invade large areas of the lake. Catch rates and fish sizes in the lake are in the middle of the range of nine and sixteen lakes respectively (Scott 1996).

Monitoring trout populations in the largest and most important lakes is a difficult task, requiring considerable resources. Estimates of the population of trout in Lake Taupo, using sonar equipment, have revealed spring populations ranging from 68,000 to 205,000 adult (> 350 mm) trout over a fifteen-year period beginning in 1988 (Maclean *et al.* 2002). Sonar methods are best suited to deep, single species lakes like Taupo, but may not be as suitable for estimating trout numbers in other more shallow lakes or those with irregular bathymetry or substrate irregularities, such as those in the South Island.

Studies of brown trout in Lake Brunner over the last decade suggest an increase in average size (Fig. 41.1), which—together with an increase in weight and abundance—has occurred without any change in the relatively poor condition of the population. Fishing competition results show a similar increase up to the 2002 season, after which a decrease in length, weight and condition is apparent. However, the data are insufficient to determine whether an unnatural change is occurring in the lake, or whether the observed trends simply reflect naturally occurring cycles. Food availability has been suggested as a contributing factor, but spawning, growth, and external inputs to the lake could also be important.

SALMON FISHERIES

Acclimatisation and distribution

Chinook salmon are the largest members of a group of fishes known as Pacific salmonids (family Salmonidae), and are native to the North Pacific rim from California to northern Japan (Healey 1991). They are a cool-water species, with their latitudinal distribution strongly related

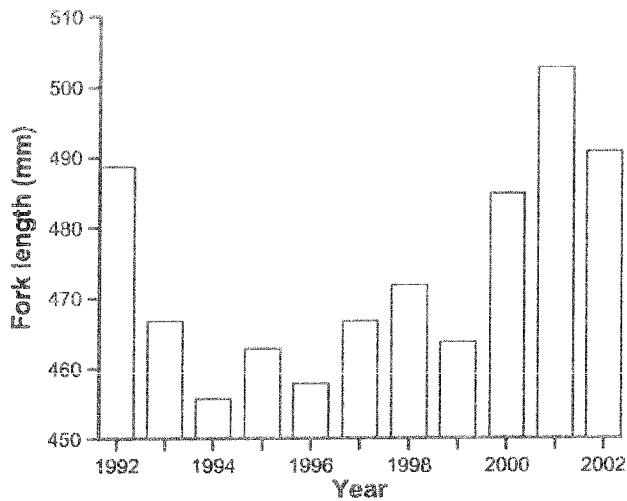


Figure 41.1 Mean fork length of brown trout caught during the annual Lake Brunner fishing competition, 1992–2002.

to ocean temperatures. They are strongly anadromous, philopatric, and semelparous, with spawning and juvenile rearing occurring in fresh water, and the remainder of the life cycle at sea.

The most important New Zealand stocks are those of the large, glacier-fed, braided east coast rivers draining the South Island main divide, primarily the Waimakariri, Rakaia, Rangitata and Waitaki (Quinn and Unwin 1993), which support annual spawning populations of up to 20,000 fish. Moderate runs (up to ~1,000 fish) occur in the Clutha River, having been greatly reduced in number after completion of the Roxburgh Dam in 1956, and in smaller east coast rivers such as the Waiau, Hurunui and Opihi. Sporadic runs are recorded well outside this range (McDowall 1990), particularly on the South Island west coast, where spawning populations of up to several hundred fish return to the tributaries of Lakes Paringa and Mapourika. Since 1987, there have also been occasional runs in Southland and the Wairarapa, following escapes of commercially farmed salmon from sea cages on Stewart Island and the Marlborough Sounds.

Relatively little is known about the oceanic phase of the life cycle, other than what can be inferred from by-catch records for commercial trawlers and gill netters. Adults do not travel far offshore, generally being captured within 50 km of the east coast of the South Island (Unwin and James 1998), where they feed opportunistically on sprat (*Sprattus muelleri*), red krill (*Munida gregaria*) and juvenile hoki (*Macruronus novaezealandiae*). Although these records are strongly influenced by the distribution of commercial fishing efforts and are not necessarily representative, a

relatively coastal marine distribution is consistent with the behaviour of present-day Sacramento River Chinook.

Landlocked Chinook occur in the Clutha River source lakes and downstream hydro lakes, and in Lake Coleridge, where they provide small but useful recreational fisheries. These fish typically mature at around 1 kg, compared to 5–10 kg for their anadromous counterparts, reflecting the relatively slow growth that occurs in fresh water.

Sports fishery

The salmon sports fishery is the backbone of the recreational fishery in the North Canterbury Region, and is also important in Central South Island. Collectively the four main fisheries accounted for 164,000 angler-days² in the 1994/95 angling season, representing 21% of the total effort devoted to river fishing in New Zealand (Unwin and Brown 1998), and 110,000 angler days in 2001/02 (Table 41.1). The Waimakariri River is one of the two most heavily fished rivers in the country, its usage being equalled or exceeded only by the Mataura River. During peak salmon migration angler densities on all four rivers can be high, particularly at their mouths, where the salmon are in prime condition. Although there is potential overcrowding and a reduced quality of experience, salmon anglers are less affected by this factor than trout anglers (Martinson and Shelby 1992), as they are generally more interested in catching fish than in a remote outdoor experience.

Since 1993, FGNZ has monitored annual trends in stock abundance by using telephone sample surveys to estimate angler harvest, and repeated aerial surveys of key spawning waters (West and Goode 1987) to estimate spawning escapement. These data show considerable annual variation in run strength, with highs and lows tending to coincide in all four rivers. However, angler harvest rates vary relatively little, and show no obvious relation to run size (Table 41.2). Spawning runs since 2000 have been unusually poor, the 2001 season generally being regarded as the worst ever. With some evidence of a slight improvement in 2003, runs over the next few years will be monitored with great interest.

Biology and population dynamics

New Zealand Chinook salmon have been the subject of numerous research studies, with over 60 papers published in the scientific literature. This research was initially centred on Glenariffe Stream, in the Rakaia headwaters, where spawning populations have been monitored annually since 1965 (Quinn and Unwin 1993), and focussed on basic life history studies and population dynamics (e.g., Unwin 1986). A second phase began around 1980, when increasing interest in rivers such as

² An angler-day is one angler fishing on one day, irrespective of the number of hours fished.

Table 41.2 Annual runs of Chinook salmon in the four main salmon rivers, 1993–2003. For the first three rivers the table shows the total run and angler catch (in thousands of fish), and the angler catch as a percentage of the total. For the Waitaki River, data shown are the estimated Hakataramea River spawning run (number of fish), and the catch (thousands of fish) for Central South Island anglers.

Year	Waimakariri River		Rakaia River		Rangitata River		Waitaki River	
	Total run	Angler catch	Total run	Angler catch	Total run	Angler catch	Hakataramea spawning run	Angler catch
1993	2.4	1.1 (45%)	3.9	1.1 (26%)			1173	
1994	4.1	1.6 (39%)	19.9	7.9 (40%)	11.4	3.3 (28%)	6301	3.4
1995	10.4	4.4 (42%)	11.6	3.1 (27%)	8.5	3.2 (38%)	1183	7.1
1996	13.5	6.0 (45%)	22.6	9.0 (40%)	16.6	5.4 (33%)	5962	2.2
1997	9.1	3.9 (43%)	18.5	8.5 (46%)	15.8	5.8 (37%)	934	3.1
1998	5.8	2.8 (48%)	6.1	2.6 (42%)	6.4	2.6 (40%)	367	2.3
1999	7.1	4.7 (66%)	6.5	2.6 (39%)	7.8	3.5 (45%)	829	1.9
2000	3.5	2.6 (72%)	4.9	3.0 (61%)	4.4	2.1 (49%)	282	1.1
2001	1.9	1.1 (56%)	1.6	0.8 (53%)	1.0	0.3 (32%)	12	0.5
2002	3.3	1.1 (34%)	2.5	0.6 (23%)	1.0	0.2 (24%)	79	0.6
2003	3.6	1.8 (49%)	3.1	1.7 (55%)	1.6	0.7 (45%)	12	0.8

the Rakaia and Waitaki for irrigation or hydroelectric development created a need for better information on mainstem habitat usage (by adults as well as juveniles), so as to determine minimum flow requirements. A third phase, during the 1990s, was based on the increasing recognition that, as a recently introduced species, New Zealand Chinook were subject to ongoing evolutionary change as they adapted to local habitats. This research exploited the fact that, as an introduced population derived from a well-known and clearly delimited source stock, New Zealand Chinook provided a natural laboratory for studying the patterns, rates, and processes associated with local adaptation. From 1994 to 2000, a series of experiments were conducted to determine whether or not, in the 90 years (c. 30 generations) since their introduction, evolutionary change had been rapid enough to allow locally adapted stocks to develop in different New Zealand rivers (see Quinn *et al.* 2001 and references therein). A key finding was that, when salmon of Glenariffe and Waitaki origin were released into Glenariffe Stream, survival rates for Glenariffe fish were 50% higher than for Waitaki fish, providing conclusive evidence of local adaptation (Unwin *et al.* 2003).

Salmon spawning habitat in the main salmon rivers is primarily confined to the upper reaches, where stable spring-fed streams draining the stabilised margins of the wider flood plain provide ideal spawning gravels. These waters (which include Glenariffe Stream) have traditionally accounted for over 90% of the spawning population in the Rakaia River, and 70% in the Rangitata River. The Waimakariri spawning tributaries are more diverse, ranging

from small spring-fed streams similar to their Rakaia and Rangitata counterparts to larger sub-tributaries such as Winding Creek and Slovens Creek. One such tributary, in the upper Poulter River, is noteworthy in that it is one of the few places in New Zealand where Chinook spawn in a forested stream similar to most North American habitats (Unwin *et al.* 2000). Salmon returning to the Waitaki River spawn in the Hakataramea River, but also use braids of the main stem to a greater degree than in the other rivers.

There is considerable variation among populations in the timing of the upstream spawning migration (Quinn *et al.* 2001), with the earliest runs (in the Rangitata River) peaking about seven weeks earlier than the latest (in the Waitaki River). However, the median date of entry to the headwater spawning tributaries varies little between the three rivers, occurring in late April or early May (Quinn and Unwin 1993). The latest population to spawn is that of the upper Poulter River, where peak spawning occurs in late May (Unwin *et al.* 2000). Redd construction takes several days, during which each female buries an average of 4500 eggs to a depth of 30–40 cm. Females usually die within a week of spawning, guarding their redd for as long as possible to discourage other females from using the same site. Males live longer, and dominant fish may mate with multiple females over several weeks.

Juvenile Chinook salmon display one of two contrasting life history strategies during their first year of life. These strategies are known as “ocean-type”, in which most of the first year is spent in the ocean, after a downriver migration lasting 3–6 months, and “stream-type”, in which juveniles spend all of their first year of life in fresh water and do not

enter the ocean until their second year. Within their native range, these variants are usually associated with two parallel adult life history types, distinguished by the seasonal timing of their upstream migration, and known as “spring-run” and “fall-run”, respectively (Healey 1991). Spring-run Chinook, and stream-type juveniles, tend to be associated with larger river systems, and have a more northern distribution. Fall-run Chinook, and ocean-type fry, are more commonly associated with smaller coastal rivers towards the southern end of their range.

Both life history variants occur in New Zealand, although the proportion of stream-type adults varies widely between populations, ranging from 29% in the Rangitata River to nearly 100% in the Poulter River (Unwin *et al.* 2000). The ancestral New Zealand stock were fall-run (McDowall 1994b), and the development of the stream-type variant here appears to be an adaptive response to local environmental conditions, which favour an extended period of freshwater residence (Unwin 1997b; Unwin and Glova 1997). In both variants, newly-hatched fry emerge from their redds in spring, about 21 weeks post-spawning, and continue to do so for up to three months (August to October, Unwin 1986). During peak emergence, in mid-September, numbers may be as high as several tens of thousands per day. In Glenariffe Stream up to 99 % of fry disperse downstream within 24 hours of emergence, at least partly in response to intense competition for rearing habitat, and take up residence in the upper Rakaia River.

Downstream dispersal of ocean-type fry in the Rakaia mainstem is well understood in qualitative terms. Fry gradually move downriver over 2 to 3 months, growing from about 33 mm fork length (FL) at emergence to 80–100 mm FL at seawater entry (Hopkins and Unwin 1987). They are continually replenished by new recruits emerging from the headwater spawning streams as the older members of each cohort leave the river mouth. They feed actively and opportunistically, taking an unusually high proportion of terrestrial invertebrates compared to brown trout fry and native fish species (Sagar and Eldon 1983). Fry first appear above the river mouth in mid October, and are present in the lower river until mid to late summer. Habitat utilisation changes with size and age. Fry of less than 55 mm FL are highly territorial, and in the Rakaia River are primarily found along the channel margins of shallow, slow-flowing water containing cover in the form of submerged riparian growth, debris accumulations, and substrate irregularities. Larger fry become less territorial and tend to school, preferring somewhat deeper water. However, very little is known about habitat utilisation and downstream dispersal for

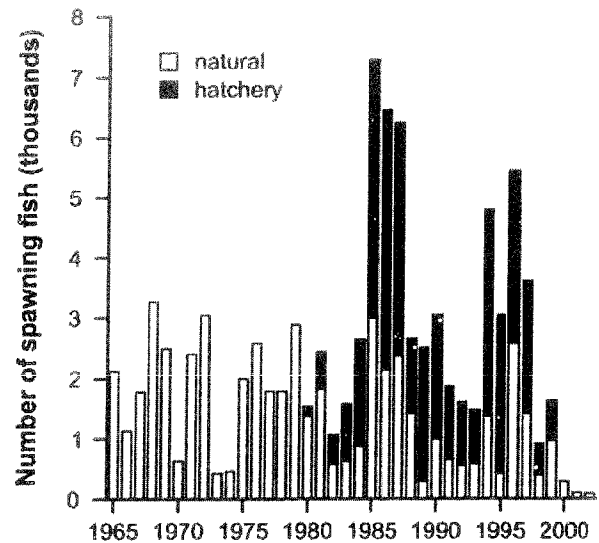


Figure 41.2 Chinook salmon spawning runs in Glenariffe Stream, 1965–2002, by origin (natural or hatchery) and year.

stream-type fry, which have been encountered only rarely in all studies to date (e.g., Hopkins and Unwin 1987; Sagar and Eldon 1983).

In Glenariffe Stream, survival from egg deposition to emergence averages 48% and varies remarkably little from year to year (Unwin 1986). By contrast, adult returns to Glenariffe Stream (Fig. 41.2) are marked by extreme fluctuations in fry to adult survival, which varies from 0.01% to 1.2%, with a long-term average of 0.08% (Unwin 1997a). Attempts to fit stock-recruitment curves to the Glenariffe data have been unsuccessful, possibly because no such relationship exists, with runs simply varying stochastically from season to season, but more probably because the underlying relationship is obscured by environmental noise. Much of this noise appears to be of marine origin, as is evidenced by the tendency for adult runs in all four main salmon-producing rivers to follow parallel trends (Table 41.2). Further evidence for a strong marine influence on survival is a high correlation between annual survival rates for wild and hatchery fish of Glenariffe origin (Unwin 1997a). Because fish of hatchery origin are shielded from most causes of natural mortality until the end of their first year of life, when they are released from the hatchery and enter the ocean, this correlation strongly implies a common marine influence.

Hatchery supplementation

Beginning in the late 1970s, considerable effort went into developing a commercial ocean ranching³ industry

³ The practice of raising young salmon in a hatchery for some or all of their first year of life, and releasing them to the ocean in the hope that enough adults will survive and return to the point of release to generate a viable business.

based on Chinook salmon. Between 1977 and 1997, 46 million hatchery salmon were released into South Island rivers, peaking at 6.6 million in 1987. Most (71%) were associated with commercial farms, with a further 25% released as part of MAF or NIWA research programmes. Angler-funded programmes aimed at enhancing the sports fishery made up 4% of the total. Releases took place from 55 locations in 21 catchments, with roughly 10% of the fish released being marked with coded-wire micro-tags, enabling returning adults to be identified by origin and point of release, and allowing information such as survival rates and age at return to be estimated directly. Survival rates ranged from 0% to 9%, with rates varying greatly between years and release locations, and showing a strong tendency to increase with increasing age (and hence size) at release (Fig. 41.3). However, their distribution was highly skewed, with the median (0.33%) less than half the mean (0.79%). These returns proved insufficient to sustain a viable ocean-ranching industry, and no such ventures lasted beyond 1990.

Returns of hatchery fish had a significant impact on the recreational salmon fishery, with at least 42,000 fish taken by anglers as far away as Taranaki. In Glenariffe Stream hatchery fish had a marked effect on spawning runs, making up 59–88% of returns from 1983 to 1991 (Fig. 41.2). Over the peak of the ocean ranching boom (1985–1987) hatchery returns contributed over 12,000 extra fish to Glenariffe Stream (Unwin and Glova 1997). These fish, particularly males, tended to mature one year earlier than their natural counterparts, and also tended to be smaller than natural fish of the same age. In Glenariffe Stream these effects combined to produce a decline in both the mean age and mean length of male spawners from 1967–1994.

A consistent pattern in many hatchery programmes was for a period of early successes to be followed by a steady decline in survival rates. Elsewhere in the world there is a lively debate as to whether hatchery releases weaken the gene pool by husbanding large numbers of young fish in a

protected hatchery environment, so that undesirable genes that would otherwise be weeded out by natural selection are allowed to propagate through to subsequent generations, and a similar effect cannot be ruled out here. It is also unclear whether hatchery releases actually increased the total number of salmon returning to freshwater, or merely replaced fish of natural origin with those of hatchery origin. Increases in spawning runs in Glenariffe Stream appear to have been accompanied by a decline in spawning populations elsewhere in the Rakaia (Unwin 1997a), as might be expected if production at some critical phase of the life cycle were density-dependent.

A more fundamental problem is that, to achieve any success at all, hatchery programmes were forced to adopt release strategies that were, by North American standards, unconventional. For fall-run Chinook, a standard practice is to release fish of 5–10 g (or “90-day smolts”), consistent with the seaward migration pattern of ocean-type fry. Here this strategy failed: of 46 such releases between 1977 and 1981, the median survival rate was 0.006%. This led to hatcheries holding fish for up to one year, typically releasing them at 8–12 months of age and weights of up to 150 g, and boosting the median survival rate to 0.34%. However, these returns were remarkably poor when compared with 0.079% for naturally produced fry migrating downstream within 24 hours of emergence, at a weight of 0.3 g. More than any other single factor, these continued poor returns, over nearly 20 years, severely limit the viability of hatchery supplementation as a management tool for the salmon fishery.

COARSE FISHERIES

Coarse sports fish are perch, tench and—in the Auckland Waikato region only—rudd (*Scardinius erythrophthalmus*). Coarse fishing is the most popular form of fishing in the UK and parts of Europe, undertaken mainly in the still or slow-moving water bodies favoured by these species, using bread or live bait such as worms, and keeping fish for release at the end of the day. New Zealand has a limited tradition of coarse fishing, although perch and tench have been part of the freshwater fauna in some parts of the country since the 1860s. Recent immigration has rekindled interest, with coarse angling clubs in the Canterbury, Waikato, and Manawatu regions, and a national body (the Federation of Coarse Anglers) formed to advocate coarse angling interests. Conflict has arisen over a desire by the coarse angling fraternity to enhance coarse angling opportunities in regions such as Nelson, which, until recently, lacked these species. Authorities such as the

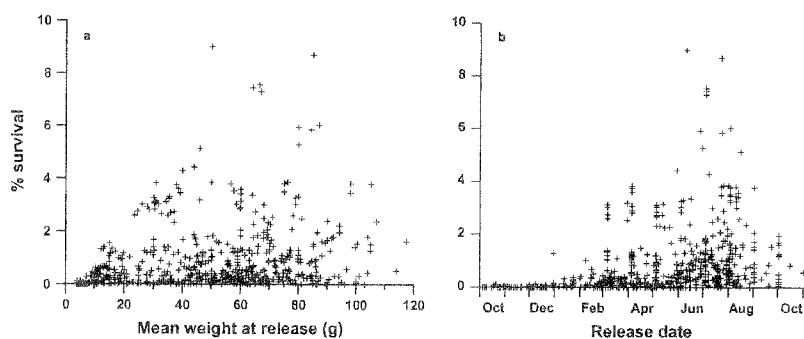


Figure 41.3 Survival to adulthood for 660 groups of hatchery-reared Chinook salmon released from 55 South Island locations between 1977 and 1997 vs. mean weight at release (left), and month of release (right).

Department of Conservation and Fish and Game Councils have seldom been as keen to authorise new species introductions. It appears some coarse anglers have taken matters into their own hands and liberated coarse fish into new areas without authorisation, sometimes with pest fish such as *Gambusia* and koi carp. Controlling unauthorised spread of coarse or pest fish, while facilitating coarse fisheries where appropriate, has proven a challenge. Little research has been undertaken on coarse fisheries, an exception being a study of the impacts of perch on native fish in the lower Taieri catchment (Goldsmith 2003).

WHITEBAIT FISHERY

Whitebait, the juveniles of five *Galaxias* species and common smelt, are a prized New Zealand delicacy. Although many New Zealanders have sampled the tiny and highly sought-after fish—arguably the most valuable fishery in the country on a cost per weight basis—few understand anything about where they come from. Their management is also unique, being the only fishery managed exclusively by the Department of Conservation. They are managed as a recreational fishery, with restrictions on season and fishing methods, but there are no limits on the take, and catches may be sold by anyone without requiring any permit or quota.

Maori were first to appreciate the delicacy, exploiting the runs of fish using fine nets and manuka branches immersed into the water. In the early days of settlement whitebait catches were reputedly so large they were sometimes ploughed into the ground as fertiliser. While this may have reflected a lack of refrigeration and the well-known variability in whitebait runs, the size of runs in the early days were sometimes vast. Catches declined in many parts of the country during the 20th century, but not until the investigations by Bob McDowall and his colleagues on the West Coast during the 1960s was the biology of these species understood (McDowall 1965). Prior to this work it was not even recognised as a multi-species fishery.

Reductions in the runs are now considered to reflect the massive changes wrought on freshwater fishery habitats since European settlement, particularly in the loss of lowland wetland habitats for inanga (*G. maculatus*), the most abundant species. Estimates vary as to the extent of wetland loss but undoubtedly it was severe, probably averaging 90%, and possibly as high as 98% in lowland parts of the Tasman Region that have been investigated in detail. Attempts are now being made to develop whitebait farming, in a venture near Raglan (Mitchell 2002). A major difficulty is housing large numbers of developing juveniles, and, if ocean ranching, having them return to their point of origin.

The fishery is not managed intensively. DoC's main

priorities are the protection of known inanga spawning areas, where suitable vegetation coincides with water levels at Mean High Water Spring tides, and compliance with regulations. Two sets of regulations exist, one for the West Coast and one for the rest of the country. These differ little except for the length and timing of the season, and have changed little in recent years. Recent attempts to change the West Coast season timing to reduce the catch of the threatened giant kokopu (*G. argenteus*) were thwarted by determined resistance from whitebait fishers to any change in the status quo.

BIODIVERSITY AND SPORTS FISHERIES

Over the last decade there has been a remarkable development in attitudes towards New Zealand's indigenous flora and fauna, epitomised by the extent to which "preserving biodiversity" has now acquired the status of a political catch phrase. One consequence is that native fish species, after many years of being virtually ignored by the general public, are finally being given due recognition as a resource to be protected and conserved. However, this has also created a tendency for native species to be considered as inherently more important than introduced species, to the extent that the prevalence of (and management for) exotic sports fish in New Zealand is sometimes considered inappropriate. While there is little doubt that successful introductions of any new species will inevitably lead to changes in the pre-existing ecological communities, it is also true that exotic species, by virtue of creating a highly valued recreational resource, can acquire a conservation value in their own right. One of the challenges which has faced FGNZ over this period has been to continue to promote the needs of the sports fishery, while also recognising the need to protect indigenous biodiversity.

The conservation value of the salmonid fishery is clearly demonstrated by the many successes of FGNZ in advocating the protection of freshwater habitats, and hence (implicitly) for native freshwater biodiversity. Trout, for example, demand more water, of higher quality, than do any native species. Impediments to migratory trout and salmon from dams are often also impediments to native fish movement. Arguments to retain river flows or water quality in rivers are usually more readily made where salmonid fisheries exist than where they do not. Protection of freshwaters in general, and angling waters in particular, arises from the substantial political influence of some 150,000 freshwater anglers. By comparison with other countries, like Australia, New Zealand gives greater recognition, both in society and in legislation, to the intrinsic and other instream values of freshwater environments rather than just the consumptive use of water. FGNZ independence from direct government control, together

Table 41.3. River fisheries (for trout unless otherwise stated) for which FGNZ has sought permanent protection via a Water Conservation Order. Regional plans under the Resource Management Act are also able to recognise and provide protection for river fisheries.

River	Status
Rakaia (salmon)	Gazetted 1988
Ahuriri	Gazetted 1990
Grey	Gazetted 1991
Rangitikei	Gazetted 1993
Mataura	Gazetted 1997
Kawarau and tributaries	Gazetted 1997
Buller and tributaries	Gazetted 2001
Motueka	Recommended
Mohaka	Recommended
Rangitata (salmon)	Under appeal

with their statutory responsibilities, enables them to be a forthright advocate for improved legislation and environmental administration of freshwaters. Of the 23 Water Conservation Orders sought since the introduction of the “Wild and Scenic Rivers” legislation in 1981, seventeen have been sought by FGNZ or its predecessor Acclimatisation Societies, of which ten have proceeded to at least the stage of a draft Order (Table 41.3). While these rivers also have values for native fish, it is doubtful if any of these would have been sufficient to justify the resource as nationally outstanding, and hence deserving of protection via a Conservation Order.

Historically, salmonid fishery management included several actions that were potentially negative for native species. For many years, Acclimatisation Societies paid anglers a bounty to kill eels, in the misguided belief that this would improve the trout fishery (McDowall 1994a). However, this bounty probably had a negligible effect on eel populations, which have experienced much greater reductions since the advent of the commercial eel fishery in the 1960s. Moreover, fishery managers now recognise that trout fisheries that lack eel populations (such as those above dams or dewatered reaches) often have large numbers of small fish, suggesting that healthy eel populations can potentially benefit anglers by preying on juveniles and increasing the average size of harvested fish (cf. Burnet 1968). Many former Societies also advocated killing native shags, believing them to be another significant trout predator, although their true effect on trout populations—if any—has been little illuminated by good research (McDowall 1994a).

There may also be secondary effects on native biodiversity due to trout fishery management. For example, native koaro populations in the central North

Island lakes were significantly reduced by trout predation. After their main food source was reduced, trout growth in these lakes was seriously hindered, leading to the introduction of native smelt, out of their natural range, as an alternative trout food source. This food source is now self-sustaining, and has been spectacularly successful for the trout fishery. The effects of smelt introductions on native fish have been little investigated, however, except that the unauthorised introduction of smelt to Lake Rotopounamu near Turangi appears to have extinguished the koaro population in the absence of salmonids (Rowe 1993).

Some of the strongest evidence for direct negative interactions between trout and native fish comes from studies of species distributions, which have often shown little or no overlap between exotics and natives, and suggest that many native fish populations are isolated and fragmentary. Recent investigations in Otago have shown little overlap in distribution between brown trout and giant kokopu (*Galaxias argenteus*) (David *et al.* 2002), providing circumstantial evidence of negative effects of trout on giant kokopu populations. Likewise, shortjaw kokopu (*G. postvectis*) are often considered one of the native species most susceptible to predation by trout, because their distributions rarely overlap (e.g., McIntosh 2000). Brown trout are an efficient predator on smaller fish such as non-diadromous galaxiids, and their presence has clearly been a significant factor in the demise of some Otago populations (Townsend and Crowl 1991).

However, further research is needed to ensure that these results are interpreted correctly. For example, our present knowledge of the distribution of shortjaw kokopu may be confounded by differences in sampling methods. Concerns about the effects of trout on shortjaw kokopu have been based primarily on observations using electric fishing machines, but more recent surveys using spotlights (which appear better suited to catching shortjaws) have shown that the two species frequently coexist, particularly when the trout are mostly similar in size to shortjaws. It may even be that shortjaws are better adapted to some rivers than trout. Interactions between koaro (*G. brevipinnis*) and trout, in the Canterbury high country, could also be interpreted in the same way (McIntosh 2000). Koaro are preyed on and out-competed by larger brown or rainbow trout in some areas, but survive in other steeper, unstable parts of the same streams or with smaller trout. There is, arguably, a form of co-existence between these two species, in which both appear to be maintaining self-sustaining populations, despite the evidence of trout predation or competition. This can also be seen in many other parts of the country, where fisheries surveys and database records show viable populations of koaro, and sometimes other galaxiid species, surviving in good numbers in locations inaccessible to trout. Similarly, in Otago, giant kokopu are more

abundant in some areas than trout (David *et al.* 2002), suggesting some niche separation. Further research would be required to determine whether the long-term survival of these populations is seriously threatened by trout alone, or whether other habitat changes may be affecting one or both of these species. The presence of isolated and fragmentary native fish populations elsewhere in New Zealand may also involve factors such as habitat alterations, dams, or the transfer of water between catchments, and possibly the presence of predatory eels, koaro, or shags (McDowall and Allibone 1994). Apparently isolated populations of some migratory native fish may be sustained by nearby viable population sources. Finally, there have been instances where trout have had beneficial effects on native species, such as in Northland's Kaiwi Lakes, where rainbow trout appear to help control the exotic pest fish *Gambusia*, and thus assist the survival of the threatened dwarf inanga (*G. gracilis*) population (Rowe in press).

Pragmatic agreement has been reached between DoC, as the primary advocate for indigenous biodiversity protection, and FGNZ, as the advocate for sports fisheries, over the issue of biodiversity and sports fish. FGNZ has agreed, by way of binding National Policy, that sports fish should not be introduced into natural waterways that lack those species. On the other hand, through the National Parks and Reserves Policy and regional Conservation Management Strategies, DoC has agreed to recognise and allow for the management of sports fisheries where these presently exist, including restocking as appropriate. In some areas, such as Otago, FGNZ allows research into the effects of removing trout from small streams that hold rare native fish. These agreements enable both parties to focus on common concerns such as water abstraction, pollution and freshwater biosecurity.

Nonetheless, tensions are likely to remain about the appropriate management of salmonid and indigenous fish species where these species occur in the same environment. Research into the ecological relationship between native and introduced species will continue, but—for the foreseeable future at least—freshwater environments will continue to be managed to provide for sports fisheries as well as for indigenous biodiversity. In most instances there is relatively little conflict between these two objectives, particularly when compared with the effects of other activities on the freshwater environment. For further perspectives on interactions between native and sports fish, see McDowall (1990), Allibone and McIntosh (1999) and Hayes and Hill (in press).

CURRENT ISSUES

Major challenges remain in sports fishery management in New Zealand. While the focus during the early days of settlement was mainly on liberating and restocking trout,

or trying to establish more salmon fisheries, for more than 50 years habitat protection has been the major focus for management. “Look after the habitat and the fish will look after themselves” has been the catch cry. Acclimatisation Societies pushed for passage of the “Wild and Scenic Rivers” legislation as an amendment to the Water and Soil Conservation Act in the early 1980s, which has since been incorporated into the Resource Management Act 1991. This legislation has enabled recognition of, and provided protection to, trout and salmon fisheries by resource managers (Table 41.3), although it has not been the universal panacea for which fisheries managers and anglers might have hoped. Effort will need to be maintained to ensure the legislation is retained and remains effective.

Active research-based management of freshwater fisheries remains the major priority of Fish and Game Councils and DoC, at least in the case of the Taupo trout fishery. Arguably, the Taupo fishery has the resources to undertake the most comprehensive management of any fishery, freshwater or marine, in the country. Their management attempts to estimate fish production, growth rates and fish numbers; angler harvest and satisfaction; and the underlying environment, in a way that enables fisheries models to predict harvest and apportion it, based on anticipated angling pressure. This approach has enabled the Taupo fishery managers to make an unpopular decision to reduce bag limits from eight to three fish in 1990, yet be able to justify this change. Bag limit regulations set by Fish and Game Councils have generally been reduced over time, partly in response to reduced fish stocks (Maclean *et al.* 2002), but also to reflect angler preferences. Some rivers have had “slot limits” imposed to enhance the survival of larger fish. There is evidence that this has been at least partially successful on the headwaters of the Rangitikei River in the centre of the North Island, where a population of large rainbow trout is intensively fished (Barker *et al.* 2002).

Fish and Game Councils aspire towards managing their fisheries at a similarly high level. Eastern Fish and Game region will continue to use new sonar technology to measure fish numbers in its lakes, while carefully managing the stocking regime to take advantage of growth rates, which are the best in the country.

FUTURE ISSUES

Sports fisheries in New Zealand have developed to their present state due to favourable geology, climate, and a relative low density of human settlement. The geological factors remain unchanged, but changes in human settlement, especially agricultural development, have had significant impacts on sports fisheries over the past 50 years. Declines in the quality of sports fishing, as measured by angler opinion, appear to be most marked in eastern

areas of New Zealand, as a result of past and recent changes in land use and increasing irrigation (Jellyman *et al.* 2003). Measured declines in trout numbers have been documented in a few rivers, such as the Horokiwi Stream (Jellyman *et al.* 2000) and the Pomahaka River (Young and Hayes 1999). Other streams, however, appear to have retained their fishing quality, or even improved—e.g., the Waiau River in Southland (see Table 41.1), where the flow has been increased following settlement of new consents issued under the Resource Management Act 1991 (Rodway 2001).

Agricultural development continues to pose a threat to the maintenance of sports fisheries (e.g., Shearer and Hayes 2003), but research into ways of preventing habitat loss in rivers and protecting water quality is well supported by government funding and should provide mechanisms to reverse current trends. Whether these are adopted remains a challenge for water managers in the 21st century. Identification of precise habitat requirements is also needed, but our understanding of these requirements is progressing (Hayes *et al.* 2000).

Although degradation of sport fish habitat poses the major threat, other issues are also of concern. The foremost of these is the growth in tourism, with New Zealand attracting ever increasing numbers of overseas tourists who come seeking to fish for trout in a wilderness setting. Some headwater and wilderness rivers now receive more angler visits from overseas anglers than from New Zealand anglers, to the extent that angler encounter rates are above desirable levels on some waters (Walrond 2002). The practise of catch and release, while necessary to retain a population of large fish in many headwater streams, has altered the behaviour of brown trout and made them more difficult to catch (Young and Hayes 2002). There are also concerns about the long-term survival of these fish, and about animal ethics (Dedual 1996).

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Chapter 40

Wetland management and restoration

Brian Sorrell, Paula Reeves and Beverley Clarkson

INTRODUCTION

In recent years a wide range of individuals and organisations in New Zealand have taken on the challenge of protecting and restoring wetlands in New Zealand. Wetland management has been a statutory responsibility for Government agencies such as the Department of Conservation, Fish and Game, and Regional and District councils for some time, but the last two decades have also seen an increasing interest by community organisations, iwi and private landowners in taking responsibility for local initiatives in wetland husbandry. Their actions are driven by a body of both local and international scientific research that offers irrefutable evidence of the benefits of the presence of wetlands in the landscape. Wetlands are habitat for desirable fauna such as fish, birds and invertebrates, they trap sediment and nutrient runoff and hence reduce downstream pollution, they prevent erosion of adjacent rivers and hillslopes, and in some cases they may mitigate flood peaks (see Bullock and Acreman (2003) and Chapter 20 for recent assessments of the role of wetlands in the hydrological cycle).

The protection of existing wetlands in the New Zealand landscape and the restoration of degraded systems are further justified by the well-documented destruction and loss of wetland area since European colonisation. A frequently quoted figure is that approximately 90% of the wetland area of New Zealand in 1850—more than 3 million hectares (Cromarty and Scott 1996)—has been lost in the subsequent 150 years. Furthermore, a disproportionately large fraction of the remaining wetland area is in montane and alpine regions, so the relative loss of lowland and coastal wetlands is even greater. Most of the loss has been due to drainage and development for agriculture and to urban development, and the pressure for land-use change remains remarkably strong, given the increased public awareness of the value of wetlands. At the

time of writing, instances of drainage and ploughing of large significant wetlands still occur (e.g., McKinley 2003), and many District and Regional plans offer little protection for smaller, fragmented wetlands on private land. Given that fragmentation into small, isolated units has been a major consequence of past wetland drainage in New Zealand, much of the remaining wetland resources are still under threat.

The protection and restoration of biodiversity in New Zealand is strongly linked to the concept of *natural character*—the provision of environmental conditions similar to those of wetlands in an undisturbed state (e.g., in size, integrity, physical properties and chemistry) that allow a relatively natural community of native species to assemble, with a minimum representation of exotic species.

Amongst freshwater ecosystems, wetlands are particularly important for protection of natural biodiversity because they have much greater species diversity than rivers and lakes, at least amongst the most well understood groups such as vascular plants and birds. The reasons for this are explained in Chapter 28 (Wetland Ecosystems)—flooding has been a powerful selective force in freshwater environments, and species richness decreases with water depth and velocity. Many wetlands that have a higher diversity of dry and moist habitats than rivers and lakes have a more diverse vascular flora, and this is likely to extend to less well-understood groups such as algae and invertebrates.

Wetland management involves four activities that can lead to improvements in the condition of wetlands and increases in the number of wetlands and of wetland area in the landscape—wetland *protection, conservation, restoration* and *creation*. *Protection* refers to the regulatory framework put in place to prevent or reduce human activities likely to reduce their natural character. The hydrological regime that gives wetlands their distinct aquatic character—

shallow and fluctuating water tables (Chapters 15, 20, 28)—makes them particularly vulnerable to disturbance compared with other freshwater ecosystems. Changes to the hydrological regimes of rivers and lakes can have dramatic effects on their biological properties, but few would argue for a hydrological modification that led to their complete disappearance from the landscape. The development of a regulatory framework for wetland protection has therefore been a logical response to the historical loss of wetlands both here and overseas, although recent social research has demonstrated that wetlands on private land can also be successfully protected by co-operation and consultation, rather than litigation, with landowners (Jones *et al.* 1995). *Conservation* refers to management practices that maintain a high natural character in existing wetlands, particularly in relation to hydrology and nutrients, the two main driving forces of wetland ecology. *Restoration* involves environmental management, again usually of hydrology and nutrients, to restore the natural character of degraded systems, and may include additional interventions such as planting, weeding, and even re-introducing flora and fauna that have become locally extinct. It includes the re-establishment of wetlands previously lost entirely from the landscape. *Wetland creation* involves the establishment of a wetland in an area where none previously existed, and will generally require even greater intervention, involving earthworks and construction of weirs to provide a hydrological regime suitable for wetland development. Created wetlands, although a greater technical challenge than conservation and restoration projects, are the only solution that can reverse the loss of wetlands, rather than merely slowing or stopping the loss. Wetlands usually require periods of hundreds to thousands of years to form naturally, an inadequate timescale for contemporary humanity.

The enthusiasm for wetland restoration in recent years has resulted in a considerable body of scientifically-based practical advice for wetland enthusiasts, available partly from experts in both government and non-governmental agencies, but also from informal networks of active practitioners. Many of these organisations have produced pamphlets and factsheets providing management guidance. Major international texts on wetland ecology such as Mitsch and Gosselink (2000) and local restoration guidelines such as those of Tanner and Kloosterman (1997)* and Buxton (1991) are also important sources of

information. Yet many wetland restoration projects still fail to achieve the desired outcomes, both in New Zealand and elsewhere (Malakoff 1998), suggesting that there is still a considerable gap between the scientific understanding of wetland processes and its implementation in successful management (Zedler 2000).

This chapter considers how the principles of hydrology, chemistry and ecology governing the values and functions of wetland ecosystems (described in Chapter 28) relate to the degradation of wetlands by human activities, and how they are applied in reversing the damage. In particular, it focuses on the restoration actions that are most likely to enhance natural character and native biodiversity.

DISTURBANCE AND LOSS OF WETLAND VALUES AND FUNCTIONS

Wetlands have desirable *values* and *functions* that can be defined and are known to be affected by disturbances. The distinction between the terms is that *values* are features that are recognised as being useful for humans, and these values have in the past been the predominant justification for protection and restoration actions. *Functions* are ecological attributes and processes, as described in Chapter 28. Pre-European iwi valued wetlands for a variety of reasons, ranging from food and fibre sources to strategic defence of settlements. Pioneering Europeans, however, saw little value in wetlands (although flax milling was one use); they later valued them as habitat for animals exploited in fishing and hunting. Very recently, wetland values have included the conservation of biodiversity, pollution control and flood mitigation. All of these values, as well as the functions of wetlands, can in principle be quantified and be used to determine the ecological and economic significance of wetlands in a landscape. The extent to which they are damaged by human modification is easily measured.

Human modification of wetlands includes drainage (water abstraction), flooding (excess water), eutrophication (nutrient enrichment), burning, and the introduction of weeds and pests (Clarkson *et al.* 2003). The effects of these practices are consistent and predictable.

Drainage

As the water table is lowered, air is drawn into the soil pores and the redox potential (Chapter 28) increases within

* The difference between a *created* wetland and a *constructed* wetland is an important distinction in wetland management. Created wetlands are wetlands designed for natural values such as biodiversity enhancement, and generally aim to have as great a natural character as possible. Constructed wetlands (also called *treatment* wetlands) are artificial systems designed primarily for wastewater treatment—they may require highly unnatural features such as gravel substrates to enhance hydraulic flow, planting with single highly productive plant species, and extreme nutrient loads. Some constructed wetlands can nevertheless provide valuable habitat for fish and birds. The design and management of constructed wetlands for wastewater treatment in New Zealand are described in detail by Tanner and Kloosterman (1997) and are not covered in this chapter.

a matter of days, as aerobic microbial processes such as nitrate respiration and oxygen respiration replace anaerobic processes. Nitrification is enhanced, increasing nitrogen availability to plants, and competitive and ruderal strategists (see Chapter 14) can invade. The establishment of the oxidizing soil environment also allows dryland species with roots that require external oxygen to survive to colonize the site (see Chapter 28). Blackberry and bracken are examples of species that often begin invading drained wetlands from the edges and spread inwards as the soil dries. Many of the herbaceous exotic weeds that appear in wetland communities after drainage can invade for this reason. Drainage also often involves wetlands becoming isolated from nearby streams and rivers, reducing connectivity for fish migration. Loss of standing water in wetlands reduces the availability of associated habitats providing suitable nesting sites for waterbirds, and allows terrestrial predators such as rats, cats and mustelids easier access for predation of eggs and chicks.

Flooding

Examples of loss of wetland character due to water tables being unnaturally raised are common internationally, due to the diversion of floodwaters and agricultural and industrial runoff into palustrine basins, and the damming of outflows. Increased mean water levels usually result in a loss of plant species diversity, as fewer species are adapted to deep water (Chapter 28). Ponding of water may also increase sedimentation rates, as the lower water velocities allow suspended materials to settle, increasing infilling. Sedimentation and infilling may also occur because ponding often involves the placement of weirs and other barriers on wetland outflows, preventing the litter and detritus produced by plants growing in the wetland from being carried downstream. Because drainage is such a widespread problem in New Zealand, flooding has received less attention, although it is probably under-reported and responsible for many examples of wetland degradation—especially the damming of ephemeral wetlands and wetland seeps in small catchments for farm ponds. Overseas, excessive flooding and elevated water levels are involved in many famous examples of degradation of large wetland ecosystems, such as the Florida Everglades and the reed swamps of the Danube delta. The excess water diverted into wetlands is often nutrient-enriched floodwater or wastewater, so the effects of flooding are often compounded by eutrophication (Fig. 40.1)

Eutrophication

Nutrient enrichment of wetlands can be caused by surface floodwaters, but also occurs via the groundwater received from the catchment, and even by aerial drift of fertiliser applied to adjacent farmland. The tendency for



Figure 40.1 Limited hydrological flushing in this rural wetland has allowed deep layers of highly reducing, anaerobic sediment to accumulate, killing desirable native species and allowing high levels of *Typha* biomass to further alter its natural character and, possibly, ecosystem functioning.

Photo: H. Brix

eutrophication to result in a change in plant species composition was discussed in Chapter 28. It also causes phytoplankton and periphyton blooms in wetlands. Although these have been studied mainly in rivers and lakes in New Zealand (Chapters 15, 21), they have also been recorded in wetlands, but the processes controlling wetland algal proliferations are not well understood. Eutrophication also stimulates microbial activity in the sediment, making it more anaerobic and hence stressing vegetation through low oxygen levels, and increased concentrations of soil phytotoxins such as acetate, propionate and sulphide.

Burning

Unsurprisingly, extensive loss of natural character occurs in wetlands that have been burned. Fast-growing species such as raupo (*Typha orientalis*) produce standing detritus that burns readily, especially in late summer when the peak biomass coincides with hot, dry weather. Other native wetland species that burn readily are the “turpentine” bushes (*Dracophyllum* spp.). Most New Zealand wetlands would have had few natural fires in prehistoric time, so the vegetation is not adapted to fire, and some of our most important and characteristic wetland plants (e.g., tussocks

of *Carex* spp., *Empodisma minus*) are very fire-sensitive and recover only slowly after fires. Fire also changes nutrient regimes, as nitrogen is lost from soils and plant tissue as volatile products of burning whereas phosphorus is retained, and organic matter is mineralised when burnt, often increasing the availability of inorganic nutrients. Fire debris such as charcoal and soot can be toxic to fish and invertebrates. Burning was commonly used to convert wetland to pasture, and has occasionally even been proposed as a management method to remove “excess” biomass, although the harm caused by fire outweighs any possible benefit. Currently, unnatural fires in New Zealand wetlands are mainly due to deliberate vandalism rather than planned management actions. Occasional natural fires did occur in pre-human times in some bogs (ca. once every 100 years), creating areas of open habitat that allowed often rare, early successional species to be maintained. Research has also shown that fish (mudfish) can survive fire by burrowing in the mud, and fernbird populations also quickly recover in a burnt area. However, this requires unburnt source areas in the vicinity, which is often not the case in today’s fragmented landscape.

Weeds and pests

The biology of weed and pest species in New Zealand freshwaters is considered in Chapter 27. In addition to the indirect encouragement such species receive from hydrological and nutrient modification, a number of species can invade even relatively pristine wetlands, because of the absence of consumers and predators, or because there are no native species of similar growth form (the “empty niche” phenomenon—see Chapter 27 and Howard-Williams *et al.* 1987). The most important and destructive examples in New Zealand wetlands are willows (*Salix* spp.), which can totally exclude all native wetland plants and rapidly dry out wetlands due to their high transpiration rates. New Zealand has no wetland tree species that are as fast-growing or flood-tolerant—kahikatea, the nearest equivalent, prefers shallower water and grows relatively slowly. Crack willow (*S. fragilis*) and grey willow (*S. cinerea*) are the two most widespread weed species, the latter of particular concern because it can set seed in New Zealand, whereas the others are represented by only one sex and rely on vegetative reproduction. Other fast-growing weed species present in New Zealand that can seriously impair wetland functioning include reed canary grass (*Phalaris arundinacea*)—a particular problem where nitrogen enrichment occurs (Green and Galatowitsch 2002), purple loosestrife (*Lythrum salicaria*), and yellow flag iris (*Iris pseudacorus*)—particularly invasive in strongly reducing, anaerobic sediments. Common reed (*Phragmites australis*) has a limited distribution in New Zealand at present, but it is known to have all the adaptations and

characteristics for survival and rapid growth in wetlands, and is a large, robust species capable of totally destroying natural character where it invades (Farnsworth and Meyerson 1999; Chambers *et al.* 2003).

Few of these problems occur in isolation—generally wetland management and restoration involves dealing with one or more stress factors simultaneously. As noted above, nutrient enrichment is usually accompanied by weed invasion, and drainage from agricultural catchments usually contains nutrients for point-source discharges and nutrient-enriched groundwater. Management plans for wetlands therefore need to consider the possible outcomes of management actions on a number of related processes. The better the understanding of the system and its ecological functioning before starting, the more likely the project is to succeed.

PLANNING AND MONITORING OF RESTORATION PROJECTS

The over-riding aim of restoration actions in wetlands is to achieve *functional equivalence* between the managed system and a comparable unmodified or pristine system—this includes wetlands under protection and conservation plans, as well as those being actively restored or created. Functional equivalence refers to the managed system having biological production (both primary and secondary), biomass, species composition, nutrient cycling and soil carbon storage similar to that of an unmodified reference system. Although success and failure are value concepts, the “failure” of a wetland management project usually refers to the loss of functional equivalence or the failure to regain it in response to management methods. Often, failure occurs because the attaining of functionally equivalent attributes was never really feasible, or because the nature of the unmodified systems used as a target was not understood.

How much functional equivalence can be practically regained depends on how far the system has diverged from a natural condition. The first and most important stage of any wetland management or restoration plan is therefore to define the goals of the project clearly. The goal of managing a wetland may be flood control, stormwater treatment or nutrient stripping, wildlife or fisheries resources, or re-establishment of natural wetland ecosystems to enhance native biodiversity. The landscaping, hydrological and planting actions for these goals are very different. Figure 40.2 is a protocol for adaptive management, relating project goals and management actions to the monitoring of change, success and background information.

The plan should also include a clear statement of the

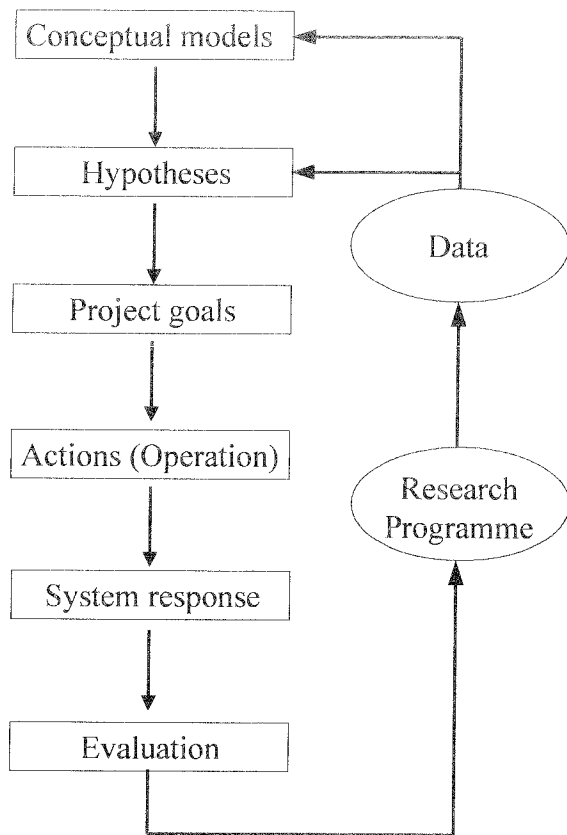


Figure 40.2 Flow diagram describing relations between restoration goals, actions and monitoring, and research information. Data from research programmes should be used to develop general conceptual models of ecological functioning and to define the hypotheses guiding restoration plans. Specific actions from the plans result in a system response, which is evaluated by monitoring and used to guide future research.

hypotheses and the expected outcomes that will occur in response to management actions. Some examples could be:

- “Mechanical removal of willows will increase light penetration and open space, allowing native species present in the seedbank such as cabbage tree and flax to germinate and recover.”
- “Excavating breaches in the stopbanks will allow river water to flush the wetland, removing nutrient-rich sediment, and allow the native *Carex* plants that are currently stressed to recover and native fish to re-establish viable populations.”
- “Raising the mean annual groundwater level by 0.5 m will result in permanent soil saturation and exclude the terrestrial weeds from 80% of the wetland area”.

Note that these statements all relate an outcome to an action, and in each case there is an understanding of the ecological processes and organism life histories that drive change.

Reference systems

Once the goals have been defined, the next step is to develop a management plan that specifies clearly the desired outcome of the management actions, preferably in relation to a *reference system*—i.e., a comparable site that is as close to pristine and unmodified as possible. Ideally, this would be one or more wetlands in the same district or region that have had minimal modification and can be monitored in parallel with the managed system. Where there has been widespread and extensive damage to the local wetlands, so no suitable reference sites are available, it may be a theoretical reference point based on historical data (e.g., on species composition and wetland extent). In either case, the significance of the reference system is that its attributes (nutrient status, flora and fauna etc.) are used to set *targets* for the management actions (Fig. 40.3). The targets need to be realistic. The ideal aim would be to restore the site to reference condition (intersection of dashed lines), but a more realistic outcome is a system that improves according to some criteria but has some different properties and structure to the reference condition. Community composition may be relatively easy to establish, but desired functions (e.g., peat accumulation, nutrient storage) often require greater timescales and greater intervention. Some properties of natural wetlands that take thousands of years to develop, especially those of soils, may not be feasible to replace, even at great expense or intervention. For example, organic carbon is lost rapidly—over years to decades—by the oxidation that occurs when soils have been drained, but it may take centuries of wetland growth to re-establish organic carbon levels. The targets may therefore be set at a level less natural

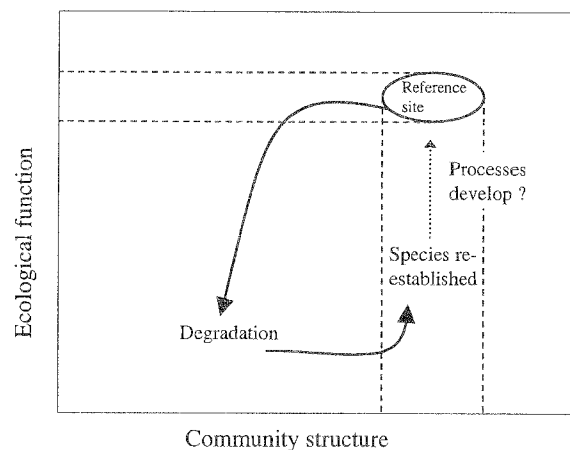


Figure 40.3 Loss of natural character of wetlands due to degradation, and restoration to target reference condition. The reference condition is determined by specific interactions of community structure and ecological function, which are lost due to human activities.

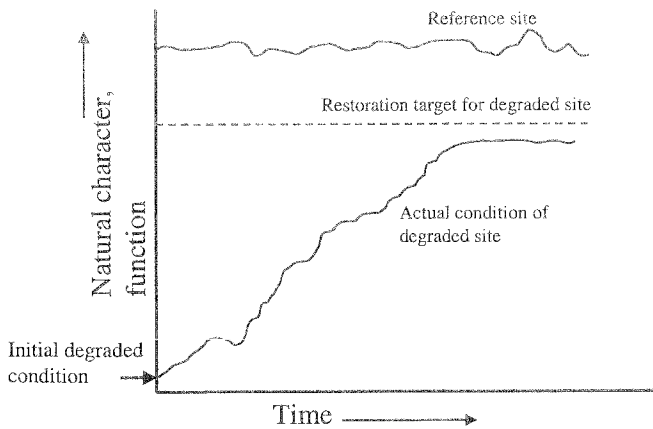


Figure 40.4 Monitoring of progress of the condition of a restored site towards a restoration target defined in terms of aspects of ecosystem structure or function.

than the reference system (Fig. 40.4). The targets must also be measurable. Examples could include species composition (defined in terms of percent cover of specified plant species, numbers of additional plant species per square metre, numbers of native fish species present, number of waterfowl species identified in standard five-minute counts), soil properties developing in relation to re-flooding (development of a specified organic matter content after a given number of years, change from a visibly oxidising to anoxic colour), or reductions in dissolved nutrient concentrations once natural hydrological flushing is restored.

Great care must also be taken to specify targets that are achievable, given the contemporary landscape and land use. For example, attempting to re-create an oligotrophic cushion bog in a landscape that once contained extensive peatland cover but is now primarily agricultural is unlikely to succeed, because the nutrient input may now be too high to sustain such a specialised, oligotrophic ecosystem. In general, therefore, contemporary wetland systems in the area should be used as the reference systems, and their properties used to set targets.

Another important role of reference systems is to give managers information on the nature of the original wetlands, and allow the restoration of a representative array of wetlands in the landscape. For example, some wetland managers appear to equate “wetland restoration” with the creation of open-water ponds, especially as these often appeal to recreational hunters. However, few of New Zealand’s natural wetlands are ponds, and it is unlikely that ponds were the main wetland type lost in any given area. If representativeness is important, re-establishment of vegetation such as flax and *Carex* meadows in palustrine basins is more likely to be an appropriate target than artificial ponds. Inclusion of some open-water areas can

still be specified in the targets, in a plan that includes a more sophisticated heterogeneity than simple ponds.

Monitoring

Once targets have been set and appropriate methods used, a monitoring plan is needed to document the progress of the site towards the desired target. Monitoring therefore records success, and also identifies the need for additional management intervention if progress towards the targets is not being achieved (Fig. 40.4). The targets will be described as some specific aspects of natural character or function (e.g., species composition, nutrient removal rates) that are desired, and the wetland monitored to follow this. The timing and frequency of monitoring differ, depending on the desired goals. Following re-flooding, terrestrial species, for example, can be lost in a matter of weeks, whereas natives may take months or years to germinate and re-grow. The former would therefore be monitored more frequently and over a shorter timescale than the latter. Determination of plant species composition may be practical only at some times of the year (e.g., in late summer, when most species are flowering or fruiting), and such practicalities also define monitoring programmes. Downstream nutrient removal may be greatest at times of highest plant growth, and consistency in timing of monitoring will also be important. Hydrological monitoring can now be carried out with great precision using automatic recorders (see Chapter 20), and the frequency and duration of datalogging with these instruments can also be varied, depending on project goals. A handbook for monitoring the condition of wetlands, based on the pressure-state response model, has recently been produced (Clarkson *et al.* 2003) and is suitable for monitoring the success of restoration actions as well as the condition of intact wetlands.

The monitoring results form the basis of future management actions, in light of current models of ecosystem functioning based on research (Fig. 40.2). The dynamic, often rapid, change that can occur in wetland ecosystems means that principles of adaptive management should be applied—especially if the wetland functioning has rapidly switched into an undesirable path that may be difficult to reverse (e.g., the spread of difficult-to-eradicate weed species).

METHODS FOR RESTORATION

Ombrotrophic (rainwater-fed) systems

The restoration of peatlands (rain-fed raised and blanket bogs) both in New Zealand and overseas has received increasing attention in recent years. The main focus for restoration is often to minimise biodiversity losses, but other roles are becoming increasingly important. For

example, natural peatlands are sinks of atmospheric carbon, but modification and changes in land use have led to large losses of carbon to the atmosphere (estimated at 30–370 Mt C/year globally; Armentano 1980), which have major ramifications for world carbon budgets. In New Zealand there has been a significant loss in peatland biodiversity and associated functions; many bogs were drained during settlement of land over the past 150 years and most remaining areas are modified to some degree. The major threats to peatland ecological integrity are drainage, nutrient enrichment, weed and pest invasion, domestic stock damage, fire, and peat mining. Many bogs, particularly in the lowland zone, require urgent management or restoration if their multiple values and functions are to be maintained.

The role of scientific research

Attempts to manage and restore peatlands, both in New Zealand and overseas have often failed and suffered costly delays, due to a lack of knowledge of the basic processes of peatland functioning (Basu 1997). In some cases, plantings of desired peat-forming species have been unsuccessful because the nutrient and hydrological conditions were more suited to undesirable species that rapidly colonised the site and displaced the planted species. It is now widely accepted that restoration of peatlands is more likely to be successful if it is based on ecological theory (van der Walk 1998; Cole 1999; Keddy 2000). As peat accumulates and the surface rises above the influence of the groundwater, bogs develop from a minerotrophic wetland (one that is dominated by groundwater and nourished by minerals) to an ombrotrophic wetland (one that is dominated by precipitation and deficient in minerals) (Mitsch and Gosselink 2000). Any disturbance resets bog developmental processes back to an earlier successional “stage”, typically with higher nutrients, more fluctuating water tables, and different species compositions. Therefore, an understanding of the physical, chemical and biological characteristics of the full range of successional phases is the key to appropriate restoration techniques.

Restoration approaches

In systems that have been severely degraded, restoration centres on three overlapping phases—rewetting (manipulation of hydrology), renaturation (recolonisation by plants to form natural communities), and regeneration (re-establishment of ecosystem processes such as peat formation) (Heathwaite *et al.* 1993). New Zealand peatlands that have been over-drained or mined for peat come into this category. The key to restoration success is the establishment of appropriate peat-forming vegetation, which indicates bog ecological processes have been initiated (Lindsay 2003).

Restoration projects in less degraded systems usually centre on restoring the water regime, e.g., by damming drainage channels, and removing troublesome exotic weeds such as willow that can potentially overtop and displace native peat-forming species. Compared to older bogs, young systems have higher nutrient levels and shallower peat (with a higher hydraulic conductivity), and tend to be more susceptible to weed invasion and water table draw-down. Whangamarino wetland is a young bog (<2000 years old), and has been declared internationally significant under the Ramsar Convention on Wetlands. Here a river weir has been installed to raise water levels and willow control projects are being undertaken. Individual wetlands may also require fencing to exclude domestic stock, pest control (possums, mustelids, and cats) for protection of birds or palatable plants, and avoidance or mitigation of nutrient inputs.

Fire caused by natural processes, e.g., lightning strike, has historically always been part of the bog ecosystem, and community recovery is usually relatively rapid (6–12 years; Clarkson 1997) provided a seed source of unburnt vegetation is nearby. However, too-frequent fires may result in extirpation of fire-sensitive peat-forming species, e.g., *Empodisma minus*, *Sporadanthus* spp., *Sphagnum* spp. Priorities for restoration in these cases would centre on the re-establishment of key peat-forming vegetation, preferably obtained from a population source nearby, as well as on the removal of exotic species that could potentially change natural developmental pathways.

Restoration of a mined peat bog

Peat is mined for horticultural use at Torehape, an old (>6000 years) peat bog in the Hauraki Plains. The miners are required to restore the area to the original bog vegetation after a permitted depth of peat (one metre) has been removed. Once mining has been completed, some 4 to 5 metres of peat will remain. Restoration trials were established in an area of 0.2 ha on the mined surface in 1998, using various combinations of water table, nutrients, and bog species characteristic of both early and late successional stages of bog development (Schipper *et al.* 2002). The most effective treatments were mounds (“islands”) of processed peat seeded with branches of manuka slash laden with ripe capsules, which reached 100% vegetation cover within two years. The miners used this approach to restore a larger area (200 ha) of mined peat. They created small islands (averaging 5 m in diameter and 30 cm high) of processed peat spaced about 30 m apart, and added one or two branches of manuka slash. Monitoring of some of these islands showed that key late successional, peat-forming species such as *Sporadanthus ferrugineus*, *Empodisma minus*, and occasionally, sphagnum (originating from an adjoining buffer/reference site), had

established within two years, indicating the developing manuka shrubland was acting as a nurse for other bog species. In addition, as vegetation cover increased, invertebrate composition (determined by pitfall trapping) on the islands was shown to become more similar to that found in the reference site. Many older islands now provide a seed and invertebrate source for further species spread via wind and birds onto areas between the islands. These results show that the island technique is effective in initiating the restoration of both plant and invertebrate assemblages in cutover bogs (Clarkson and Watts 2003), and it is also proving to be effective in many restoration projects in minerotrophic systems.

A major challenge for the future will be the restoration of nutrient-enriched agricultural peatlands to low-nutrient peat-forming bog ecosystems to provide buffer zones for vulnerable fragments, to increase peatland size, or to reinstate other peatland functions.

Minerotrophic (groundwater and surface water-fed) systems

No single list of prescribed rules and actions could ever capture the management subtleties needed for the large variety of minerotrophic wetland ecosystems in New Zealand (Johnson and Gerbeaux 2004). The range of landforms and hydrological regimes occupied are almost infinite, and much of the landscaping and hydrological planning will need to be tailored for a specific site. Nevertheless, experience here and overseas has shown that there are several general principles that, if followed, may allow a project to achieve its aims and approach management and restoration targets (Zedler 2000; Mitsch and Gosselink 2000).

Landscaping, soil properties and engineering

Successful restoration and creation of minerotrophic wetlands is very much a function of wise initial site selection. Restoring a degraded system is usually easier than *de novo* creation, and restoration projects have a much better track record of achieving their targets than creation projects (Weinstein *et al.* 2001). Success is more likely in sites where the soil is frequently or can readily be waterlogged, and in sites close to rivers where desirable plants and animals are found. The soil properties need to be suitable for wetland development—not too permeable, preferably without a history of industrial pollution that could prevent a diverse wetland community from developing, and with nutrient availability appropriate for wetlands. Many of the substrates that have been used in wetland projects are too coarse, rocky and infertile for good plant growth, and gravelly substrates may never develop biogeochemical parameters similar to those of reference sites. Other practical considerations are size (small wetlands are more ecologically fragile and prone to failure),

remoteness (access to services and public use and involvement on the one hand, security from vandalism on the other), and the suitability of use of the surrounding land when a wetland is the receiving water body.

The temptation to carry out extensive engineering interventions (use of weirs and stop banks, straight-sided morphology and rectangular basins) is always strong in wetland management, but is often unnecessary and undesirable. Functional equivalence is more likely to be attained in a system where natural contours, natural connections upstream and downstream, and natural energies are mimicked (Mitsch and Jørgensen 1989). This is likely to be less expensive in the long term, and may in any case be essential for colonisation of fish and birds. Natural disturbances such as storms and droughts must also be factored into the design.

Hydrological regime

Until recently, restoring water supplies to drained wetlands was largely carried out using a “just add water” approach, involving the haphazard blocking of drains and installation of weirs, with little attention to the complexities of variations in hydroperiod required by specific ecosystems. Any increase in the duration and depth of inundation is likely to assist in eradicating semi-terrestrial weeds, but native wetland species that differ in flooding tolerance have very different water regimes and are sensitive to subtle differences in hydroperiod. Vegetation types develop in response to a range of soil and hydrological parameters that respond to water management, and developing a better understanding of these linkages (Fig. 40.5) will help in the development of suitable water management plans.

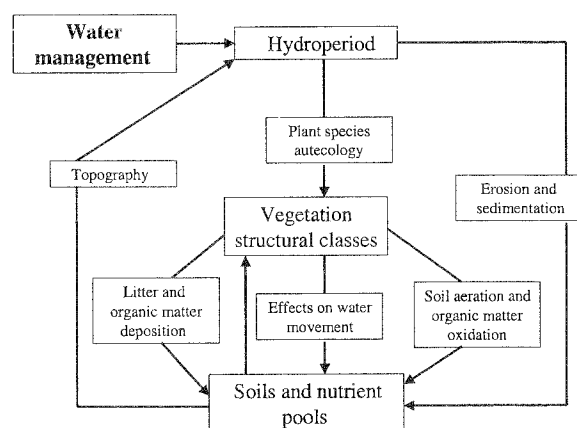


Figure 40.5 Conceptual model of processes underlying the responses of vegetation types to variation in water management practices during restoration activities, and their effects on processes controlling soil development, including feedback effects on vegetation and hydrology. Modified from Ross *et al.* (2003).

Success in restoration can be improved by developing a better understanding of how the hydrology has changed, before beginning any earthworks or construction. This can include using information on the hydrology of reference systems, and using modelling techniques to determine historical water movements, including flooding frequency. Restoring the hydrology to an exact duplicate of its pre-modification state may often not be practical, as other larger-scale changes upstream and downstream may limit available water supplies and discharges. Partial restoration of the hydroperiod will often be sufficient to re-establish wetland biogeochemistry and plant communities, but just how much replication of a former hydrological regime is needed is not clear. It is apparent that many of the common vegetation types in New Zealand wetlands—particularly those dominated by the genera *Carex*, *Juncus* and *Baumea*—favour wetlands where the water level is close to the soil surface most of the time, with only short periods (days or at most weeks) where there is either standing water on the one hand or water tables more than a few millimetres below the surface on the other.

Incorporating the natural daily and seasonal variability in water regime is critical for maximising species diversity, if biodiversity is a restoration goal. In particular, long-term water level fluctuations over weeks, months, or even years that create a mosaic of wet and dry sites, and a cycle of soil wetting and drying, are particularly suitable for many desirable types of vegetation. In wetlands with a visible depth gradient, the “varial” zone over which the water surface is fluctuating is always rich in species. Very stable water levels, which are a feature of many wetland restoration projects and management schemes, are highly unnatural and “squeeze” the vegetation into a dryland, non flood-tolerant community above the water line and an aquatic community below the waterline, with very limited habitat for the semi-aquatic species that provide most diversity in wetlands. On the other hand, large extreme fluctuations (over tens of centimetres or more, and over short time periods) are very destructive to vegetation and usually lead to an unvegetated desert in the varial zone.

Nutrient status, productivity, species richness and invasive species

Although wetlands have a naturally wide range of fertilities, minimising nutrient fluxes during restoration is important to reduce the number of weed species—the density of weed species and their growth rates are clearly related to nutrient status (Fig. 40.6). Minimisation of nutrient input requires two control methods—the use of buffer zones and the use, where possible, of clean water supplies. The latter is a common omission in restoration projects, because water that has already been used for other purposes (e.g., irrigation effluent and storm water) is often that which is most readily available for environmental

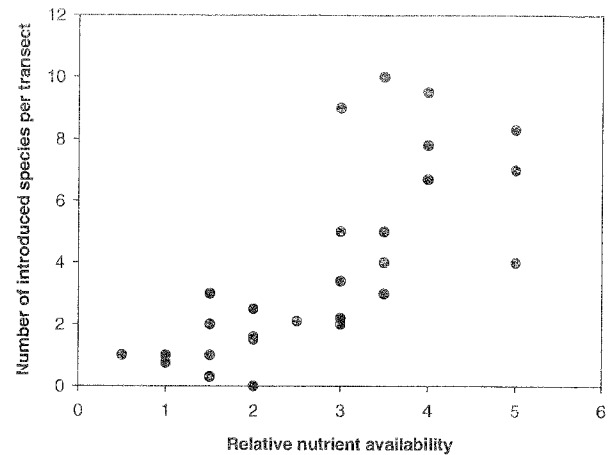


Figure 40.6 Relationship between the presence of weed species (measured as number of species in vegetation transects) and relative nutrient enrichment, for 28 fen and swamp wetlands in the West Coast region. Relative nutrient availability is based on plant and soil nitrogen and phosphorus concentrations. Authors’ unpublished data.

purposes. Clean water is particularly important during the early stages of restoration to reduce the aggressiveness of invasives—a climax community is more resilient to invasion and can cope with higher nutrient fluxes without losing natural character. Incorporating a treatment wetland upstream of a restoration site is one way of providing a cleaner water supply for the project.

Some international research has suggested that species richness is generally highest in wetlands with intermediate fertility (Keddy 2000; Mitsch and Gosselink 2000). This trend is also observed in New Zealand. As there are large changes in species composition across fertility gradients, the protection of sites with a wide range of nutrient availabilities will ensure the most comprehensive protection of biodiversity.

Seed banks, dispersal, planting and succession

Whether or not species will naturally re-establish when hydrology is restored, nutrient inputs are managed, or weeds and pests are removed, is one of the most important questions in wetland restoration. Planting and landscaping are expensive, and allowing the system to “self-design” by natural immigration or by pre-existing seed banks has become desirable for most restoration and creation projects (Mitsch *et al.* 1998). How well self-design can mimic the pre-existing system depends on the availability and recruitment potential of the seeds and propagules, given the hydrological regime that is established (Middleton 1999). In most cases, a seed bank containing some fraction of the wetland flora is usually present and many native species will become re-established. For example, recent

projects in New Zealand wetlands have found that native species such as cabbage trees (*Cordyline australis*), flax, rushes and sedges will re-establish well after large-scale removal of willows, provided the site is contoured to provide the depth ranges required by these species (Chapter 28). When the important, dominant species fail to re-establish, it may be because of a lack of proper hydrological connections or distance from less disturbed sites that could act as a source of propagules (Wienhold and van der Valk 1989; Middleton 1999). In general, seeds of herbaceous species seem to remain viable longer in seedbanks than those of woody species (Middleton 2003). Rarer species, especially those with limited dispersal capacities, are much less likely to spontaneously re-colonise (van der Valk 1998). Although overall species diversity and biomass may rapidly approach that of reference sites, the composition can remain different for many years after re-flooding (Kellogg and Bridgham 2002).

Planting is important in many cases, because random colonisation may not always favour target species as determined from reference sites, especially in small isolated wetlands. Planting can also improve the rate at which a system approaches reference condition. Several factors are important when planting. The spacing of individuals (ca. 0.5–1.0 m for most herbaceous sedges and rushes) must be appropriate to exclude invasive species while allowing enough space for neighbours not to interfere. Contouring and landscaping may be needed to provide the appropriate depth range. During the initial establishment phases, the plantings may need maintenance and husbandry. The nature of the substrate (nutrient status and particle size) also needs to be within the range suitable for the planted species. A disadvantage of planting is that it is likely to lead to more genetically uniform populations than natural colonisation, as it has recently become clear that most common wetland plants have a large amount of genetic variability, even on small spatial scales (Clevering and Lissner 1999). Genetically uniform plantings are more susceptible to disease and weed invasions, and may not be representative of natural biodiversity within species. Another risk of planting is that vigorous stands of planted species may competitively exclude some species that would otherwise colonise naturally.

Self-design is an important principle for fauna as well as flora. If hydrological connections and good water quality are maintained, most invertebrate and native fish species will re-colonise by themselves. Bird species will also re-colonise if the structure of the vegetation and shorelines are suitable.

Above all, it should be recognised that succession is an important feature of wetlands, like all ecosystems, and changes in plant species composition over time should be accepted as part of the management plan and not feared.

If undesirable features (loss of key species, invasion of weeds, disease, mortality) do occur, this can usually be traced to stress due to hydrological and nutrient management.

Size and connectivity

The greater the area of a wetland, the greater its contribution to ecosystem services such as biodiversity and nutrient storage. Larger areas always incorporate a greater range and types of vegetation, which allows a greater total number of species because the number within any single type is limited. A greater range of vegetation types also provides greater habitat diversity for fish and birds. Some bird species require some minimum area for nesting (e.g., bittern) and hence are found only in large wetlands. Wetland size is therefore one important criterion for assigning conservation priority.

For conservation, large reserves are considered to be of greater value than small reserves. This has frequently been justified on the basis of the equilibrium theory of island biogeography (MacArthur and Wilson 1967), which relates species numbers to immigration and extinction rates calculated on the basis of distance to the species source pool and on population sizes (Simberloff 1972). All else being equal, extinction rates will be greater in small reserves because they have smaller population sizes, implying that larger areas will have more individuals and more species. However, it has also been noted that a group of small reserves can contain just as many species as a single large one, and the species-area relationship does not always demonstrate an unambiguous relationship between size and diversity (Simberloff 1972; Simberloff and Abele 1976). Prioritising large wetlands is still a useful principle because of the range of habitats represented, but most important is to ensure the habitat requirements of the target species are provided.

A maximum diversity of species across a range of protected, managed or restored wetlands in a region will be particularly enhanced if they are linked by corridors and surrounded by a protective buffer in the landscape. Wetlands should not be protected and restored as isolated fragments in the landscape, as this increases the probability of local extinction of species, but managed in conjunction with the connecting waterways and catchments. Connecting waterways are particularly important for fish movements (Chapter 17), and connectivity is maximised by both removing physical barriers to migration and protecting stream water quality. Buffer zones for wetlands involve providing marginal areas that can be wetted and dried by natural fluctuations so that there is a gradual transition from a terrestrial to aquatic habitat (the ecotone—see Chapter 28). Abrupt transitions from urban and agricultural landscapes into wetlands allow easier access for predators, and are likely to increase rates of

nutrient input. Another justification for large reserves with a range of community types and buffer zones is that this provides for a range of species with a greater diversity of function than small homogeneous environments, a factor that is more important for ecosystem functioning than species diversity per se (Díaz and Cabido 2001).

PREDICTIVE MODELS AND FUTURE DIRECTIONS

In this chapter, and in Chapters 20 and 28, we have demonstrated that the wide range of wetland types in New Zealand is controlled by a relatively small group of environmental factors—water levels, soil fertility, burning, grazing and disturbance. There is already much information about the mechanisms by which these factors control ecosystems, but it is only loosely applied to management and restoration. There needs to be much better integration between researchers, managers and landowners to ensure that scientific principles are being applied in management, and in a measurable way. Researchers need to be better at assembling information in forms that can be used in management, and in developing the predictive models that can relate outcomes to changes in the important parameters that change in response to management actions. These types of models exist for river ecosystems and lake ecosystems, but are not currently in wide use for wetlands.

One of the current gaps in information relates to the aspects of plant biology controlling survival in different conditions. Species often have very specific ecological requirements that need to be better understood for successful restoration. Information on species germination, establishment and survival under different conditions is sparse, as is specific information on their tolerances of water level fluctuations. The seedbank is another aspect of plant biology that is poorly understood. Once plants have become established in managed sites, they almost inevitably go through succession processes, and these are often accelerated in lowland systems due to high inputs of nutrient or sediment. Learning how to manage nutrients and sediments is one of the greatest current challenges. It is clearly linked to weed colonisation (Fig. 40.6), and is likely to be a major factor in the weed invasions that tend to occur after hydrological restoration. Experiments looking at the effect of using cleaner water supplies in restoration projects on weed colonisation would be of great interest.

Focusing on plants is essential in wetland ecology, given the dominant role they have in structuring wetland ecosystems. Successful colonisation by other groups of organisms is largely driven by the composition and, in particular, the density and structure of the vegetation.

Nevertheless, other questions, such as the role of connectivity and disturbance in processes such as fish passage, also require further attention. Much of our missing understanding of wetlands is likely to be advanced if we cease to treat them as separate, individual fragments, managing them instead in the context of their links with other freshwater habitats and the surrounding catchment.

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Chapter 39

Lake restoration

David Rowe

INTRODUCTION

Over the past 50 years, the water quality has declined in a number of large New Zealand lakes, including Pupuke, Rotorua, Rotoehu, Rotoiti, Tutira and Horowhenua in the North Island, and lakes Ellesmere and Forsyth in the South Island. These lakes have become turbid and are periodically affected by both weed problems and algal blooms. Potable water supplies have been reduced, fisheries have deteriorated, and recreational use, navigation, and tourist activities in these waters have declined. When such problems occur in popular and well used lakes, it quickly catches the public's attention, but many smaller lakes that are less well known have also been degraded over the past 50 years, and this loss has gone largely unnoticed. Now, however, there is an increasing local and national awareness of the need to protect lakes and to restore those that have deteriorated.

Concerns with lake deterioration raise the question—what can be done? Can their deterioration be stopped? More significantly, can degraded lakes be restored? The answer to these queries requires knowledge of what has gone wrong and why. Limnologists have learned much about the causes of lake degradation over the past 100 years and are steadily producing a range of restoration technologies. Both Dunst *et al.* (1974) and Cooke *et al.* (1986) have provided excellent reviews of these, but they were mainly for North American lakes, and the problems and methods applicable to New Zealand lakes often differ from those in North America or Europe.

Control of eutrophication caused by an oversupply of plant nutrients was the major concern with New Zealand lakes in the 1970s. This problem was comprehensively reviewed in Vant (1987) and the main methods of nutrient control used in New Zealand were described by Howard-Williams (1987). Although phosphorus was the main nutrient affecting most Northern Hemisphere lakes at that

time, it was apparent that the concentrations and types of nutrients affecting the phytoplankton of New Zealand lakes don't always follow the patterns of the Northern Hemisphere lakes (White 1983; Howard-Williams and Kelly 2002). New Zealand lakes also contain a less diverse fauna and are more vulnerable to invasions by exotic plants and fish than Northern Hemisphere lakes. However, unlike many lakes in the United Kingdom and Northern Europe, lakes in New Zealand have not suffered from the acidification resulting from industrial pollution. The problems facing lakes in New Zealand therefore differ markedly from those in the northern hemisphere and their restoration requires a different perspective and approach.

This chapter reviews the types of environmental problems encountered in New Zealand lakes, the science underpinning our knowledge of the causes, and the technologies that have been and are being developed to restore the lakes. It then outlines some examples of restoration techniques that have been applied in New Zealand lakes and previews likely advances in restoration.

PROBLEMS REQUIRING RESTORATION

In general, lake restoration is needed when one or more of the main ecological processes affecting the functioning of lakes is compromised, resulting in "lake-wide" problems. For example, the processes responsible for a decline in water quality are often linked with those causing blue-green algal blooms and tainted or toxic water, but differ from those responsible for the loss of macrophytes, or the proliferation of exotic weeds, or the collapse of a fishery. However, the processes are all interlinked to some extent, so it can often be hard to identify the main "drivers" of ecological changes in lakes. Lake restoration therefore requires a clear understanding of the problems and the goals to be achieved so that remedial works are not misplaced or ineffective. In this section, the types of

Table 39.1 Some problems affecting New Zealand lakes and the main causes

Problems affecting lakes	Potential causal factors
1. Eutrophication	<ul style="list-style-type: none"> – increase in external nutrient supply – increased internal nutrient cycling through hypolimnetic deoxygenation – collapse of littoral vegetation – exotic fish introductions
2. Increased turbidity	<ul style="list-style-type: none"> – increase in planktonic algae – increase in re-suspension of silt – turbid inflows – lake shore erosion – blue-green algal blooms – increased lake level fluctuation
3. Invasion by exotic plants	<ul style="list-style-type: none"> – deliberate introduction – accidental introduction on boats, trailers, nets – downstream colonisation – water diversion
4. Invasion by exotic fish	<ul style="list-style-type: none"> – deliberate stocking – accidental transfer – downstream colonisation – water diversion
5. Decline in fish or fisheries	<ul style="list-style-type: none"> – hypolimnetic deoxygenation – introduction of new fish species – increased turbidity – loss of prey species – chemical contamination
6. Chemical contamination	<ul style="list-style-type: none"> – natural geothermal inputs – catchment land use (e.g. farming, horticulture) – waste discharges to inlet streams – road runoff drained into lakes

problem affecting lakes and their main causes are identified (Table 39.1) and the current scientific understanding of the processes responsible is overviewed.

Eutrophication

Eutrophication occurs when plant nutrients (mainly nitrogen and phosphorus) accumulate in lake water and increase its biological productivity and trophic status. Oligotrophic lakes are characterised by low nutrients and low biological production. They have low levels of chlorophyll *a* (reflecting their low density of phytoplankton), contain characteristic phytoplankton, zooplankton and benthic invertebrate communities, and their

waters are relatively clear. In contrast, eutrophic lakes have high nutrient levels and high biological production. Chlorophyll *a* levels are high, blue-green algae blooms occur more frequently, and both plankton and suspended silt contribute to high turbidity and low water clarity.

Eutrophication occurs naturally and slowly in lakes over geological time scales, but this process can be greatly accelerated by human activities and lead to a sudden deterioration in water quality. Lake managers therefore need to know and monitor the trophic status of lakes. Burns *et al.* (2000) have developed a trophic level index (TLI) for New Zealand lakes based on values for chlorophyll *a*, water clarity measured using a secchi disc, total phosphorus and nitrogen. In addition, Clayton and Edwards (2002) have recently produced a new method (LakeSPI) for assessing lake health based on changes in the native and exotic macrophyte community.

Vollenweider (1968) initially developed indices that help define the fundamental relationship between nutrient levels and chlorophyll *a* concentrations in Northern Hemisphere lakes. A similar relationship exists for New Zealand lakes (Pridmore and McBride 1984; Pridmore 1987), and the respective roles played by both phosphorus and nitrogen in the eutrophication of New Zealand lakes are discussed by Vant (1987) and Howard-Williams and Kelly (2002). The realisation that both high nitrogen and phosphorus concentrations promote phytoplankton growth and increase a lake's trophic status underpins most lake restoration efforts. In theory, the aim is to identify which of these two nutrients is controlling phytoplankton levels and to then reduce inputs of this nutrient into the lake. In practice, identifying the nutrients limiting phytoplankton growth is not easy, as a wide range of phytoplankton species inhabit lakes at any one time and their nutrient requirements vary (Pridmore 1987). Blue-green algae can utilise atmospheric nitrogen, so need not be limited by nitrogen inputs, and the availability of silica or iron may limit some phytoplankton groups such as diatoms. Furthermore, the limits set by nitrogen versus phosphorus availability can vary among phytoplankton species, as well as varying between lakes and seasons (Pridmore 1987; Howard-Williams and Kelly 2002).

Nutrient budgets for both nitrogen and phosphorus therefore need to be constructed for New Zealand lakes, with measurements of the mass concentrations of both nutrients in all the main inflows, including rivers, streams, groundwater seeps, geothermal and natural springs, and stormwater inflows. Once the major sources of nutrients have been identified, appropriate measures can then be taken to reduce them. However, a reduction in the "external" supply of nutrients to a lake won't necessarily reduce phytoplankton and restore the lake, at least not in the short term, because nutrients present within lakes (i.e.,

the “internal” supply) may, under some circumstances, be recycled back into the lake water.

In oligotrophic lakes, some nutrients remain within lake sediments and can steadily accumulate over many decades with little noticeable effect. However, once a lake becomes eutrophic, and hypolimnetic deoxygenation results in anoxic conditions above the sediments, chemical changes release these nutrients back into the water column, resulting in more phytoplankton and increased hypolimnetic deoxygenation. This raises the depth of the deoxygenated zone so that a greater area of lake sediment is then exposed to anoxic conditions (Fig. 39.1). In other words, once hypolimnetic deoxygenation begins, a positive feedback cycle is initiated, resulting in a steadily increasing release of nutrients from lake sediments back into the water column. This can continue until the entire hypolimnion becomes anoxic for most of the time that the lake is stratified (Fig. 39.1).

Emergent plants, such as rushes, grow around the margins of many lakes and extract nutrients from the groundwater. Furthermore, inflowing streams may pass through large wetlands of emergent plants before entering lakes. Below the lake surface, aquatic macrophytes in the littoral zone absorb nutrients from both the lake sediments and the water column (Schultz and Malueg 1973), and luxury uptake of nutrients (Gerloff 1969) can occur at times, producing high nutrient concentrations in some macrophytes. These plant beds (both emergent and aquatic) accumulate nutrients in their biomass and are another potential nutrient sink to complement that in the sediments. If these plant beds were to collapse, their nutrients would be released back into the lake water column and further increase algal growth and eutrophication.

In large, deep lakes, where the macrophytes and rush beds are a small component of the total lake ecosystem, the impact of their decline may not be so great. However,

in shallow lakes, where the rushes and emergent plants cover a large area of the lake bed and are a major ecosystem component, or where emergent plants form an extensive wetland that adjoins the lake (e.g., Lake Te Riu - Fig. 39.2), their collapse could lead to a large and sudden influx of nutrients into the lake water column.

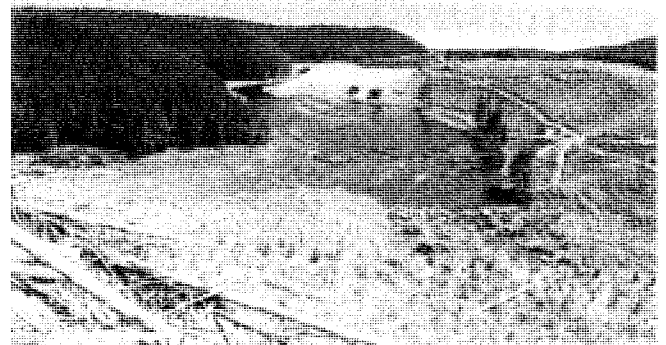


Figure 39.2 Lake Te Riu (Northland) is a shallow lake with a large adjoining wetland (foreground) containing a high biomass of emergent plants. Photo: Dave Rowe

Increased turbidity

Inorganic turbidity is produced by suspended clay, by particulate matter (e.g., glacial flour), or by suspended silt—it reduces light levels in lakes and contributes to the decline of macrophytes. This type of turbidity occurs when inflowing streams or rivers contain high levels of suspended solids because of changes in land use within the catchment, or when silt-laden waters are diverted into lakes (e.g., for hydro-electric developments). Similarly, increased erosion of lake edges, related to greater lake level fluctuations or to the loss of wave damping and buffering by emergent rush and raupo beds, could also result in increased turbidity and so reduce light levels.

In some lakes, a reduction in light levels caused by increased turbidity may precipitate the collapse of macrophytes and further exacerbate eutrophication. For example, a rise in the lowest depth at which photosynthesis can occur (i.e., the photic depth) will raise the lower depth limit for macrophytes and increase the area of open sediment (Fig. 39.3). When the deeper-dwelling macrophytes are reduced in this way, nutrients present in their tissues are released back to the lake water and can result in more algae. The greater exposure of the lake bed to disturbance by either wind-driven currents or by fish foraging activities can also lead to an increase in turbidity from silt re-

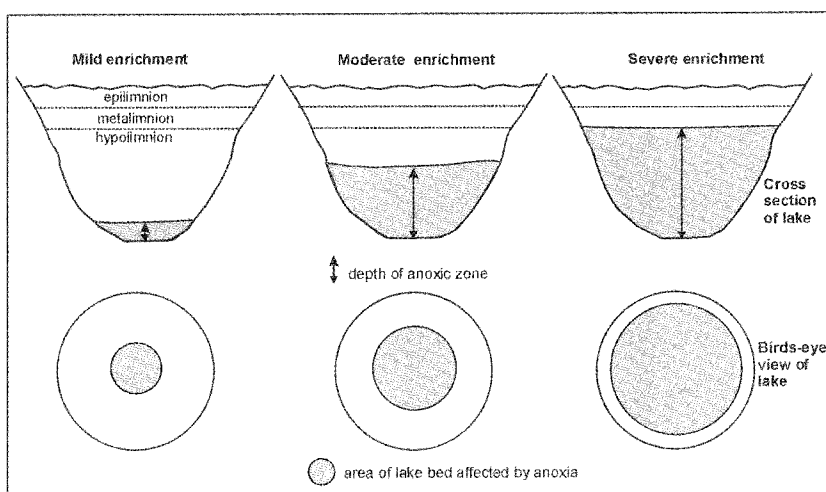


Figure 39.1 Effects of increasing hypolimnetic deoxygenation on the depth of water and area of lake bed exposed to anoxic conditions.

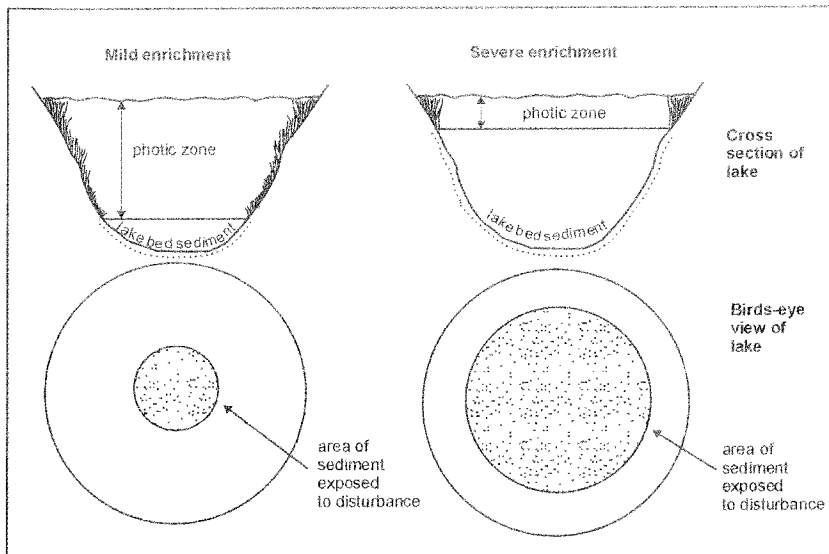


Figure 39.3 Effects of photic zone decrease on macrophyte distribution and the area of lake-bed sediment exposed to disturbance by internal currents or fish foraging.

suspension. Therefore, when macrophyte cover is reduced, increases in both nutrients and silt in the water column can compound the loss of light that initiated the macrophyte decline. Another feedback cycle is then initiated that can eventually lead to macrophyte collapse and lakes “flipping” from a macrophyte-dominated, clear-water state to an algal-dominated, turbid state. Models that predict the lowest depth for macrophytes in lakes, depending on light levels and the various factors that influence this, have been recently produced (e.g., for charophytes by Schwarz *et al.* 2000) and provide managers with guidance on the minimum light levels needed to maintain such plants.

Turbid, algal-dominated lakes are prone to blooms of blue-green algae during summer months and these further increase turbidity, especially during calm conditions. When water movement in lakes is minimal, many algal cells sink below the photic zone. However, blue green algae can produce gas vacuoles, which make them buoyant. During calm conditions their cells can ascend to the photic zone and they then concentrate in surface waters, forming a large algal bloom. Blue-green algal blooms impart a muddy flavour to both water and fish through contamination of the benthic food web and lake water with geosmin, an organic chemical with a distinctive earthy odour. At high densities, and under circumstances yet to be discovered, some species may produce toxins, which create a health risk.

Invasion by exotic plants

Since the 1960s, New Zealand lakes have been invaded by a succession of exotic macrophytes. The main problem species are hydrocharaceans (or oxygen weeds) such as

Elodea canadensis, *Lagarosiphon major*, *Egeria densa* and *Hydrilla verticillata*. However, *Ceratophyllum demersum* (a hornwort) is also proving to be a problem species in some lakes. Initially, these weeds were introduced by aquarists. They don't reproduce sexually in New Zealand, but they do grow vigorously in lakes and ponds and were initially spread by anglers in a misguided attempt to improve fish habitat. Accidental spread by boat trailers and commercial fisher's nets now compounds deliberate introductions (Champion *et al.* 2002).

Prior to these invasions, aquatic plant communities in lakes consisted of relatively low-growing native species that formed littoral stands generally no higher than 2–3 m (Brown 1975; Howard-Williams *et al.* 1987). These native plants

present few problems for swimming, navigation or water supply. However, the invasive, exotic species out-competed the native plants in many lakes and replaced them with much taller (up to 7 m high), mono-specific stands that grew to the water surface (Clayton 1996). These dense beds of exotics prevented swimming and navigation and reduced habitat for fish. Furthermore, because of their height, they are prone to dislocation by wave action, and, following storms, large swathes of weed are periodically washed onto the shorelines of the lakes where they rot (Fig. 39.4). Water intakes for power stations are affected by the drifting weed. In some very shallow (< 2 m deep) lakes, strong winds have been sufficient to break up weed beds to the point where turbidity (from silt and phytoplankton) increases and the weed beds collapse completely (Coffey 1987).



Figure 39.4 Large mats of rotting weed (mainly *Egeria densa*) washed ashore in Lake Rotorua.

Photo: NIWA

These exotic macrophytes provide a ready food source for black swans and have encouraged greater densities of these birds on some lakes. At times, large flocks of swans create problems for farmers because they damage adjoining pasture, but in shallow lakes they may contribute to eutrophication by increasing nutrient cycling (Mitchell *et al.* 1988). Their feeding also results in a constant drift of weed fragments to water intakes in some lakes.

Invasion by exotic fish

The major impact of exotic fish introductions to New Zealand lakes has been a reduction in certain native fish species (mainly Galaxiidae) through competition, predation and interactive segregation. However, the interplay of such biological interactions is far from simple—they are often specific to a particular lake and are not readily predictable or fully understood yet. For example, the high mortality of dune lake galaxiids (*Galaxias gracilis*) in Lake Waikere, Northland caused by interactions with gambusia (*Gambusia affinis*) was restricted, because gambusia were abundant only in marginal rush beds during autumn, when food for the dune lake galaxiid was scarce elsewhere (Rowe *et al.* 1999). Similarly, the introduction of smelt (*Retropinna retropinna*) into Lake Rotopounamu resulted in the extinction of koaro (*Galaxias brevipinnis*) in this, but not other lakes, because there was no deep-water refuge for the koaro in Lake Rotopounamu. The koaro were therefore more vulnerable to predation and competition with the smelt in this lake (Rowe 1993).

The main exotic predators affecting native fish in New Zealand lakes are salmonids. Near complete removal of koaro has occurred in large South Island lakes where the highly piscivorous brown trout (*Salmo trutta*) is dominant, but not in the large North Island lakes where the less piscivorous rainbow trout (*Oncorhynchus mykiss*) is prevalent (Rowe *et al.* 2003). Rainbow trout have reduced rather than displaced the koaro in lakes, and it was the later introduction of smelt that resulted in the virtual disappearance of koaro from the large North Island lakes (Rowe 1993). Introductions of native fish into lakes have also created large changes in other fish species. The inadvertent transfer of koaro into Lake Mahinerangi is thought to have resulted in the displacement of the common river galaxiid *Galaxias vulgaris* (McDowall and Allibone 1994). Furthermore, eel stocking can potentially reduce populations of small fish such as the dune lake galaxiid and the common bully in lakes (Rowe and Graynoth 2002).

Although changes in fish interactions following the introduction of a new species into lakes are a result mainly of predation and competition, the stocking of rudd (*Scardinius erythrophthalmus*) resulted in the collapse of a trout fishery in a small lake because, in the absence of

natural predators such as pike, they proliferated and the large number of stunted rudd then took trout angler's lures before the trout could (Rowe and Champion 1994). Adult rudd are herbivores, so they may also prevent the re-establishment of macrophytes in some lakes (Lake *et al.* 2002).

Northern hemisphere research has recently established the importance of the "top-down" influence by plankton-eating fish on algal biomass (and hence on lake water clarity) that complements the "bottom-up" effect of nutrient supply (Carpenter *et al.* 1985; McQueen *et al.* 1986; Northcote 1988). This top-down influence is based on control of zooplankton by planktivorous fish. When planktivores are reduced (e.g., by a piscicide, or predator) and predation pressure on zooplankton lifts, larger bodied zooplankton species such as cladocera, which are more efficient algal grazers (Burns 1969) increase, and phytoplankton densities then decline, resulting in clearer water. Piscivorous fish (including salmonids), can play a significant "top-down" role in some lakes through their predatory effect on planktivorous fish (e.g., Scavia *et al.* 1986). Such top-down effects by fish have been clearly demonstrated in a number of northern hemisphere lakes (e.g., Reinertsen *et al.* 1990), but a number of these fish manipulation experiments have also failed. Therefore, lake restoration by manipulation of planktivores and their predators is complex and not easily predicted. The potential for "top-down" effects by planktivorous fish in New Zealand lakes has been commented on by Rowe (1994; 1999a), Rowe and Graynoth (2002), Jeppesen *et al.* (1997), and Burns (1998). Such "top-down" bio-manipulation is unlikely to be effective in New Zealand lakes because they lack large cladocera such as *Daphnia*. However, there is potential for planktivores to change the zooplankton and phytoplankton communities within lakes (Jeppesen *et al.* 1997), and the impact of this is yet to be properly documented and explored.

The role of benthic fish, particularly cyprinids, in the decline of lake trophic status through "bottom-up" processes has also been studied (e.g., Andersson *et al.* 1978; Shapiro and Wright 1984; Northcote 1988). A number of fish removal experiments have now clearly demonstrated that benthic fish can also influence lake water clarity (e.g., Bjork 1988; Richardson *et al.* 1995; Tatrai *et al.* 1996, 1997; Berg *et al.* 1997; Bergman *et al.* 1999; Perrow *et al.* 1999; Hamrin 1999). This effect is thought to be related to a combination of bio-perturbation of sediments and silt re-suspension created by one or more of fish foraging activities, macrophyte reduction (e.g., by herbivores), or by increased nutrient re-cycling. For example, the common carp and the brown bullhead catfish both excrete significant amounts of phosphorous compared with other fish (Cooke *et al.* 1986). Surprisingly, even stocked populations

of salmonids have been implicated in increased cycling of phosphorus through their heavy feeding on benthic and terrestrial prey (Schindler *et al.* 2001). However, these effects aside, the juveniles of many of these benthic species are planktivorous, so “top-down” effects may also be involved. The overseas evidence for the impact of benthic and macrophytic fish on water clarity is strong, and control of such fish by either predators (e.g., pike), pond drainage, or by chemical methods (rotenone) has resulted in significant improvements to the water clarity of some affected lakes (e.g., Van Donk *et al.* 1987; Giussani *et al.* 1990; Reinertsen *et al.* 1990; Annadotter *et al.* 1999).

A number of warm-water cyprinid species are now present in New Zealand (e.g., koi carp, rudd, tench, goldfish), and other large, exotic species (e.g., catfish, perch) are also capable of contributing to the processes that result in water quality decline in lakes. More importantly, all of these species are invaders and, in the absence of natural predators, they have the ability to produce large populations and change the food web structures within New Zealand lakes (see Chapter 23). At present, there is little direct evidence that such exotic fish can degrade the water quality of New Zealand lakes, but an analysis of water transparency in northern New Zealand lakes revealed a marked decline in the clarity of lakes containing large (non-salmonid) exotic fish compared with lakes lacking them (Fig. 39.5). In some lakes, historical data indicated that this decline in water transparency occurred after the exotic fish introductions, suggesting that these fish have in some way contributed to the decline in water clarity. Given the recent and increasing spread of such exotic species to many small New Zealand lakes, this is of some concern.

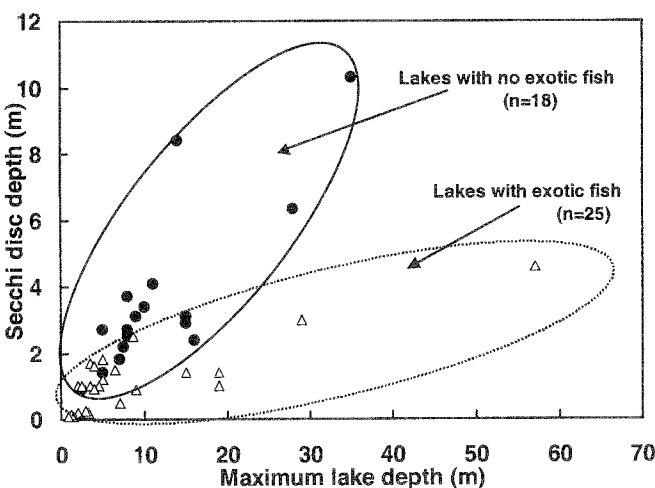


Figure 39.5 Effects of non-salmonid exotic fish on the water clarity of northern New Zealand lakes (ellipses delineate lakes with and without exotic fish).

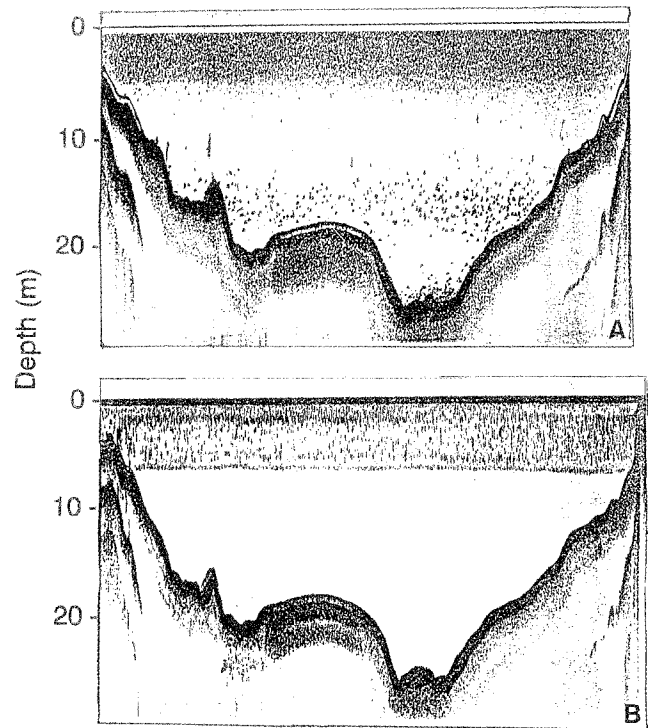


Figure 39.6 Effect of hypolimnetic reduction on fish distributions in Lake Ngapouri (Rotorua); a) Winter echogram showing smelt and trout between 10–20 m b) Summer echogram showing confinement of all fish to water above 6 m.

Decline in fish or fisheries

Although introductions of new fish species to lakes can reduce rare native fish, or degrade fisheries, habitat loss is also a major factor. Hypolimnetic deoxygenation leads to a progressive loss of both benthic and hypolimnetic habitat in lakes over summer months. It kills benthic invertebrates on the lake bed and prevents fish from using the hypolimnion, thereby creating a large lifeless zone. In Lake Ngapouri near Rotorua, hypolimnetic deoxygenation over summer reduced the water depth inhabited by trout and smelt from 26 m to 6 m (Fig. 39.6). During summer, all fish were concentrated in a thin layer near the surface.

This habitat loss compounds the effects of eutrophication on fish. In several Rotorua lakes, the growth rate of rainbow trout was lower in eutrophic lakes than in oligotrophic lakes (Fish 1968) and this was caused by a combination of reduced trout habitat (Rowe and Chisnall 1995), a decline in trout feeding on smelt related to increased turbidity, and changes in their main prey species, related mainly to the impacts of siltation on fish habitat (Rowe 1984; Rowe *et al.* 2003).

The effects of increased turbidity from organic suspended solids on macrophytes were discussed above. However, inorganic suspended solids are heavier and settle on the lake bed, where they can smother benthic habitats.

For example, in Lake Tekapo, which has high loads of inorganic suspended solids, common bullies were absent below 25 m, even though they generally occur at depths down to at least 70 m in other lakes where the lake bed is unaffected by the deposition of inorganic solids (Rowe *et al.* 2001). Furthermore, common bullies were much less abundant in turbid lakes whose beds were coated in a layer of inorganic suspended solids than in the control lakes (Rowe *et al.* 2003). Such differences in fish distribution and abundance are assumed to have occurred because the fine sediment from glacial rivers effectively seals the lake bed with a heavy layer of mud and this then prevents colonisation by interstitial invertebrates and the development of a detrital food chain.

Chemical contamination

New Zealand lakes are not generally affected by acidification as has occurred in many northern Europe lakes. In addition, in some northern hemisphere lakes contamination of lake food webs and hence fish stocks by metals such as mercury and organic compounds, such as PCBs, is a problem. These substances accumulate up the food chain in lacustrine organisms, and tissue concentrations in top predators increase with food-chain length (e.g., Rasmussen *et al.* 1990; Berglund *et al.* 2000). Advisories are issued to anglers in some North American lakes to limit consumption of fish with high mercury or PCB levels.

Some New Zealand lakes also contain such contaminants. For example, mercury (and arsenic) occur within the food webs of many central North Island lakes and reservoirs that have natural geothermal springs within their catchments (Rowe and Graynoth 2002). However, some contamination from industrial discharges is also likely (e.g., from thermal power stations and wood treatment plants), and drainage of runoff from roading networks also introduces chemicals into some lakes. Although top predators, such as trout and eels, contain high levels of mercury in geothermally affected lakes, this is not regarded as a serious enough problem in New Zealand to warrant the issue of health warnings, let alone to contemplate diversion or treatment of sources.

Summary

The types of problems in New Zealand lakes and their causes vary greatly and, before undertaking restoration, it is important to identify all the problems and their cause(s) so that the goals for restoration can be set. Failure to fully identify all causes and to set realistic goals is probably the main reason restoration sometimes fails to meet expectations (National Research Council 1992; Howard-Williams and Kelly 2002). For example, failure to identify long-term nitrogen accumulation in the underground aquifers that feed the spring inflows to central North Island

lakes such as Rotorua and Taupo is one of the reasons why the water quality in Lake Rotorua failed to improve following diversion of the city effluent to land treatment.

APPROACHES TO LAKE RESTORATION

Reduction of phosphorus and nitrogen to control the growth of phytoplankton is often the main goal of restoration in eutrophic lakes (Harper 1992). However, this requires a nutrient budget to identify the main sources, sinks and transport mechanisms. A nutrient budget was prepared for Lake Rotorua in the late 1980s (Hoare 1987) and is the most comprehensive budget prepared for a New Zealand lake to date. Surprisingly, the highest levels of nitrogen and phosphorus occurred in the crystal-clear waters of the Hamurana Spring and not in the more turbid streams draining farmland. Although the nutrient levels in all inflowing waters were determined, nutrient inputs from internal sources, such as the sediments, were not. Cullen *et al.* (1978) provided a comprehensive phosphorus budget for a small Australian lake, which included the phosphorus contribution from sediments, but only under aerobic conditions.

The methods used to reduce nutrients depend on the sources. Historically, inadequately treated sewage was discharged into many large lakes—this is now rare, and most discharges of waste water from treatment facilities into lakes have ceased. Attention has now shifted to diffuse sources of nutrients entering lakes via inlet streams, groundwater, and urban storm-water drains. Riparian planting is being increasingly used to reduce nutrient inputs to streams from land used for agriculture, forestry and horticulture. A recent review of the effectiveness of such measures in the Lake Rotorua catchment indicated that, although particulates were reduced and phosphorus levels remained the same, nitrogen levels had increased (Rutherford *et al.* 2003). Sophisticated GIS-based models are now available to predict the effects of changes in land use on nutrient concentrations in inlet streams (Elliot and Sorrell 2002). In some cases, diversion of streams with high nutrient loads is required (e.g., Lake Tutira, Hawke Bay). Diverting such streams through constructed wetlands may also reduce nutrient inputs to lakes.

Even if the external addition of nutrients to lakes can be reduced, internal concentrations and recycling may need to be addressed. The major methods for this include; (a) dilution, or flushing and the selective abstraction of nutrient-rich waters from the lake, (b) sediment removal, (c) chemical treatment of the lake water to precipitate phosphorus, (d) treatment of sediments to physically seal phosphorus within them, (e) chemical treatment to bind phosphorus in the sediment and prevent its release when anoxic conditions occur and (f) sediment oxidation. Nutrient reduction may also involve varied approaches to

harvesting, but these may only be viable in certain circumstances and in conjunction with other nutrient reduction measures.

Flushing requires a large source of water low in nutrients to replace the nutrient-rich water in the eutrophic lakes. It has been successfully used in several small North American lakes (Dunst *et al.* 1974; Cooke *et al.* 1986). In reservoirs, siphoning off nutrient-rich, hypolimnetic waters from some lakes is feasible and has been attempted with some success. This also reduces the anoxic zone and so may help reduce nutrient fluxes from the sediments to lake water. In Snake Lake, Wisconsin, enriched lake water was diverted to a land disposal site, and while this restoration effort was not successful because of high nutrient levels in the sediments, it demonstrated the feasibility of the method for other small lakes (Dunst *et al.* 1974). Impounding nutrient-rich inflowing streams in wetlands (natural or artificially constructed) may also be possible.

In New Zealand, flushing of nutrients would be limited by lake size and water residence times, pumping costs, and sites for the temporary storage or treatment of the nutrient-rich water. Inadvertent flushing occurs in Lake Rotoaira (North Island), where the water residence time was reduced from 247 to 28 days by diversion of river flows into the lake for hydroelectric power generation. This increased flushing reduced the incidence of both summer stratification and hypolimnetic deoxygenation, and it may have contributed to the reduced occurrence of blue-green algal blooms in this lake (James *et al.* 1999).

Phosphorus concentrations in lake water can be reduced by flocculation and precipitation, but phosphorus release from sediments can still provide a major source of internal nutrients to replace those lost from the water (Howard-Williams 1987). In such cases, sediment removal or treatment is needed. In some small, shallow lakes, sediment has been successfully removed to reduce the internal nutrient levels and to improve water quality. Dredging can be carried out by dragliner, bucket dredge, or hydraulic suction dredge, depending on the lake depth (Dunst *et al.* 1974; Cooke *et al.* 1986). As nutrient concentrations in sediments can increase with water depth (Barroin 1980) and only the deeper parts of lakes are affected by hypolimnetic deoxygenation (Dominic 1980), only the deeper sediments may need to be removed or treated. Although sediment removal may remove a major source of nutrients, it can also create temporarily higher nutrient levels in lake water, and it may also replace nutrient-sorptive sediments with non-sorptive ones (Dunst *et al.* 1974). Furthermore, dredging may be more costly than chemical treatment.

In small lakes that are too deep for dredging, nutrients in the sediment can be physically sealed by covering the affected areas of lake bottom with sand (Dunst *et al.* 1974). Mineral soils, fly ash, clays and hydrous metal oxides may

also be used (e.g., Theis and McCabe 1978), but such techniques are not widely used. Cooke *et al.* (1986) observed that while fly ash was useful for phosphorous inactivation, it could introduce potentially toxic metals into lake ecosystems. Chemical inactivation of nutrients in sediments is generally carried out using alum salts (e.g., aluminium sulphate). However, a range of iron, aluminium, calcium and zirconium salts may also be potentially useful (Jenkins *et al.* 1971, Sanville *et al.* 1982). In Lake Morillon, France, the sediments were initially ploughed to degass them, and alum was then injected into the top 15 cm of sediments to bind the phosphorus. Alum was subsequently spread over the lake surface to precipitate phosphorus in the water column, to flocculate suspended solids (including phytoplankton), and then to settle this floc on the lake-bed, thus helping to seal the sediments (Barroin 1980). Such comprehensive sediment treatment may require supplementary chemical treatment to buffer the pH of the lake water, and is restricted (mainly by cost) to small lakes. However, Annabessacook Lake (Maine, USA) is a relatively large lake (574 ha), and the 150 ha area of lake-bed affected by anoxic waters was successfully treated with alum (Dominic 1980). In New Zealand, Environment Bay of Plenty are planning to treat the sediments of a small, relatively shallow eutrophic lake near Rotorua (Lake Okaro) with alum salts to help improve its water quality.

More recently, sediment oxidation has been suggested as a technique to bind phosphorus with iron in lake sediments (National Research Council 1992). Calcium nitrate is injected into bottom sediments using an underwater "harrow" device and this stimulates denitrification, which results in the phosphorus binding with iron (Cooke *et al.* 1986). In lakes that lack iron, or where the pH is too low, applications of ferric chloride and lime may also be required to complete denitrification. This method has been applied to Lake Lillesjön in Sweden and, although it reduced phosphorous levels in the sediment, the longer-term extent of the phosphorous inactivation is uncertain (Cooke *et al.* 1986).

Harvesting of algae, macrophytes and fish have all been considered as nutrient removal techniques and may be feasible in a few specific lakes, but only where such "biological" nutrient stores are large and readily removed. Physical filtration of algae has been suggested for small lakes (and ponds) where the algae are concentrated in surface waters (Toerien 1977), and this worked on a small scale in Clear Lake, California, USA (Oswald 1976). However, such techniques are not commonly used. Biological filtration has also been suggested, because of the impact of invasive molluscs (e.g., zebra mussels) on water quality in some North American lakes. Freshwater mussels present in New Zealand lakes could theoretically be

stocked in large enough numbers to reduce the phytoplankton of some shallow lakes. However, practical considerations (e.g., low growth rates necessitating high stocking densities) and biotic factors (e.g., calcium limitation) are likely to limit this approach in practice. Analysis of the nutrient concentrations in fish indicates that there is scope for significant phosphorus removal only when most fish are eliminated (e.g., by piscicide treatment) (Dunst *et al.* 1974). In general, too little is known about harvesting options to determine their usefulness in lake restoration and consequently such methods are rarely used.

Where nutrients cannot be reduced to the point where water quality improves, then a number of methods are available to alleviate some of the main symptoms of eutrophication (Dunst *et al.* 1974; National Research Council 1992; Harper 1992). De-stratification and aeration of the hypolimnion are the main techniques—these are both designed to introduce more oxygen to the hypolimnion. This increases fish habitat and/or prevents the chemical changes that release nutrients in sediments back into the water column.

In shallow lakes, de-stratification may involve the use of floating mechanical aerators (Fig. 39.7). However, in deeper lakes, compressed air is piped to the lake bottom. In one type of compressed air application, air is released through small perforations in lengths of hose laid on the lake bed. The resultant curtain of bubbles causes localised up-welling of water. This technique was routinely used in a number of Auckland's water reservoirs to prevent hypolimnetic deoxygenation (Howard-Williams 1987). The other main application, the airlift, involves the release of bubbles into a large diameter PVC tube positioned vertically in the lake and extending from the hypolimnion to the epilimnion. As each bubble moves towards the lake surface, it expands to fill the tube's diameter, pushing water



Figure 39.7 Mechanical surface aerator used to disperse algal scums and to increase water circulation in Lake Te Koutu (Cambridge). Photo: Dave Rowe

above it and pulling water behind it to create a vertical current. This lifts the colder hypolimnetic water to the lake surface. In some airlifts, inclusion of an Archimedes screw within the tube increases efficiency. Provided enough airlifts are installed and their lifting power is sufficient, they can fully mix the lake water and de-stratify the lake (Harper 1992).

Although de-stratification can reduce the volume of anoxic water and thereby improve fish habitat, it may increase phosphorus concentrations in lake water by warming the sediments and by increasing the physical transfer of phosphorus from the sediment-water interface to the lake water (Dunst *et al.* 1974; Cooke *et al.* 1986). This is overcome by aerating or oxygenating the hypolimnion rather than by de-stratifying the lake. Airlifts used for de-stratification can be modified for this purpose. Hypolimnetic water is brought to the lake surface by the airlift, where it is exposed to the atmosphere in a floating container. This container prevents it from mixing with surface water and aerates it. The aerated water is then returned to the hypolimnion via return pipes. Another system of airlift aerates hypolimnetic water without bringing it to the surface. The latter device is more popular, but the former is more efficient (Cooke *et al.* 1986). With aeration of just the hypolimnetic water, the lake remains stratified, and oxygen levels in the hypolimnion are increased. In theory, direct infusion of oxygen to hypolimnetic waters could also accomplish this. Failure of such systems to work adequately are generally related to the use of undersized systems. The chemical and biotic changes arising from de-stratification and hypolimnetic deoxygenation are both complex and profound and can be expected to vary from lake to lake. Much more research is required to predict the full effects of such measures on the biota of lakes (Dunst *et al.* 1974; Cooke *et al.* 1986).

Various types of biomanipulation are also used to alleviate the symptoms of eutrophication—they include methods to address weed problems, to reduce algal blooms, and methods to reduce fish species responsible for “top-down” effects on plankton, or for increased nutrient cycling and sediment re-suspension.

A large amount of research has been carried out into the invasion and spread of exotic plants in New Zealand lakes, as well as into the problems the various species create, and the control options available (e.g., Clayton and Tanner 1988; Champion *et al.* 2002). The main species that cause problems only reproduce by vegetative propagation (because only one sex is present in New Zealand) but the introduction of even a small plant fragment can result in the establishment of the species. Restoration of lakes infested with exotic macrophytes involves either on-going control of the problem species through the periodic use of herbicide, or harvesting, or the removal of all macrophytes

by a biological control agent such as grass carp (*Ctenopharyngodon idella*) (Champion *et al.* 2002). Spraying with herbicides, such as diquat, has proved effective for some weed species in some New Zealand lakes (Clayton and Tanner 1988; Clayton and Wells 2001). However, its use is limited to clear rather than turbid waters, and successful application generally requires calm, still conditions. Mechanical harvesting and spot removal by diver-operated suction dredge is suitable in some shallow lakes where the cost of ongoing control is warranted. Grass carp stocking has now proved successful in removing all plants (and in eliminating a range of exotic species) from a number of New Zealand lakes (Rowe and Hill 1989; Rowe and Graynoth 2002). These carp readily eradicate plants that reproduce vegetatively. Once the exotic weed species are eradicated, native plant communities regenerate (from seeds present in lake bed sediments), but this requires a reduction in browsing pressure by the grass carp. Grass carp will maintain total control over plant growth for many years after stocking and can live for 10–15 years in the wild. Densities therefore need to be reduced, before native species establish, but these fish are difficult to remove from lakes without the use of piscicides. Recent research using formulations of toxic bait shows some promise for this (Rowe 1999b).

Fish that feed on phytoplankton are sometimes used to reduce blooms of algae or to prevent their occurrence in lakes. Prowse (1969) suggested that silver carp, grey mullet and certain species of tilapia could be used for this purpose. This form of biomanipulation has been used in a number of Israeli lakes to reduce algal blooms (Leventer 1981). It achieved a useful degree of control in a Brazilian lake (Starling and Rocha 1990) and is thought to show some promise for Chinese lakes (Wu *et al.* 1997). It has also been tried, with some success, in a New Zealand lake (Carruthers 1986). Such methods are restricted to lakes from which these fish cannot escape and they can require high stocking densities, which are not always achievable.

Restoration of lakes where problems are caused by introduced exotic fish is mainly carried out using a piscicide such as rotenone (Rowe 2003). However, stocking of predatory fish to reduce zooplanktivorous species has also been used with some success in European lakes to increase “top-down” influences by zooplankton on phytoplankton and hence to improve water clarity (Berg *et al.* 1997; Bergman *et al.* 1999; Perrow *et al.* 1999; Tatrai *et al.* 1997). In Lake Trummen (Sweden), annual reduction of common carp was eventually required to improve lake water clarity after diversion of nutrient inputs and sediment skimming reduced external and internal nutrient levels but failed to improve water quality in the long term (Bjork 1988). In New Zealand, rotenone was successfully used to eliminate rudd and tench from

Parkinson’s Lake near Pukekohe (see below). Other measures that can be used to reduce exotic fish and to restore native fish in lakes involve habitat modification. For example, water level reductions reduced the spawning, and hence the abundance of a cyprinid in Lake Kinneret (Gafny *et al.* 1992), and northern pike in Lake Mendota (National Research Council 1992).

EXAMPLES OF LAKE RESTORATION IN NEW ZEALAND

Lake Tutira

Lake Tutira (area 1.47 km², max. depth 42 m), in the Hawke Bay area, became noticeably more turbid and eutrophic in the 1960s and 1970s because of high nutrient inputs from the surrounding catchment. The catchment changes included the historic clearance of native forest, and then the creation and management of pasture for sheep farming. Concerns over the state of the lake were raised by trout anglers and resulted in initial studies by the Ministry of Agriculture and Fisheries. These revealed oxygen depletion in the hypolimnion and culminated in the installation of a number of airlifts to de-stratify the lake during summer months. Six 13-m-long vertical tubes (approx 30 cm diameter) were installed in the lake by SCUBA divers in 1975 and extended from close to the surface down to the hypolimnion (Teirney 1980). A shore-based compressor pumped air to the base of each tube. Ascending air created a vertical water current in the tube. This resulted in de-stratification and increased oxygen levels in the hypolimnion between 1975–1979. The ongoing cost of running the air compressor prohibited continuation of this treatment.

Attention then turned to identification of the main source of nutrients, specifically the streams entering the lake. Sandy Creek, the main inflowing stream, was subsequently identified as a major source of nutrients and was diverted around the lake so that it discharged close to the lake outlet (Hawke Bay Catchment Board 1986).

Lake Rotorua/Rotoiti

The water quality in Lake Rotorua (area 79.78 km², max. depth 44.5 m) declined during the 1950s and 1960s, primarily because of the discharge of partially treated sewage from the city of Rotorua into the lake. On the basis of northern hemisphere experience, overseas consultants assumed that phosphorus was the main problem and recommended chemical treatment of the city’s waste water. However, this was not particularly effective. Subsequent research to identify the major sources of nutrient inputs to the lake revealed that, apart from the partially treated sewage from the City of Rotorua, the crystal-clear waters of the Hamurana springs contributed a large fraction of

the total nitrogen and phosphorus (Fish 1975; Hoare 1987). Nevertheless, it was concluded that better treatment of waste water would reduce a major source of nutrients. Eventually two measures to reduce nutrients entering the lake were approved. The first involved land disposal of the treated effluent by spray irrigation in the Whakarewarewa Forest. The second measure was the implementation of the Kaituna Catchment Scheme. This was designed to reduce non-point sources of nutrients entering the lake's tributary streams from activities such as farming. The scheme encouraged the gradual planting of native trees along the edges of many of the main inflowing streams, both in Lake Rotorua and Lake Rotoiti.

The lake did not recover as quickly as anticipated, probably because it still contained very high internal nutrient levels in its sediments. Furthermore, nitrate levels in the streams have continued to increase (Rutherford *et al.* 2003). Howard-Williams and Kelly (2002) estimated that just four days of anoxic conditions over the lake's sediments would produce the same amount of nitrogen and phosphorus that are brought into the lake by all external sources for an entire year. Because Lake Rotorua is relatively large and shallow, it is usually well mixed. However, it stratifies for brief periods during calm conditions in summer (i.e., it is polymictic) and anoxic conditions can develop in certain areas of the lake during these times. As the lake bottom is relatively flat, such areas vary, but are readily detected by routine echo-sounding—fish echoes are absent in the pockets of anoxic water (author's unpublished data). Nutrient release from lake sediments can be expected when anoxia occurs and this release may now contribute to the lake's continued high productivity and failure to recover quickly. However, the restoration measures that have been completed will have reduced the rate of deterioration in this lake.

The downstream effects of Lake Rotorua water on Lake Rotoiti are now of concern. This large (area 34.34 km²), deep (max. depth 93.5 m) lake receives a large proportion of the nutrients leaving Lake Rotorua in its outflow, and it has a relatively long water residence time. Its hypolimnion is deoxygenated for an extended period (3 months) each year, which means that it is potentially more vulnerable to nutrient increases and the compounding effects of hypolimnetic deoxygenation than Lake Rotorua. Studies by the DSIR in the 1980s (Gibbs *et al.* 1983; Vincent *et al.* 1986) indicated that the water exiting Lake Rotorua through the Ohau channel (which connects these two lakes) often penetrated well into Lake Rotoiti. Over time, the nutrient-laden waters from Lake Rotorua have increased the nutrients deposited in the sediments of Lake Rotoiti. This caused little noticeable problem until hypolimnetic deoxygenation increased and released the nutrients in the sediments back into the water column. Hypolimnetic deoxygenation has increased markedly

between the 1950s and 1980s, and Lake Rotoiti's water quality has deteriorated rapidly over the past decade, with a noticeable increase in algal blooms over the past five years. In 2003, health warnings were issued for the entire lake, recommending no water contact. The increased hypolimnetic deoxygenation in Lake Rotoiti has also resulted in a major loss in trout habitat in this lake (Rowe and Chisnall 1995). In late summer, the trout are now "sandwiched" between the warm surface waters, which are too hot for them, and the cooler, deeper hypolimnetic waters, which are deoxygenated and uninhabitable. Such "habitat squeeze" reduces the volume of habitable water for fish and if it continues could eventually limit trout production here.

The blue-green algal bloom and lake closure in 2003 galvanised authorities to take action to restore Lake Rotoiti. Although further nutrient reduction in Lake Rotorua will be a major long-term goal, a number of shorter-term, alleviative measures are also under investigation. These include the diversion of Lake Rotorua water around Lake Rotoiti to its outlet, the feasibility of hypolimnetic aeration, and the possibility of direct oxygenation of the hypolimnion (pers. comm. M. Gibbs). The contribution of nutrients from within the Lake Rotoiti catchment, including those produced by local farm runoff (e.g., following aerial top-dressing of pasture), recreational users (e.g., fleets of trailer sailor yachts lacking holding tanks), septic tanks leaching into groundwater, and pine forest management (e.g., harvesting and soil fertilisation) will also need to be more closely examined. As internal cycling of phosphorus from the sediments may be a major problem, the lake sediments may also need to be investigated to determine the feasibility of treating or sealing areas with high nutrient concentrations.

Parkinson's Lake

Although Parkinson's Lake near Waiuku (area 0.02 km², max. depth 8 m) was moderately enriched, eutrophication was not the main problem in this lake. It was affected firstly by the introduction of the exotic weed *Egeria* and its rapid expansion across much of the lake, and secondly by introductions of exotic fish—rudd and tench (Rowe and Champion 1994). These fish and plant introductions ruined the "put-and-take" trout fishery in this lake, mainly through their effects on angling efficiency. But they also created problems for water intakes and swimming.

Restoration involved elimination of all the exotic problem species and then re-establishment of native species. *Egeria* was firstly eliminated by grass carp—this was feasible because *Egeria* occurs as a single-sex cultivar and only reproduces vegetatively. There were no seeds to re-establish its populations once all vegetation was removed. This was accomplished over a period of two to three years, and the resulting lack of cover for fish resulted

in increased shag predation on the larger fish species, especially rudd and tench (Rowe 1984; author's unpubl. data). Although this reduced densities of these exotic fish species, it did not eliminate them, and rotenone was later used to completely remove all fish. Following the removal of all fish, water clarity improved, and the native macrophyte fauna soon re-established itself in the littoral zone from seeds present in the sediment. The lake was later restocked with native fish species (Rowe and Champion 1994) and the trout fishery re-established by stocking. This fishery produced larger trout than had occurred in the past, for at least four years.

The restoration of this lake represents an example of integrated biomanipulation, where weed problems and fishery problems were interlinked and where the addition of a new species (grass carp) was needed to eliminate other exotic species. Eradication of the exotic fish would not have been possible without the prior removal of all plants by grass carp.

Lake Waingata

Lake Waingata (area 0.12 km², max. depth 7 m) in Northland contained a low-density population of the threatened dune lake galaxiid *Galaxias gracilis*. The main reason for the low density of this species was thought to be the complete displacement of native plants by the exotic weed *Elodea canadensis*, and the effects of its prolific growth, from the lake bed to the water surface, over much of the lake. However, eutrophication was also a potential factor. Grass carp stocking was used to eliminate the *Elodea* from this lake and the density of the dune lakes galaxiid subsequently increased (Rowe *et al.* 1999a). The grass carp continued to maintain the lake in a de-vegetated state, despite a decline in their abundance from natural mortality, so they were reduced by a combination of netting and then experimental methods using Prentox bait (e.g., Rowe 1999b). Once carp numbers were reduced to a level where macrophyte production exceeded the total browsing, native plants began to regenerate from seeds in the sediment and plant biomass increased in the littoral zone. This restoration of the native macrophyte flora is being accompanied by an increase in water clarity and an increase in the dune lake galaxiid.

Lake Orakai

Lake Orakai is a small (4 ha, deep) hyper-eutrophic lake close to Lake Tutira in Hawke Bay—it was subject to frequent blooms of blue-green algae. Silver carp were experimentally released in this lake in 1978 to test the ability of this phytoplanktivorous fish to control algal blooms. Using a barrier net, the lake was split into a control section with no fish, and an experimental section with fish present. Initial results were promising and control was clearly established for a time in the experimental section.

This occurred even when the control and experimental sections were reversed (Carruthers 1986). However, control in this lake may have occurred because of the high density of blue-green algae, as well as the high stocking rates of silver carp, and so may not occur in other lakes with lesser blooms. Furthermore, the duration for effective control was not determined, because algal biomass was not monitored. The implications for lake ecology, particularly for the pelagic food web and larval fish densities, were also not investigated (Rowe 1989). Although these initial results showed some promise, they were not continued because of budget constraints and the need to address other priorities.

LAKE RESTORATION IN THE FUTURE

Modelling tools

As limnologists' understanding of the processes responsible for lake decline increases, modelling has become an important tool, as it allows the integration of the wide range of variables involved. Harper (1992) provides a good account of some of the dynamic models, including CLEAN, that have been recently developed to incorporate and integrate the effects of the numerous ecological relationships in lakes on water quality. Other models are based on hydrodynamics and thermal mixing patterns. In addition to models of lake ecosystems, a number of sophisticated models are also being developed to predict the effects of changes in land use in New Zealand lake catchments on the nutrient concentrations of their inlet streams (e.g., Elliot and Sorrell 2002). Integration of both these lake and catchment models will eventually pave the way for better prediction and prevention of problems with nutrients in lakes, as well as the identification of other restoration options.

Food web theory and biomanipulation

Lake restoration is not simplistic and confined to just nutrient control or macrophyte loss. Knowledge of the key roles of benthic and planktivorous fish in accelerating eutrophication is already resulting in an increase in fish biomanipulation experiments in Northern Hemisphere lakes. These experiments are limited at present and have met with varying success because knowledge of the ecological role of each fish species in a given lake is lacking. Research can be expected to continue in this field because of its fundamental importance to lake restoration. Meanwhile, important links in the food webs of lakes to be restored need to be established, along with the main distributions and relative abundance of benthic or planktivorous fish species that could contribute to top-down or nutrient cycling effects. Modelling techniques can then be used to identify the potential role of these fish species and to determine whether control is required.

Integration of lake management and planning

Perhaps the greatest challenge for future lake restoration, both in New Zealand and overseas, will be the need for planning and interagency co-operation and co-ordination. Lake restoration requires a holistic, ecological approach to lake management, and this cuts across the administrative boundaries for water management, for lake-bed control, for recreational use, for angling, for the conservation of native species, and for customary uses and guardianship. The high cost of restoration, together with the fragmented administration of lake management, can result in institutional barriers to progress, and this has stymied the restoration of some Northern Hemisphere lakes (e.g., Dunst *et al.* 1974, National Research Council 1992). Such administrative barriers have been overridden by central government, which can establish independent authorities charged with co-ordinating restoration efforts. However, even this approach can be limited by legislation and there is a danger that restoration will be inhibited because of both the fragmented control over lakes and the resultant political side-stepping. It is clear that the Ministry for the Environment and Regional Councils will have an important leadership role in this area in New Zealand. Improved planning, especially involving local communities, can potentially play an important role in co-ordinating restoration efforts by the various administrative agencies involved while simultaneously raising public support. The recent efforts by Environment Waikato to secure widespread agreement over the management of Lake Taupo provides an example of such planning.

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Chapter 38

River restoration

Alastair Suren, Shelley McMurtrie and Leanne O'Brien

INTRODUCTION

What is river restoration?

As New Zealanders, we value our rivers and streams for many reasons. They are an integral part of the natural environment, and the setting for recreational activities such as fishing, jet boating, rowing, kayaking and swimming. They have social importance, with cultural uses such as the provision of mahinga kai, and historical and aesthetic values. Rivers also provide essential services to our economy, including power for our hydro-electricity schemes and water for irrigating land (Chapter 37). Land uses such as urban development, forestry, mining, and farming directly or indirectly impinge on our waterways to varying degrees (Chapters 34-37). Consequently, our use of waterways and their catchments can directly compromise the values we associate with them (Fig. 38.1). For example, urbanisation, agriculture and mining reduce stream habitat quality to the detriment of aquatic invertebrate and fish communities, leading to a reduction in the fisheries value of many lowland streams (Craig 2003). A dichotomy thus exists between how we value rivers and how our activities impinge on these values. The field of river restoration is concerned with recognising the adverse effects of our activities and developing methods to mitigate or remedy them to help protect a river's values.

Restoration is a relatively new field of ecology, and in New Zealand most restoration efforts have been implemented only over the past 10 years. Most of these have concentrated on restoring riparian vegetation, although there has also been some restoration of banks and instream habitat, as well as the flow regimes of rivers used for generating electricity. This chapter examines factors such as scale, disturbance, and potential limitations to restoration, describes the general planning necessary for successful restoration activities, and discusses problems relating to various restoration methods.

At this point it is useful to clarify two commonly used terms—*restoration* and *rehabilitation*. Although they are often used interchangeably, *restoration* refers to returning a degraded river to something near its original state, whereas *rehabilitation* refers to improving the conditions of a degraded river to a less degraded, but not original state (Bradshaw 1996; Harper *et al.* 1998). Given these definitions, much of what is termed “restoration” might be more accurately termed “rehabilitation”, as it is debatable whether degraded ecosystems can ever be fully restored. Indeed, at best, the outcome of restoration activities is thought to be the re-creation of features or communities that are merely similar to those that once existed, rather than complete ecosystem recovery (Boon 1998). In this chapter, we use the term “restoration” to cover both restoration and rehabilitation, in keeping with international usage. The underlying premise of restoration activities is that by improving instream habitat, the biological communities will recover. This has been called the “field of dreams” hypothesis—“if you build it, they will come” (Palmer *et al.* 1997).

Resilience and resistance

New Zealand's rivers and streams are highly changeable, disturbed, and patchy environments (Townsend 1989), where aquatic communities experience large-scale natural disturbances such as floods, droughts, and sediment inputs from landslides or bank erosion. Disturbances are best defined as any discrete event that disrupts populations, communities or ecosystem structure, often by changing the physical environment or resource abundance (Resh *et al.* 1988; Lamberti *et al.* 1991). Disturbances can be predictable (e.g., seasonal floods) or unpredictable (e.g., sedimentation from slips) and can vary in magnitude from disturbances that influence local stream conditions to those affecting all streams throughout a catchment. Large

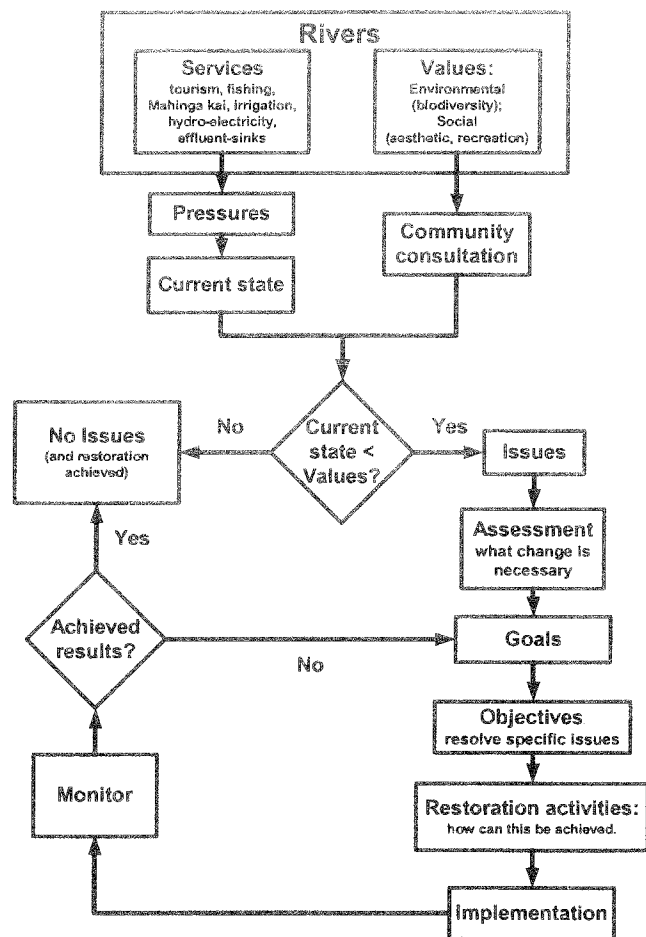


Figure 38.1 Suggested planning steps for river restoration, modified from Hobbs and Norton (1996).

disturbances can significantly change biological communities in rivers, although these changes are often minimised as a result of two natural properties of biological communities—their *resistance* and *resilience* (Winterbourn 1997).

Resilience is characterised by the ability of stream communities to recover quickly to their pre-disturbed state following a disturbance (e.g., Winterbourn 1997; Fig. 38.2a). Ecosystem *resistance*, in contrast, is the ability of communities to oppose the effects of disturbances, so that there is little change in community structure (Fig. 38.2b). These two traits can refer to ecosystems, communities and individual species, and have important implications for river restoration. For example, land-use activities (e.g., mining, agriculture, forestry, urbanisation and irrigation) can significantly disturb rivers and streams, and have major effects on their ecosystems by reducing habitat complexity, altering energy inputs from riparian vegetation and changing water quality and quantity. However, due to the resistance or resilience of aquatic communities, recovery is possible even after relatively large manmade disturbances,

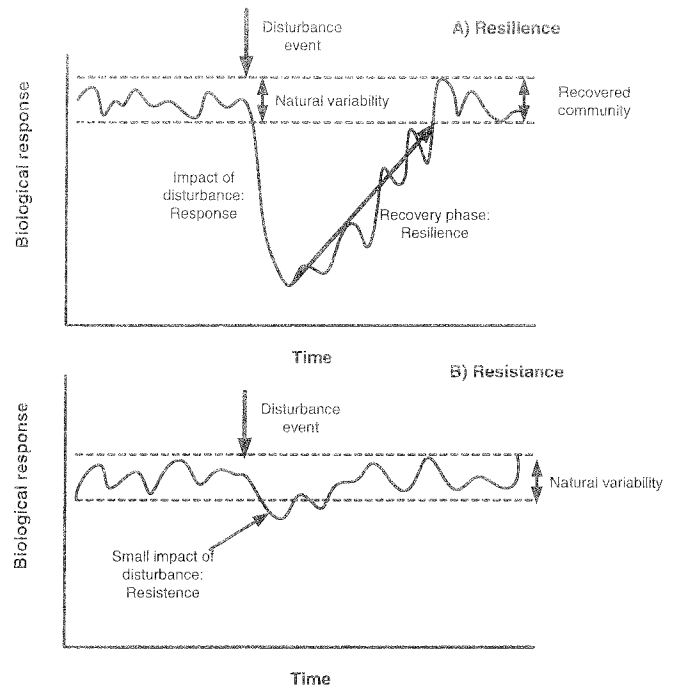


Figure 38.2 Conceptual diagram illustrating A) resilience and B) resistance of biological communities to disturbances.

if the effects of these disturbances are minimised or removed, and if there is a pool of plants and animals that can recolonise the stream. This natural “self-healing” process of river ecosystems is a basic factor determining the success or failure of restoration activities.

Steps to river restoration

Identifying problems

Restoration activities are usually initiated to resolve conflict between the current conditions of a stream and what society values it for. If an individual stream has no great value to society, then there may be no incentive for society to restore it. The first steps of river restoration are to identify the pressures placed on a river by human uses, and to characterise the river’s current state—its physical, chemical and biological conditions.

Once the river’s current state has been characterised, this can be compared with the values placed on the river. These values can be identified through community consultation. Note that a river’s values may not always be obvious to the greater community: education is thus an important aspect in determining the value, or potential value, of a stream. Management may be needed if the existing state of the river is poorer than the community expects (Fig. 38.1). For example, urban development commonly causes deterioration in the physical, chemical and biological conditions of streams (Chapter 35), so continuing urban

development may become a management problem for some streams. If such development occurs in catchments dominated by intensive agriculture where the streams are already degraded and have little value to society, further degradation associated with urbanisation may not be a significant issue. However, urban development within a catchment where streams have high recreational, aesthetic or fishery values may result in significant issues. Under such conditions, stream restoration techniques may be used to mitigate or remedy the adverse impacts of urbanisation.

Setting goals

Once management problems have been highlighted, it is necessary to assess the changes needed to mitigate further damage. The assessment should consider historic and present pressures placed on a stream, as well as other factors (e.g., the length of time over which a disturbance has occurred, the size and geomorphology of the stream, and the spatial, social and fiscal constraints on potential restoration activities). Once this assessment has been completed, specific restoration goals can be identified. These goals will determine the nature of the restoration objectives, and the actions needed to achieve these goals.

Restoration is a multi-disciplinary science, incorporating the skills of terrestrial and aquatic ecologists, environmental engineers, hydrologists, landscapers, planners, councils, interest groups, and local communities. These groups may have differing opinions on specific restoration goals. For example, Collier *et al.* (2001) predicted that planting the riparian zone with pine trees along a number of streams in a hill-country pasture catchment in the North Island would reduce stream water temperature and improve instream biological communities. If this were the goal behind the restoration, then riparian planting would have met this goal. However, they also predicted that sediment yield from the planted stream catchments would increase over a 25-year period, as the stream channels widened in response to the increased shade. Therefore if the goal of the restoration were to rehabilitate an estuary by minimising sediment input, then this type of riparian planting in the catchment would not achieve this goal. Proper identification of restoration goals is therefore essential for any restoration programme.

Once specific goals have been established, clear objectives that will help achieve these goals (Fig. 38.1) need to be developed. These objectives are met by specific activities. For example, planting the riparian zone is expected to improve instream conditions by providing shade, reducing overland runoff, minimising bank erosion where excessive shade is not a problem, and introducing coarse organic matter such as woody debris. Other activities include adding coarse inorganic substrates to streams, con-

stricting the channel in places to create small-scale variations in velocity, and adding flow refugia where animals can seek shelter during extremes of high or low flow (Zika and Peter 2002). Such actions are aimed at improving instream habitat conditions, especially in urban streams where engineering activities such as bank and channel reinforcing have greatly reduced the quality of the instream habitats.

Implementation and monitoring

Once restoration activities have been identified, they need to be implemented. Although local councils or other statutory bodies can carry out restoration activities, it is often beneficial to involve the wider community, using environmental groups and schools in activities such as tree-planting, rubbish removal, and riparian fencing. Such public involvement raises the awareness of the importance of the local stream, and gives the community some sense of guardianship over a section of stream.

The final phase of the restoration cycle is monitoring to validate the effects of restoration activities (Fig. 38.1). Ideally, a monitoring programme should be planned and implemented at least one year before restoration activities begin, especially if information on the effects of restoration efforts on sediment, nutrient transport, and hydrology are required. It is also desirable to establish a reference or control site outside (and preferably upstream) of the restored reach to compare with observed changes in the restored site. The most effective monitoring strategy is to simultaneously monitor control and restoration reaches, with data collected both before and after restoration—the Before-After-Control-Impact (BACI) design (Underwood 1991). Monitoring can be carried out either by trained professionals or by community groups using many of the waterway assessment tools currently available in New Zealand, e.g., SHMAK (Stream Health and Monitoring Assessment Kit; Biggs *et al.* 1998) or Stream Sense (E.W.) 1999. The choice of monitoring regime will of course depend on the goals of the restoration.

Monitoring the effects of restoration activities can be as important as the restoration itself, as it will determine the success of the restoration, the return on the financial investment, and will help educate the community on the relationships between physical habitat and biological community structure. Monitoring will also assist the implementation of future restoration projects by identifying activities that have succeeded or failed, and determining reasons for this. Restoration is not an overnight phenomenon, and long-term monitoring (3 to 5, 10 to 20+ years) may often be required to detect changes in biological communities. This is primarily because the resilience and resistance of biological communities in degraded streams may not be as high as those in non-

degraded streams. Accordingly, the expectations of land-owners, government, and the public need to be set at realistic levels in terms of the time required to achieve the desired effects (Howard-Williams and Pickmere 1999).

River restoration activities must take into account differences in the rates that riparian vegetation, aquatic plants, invertebrate and fish communities can recolonise a stream. Riparian succession may be relatively slow, even where riparian areas are planted. Factors such as floods, fires, and weed invasion will further slow this process, as will allowing plants to become established via natural dispersal. In comparison to riparian plants, aquatic plants will colonise restored streams much more quickly. Algal biomass and species composition can recover within 4–8 weeks following a disturbance (see Stevenson *et al.* (1996) for a review), while macrophytes can colonise restored stream sections within 10–20 weeks, as fragments drift downstream and become rooted in the streambed (Riis and Biggs 2003). Fish and invertebrate species may recolonise areas relatively quickly following stream restoration activities, providing there are no barriers to successful recolonisation.

Importance of scale

Land-use activities often affect individual species and communities, as well as ecosystem structure and function, in a hierarchical manner. A very localised land-use activity may initially place one stress on an ecosystem, adversely affecting only a small component. Increasing the intensity or extent of the activity may introduce additional stresses, which in turn will affect more components of the ecosystem. For example, removing overhanging riparian vegetation will increase the amount of sunlight reaching a stream, leading to an increase in water temperature and possibly an increase in algal biomass. In addition, the biomass of leaf litter entering the streams will decline. If stream temperatures exceed the critical thermal thresholds for stoneflies (i.e., $> 19^{\circ}\text{C}$: Quinn and Hickey 1990), or if their food resources decline, then these animals may disappear from the streams. Further catchment development such as conversion to pasture, use of fertilisers, and stock crossing streams will result in even greater changes to stream conditions. Such changes can lead to a loss of other taxa such as mayflies and caddisflies, leading to very large changes in the invertebrate community composition. These changes to the invertebrate community composition often mirror large alterations in the structure and function of the resultant ecosystem. For example, increasing catchment development reduces litter retention in

streams, as well as increase nutrient spiralling lengths (Young *et al.* 1994).

Restoration targets can be set at multiple scales (Bradshaw 1996; Fig. 38.3). At the smallest scale are efforts to bring back specific species (e.g., species of cultural importance such as freshwater crayfish, freshwater mussels, eels or mudfish). The next level in the restoration hierarchy involves efforts to restore specific communities (e.g., to restore an invertebrate community dominated by mayflies, caddisflies and stoneflies). Finally, higher levels of restoration deal with restoring ecosystem structure or function. Not surprisingly, the higher the goal in the hierarchy, the harder it will be and the more resources will be required for success. A cautionary note was given by Palmer *et al.* (1997), who suggested that re-establishing all potential species to a restored site will generally not be possible and recommended a degree of pragmatism when developing restoration goals. Conversely, the danger of single-interest restoration goals (especially for fisheries) is that activities that initially provide local improvements to habitat structure may later prove to be unsustainable, because they have taken insufficient account of other ecosystem processes, including those at the catchment scale (Boon 1998).

Restoration goals are also influenced by physical processes that occur at different scales, so restoration activities must be targeted at these scales (Fig. 38.3). Practitioners need to be aware of the spatial extent of their restoration activities and how these link to restoration goals. Restoration activities can occur at a small-scale, by manipulating local instream habitat conditions (e.g., by adding debris jams or planting riparian vegetation). These small-scale activities are those most commonly implemented, but they are largely driven by perceptions of what small-scale habitat features are important in a stream, or by budget or social constraints, rather than by scientific knowledge of the processes that led to the degradation of a particular community. If there is a considerable disparity

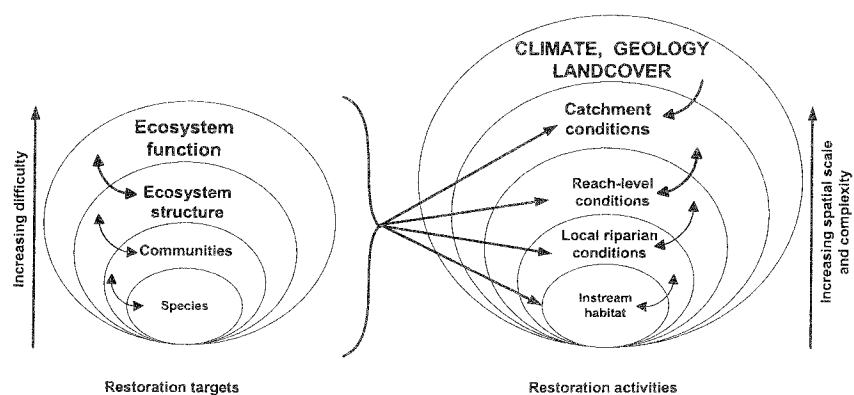


Figure 38.3 Conceptual diagram showing the importance of scale for both restoration targets and activities.

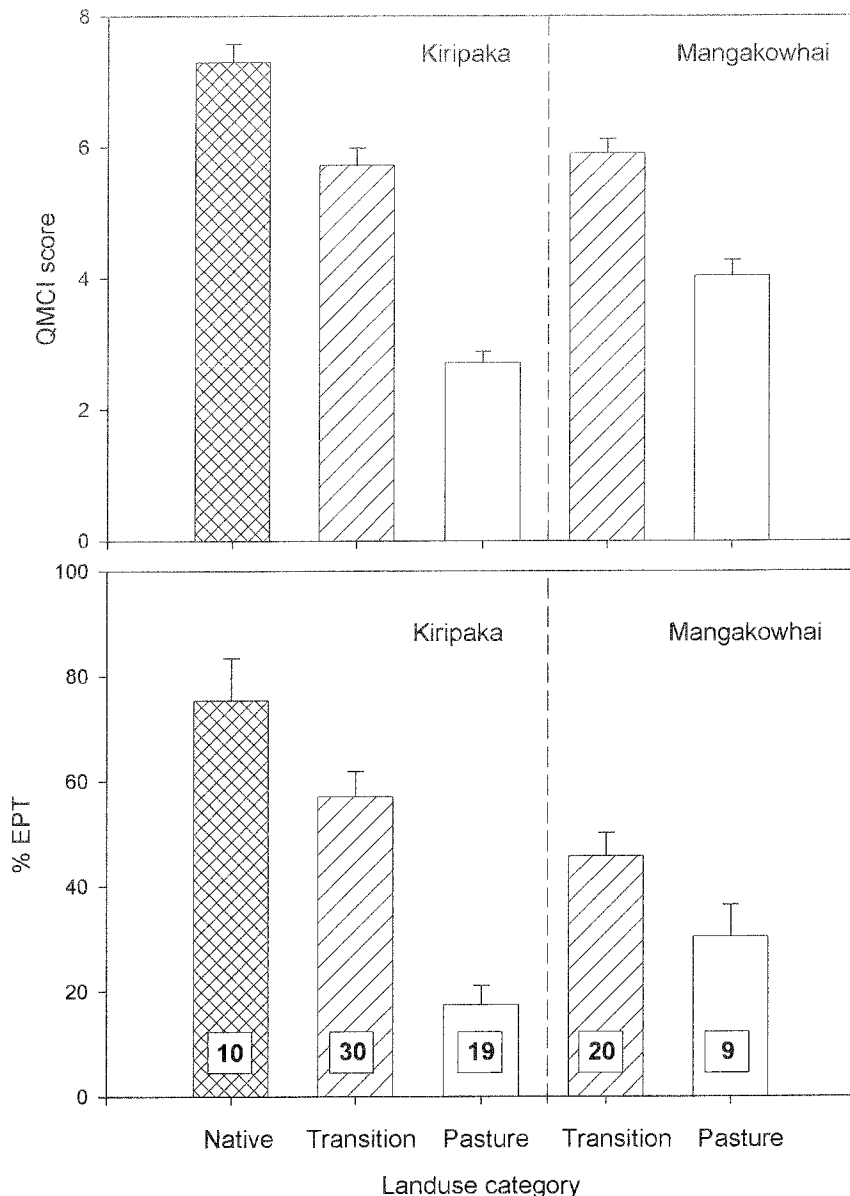


Figure 38.4 The effects of different riparian management strategies (undisturbed native bush, bush remnants [Transition], pasture) on the Quantitative Macroinvertebrate Community Index (QMCI) and the % Ephemeroptera, Plecoptera and Trichoptera (EPT) in two hill-country catchments near Hamilton ($\bar{X} + 1$ SE, n as per number in each bar). (Data from M. Scarsbrook.)

between the extent of habitat restored and the requirements of individual animals or communities (Bond and Lake 2003), small-scale restoration activities may fail to achieve their goals.

Restoration of reach-scale conditions is more problematic, but can be achieved, especially for physical properties such as stream shade and temperature. Studies have shown that shade and water temperature of open pasture streams that flowed through native forest remnants recovered to levels comparable to streams where no open pasture was present (Storey and Cowley 1997; Scarsbrook and Halliday 1999). Invertebrate community composition also changed

after a few hundred meters along the stream into these forest remnants (Fig. 38.4), suggesting that reach-scale restoration such as riparian planting of pasture streams can influence invertebrate communities. However, Scarsbrook and Halliday (1999) found that nitrogen processing, even 300 m into a forest remnant, was still much lower than in nearby control streams, suggesting that restoration at the reach level via riparian planting would not be successful in reducing total nitrogen loads in streams.

Being aware of catchment-wide processes, even when restoring on a small scale, is often necessary to achieve success. In streams in northwest Montana, USA, the decline of bull trout (*Salvenius confluentus*) and cutthroat trout (*Oncorhynchus clarki*) was originally attributed to a lack of suitable habitat (Bohn and Kershner 2002). However, a more comprehensive study of the catchment conditions showed instead that sediment inputs throughout the catchment were having a greater effect on these fish species than the lack of small-scale habitat. Appropriate restoration efforts were thus developed at the catchment scale to reduce these sediment inputs. Catchment planning can also help minimise habitat fragmentation and maximise connectivity along a river corridor, thereby maximising the outcome of the restoration project (Roni *et al.* 2002).

Approaching waterway restoration with a view of multiple scales will improve the likelihood of restoration success, and will help communities

develop more realistic restoration goals, based on a better understanding of what is possible. A catchment-wide approach to restoration does not necessarily imply that there will be a successful outcome, but it will allow any processes that may limit the effectiveness of the restoration activities to be identified.

Limitations to restoration

Bond and Lake (2003) recently reviewed habitat restoration in streams, and highlighted the fact that, despite the existence of well-defined planning for restoration, less attention has been paid to the ecological processes that

ultimately dictate the outcomes of the restoration work. In particular, they identified five broad aspects that need to be considered—barriers to dispersal of biota, temporal changes in habitat use, introduced species, long-term or large-scale driving processes, and inappropriate scales of restoration. These factors, and their relevance for restoration activities within New Zealand are discussed below.

Barriers to dispersal

Dispersal barriers that can limit the success of restoration activities can be regarded as either hard or soft (Bond and Lake 2003). Hard barriers include physical structures that prevent the longitudinal movement of animals along a waterway. Many New Zealand native fish require access to the sea as part of their life cycles, so the presence of dams, weirs or natural waterfalls within rivers can greatly affect the types of fish found in particular locations. Knowledge of these hard barriers and their influence on fish movements is helpful in setting restoration goals. For example, a 400-m section of piped waterway downstream from a restored urban stream in Christchurch was identified as the most likely reason why no fish were found in this site five years after restoration. The lack of response by the fish community here was in sharp contrast to four other restored urban streams in the same area, where the densities of most fish increased markedly (Table 38.1).

Soft barriers represent the physical distance between restored sites and the nearest potential colonists. Many stream restoration projects often fail to consider the need to maintain a degree of connectivity between restored sections and areas where animals and plants are naturally present. Smith and Collier (2001) studied the dispersal abilities of the New Zealand caddisfly *Orthopsyche fimbriata* in Auckland, and found distinct populations above and below the Auckland isthmus. They suggested that movement of *Orthopsyche* occurred primarily along

waterways, so the low number of streams in the heavily urbanised Auckland area would have hindered dispersal across the isthmus. Suren and McMurtrie (2003) also hypothesised that habitat isolation was a likely factor in the lack of change in invertebrate communities five years after habitat restoration in three Christchurch urban waterways. Between-catchment colonisation via aerial dispersal through land corridors would have been the only mechanism for recolonisation to these restored stream sections, but this was thought unlikely, especially given the lack of suitable riparian vegetation in the urban environment (A. Suren, pers. obs) that could act as habitat for adult aquatic insects (Collier and Scarsbrook 2000). Moreover, other conditions within the urban environment may hinder adult dispersal—for instance, bridges and culverts across streams may hinder upstream migration patterns (Blakely *et al.* 2003), lights from buildings and vehicles may disorient individual animals, and there may be a high mortality of adults from vehicle traffic if roads are close to flight paths.

Small, isolated patches of enhanced streams, especially in urban areas, are thus unlikely to be recolonised by sensitive aquatic invertebrates unless corridors of naturalised waterway sections and riparian vegetation link stream reaches throughout the urban landscape. Greater emphasis should therefore be placed on enhancing areas within selected catchments to ensure a continuous corridor of improved instream habitat and riparian zones where aerial recolonisation is likely. Careful catchment planning that minimises habitat fragmentation and maximises connectivity along a river corridor will maximise the outcome of the restoration project (Roni *et al.* 2002).

Habitat use

Streams contain a mosaic of habitats, not all of which are equally suitable for supporting particular organisms. Many fish and invertebrate species have quite different habitat requirements during different stages of their life

Table 38.1 Density of fish per 100 m² in control and restored sections of four Christchurch urban streams. Bold figures indicate sites where there was a significant difference ($P < 0.05$) in fish densities. Densities were generally higher in restored streams, with the exception of small shortfin eels in Smacks Creek.

Fish	Nottingham		Papanui		Smacks		Steam wharf	
	Control	Restored	Control	Restored	Control	Restored	Control	Restored
Shortfin eels (< 300 mm)	2.8	9.1	8.4	41.1	32.8	10.9	13.8	32.9
Shortfin eels (> 300 mm)	7.0	10.9	1.4	9.6	11.8	6.2	8.1	2.9
Longfin eels	0	5.2	1.4	7.5	1.6	2.7	0	0
Upland bully	71.8	50.3	4.1	18.3	0	0	0	0
Inanga	0	0	0	0	0	0	0	3.3

cycle. For example, studies of the habitat preferences of shortfin eels in four New Zealand rivers has shown that small individuals (< 100 mm) prefer shallow water, with a velocity of less than 1.0 m/sec and a coarse substrate. Larger shortfins (> 499 mm) however prefer deeper water, with a velocity of < 0.5 m/sec and a substrate of fine material (Jellyman *et al.* 2003). Any attempts at restoring instream habitat for shortfin eels would therefore need to consider these differing habitat requirements, and either try to incorporate a mosaic of different habitat types in the restored stream section, or else concentrate on creating habitat conditions for a particular life stage of the animal.

Animals also change their habitat requirements during times of high flow. Lancaster and Hildrew (1993a) showed that different habitats in an English stream had different degrees of shear stress with increasing discharge, and they identified patches of fast, variable and slow flow. The latter patches were considered to be flow refugia, where animals could escape high shear stresses during floods. Lancaster and Hildrew (1993b) subsequently showed that invertebrate densities, and densities of large stoneflies, remained higher in the flow refugia patches after periods of high and fluctuating discharge. The existence of similar small-scale refugia within New Zealand streams has not been fully established, although Francoeur *et al.* (1998) showed that microform bed clusters helped periphyton communities resist a large flood in an east-coast South Island gravel-bed river. They suggested that in other flood-prone gravel-bed streams, preserving or increasing the abundance of these microform bed clusters may be useful for increasing primary and secondary production. Small-scale refugia patches that are needed infrequently by biota may be critical to ensuring population stability in the face of natural disturbances such as floods. Restoration activities must therefore strive to identify such patches and consider reintroducing them if they are absent from streams.

Biotic interactions

Resources such as food and shelter in streams are often limited, leading to a degree of competition for them between animals and plants. Competition can exist among individuals of the same species, as well as between individuals of different species, resulting in reduced growth, a small final body size and fewer offspring for the weaker individual. The introduction of exotic species can add another dimension to competitive interactions in streams. Bond and Lake (2003) suggest that exotic species can potentially out-compete native species in restored sites, and indeed there are some examples where exotic species have out-competed native biota (Zelder 2000). Consequently, it may seem desirable to minimise the chances of introduced species arriving in restored streams. However, in some circumstances, the arrival of exotic species in

restored streams may be considered a success, especially if these species were not found in the sites prior to restoration. Within New Zealand, introduced salmonids have a high social value: indeed the Resource Management Act (RMA 1991) has explicit requirements for the protection of salmon and trout habitat. It has been argued that protection or restoration of salmonid habitat may also benefit other fish and aquatic invertebrate fauna that rely on similar habitat. However, there is also evidence that trout have a detrimental affect on some native fish populations, through competition and predation (Crowl *et al.* 1992; Chapter 17). Thus, the establishment of small vulnerable native species such as galaxiids may be inhibited by the presence of large salmonids. Consideration of such biotic interactions may thus be important in restoration activities that focus on re-establishing either native or exotic fish.

Although interactions between native and introduced species may affect restoration, biotic interactions among native species also deserve consideration. For example, the endangered Canterbury mudfish (*Neochanna burrowsius*) is unable to successfully establish or survive where other fish (either native or introduced) are present. McDowall (2002) contends that efforts to restore populations of the freshwater mussel (*Hyridella menziesii*), or kakahi, in rivers and lakes around New Zealand may well depend not only on meeting their general habitat requirements, but also on restoring populations of their host fish, koaro (*Galaxias brevipennis*). This is because the kakahi life cycle is dependent on the small parasitic larva (the glochidia) attaching to a host fish, before dropping off to establish its sessile life stage in soft sediment. The presence of the host fish, koaro, may in turn be inhibited by the presence of introduced brown trout. Indeed, the introduction of trout into Lake Taupo has severely depleted the koaro population (McDowall 2002). Thus, kakahi populations in Lake Taupo may never recover unless a suitable fish host can also successfully be re-introduced.

Large-scale factors

Most restoration activities are based on relatively small changes to riparian conditions or instream habitat. However, restoration practitioners must realise that processes operating at a catchment scale may limit the effectiveness of restoration activities. Although the "if you build it, they will come" hypothesis appears to be valid for stream restoration works in rural areas (Gortz 1998; Laasonen *et al.* 1998; Zika and Peter 2002), its applicability to urban areas is questionable (Hilderbrand *et al.* 1997; Suren and McMurtrie 2004). The lack of response of biological communities in urban streams to stream restoration programmes may reflect a combination of several factors.

- Factors such as catchment imperviousness may result in

large-scale changes in flow regimes, making it difficult for many animals and plants to persist.

- Stormwater runoff from buildings and roads is transported directly to waterways, resulting in high concentrations of heavy metal and polycyclic aromatic hydrocarbons (PAH), and sediment inputs in urban stream water may limit the survival of sensitive invertebrates.
- There may be difficulty in achieving connectivity with other restored or unaffected reaches for natural recolonisation by many aquatic species.
- Modifications to the riparian zone may create unfavourable conditions for adult aerial phases of many insect species.

These limitations imply that successful restoration of urban streams may be highly problematic, and can occur only if these constraints are somehow mitigated or remedied.

Inappropriate scales of restoration

Restoration activities often do not take into account what is an appropriate scale to restore. The spatial extent and quality of the restored section will undoubtedly influence the complexity of aquatic communities that can be supported. Consequently, even where the habitat has been improved in a section of river, aquatic communities may not respond because of insufficient habitat modification or the small size of the restored area. In urban catchments, in particular, close proximity of buildings or roads may limit restoration because it is impossible to recreate or reclaim the floodplain area, remove channel reinforcing to allow the channel to meander, or to establish any substantial riparian zone. In these sites, restoration may be limited to improving the landscape or social aspects, as opposed to any significant biological improvement. Purcell *et al.* (2002), for example, found that although adult caddisflies were frequently trapped along a restored section of an urban waterway in California, their aquatic larval stages were continually absent from benthic samples. This absence was thought to be due to inadequate habitat size for larval survival and population establishment, insufficient resources to support these animals, or lack of suitable egg-laying sites for adults.

RESTORATION ACTIVITIES

Restoring the riparian zone

The importance of the riparian zone to aquatic ecosystems cannot be overemphasised (Collier *et al.* 1995), and riparian restoration is an integral part of any waterway restoration project. Riparian zones influence the transfer of both energy (light and heat) and material (inorganic and organic) to streams, help modify waterway shape on a local scale, provide habitat for aquatic insects and fish, and help buffer aquatic systems from activities on the adjacent land. In rural areas many Regional and District Councils, and community groups, restore riparian areas either by fencing to exclude grazing stock or by planting. Riparian restoration stems from the simple concept that healthy riparian conditions lead to healthy streams. Howard-Williams and Pickmere (1999) showed that retirement and adaptive management of the riparian zone along Whangamata Stream in Taupo over a 24-year period led to an established, continuous native riparian wetland corridor along the entire waterway reach. This had attracted native rare birds such as fernbirds, controlled bank erosion and proliferations of aquatic plants, and protected trout spawning habitat. Collier *et al.* (2001) predicted that reforestation of the entire 259-ha catchment of the upper Mangaotama Stream near Hamilton would result in a 25% increase in Macroinvertebrate Community Index (MCI) values within 15 years. They also predicted that MCI scores would increase by proportionally less if less riparian planting was done (Fig. 38.5). Merely enhancing riparian zones does not, however, always cause an improvement in stream health. For example, Suren and McMurtrie (2004) observed little change to invertebrate communities in three Christchurch urban stream sections where riparian planting occurred five years before.

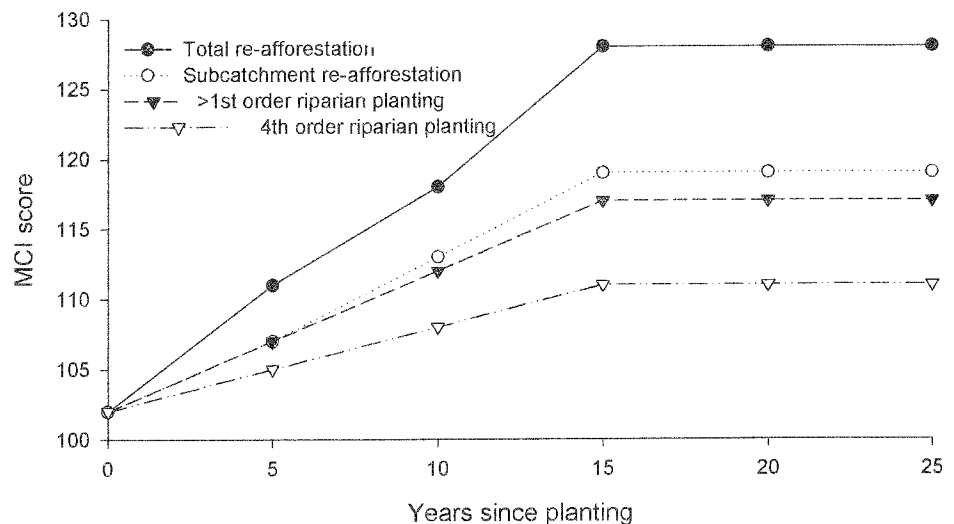


Figure 38.5 Predicted streambed area-weighted MCI scores for a small pasture stream in the upper Mangaotama Stream subjected to four differing riparian planting scenarios over a 25-year period (from Collier *et al.* 2001).

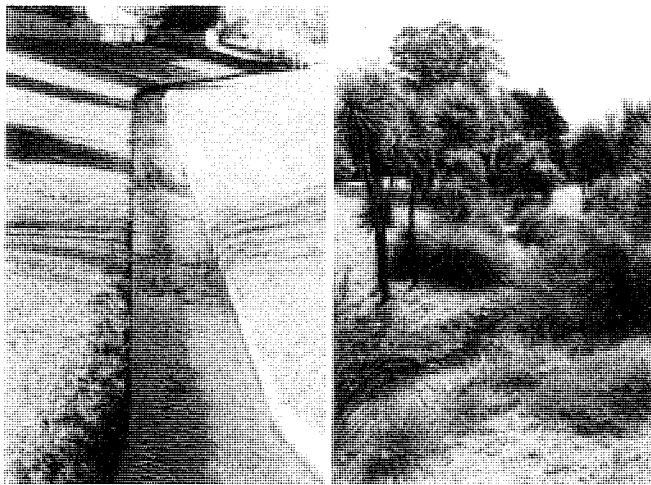


Figure 38.6 The aesthetic value of properly planned riparian planting is clearly evident in Jacksons Creek, Christchurch : A) an open utility concrete-lined section without riparian enhancement and B) an enhanced section 5 years after the re-instatement of a meandering natural channel and riparian planting.

Photos: Christchurch City Council and Shelley McMurtrie

Notwithstanding the ecological benefits, few people would doubt the visual and social improvements to streams that have undergone riparian restoration (Fig. 38.6). The effectiveness of riparian restoration will depend on many aspects, including the size of the riparian zone, the types of vegetation planted, and how much shade is generated as a result of planting.

Size of riparian zone

The influence of the riparian zone is greater in small streams than in larger rivers, reflecting the higher ratio of bank area to stream area in the former. The importance of riparian vegetation will also vary with its width, as wider riparian zones will be more effective in minimising overland runoff and removing nutrients than narrow zones. Valley morphology will also affect the degree to which riparian vegetation influences streams, with hill-country streams having more overland flow and higher stream power than lowland streams. The optimum width for the riparian zone will therefore depend on waterway size, catchment and geomorphology conditions, and the restoration goals.

Type of riparian vegetation

Terrestrial matter entering waterways is transformed into energy via leaching, microbial breakdown, and consumption by secondary consumers (mainly detritivorous invertebrates). However, litter inputs from exotic and native species differ in their timing, quantity and quality (Scarsbrook *et al.* 2001), and these differences

may influence river ecosystems. For example, the widespread planting of willow trees along many streams throughout New Zealand (West 1993) causes large inputs of leaves into streams in autumn, as opposed to the more even inputs (with a slight summer peak) of leaf litter from native evergreen species. New Zealand's aquatic invertebrates may not make effective use of the high autumn litter input from exotic deciduous trees, as large populations of shredding detritivores are generally rare. Indeed, large accumulations of decaying leaf litter can cause low oxygen levels in small streams during low flows (Hicks and Laboyrie 1999), which will be detrimental to many fish and invertebrates.

Leaf litter that falls into streams is broken down by a variety of fungi and bacteria (Winterbourn 1995), and invertebrates enhance breakdown rates (Rounick and Winterbourn 1983; Collier and Winterbourn 1987). Large differences exist in breakdown rates among different plants, although there appear to be no consistent differences between the breakdown rates of native versus exotic plants (Table 38.2). Moreover, studies have shown that many invertebrates show no clear preferences for leaf litter derived from either native or exotic trees (Linklater and Winterbourn 1993; Parkyn and Winterbourn 1997; but see Harding and Winterbourn 1995). These results suggest that it may be possible to plant a mixture of native and exotic vegetation as part of riparian restoration. A potential additional benefit of planting a mix of native and introduced species is that invertebrates can exploit periods of both autumnal leaf fall from exotic species and peaks in summer leaf inputs from native species (Cottam 1999; Scarsbrook *et al.* 2001). Finally, the nutritional and habitat value of particular leaf types is largely related to microbial activity, decay rates, and stream retention rates. Based on this, Hicks and Laboyrie (1999) and Quinn *et al.* (2000) suggest that vegetation with fast-decaying leaves be planted along streams characterised by floods and poor retentive capacity, because these leaves release their energy more quickly. Plants with slow-decaying leaves could be planted along streams that retain leaf litter for longer periods or in streams where longitudinal transport may benefit invertebrates in downstream unvegetated stream sections. Moreover, slow-decaying leaves can also provide substrate/habitat for invertebrates, which may be important in streams where a diverse substrate is lacking.

In addition to their influence on food and habitat resources in lotic systems, riparian plants are important for the terrestrial life stages of some aquatic insects. Although some adult insect species have been closely linked to either open native tussock/grassland areas or forested areas (Collier and Scarsbrook 2000), the relationships between adult insects and riparian plant species are still largely unknown. Riparian plants are also utilised by

Table 38.2 Rates of decay (k day⁻¹) of different native and introduced plant material placed into natural streams. Most of these plants are common in the riparian zone of streams throughout New Zealand. Information on decay rates obtained from published material (listed under Source).

Plant material	Type		Max k (day ⁻¹)	Source
Tussock (<i>Chinochloa</i> sp.)	Native	Grass	-0.005	Young <i>et al.</i> 1994
Wineberry (<i>Aristotelia serrata</i>)	Native	Deciduous	-0.0547	Quinn <i>et al.</i> 2000
Rewarewa (<i>Knightia excelsa</i>)	Native	Evergreen	-0.0167	Quinn <i>et al.</i> 2000
Poplar (<i>Populus x canadensis</i>)	Introduced	Deciduous	-0.0374	Quinn <i>et al.</i> 2000
Mountain beech (<i>Nothofagus solandri</i> var <i>cliffortioides</i>)	Native	Evergreen	-0.0055	Parkyn and Winterbourn 1997
Red Beech (<i>Nothofagus fusca</i>)	Native	Evergreen	-0.0225	Parkyn and Winterbourn 1997
Mahoe (<i>Meliccytus ramiflorus</i>)	Native	Evergreen	-0.0135	Parkyn and Winterbourn 1997
Oak (<i>Quercus coccolinea</i>)	Introduced	Deciduous	-0.0099	Parkyn and Winterbourn 1997
Elm (<i>Ulmus procera</i>)	Introduced	Deciduous	-0.0269	Parkyn and Winterbourn 1997
Willow (<i>Salix</i> sp.)	Introduced	Deciduous	0.016	Parkyn and Winterbourn 1997
Willow (<i>Salix babylonica</i>)	Introduced	Deciduous	0.0139 ¹	Collier and Winterbourn 1986

¹ Value is the mean of four results, obtained from two streams and using two different mesh sizes

some native fish species, and the spawning habits of inanga (*Galaxias maculatus*) among thick-growing grasses or native rushes in tidal reaches are well documented (Chapter 17). Native riparian planting can also enhance native bird habitat, as they provide food sources for birds during autumn and winter—times when most exotic species have lost their foliage and fruit (Christchurch City Council 2003).

Established riparian vegetation also contributes large woody debris to waterways, which influences channel morphology and retentive capacity. The size and effectiveness of woody debris varies greatly, depending on its source. For example, pines contribute more large woody debris to streams than native trees, but woody debris derived from native trees has a greater influence on channel morphology and hydraulic diversity (Baillie and Davies 2002).

Selecting plant species and genetic strains from the local area will enhance the chances of plant survival and reduce the risk of diminishing the local gene pool (Ministry for the Environment 2001). Thus, the practice of “eco-sourcing” native plants that are adapted to local conditions is recommended for riparian restoration.

Level of shading

Shading is particularly beneficial when restoring small headwater streams, as it can maintain low water temperatures, which in turn increases dissolved oxygen levels. Storey and Cowley (1997), for example, found that water temperature and dissolved oxygen levels of a stream flowing predominantly through pasture returned to forest-

stream levels within 300 m of entering a small remnant bush section. Rutherford *et al.* (1997) developed models predicting stream temperature increases along a 560-m reach of a small pasture stream near Hamilton as it flowed from native bush. Temperatures were predicted to increase from 8° C to 24° C on a sunny day where there was no riparian shade (Fig. 38.7). However, stream temperatures increased less as the amount of riparian shading increased (Fig. 38.7): a canopy shade factor of 50% would result in temperature increases to only 20° C, and a canopy shade of 90% was sufficient to prevent any increases in stream temperature from where it left the native bush (Rutherford *et al.* 1997).

When considering the desired extent of riparian shading, it is important to recognise potential erosion problems. Heavy shading of some streams can accelerate bank erosion, as thick ground cover such as grasses can die under heavy shade. Ground cover is often suppressed beneath coniferous trees as fallen pine needles accumulate. To counter these problems, Ministry for the Environment (2001) recommends shading be kept below 50–70 % along areas where streambank erosion is a problem. However, the erosion potential will largely depend on soil type, stream power and bank slope. Little can be done about the first two variables, but the bank slope can be reduced by re-shaping. Planting shade-tolerant native ground cover may also help stabilise banks to minimise bank erosion where heavy shade is preferred. The extensive root mats of some native trees can also help stabilise banks.

Shading can also suppress nuisance algal and macrophyte growth, as these plants are shade intolerant

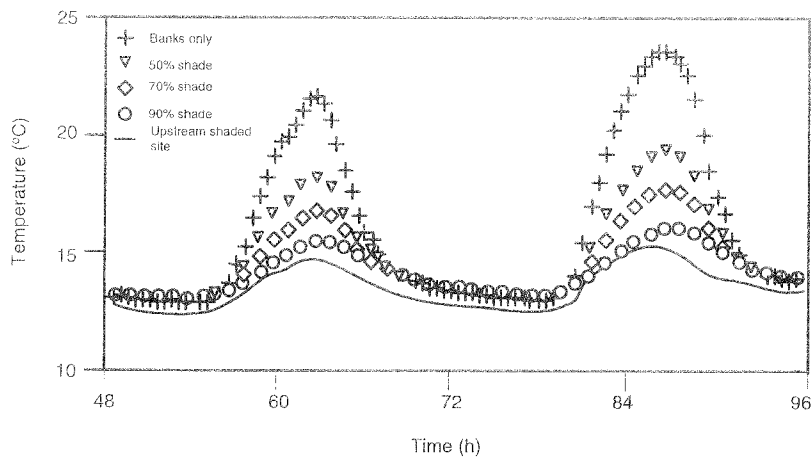


Figure 38.7 Predicted stream water temperatures under different planting scenarios (from Rutherford *et al.* 1997).

and can be maintained below nuisance levels with more than 50% shading of the waterway (Dawson and Haslam 1983). Shading may therefore be a more cost-effective means to minimise excessive macrophyte growth than mechanical removal. However, reducing algal and macrophyte growth can have detrimental effects on nutrient levels in streams. For example, in a 24-year study of vegetation and water quality changes in a retired pasture stream in Taupo, the highest nutrient levels were recorded at the end of the study, when riparian vegetation was substantially shading the stream and preventing macrophyte growth that would have otherwise used these nutrients (Howard-Williams and Pickmere 1999). Such a scenario may or may not be desirable, depending on the restoration goals.

Riparian influences on water quality

A restored riparian zone can have positive effects on water quality through its ability to remove nutrients, sediment particles, and toxic substances in overland runoff and groundwater flow via biological (adsorption) or mechanical (trapping) means. Haycock *et al.* (1997) provide an international overview of the efficiency of riparian zones for protecting waterways from sediment, phosphorus, nitrogen and pesticides.

The types of riparian vegetation also influence water quality. Parkyn *et al.* (2000) suggested that grass buffers may be more efficient than trees for filtering sediment and nutrients from overland flow, reflecting the greater surface contact of grasses to overland flow than trees. Conversely, Storey and Cowley (1997) found that remnants of native forest along New Zealand pastoral streams were capable of restoring water quality and aquatic invertebrate communities within a 600-m longitudinal distance. Burns and Nguyen (2002) found that plants rooted within a 1-m subsurface flow path could remove over 90% of nitrate ($\text{NO}_3\text{-N}$), whereas the nutrient-stripping capacity of

surface flows during heavy rain was negligible, largely due to the limited residence time and low interaction with organic soils. Ultimately, the efficiency of the riparian zone in removing pollutants from surface and groundwater flows is largely dependent on the interaction between surface flows with vegetation and litter; and groundwater flows with root systems and organic matter (Correll 1997).

Although riparian zones can provide some level of improvement in water quality, it is more effective to treat contaminated water in specialised water treatment systems, especially in urban areas, where stormwater systems discharge water directly into waterways, circumventing any interaction with the riparian zone.

Restoring habitat quality

Habitat conditions within streams draining unmodified catchments are commonly diverse and complex, with natural meanders and rough banks, overhanging riparian vegetation, partially or fully submerged debris jams, and streambeds with a variety of substrates of different sizes. A common consequence of land-use activities is a reduction or removal of instream habitat for fish and invertebrates. For example, many urban and rural streams have been straightened to improve their drainage efficiency, and gravel streambeds are often smothered by fine sediments from activities such as construction in urban catchments or stock-induced bank erosion in rural catchments. Consequently, in addition to riparian planting, restoration of instream habitat conditions is a high priority. Restoration of habitat quality can focus on both a stream's physical aspects (e.g., its morphology and the nature of the substrate), as well as the processes that operate within streams (e.g., providing refugia from disturbance, increasing instream retention).

Channel morphology changes

Unmodified waterways may have meandering channels, riffle-run-pool sequences, variations in width, and natural banks. Re-creation of these morphological attributes is often a major focus of restoration (Fig. 38.8). Restoring a river's characteristic patterns of stream morphology can increase a waterway's functional length, increase hydraulic energy dissipation, and improve habitat heterogeneity and nutrient spiralling properties. Baseflow water depth, velocity, and substrate size are the principle instream physical components determining benthic invertebrate distribution (Jowett 2000), and these factors can be manipulated by altering channel morphology.

Reinstating channel meandering and riffle-run-pools should also ultimately decrease bank erosion, as the resulting increased water retention time and hydraulic energy dissipation (especially in riffles) will mean there is less energy available to erode banks.

Reinstatement of meanders can involve large-scale changes to the whole stream channel or enhancing meandering patterns within an existing stream channel. In areas with sufficient floodplain area and no bank reinforcement, a waterway will naturally revert to a meandering pattern. This tendency can be encouraged with careful placement of flow deflection structures such as boulders, large woody debris, macrophytes and marginal plants. Halting maintenance and observing where natural areas of scour and deposition appear can facilitate correct placement of such structures. The time taken for natural meanders to develop can be too lengthy from a management perspective, so mechanical means may need to be employed to ensure meandering occurs in channelised streams (Fig. 38.8).

If restoration efforts involve channel modification, then theoretical equations can be used to help determine an appropriate meander frequency and riffle-run-pool placement (Chapter 8). Completely prescriptive methods of channel design and riffle-run-pool placement are not critical, as the channel will continue to alter its form, provided management practices do not interfere. The ability to create a meandering channel with riffle-run-pool habitats will also depend on the amount of land available and on the stream slope, which influences velocity/depth relationships and the presence of riffles, runs and pools (Jowett 1993).

Many urban streams are much wider than rural streams, and wider than their catchment size would normally dictate (Chapter 35). As a result, there is often insufficient

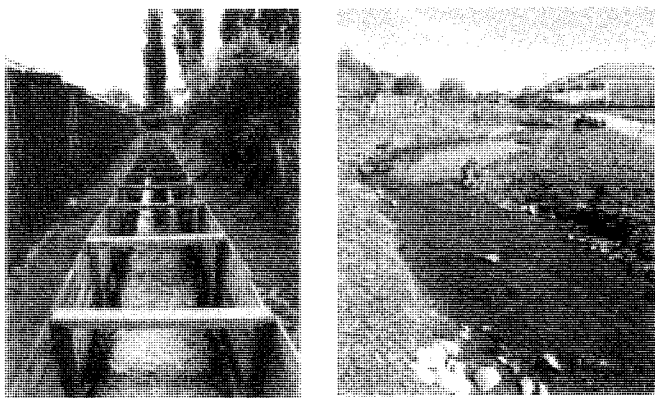


Figure 38.8 Papanui Stream, Christchurch: A) before restoration and B) after restoration. Urban streams are often straightened to increase their hydraulic efficiency, but their reinforced banks preclude the formation of natural meanders.

Photos: Shelley McMurtrie and Christchurch City Council

water in the channel to maintain suitable habitat during low flows. Varying channel widths and creating a low-flow channel (thalweg) in such streams will increase channel and hydraulic heterogeneity, and may ensure there is sufficient water depth for aquatic fauna, even during times of low flow. Because of the interrelationships between water depth, velocity and substrate type, and habitat preferences of invertebrates and fish, it is problematic to determine optimal flow conditions for biota. Jowett (2000) provides an overview of methods that can be used to determine flow preferences for particular benthic invertebrate species, showing depth and velocity preferences for some invertebrate species in small, medium and large waterways. These preferences should be considered when flow conditions are altered during restoration activities.

Reducing bank slope can reduce the likelihood of bank failure and thus reduce the quantities of sediment entering streams. Gently sloping banks also increase the bankful width of the stream channel, and provide a discrete floodplain area. Interactions between bank vegetation and other bank roughness elements will reduce water velocity, facilitate sediment deposition in the floodplain (instead of transporting it downstream) and dissipate energy that would otherwise erode banks. In urban areas where floods can be exaggerated and cause damage to nearby buildings, side-slope reduction is often necessary to compensate for the loss of flood capacity and increased drag caused by the establishment of riparian vegetation.

Large woody debris (especially debris dams) can be used to alter channel morphology, with hydraulic diversity created by scour pools downstream of the woody debris, or dammed pools upstream of it. Debris aggregates may also be more effective in creating pools than single pieces of large wood placed in streams. Moreover, large woody debris aligned perpendicular or obliquely to stream flow, and positioned on or angled onto the streambed from the bank, will influence channel morphology (Hilderbrand *et al.* 1997; Baillie and Davies 2002) and help reintroduce hydraulic variability in modified channels.

Inorganic and organic substrate additions

Substrate size influences benthic invertebrate abundance and taxonomic richness, so the size of inorganic substrate (e.g. stones) added to the streambed will greatly influence what fauna can colonise the area (Death 2000). In particular, sandy/silty substrates support the least diverse and abundant invertebrate communities, while gravel and cobble substrates support the most diverse. Similar relationships have been found for some native fish species (Richardson and Jowett 2002). Jowett and Boustead (2001) also showed experimentally that upland bully densities were positively related to the cover provided by cobbles. Placement of such substrates in streams where this material has been smothered by fine sediments may help

restore diverse fish and invertebrate communities, as long as further sediment inputs are minimised.

In areas with high water velocity and with sufficient slope, placement of coarse substrata will help form riffles. Pools, however, are generally depositional areas, so the addition of coarse substrata to these habitats may not be as useful—indeed the interstitial spaces between cobbles may fill with fine sediments. The size of any substrate material added to a stream as part of restoration needs to be carefully calculated, as small material may quickly scour away, while larger material may look unrealistic from an aesthetic perspective. Substrate stability depends on the gravitational restraining force exceeding the bed shear stress, and guidelines to determine bed shear stress and suitable substrate size are readily available (e.g., Christchurch City Council 2003).

Adding larger substrates (woody debris, large cobbles, or even boulders) to the stream edge or within the main channel may create stable habitat, as well as egg-laying sites for adult aquatic insects (Reich and Downes 2003) and spawning sites for fish. Giant kokopu (*Galaxias argenteus*), New Zealand's largest native galaxiid, are commonly found among accumulated logs and debris, which provide dense instream cover and areas of low velocity (Bonnet 2000). If used correctly, large woody debris can be used as a natural way to support banks and provide overhangs. For example, the Christchurch City Council has used hardwood tree stumps secured upside-down on the outside of bends in a recent waterway restoration project (Fig. 38.9).

Establishing aquatic vegetation

Aquatic macrophytes can play an important role in restoration, by providing in-stream structure and habitat. Macrophytes provide stable habitat for invertebrates and fish, and are of particular importance in slow-flowing



Figure 38.9 Large hardwood tree stumps were placed upside-down in the banks of Papanui Stream in Christchurch to create a stable overhang and interesting landscape feature.

Photo: Shelley McMurtrie

streams with soft substrates that otherwise have low habitat value. Unfortunately, in New Zealand the introduction of invasive macrophytes that can impede drainage has necessitated costly manual and mechanical removal in many rural and urban streams (Chapter 14). Although increasing stream shade may control these nuisance growths, active planting of native macrophytes may also minimise the growth of introduced species. This was the rationale behind the introduction of native macrophytes in Papanui Stream, a recently restored urban stream in Christchurch. Here, three species of native macrophyte (*Callitriche petriei*, *Myriophyllum triphyllum*, and *Potamogeton cheesemani*) were transplanted into riffles, runs and pools in a 100-m reach of the newly reshaped Papanui Stream. An upper 100-m unplanted reach acted as a control. During the 8-month growing season until July, when all the macrophytes naturally decayed, the planted section had a lower cover of the invasive *C. stagnalis* than the control section (Fig. 38.10). Moreover, the *C. stagnalis*

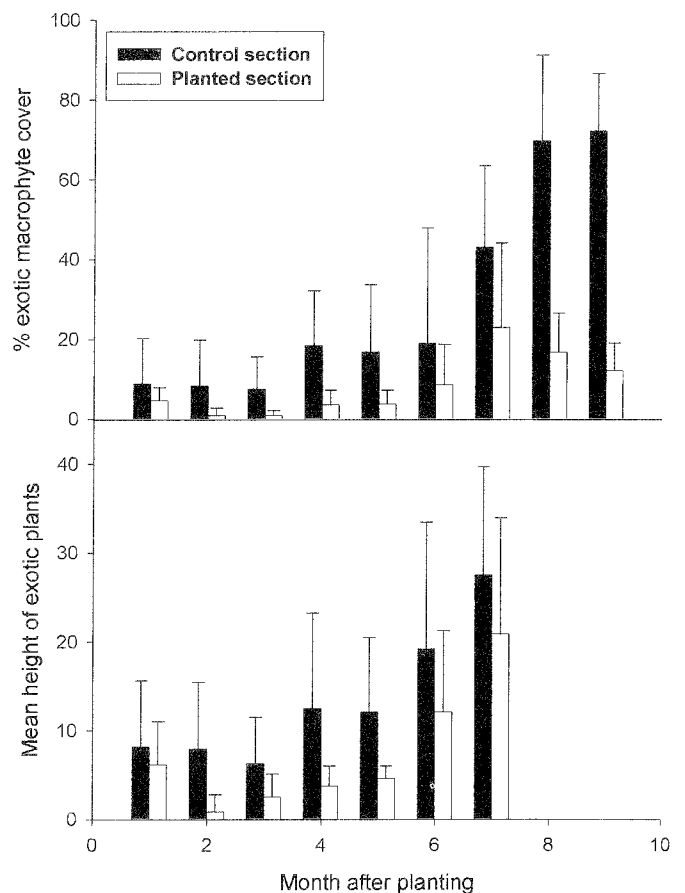


Figure 38.10 The effects of planting native macrophytes in Papanui Stream in Christchurch. Top: mean % cover ($x + 1$ se, $n = 12$) of the streambed of exotic macrophytes found in the section that was planted with native macrophytes (grey bars) or left as an unplanted control section (black bars). Bottom: the mean height of the introduced macrophytes that grew in each section.

that did grow in the planted section was lower than in the control reach (Fig. 38.10).

Restoration activities in Okeover Stream, in the University of Canterbury, have included selective maintenance of macrophytes. These activities have facilitated channel meandering, and the resulting hydraulic change has helped remove fine sediment and created patches of fast-flowing water over coarse substrates (Christchurch City Council 2003). Watercress has accumulated silt within its roots, further trapping sediment and re-directing flow to the thalweg of the channel. This has resulted in the formation of a deep, fast-flowing main channel, and side berms that can be planted with wetland plants.

Bryophytes are an overlooked aquatic plant in habitat restoration programmes. They provide optimal “nursery” habitats for early instars of many aquatic invertebrates (Suren 1991), have a high ability to retain organic matter (Suren 1992; Muotka and Laasonen 2002), and can alter near-bed flow regimes (Nikora *et al.* 1998) and provide hydraulic refugia during times of high flow (Lancaster and Hildrew 1993a). Indeed, Muotka and Laasonen (2002) concluded that, three years after restoration of several channelised forested headwater streams, the lack of bryophytes was likely a key contributor to the lack of recovery of detritivorous invertebrates to pre-channelisation levels. Addition of bryophyte-covered boulders to streams as part of enhancement activities may thus prove beneficial to invertebrate biodiversity in some streams.

Disturbance refugia

Refugia are stable areas that can be used by aquatic fauna during times of disturbance. They are important for the resistance or resilience of aquatic communities, as invertebrates and fish from these refuge areas can quickly recolonise a stream following a disturbance (Lake 2000; Collier and Scarsbrook 2000). Refugia can be created in waterways by adding coarse, stable substrates, large woody debris or bryophytes, and by preserving the hyporheic zone. Channel alteration may also create suitable backwater areas or other areas with low water velocities, especially during times of high flow. The importance of interstitial spaces within the streambed to a wide range of invertebrates is also widely recognised by ecologists, and their filling by fine sediment is an insidious consequence of urban and rural land use (Brunke 1999). However, restoring the interstitial spaces beneath cobbles and in the hyporheic zone is a particularly difficult challenge, as this zone is sometimes over-compacted by heavy machinery used in the channel, or can be smothered by fine sediment (Brunke 1999). However, enhancement of meanders and riffles may promote the scouring and sediment migration that can help maintain silt-free areas within the streambed.

Restoration methods need not completely mimic

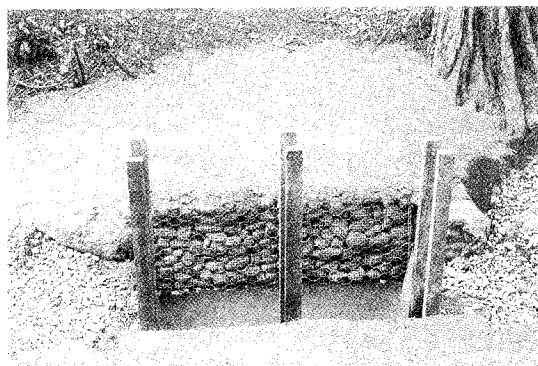


Figure 38.11 Gabion baskets in Okeover Stream in Christchurch were designed as low flow refugia for invertebrates and fish, within the streambank. Additionally, flow was redirected to create a still “backwater” habitat (top right).

Photo: Leanne O'Brien

natural conditions. For instance, wire cages filled with large river cobbles (i.e., gabion baskets) have been trialled in Okeover Stream in the University of Canterbury (Fig. 38.11) to create low-velocity backwater areas suitable for invertebrates and fish. Although these baskets are frequently regarded as an artificial solution to increase bank stability, the large interstitial spaces between the cobbles within them can be used by many different aquatic species.

Retentive capacity

Waterways need to retain organic material to provide habitat and food for aquatic invertebrates. Not surprisingly, the retentive capacity of a waterway influences the composition and density of its aquatic community. For example, shredding invertebrates are often absent or rare in streams with poor retention characteristics (Linklater and Winterbourn 1993 ; Winterbourn 1995). Modified waterways such as urban streams, especially those with uniform u-shaped channels with a low width/depth ratio, also have low retentive capacity as a consequence of their high drainage efficiency, and this can limit the diversity of invertebrate communities (Cottam 1999).

Improving retentive capacity is an important aspect of waterway restoration, and can have strong bottom-up effects on populations of detritivorous invertebrates (Haapala *et al.* 2003), with flow-on effects to fish populations. Probably the most important aspect of improving retention is to mimic the physical complexity of natural waterways. For example, Muotka and Laasonen (2002) found that structures used in the restoration of channelised forested headwater streams improved retentive ability compared to pre-restoration levels, but failed to achieve the retentive capacity of unmodified headwater streams. They attributed this to a lack of physical complexity in the restored sections (Fig. 38.12). Although alterations to

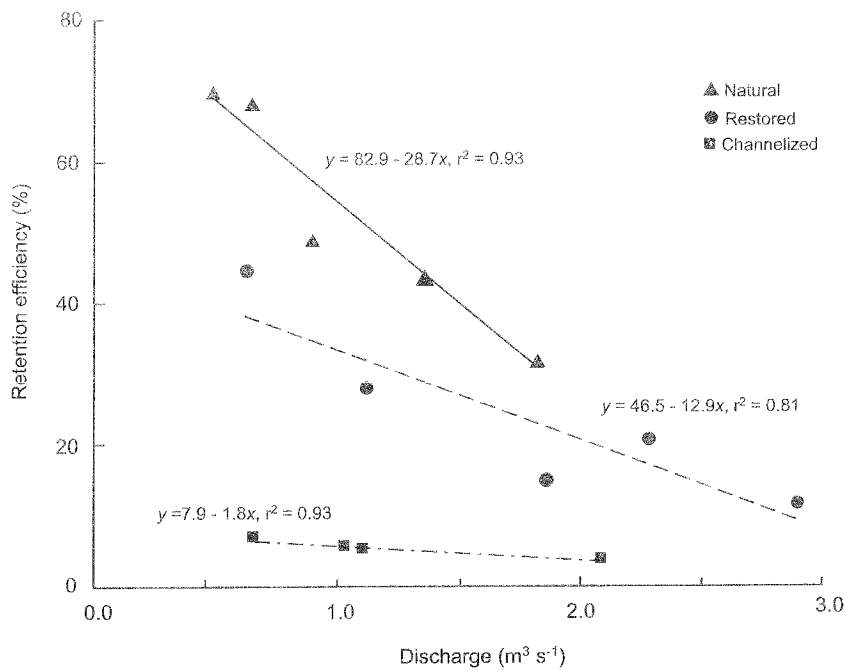


Figure 38.12 Comparison of retention efficiency at different discharges in natural, channelised, and restored sections in eight forested headwater streams in north-eastern Finland. The failure in the restored sections to attain retentive characteristics similar to that of the natural streams was attributed to a failure to mimic the physical complexity of the natural waterways (from Muotka and Laasonen 2002).

channel morphology to increase hydraulic heterogeneity will also promote scour and retention areas within the channel, retention structures must be added to fully maximise the retentive capacity of streams. Retentive structures can include biological features such as aquatic plants (macrophytes and bryophytes) and large woody debris, as well as the channel substrate, margins and backwaters. The substrate itself can be an effective retentive structure, although retentive capacity varies with substrate size and is negatively correlated with discharge. Substrates between 10–20 mm retain more small-sized detrital matter than other sizes (Rabeni and Minshall 1977). Coarse substrates (e.g., cobbles) also appear to be the most important retentive structures in open tussock grassland streams, where large woody debris is naturally absent (Scarsbrook and Townsend 1994). In forested headwater streams, woody debris (in particular debris dams) or, in some cases, bryophytes, are significant retention structures of sediment and organic matter, even during times of high flow (Baillie and Davies 2002).

Sediment control

The degradation of rivers by deposited and suspended sediment is of considerable concern both in New Zealand (Schofield *et al.* 2000) and worldwide (Waters 1995). High sediment loads can adversely affect aquatic invertebrate and fish communities (Ryan 1991; Richardson and Jowett 2002), despite otherwise effective habitat restoration. Controlling sediment loads requires decreasing sediment inputs, while promoting sediment movement and removal. To reduce sediment inputs, sediment sources must be identified and controlled, and localised bank erosion reduced through bank stabilisation techniques such as riparian planting and exclusion of grazing animals. Natural methods of bank reinforcement (e.g., the use of rocks or stumps) and bank slope reduction can also be used. In urban areas sediment inputs may be controlled by stormwater treatment systems, designed to remove sediment before it enters river systems.

Channel construction and riparian vegetation modification can also contribute significant sediment to waterways. Provided there are no significant upstream sources of sediment, river restoration usually produces an initial sediment pulse during the construction phase, and continued elevated levels of sediment derived from bare banks. This high sediment yield will gradually return to its pre-disturbance levels as vegetation slowly covers the stream banks (Fig. 38.13). Methods of reducing

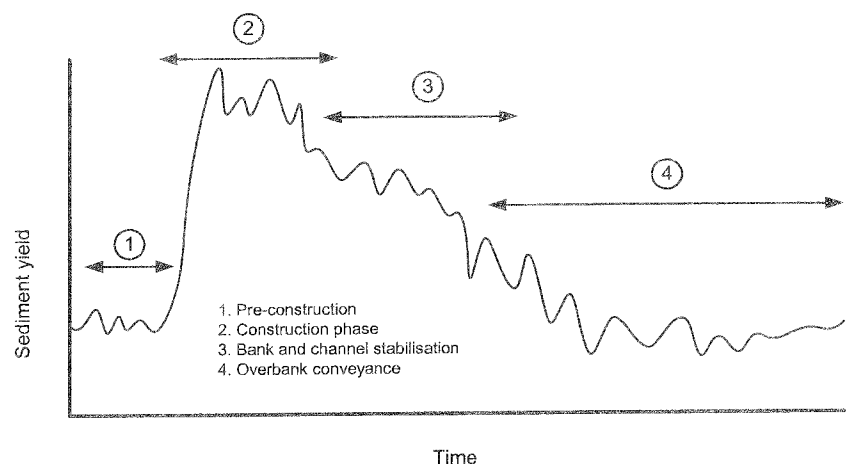


Figure 38.13 Model of sediments yields resulting from river restoration (modified from Sear *et al.* 1998).

construction-derived sediment include constructing as much of the new channel as possible without any surface water connections, and using artificial ground cover (e.g., terramat, woolmat) on stream banks to reduce surface soil erosion while riparian plants become established. In addition, containment measures to prevent construction-derived sediment from entering the channel should be undertaken within the immediate area (Auckland Regional Council 1999).

Once sediment is in a waterway, it can be partially controlled by facilitating its movement to areas where it may be more easily removed. High velocity areas can be developed to reduce the smothering of substrata, and pools can then be used as “sediment traps”, especially on the approaches to bridges and culverts. This can reduce the amount of sediment entering other areas of the waterway (Ministry for the Environment 2001). Sediment traps must be cleaned regularly, especially in urban areas where the sediment may contain pollutants such as heavy metals. However, discrete patches of sediment are not always necessarily detrimental to stream health, as such patches may be vital to species such as lamprey and large shortfins (> 499 mm), both of which show distinct preferences for substrates dominated by fine material (Jellyman *et al.* 2003).

Improving connectivity

Culverts, weirs, dams and other constructed obstacles along waterways can have profound consequences on aquatic fauna. Although these barriers can hinder the movement of fish and invertebrates, the effects are not always negative. For example, the removal of fish migration barriers downstream from a restored section can help to facilitate migration of fish species into a restored reach. However, it may also allow the immigration of other fish species that may out-compete or prey upon existing species. In many streams in Otago, non-migratory species of galaxiids are found only above barriers that prevent the immigration of trout (Townsend and Crowl 1991). If such barriers were to be removed, trout would gain access to these areas, to the detriment of the native fish populations.

The swimming ability of fish varies widely among species and life history stages, and is influenced by the swimming mode used. To develop “fish-friendly” weirs and culverts, the size and swimming ability of migratory fish species should be considered, and the structures should be designed to allow the upstream passage of the least powerful swimmers (Boubee *et al.* 1999). In New Zealand, inanga (*Galaxias maculatus*) are regarded as the weakest swimmers of the migratory native fish. Research into their swimming and jumping ability has highlighted that even a 150-mm weir can severely restrict their upstream movement (Baker and Allibone 2002). Fish-friendly

culverts should include features such as resting pools and weirs, or tailwater control devices that provide suitable slopes for fish passage (see Boubee *et al.* 1999; Christchurch City Council 2003).

CONCLUSIONS

Restoration ecology is a new, multi-disciplinary science that has only come of age as the pressures on our waterways have increased, and as different parts of the community have witnessed the things they value come under threat. Presently, restoration ecology is more of an art than a science, although its practitioners are beginning to rely more heavily on basic science to help answer the questions of what needs to be done to restore a waterway, what goals are realistic, and what are the natural constraints that may limit the effectiveness of restoration activities. Although the scientific and planning community can identify problems, set goals and objectives, and implement restoration activities, it is the natural resilience and resistance of biological communities that ultimately determine the success of restoration activities.

To ensure success, practitioners of restoration ecology must consider a number of aspects. Connectivity of a restored section of waterway to undisturbed populations is vital, so that individuals can immigrate to newly restored areas. Ongoing perturbations within a catchment may continue to affect restored sections, thereby preventing full restoration. It is important to consider the entire life stages of species or communities, as populations may fail to establish, persist and thrive if particular life stages are constrained. River restoration also needs to include the surrounding terrestrial environment. Lack of attention to these issues may explain the apparent lack of success of many waterway restoration activities, as physical, hydrological or chemical factors, or lack of potential colonists from unaffected sources may prevent the normal resilient recovery of communities from disturbances.

Although we cannot always halt development and the increasing pressures on our water resources, we can take definite steps to mitigate and remedy any adverse effects of our activities on waterways through restoration. Yet as restoration ecology is a developing science, so many restoration programmes do not achieve their desired goals. Scientists and restoration practitioners should relish the challenge to work out why, and develop ways to maximise the effectiveness of future programmes. Redefining restoration goals is also essential following a failed restoration attempt, as the original goals may have been unrealistic.

Restoration will never be a total cure for river degradation. Because of physical, financial and political constraints, restoration efforts may be sporadic, and at best

can only hope to create conditions approximating the pre-degradation environment. In many cases, diverting resources to conserving and preserving systems *before* they become too degraded will return better results for less effort than trying to restore systems once they have become degraded. The real challenge—and indeed one with great potential reward—lies in the wise management of our water resources, which will enable us to avoid their degradation in the first place. While river restoration will never become an obsolete activity, it would be nice to think that for some systems it will be unnecessary, given a more holistic approach to the use of our water resources.

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Chapter 37

Impacts of hydro-dams, irrigation schemes and river control works

Roger Young, Graeme Smart and Jon Harding

INTRODUCTION

Many New Zealand rivers are modified by large water projects. Water is used for irrigation, water supply and industrial purposes throughout New Zealand, but the amounts of water abstracted are often relatively small (less than 1% of New Zealand's total water resources; Statistics New Zealand 2000). Large-scale abstraction and storage of water for irrigation or water supply occurs primarily in drier parts of the country, or close to larger cities and industrial areas. In contrast, hydro-electric power schemes use 16–33% of New Zealand's total water resources (Statistics New Zealand 2000), but generally allow water to be returned to the same river system.

Water storage is often a key feature of large water projects. Storage can be “online”, where water is held back within the river channel by a dam, or “off-stream”, where water is diverted to a reservoir away from the river channel. The effects of large water projects depend to a great extent on the type and volume of storage involved. Online storage requires the construction of large dams and interferes with many natural processes within the river channel. In contrast, off-stream schemes might be expected to have more benign effects on the environment, but such schemes are uncommon due to the difficulty in locating suitable water storage areas away from the river channel and the costs of transferring water out of the river channel.

Other large projects associated with river systems include flood protection and river control works. Here floodwaters and channel movement are confined to a restricted area to protect the surrounding land from flooding and erosion. Such projects have been instrumental in providing sufficient security for the development of land within river floodplains and have resulted in significant economic benefits to New Zealand. However, confinement

of rivers can have negative effects on the ecological functioning of river systems, reducing the amount of available habitat and also altering natural geomorphic processes.

This chapter reviews the history of the development of large water projects in New Zealand, comments on current trends in project development and summarises the effects of large water projects on flow regimes, lake levels, sediment movement and water quality. Changes to the life-supporting capacity of habitats affected by large water projects are also examined, along with their effects on cultural, aesthetic and recreational values that are cherished by many New Zealanders. We also provide a short summary of how the impacts of large water projects are managed within the context of the Resource Management Act (RMA).

HISTORY OF LARGE WATER PROJECT DEVELOPMENT

Hydroelectric development

The hydroelectric potential of New Zealand rivers was recognised in the early 1900s. At first various small schemes were operated by private companies or local bodies, then the growing demand for electricity led to the construction of the first government power scheme at Lake Coleridge, which was completed in 1914 (Table 37.1). Development continued throughout the country, with Mangahao and Tuai supplying power to the lower North Island, Arapuni supplying Auckland, and Waitaki supplying various parts of the South Island. After a delay in construction activities through World War II, further stations were constructed on the Waikato, Clutha, Waitaki and Waiiau rivers. Much of the Tongariro Power Development occurred during the 1970s, with diversion of water

Table 37.1 Moderate and large (>10MW) hydroelectric power schemes (as of 2003).

Name	First operated	Installed capacity (MW)
Karapiro	1947	90
Arapuni	1929-46	197
Waipapa	1961	51
Maraetai	1953-62	360
Whakamaru	1956	100
Atiamuri	1958	84
Ohakuri	1961	112
Aratiatia	1964	84
Tokaanu	1973	240
Rangipo	1983	120
Kaitawa	1949	37
Tuai	1929-39	52
Piripaua	1943	44
Mangahao	1925	19
Matahina	1967	72
Patea	1984	31
Lloyd Mandeno	–	16
Ruahihi	1983	20
Wheao	1980	26
Cobb	1944-55	32
Branch	1983	11
Lake Coleridge	1915-29	45
Highbank	1945	25
Tekapo A	1951	25
Tekapo B	1977	160
Ohau A	1979	264
Ohau B	1983	212
Ohau C	1985	212
Benmore	1965	540
Aviemore	1968	220
Waitaki	1935-49	105
Paerau	–	10
Clyde	1992	432
Roxburgh	1956-61	320
Manapouri	1969-71	680
Waipori 2	–	54

from the Whanganui and Rangitikei headwaters into the Rangipo and Tokaanu power stations; this also resulted in increased generating capacity of the stations on the Waikato River downstream. The major development in the 1980s was the complex system of canals connecting the waters of Lakes Tekapo, Pukaki and Ohau in the upper Waitaki River. The last major development was the Clyde Dam on the Clutha River, which was completed in 1992 (Table 37.1).

Irrigation development

Prior to 1950, the majority of irrigation schemes were developed and financed solely by the government. Almost 100,000 hectares of land were irrigated by these schemes, the majority of which were dependent on mining rights to water in Central Otago, or the development of the Rangitara diversion race in mid Canterbury (MAF 2001). Between 1950 and 1980, problems with the use of existing schemes by farmers became evident. Future schemes needed prior commitment by farmers to ensure that benefits from the schemes were maximised, and it was recognised that the beneficiaries of schemes should contribute to the costs. Large areas of land were irrigated through this period. From 1980 onwards, the development of large community irrigation schemes slowed due to the removal of subsidies and loans to farmers and the tight financial conditions that faced farming during the late 1980s and early 1990s. However, with very dry summers in 1982 and again in 1984–85, there was rapid growth of irrigation on the Canterbury Plains, as individual farmers sank deep groundwater wells (deeper than 30 m) and used spray irrigation for crops and pasture. Central government no longer funded community irrigation schemes and local body reform and the implementation of the Resource Management Act virtually halted communal irrigation schemes (MAF 2001). Two notable exceptions to this were an 11,000-hectare development in the Waimakariri area and the 16,000-hectare Opihi augmentation scheme in South Canterbury. By 2002, around 500,000 hectares of land were under irrigation, largely in Canterbury, Otago and Hawkes Bay (Table 37.2).

Table 37.2 The area of land irrigated for agriculture in 2000 (from Hegarty *et al.* 2002)

Region	Area irrigated	% of total area
Northland	4,040	0.8
Auckland	6,500	1.3
Waikato/King Country	4,500	0.9
Bay of Plenty	9,341	1.8
Gisborne	5,000	1.0
Hawkes Bay	23,242	4.6
Taranaki	2,000	0.4
Manawatu/Wanganui	8,000	1.6
Wellington/Wairarapa	9,273	1.8
Tasman	7,920	1.6
Marlborough	12,087	2.4
Westland	0	0
Canterbury	347,022	68.6
Otago	65,090	12.9
Southland	1,500	0.3
TOTAL	505,514	100

Flood protection and river control works

The importance of flood protection and drainage works became evident as soon as European settlers attempted to develop river floodplains for agriculture. River Boards were set up in some regions as early as 1868, with the River Boards Act being passed by the central government in 1884. Numerous amendments and new acts followed, however most of the Boards dealt with only small parts of river systems. In some cases one Board would control one side of a river, while another Board would control the other side. To overcome this problem, Catchment Boards were set up under the Soil Conservation and Rivers Control Act 1941. Each Catchment Board exercised control over one or more large catchments and was involved with catchment-wide river control planning. Under the Resource Management Act 1991, regional and unitary councils are responsible for the avoidance or mitigation of natural hazards such as flooding. Throughout the country over 650,000 hectares of floodplains have been protected from flooding (Williman and Smart 1987).

CURRENT TRENDS IN PROJECT DEVELOPMENT

Low rainfall in the catchments of the southern hydro-lakes during 1992, 2001 and again in 2003, resulted in electricity crises involving substantial increases in the spot price of electricity, widespread calls for power savings by consumers, and subsequent calls for increased generation capacity. Knowledge that the Maui gas fields will run out earlier than originally expected has also prompted reviews of New Zealand's future electricity generation. The Kyoto Protocol's focus on reducing emissions of greenhouse gases will also encourage further moves to develop renewable sources of energy.

Several new hydroelectric schemes have been proposed. The largest is Meridian Energy's "Project Aqua" on the lower Waitaki River. This project is projected to cost around \$1.2 billion and will involve the construction of a 60-km canal between Kurow and State Highway 1, with approximately 70% of the river flow being diverted down the canal. The balance of the water is to remain in the existing river bed. Six power stations along the canal, each generating up to 90 MW of power, are proposed.

Brown (2002) reports that up to 400,000 hectares of land are currently under consideration for communal irrigation schemes. Market forces are expected to result in at least a 28% increase (144,000 ha) in area irrigated by 2010, with the majority of the increase predicted to be in Canterbury (57,000 ha increase) and Otago (36,000 ha increase) (Hegarty *et al.* 2002). Substantial increases in the amount of irrigated land are also expected in Waikato, Hawkes Bay, Wairarapa and Marlborough (Hegarty *et al.*

2002). Meridian Energy has proposed that Project Aqua be linked with several new irrigation schemes in North Otago, potentially irrigating up to 39,000 ha of land that is not currently irrigated. Integration of hydro-power and irrigation projects will probably become more common, as there are often mutual benefits in joint development.

EFFECTS OF LARGE WATER PROJECTS

Downstream flow regimes

Large water projects have caused dramatic changes to the flow regimes of many New Zealand rivers. The diversion of water to Doubtful Sound as part of the Manapouri Power Scheme, for example, has substantially reduced flows in the Waiau River downstream of Lake Manapouri, while diversion of water from the headwaters of the Whanganui and Rangitikei rivers, as part of the Tongariro Power Development, has increased flows in the Waikato River. Water storage reservoirs tend to attenuate flood peaks and change the annual distribution of flows by storing water at one time and releasing it at another. Flows in the Waitaki River, for example, are highly moderated by the presence of control structures on Lakes Tekapo and Pukaki. These lakes are capable of absorbing large floods and generally augment flows in winter when they would naturally be at their lowest (Waugh and Payne 2003). Fluctuations in the spot price of electricity mean that generation generally peaks for a few hours during weekday mornings and again in the late afternoon when demand is highest. If storage is available, water is held back during weekends and at night. These fluctuations in demand and generation result in substantial fluctuations in flow over short periods. Further details of the effects of large water projects on flow regimes are given in Chapter 7.

Lake levels upstream

Fluctuating water levels are a feature of all lakes and are controlled by a combination of factors, including catchment size, topography, climate, lake size and the characteristics of the lake outlet. In natural lakes, levels generally fluctuate by less than 5 m, while the levels of storage lakes controlled by large water projects tend to fluctuate more widely (Mark 1987; Fig. 37.1). For example, Lake Hawea historically fluctuated by around 3 m prior to control, and then by over 20 m after the lake was controlled (Mark 1987). Such large fluctuations in lake level are no longer considered acceptable, and the normal operating range is now restricted to 8 m (Freestone and Payne 2000).

Fluctuations in lake levels can have a variety of effects. High lake levels can cause erosion of shorelines, the inundation and mortality of terrestrial vegetation around the shore, and a decrease in available light to littoral and



Figure 37.1 The Cobb Reservoir after a prolonged period of dry weather.

Photo: Roger Young

benthic aquatic plant communities (James *et al.* 2002). Low lake levels can also increase rates of lake shore erosion and expose littoral communities to freezing, dessication and wave action (James *et al.* 2002). The ecology of lakes, and the effects of level fluctuations, is discussed further in Chapters 23, 24 and 25.

Storage dams also have the potential to increase the risk of flooding for communities upstream. For example, the presence of the Roxburgh Dam, and sediment deposited in the upper reaches of Lake Roxburgh, appear to have increased the risk of flooding in the town of Alexandra just upstream (Mackay *et al.* 2000). To address this problem, the level of the lake is lowered prior to floods and increased water velocities are used to flush sediment further down the lake. Some sediment is also carried downstream past the dam. This flushing program and the practice of drawing down the lake in advance of a flood has combined to reduce flood levels at Alexandra by approximately 1.7 m (Mackay *et al.* 2000).

Morphologic effects

The largest New Zealand rivers move millions of tons of sediment to the sea each year (see Chapter 12). Changes brought about by large water projects can significantly modify sediment transport, and hence river geomorphology (Fig. 37.2). The size of river sediment ranges from suspended clay particles that remain in the water column, to larger substrate materials that move along the riverbed. The finer the particles, the further they travel after being entrained into the river flow. In a flood, entrained clay particles may travel through the river system and out to sea, whereas gravel may only be transported from one river bar to the next. Thus, water projects can have both local and remote morphological effects.

The total sediment load carried by a river is often evaluated in terms of the suspended load and the bed load. At a specified location and flow there is an upper limit to the bed load that a river can transport. Scour and erosion can take place when the actual bed load is below the local transport capacity. Deposition occurs if the transport capacity is exceeded. Suspended load can be deposited in backwaters, on floodplains or within voids between bed particles. The types and rates of sediment erosion and re-deposition determine the morphology of a river system. Rivers can be single thread, meandering, braided or a sequence of steps and pools. Morphology is relatively stable when the long-term supply of sediment to a river is matched by a river's ability to transport sediment, but water projects can alter the balance between sediment supply and transport capacity and bring about serious morphological changes. More details about sediment transport in rivers can be found in Chapter 12.

Dams can result in improved water clarity downstream, as their reservoirs trap most of the sediment that previously passed through. Sediment trapping can cause adverse effects both upstream and downstream. Sediment deposition within the reservoir can reduce the intended storage volume and may raise the riverbed upstream. For example, the Waihopai Dam in Marlborough has trapped 10 million cubic metres of sediment since its construction in 1926 and increased bed levels upstream by up to 25 m. Downstream of a dam the combination of clear water and modified flow patterns can increase erosion and lower the river bed. Lowering of the river bed may endanger the structural stability of the dam, undermine banks and bridges and lower groundwater levels. Changes to the channel will eventually cease when the downstream river slope becomes flatter or when the bed becomes armoured with stones large enough to resist erosion. Figure 37.3 shows bridge piles that have been exposed as a result of falling bed levels in the Waikato River. Parts of this river are degrading because sediment supplies from upstream are trapped behind dams, and sediment starvation allows

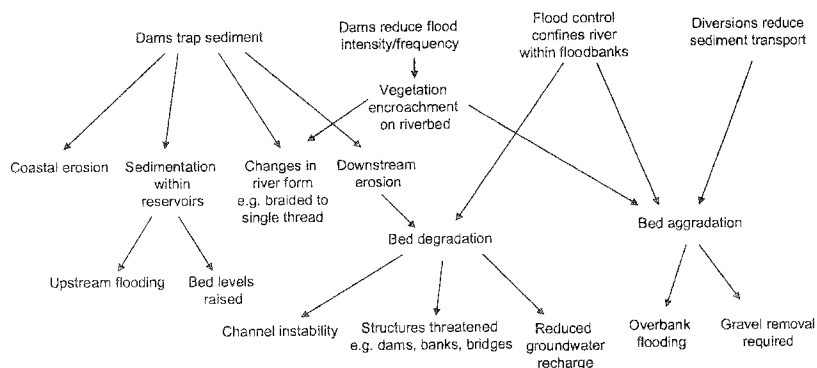


Figure 37.2 Conceptual diagram of the potential effects of large water projects on river morphology



Figure 37.3 Piles exposed beneath the piers of the Claudelands Rail Bridge on the Waikato River, Hamilton.

Photo: Graeme Smart

the river to scour and transport material from the bed and banks downstream of the dams, producing the effects shown in Figure 37.3.

Water diversion intakes can also affect the sediment regime of a river. If too much water is taken, the flows downstream of an intake may not be sufficient to move incoming sediment and sediment deposition will occur, raising the bed level. Channel instability and over-bank flooding may also occur. Sediment may also be diverted along with the water, causing problems. This occurred at the Poutu Intake of the Tongariro Power Development in the late 1980s, when 42,000 tons of sediment per year were diverted from the Tongariro River and had to be excavated from canals.

Islands within river systems are particularly susceptible to changes associated with flow regulation. For example, changes in sediment dynamics may cause island growth, migration or erosion. Some low-lying islands may be drowned when river depths increase. Conversely, if flows and depths are reduced, previously inaccessible islands may be easily accessed by the public and by introduced predators. This is a particular concern for local iwi, since many riverine islands are waahi tapu or offer sanctuary to rare species (MfE 1998).

Effects on coastal erosion

Where a dam intercepts the natural export of sediment to the ocean, severe effects can be seen in the coastal zone, as beaches may be depleted and coastal erosion accelerated. Concerns have been raised about the effects of sediment trapping by the large dams on the Clutha and Waitaki rivers, since both rivers historically contributed significant amounts of sand and gravel to the Otago and South Canterbury coasts. The potential yield of sand and gravel from Clutha River tributaries to the coast has been reduced

by 95% since the Roxburgh and Clyde dams have been in place (Hicks *et al.* 2000). However, the effects of this reduction on rates of coastal erosion are not clear. Rates of coastal erosion north of the Waitaki River mouth appeared to increase immediately after the construction of the Waitaki Dam, but have now returned to pre-dam levels (Hicks *et al.* 2002).

Geomorphic effects of flood control schemes

By protecting land from inundation, flood control schemes prevent flood flows from covering parts of their original floodplain. Floodwaters and their associated sediment load are concentrated into a channel or floodway. Not only is sediment that would have been deposited on the floodplains conveyed further downstream, but also the concentrated flows have a greater ability to erode and transport sediment. Local degradation and downstream sediment accumulation can occur. Narrowing of braided rivers sometimes has the opposite effect and causes rapid aggradation in the riverbed between the floodbanks. Because of this process, reaches of the North Ashburton River are now perched above their original floodplain. Substantial aggradation of riverbeds between floodbanks decreases the level of flood protection offered by the banks, while substantial degradation has the potential to undermine the foundations of the floodbanks, thus requiring expensive bank protection works. Riverbed levels also influence the rate of recharge to surrounding aquifers. For example, a 0.5 m drop in the lower Motueka riverbed is predicted to reduce summer recharge of the Motueka Plains aquifer by 24% (Basher 2003). Changes in the roughness of channel banks, such as occur when replacing scrub-covered banks with grass or rock, can also bring about large changes in the sediment-transporting characteristics of a river channel (Yu and Smart 2003).

Effects on water quality

Effects of storage on water quality

Storage of water in reservoirs often results in changes to the physical and chemical characteristics and temperature of the water released downstream (Table 37.3). The changes depend on a variety of factors, including the residence time (i.e., the average length of time taken for water to pass through the reservoir), the position of the reservoir within the catchment (i.e., in the headwaters or lower reaches), whether the water within the reservoir becomes stratified and anoxic, and the level of the reservoir outlet (surface or deep).

Generally, the longer the residence time, the greater the potential effects of water storage. As mentioned above, most of the suspended sediment washed into a deep reservoir will be trapped within the reservoir. Therefore water released downstream will generally be clearer, with

Table 37.3 Changes in downstream water quality associated with water storage

Parameter	Change	Comments
Suspended solids	Reduced	Most marked effects in large deep lakes. Sediment may be re-suspended by wind in shallow lakes.
Water clarity	Increased	Dense blooms of phytoplankton in lakes may reduce clarity at times Release of iron and manganese in some reservoirs may reduce water clarity downstream
Organic matter	Reduced	However, high quality organic material is often released from reservoirs
Water temperature	Reduced daily temperature fluctuations Increased summer daily mean temperatures in headwater reservoirs Seasonal buffering of temperatures in large lowland reservoirs Delayed seasonal minima and maxima	
Dissolved oxygen	Reduced	Depends on the degree of stratification
Phosphorus, Iron and Manganese	Increased	Particularly if bottom waters of reservoir are anoxic
Nutrients	Increased	Especially after inundation of fertile soils by a new reservoir

lower concentration of suspended sediment. The classic examples of this in New Zealand are the Roxburgh and Clyde Dams, which intercept much of the sediment from the Kawarau and Shotover rivers upstream. As well as geomorphic effects, the changes in water clarity and suspended sediment concentration can have marked effects on the ecology of river systems downstream.

Organic matter (e.g., leaves, wood and other detritus) can also be trapped within reservoirs, altering the composition and delivery of material to downstream reaches. Organic matter from upstream is often a key food source for many riverine organisms (Fisher and Likens 1973; Chapter 13). The interception of organic matter may interfere with the natural longitudinal changes in river ecosystems and the productivity of downstream reaches (Vannote *et al.* 1980). Ward and Stanford (1983) proposed the Serial Discontinuity Concept, which suggested that natural and artificial lakes “reset” many abiotic and biotic conditions below impoundments (Fig. 37.4). The downstream or upstream shift of a particular abiotic or biotic parameter caused by impoundment is defined as the discontinuity distance, while the magnitude of change is referred to as the parameter intensity (Fig. 37.4). The construction of a dam might be expected to reduce

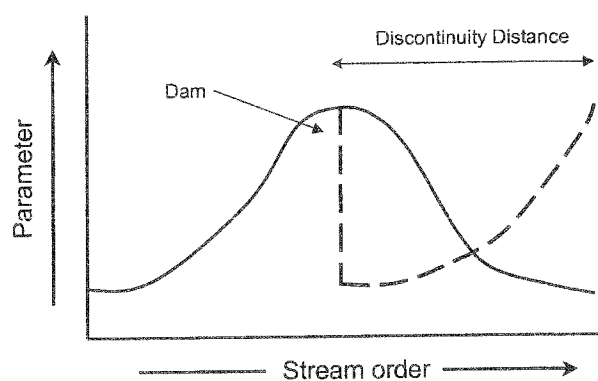


Figure 37.4 Conceptual diagram of the effect of dams in resetting biotic and abiotic parameters compared to a natural river system (redrawn from Ward and Stanford 1983)

biodiversity in the downstream river, for example, and maximum biodiversity may not return to pre-impoundment levels for many kilometres downstream.

A reservoir near the headwaters of a river catchment is predicted to have markedly different effects than one in the lower reaches (Ward and Stanford 1983). For example,

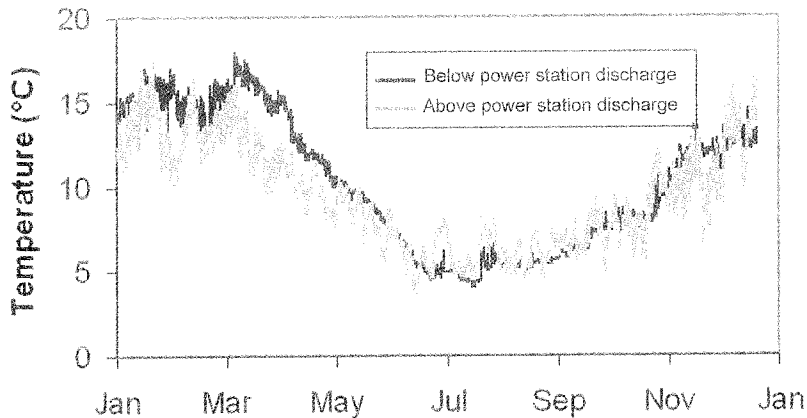


Figure 37.5 Changes in the annual water temperature regime as a result of storage in the headwaters of a river catchment (Takaka River, Golden Bay).

the water temperature of forested headwater streams is generally cool throughout the year due to shading by overhanging riparian vegetation and the buffering influence of groundwater contributions (Poole and Berman 2001). A dam in the headwaters of a river may allow more solar heating of the water, especially during summer. The Cobb Power Scheme in Golden Bay dams the headwaters of the Cobb River and water released from the dam is up to 4°C warmer than riverine water from a neighbouring catchment during summer, but very similar during winter (Young *et al.* 2000; Fig. 37.5).

In contrast, a large-volume reservoir in the lower reaches of a river will act as a thermal buffer. Water released will often be cooler in the summer, and warmer in the winter, than would have been expected naturally. Historically, the Green River in Utah, U.S.A., had extremely variable flow, sediment concentrations and temperature, and evolved a unique assemblage of native fish species adapted to these extreme conditions. The construction of the Flaming Gorge Dam resulted in major changes to the thermal regime of the river downstream, with much cooler temperatures in the summer and slightly warmer temperatures in the winter. This effect is exacerbated, as the water released downstream is from deep in the reservoir. Many of the native fish species that relied on warm summer temperatures as cues for spawning have been displaced downstream. Introduced cold-water species, such as rainbow trout, now flourish downstream of the dam (Filbert and Hawkins 1995). Such dramatic changes in thermal regimes have not been documented in New Zealand hydro dams. However, significant buffering of temperatures is expected to occur below large natural and hydro lakes.

Dissolved oxygen concentrations in water bodies are controlled by the balance between the supply of oxygen

from the atmosphere and photosynthetic inputs, and the consumption of oxygen by biological and chemical oxidation processes. Oxygen exchange with the atmosphere is reduced in deep reservoirs, relative to rivers, and may result in the deeper parts of reservoirs becoming anoxic, especially if the water column becomes thermally stratified. In addition, organic matter will tend to sink to the bottom of reservoirs, where it eventually decomposes, increasing the demand for dissolved oxygen. Water released from the deeper parts of reservoirs may be depleted in dissolved oxygen, with direct effects on biota downstream. However, exchange with the atmosphere will eventually bring oxygen back to equilibrium concentrations.

This may happen very quickly if water is passed through turbines, resulting in turbulent mixing of water and air.

The concentration of dissolved oxygen near the bottom of a reservoir also influences the concentrations of phosphorus, iron and manganese in the water (Wetzel 1983). As the concentration of dissolved oxygen declines near the sediment surface, the release of phosphorus, iron and manganese increases markedly. The change in redox conditions, caused by low oxygen concentrations, results in the reduction of insoluble iron and manganese hydroxides to more soluble forms, and the mobilization of adsorbed phosphorus. Dissolved iron and manganese will precipitate out of solution once oxygen concentrations increase. Therefore, iron and manganese flocs are sometimes seen on substrates downstream of power station discharges (Young *et al.* 2000). Increases in phosphorus concentrations may stimulate periphyton growth downstream if the availability of phosphorus limits their growth.

Water passing through or over dams may become supersaturated with gases. The pressure inside the turbines may be sufficient to force large quantities of gas to dissolve into solution. Alternatively, water plunging over a spillway will contain air bubbles that may be carried to a considerable depth, where the water pressure is great enough to force the gas into solution. If fish respire water that is supersaturated with gases, the excess gas may come out of solution in their blood as bubbles, resulting in "gas-bubble disease", which resembles the "bends" experienced by some divers. These bubbles lodge in various parts of the body and can cause injury or death. Major fish kills have resulted from this super-saturation below several dams overseas (Backman *et al.* 2002). To our knowledge, mortality due to gas supersaturation has not been documented in New Zealand.

Effects of diversion on water quality

Diversion of water from one catchment to another also results in the diversion of sediment, nutrients and organic matter. Therefore, any benefits or problems resulting from this material may also be transferred between catchments. Nutrient availability often determines phytoplankton production around parts of New Zealand's coast. Therefore, the productivity of some coastal areas around river mouths may depend on the supply of nutrients and organic matter from adjacent rivers, and could be affected if this supply is diverted elsewhere. This concern has been raised in relation to the Manapouri Power Scheme's diversion of Waiau River water to Doubtful Sound, and the potential effect on the ecology of the Southland coast.

Diversion of water also reduces the ability of a river to dilute wastes. This is particularly a problem with irrigation schemes, where the diverted water allows more intensive agriculture in the surrounding catchment. Runoff from this irrigated land then makes its way back into the river and is usually highly enriched with nutrients and other contaminants. The effects of diversion are essentially doubled, since the reduced flow of the river is less able to dilute these elevated levels of contaminants in the runoff.

Reductions in flow may also alter dissolved oxygen and temperature dynamics in rivers. Lower, more stable flows and slower velocities tend to allow higher biomasses of periphyton to accumulate. Assuming that more biomass means more photosynthesis, the amount of oxygen released during daylight and taken up at night by periphyton mats will also be increased. Under low flow conditions, this increased flux of oxygen will be dissolved in a smaller volume of water, potentially resulting in larger daily fluctuations in dissolved oxygen concentration if the amount of exchange of oxygen through the river surface is not altered. The thermal buffering effect of a large water volume will also be reduced at lower flows, potentially leading to larger daily fluctuations in water temperature.

Ecological effects

New lake formation

The impoundment of rivers and creation of artificial lakes causes dynamic shifts in pelagic and benthic communities in these new water bodies. Over time the community composition changes, with a reduction in lotic (running water) species and an increase in lentic (still water) ones. The formation of Lake Dunstan in Central Otago in 1992 resulted in the flooding of surrounding river terraces, which caused significant changes in energy transfer within the new lake, as terrestrial vegetation and organic material associated with the land decomposed. During the early years of the lake, the benthic fauna was dominated by chironomids and oligochaetes. However, as this source of enrichment dissipated, and aquatic macro-

phytes invaded, the community changed to one dominated by snails and caddisflies (Strickland *et al.* 2000). New reservoirs will usually be invaded within a few years by aquatic macrophytes. Lake Dunstan had not been completely filled when *Elodea canadensis*, *Myriophyllum* spp. and *Lagarosiphon major* were observed in the lake. By 1998, six years after the dam have been completed, a stable community of 16 aquatic plant species had formed throughout the lake. *Lagarosiphon* dominated in the 2–4 m zone (Strickland *et al.* 2000).

Not surprisingly, the formation of a new lake will create habitat for aquatic birds. Eight years after Lake Dunstan was filled, 25 species occurred around the lake, compared with 14 recorded prior to lake formation. New colonisers included the New Zealand scaup, Canada geese, Black swans, Australian coots, and Black and Little shags. Lake-level fluctuations resulting from changes in hydro-electric power generation have positive and negative effects on bird communities. These are discussed in more detail in Chapter 26. The most important impacts are on nesting activities of species such as grebes, coots, pukeko and black swans, which construct floating nests or nest close to the shoreline. Feeding can also be affected, as the raising and lowering of lake levels may reduce the availability of aquatic plants and benthic invertebrates as food, or alternatively make terrestrial food that was previously flooded available (Sanders 1996). However, these effects vary markedly with the lake morphology and riparian vegetation of the lake deltas.

Planktonic communities in artificial lake outlets

Regulated lake outlets frequently receive significant inputs of planktonic material from lakes upstream. Phyto- and zooplankton biomass in outlet rivers may be greatly elevated, depending on the type of impoundment and season. If surface water is released, it will contain plankton representative of lake surface waters, whereas hypolimnetic (deep-release) lakes may be devoid of plankton or have communities representative of the lake pelagic zone. Plankton biomass in outlet rivers decreases dramatically downstream, as material is either deposited on the riverbed or consumed by abundant benthic communities associated with the outlet (Monaghan *et al.* 2001; Fig. 37.6).

Benthic algae

Benthic algae or periphyton may proliferate after river regulation. Lake outlets generally provide an ideal habitat for algae, with reduced flood intensities, a moderate supply of nutrients from the lake, and high water clarity, allowing plenty of light for photosynthesis on the river bed (Young 1998). Furthermore, water temperatures are often buffered, and the stable "armoured" river beds that are characteristic of many regulated lake outlets provide an ideal substrate for periphyton attachment.

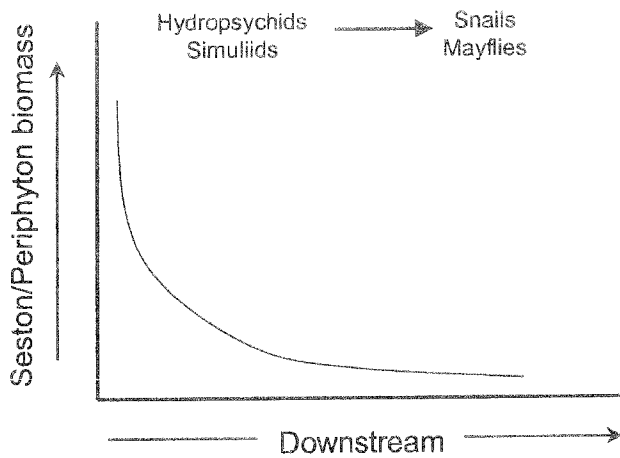


Figure 37.6 Model of the characteristic decline in seston downstream from the lake, and changing benthic community.

Diversion of water may also increase periphyton biomass downstream. Reductions in flow reduce the velocities and associated shear stresses on algal mats, allowing thick mats to accumulate. For more details on periphyton and factors controlling periphyton growth see Chapter 15.

Benthic invertebrates

Considerable research has been done on the response of benthic invertebrate communities to river impoundment (Fig. 37.7). Again, responses vary with the type of outlet. In general, outlets releasing surface water have communities with relatively low species richness, but exceptionally high densities and biomass. Production studies of lake outlet species have recorded high productivity, which is a response to the availability of

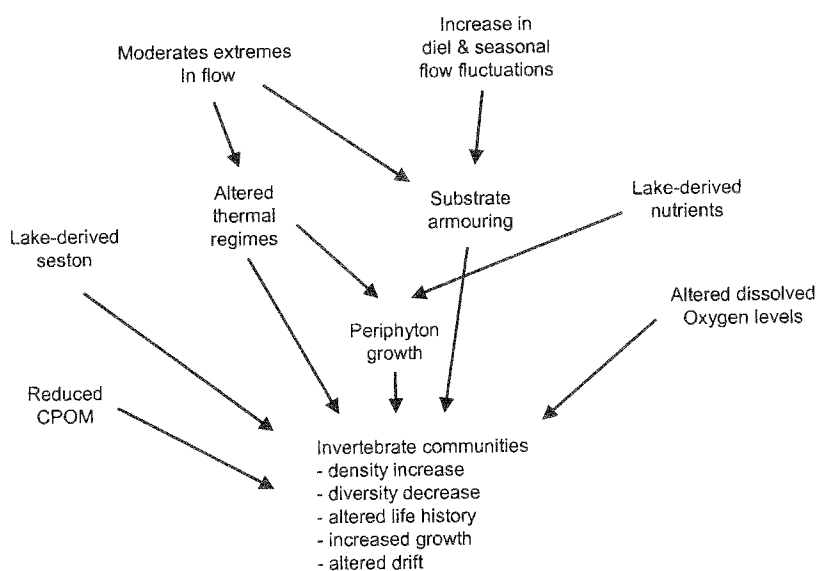


Figure 37.7 Conceptual diagram of the potential effects of river regulation on downstream benthic communities.

plentiful, high-quality food (i.e., lake plankton), relatively stable flow conditions and substrate, and buffered water temperatures, moderated by the upstream presence of a lake. These high densities of invertebrates can occur for some distance downstream in extremely large outlet rivers, however more frequently this response is relatively localised (e.g., 500 m to 1 km downstream). These low-diversity outlet communities are frequently similar to those found in natural lake outlets where similar physical conditions occur. The communities are usually dominated by filter-feeding species such as Simuliidae, and the net-spinning Hydropsychidae (Harding 1994). Densities as high as 10,000/m² have been recorded, and these are usually highest at the lake outlet, declining rapidly downstream. The plentiful food supply in these habitats has been shown to facilitate the co-existence of conspecifics, which might be expected to occur only under certain circumstances. Harding (1997) showed that two species of the Hydropsychid *Aoteapsyche* were able to co-exist on the same boulders in a lake outlet. One species, a more aggressive, territorially-dominant caddis *Aoteapsyche rarururu* occupied the high-quality food sites on the tops and sides of stones, while its conspecific *Aoteapsyche colonica* was restricted to the undersides of stones, where presumably there was less food. Both simuliids and hydropsychids have been shown to aggressively defend territory against other members of their species. However, the abundant food in lake outlets seems to suppress this competition in some species.

Benthic communities downstream of deep-release dams appear to be markedly different to those downstream of surface-release outlets. In these conditions, relatively little planktonic food may be released into the river, while water quality may be poor, with low dissolved oxygen, high concentrations of iron and manganese, and cool water temperatures. In extreme cases, these outlets may be devoid of benthic invertebrates or have communities almost entirely dominated by a few taxa. For example, Lake Waitaki has a deep-release outlet and 87% of the invertebrate community just downstream is a single net-spinning caddis species, *Aoteapsyche rarururu* (Harding 1994; Fig. 37.8).

As discussed earlier, the Serial Discontinuity Concept (Ward and Stanford 1983) predicts downstream changes in abiotic, and hence biotic, communities. As we have already seen, lake outlet communities are affected by several unique conditions created by the presence of the lakes (Harding 1992). The presence of a lake may enhance conditions for some

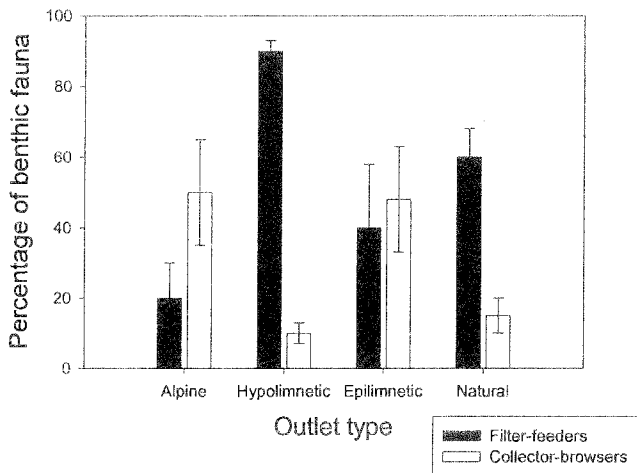


Figure 37.8 Relative importance of filter-feeders and collector-browsers in different lake outlet types (after Harding 1994). Alpine outlets were >1000 m asl, and Natural outlets <500 m asl.

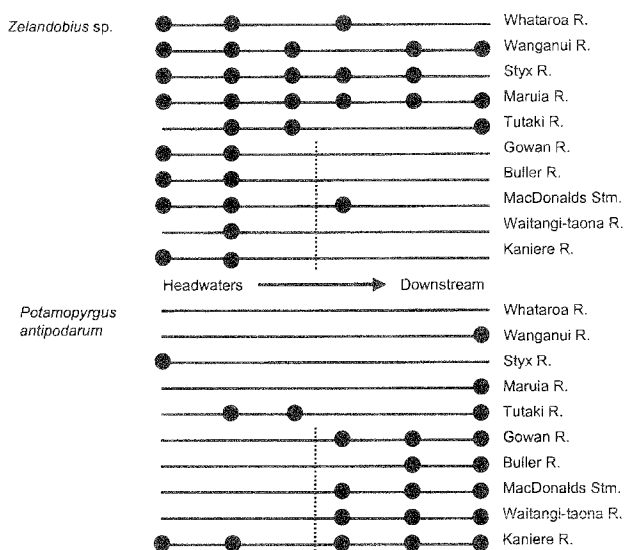


Figure 37.9 Discontinuities in the distribution of the stonefly *Zelandobius* sp. and the snail *Potamopyrgus antipodarum* down a river continuum due to the presence of an impoundment (Harding 1992). Dashed line indicates a lake or impoundment within the river system. Solid circles indicate sites at which species were present.

species, while creating unsuitable conditions for others (Fig. 37.9).

Regular fluctuations in flow resulting from changes in electricity demand may have detrimental effects on benthic invertebrates, particularly along the varial zone—the edges of river channels that are exposed to repeated wetting and drying. The effects are expected to be greatest in rivers with gently sloping margins, resulting in large areas of riverbed

being repeatedly inundated and exposed. Despite the potential importance of this effect on benthic invertebrates, there have been surprisingly few studies of it conducted in New Zealand. Irvine and Henriques (1984) found that invertebrate drift densities increased in response to flow changes in the Hawea River, while Irvine (1985) reported similar increases in drift and eventual depletion of the benthos after repeated fluctuations in experimental channels in the lower Waitaki River. More recently, a survey of fish-stranding on the lower Waitaki River reported lower invertebrate densities in areas that are regularly exposed during the fluctuating flow cycle compared with areas that were exposed only during an extreme low flow trial (Strickland *et al.* 2002).

Fish

Large water projects have had major impacts on fish communities in New Zealand. As well as indirect effects on water quality and food supplies, large water projects can directly affect fish migration, habitat availability, and juvenile survival, and can increase competitive interactions among species. Transfer of water between catchments also provides the opportunity for some species to colonise areas where they would not have occurred naturally.

Many of New Zealand's native fish species are diadromous, requiring access to the sea for part of their life cycle (see Chapter 17). Sports fish, like salmon and sea-run trout, also require access to and from the sea at times during their life cycle. Impoundments can thus severely disrupt migration and alter the distribution of species within and between catchments. With no recruitment of juveniles, eel populations upstream of dams will eventually disappear, although this may take many years. For example, eels currently found in the Cobb Reservoir are remnants from before dam construction in 1955 (Young *et al.* 2000). New Zealand's only sockeye salmon population in the Waitaki River initially benefited from hydro-power development, with Lake Benmore providing productive feeding grounds. However, the construction of dams on the Ohau River cut access between Lake Benmore and spawning streams at the head of Lake Ohau. Spawning runs of this fish have been reduced from over 18,000 in the late 1970s to fewer than 100 fish, which may be insufficient to maintain wild stocks (Graynoth 1995). The construction of the Waitaki Dam in 1934 on the Waitaki River and the Roxburgh Dam in 1956 on the Clutha River had similar dramatic effects for the large chinook salmon populations that once migrated from the sea up to the spawning tributaries and productive lakes in the upper part of these catchments.

Fish passes have been constructed on many New Zealand dams to allow upstream passage (Fig. 37.10). A few have been relatively successful for some species

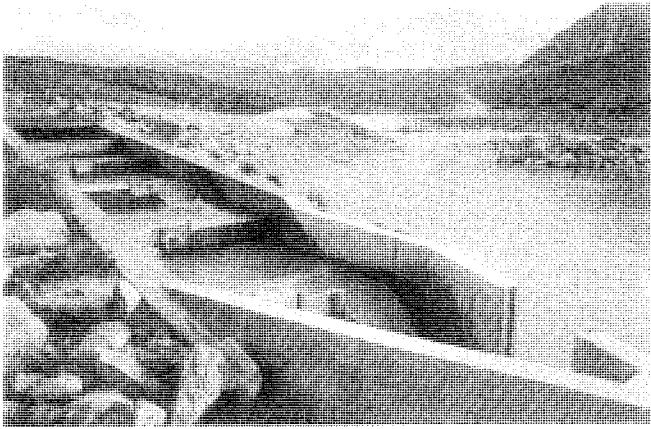


Figure 37.10 The Ohau River fish pass near the outlet of Lake Ohau.

Photo: Jon Harding

(e.g., the eel pass at the Patea Dam, the new vertical slot fish pass at the Mararoa Weir, Manapouri Control Structure), while most have been failures (e.g., the fish ladder on Waitaki Dam). In rivers with multiple dams (e.g., Waikato, Waitaki, Clutha, Rangitaiki) it appears to be more efficient to capture juvenile eels and manually transfer them upstream past dams (Boubee *et al.* 2001). However, downstream passage also needs to be considered if the aim is to provide future brood stock, rather than just maintain an eel fishery (Boubee *et al.* 2001). Many large eels suffocate on intake screens, or are killed as they pass through turbines. Nevertheless, some fish survive downstream passage. The sea-run quinnat salmon fishery in the lower Clutha River, for example, appears to be dependent on successful downstream migration of juveniles from landlocked populations in Lakes Wakitipu and Wanaka (James and Dungey 2000). Juvenile fish migrating downstream are also susceptible to being diverted into irrigation race intakes and may eventually end up stranded on irrigated pastures.

Diversion of flow by large water projects reduces the wetted width of residual river channels, and the depth and velocity of the water in that channel. Changes in wetted width reduce the total area of available habitat, while changes in depth and velocity affect the suitability of specific locations for particular species. Available habitat for species that prefer fast and/or deep water (e.g., adult salmonids, torrentfish, bluegill bullies) will be most affected by flow reductions. Fish, such as trout and koaro, that depend on drifting invertebrates for a large proportion of their energy intake are particularly susceptible to reductions in water velocity, since the rate of their food delivery directly depends on water velocity. Flow reductions may also make sections of rivers too shallow for fish migration, a particular problem for chinook salmon attempting to migrate up the shallow braids of Canterbury rivers.

Rapid flow fluctuations resulting from changes in electricity demand will also affect the amount of available habitat. The depth and velocity at a particular location may provide ideal habitat at one flow, but be too shallow or slow during the low-flow phase of the cycle and/or too fast during high flows. Mobile fish species may be able to deal with these flow fluctuations by moving laterally in response to changes in flow. However, their food resources (invertebrates) are not so well equipped and may be exposed during low flows or dislodged by high flows. Stranding may also occur, particularly of juvenile fish and species that tend to hide in river gravels (Almodovar and Nicola 1999).

An example of the complex effects of impoundment on fish populations is reported in Allibone (1999). The formation of Lake Mahinerangi on the Waipori River in Otago resulted in a previously diadromous population of koaro becoming landlocked. The change to lake rearing of juveniles, and decreased migration distance, allowed greater recruitment of koaro into the tributaries of the lake. This increase in koaro abundance has been linked with displacement of two species of non-migratory galaxiids from tributaries of the lake (McDowall and Allibone 1994). There now appears to be a dominance hierarchy in lake tributaries—brown trout are the most dominant species, but are only found downstream of waterfalls. Koaro are the next most dominant and can ascend waterfalls and displace non-migratory galaxiid populations further upstream (Allibone 1999).

Tunnels or water races built to transfer water from one catchment to another also act as conduits for fish to move between catchments. Such movement allows species to invade areas where they would not have occurred naturally. In several locations, aggressive species like brown trout and koaro appear to have wiped out non-migratory galaxiid populations, which would otherwise have been protected by downstream barriers (McDowall and Allibone 1994). Fish movement between catchments also allows interbreeding of closely related species, such as the various non-migratory Otago galaxiids, which almost never co-exist naturally (Esa *et al.* 2000).

Ecological effects of river control schemes and river confinement

The ecological importance of interactions between rivers and their floodplains has become more widely recognised over the last few decades (Bayley 1995). When floodplains are inundated, the habitats and food resources of the flood plains become available to riverine biota, potentially enhancing the biodiversity and productivity of river systems. For example, a large unconstrained floodplain in the upper reaches of the Taieri River is a major source of organic matter for the river downstream

(Young and Huryn 1997). During periods when the floodplain is inundated, fish move out of the river channel to exploit the abundant food resources on the floodplain. Stable isotope analyses of invertebrates and fish in this reach of the river indicate that the floodplain is the primary contributor of carbon and nitrogen to the riverine food web (Huryn *et al.* 2002). River control schemes break the connection between rivers and their floodplains by containing floodwaters within a narrow zone inside the floodbanks (Fig. 37.11). The importance of this break in ecological connectivity will largely depend on the geomorphology of the river system, since inundation has to occur regularly and for sustained periods to accrue significant ecological benefits. Therefore, the ecological effects of river control schemes are likely to be greatest in rivers with low-gradient floodplains and prolonged floods, but minimal in rivers with steep gradients and flashy hydrographs.

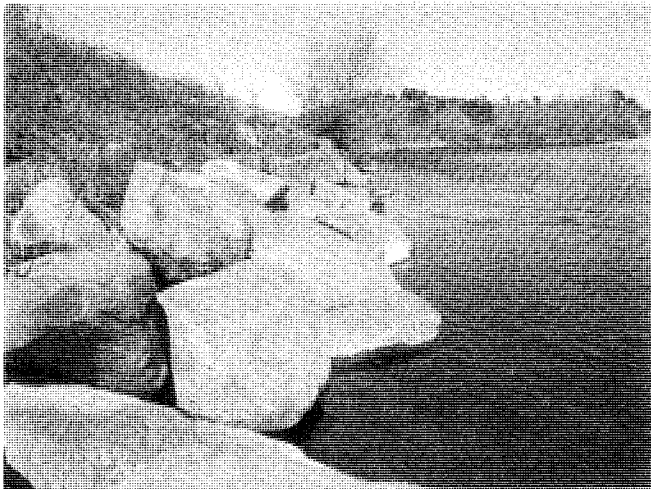


Figure 37.11 Bank control works confine the lower Motueka River to a single narrow channel.

Photo: Roger Young

Effects on people

People value waterways for a variety of reasons. A river may have important cultural values (mauri, mahinga kai, waahi tapu), be important for recreation (fishing, whitebaiting, kayaking, rafting, swimming, jet boating), and/or make a major contribution to the aesthetics of the surrounding landscape (MfE 1998). Large water projects have the potential to degrade these values if safeguards are not put in place. For example, flow regulation changes the seasonal pattern of flows, the strength of the connection between the mountains and the sea, and the ability of a river to carry sediment and dilute contaminants. All these changes will affect the mauri or life-essence of a river. Diversions resulting in the mixing of water from one catchment with another will also desecrate the mauri of rivers (MfE 1998). River-mouth blockages associated with

reduced flows, along with dams themselves, can adversely affect the passage of mahinga kai species between spawning and rearing areas. Many waahi tapu are located near rivers and lakes, and can be inundated by storage dams, or overgrown with weeds due to reductions in flood frequency caused by river control upstream. Kai Tahu considers that an estimated 90% of waahi tapu sites adjacent to Lake Hawea, Lake Dunstan and Lake Roxburgh have been lost because they were sited next to the river and are now drowned under the hydro-electricity lakes (Contact Energy 2001).

Many of New Zealand's large water projects have improved public access and created reservoirs, which are recreational assets for anglers, swimmers and boat owners. The network of lakes in the Waitaki Valley, for example, attracts numerous holidaymakers every summer. However, these improvements in recreational facilities have often been offset by the loss of recreational activities using the flowing waters inundated by reservoirs. For example, rapids on the Kawarau River that were popular for rafting and kayaking have been inundated by Lake Dunstan (Egarr and Egarr 1981). Changes in flow regime may also influence the quality of the recreational experience on some waterways. Reductions in water depth may expose debris, such as logs and rocks, posing serious risks for swimmers and paddlers. The safety of some swimming holes/jumping sites that require deep water may also be affected by reduced flows. Changes in water quality as a result of large water projects may also affect people's enjoyment of an area. An interesting example is the Tekapo River, where water clarity has increased now that turbid glacial water is diverted down the hydro-electric canals. While this increase in clarity and reduction in flow fluctuation has improved the trout fishery in the Tekapo River, local iwi are concerned that the stabilised flows have benefited salmonids and thus increased predation and competitive pressures on native fish species (MfE 1998).

Aesthetic values are determined by the landscape's physical and natural properties, coupled with the cultural values of the person experiencing the landscape (MfE 1998). Large water projects may affect the physical properties of a river by directly altering flow variability, water colour and clarity, hydraulics (width, depth and velocity), and the amount of riverbed exposed during low flows. Indirect physical changes associated with large water projects include modifications to riverbed vegetation and channel type (braided versus single thread) caused by changes in the sediment supply or flooding frequency. The extent of physical effects on aesthetic values will depend on the channel shape, with the most apparent effects on single-thread rivers with gently shelving margins. Very large changes in flow are required to change the wetted area of a river with near-vertical banks, while braided rivers

are naturally dominated by exposed gravel banks, so additional exposed areas are less conspicuous (MfE 1998). Other perceptual qualities of a river may also be affected by large water projects. Artificial structures, like dams and canals, will reduce the natural character associated with a river, while reductions in flow may influence the sound, smell and overall mood of a river. The wild and scenic properties of a large, raw, untamed river will be degraded by flow regulation and reduction (Godman 1989). Chapter 8 provides further discussion on “riverscapes”.

MANAGEMENT OF IMPACTS

Conflicts between the needs of ecosystems and human needs for freshwater resources are becoming increasingly common throughout the world (Poff *et al.* 2003). In New Zealand debate has intensified recently with the increasing demand for development of large water projects related to hydro-power and/or irrigation. Partly in response to these demands, Fish and Game New Zealand and other environmental agencies have applied for water conservation orders on several major river systems to recognize and sustain the outstanding values associated with these systems. The continuing development of regional water plans by regional councils has also heightened the level of interest in freshwater management.

With the introduction of the Resource Management Act (1991), consents for the majority of existing large water projects expired in 2001. The act requires that any effects of resource use should be avoided, remedied or mitigated. Therefore, substantial amounts of consultation and assessment of environmental effects have been required as a part of consent renewal. This has allowed many different groups to express their opinions on the pros and cons of specific schemes, some of which were developed at a time when little consideration was given to any associated ecological, social and cultural impacts. Consents that have been renewed have often included conditions to mitigate observed effects. A good example is the Manapouri Power Scheme, which initially involved the diversion of almost all of the Lake Manapouri inflow (mean 343 m³/s) to Doubtful Sound via the Manapouri power station. For much of the time only a small flow (0.29 m³/s) was released past the Mararoa Weir through the existing fish pass. Not surprisingly, this flow provided little in-stream habitat for aquatic life in the reach downstream, until flows were augmented by tributaries. After much consultation, new resource consents were granted for the scheme in 1996, and included conditions requiring a minimum discharge past the weir of 16 m³/s over summer and 12 m³/s over winter. The trout population has increased dramatically since the initiation of these minimum flow requirements (Fig. 37.12), although most of the change has resulted

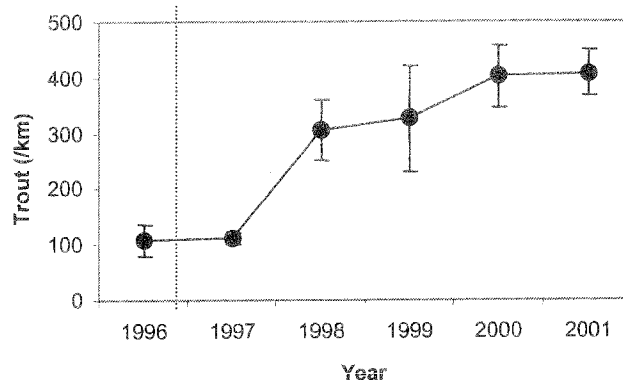


Figure 37.12 Trout abundance in four reaches of the Waiiau River downstream of the Mararoa Weir. Minimum flow releases from the Mararoa Weir began in December 1996 (Dotted line). Data from Moss (2001).

from increases in the number of small and medium-sized fish (Moss 2001). The consent also included conditions requiring monthly flow releases over summer for recreational purposes, and installation of an effective fish pass on the weir. Groups such as the Waiiau Working Party and the Guardians of Lakes Manapouri and Te Anau have been instrumental in the consultative process that led to decisions on the management of the Manapouri Power Scheme (Sutton 2002).

We are not aware of any existing large water projects in New Zealand that have failed, or are likely to fail to renew their consents. Nevertheless, this is a possibility and will become more likely as projects approach their life expectancy. Decommissioning of large water projects is becoming relatively common overseas, particularly in the United States, where more than 500 dams have been removed to achieve specified ecological goals (Poff *et al.* 2003). Dam removal is expensive, with both monetary and environmental costs. However, these costs are undoubtedly low compared with the potential costs of dam failure, as was demonstrated by the failure of the partially completed Opuha Dam in February 1997. The only dam removals in New Zealand have been of defunct water supply dams such as the Brook Reservoir near Nelson and the Wainuiomata River near Wellington.

Large water projects will continue to be proposed and involve considerable debate on the benefits and costs of such schemes. The Resource Management Act provides a framework encouraging consultation between developers, resource managers and other interested parties, and mechanisms for addressing adverse effects. The RMA also provides for the development of statutory Water Management Plans or Regional Plans, where issues of water allocation between competing demands, water use priority, instream flow requirements, a minimum flow

regime, flow sharing rules, and a cap on the total abstractive allocation can all be dealt with on a catchment or regional basis. Future advances in scientific understanding and model development may allow improved prediction of the impacts of projects, allowing the costs and benefits of schemes to be more accurately assessed prior to their construction. Striking the balance between costs and benefits is very important, since New Zealand's economy is highly dependant on the use of our water resources, while most New Zealanders want to ensure that the integrity of our freshwater ecosystems and recreational amenities remain intact for future generations to enjoy.

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Chapter 36

Impacts of mining

Jon Harding and Ian Boothroyd

INTRODUCTION

New Zealand has a long history of mineral extraction, beginning with the discovery of gold in the Coromandel Peninsula in 1852 and Central Otago in the 1861, and coal on the West Coast in the 1860s. Coal was first mined in 1872 at the Albion mine, and soon after underground mines were extensively developed along the Stockton-Denniston Plateau, North Westland, and in the Reefton area, where coal was discovered in close proximity to gold. In Central Otago widespread alluvial gold mining began in the 1860s, while the Ohinemuri goldfield opened in the Coromandel in 1875. Most of the early activity occurred in the Karangahake area, but in 1878 the rich gold-bearing quartz at Martha Hill in Waihi was discovered. Placer mining and gold dredging operations became widespread in Central Otago and Westland in the 1870s and these areas continue to be reworked as both technology and the price of gold improve (Fig. 36.1).

Other minerals that have been mined in New Zealand include gravel, tin, copper and uranium.

Of the mineral extraction methods, underground mining is the most widespread in New Zealand and has been used extensively in the coal industry, particularly on the West Coast. Underground mines involve excavating either an inclined shaft (a decline or drift) or a horizontal shaft (an adit), followed by parallel shafts (bord-and-pillar or longwalls), which enable the maximum amount of a coal seam to be extracted. The introduction of hydro-mining (using high-pressure water blasted against the coal face) improved the efficiency of coal extraction, while exacerbating the effects of mine drainage.

Several large opencast mines have been developed, including the Waihi (extracting gold) and Stockton (coal) mines. Opencast mining involves the removal of surface topsoil and rock to expose shallow mineral seams. It creates considerable waste material or overburden, which must

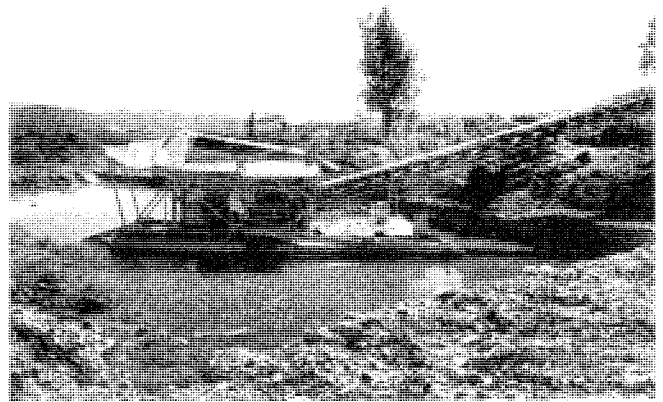


Figure 36.1 Placer mine gold operation in Carton Creek near Reefton.

Photo: Jon Harding

then be disposed of. Excavated soil and rock take up 50% more volume than the in-place material, hence back-filling of opencast mines still leaves substantial quantities of excess overburden (Fig. 36.2).

Placer or alluvial mining is the other major extraction technique used in New Zealand. Much of the current gold



Figure 36.2 Overburden dumps, Stockton opencast mine.

Photo: Jon Harding

mining in the South Island is alluvial mining. Gold fines eroded from seams in the mountains are washed naturally down to the valley floors. These heavy gold fines filter into the deep riverbed gravels, and alluvial mining operations excavate the riverbed, often down to bedrock, to sift out the gold.

Each of these three extraction methods can cause marked changes to mining landscapes, and consequently surface water and groundwaters associated with mines may be significantly affected by mine leachate, sediment and by mine operations.

In this chapter we discuss the effects of mining activities on the water quality, physical morphology and biotic communities of freshwater ecosystems. Increasing pressure for improved environmental management has resulted in a greater emphasis on the restoration and remediation of sites affected by current and historic mining activities, hence research is now beginning to focus on these problems.

Much of the mining activity in New Zealand is associated with running water ecosystems, and we will focus on these. No major lakes in New Zealand currently receive mine discharges.

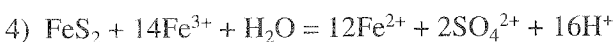
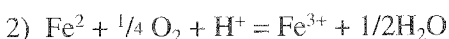
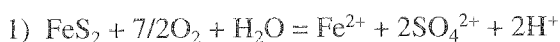
HYDROLOGICAL EFFECTS

Many early mining activities relied heavily on a continual supply of water, either to assist with excavation (as in hydro-mining) or to transport mined material to the surface. In Central Otago, for example, hundreds of kilometres of shallow canals were constructed to transport water from foothill streams to gold-mining digs. Initially, this water was used for washing and sluicing gold fines. Later hydro-mining was developed, in which high-pressure water jets were used to carve soil and gravels from the hillside. In addition, the manipulation of flow conditions is common in placer mining, where flow is temporarily diverted to expose streambeds to be re-worked for alluvial gold. The long-term hydrological impacts of these flow diversions have not been well studied. Prior to and during mining the riverbed is usually cleared of vegetation and riverbed gravels are turned over. In at least one instance on the West Coast, channel diversion lowered the water table, causing the diverted river to dry in late summer (Harding and Greenwood 2003). Placer mining may also involve diversion and storage of water in dredging and sediment-settling ponds, which may also have significant short-term effects on the water table.

In underground mining, the diversion of surface water and groundwaters is also common. However, the long-term effects of altered hydrological regimes are not well documented.

WATER QUALITY

Water chemistry associated with mining activities has been the focus of much investigation. Surface water and groundwaters associated with mines are often affected by acidification, the presence of toxic metals, and sedimentation. Discharges emanating from coal mines can cause the most severe problems. Many coal seams, such as those in the Brunner Coal measures in North Westland, contain pyrite, which has high levels of sulphur. When pyrite is exposed to water and oxygen, several well documented reactions can occur (Singer and Strumm 1970), resulting in the acidification of mine waters.



Thus the exposure of pyrites to oxygen is a crucial step in the generation of acid mine runoff (sulphuric acid), whereas in natural undisturbed coal formations, where exposure to oxygen is uncommon, acid generation is rare. Acid mine drainage is frequently cited as one of the most important environmental side effects of coal mining.

In environmental monitoring of acid mine drainage, acidity is usually assessed by measuring pH. However, pH is not a true measure of acidity, nor is it necessarily an accurate indicator of the extent of acid mine generation occurring in a system (Kelly 1988). More precisely, pH is a measure of the concentration or activity of hydrogen ions in a solution. In acid generation, the crucial factor is the *availability* of hydrogen ions to neutralise bases, hence, the determining factor controlling acidity is the excess of hydrogen ions over other ions. Thus “total acidity” (measured as CaCO_3) is a more accurate measurement of acidity than pH. However, at low acidity there is often a correlation between total acidity and pH, and realistically pH is easy to measure with standard field meters (Fig. 36.3). Above about pH 7 there is rarely any acidity.

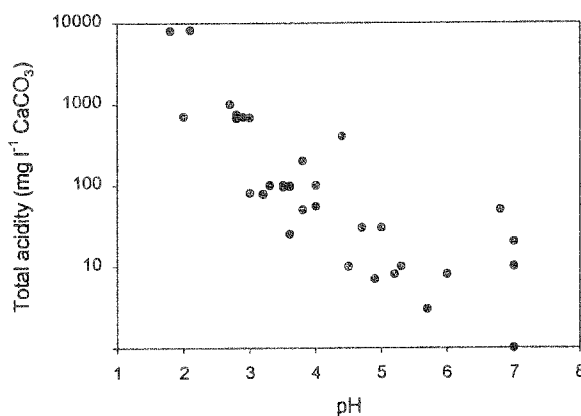


Figure 36.3 Relationship between pH and total acidity (after Kelly 1988).

Acidification can have severe effects on freshwater biota, however from a chemical perspective one of the most important side effects may be the reduction in the bicarbonate buffering capacity of the water. Once pH falls below about 4.2, carbonate and bicarbonate are converted to carbonic acid. The upshot of this is twofold. Firstly, stream water loses its capacity to buffer changes in pH, so that additional acid mine effluents entering a stream may decrease the pH markedly. Secondly, the loss of bicarbonate will have a negative impact on the many photosynthetic organisms that require inorganic carbon. Algae and bryophytes living at pH <4.2 need to be able to process free carbon dioxide in the absence of bicarbonate. Furthermore, even after acid mine inputs have ceased, it may take a stream or river many decades to recover from a loss of buffering capacity.

Lowering of pH can also have other chemical effects. Acidification tends to increase the rate of precipitation of silt and clays, and thus may increase sedimentation on the riverbed. However, depending on the pH, this may also have the positive side effect of reducing the turbidity of the water. Furthermore, increased acidification increases the decomposition of minerals, including feldspars and carbonates, resulting in the release of metals such as aluminium, and the release of silica. The release of silica may stimulate the growth of acidophilic algae, particularly diatoms, whereas aluminium may have several negative impacts. As the sources of acidity in mine drainages are strong mineral acids, acidification in these systems is usually associated with high conductivity (Fig. 36.4).

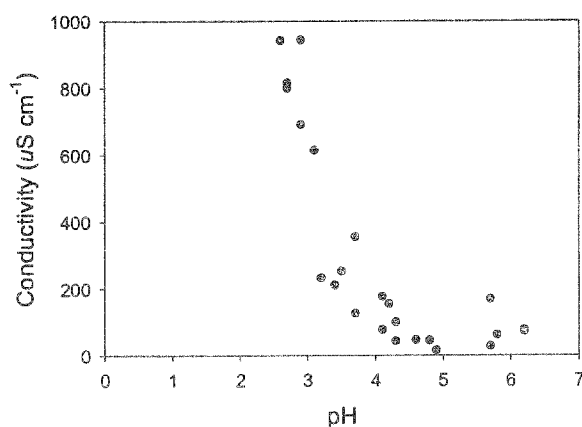


Figure 36.4 Relationship between stream pH and conductivity in 24 South Island streams (after Winterbourn *et al.* 2000).

Mine discharges may markedly reduce the pH of receiving waters, and in New Zealand adits with a pH <2.9 are not uncommon. A range of metals may also be associated with coal formations, including aluminium, arsenic, copper, iron, lead, nickel and zinc (Fig. 36.5). High

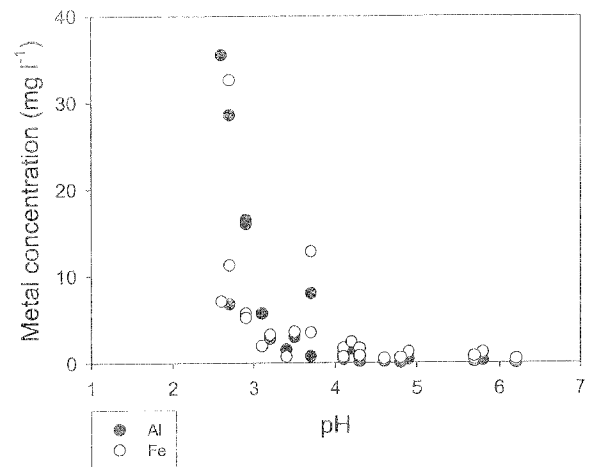


Figure 36.5 – Relationship between pH, and total reactive aluminium and dissolved iron, in 24 South Island streams (after Winterbourn *et al.* 2000).

acidity enhances the solubility of these metals, however as pH increases the chemical nature of the metals is affected. At very low pH, e.g., <3, any metals present remain in solution, however, as pH increases flocs of these metals may form, and if the metals are present in high concentrations precipitates form on the streambed. In particular, the yellow or ochre precipitates of iron (Fe^{3+}) (“yellowboy”) are commonly found in streams where iron is associated with exposed coal. This precipitate is composed primarily of iron hydroxide ($\text{Fe}(\text{OH})_3$), but may also include a complex of hydrates.

The critical pH value for iron precipitate formation ranges from about 3.5–4.3 (Table 36.1). Below this, Fe^{3+} remains dissolved in stream water and although it is toxic to some biota at high concentrations in this state, it does not cause sedimentation problems. Iron hydroxide precipitate can cause significant modification of the streambed by cementing substrata, clogging interstitial spaces, and covering biota. The substrate “armouring”

Table 36.1 Minimum pH values for precipitation of metal ions as hydroxides (Kelly 1988; Niyogi *et al.* 1999)

Metal	Minimum pH-hydroxide
Sn	4.2
Fe^{3+}	3.5–4.3
Al	4.9–5.4
Pb^{2+}	6.3
Cu^{2+}	7.2
Zn	8.4
Ni	9.3
Fe^{2+}	9.5
Cd	9.7
Mn^{2+}	10.6

effect caused by precipitate reduces refugia for invertebrates and fish, making them more susceptible to floods and disturbance and increasing the potential for predation and competition. Other metals precipitate in streamwater at differing pHs (e.g., aluminium), and several critical pH values are shown in Table 36.1. When acid inputs to a stream are stopped, these precipitates may re-dissolve and the substrate may recover in as little as six months (Niyogi *et al.* 1999). As we discuss later, Fe, Al, and other metals, both dissolved and as precipitates, may be directly toxic to freshwater biota.

Aluminium hydroxide precipitate, seen as a white coating on stones, has been widely documented in North America, whereas in New Zealand it has rarely been reported. However, dissolved Al is a significant problem in many catchments. On the West Coast of South Island naturally acidic brown-water streams are common; this acidity is produced by natural tannins (humic and fulvic acids) derived from decomposing vegetation in temperate rainforests. These naturally low pH waters (down to about pH 4) may have relatively high levels of dissolved Al, which is less toxic than in acid mine drainage waters because the Al is bound with organic carbon (Collier *et al.* 1990; Stenzel and Herrmann 1990).

Different metals have highly variable degrees of toxicity on stream biota, depending on localised conditions. Several workers have shown that Cu can be 13 times more toxic than Zn; however when these two metals occur together the cumulative toxicity can be far greater (Gray 1998). Similarly, numerous studies have shown that dissolved Al is more toxic than dissolved Fe at comparable concentrations. Winterbourn *et al.* (2000) reported concentrations of Fe that were three times greater than Al in the tissues of insects from 24 West Coast Streams. Fe can be photo-reduced by sunlight, and therefore open streams are liable to have Fe at less toxic levels (Niyogi *et al.* 1999). Not surprisingly, seasonal variations in discharges from mines alter the concentration and toxicity of receiving waters. Thus, a stream may be toxic at one time of the year and significantly less toxic at other times. Furthermore, heavy metals are transported in the water column both as dissolved free metal ions and complexes, and as metals bound to suspended sediments. In the Waihou River, Waikato, Cu, Mn and Zn were mainly in dissolved form in river water, whereas Fe was predominantly bound to suspended acid-soluble particulates in neutral pH waters (7.0–7.9). However, in the Waihou River estuary, almost all Fe and Mn had flocculated out, so high sediment metal concentrations occurred there (Webster 1995).

So far we have primarily discussed conditions associated with coal mining. However, alluvial mining may also release minerals associated with buried riverbed material

and cause significant turbidity in receiving waters. The successful use of a cyanide process for extracting precious metals was crucial to the further development of gold mining. However, the extraction process also results in other toxic materials (e.g., arsenic, antimony) being released. For example, X-ray diffraction analysis of sediments showed quartz, magnetite, hypersthene, ilmenite, anglesite, hornblende and cummingtonite were present in sediments in the Karangahake region of the Waihou River (Sabti *et al.* 2000). Waste products from cyanide-based metal extraction include toxic sludge. In 1895, following sustained pressure from mining companies, the Government declared (by Proclamation) the Ohinemuri and Waihou Rivers in the Coromandel as sludge channels (Watton 1995). This allowed the discharge of an estimated 250,000 tonnes per annum of mine tailings into the Ohinemuri River.

Placer gold-mining frequently results in significant increases in suspended sediment levels downstream. For example, turbidity has been shown to increase from 2.4 to >100 NTUs (Nephelometric Turbidity Units) in West Coast streams above and below placer operations (Davies-Colley *et al.* 1992). Increased turbidity reduced light penetration by as much as 400% and affected both benthic algal biomass by reducing photosynthesis and benthic invertebrate densities by degrading food quality (Quinn *et al.* 1992).

MODIFICATION TO THE PHYSICAL HABITAT

Gold dredging produces large amounts of sifted gravels, which historically, were deposited as tailings. Numerous West Coast and Otago rivers (e.g., the Grey, and Taramakau) have extensive areas of modified riverbeds. These tailings form convoluted ridge and valley hummocks along the river banks, frequently creating small artificial ponds, reducing or preventing riparian re-vegetation along rivers, altering the river channel morphology, and exposing minerals to weathering and erosion.

In contrast, opencast mining results in different, though no less obvious, modification of the landscape. Overburden dumps where excess pit spoil is piled can form new hills within the mine terrain (Fig. 36.2). These overburden dumps frequently act as sources of leachate and sediment, which enter surface waters associated with the mine. Capping and remediation of overburden dumps is an emerging research challenge.

Sedimentation of waterways can be a major problem associated with excavation and roading at mine sites, as suspended sediment can smother algae, benthic invertebrates and the substrate, reducing substrate heterogeneity (Fig. 36.6).

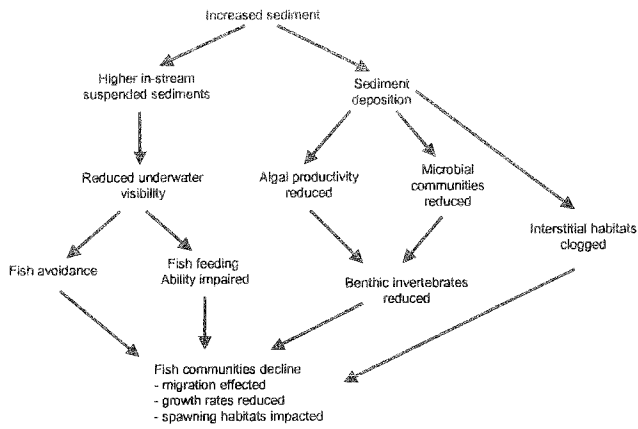


Figure 36.6 Model of the pathways by which suspended sediment might affect components of the stream ecosystem (after Rowe and Dean 1998).

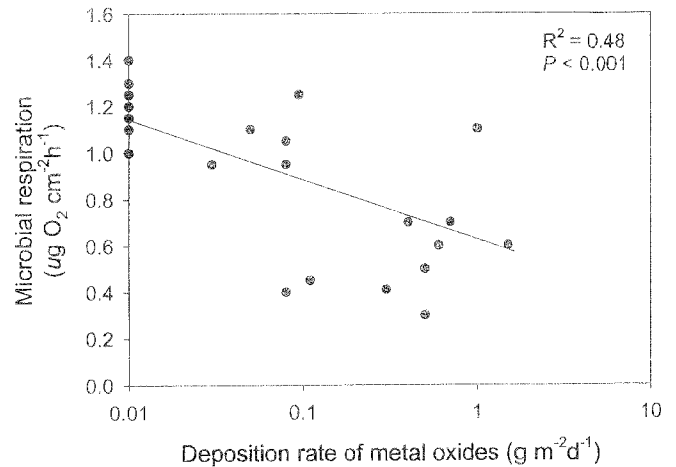


Figure 36.7 Microbial respiration and deposition rate of metal oxides (after Niyogi *et al.* 2001).

BIOTIC RESPONSES

Bacteria and fungi

Alterations in water chemistry, particularly increased acidity and the release of metals, can have a profound effect on microbial communities. Experiments on decomposition rates of leaves in coal mining streams have generally shown greatly reduced microbial activity, although Harbrow (2001) found highly variable breakdown rates in six West Coast streams. Winterbourn *et al.* (1985) observed a lack of fungi on stones in a stream with a natural pH 4.3, while Hildrew *et al.* (1984) noted impaired cellulolytic decomposition in streams with pH < 5.6. The mechanisms that lead to a reduction in microbes at low pH are unclear, but there are at least two. Firstly, high acidity inhibits microbial enzymes. For example, the activity of pectin-degrading enzymes such as pectin lyase, which is involved in softening plant tissue in the early stages of leaf breakdown, is lower at low pHs (Suberkropp and Klug 1981). Secondly, reduced litter breakdown rates have been linked to metal oxide deposition, and Niyogi *et al.* (2001) found that microbial respiration decreased with increasing rates of deposition of metal oxides on leaf litter (Fig. 36.7). Thus, the precipitates themselves may either smother microbial complexes or be directly toxic to them.

Consequently, in lakes and rivers affected by mining and not subject to flushing flows from floods, coarse particulate organic matter may accumulate on the bed because natural decomposition is suppressed. Reduction in microbial processing of allochthonous inputs may also significantly reduce the availability of dissolved and fine organic matter within the system. These interrelationships, when coupled with a reduction in the numbers of invertebrate shredders due to toxic conditions, can combine to reduce overall allochthonous breakdown in mine drainage streams (Fig. 36.8).

The importance of metal oxide deposition has been confirmed in post-remediation studies in which litter breakdown rates failed to recover even after water quality improved. In some instances the presence of metal oxides may have continued to limit microbial activity. Poor or depauperate food resources frequently have been cited as additional confounding factors in the recovery of benthic invertebrate communities in mine drainage streams after remediation, however this has not been well studied in New Zealand.

The presence of high concentrations of dissolved metals has little effect on some species, particularly iron bacteria that occur in some systems affected by mining. Winterbourn *et al.* (1985) found the iron bacterium *Leptothrix* sp. dominated a stream with total iron concentrations of 6 g m^{-3} , and was eaten by a chironomid that lived on it. Little is known about the ability of other organisms to use these food resources.

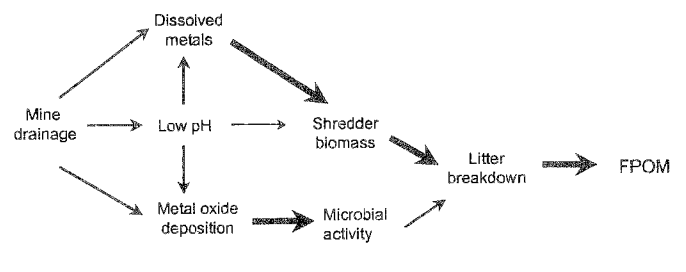


Figure 36.8 Model of the effects of mine drainage on leaf litter breakdown and the production of fine particulate organic matter (FPOM). The thickness of the arrow indicates the strength of the effect (modified from Niyogi *et al.* 2001).

Aquatic plants

Vascular plants are usually absent from acidic streams, but bryophytes and periphyton can be locally very abundant. Some algae are acidophilic and can occur in high densities in acidic sites with stable flows. Where acidic algae proliferate, communities are frequently dominated by a few species, with high biomass. Acid-tolerant algae, such as *Ulothrix* sp., have been shown to account for 99% of the algal biomass at mine discharges, with other filamentous algae such as *Microspora*, and *Tribonema* common (Niyogi *et al.* 1999; Winterbourn *et al.* 2000) (Fig. 36.9).



Figure 36.9 Algae and iron hydroxide precipitates at Sullivans West adit, Denniston Plateau, North Westland

Photo: Jon Harding

Like microbial communities, algae may proliferate at sites with low pH, stable flows and low metal oxide deposition (e.g., mine adits), but as soon as metal deposition rates increase (e.g., when pH rises above 3.5–4.3) algae can survive only if they are able to grow faster than the rate at which oxides smother their surfaces. Experimental manipulations of pH have shown that algal biomass can increase until Al precipitation occurs (at about pH 4.9) and periphyton growth is inhibited.

Where the substratum is relatively stable, bryophytes (mosses and liverworts) occur in naturally and mining-affected low-pH streams. The liverworts *Lophocolea*, *Jungermannia* and *Riccardia*, and mosses such as *Blinda* and *Sphagnum*, have been recorded in low pH streams in New Zealand (Winterbourn *et al.* 2000). Some bryophytes accumulate metals and, in particular, Fe. Winterbourn *et al.* (2000) reported concentrations of Fe 10 times higher than Al in bryophyte and algal tissue in West Coast streams. Concentrations of metals can reach high levels; Englemann and McDiffett (1996) reported concentrations of 17.3 and 9.1 $\mu\text{g/g}$ dry mass of Fe in two species of bryophytes. Fe^{3+} is practically insoluble, and moderately toxic to plants. It competes with other substances for binding sites on cell membranes and is then taken into the cell where it accumulates (Englemann and McDiffett

1996). It has been suggested, however, that Fe can actually decrease the toxic effect of other metals by competing for binding sites, and by the partial sequestration of other metals by Fe colloids. Al has also been shown to accumulate in bryophytes, however Al absorption in moss usually peaks in tissue at intermediate pHs, i.e., 5.2–5.8 (Englemann and McDiffett 1996; Winterbourn *et al.* 2000). The reason for this is probably that the solubility of Al increases markedly below pH 5, where the more toxic Al^{3+} is more prevalent. Hence the higher concentration of Al in plants at intermediate pH may be a result of bioaccumulation during periods of lower pH, and poor solubility as pH rises. The low accumulation of Al at low pH may be due to the domination of Al^{3+} , which has been shown to cause mucilage production and root necrosis in duckweed (Crowder 1991). Where they are able to survive, higher-order aquatic plants can also accumulate metals. Sabti *et al.* (2000) found that aquatic macrophytes (*Egeria densa*) in the Ohinemuri and Waitekauri Rivers contained gold (302 and 672 $\mu\text{g kg}^{-1}$ dried matter respectively), and suggested that plants may be useful as bioindicators of heavy metal contamination.

Benthic invertebrate communities

The impacts of mine drainage on stream invertebrates are almost entirely negative and have been termed “acidaemia” by Kelly (1988). The effects range from acute and direct toxicity caused by combinations of low pH and/or the presence of toxic metals, to indirect effects in areas where food resources are limited (by reducing organic matter processing and algal growth), to altered in-stream habitat (by armouring and clogging of the riverbed substrata).

In naturally acidic brown-water streams (pH 4.3–5.7) benthic invertebrate communities often have fewer species, lower densities and altered community composition compared to communities in similar-sized natural streams (Collier and Winterbourn 1987). Collier and Winterbourn (1987) suggested that the depauperate state of naturally acidic brown-water streams was probably a result of changes in the food supply, particularly a reduction in periphyton biomass (dominated by diatoms such as *Eunotia* and *Fragilaria* spp.). By comparison, streams affected by acid mine drainage may be almost devoid of species (Fig. 36.10), and where organisms are present their densities may be as low as a few animals per square metre.

Naturally acidic waters are not uncommon on the West Coast of New Zealand, and it is apparent that some benthic invertebrate species are well adapted to the conditions found in low pH waters (Winterbourn and McDiffett 1996). Naturally acid brown-water streams in South Westland typically have communities dominated by the common leptophlebiid mayfly *Deleatidium*, Chironomidae, the elmid beetle *Hydora*, and the stonefly

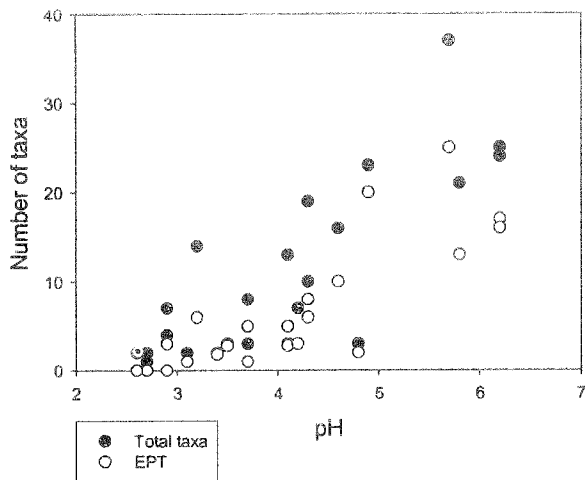


Figure 36.10 Relationship between total taxa and pH in 23 streams in South Island (after Winterbourn *et al.* 2000).

Zelandobius. However, in streams contaminated by acid mine drainage in North Westland (pH 2.6–4.2), a range of taxa were recorded in low densities, including the mayflies *Deleatidium* and *Austroclima sepia*, the stonefly *Spaniocercoides philpotti*, the caddisflies *Psilochorema* and *Oxyethira albiceps*, scirtid beetles and dipterans, including *Chironomus zealandicus* and *Eukiefferiella* (Boothroyd 2002; Harding 2002). Sites with pH <3 in the Denniston and Stockton Plateaus were dominated by chironomids and scirtids, but included two caddisflies—*Pseudoeconesus* and *Kokiria miharo* (Winterbourn 1998). Although Crustacea are frequently considered intolerant of acidity, Anthony (1999) found the amphipod *Paraleptamphopus* below pH 4, and the crayfish *Paranephrops* occur in naturally low pH waters (Collier *et al.* 1990).

Exposure to low pH and elevated metal concentrations may cause a number of physiological stresses to benthic invertebrates. Numerous studies have shown osmoregulation in many species is impaired by the disruption of acid base and ion balances, nitrogen excretion and respiration (Sutcliffe and Hildrew 1989). Low pH also causes a reduction in sodium uptake, detrimental to sodium-sensitive species such as crayfish (Haines 1981). Similarly, Cl, Ca and K can all be lowered in high-acidity waters (Rowe *et al.* 1989), while Bell (1971) showed that low pH could be lethal at critical junctures of the life cycles of stream insects. Thus, trichoptera adults had only 50% successful emergence at pH 4, while mayflies showed 50% adult emergence failure at pH 5.9. However, a number of New Zealand mayflies and caddis are clearly capable of successful emergence at low pH, although this may differ between natural and mine waters. The dominance of benthic communities by Chironomidae at low pH may relate to the ability of these taxa to tolerate ion imbalance (Forsyth 1983; Boothroyd 2002).

Low pH may also affect some species by making them more susceptible to disease, infection and parasitism. Leuven *et al.* (1986) found that amphibian eggs were unable to develop at pH <3.5 due to infection from pathogenic fungi, while the crayfish *Orconectes virilis* suffered egg mortality from protozoan parasites when the pH dropped below 5.6 (Schindler and Turner 1982). A high frequency of infections (visible sores) has been noted in benthic invertebrates from streams receiving discharge from abandoned gold mines near Reefton. This suggests that toxins in these mine waters may weaken benthic organisms, making them susceptible to disease and infection.

Toxicity of metals can be a major problem in many mining-affected waters. Toxicological studies have shown insects, crustacea and fish are frequently susceptible to toxic metals. Hickey and Clements (1998) noted that net-spinning hydropsychid caddis and orthoclad chironomids dominated at sites high in metals in the Coromandel Peninsula. Analysis of taxonomic richness data from North Westland shows that relatively few species are found in waters with high concentrations of dissolved Al and Fe (Fig. 36.11).

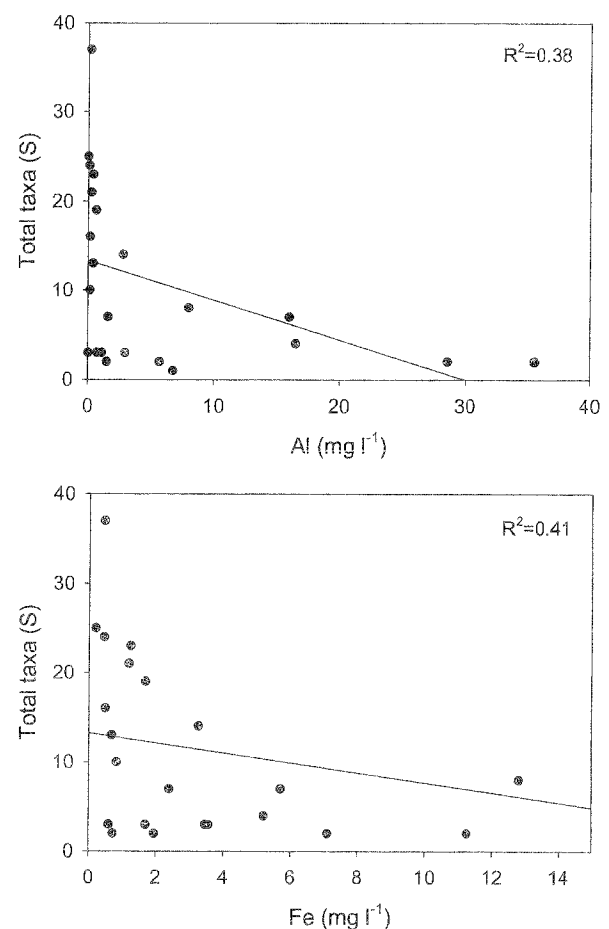


Figure 36.11 Relationship of total taxa to mean total reactive aluminium (top) and total dissolved iron (bottom) in 19 streams in North Westland (after Harbrow 2001).

Fish communities

Most fish species are negatively affected by acidification and mine leachate. These toxic effects may be acute—causing death, or chronic—resulting in impaired health (e.g., mucous secretion on gills impairing gas exchange), or physiological—from stress that reduces fish condition. Accumulation of heavy metals within the flesh of fish may have long-term toxic effects (Table 36.2). Furthermore, the presence of mine discharges may create a chemical barrier to diadromous species, reducing or preventing their migration to and from the sea. Several diadromous native fish have been recorded in low pH waters (Main 1988) and have been shown to be able to detect pH gradients and display pH preferences. In particular, short-finned eels (*Anguilla australis*) and two whitebait species, koaro (*Galaxias brevipinnis*) and banded kokopu (*Galaxias fasciatus*), prefer waters <6.5, whereas other species avoided low pHs in laboratory trials (West *et al.* 1997). These pH preferences were stronger in adults than juveniles, and probably account for the presence of galaxiids in naturally brown-water streams on the West Coast. The freshwater shrimp *Paratya curvirostris* seemed unable to detect pH changes, and had high mortality in high pH waters. These findings are consistent with Collier *et al.* (1990), who reported widespread tolerance of low pH in naturally acidic streams in Westland, with 9 out of 14 native fish species reported in pH <5, and 7 species at pH <4.5. Particularly low pH-tolerant species were inanga (*Galaxias maculatus*), giant kokopu (*G. argentus*), long- and short-finned eels and banded kokopu. The fish fauna in mine-contaminated streams have not been well studied. Historical anecdotal evidence from local residents in the Ngakawau River, north Westland, indicate that prior to mining crayfish and koaro were abundant in this system, however neither species have been recorded post-mining.

As mentioned previously, the presence of high concentrations of suspended sediments derived from mining (e.g., from placer mining, roading) can affect fish communities. Behavioural experiments by Boubée *et al.* (1997) on native juvenile migratory species have shown that banded kokopu can be sensitive to turbidities >17 NTUs, whereas koaro and inanga are less sensitive and avoid only much more turbid silt-laden waters (>70 and 240 NTUs, respectively). In contrast, short-finned and long-finned eels, and red-finned bullies were not effected by turbidities > 1000

Table 36.2 Mean copper concentrations in muscle tissue of rainbow trout from the Waitekauri River, Golden Cross (All data stated as mg/kg wet weight).

Year	Upstream	Immediately downstream	5 km downstream	Control site
1993	-	1.39 ± 0.09	1.22 ± 0.29	0.62 ± 0.08
1994	-	1.15 ± 0.14	0.76 ± 0.07	0.64 ± 0.04
1996	0.31 ± 0.06	0.32 ± 0.08	0.41 ± 0.15	0.29 ± 0.04

NTUs (Boubée *et al.* 1997). A possible explanation for these preferences was offered by Rowe and Dean (1998), who demonstrated that feeding rates of banded kokopu and inanga declined significantly as turbidity increases. Banded kokopu feeding rates were 40% lower at 20 NTUs than in clear water, whereas koaro feeding rates were not affected, even at the highest turbidity.

Ecosystem interactions

As we have seen, mining activities can have a profound effect on water chemistry, physical conditions and the biota of freshwater systems. The loss or reduction of bacteria and shredding insects means that the processing of organic matter is significantly reduced in mining-affected streams. Similarly, impacts on algal communities, which form the food base of many New Zealand stream food webs (especially outside forests), will have a cascade effect on invertebrate grazers, predators and fish communities. Figure 36.12 summarises the major chemical, physical and biological responses from the effects of mining.

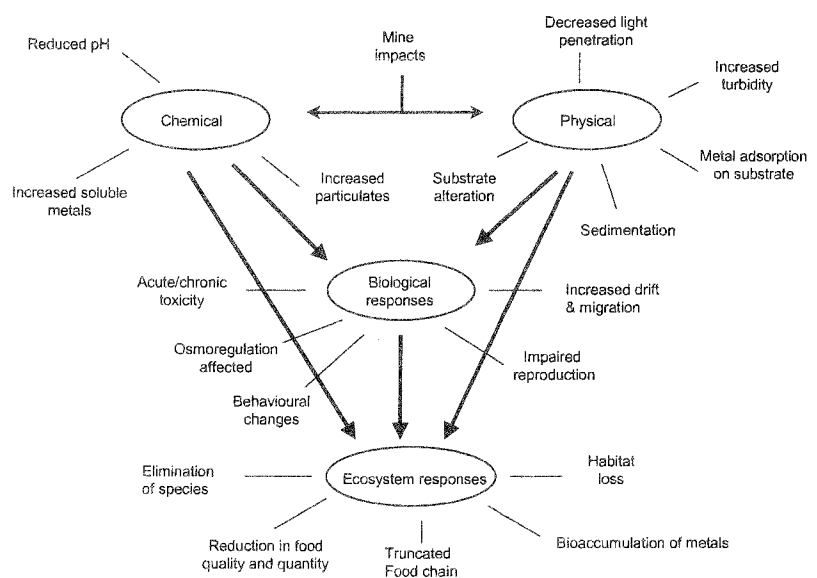


Figure 36.12 Model of the primary ecosystem factors influenced by mine drainage inputs (modified from Gray 1997)

Management and restoration strategies

The chemical and geological complexity of the effects of mine drainage presents significant challenges to the effective management of mining landscapes. The nature of the industry (i.e., large-scale extraction of minerals from the ground) means that environmental impacts are a byproduct of the process, and moderating these impacts has become the focus of research. Management strategies centre on reducing toxic inputs, remediation of effects, restoration and monitoring. While a number of remediation techniques have been developed internationally, the success of many techniques—artificial wetlands, limestone dosing, dilution and buffering, and bacterial reduction of leachate—have been highly variable, and have not been adequately tested in New Zealand.

SUMMARY

Considerable knowledge exists about the chemical processes involved in mine drainage, and its effects on water chemistry, algae and benthic communities have been well documented. The complexity of these relationships continues to present significant challenges to researchers, and those charged with the task of remediating mine impacts.

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Chapter 35

Impacts of urbanisation on streams

Alastair Suren and Sandy Elliott

INTRODUCTION

Humans are highly social, and the city is one of the greatest expressions of their behaviour. History is replete with tales of extraordinary cities that captured the imagination and spirits of those who saw them. Rivers represented the life-blood of early cities, providing water for domestic use, transportation of goods, and a convenient way for city dwellers to dispose of their sewage and industrial wastes (Walsh 2000). They still fulfil all or many of these important roles today.

Cities are unique ecosystems, as they are unable to function solely on local resources. Instead they rely on appropriation of materials from both terrestrial and aquatic ecosystems that are geographically distant (McDonnell and Pickett 1990). Not surprisingly, the impact of cities on ecosystems is enormous—indeed modern European cities require the ecological services of between 500 to 1000 times the land area they occupy (Folke *et al.* 1997). The equivalent figure for New Zealand cities has not been calculated, but is likely to be of a similar order of magnitude. Cities thus have the potential to consume vast amounts of natural resources, and produce equally vast amounts of waste material. Such use and disposal of resources has significant implications for ecosystems, especially considering the increasing proportion of the world's population that is becoming urbanised (Paul and Meyer 2001). The ecological impacts of cities extend both to the surrounding land, and to rivers and streams. Worldwide, there is a well-documented decline in the habitat and water quality of urban streams, as well as declines in their ecological health, and aesthetic and recreational values (Hall *et al.* 2001; Paul and Meyer 2001; Beach 2002).

By world standards, New Zealand's cities are small in area, relatively young, and have small populations—it may thus be tempting to assume that this would somehow

buffer the adverse effects of urbanisation on streams. Unfortunately, New Zealand's urban streams have been greatly affected as cities have grown, and they often have poor water or sediment quality (Williamson 1993) and degraded biological communities (Hall *et al.* 2001). There is also growing concern that the encroachment of cities into rural hinterlands will degrade rural streams as their catchments are urbanised. For example, in Christchurch many of the spring-fed streams on the northern edge of the city currently support relatively diverse invertebrate assemblages of caddisflies (A. Suren pers. obs.). Urbanisation of these catchments may result in the loss of these animals.

Until relatively recently, urban streams were valued principally for drainage, and their management focused on maintaining high hydraulic efficiency. However, urban streams have now become important to the public for recreation (e.g., walking, running and cycling tracks following waterways), aesthetics, and as places of cultural and historic value (Greenaway and Watts 1999). If they are properly managed, urban streams can also provide places where diverse native plants and animals may be found within the matrix of urban development. However, to minimise the adverse effects of urbanisation on streams, we need to identify the processes linking urbanisation to the degradation of the physical and chemical properties, and ecological integrity of streams. We must identify what can be done about these processes to minimise the effects of future urban expansion. Such knowledge is particularly relevant given the ever-growing need of urban dwellers to use green-spaces within cities for recreation and relaxation. The Resource Management Act (1991) also embodies the concept of sustainable development, and focuses on ways to avoid, mitigate or remedy adverse impacts of urbanisation on streams.

Urban development is a good example of a pressure-

state-response model—the pressure of urbanisation results in distinctive and highly predictable changes to the state of urban waterways, both in their physico-chemical conditions and to their biological communities (Fig. 35.1). Pressure on waterways is ultimately dictated by social factors, such as increased population and economic growth, as well as changes in land-use policy both within urban environments (e.g., in-fill housing) and the rural hinterlands that surround cities (e.g., “green-field” developments). For example, within Christchurch, 2350 new building consents were issued in the 2002–2003 year for new residential and commercial developments, both with the existing urban boundaries and in new green-field developments in areas that were historically market gardens or orchards. An increasing population is, however, only one of the many pressures affecting urban streams. Socio-economic factors can also have a profound influence on stream degradation in urban areas. For example, Herald (2003) found that streams draining lower socio-economic areas of catchments within the Waitakere Ranges had high zinc concentrations (110 µg/L) at base flow. Zinc is derived from vehicle traffic (brake pads and tyre wear), and from the corrosion of galvanised roofs, pipes and other metal fittings. Vehicle traffic within this particular catchment was relatively light, so Herald (2003) attributed its source to the poorly maintained galvanised iron roofs, garages and fences that are common in this catchment.

This chapter first examines the factors that are responsible for the degradation of urban streams, and then examines the resulting biological effects. Potential

mitigation measures are also discussed, as many of the causative factors can be avoided, mitigated or remedied by proper management practices. This chapter deals with the problematic issue of stormwater and its effects on streams, as sewage management in most large urban areas in New Zealand is well advanced and its effects on the receiving waters are relatively well understood.

CAUSATIVE FACTORS

Changes to the land surface, stormwater drainage, and stream channels

The most obvious features of the land surface in an urban area are the impervious surfaces, such as roads, roofs, driveways and parking areas. In a typical urban residential area, impervious surfaces cover about 40% of the land area—about half of that is roads and driveways. In commercial or light industrial areas impervious surfaces may cover 80% or more of the area. Increasing the impervious cover within a catchment changes a stream’s hydrological regime, as most of the rain from a storm is converted immediately to surface runoff, instead of soaking into the ground. This rapid and efficient delivery of water from the catchment to the stream also affects both its physical habitat and water quality. This change to the hydrological regime is so fundamental that the fraction of impervious area in a catchment is commonly used to indicate the intensity of urbanisation and its likely effects on streams.

Urbanisation not only increases impervious surfaces, it

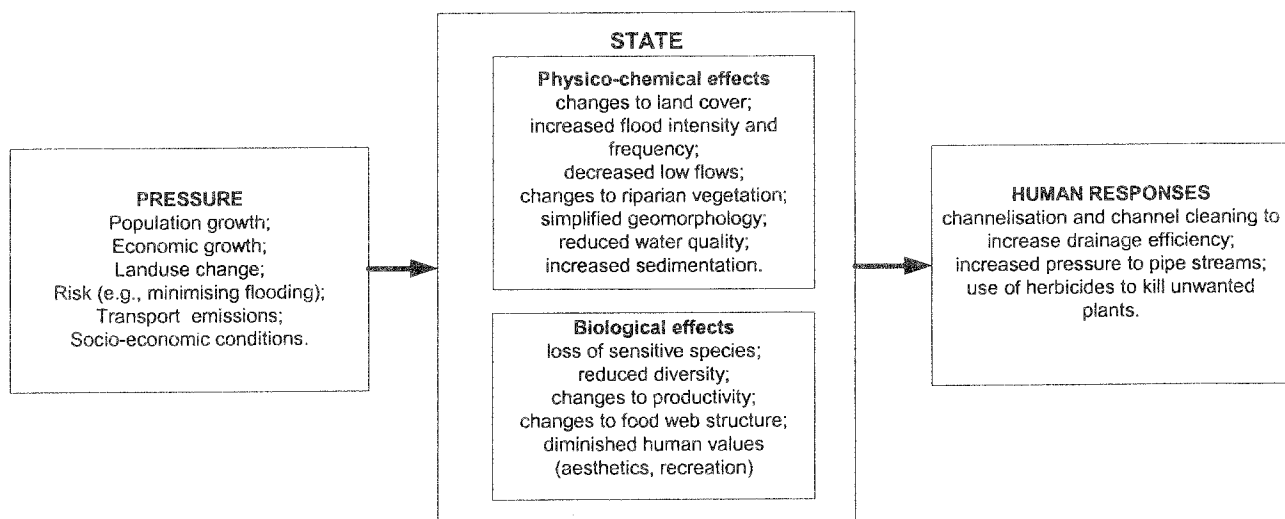


Figure 35.1 Pressure-state-response model outlining relationships between the pressures behind urban expansion, their physical and chemical effects on the state of urban waterways, and the human responses to these states.

also alters the nature of the pervious surfaces. Typically, urbanisation involves removing existing vegetation and topsoil, re-contouring the land, compacting the subsoil with heavy machinery, replacing the topsoil, and establishing lawns and gardens. Such activities can greatly reduce the water storage and holding capacity of topsoil, increasing runoff from urbanised areas (Schueler *et al.* 1999).

Extensive networks of stormwater gutters and pipes are usually constructed to quickly and efficiently transport stormwater into streams or to the coast. These stormwater networks become larger and more extensive as urban areas incorporate more impervious land (Fig. 35.2). Developers often pipe headwater streams (particularly ephemeral ones) to increase the useable land area, and to ensure the efficient transport of stormwater. Such piped drainage systems are usually designed for floodwaters from a 10-year return period storm. Unlike the headwater tributaries, the lower parts of a stream network may be left in a more natural form to carry large floods. However, the channels are often modified to convey flows more efficiently and to prevent

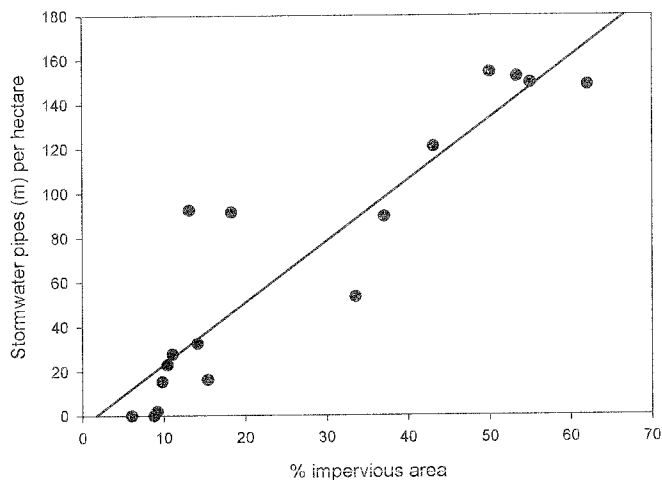


Figure 35.2 Length of public stormwater drainage per hectare as a function of percent impervious surface area in four subcatchments in the Waitakeres (from Herald 2003).

storm flows from overtopping the banks. Modifications such as channel straightening, removal of obstructions and riparian vegetation, re-contouring of banks, and even reinforcing an entire channel with concrete (Fig. 35.3) are common in urban streams (e.g., Riley 1998; Table 35.1). However, such modifications are becoming less common in new urban developments, where protected riparian zones often are being established along streams.

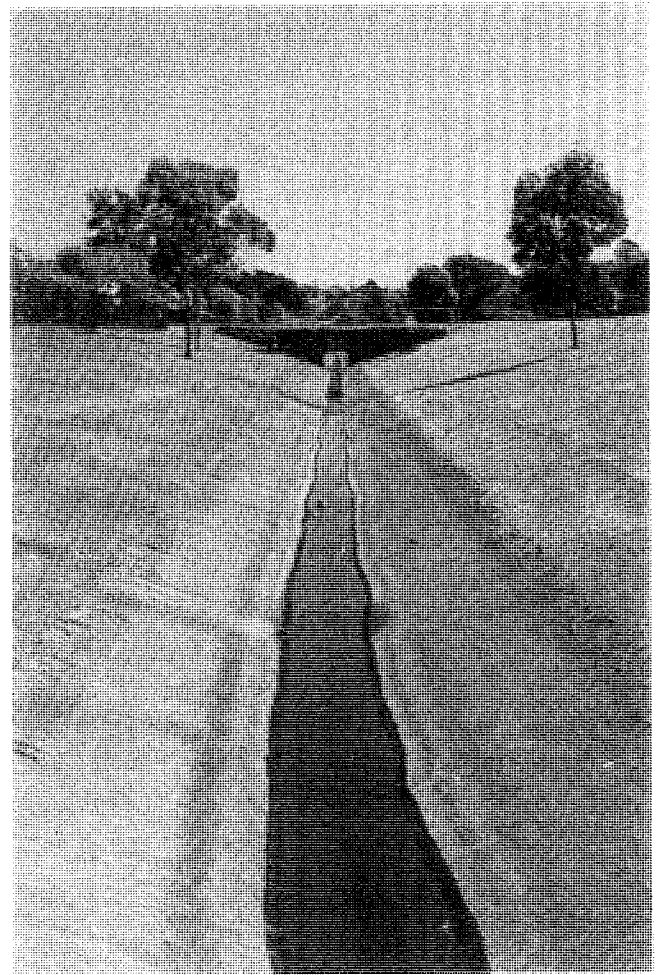


Figure 35.3 Oakley Creek, Auckland has been re-contoured, with a concrete-lined low-flow channel. Minimal habitat is available for stream biota.

Photo: Alastair Suren

Table 35.1 Imperviousness and flow data for three catchments in the Waitakere Ranges. Flow characteristics from July 2002 – February 2003 and the percentage of the stream channel that is modified are also shown (data from Herald 2003).

Catchment	Area (ha)	% impervious	% stream channel modified	Flood flow (m^3s^{-1})	Base flow (ls^{-1})	% stream channel modified
Cantwell	76	6	0	1.2	3	0
Waikumete	54	16	15	2.1	1.1	15
Tangutu	84	34	50	6.7	Ceased for 15 days	50

In several regions in New Zealand, urbanisation has involved the draining of swampy areas and wetlands in the headwaters of streams by installing ground drains to lower the local water table, and by excavating and backfilling wetland soils. Removal of these hydrological “sponges” can drastically alter the natural responses of catchments to rainfall, and lower the overall diversity of waterways found within urbanised catchments.

Changes to the hydrologic regime

Impervious surfaces produce a greater volume of storm runoff than pasture or bush-clad land (McConchie 1992; Booth and Jackson 1997). Storm runoff also reaches streams more quickly through an efficient network of gutters and pipes. The greater volume of runoff and more efficient conveyance of flows results in much greater flood flows in urban streams (Table 35.1). Typically, the mean annual maximum flood increases by a factor of three in areas of residential development, and flood flows also become more frequent (Fig. 35.4). Flow variability also increases.

Urbanisation not only increases high flows, it can also reduce baseflows (Paul and Meyer 2001), because of reduced groundwater recharge beneath impervious surfaces such as roofs and roads (Fig. 35.5; Table 35.1). However, such changes are not always observed (Herald

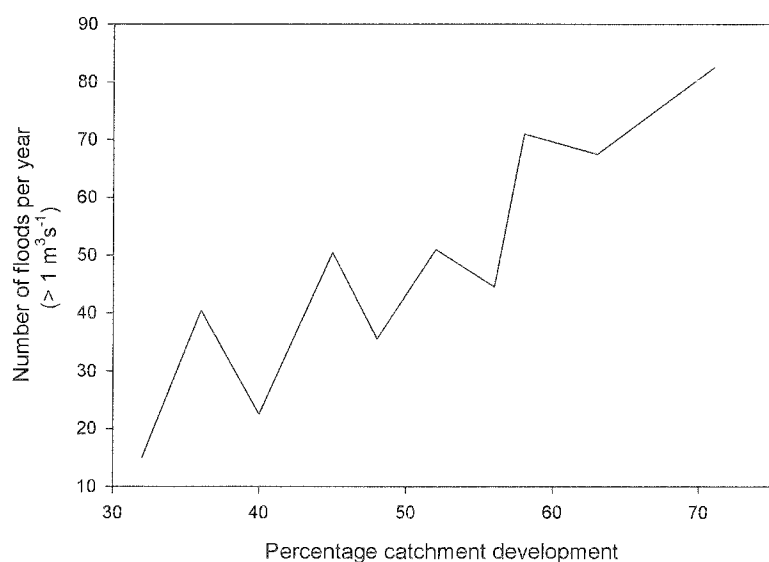


Figure 35.4 Increase in the frequency of small floods associated with increased development in the Wairau Creek catchment (North Shore, Auckland) between 1962 and 1975 (from Williams 1976).

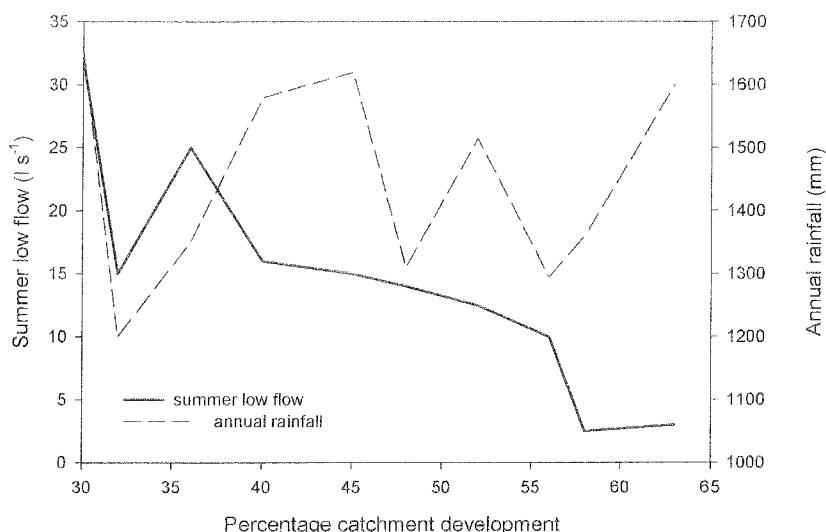


Figure 35.5 Instantaneous annual minimum flow (solid line) and its relationship with catchment development in the Wairau Creek (Auckland) catchment, 1962 to 1971. The annual rainfall (Whenuapai) is also shown (dashed line) to show that the reduced flow was not due to reduced rainfall (from Williams 1976).

1989), perhaps reflecting changes in the baseflow from pervious portions of a catchment. Reduced baseflow results in shallower, slower flows in the streams, and headwater springs or seeps often move down the drainage network as the water table is lowered (Suren 2000).

Stream flow, and flow-related variables such as velocity, shear stress and turbulence affect stream ecosystems (Death and Winterbourn 1995; Clausen and Biggs 1997), and influence many or all of the variables that control instream communities. Indeed, it is almost axiomatic that flow is fundamental to structuring instream benthic communities (Resh *et al.* 1988; Poff and Ward 1989; Jowett 2000). Both floods and low flows can influence invertebrate communities, and both are altered by urbanisation.

Geomorphological changes

Urban streams represent some of the most physically modified running water ecosystems in New Zealand. Streams that once had meanders, complex channel morphology, a heterogeneous substrate and dense riparian vegetation have often been reduced to straightened concrete or wood-lined channels (Suren 2000; Paul and Meyer 2001). Streambeds have also been modified extensively, with sedimentation lowering the habitat value of low-gradient urban streams, and reinforced concrete channels replacing natural boulders, cobbles and gravels in higher gradient urban

streams. Such modifications greatly diminish the usable habitat of urban streams, and their capacity to support diverse biological communities (Wilding 1999; Allibone *et al.* 2001).

To accommodate increased storm flows, urban stream channels increase in size following catchment development (Fig. 35.6). Typically, the bank-full cross-section increases by 2–3 times, and the width increases by 50–100%, but there is considerable variation in the way in which streams adjust, depending on the local setting. For example, some streams may scour down to bedrock and therefore increase their area mainly by widening. Alternatively, deep scour pools may form in streams where there are longitudinal variations in bank or bed strength, although such channel adjustments may take decades to occur (Hammer 1972). There may also be a tendency for stream morphology to change from a series of pool/riffle sequences to more uniform runs, further reducing habitat diversity and quality.

Lastly, because bank erosion can eat into stream-side properties or threaten buildings, banks are often strengthened by installing rock or timber walls, or by lining the banks with concrete, to the detriment of habitat values (Riley 1998).

Loss of riparian vegetation

Riparian vegetation plays a number of important roles in streams, including providing shade, energy inputs in the form of leaf litter, instream habitat and sites for oviposition. However, the riparian conditions of urban streams are often altered dramatically and little vegetation may remain. For example, a survey of 30 randomly selected stream sites within the Auckland city area showed that

nearly half of them had no canopy cover, and only five sites had > 75% canopy cover (A. Suren, pers. obs). The loss of overhead canopies is further exacerbated by management activities that often focus on the removal of other bankside vegetation (Fig. 35.3) in an effort to maximise hydraulic efficiency. As a consequence, the services provided to urban streams by riparian vegetation are greatly reduced.

Reducing the complexity and cover of riparian vegetation greatly reduces stream shade, and lowers energy subsidies in the form of leaf litter to a stream (Fig. 35.7). Loss of riparian vegetation also reduces habitat quality by eliminating overhanging vegetation, and branches and logs that otherwise may fall into the water and provide cover for fish and invertebrates. Water temperature is affected by shading (Rutherford *et al.* 1997) and is typically higher in open than in shaded streams. For example, Webster (2000) recorded stream temperatures of 29.2°C at a site without any shading in Oakley Creek, Auckland, and noted that the stream temperature was regulated more by direct solar radiation and cloud cover than by air temperature *per se*. High water temperatures may reach lethal levels for some fish and invertebrates (e.g. Quinn *et al.* 1994). Moreover, many urban surfaces (roads, roofs and carparks) hold and store heat from solar radiation efficiently, and this heat can be quickly transferred to urban streams during brief summer thunderstorms, further exacerbating the lack of shade by riparian plants.

Lack of shade, combined with high water temperature, can cause extensive algal blooms, reducing dissolved oxygen concentrations and bringing about large diurnal changes in pH, further stressing aquatic life (Fig. 35.7). Stream water pH as high as 9.4 was recorded during the day in Oakley Creek during summer, most likely as a result of algal photosynthesis (Webster 2000). Changes in the riparian zone also have repercussions for aquatic insects whose terrestrial adults use riparian vegetation (Collier and Scarsbrook 2000), and on galaxiid fishes such as *Inanga*, which spawn among overhanging *Carex* sedges at high tide (McDowall 1990; Eldon and Taylor 1990).

Inappropriate riparian planting is also a problem in many urban areas. For example, the early settlers of Christchurch planted many exotic trees along streams, including weeping willow (some of which grew from cuttings from Napoleon's tomb), Lombardy poplar, sycamore, Douglas fir and laburnum (Lamb 1981). In this way the original riparian wetland vegetation was "beautified" by a mix of exotics, and

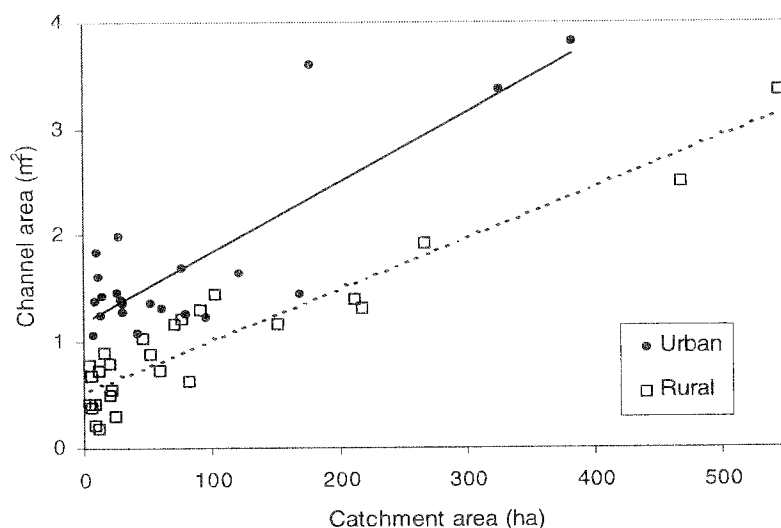


Figure 35.6 Comparison of channel bank-full cross-sectional area for urban (closed circles) and rural streams (open squares) on the North Shore, Auckland. The lines are linear fits to the data. (from Herald 1989)

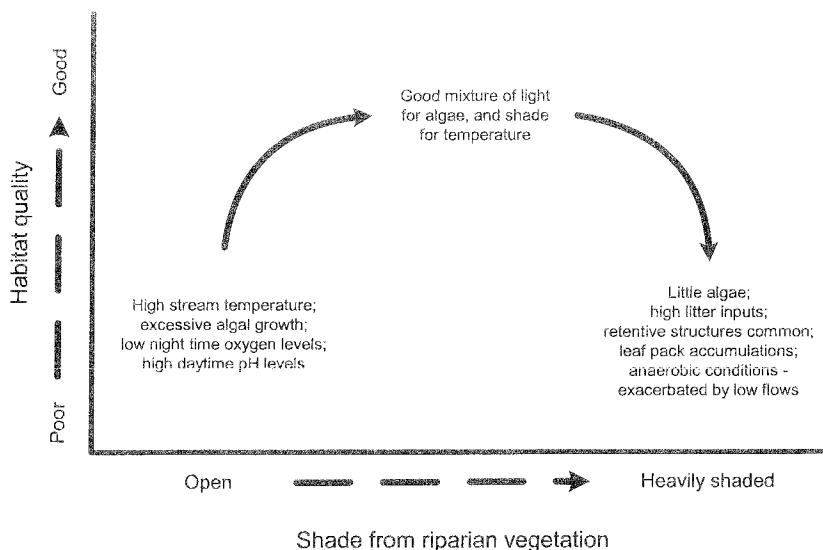


Figure 35.7 Hypothetical relationship between habitat quality and shade from riparian vegetation.

mown riverbank grasses, which served to remind early settlers of their English homes. As a consequence, the waterways in Christchurch bear little resemblance to the original meandering streams with their native riparian zones of flax (*Phormium*) and *Carex* sedges (Lamb 1981; Morland 1996). Although many exotic deciduous trees shade streams in summer and ameliorate water temperatures, large quantities of leaf litter enter the streams in autumn and can lead to anaerobic conditions where leaf packs block a stream (Fig. 35.7). Too much shade also suppresses algal biomass, which is a primary food of many stream invertebrates.

Water quality changes

Water quality generally decreases as the percentage of impervious area increases within a catchment (e.g., Williamson 1993; Lenat and Crawford 1994). Much of this reflects the increased efficiency with which rainfall leaves the catchment as runoff, efficiently transporting pollutants that have accumulated on impervious surfaces. These pollutants come from many sources, including transport infrastructure (roads, carparks, and railways), erosion and runoff from developing subdivisions, industrial spillages, roof run off, discarded rubbish, accidental or deliberate discharges into stormwater systems, and animal faeces in streets or parks. There are five major types of pollutants in urban stormwater—suspended sediments, toxicants, nutrients, bacteriological contamination, and general rubbish (Williamson 1993).

Suspended sediments and toxicants are of particular ecological concern in waterways, and can cause profound changes in instream biota. High suspended-sediment levels are common in urban catchments and result from bank

erosion as stream channels adjust to the increased runoff from the catchment. Building and road construction within catchments also results in high sediment inputs to streams when expansive areas of bare earth are exposed to rainfall and subsequent erosion (Hogg and Norris 1991), or when groundwater is pumped from foundations as they are being dug. High suspended-sediment loads are often transient, lasting only during rainfalls or during construction. The impacts of high suspended sediment on streams are often substantial, however, even though the input is transient. High suspended-sediment levels can affect fish feeding and behaviour (Rowe and Dean 1998), and reduce instream algal productivity by increasing light scatter within the water

column (Davies-Colley *et al.* 1992). Turbid streams also have low aesthetic appeal, and can be considered inherently unhealthy, especially to Tangata Whenua (Tipa and Teirney 2003). Suspended sediments also settle on the streambed, smothering gravel and cobble substrates and filling interstitial spaces between particles (Brunke 1999), thus reducing the amount of habitat available to fish and invertebrates (Suren and Jowett 2001). Sediment can also settle in the lower reaches of a stream network, raising the bed and reducing the flood capacity of the stream. Delivery of sediment to estuaries also increases, increasing the rate of estuarine infilling and smothering estuarine organisms. Stream widening, which typically continues for some time after urbanisation of a catchment (up to decades), provides a continuing source of sediment to stream and estuaries. Once catchment development stops and stream widths have changed, sediment loads return to pre-development levels or lower. Sources of sediment such as slips are likely to be reduced as a result of bank stabilisation and reduced groundwater recharge.

Toxicants in stormwater include pesticides, heavy metals and various types of hydrocarbons. Pesticides include herbicides, insecticides and fungicides, all of which are commonly used by residential and commercial land-owners. Reviews in the United States and France have shown that pesticide use and contamination of urban stream sediments frequently exceed those in agricultural streams (Paul and Meyer 2001), and there may be a similar situation in New Zealand, especially given the ready access to pesticides and their use by home owners.

The commonest heavy metals in New Zealand streams are lead, zinc and copper. However, environmental lead concentrations are expected to decline in New Zealand, reflecting the phasing out of lead-based paints and petrol

products. Zinc, one of the more problematic heavy metals in streams (Herald 2003) is derived from a combination of road runoff (from brake linings and tyre wear) and deteriorating zinc-coated, or galvanised, roofs and fences. Hydrocarbon contamination of waterways most likely comes from direct runoff from service stations, parking lots and garages of petrol and oil that drip from sumps, fuel lines and the like.

Surveys of streams that flow through New Zealand urban centers have shown that concentrations of dissolved metal (i.e., metal present in the <0.6 µm phase) in urban streams during baseflow are often below those at which significant biological effects occur (Mike Timperley, NIWA Auckland, unpublished data; Table 35.2). However, dissolved metal concentrations are likely to reach higher levels during short-lived floods, and acute pulses may have adverse effects on some aquatic species. Unlike dissolved metals, bed sediment concentrations of metals often exceed ANZECC interim sediment quality guideline (ISQG) values (Table 35.2). Accordingly, bed sediments may have detrimental impacts on aquatic species. Moreover, recent surveys of metal contamination of biofilms coating aquatic plants in Auckland, and cobbles in Christchurch streams have shown that they also contain high levels of heavy metals (Table 35.3).

BIOLOGICAL EFFECTS

Changes to the physical, hydrological, and water quality conditions of urban streams profoundly influence their ecology. Many of the original invertebrate species that lived in these streams have been lost and the remaining invertebrate communities tend to be characterised by a relatively few hardy species. As noted by Paul and Meyer (2001), ecological responses to urbanisation are less studied than physical responses. Within New Zealand, only a few studies dealing with invertebrates in urban streams have been published (Suren 2000; Hall *et al.* 2001; Maxted *et*

Table 35.3 Heavy metal concentrations (mg per kg of biofilm) in organic biofilms collected from cobbles in three Christchurch urban streams and one rural stream near Christchurch: * = exceed ANZECC ISQC – low trigger value guidelines (65 mg/kg for Copper, 200 mg/kg for Zinc, 50 mg/kg for lead); ** exceed ANZECC ISQC – High trigger value guidelines (270 mg/kg for Copper, 410 mg/kg for Zinc, 220 mg/kg for lead).

Site	Land use	Copper mg/kg	Zinc mg/kg	Lead mg/kg
St Albans Stream	Old residential properties	51.5	1030**	281**
Waimari Stream	Old residential properties	52.7	180	176*
Curletts Road	Industrial estate	305**	1940**	105*
Silverstream	Rural	6.3	43.5	10.4

Table 35.2 Median values of various heavy metals, total suspended solids (TSS), and PAHs collected from 5 urban streams. Stormwater samples were collected weekly for 20 weeks, while sediment samples were collected once at each site from a composite of sediment in a 5 m length of stream; * = exceed ANZECC ISQC – low trigger value guidelines; ** exceed ANZECC ISQC

Site	Riparian landcover	Copper		Zinc		TSS	Lead	PAH	
		Stormwater (mg/m ³)	Sediments (mg/kg)	Stormwater (mg/m ³)	Sediments (mg/kg)	Stormwater (mg/m ³)	Sediments (mg/kg)	Stormwater (mg/m ³)	Sediments (mg/kg)
St Albans Stream (Christchurch)	Old residential properties	0.9	63.3	12	855**	5.8	324.3**	657	124 000**
Pakuranga Stream (Auckland)	Park, sports fields	2.2	44.7	38	425.3**	8.6	69.2*	150	885
Botany Downs Stream (Auckland)	Riparian reserve	2.5	17.1	13	173.2	8.7	24.1	229	228
Addington Stream (Christchurch)	Park, sports fields	1.4	134.4*	18	959**	2.3	540.9**	215	18 400*
Waitawhiriwhiri Stream (Hamilton)	Riparian reserve	1.0	56.2	20	821.3**	28	121.5*	794	741

al. 2003), and no studies dealing with fish or aquatic plants in urban streams. There have, however, been many unpublished contract reports written for local or regional authorities that summarise the biological communities of urban streams.

Macrophytes and bryophytes

New Zealand has approximately 107 endemic and introduced species of aquatic macrophytes, including free-floating ferns, floating-leaved macrophytes, submerged plants and charophytes (Chapter 14), and at least 96 species of aquatic bryophytes (Suren 1996). Despite such a diverse flora and the importance of these plants to stream ecosystems, there have been few studies of plant communities in urban streams. This is surprising, especially considering the fact that excessive macrophyte growth can impede stream drainage (Newell 1997) and interfere with recreational values (Schwarz and Snelder 1999). The latter is of particular concern to cities such as Christchurch, where each year, over \$500,000 is spent removing excess macrophytes from sections of the Avon, Heathcote and Styx rivers as part of the waterways management programme (Owen Southern, Christchurch City Council, pers. comm.). Macrophyte removal within Christchurch has become increasingly important since the discovery of the noxious macrophyte *Egeria densa* in the Avon River at Kerrs Reach, a popular rowing area. This plant was probably introduced accidentally on rowing equipment from the North Island, where it is relatively widespread. Trials using the herbicide Diquat have been successful in eradicating *E. densa* from the Avon River, but constant vigilance is needed to prevent further invasions.

Invasions of introduced macrophytes are common in many urban streams, although good long-term data on their prevalence are lacking. An exception is the Styx River, which flows through a semi-urbanised catchment near Christchurch and was first surveyed botanically in the early 1950s (Connor 1953), and resurveyed from 1977 onwards at approximately five-year intervals (Robb 1980; Carroll and Robb 1986; Robb 1989; Robb *et al.* 1994). During this 50-year period, large changes have been apparent in the tall-growing submerged aquatic macrophyte vegetation. Cover of two native species (*Potamogeton cheesemanii* and *Myriophyllum propinquum*) has decreased dramatically, while cover of the introduced *Potamogeton crispus* has increased greatly. *P. crispus* was first observed in a small drain entering the Styx in the mid-1970s; it subsequently spread throughout the lower reaches of the river and is now co-dominant with a much longer-established, introduced species, *Elodea canadensis*.

Surveys of Auckland urban streams have also shown a preponderance of introduced plants, including *P. crispus*, *Lagarosiphon major*, *E. canadensis*, *E. densa* and *Vallisneria spiralis*. These plants have often displaced native plants

and can choke a stream channel completely. Some of these plants (e.g., *E. densa* and *V. spiralis*) are classified as noxious plants, such is their potential to degrade instream conditions and spread to other areas. Lack of overhanging shade in many urban streams further exacerbates the degree to which these plants grow and choke stream channels.

Another group of plants found in urban streams is the bryophytes (mosses and liverworts). Unlike macrophytes, these non-flowering, small-statured plants lack roots, but instead attach themselves firmly to the substrate by small rhizoids. Bryophytes are particularly common on stable substrates such as bedrock (Suren and Duncan 1999), and grow on concrete drains and gutters in urban streams. Although many species of aquatic bryophytes are found in New Zealand (Suren 1996), only a few species appear to be common in urban areas (e.g., *Acrocladium cuspidatum*, *Bryum laevigatum*, *B. blandum*, *Fissidens rigidulus*, *F. berteroi*, *F. taranga*, *Leptodictium riparium*, *Racopilum strumiferum*), with *Fissidens rigidulus* and *Leptodictium riparium* being particularly widespread. Aquatic bryophytes often provide the only instream cover for invertebrates in concrete-lined urban streams, and can support high densities of invertebrates, mostly midge larvae (Suren 1991). Thus, they represent an important habitat within urban streams, especially in concrete-lined streams (Wilding 1996; Linhart *et al.* 2002).

Even the most highly urbanised streams can still have high biodiversity, as evidenced by the occurrence of a nationally endangered moss *Fissidens berteroi* in the aptly named Motions Creek, Auckland. This stream flows from Western Springs, through the Auckland zoo (where it historically received large nutrient inputs from the hippopotamus enclosure) and into the Waitemata harbour. It has bedrock substrate along much of its lower length, where an extensive population of *F. berteroi* has been found (Beever 1995).

Invertebrates

Suren (2000) emphasised the loss of many mayfly and stonefly taxa in urban streams, and shifts in the invertebrate community to one dominated by taxa such as Oligochaeta, the snail *Potamopyrgus antipodarum*, orthoclad and chironomid midges, ostracods and the blackfly *Austrosimulium*. These faunal changes have occurred as a result of changes to the physical and water quality conditions caused by urbanisation (Fig. 35.8), so only taxa that can tolerate these conditions are found.

Effect of imperviousness

An emerging paradigm from overseas studies is that adverse impacts of urbanisation on invertebrate communities can occur with as little as 10–20% of the land covered by impervious surfaces (Paul and Meyer

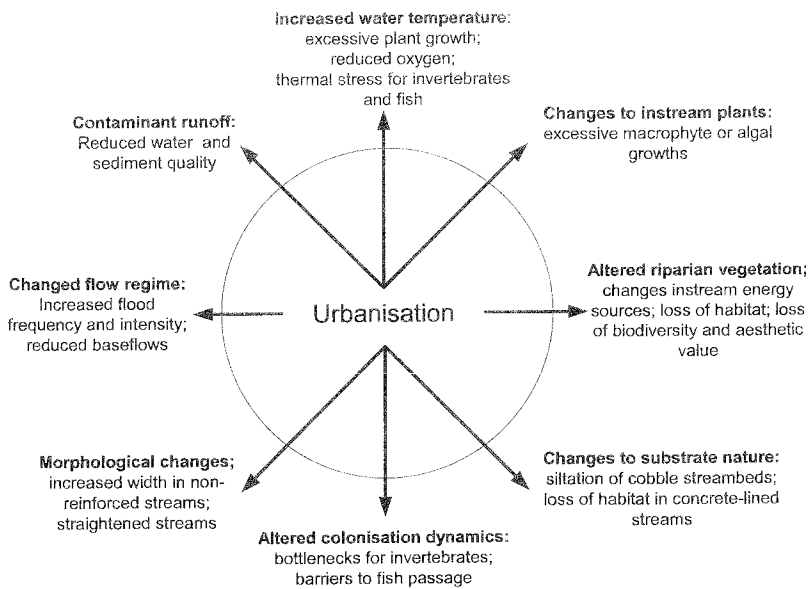


Figure 35.8 Summary diagram of the effects of urbanisation on a number of inter-related habitat factors.

2001; Beach 2002). Many studies have illustrated a negative correlation between percent impervious land cover and biological community health, and a particularly sharp decline in biological indices even at this relatively low level of imperviousness (Jones and Clark 1987; Lenat and Crawford 1994). Similar patterns are apparent in New Zealand, with Herald (2003) reporting a decrease in the Macroinvertebrate Community Index (MCI; Stark 1985) with increasing % impervious area among streams draining the Waitakere Ranges (Fig. 35.9). Allibone *et al.* (2001) also reported a reduction in the number of Ephemeroptera,

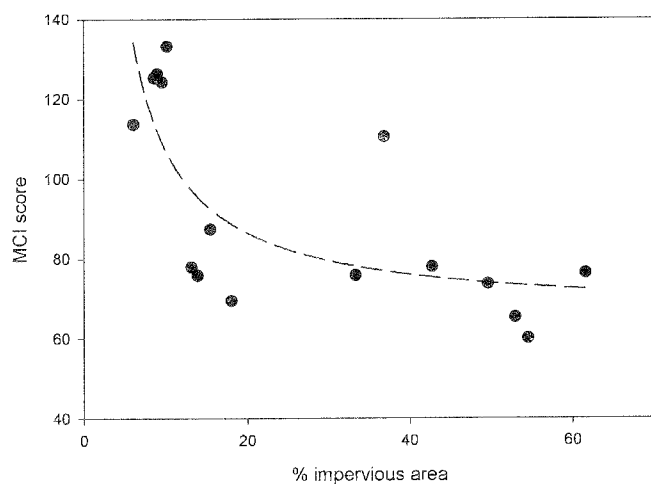


Figure 35.9 Relationship between invertebrate community composition (expressed as MCI scores) and % impervious area of four catchments in the Waitakere Ranges (from Herald 2003).

Plecoptera and Trichoptera (EPT) taxa from 11 at sites with low (10%) catchment imperviousness, to only 1 and 2 at sites with c. 30 – 40% imperviousness (Fig. 35.10). At sites with > 40 % catchment imperviousness, no EPT taxa were found.

Despite this strong trend within Auckland, Suren *et al.* (1998) found no relationship between percent impervious catchment and any biological metric in a nationwide survey of 59 urban streams (see Fig. 11.3 in Suren 2000). The lack of a strong response to urbanisation, even within a particular city, was explained by the fact that some of the streams flowed through highly modified peri-urban areas, where both habitat and water quality were degraded as a result of historic drainage practices and contemporary agricultural activities. As such, the faunas in these essentially rural catchments were already

highly degraded.

Effect of hydrological changes

Urbanisation can have strong effects on both high and low flows, and may alter both their intensity and duration. High flows can wash away invertebrates, especially in sites such as concrete-lined streams, where there is no shelter from fast-flowing water, and where only very small animals can live (Wilding 1996). High flows can also cause extensive substrate movement, which has detrimental effects on invertebrate communities (Death 1995; Townsend *et al.* 1997). Low flows are likely to have strong

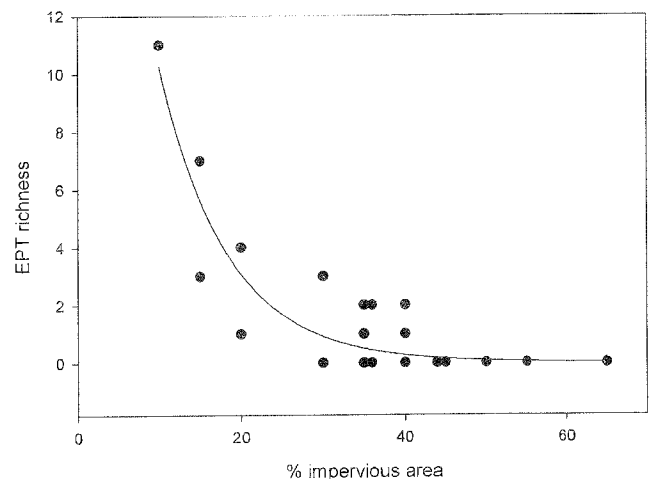


Figure 35.10 Relationship between EPT richness and % impervious area of 20 urban streams in the Auckland city region (from Allibone *et al.* 2001).

indirect effects on invertebrate communities, as they are often associated with increases in water temperature, low dissolved oxygen content, and excessive algal or macrophyte growth. Sediment deposition is also commonly a problem during low flows, causing further stress on invertebrate communities.

Effects of contaminated sediments and water quality

Sediments and biofilms in many urban streams contain high levels of heavy metals and polycyclic aromatic hydrocarbons (PAHs). The effects of heavy metals on instream invertebrate communities has been investigated in streams polluted by mine drainage (Chapter 36), but few studies have focused on how contaminated sediments affect invertebrate communities within New Zealand.

A recent study (Golding 2002) examined the chronic (28-day) toxicity of dissolved and particulate metals (copper, lead and zinc) in stormwater entering two Auckland urban streams by measuring the survival of adult and juvenile freshwater snails (*Potamopyrgus antipodarum*), in small *in situ* cages. Snail survival was lower in the two urban streams than in a reference stream in a native bush catchment. Juvenile snails were more sensitive than adults, with 100% juvenile mortality at half of the sites after 28 days. Adults were more tolerant, but still only 10% survived in some of the cages placed in the urban streams. Measurements of dissolved O₂, pH and temperature showed that these variables were within the tolerance limits of these animals—their low survival was attributed to a toxicity due to metal contamination during a storm (Golding 2002). In addition, exposure to elevated dissolved metal concentrations under baseflow conditions in the urban streams may have also contributed to the high mortality rates. The high mortality of caged *P. antipodarum* in these Auckland study sites emphasises the potential adverse effects of stormwater on urban streams, particularly as *P. antipodarum* is regarded as one of the more robust of our freshwater invertebrates, and is highly tolerant of many different land-use activities (Quinn 2000; Suren 2000). The fact that this animal was absent from the majority of

the urban sites suggests that instream conditions constrained the invertebrate communities there.

In situ cages were used to test the effects of contaminated sediments on invertebrates in three urban streams in Christchurch—here the water quality is relatively high, reflecting the streams' spring-fed sources. However, organic biofilms on cobbles can be highly contaminated with heavy metals (A. Suren, unpublished data: Table 35.3), which could be partially responsible for the absence of sensitive invertebrates such as *Deleatidium* from Christchurch streams. To test this, cobbles with contaminated biofilms were collected from riffles in three shaded Christchurch urban streams and placed into cages, which in turn were placed in fast-flowing areas of each stream. Cobbles with uncontaminated biofilms were collected from a rural site, and placed in other cages to act as controls. One hundred *Deleatidium* nymphs were placed in each cage and their survival assessed after 10 days. Nymph survival was low in the control cages at St Albans and Curletts Road Creeks, most likely reflecting the high accumulation of sediment on the cobbles in each cage. However, survival of nymphs exposed to the contaminated cobbles in these sites was even lower (Fig. 35.11). This suggests that metal contamination may be further stressing these animals. Cages at a third stream (Waimairi) had accumulated much less silt, and so

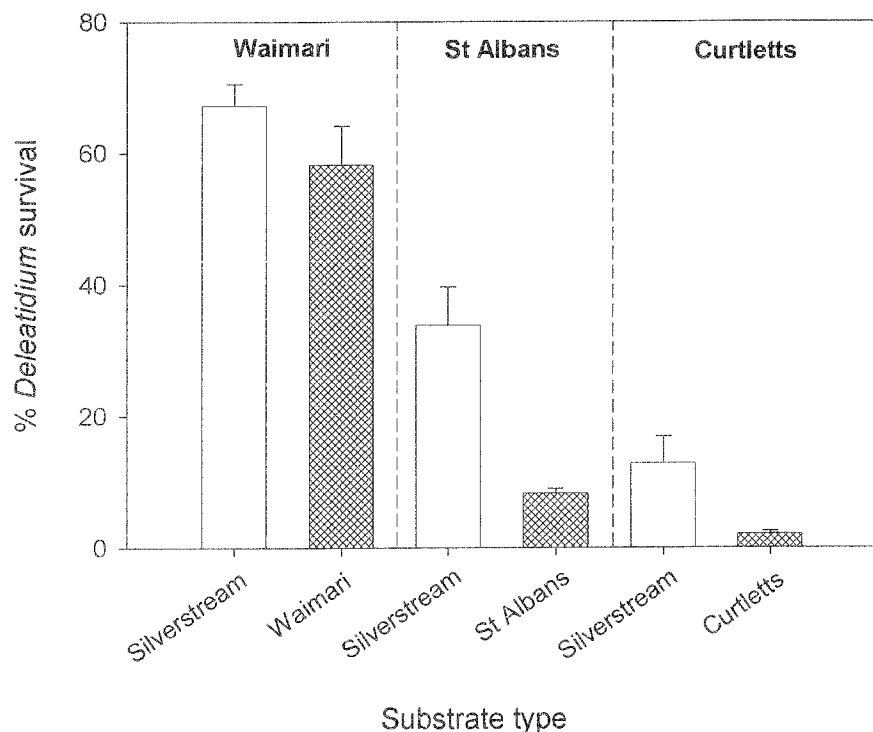


Figure 35.11 Survival after 10 days of *Deleatidium* nymphs placed in experimental cages ($\bar{X} + 1$ SE, $n = 5$) in three Christchurch urban streams. These cages contained either cobbles with contaminated biofilms (hatched bars) or cobbles with uncontaminated biofilms (open bars).

Deleatidium survival in the control cages was relatively high (Fig. 35.11). However, survival was lower on the contaminated cobbles, again implicating the effects of metal-contaminated biofilms. The absence of *Deleatidium* from many Christchurch urban streams thus most likely reflects a combination of factors, including metal-contaminated biofilms and sediments and high sedimentation levels that smother otherwise suitable habitats.

Fish

We could find no published accounts in the scientific literature of the fish communities within New Zealand urban streams, although a large number of contract reports have been prepared for statutory bodies. However, overseas studies have shown a sharp decline in fish diversity, abundance and recruitment in catchments with > 10–15% impervious surfaces (Beach 2002). Detailed studies by the Ohio Environmental Protection Agency have shown three levels of fish response to urbanisation: with 0–5% urban land use, sensitive species are lost; from 5–15%, habitat degradation and loss of benthic invertebrates cause further losses, and > 15% urban land use results in severe degradation of the fish fauna. Although this model has not been verified, many studies have reported consistent decreases in the fish multimetric index of biotic integrity (IBI), with “precipitous declines in fish IBI metrics between 0–15% imperviousness, above which fish communities remain degraded” (Paul and Meyer 2001). The situation in New Zealand is not clear-cut, and as no equivalent of the IBI exists, functional relationships between the integrity of the fish community and the degree of urban development are unclear.

There are c. 35 native freshwater fish in New Zealand, as well as 21 introduced fish (see Chapter 17). Many of the native species are diadromous, and need free access to the sea to complete their life cycles. Consequently, these species may be absent from even the most pristine streams if there are barriers to free fish passage (Joy and Death 2000). Other species are relatively weak climbers and swimmers, and cannot penetrate far inland. Therefore, distance to the sea, stream altitude and slope are important factors influencing fish distributions throughout New Zealand. Because most New Zealand cities are coastal, urbanisation may have negative influences on New Zealand’s fish communities, as it does overseas.

Examination of the New Zealand Freshwater Fisheries Data Base (NZFFBD) and commercial reports (e.g., Eldon and Taylor 1990; Sides and Bennett 1998; Allibone *et al.* 2001; McMurtrie and Taylor 2003) have shown that there is considerable fish diversity even within urban centers such as Auckland and Christchurch (Table 35.4). Within

Christchurch, the most common fish are common bully, shortfin eel, upland bully, longfin eel, bluegill bully, brown trout, inanga, and lamprey. Shortfin and longfin eels are widespread throughout Auckland urban streams, being found in 47 (shortfin) and 27 (longfin) of 64 streams surveyed by Allibone *et al.* (2001). Other widespread fish were common bully, banded kokopu and inanga, which were found at c. 25% of sites surveyed. In the Auckland area even relatively uncommon fish such as banded and giant kokopu can be found in urban streams, as long as these streams have remnant riparian vegetation, pools and good instream cover (Sides and Bennett 1998; Allibone *et al.* 2001). Indeed, the presence of riparian understory and instream cover were strong predictors of the presence of fish in Auckland streams, along with site slope, altitude and stream size, which are not greatly affected by urbanisation. Given the occurrence of species with high conservation value in urban streams, observed relationships between percent imperviousness of the catchment and fish diversity in streams may be less applicable in New Zealand than in countries such as the United States.

Importance of fish passage

Habitat modifications such as piping and culvert construction within urban areas can strongly influence fish communities in streams. For instance, the presence of a 600-m-long pipe downstream of an enhanced stream site in Christchurch was thought to have prevented the upstream migration of fish species that might otherwise have colonised it (A. Suren. pers. obs.). Similarly, the presence of a small weir across another Christchurch stream was thought to have prevented the upstream migration of the common bully (Sykes *et al.* 1998). Pipes do not necessarily have negative impacts on fish faunas, however. Herald (2003) observed banded kokopu in the headwaters of three streams in the Waitakere Ranges that had regenerating native bush in their riparian zones, even though these fish had to negotiate degraded lower reaches, including a 400-m length of culverted channel in one of the streams. In fact pipes appear to have few adverse impacts on the upstream movements of banded kokopu within Auckland streams in general, as large populations of these fish can be found in sections of small shaded streams that are located upstream of pipes and culverts (Allibone *et al.* 2001).

The presence of pipes and culverts has even helped conserve fish in other streams. For example, Taylor (2002) found a very large upland bully (*Gobiomorphus breviceps*) in a small urban stream that flows into the Styx River, north of Christchurch, via a large pipe. This pipe effectively prevented the upstream migration of predatory fish such as eels and trout, and allowed the non-migratory upland bully to thrive.

Table 35.4 List of fish species found in urban waterways: data from the New Zealand Freshwater Fisheries Data Base and numerous reports for regional and local body councils.

Fish species	Auckland	Hamilton	Wellington	Nelson	Christchurch	Dunedin	Invercargill
Banded kokopu	*	*	*	*	*	*	
Black flounder					*	*	
Brown trout		*	*	*	*	*	
Bluegill bully					*		
Catfish	*	*					
Canterbury mudfish					*		
Chinook salmon					*	*	
Common bully	*	*	*	*	*	*	
Common smelt	*	*	*	*	*		
Crans bully	*						
Giant bully	*		*	*	*		
Giant kokopu	*		*			*	
Golden orfe	*				*		
Goldfish	*	*	*	*	*		*
Grey mullet		*					
Inanga	*	*	*	*	*	*	
Koaro	*		*	*		*	
Koi carp	*	*	*	*			
Koura	*		*	*		*	
Lamprey			*		*		
Longfin eel	*	*	*	*	*	*	
Mosquitofish	*	*					
Perch	*	*			*	*	
Rainbow trout	*						
Redfin bully	*		*	*	*	*	
Rudd	*	*			*		
Shortfin eel	*	*	*	*	*	*	
Shortjaw kokopu			*				
Tench	*						
Torrentfish	*	*		*			
Triplefin	*				*		
Upland bully				*	*		
Yelloweyed mullet	*				*		
Species richness	24	15	16	15	21	13	1

Importance of habitat

The occurrence of kokopu in some Auckland streams reflects the presence of both good riparian vegetation cover around these streams and instream cover (Sides and Bennett 1998; Allibone *et al.* 2001)—indeed fish in general are absent from, or scarce in, most open concrete waterways throughout Auckland. However, instream cover does not have to be natural to provide beneficial habitat for fish. The value to fish communities of four different bank types—stone-clad banks with or without undercuts, and unclad banks with overhanging *Festuca* or *Carex*, or with short grass—was assessed in the Avon River in central

Christchurch (McMurtrie and Taylor 2003). Total fish density (number/m²) varied significantly among bank treatments and was highest among rough-edged banks with substantial *Festuca* grass overhangs. *Carex* grass banks supported somewhat lower fish densities, with the refuges created by overhang vegetation being diminished by root infilling. Unclad banks with short grass supported only low fish densities (McMurtrie and Taylor 2003) but cut-stone-clad banks with undercuts, even without a vegetation overhang, provided refuge for high densities of fish, particularly common and upland bullies. In contrast, stone-clad banks that lacked an undercut supported low

fish densities. These results highlight the importance of habitat where fish can shelter, and suggests that even artificial banks can provide good fish habitat if there are undercuts or other forms of cover within them.

MITIGATION

Traditional practices associated with urban development and stormwater management have had major detrimental effects on stream ecosystems. There is now both an increasing social awareness, and legislative requirements to enhance the “life-supporting capacity” of urban streams, as well as their aesthetic, social and recreational values. Such awareness has led to the development of methods to protect and enhance urban streams. For example, in older parts of Auckland there are still combined stormwater and sewage drains that frequently overflow into the streams and coastal areas. There are ongoing efforts to reduce such overflows, but this is expensive. In Christchurch there is an active programme of stream rehabilitation, where streams that were basically drainage ditches have been widened and their margins re-planted (Chapter 38). In new urban developments, a protected riparian area is usually established along the main streams. This not only allows for protection of riparian vegetation, but also reduces the need to modify the channel to stop bank erosion or to reduce flood water levels.

North Shore City is offering financial incentives for residents to install rain tanks for garden irrigation to reduce stormwater discharge, and in Auckland, design manuals for such low-impact development techniques have been developed (e.g., Auckland Regional Council 2000, 2003). Hopefully it will only be a matter of time before new urban developments in New Zealand will discharge all stormwater into biofilters that provide treatment, as has been achieved in Australia (Lloyd *et al.* 2001).

Some of the greatest changes to urban streams are modifications to their flow regimes—minimising the changes in flow are major challenges to drainage engineers, urban planners and environmental scientists. There is a growing trend to impose “hydraulic neutrality” on new urban developments, so that they will not change the original hydrological regime. Such a concept requires innovative approaches that focus on reducing the volume and rate of stormwater runoff, such as reducing impervious areas, protecting the infiltration capacity of soils, re-vegetating, and using shallow open drainage-ways (swales) rather than pipes. Flood detention ponds have been used in many locations to reduce flood peaks and flooding of properties in urban areas. They are of limited benefit in terms of protecting stream habitat, however, as they are usually designed for very infrequent floods that would disturb the stream even under natural conditions. More

recently, the use of larger ponds with lower release rates has been promoted in Auckland to control erosive flows (Auckland Regional Council 2003).

Better controls are needed to reduce sediment inputs from urban earthworks. Such controls should include restriction of earthworks in winter, re-grassing of bare soil, and construction of silt-trapping ponds. However, these measures can be only partially effective, and concerns about the deposition of earthworks-related sediment into estuaries is restricting the location and intensity of urban development in Auckland.

In Auckland all new developments are required to include stormwater treatment devices designed to remove 75% of the sediment (Auckland Regional Council 2003). If widely implemented, they would reduce sediment input to streams, and also reduce the rate of buildup in estuaries and enclosed marine embayments (Williamson *et al.* 1999; Williamson and Morrisey 2000). However, there are still concerns about the accumulation of fine sediments and colloidal material from existing developed areas that make it through such treatment systems.

There is also a need to reduce the amount of heavy metals and other pollutants reaching streams. Pandey *et al.* (2003) illustrated how road-side filtration devices lined with absorbent material have the capacity to bind metals such as zinc and copper, and suggested they could be used at strategic road intersections to help minimise contaminants from roads.

FUTURE RESEARCH DIRECTIONS

Urban landscapes represent ideal environments for furthering ecological knowledge, especially of ecosystem gradients (McDonnell and Pickett 1990). The gradient paradigm contends that ecosystem structure and function are a result of environmental patterns. These patterns themselves are ordered in geographical space—where there are steep changes in environmental patterns over small geographical areas, there will be corresponding steep changes in ecosystem structure and function. Such steep gradients exist in the urban-rural gradient. Urban areas have a dense, highly developed core of commercial and industrial infrastructure, surrounded by a less dense peri-urban matrix of predominantly residential infrastructure, which in turn is surrounded by rural or natural areas. Such steep urban-rural gradients are regarded as ideal situations in which to investigate many ecological questions (McDonnell and Pickett 1990).

Such questions could include:

- Are there any ecological generalities that can be made about the ability of different plant, invertebrate or fish to live in cities?
- What aquatic species disappear, and why?

- Are there any unifying processes that cause stream degradation as a result of urban development?
- Is there a threshold effect of urbanisation, as observed with the apparent decline in ecological health in overseas studies when > 15% of the catchment becomes impervious, or do communities vary in a continuous manner with increasing urbanisation?

Identification of a threshold effect, however, relies on accurately measuring the degree of urban intensity. This is a complex problem, and highly dependent on measuring parameters at relevant spatial scales. For example, is it the immediate riparian land cover that is important, or the road density, or the percentage of the entire catchment that is impervious? Identifying relevant and accurate ways to measure these parameters is also problematic, as many of the existing land cover databases soon become obsolete as new subdivisions are built. McMahon and Cuffney (2000) present a multimetric index that provides a consistent and objective index for assessing the degree of catchment urbanisation. This index integrates information about multiple facets of human influence on the urban landscape, and includes variables that describe land cover, infrastructure and socioeconomic data. The major benefit of this multi-metric approach is that it deals with many of the factors associated with urbanisation, rather than merely looking at the % impervious cover as many New Zealand studies have done to date. Its use within New Zealand may shed further light on how urban development influences instream communities.

Closely related to the need for a more descriptive index of urbanisation is to look in greater detail at the processes by which urbanisation adversely effects on stream ecosystems. Accurate identification of these processes will allow managers to develop strategies to avoid, mitigate or remedy the adverse effects. Recent research has suggested that it is not the % catchment imperviousness *per se* that is the driving force behind the degradation of stream health, but rather it may be the proportion of impervious material in a catchment that is connected to a stream by pipes or drains. Studies in the Dandenong Ranges on the eastern fringes of Melbourne (Walsh *et al.* 2001) have shown that the ecological condition of streams there was more tightly correlated with drainage connection than with the % imperviousness within a catchment. A further study showed that the distribution of the threatened stream-dwelling amphipod *Austrogammarus australis* in Dandenong streams was best described by drainage connections (Walsh *et al.* 2004). These finding suggests that it is the path by which water is delivered to a stream that is critical, and reinforces the contention that water-sensitive urban developments that minimise stormwater runoff and allow it to infiltrate into the soil or be retained in wetlands may indeed minimise the adverse effects of urbanisation on waterways. Minimising drainage con-

nection is analogous to providing riparian fencing around streams draining pasture to minimise the effects of agriculture by effectively isolating the stream from the surrounding catchment. Research of this kind should help strengthen statutory planning requirements and urban development guidelines to minimise stormwater runoff and achieve hydrological neutrality in urban areas.

Finally, it is necessary for planners and resource managers to realise that urban streams should be viewed for more than their utilitarian roles as drainage, and that achieving healthy streams is important for both the environment as well as for people and businesses. Ecologically healthy streams are aesthetically pleasing and their banks become attractive places to live and for recreation, and this will have profound effects on the social importance of these streams. Urban streams may provide the only chance some people have to glimpse nature, and as such provide excellent opportunities for integrating both social and ecological sciences. Managing these streams as valuable assets within the urban landscape may help future urban developments occur without compromising the ecological condition of these systems. An apparent paradox thus exists, in that urban streams can be extremely important for urban dwellers and educate them of the value of green-spaces. However, these same streams often become degraded during the process of urbanisation. The real challenge for planners and resource managers is to develop ways to minimise these adverse impacts through novel catchment-wide engineering approaches.

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Chapter 34

Impacts of agricultural land use

Stephanie Parkyn and Bob Wilcock

INTRODUCTION

To understand the effects of agriculture on waterways, it is important to acknowledge that streams are products of their catchments. Rainwater interacts with soil and vegetation to produce runoff from hillslopes, or it percolates into groundwater, and is transported downstream in springs and streams. Therefore, catchment land use can influence stream chemistry, morphology, and aquatic biota. Downstream environments, such as lakes and estuaries, can also be affected by what happens on the land many kilometres away. Thus, the impacts of agricultural inputs to streams can be localised, but they will also have cumulative effects downstream.

Research on the environmental effects of agriculture

One of the first problems to be identified as an effect of agricultural land use was water quality degradation. Early research on water quality was focused on point source discharges—effluents from human sewage disposal and agriculture-related industries, such as wastes from dairy factories and meatworks, as well as numerous small point sources of animal wastes from piggeries and dairy sheds (Rutherford *et al.* 1987). These provided major sources of nutrients (nitrogen, phosphorus, and organic material) to New Zealand freshwaters. Substantial improvements in waste treatment in the 1970s and 80s led to reductions in point source discharges and a change in research focus towards the problems of non-point, or diffuse, sources of pollution (see Chapter 11 for a discussion of the sources of pollution).

Initially, research into non-point source pollution from agriculture was concentrated on estimating total annual loads of nutrients per unit catchment area (called specific yields or, simply, yields, with the units kg ha/yr) for different land uses (e.g., Wilcock 1986), and determining the

yields of nutrients derived from fertiliser application and livestock wastes (Rutherford *et al.* 1987). Research showed that the total diffuse amounts of phosphorus and nitrogen reaching New Zealand rivers from pasture was greater than the total amounts from point sources. By the time of the Smith *et al.* (1993) review of agricultural effects on waterways it was apparent that diffuse sources of contaminants to streams had a major impact on water quality around New Zealand.

There had been few investigations into the effects of agriculture on the ecology of streams up until the early 1990s, but during that decade there was an expansion in studies of habitat and biota of streams in areas with differing land uses. Profound changes in habitat have accompanied deforestation and conversion to pasture, with dramatic changes to the abundance and diversity of species found in New Zealand streams (Quinn 2000).

In this chapter we review the current levels of understanding of the effects of agriculture on the water quality and ecology of streams. We look at ways of reducing the impact of farming on streams, and identify some of the important research problems for the future of sustainable land use in New Zealand.

FARMING IN NEW ZEALAND

Pasture is the dominant land cover in New Zealand, covering 38% of the country (Fig. 34.1), with over half of the land area in the North Island devoted to pasture (Statistics NZ 2003). Indigenous forest cover has been reduced from approximately 85% of New Zealand's land area to about 23% (Taylor and Smith 1997). The deforestation of large tracts of land and the use of northern hemisphere farming techniques increased after the large-scale European colonisation in the 1850s. Harding (2003) noted that 99.9% of the forest was removed from Banks Peninsula, Canterbury, in 60 years. Agricultural develop-

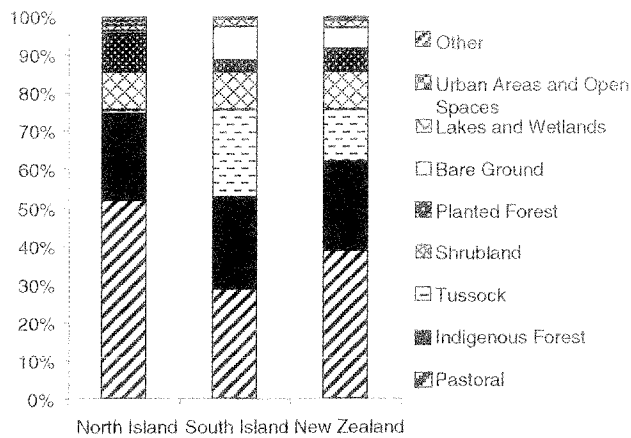


Figure 34.1 New Zealand land cover, for the 1996/1997 period (from Ministry of Agriculture and Forestry through Statistics NZ).

ment began in the lowlands, but intensified in the hill country after the development of aerial top dressing with fertilisers in the 1950s.

By 2002, pastoral land covered 12 million ha, planted production forest covered 1.8 million ha, and horticulture 110,000 ha (Statistics NZ 2003). From 1994 to 2002, the intensity of farming increased, with declining numbers of sheep and increasing numbers of dairy cattle in several regions of New Zealand (Table 34.1). Dairy cattle reached 5.2 million in 2002, with the highest number in the Waikato region. Deer farming has increased four-fold since 1994, with a total of 1.6 million deer in 2002. The Canterbury region has seen increases in dairy and deer farming, and although the number of sheep being farmed decreased between 1994 and 2002, the region still has the largest number of sheep. Canterbury, Southland, Otago, and Hawkes Bay have all shown a >150% increase in the numbers of dairy cattle between 1994 and 2002.

Stocking intensities vary from 5 to 20 stock units/ha on hill land for sheep, beef, and deer farming, while dairy

farms average about 2.4 cows/ha on flatter land (Gillingham and Thorrold 2000). A standard stock unit (SU) is equivalent to a 55-kg ewe, and a dairy cow is roughly equivalent to 6 SU in terms of pasture requirements (Metherell and Morrison 1984).

New Zealand pastures have historically been fertilised predominantly with superphosphate to address deficiencies in soil phosphorus and in sulphur content. Nitrogen fixation by forage legumes, particularly white clover, has been used to maintain adequate soil nitrogen levels. Urea is produced locally and is widely used as a source of nitrogen fertiliser. On a smaller scale, some farmers fertilise land by spray-irrigating with dairy shed and pond wastes. About 0.18 million tonnes of phosphorus, predominantly as single superphosphate fertiliser, are spread on New Zealand farm land annually (Gillingham and Thorrold 2000).

WHAT ARE THE EFFECTS OF AGRICULTURE?

A traditional approach to investigating the effects of agriculture on water quality has been to compare streams in pasture catchments with those in native forest. Native forest streams are used as a representation of natural conditions prior to the conversion to pasture and thus can show us how the streams should be functioning. Table 34.2 summarises the responses of streams to changes in land use and outlines the mechanisms and impacts of this change.

Water quality

Pasture catchments have higher runoff, water yield and storm flows than native forest catchments (Davoren 1986; Dons 1987; Cooper and Thomsen 1988). Pasture vegetation (e.g., grass) has lower interception and evapotranspiration rates than forest (Fahey and Rowe 1992), and livestock trampling compacts the soil and reduces infiltration (Trimble and Mendel 1995; Nguyen *et al.* 1998).

Pasture streams have higher amounts of suspended

Table 34.1 Numbers of livestock in 1994 compared with 2002 in selected regions of New Zealand, and totals for the North and South Islands and all of New Zealand (source: Statistics NZ 2003).

	Sheep (000)		Dairy (000)		Beef (000)		Deer (000)	
	1994	2002	1994	2002	1994	2002	1994	2002
Waikato	3,606	2,592	1,438	1,663	870	667	162	143
Hawkes Bay	4,264	3,789	32	89	523	556	118	129
Manawatu-Wanganui	7,458	6,564	308	417	825	726	129	145
Canterbury	9,747	7,758	212	543	453	505	259	412
Otago	7,636	6,121	82	205	270	267	91	197
Southland	7,851	5,951	114	356	203	204	185	348
Total North Island	22,516	18,414	3,278	3,832	3,907	3,347	615	598
Total South Island	26,846	21,132	561	1,330	1,134	1,148	616	1,046
Total New Zealand	49,466	39,546	3,839	5,162	5,048	4,495	1,231	1,644

Table 34.2 Changes to physical habitat and water quality characteristics of streams due to agricultural land use.

Response	Mechanisms	Impacts to stream and downstream environments
<i>Physical habitat and channel morphology</i>		
Native riparian vegetation cover and diversity reduced	<ul style="list-style-type: none"> • Deforestation • Livestock grazing, browsing, and trampling of remnant vegetation • Exposure to wind and sun – drying 	<ul style="list-style-type: none"> • Reduced shade, increased water and air temperature • Loss of cool-water organisms • Increased growth of nuisance plants and algae with increased light • Cumulative increases in water temperature downstream • Reduced channel stability • Reduced food supply to the stream • Reduced habitat cover for fish
Soil condition degraded	<ul style="list-style-type: none"> • Increased bare ground • Compaction and reduced water infiltration 	<ul style="list-style-type: none"> • Soil erosion • Greater surface runoff • Increased delivery of contaminants
Channel stability reduced	<ul style="list-style-type: none"> • Trampling by livestock • Tree removal 	<ul style="list-style-type: none"> • Bed siltation, local widening • Reduced in-stream habitat quality • Reduced visual appeal and amenity values
Channel width reduced	<ul style="list-style-type: none"> • Pasture grasses armour against fluvial erosion and trap sediments • Soil creep from hillslopes into channels • Channel width may locally increase at livestock crossings 	<ul style="list-style-type: none"> • Reduced benthic habitat • Reduced quality of benthic habitat
Bed sediment texture decreased	<ul style="list-style-type: none"> • Siltation of the streambed by fines 	<ul style="list-style-type: none"> • Reduced interstitial water exchange • Reduced epilithic food quality • Reduced benthic habitat quality
<i>Contaminants and water quality</i>		
Suspended sediment load and turbidity increased	<ul style="list-style-type: none"> • Trampling and grazing leading to bank erosion and sediment suspension • Hillslope instability • Reduced entrapment in riparian vegetation • Stock crossings 	<ul style="list-style-type: none"> • Altered habitat and food selection for sighted animals • May affect gills of fish, crayfish, and other invertebrates • Siltation of estuaries and lakes • Reduced visual appeal
Nutrients increased (N & P)	<ul style="list-style-type: none"> • Stock defecation in stream channel • Runoff from dung on hillslopes • Leaching of urine (N) • Reduced entrapment in riparian soils and vegetation 	<ul style="list-style-type: none"> • Proliferation of nuisance plants and algae in streams • Eutrophication of downstream lakes and estuaries
Agrichemicals (pesticides, herbicides, insecticides, fungicides)	<ul style="list-style-type: none"> • Poorly managed application of agrichemicals • Poorly managed sheep dip operations 	<ul style="list-style-type: none"> • Potential for fish, crayfish and other invertebrate deaths
Faecal microbes increased	<ul style="list-style-type: none"> • Defecation in stream channel • Stock crossings • Runoff from farm tracks and raceways • Reduced entrapment in riparian vegetation • Farm dairy effluent 	<ul style="list-style-type: none"> • Health risk to human water supply • Unsafe recreation and swimming • Health risk to domestic livestock • Increased water treatment costs • Contamination of shellfish in downstream estuaries

sediment, higher turbidity and lower water clarity than native forest streams (Dons 1987; Smith *et al.* 1993; Quinn *et al.* 1997; Quinn and Stroud 2002) because of increased runoff, hillside erosion, and bank instability caused by grazing (Trimble and Mendel 1995). Extreme flow events can produce much higher suspended sediment and turbidity in pastoral streams than in native forest streams, as can be seen from monthly water quality samples taken at Whatawhata, Waikato from 1995 to 1999, a period that included a 1-in-20 year flood (Fig. 34.2).

Levels of the nutrients nitrogen (N) and phosphorus (P) are generally higher in pasture streams than in native forest streams (Cooke 1979; Wilcock 1986; Cooper *et al.* 1987; Cooper and Thomsen 1988; Quinn *et al.* 1997; Quinn and Stroud 2002; Figure 34.2). Erosion, and fertilisers and animal wastes, are important diffuse sources of phosphorus. Despite the low use of nitrogen fertilisers in New Zealand, animal wastes, particularly urine, provide locally concentrated inputs of nitrogen that exceed the nitrogen requirements of the pasture and thus are susceptible to leaching (Rutherford *et al.* 1987). Elevated nutrient levels in streams are due to a combination of factors—increased runoff, eroded sediment, and subsurface leaching carrying excess nutrients from fertilizers, nitrogen fixed by legumes and stock excreta on pastures. Elevated levels may also be due to direct inputs from fertilizer drift and from stock excreta where animals have free access to waterways (Quinn and Stroud 2002). Turbidity and nutrient concentrations found in pasture streams often exceed water quality guidelines for aquatic ecosystems (ANZECC 2000).

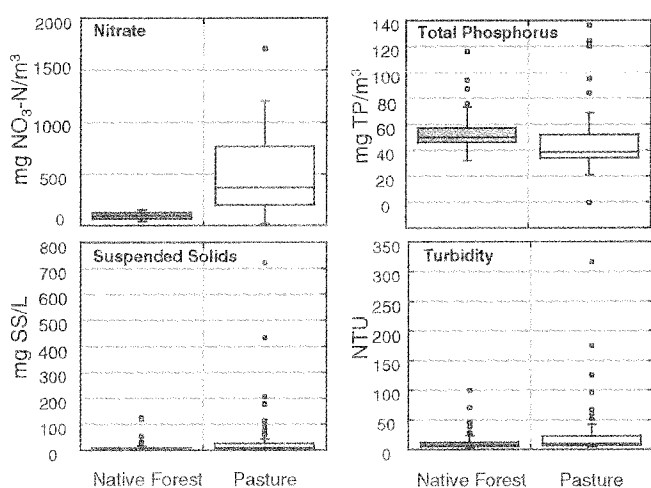


Figure 34.2 Water quality measures taken monthly from 1995 to 1999 from streams in native forest and pasture catchments (3.0 and 2.6 km² in area, respectively) at Whatawhata, Waikato (from Quinn and Stroud 2002). Open circles represent outlying values.

Pasture streams have higher levels of faecal indicator bacteria such as faecal coliform (Smith *et al.* 1993) and *Escherichia coli*, as well as pathogens including *Campylobacter*, *Salmonella*, *Cryptosporidium* and *Giardia*, than native forest streams (Donnison and Ross 1999). About 80% of the notifiable diseases in New Zealand in 2001 and 2002 were potentially waterborne and carried by humans and animals (Sneyd and Baker 2003). The presence of these harmful micro-organisms can mean that streams are unsuitable for contact recreation such as swimming, and may require a high level of treatment to be used for potable supply.

Some common pathways for diffuse pollution are surface and subsurface runoff, and direct inputs such as applications of fertiliser falling into stream channels, or cows wading streams (Nagels *et al.* 2002). Some dissolved substances, notably nitrate, are leached by rainfall to groundwater and eventually enter streams via seeps and springs. Recent studies suggest that nitrate draining through porous pumice soils in the central North Island of New Zealand may take several decades to emerge in surface waters. This makes it difficult to predict the effects of pastoral land use on water quality (Vant and Smith 2002) and to mitigate problems using conventional riparian management, as nutrients may bypass the riparian zone (Howard-Williams and Pickmere 1999).

The cumulative effects of poor water quality in streams are most often expressed in downstream lakes and estuaries, which may be affected by sedimentation and eutrophication. High phosphorus levels in lakes have resulted in blooms of toxic algae (e.g., Lake Rotoiti in the Bay of Plenty region). Algal blooms are the result of an increase in concentration of a “limiting” element, usually nitrogen (N) or phosphorus (P). If there is too little of the plant-available forms of these nutrients in the water, further plant growth is impossible. Most commonly, it is the availability of phosphorus that determines algal growth in lakes. Studies of the difference between native forest and pastoral streams help to identify the mechanisms causing increased amounts of nutrients entering lakes and estuaries.

Point sources

Farm oxidation ponds are important point sources of nutrients and faecal pollution to New Zealand streams and rivers (Wilcock *et al.* 1999). They are generally of less concern than diffuse pollution sources because of improvements in waste treatment technologies and the greater ease of regulation. However, a study of the performance of large farm oxidation ponds found that even though BOD (Biochemical Oxygen Demand, a measure of organic material), ammoniacal nitrogen, total N and faecal coliform levels were improved by 20–70% (Sukias *et al.* 2001), only half of the ponds consistently achieved

effluent quality standards. To meet guidelines for visual clarity and contact recreation, discharges from these farm ponds require dilutions of around 500-fold and >850-fold respectively. To meet nutrient guidelines, dilutions in the range of >370–925 fold are required.

Two studies in the Bay of Plenty region showed that only 30–48% of dairy pond discharges to surface water complied with consent conditions on first visits (Pickles 1999; Larsen 2000). New technologies that include aerating ponds (Sukias *et al.* 2001) and installing Advanced Pond Systems (APS) (Craggs *et al.* 2003) offer hope for improved effluent disposal to New Zealand waterways in the future.

Tile drains are subsurface drains constructed beneath flat or undulating paddocks to remove excess water from soils. These drains are a significant source of pollution to rural streams because they intercept runoff from large areas and sometimes convey strong effluent (e.g., when drained pasture is being irrigated with pond effluent), with little attenuation of pollutant concentrations (Nguyen and Sukias 2002). Furthermore, because tile drains mostly discharge directly into streams, there is little opportunity for amelioration by riparian processes, such as denitrification or uptake of nutrients by plants.

Ecology and habitat

Much of New Zealand was originally forested prior to agricultural development, and New Zealand rivers had headwaters that were heavily shaded—greater than 90% shade (Rutherford *et al.* 1999) until the stream channel widened further downstream, opening the forest streams to sunlight (> c.10 m width; Davies-Colley and Quinn 1998). Forest removal, particularly in headwater areas, has resulted in a homogenising of stream habitat conditions and the consequent loss of stream biodiversity, especially the loss of species adapted to the high shade and cool temperatures of forested headwaters (Quinn 2000).

When land is cleared and grazed, a number of factors are altered, contributing to changes in stream communities (Table 34.2). The reduction in shade leads to algal proliferations and increased temperature (Quinn *et al.* 1997; Rutherford *et al.* 1997, 1999), and there is a reduction in organic matter inputs (leaves) that are habitat and food resources (Scarsbrook *et al.* 2001). Increased nutrients add to instream plant growth, sediment concentrations may increase (Quinn and Stroud 2002), and there are changes to stream morphology and wood inputs (Davies-Colley 1997). Deepening and straightening of channels can increase stream gradients, reduce stream length and habitat diversity (Williamson *et al.* 1992), and cause increases in flow yield, variability and surface runoff (Dons 1987; Fahey and Rowe 1992).

The riparian zone, shade and temperature

The riparian zone is the three-dimensional area of land beside a stream that is a zone of interaction between aquatic and terrestrial systems. A forested riparian zone interacts with rainwater by intercepting runoff and shallow subsurface flows, and interacts with streamwater when the stream overflows into the floodplain.

Particulate contaminants (sediment and nutrients) generated on land pass through the riparian zone through surface or subsurface flow pathways. Riparian zones trap contaminants, mainly by infiltration (Lowrance *et al.* 1997), and nutrients may be utilised by plants growing in the riparian zone, or nitrate may be stripped from runoff water by a process of denitrification in riparian or wetland soils. A large proportion of the contamination of rural streams by sediment, nutrients and faecal matter occurs where livestock have access to the riparian zone and the stream channel itself. However, where riparian vegetation is in good condition and livestock are excluded or restricted, riparian zones can function as a sink—trapping contaminants from agricultural lands—rather than as a source (Collier *et al.* 1995).

Many of the factors that have changed the ecology of streams in pasture are related to the removal of vegetation in the riparian zone. Trees alongside streams provide shade and protect streams and their banks from sunlight and solar heating; they stabilise the banks to flood erosion, and provide habitat in the form of tree roots and woody debris (e.g., logs and branches) for crayfish, other invertebrates and fish. Dissolved organic matter from the catchment and leaf litter that falls into streams dominates the energy base of aquatic food webs in forest.

Fish species such as inanga and other native whitebait species require suitable riparian vegetation to spawn, as they lay their eggs in amongst grasses on stream banks, and thus are susceptible to trampling by livestock (see Chapter 17). Native fish and salmonids are also affected by the removal of trees that provide cover habitat and shade.

Studies comparing the water temperature of native forest and pasture streams generally show marked changes in the diurnal temperature maxima. Temperatures during summer can be appreciably higher in small pasture streams (by as much as 8°C) than in streams under forest shade. Temperature is one of the factors that influence the distribution of fish and invertebrates and may affect the taxa present in warm pasture streams (Fig. 34.3). Preferred temperatures for native fish species, determined from laboratory experiments, range between 16° and 27°C (Richardson *et al.* 1994). In these experiments, eels, which are abundant in pasture streams, preferred the warmer temperatures. However, New Zealand fish species are thought to have a wide temperature tolerance, and temperatures may need to be above 30°C to be lethal for

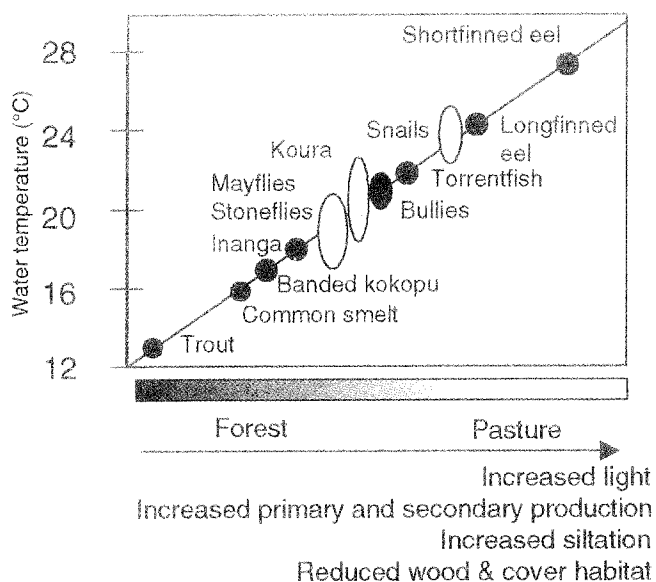


Figure 34.3 Preferred water temperatures established for fish (solid symbols) and estimated temperature ranges for invertebrates (shaded symbols) related to the potential use of waterways from native forest to pastoral land use. Data were compiled largely from experimental studies in the North Island of New Zealand (Quinn *et al.* 1994; Rowe and Chisnall 1995; Richardson *et al.* 1994; Cox and Rutherford 2000; Parkyn *et al.* 2002).

many species. Mayflies and stoneflies are affected by water temperatures greater than c. 20°C (Quinn *et al.* 1994; Cox and Rutherford 2000) and it is widely accepted that high water temperatures have caused the loss of some invertebrate species from pasture streams. The same pattern is unlikely to be found where pasture streams are cool due to inflows from groundwater springs, to climate or to altitude.

In addition to the effect of riparian vegetation on stream temperature, many stream invertebrates have adult flying stages that require specific microclimate conditions and food in the riparian zone to complete their life cycle (Collier *et al.* 1997). The longevity of adult stoneflies, *Zelandoperla decorata*, was examined for a range of diurnally varying air temperatures demonstrating a reduction in longevity with increasing temperature—adults survived for about 10 days at maximum daily temperatures of 25°C, but only to about 4 days with a maximum of 32°C (B.J. Smith, NIWA, unpubl. data). Some individuals held at cooler temperatures (maximum of 16–18°C) were able to survive for 40 to 50 days, suggesting that adults in cooler microclimates have longer to disperse, mate and lay eggs.

Livestock, bank stability, channel width, and sediment

Livestock provide direct sources of nutrient and faecal contamination by defecating in streams. Cattle particu-

larly, by virtue of their large size and affinity for water, can undermine stream bank stability and provide direct sources of sediment by trampling stream banks and riparian soils. A study on the Sherry River, Tasman District, showed a 50% increase in suspended sediment and a 400% increase in *E. coli* after a herd of cows crossed a stream to and from the milking shed (Davies-Colley *et al.* 2001). This increase was associated with the disturbance of the streambed and banks, and defecation, as the 246 cows deposited about 37 kg of faeces on two crossings of the stream.

Livestock are known to damage remnant native vegetation in the riparian zone (e.g., Sansom 1999), thereby reducing their contaminant trapping function (Belsky *et al.* 1999). Indirect effects of livestock damage are caused by trampling of the catchment and riparian soils (Belsky *et al.* 1999; Nguyen *et al.* 1998), resulting in reduced infiltration and increased contaminant runoff (Cooper *et al.* 1995).

While livestock can increase the channel width of streams locally at crossings, in general, pasture streams are narrower than streams in native forest in catchments of an equivalent size (Davies-Colley 1997). This phenomenon is largely due to the gradual development, under open light conditions, of grasses on the gravel bars, trapping sediment and armouring the banks against flood erosion (Fig. 34.4). This finding is significant for riparian management, because when shade is restored to the riparian zone to restore ecological conditions, the accumulated sediment will erode, and water quality and sediment accumulation in downstream environments may be worse for some years until the original streambed width is restored.

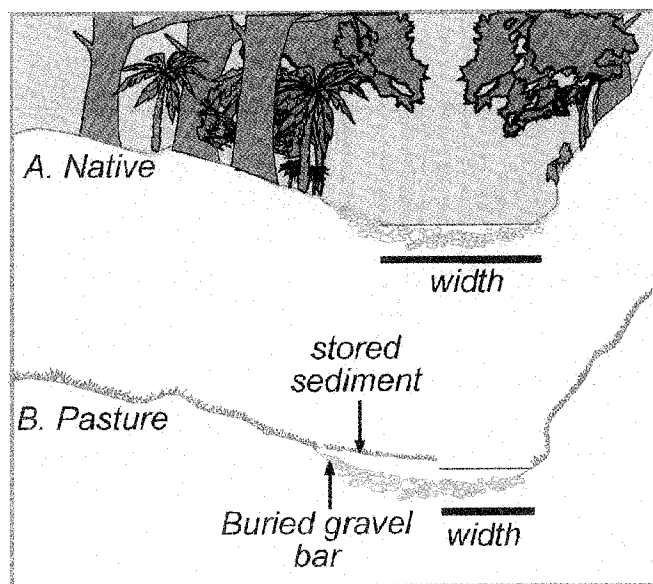


Figure 34.4 Change in stream channel width from native forest (A) to pasture (B), where pasture grasses trap sediment, resulting in narrow and incised channels (from Davies-Colley 1997).

Sediment is also generated from erosion and slips on catchment hillslopes, particularly where steep lands have been deforested and converted to pasture, or in soft-rock hill country, such as the East Cape region (DeRose *et al.* 1993; Page *et al.* 1994). Increased suspended sediment affects stream communities by reducing water clarity (affecting light for photosynthesis and vision for sight-hunting fish), and by smothering the substrate, which affects habitat and food resources for fish and invertebrates (Ryan 1991; Davies-Colley *et al.* 1992; Quinn *et al.* 1992a; Rowe and Dean 1998), as well as potentially suffocating and abrading benthic biota. Increased turbidity can affect the predator avoidance, foraging and migratory behaviour of fish (Rowe and Dean 1998; Rowe *et al.* 2000). Sensitivities to turbidity for native fish can range from thresholds of 20 NTU (Nephelometric Turbidity Units) for banded kokopu to >160 NTU. The ability of trout to use size-selective feeding can be reduced in turbid waters (Rowe *et al.* 2003). As well as visual feeding strategies, trout can effectively use lateral line detection (of movement by pressure waves) in turbid water, but high turbidities could cause changes in feeding behaviour from mobile to sedentary prey and reduce their ability to find large prey.

Periphyton

One of the most visible consequences of the removal of shade has been the increase in algal primary production. Many of the changes to New Zealand stream communities are the result of the change in the dominant food base of the stream from heterotrophic (leaf litter and microbial communities) to autotrophic (algae) production. Evidence for pastoral effects on New Zealand streams was first noted by Allen (1959), who observed that thin, brownish algal films on the streambed were replaced by filamentous and mat-forming algal cover over a 20-year period of increasingly intensive agricultural development in Horokiwi Stream, Wellington.

Excessive growth of periphyton can affect benthic faunal species richness and fish habitat, and be visually unappealing. A number of factors control the growth of algae, including light, nutrients, physical abrasion, and invertebrate grazing (Biggs 2000). Agricultural land use has been found to increase the growth of periphyton, because of higher light levels after the removal of forest trees, and through increased nutrient delivery to streams (Friberg *et al.* 1997; Quinn *et al.* 1997). To prevent periphyton biomass from reaching levels that inhibit benthic faunal species richness, Biggs (2000) suggests guideline levels of soluble inorganic nitrogen (less than 10 mg/m³) and soluble reactive phosphorus (<1 mg/m³) should not be exceeded in stable, cobble-bottomed, unshaded streams. Where nutrient levels cannot be reduced to those levels, other mitigation measures such as riparian shading may be useful. Streams that are more frequently disturbed by

floods may not require such stringent controls on nutrients—for instance, the ANZECC (2000) guideline values for upland and lowland streams are 9 and 10 mg dissolved reactive phosphorus (DRP)/m³, respectively.

Invertebrate communities

Invertebrate communities can be used as indicators of stream health, as changes in the species assemblage can reflect the cumulative effects of changes to habitat and water quality over time. Changes in land use between native forest and pasture have led to marked differences in invertebrate communities in headwater streams throughout New Zealand: see, for example, studies in Southland (Quinn *et al.* 1992b; Scott *et al.* 1994), Otago (Townsend *et al.* 1997; Riley *et al.* 2003), North Canterbury (Harding and Winterbourn 1995), and Waikato (Quinn *et al.* 1997). Differences between areas with differing land use have also been found for lowland stream faunas in Northland (Collier 1995), Waikato, and Westland (Duggan *et al.* 2002), although fewer studies have been conducted in lowland streams.

In general, pasture streams have higher total numbers of invertebrates, but fewer sensitive species such as Ephemeroptera (mayflies), Plecoptera (Stoneflies) and Trichoptera (caddisflies) (EPT taxa), and much smaller proportions of species that can feed on leaf litter. Pasture streams typically have higher numbers of taxa that are tolerant of organic pollution and sedimentation, such as worms, certain chironomid larvae, leeches, and flatworms, and increases in algal feeding taxa such as snails and the purse cased caddisfly *Oxyethira* (Table 34.3). The biodiversity of invertebrates has been linked to the amount of forest present in the catchment (Northland: Collier 1995; Banks Peninsula: Harding 2003).

Quinn and Hickey (1990) surveyed 51 medium-large rivers and found that rivers in catchments with >30% pastoral development showed stress effects that were indicated by changes to the invertebrate fauna. Typically, fewer EPT taxa were recorded and there was an increase in species that consume algae. These changes were associated with increases in summer maximum temperature, organic nitrogen concentration, and periphyton biomass. Periphyton proliferations in rivers were common in many regions of New Zealand (Biggs 1985) and, in a study of 100 rivers, a high biomass of periphyton was found to occur in catchments that were at least partially developed into pasture (>20%; Biggs 1990).

Biomonitoring is now regarded as an essential component of resource management, and the Macroinvertebrate Community Index (MCI or QMCI), developed to assess organic enrichment in stony streams, is commonly used in New Zealand (Stark 1985). The MCI-based indices, which combine the tolerance of invertebrate

Table 34.3 Generalised invertebrate community response to the change from native forest to pastoral conditions and the factors that contribute to the change. Invertebrates are useful as bio-indicators, as the invertebrate community response may differ according to the severity of changes.

Typical invertebrate community characteristics in native forest	Invertebrate community response to pastoral land use
<ul style="list-style-type: none"> • Diverse mayfly, stonefly, and cased caddisfly communities • Leaf shredding stonefly, <i>Austroperla cyrene</i> and wood cased caddisfly <i>Triplectides dolichos</i> distinctive • Taxa requiring excellent water quality found exclusively in native e.g., <i>Helicopsyche</i> sp., <i>Zelandoperla</i> sp. • Low abundance and diversity of Mollusca 	<ul style="list-style-type: none"> • Decrease in sensitive taxa, e.g., mayflies, stoneflies and most caddisflies (EPT taxa) • Increase in algal feeding species e.g., snails (<i>Potamopyrgus antipodarum</i>), some caddisflies (<i>Oxyethira</i> sp., <i>Pycnocentroides</i> sp.) • Increased abundance of total invertebrates • Increased abundance of some predatory uncased caddisflies (e.g., <i>Hydrobiosis</i> sp., <i>Aoteopsyche</i> sp.) • Abundant pollution-tolerant taxa, e.g., Diptera, Oligochaetes, Chironomids
Factors	Factors
<ul style="list-style-type: none"> • Canopy shade • Cool water and air temperatures • Leaf litter inputs as food base and habitat • Clean gravels and wood as substrates 	<ul style="list-style-type: none"> • High light • Warm water and air temperature, diurnal extremes large • Abundant algae • Silted substrates and benthic food sources • Nuisance weed growth, particularly in small or lowland streams • Organic enrichment • Low oxygen in summer in slow-flowing, lowland streams

species to organic pollution with their occurrence or abundance to produce a stream “health” score, have been reduced in streams with pastoral development (Quinn *et al.* 1992b; Scott *et al.* 1994). These indices are not designed for soft-bottomed, sluggish streams, so any effects on the invertebrate communities of lowland pastoral streams may not be revealed by the use of these indices alone.

Smith *et al.* (1993) documented that low-level inputs of nutrients and organic material (measured by Biochemical Oxygen Demand—BOD) from agricultural land may have a beneficial impact on the natural aquatic community by increasing primary and secondary production (increasing invertebrate densities) without substantially altering the community structure. However, at higher inputs, indications of ecosystem stress develop, such as sedimentation, profuse growths of algae, excessive ammonia (NH₃), and low dissolved oxygen (DO). Quinn (2000) explains these types of changes to communities in terms of a subsidy-stress conceptual model. With low levels of pastoral development certain aspects of agricultural development may benefit stream invertebrates. Above a critical level, however, they will begin to have negative effects, i.e., increasing light and nutrients can increase primary production so there is more food for invertebrates

that feed on periphyton, but light and nutrients can become a stressor when high enough to cause algal blooms and high water temperatures. Increased abundances and numbers of macroinvertebrate taxa, as well as an increase in QMCI score, were found with agricultural development of native tussock grasslands to exotic pasture in Otago, demonstrating that while there were differences from the pristine condition, the net effects of subsidies and stresses on macroinvertebrate communities were positive (Riley *et al.* 2003). In that study of cool, open, nutrient-limited streams, the degree of catchment development had not caused significant degradation (i.e., critical values were not exceeded) as might be expected with the more dramatic change in habitat from forested catchments to pasture.

Native fish communities

Land-use change has been implicated in the reduced viability and species diversity of indigenous freshwater fish, especially in the case of the extinction of the southern grayling, and apparent decline in whitebait abundance (McDowall 1990). Forest removal, intensive agriculture, wetland drainage, damming and water abstraction, and effluent discharges have all been identified as factors that affect native fish fauna (McDowall 2000).

The effects of land use on fish populations can differ according to species. Hicks and McCaughan (1997) found that eels were abundant in pastoral streams, particularly short-finned eels, but that other species such as banded kokopu were present only in forested sites. Similarly, Hancher (1990) surveyed 55 sites in the Waikato region and found that the diversity of native fish species declined in pasture sites compared to native forest, possibly due to a reduction in suitable habitats (e.g., woody debris) and increased siltation.

Half of the native fish species in New Zealand are diadromous, requiring a life cycle stage spent at sea, and hence altitude and distance inland were the two most significant environmental variables explaining the distribution of fish communities (Jowett and Richardson 2003). Barriers to fish migration can therefore have an overriding influence on fish species assemblages at a particular site. However, in their study of the relationship between environmental variables and fish distributions, Jowett and Richardson (2003) found that percentage of native forest or farming land use in the catchment can also be highly correlated with fish community assignments.

CASE STUDIES – UPLANDS AND LOWLANDS

The effects of pastoral development on streams can differ between hill-country and lowland catchments. The case studies below compare predominantly sheep and beef hill-country catchments (Whatawhata) with a lowland dairying catchment (Toenepi).

Hill country catchments – Whatawhata

The Whatawhata research station, Waikato, has been the site of extensive research into the effects of land use on streams (Fig. 34.5A, B) (Quinn and Cooper 1997). This steep to rolling hill-country farm is stocked with sheep and cattle at 13 stock units/ha and receives phosphorus fertiliser. Quinn and Stroud (2002) compared streams draining adjacent catchments and found that pasture streams had higher exports of sediment, nitrogen and dissolved organic carbon, higher stream temperatures and lower visual clarity than native forest streams (see also Fig. 34.2). Changes in water yield were not as marked between native forest and pasture as in some studies (7% increase

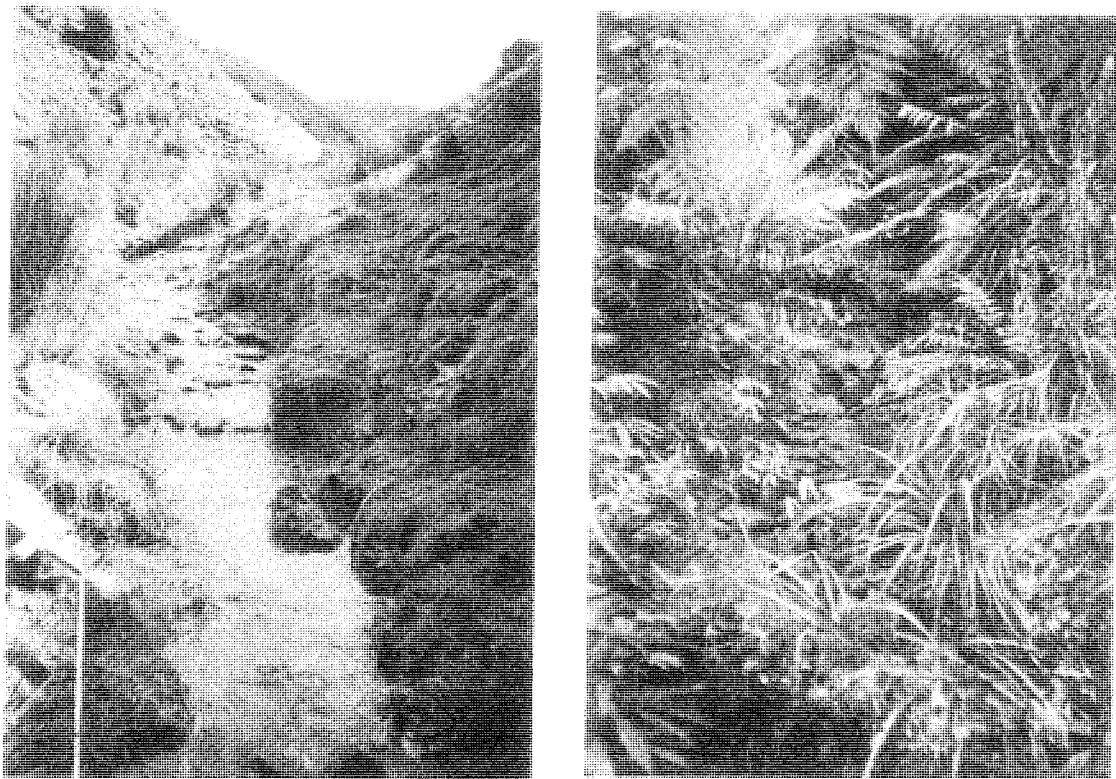


Figure 34.5 Upland pastoral (A) and native forest (B) streams at Whatawhata, Waikato.

Photos: Stephanie Parkyn

in runoff in pasture *cf.* 60% increase; Dons 1987), but specific yields of total nitrogen and phosphorus were high by New Zealand standards. Particulate forms of nitrogen and phosphorus dominated the total export yields in hill-country catchments, indicating the role of surface runoff and erosion in generating nutrients in steep pastoral sites. In contrast, particulate nutrients and sediment concentrations were much lower in a lowland pasture catchment (Toenepi). Here 63% of TP export comprised dissolved reactive phosphorus, predominantly from dairy-shed treatment systems, and nitrate leaching was the main process of nitrogen loss (Wilcock *et al.* 1999). Erosion from hillslopes also led to a much greater yield of suspended sediments from the hill country than from the flat to undulating topography of the Toenepi catchment (Table 34.4). The study by Quinn and Stroud (2002) showed that water quality in hill-country catchments can be strongly influenced by erosion, and hence management activities should focus on erosion control, such as tree planting to stabilise slips, pasture management to maintain grass cover, and riparian zone management to enhance filtration of sediment in overland flow and reduce stream bank erosion.

Lowland dairy catchment – Toenepi Stream

The problems affecting artificially drained, flat, lowland catchments can be quite different to those of hill-country catchments. The Toenepi catchment is located in an area of intensive dairy farming and has a stocking rate of 20 stock units/ha (Wilcock *et al.* 1999). Low-gradient streams like Toenepi generally have low erosion losses, so that most inputs of sediments and associated pollutants, such as

particulate phosphorus, originate close to the stream margins or within the channel itself. Thus, the sediment and associated particulate nutrient yield of Toenepi Stream is appreciably less than for the Whatawhata streams (Table 34.4). Dissolved nutrients, however, such as nitrate and dissolved reactive phosphorus, are intercepted by surface and subsurface drainage networks in lowland pasture catchments, with the result that yields of total phosphorus can be just as high as those from hilly catchments. Total nitrogen yields are much higher for Toenepi than for Whatawhata because of the higher stocking rate and the efficiency of drainage systems that collect soil leachate over much of the catchment area and discharge it into streams.

Many nutrient-rich lowland streams in sunny, open locations like Toenepi stream often have dense stands of macrophytes in summer (Fig. 34.6). Plant biomass can affect the hydraulic, physical and chemical properties of streams, and affect nutrient uptake and retention within channels (Wilcock *et al.* 1999; 2002). Photosynthesis and respiration create large diurnal extremes in dissolved oxygen and pH, with possible adverse consequences for stream life (Wilcock *et al.* 1995; Wilcock and Nagels 2001).

The amount of habitat available for biota in lowland streams can also be affected by the accumulation of sediments on streambeds. Collier *et al.* (1999) have shown that for the submerged macrophyte *Egeria densa*, the abundance of total phytophilous invertebrates increased with increasing plant biomass up to a maximum of around 400 g dw/m², but that invertebrate abundance declined as plant biomass increased further. They conclude that

Table 34.4 Comparison of water quality variables and total yields of nutrients, sediment and water for upland pasture streams and a lowland pasture stream. Measured values in native forest or guideline values are in brackets. Data from Wilcock *et al.* (1999) and Quinn and Stroud (2002).

Attribute	Upland Pasture – Whatawhata	Lowland Pasture – Toenepi
Dissolved oxygen (%)	c. 100%	< 10% in summer
Nitrate (NO ₃ -N mg/L)	Up to 2 (native forest 0.2)	Up to 6
Dissolved reactive phosphorus (DRP mg/L)	Up to 0.05 (guideline < 0.03)	Up to 0.6
Faecal coliform indicator (<i>E. coli</i> /100mL)	Median 635 (guideline < 126)	420
Temperature (°C)	Up to 25°C (native forest 19°C)	Up to 25°C
Yield		
Total nitrogen (TN kg/ha/y)	10-23 (native forest 2)	35
Total phosphorus (TP kg/ha/y)	1.5-3.2 (native forest 0.6)	1.2
Suspended solids (SS kg/ha/y)	1000-3200 (native forest 320)	142
Water (L/km ² /s)	30	20
Low flow (L/km ² /s)	5	1



Figure 34.6 Toenepi stream, Waikato, a lowland pastoral stream, is choked with the emergent macrophyte water pepper, *Persicaria hydropiper*. Photo: John Nagels

intermediate macrophyte biomass levels are likely to enhance macroinvertebrate biodiversity in sandy-bottom lowland streams.

Toenepi Stream receives several dairy pond discharges that generate occasionally high levels of ammonia nitrogen (1–2 mg/L). Most dissolved ammonia is in the form of harmless, ionised ammonium (NH_4^+) but as temperature and pH increase, so too does the proportion of toxic, un-ionised ammonia (NH_3), for which the ANZECC (2000) guideline value for the protection of 95% of freshwater species is 0.9 mg/L.

Low oxygenation, high temperature, and ammonia toxicity in lowland streams can lead to severely degraded fish and invertebrate communities, where abundance and taxa richness are reduced to only the most pollution-tolerant species. The faster flows of hill-country streams can lead to better oxygenation and possibly lower acute effects of ammonia pollution from animal wastes than in lowland streams.

HOW DO WE FARM SUSTAINABLY?

Diffuse sources of contaminants are more difficult to prevent from entering streams than point sources such as effluent from dairy oxidation ponds. Because of the link between streams and their catchments, improved land management, together with riparian management, are most likely to achieve improvements in water quality and stream habitat. Land management techniques can include avoiding overstocking and pugging of soils, retiring steep and erosion-prone land, protecting wetlands that are sites of denitrification (a process by which organic nitrogen is converted to inorganic nitrogen gas), diverting road and track runoff, which can be a concentrated source of effluent and sediments, and avoiding fertiliser application directly

to streams or when the water table is high or heavy rain is likely.

Riparian management can be viewed as a last line of defence for filtering out contaminants before they enter streams (Fig. 34.7). Fencing stock out of streams and retiring riparian margins from agricultural land use are particularly important for improving stream water quality. Buffer zones can filter contaminants and sediments from overland flow by increasing the infiltration into the soil, intercepting particulates, and removing nutrients by plant uptake. Riparian soils may also be sites of denitrification for subsurface water flow. Plantings, especially through the provision of shade, may restore the ecological function of streams. However, shade can result in the widening of stream channels, with subsequent movement of sediment downstream, and also reduce nutrient attenuation within a given stream reach as instream plants are shaded out. This may become a problem when riparian management is implemented in a piecemeal fashion and where there are sensitive lakes or estuaries downstream. The linkages within the whole catchment must be considered when designing riparian management schemes and best management practice in most cases would be to begin planting from the headwaters and continue downstream.

Riparian management is widely accepted by scientists and regulatory authorities as an effective means to improve water quality and stream habitat in rural areas. More

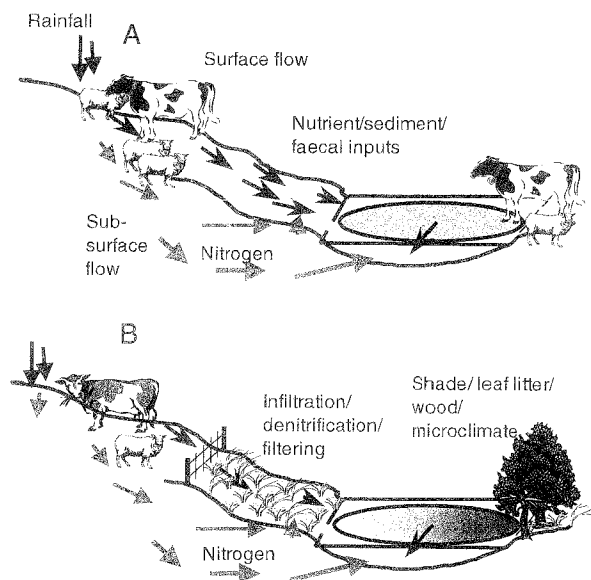


Figure 34.7 (A) Inputs of direct and diffuse sources of contaminants to pasture streams; (B) riparian management through fencing allows infiltration, denitrification and filtering of contaminants from flows (except for deep sub-surface flow), and planting provides additional ecological benefits.

information can be found in the DoC-NIWA guidelines: Managing Riparian Zones (Collier *et al.* 1995) and in the guidelines for New Zealand rural waterways, MfE (2001). A number of regional councils have also produced guides to riparian management (e.g., Auckland, Wellington and Taranaki Regional Councils).

CURRENT TRENDS AND RESEARCH GAPS

Intensification of agriculture

The current trend for an increasing intensification of agriculture, particularly the change from sheep to dairy farming, has led to concerns about water quality. In Southland, where there has been a rapid increase in dairying in the region, Hamill and McBride (2003) compared trends between 1995 and 2001 in water quality variables from river sites with changes in livestock numbers. They found that increased dairy farming had been associated with increasing concentrations of dissolved reactive phosphorus. Worsening of dissolved oxygen and oxidised nitrogen levels occurred at both dairying and non-dairying pasture sites. Specific yields of sediment, nutrients and faecal indicator bacteria increase with increasing agricultural intensity (especially stocking density). For example, total nitrogen yields from extensively grazed pasture catchments are typically 4–14 kg/ha/yr, whereas the corresponding yield from an intensively farmed dairy catchment was 35 kg/ha/yr (Wilcock *et al.* 1999). Studies of invertebrate communities in gravel-bed streams in Southland showed that intensive grazing by cattle (c. 15 stock units/ha) was associated with greater changes in invertebrate communities than extensive grazing by sheep or cattle (<10 stock units/ha). Small streams were more at risk from the effects of riparian grazing than larger streams, due to the greater impact of reduced stream shade and higher sedimentation from bank trampling by stock (Quinn *et al.* 1992b).

Harding *et al.* (1999) investigated the effects of agricultural intensity on water quality and invertebrate communities in the Pomahaka River, Otago. They found that increases in turbidity down the length of the river were out of proportion with that observed in longitudinal studies of a reference forested river. They concluded that agricultural intensity, rather than the percentage of differing land uses, was a more useful indicator of impacts within the rivers and suggested that, in the Pomahaka catchment, there was a threshold stocking intensity of c. 6–8 stock units/ha. Above this intensity, significant reductions in water quality were observed, with high nutrient and sediment levels, and changes to the macro-invertebrate assemblages were observed in the lower river.

Long-term monitoring and analysis of trends in water quality and waterway health need to be conducted and

related to catchment land use at a regional level, so that the effects of intensification or changes to predominant land uses can be assessed.

Research gaps

In a recent review, de Klein *et al.* (2003) identified the major environmental problems from deer farming to be the effects on soil and water, exacerbated by the responses of deer to confinement (in particular fence pacing and wallowing in streams). Fence pacing causes compaction and erosion losses that can lead to increased sedimentation in streams, whereas wallowing degrades the water and habitat quality of rural streams and can cause notable increases in stream faecal bacteria (*E. coli*) concentrations. Little research has been conducted on the effects of deer farming and, with the four-fold increase in deer farming since 1994, research into the effects of deer farming and the mitigation of problems is warranted.

Other areas of the stream drainage networks that have received little attention have been temporary and ephemeral waterways and wetlands. These habitats, which are wet for only part of the year, may be important treatment sites for pasture runoff before the water reaches the stream. They may thus be areas worthy of particular attention and restoration.

THE FUTURE: WHAT DO WE NEED TO KNOW?

Climate change

Kenny (2001) has summarised the effects of climate change for New Zealand, saying the greatest impacts will be as a result of climate variability and climate extremes. Eastern regions may experience more frequent and more severe droughts through increased temperatures and lower rainfall. This will increase demand for irrigation and, in turn, put pressure on water resources and affect the quality of water from runoff (surface waters) and leachate (groundwater). Western regions may be more prone to flooding and erosion from high rainfalls. The benefits gained from increased arable cropping due to warmer conditions and higher carbon dioxide levels in the atmosphere may be offset by higher fertiliser inputs. The availability of water for irrigation will be an important factor in achieving any potential gains, particularly in Canterbury, where there will be an increased risk of drought. Research is needed to refine our estimates of likely outcomes for rural waterways in response to changes in land use and should be linked with research on changes in water resources in relation to climate change.

Behavioural change in farmers

An important key to attaining better rural water quality is the rate at which farmers adopt sound environmental

practices. Regional councils have implemented incentive schemes that include subsidies for stream fences and free riparian plants, as well as offering a range of advice on riparian management. Rhodes *et al.* (2002) noted that farmers who were aware that funding was available were more likely to state their intention to implement riparian management within the next year, independent of their level of knowledge or exposure to information about riparian management. Financial factors were the most influential barrier to the adoption of permanent fencing. The research did show a positive correlation between the receipt of information and greater knowledge levels about riparian management, as well as the adoption of a wider range of riparian management measures.

There is significant evidence that agriculture affects the water quality and ecology of streams and the downstream environments. Education and acceptance by farmers that environmentally sound practices are an integral part of farming, are vital to achieve sustainable use of our land and water resources.

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Chapter 33

Impacts of forestry

Barry Fahey, Maurice Duncan and John Quinn

INTRODUCTION

Most of New Zealand was forested before humans arrived. Today native forest covers 23% of the country and plantation forests 6% (1.7 M ha), with most of the remainder in pasture. Radiata pine (*Pinus radiata* Dons) makes up 90% of the total plantation estate, and is planted mainly in areas below 800 m elevation, and where rainfall exceeds 600 mm. Douglas fir (*Pseudotsuga menziesii*) occupies about 86,000 ha, mostly at higher altitudes in the central and southern South Island. In the 1990s afforestation rates exceeded 60,000 ha per year, but these rates have declined and are now probably only half this figure. About 90% of the new plantings are on land previously in pasture (Ministry of Agriculture and Forestry 2000) after conversion from its natural vegetation (mostly native forest or scrub). Plantation forestry has made a substantial contribution to New Zealand's economic growth over the years. Plantings on land previously in pasture have also provided protection from erosion and mass movement. However, converting pasture to forestry can alter streamflow characteristics, stream habitat and water quality, which in turn can have major implications for stream biota and water users downstream.

The hydrological and ecological changes associated with plantation forestry have received much attention from water resource managers, foresters, and hydrologists, particularly since the introduction of the Resource Management Act in 1992. This act consolidated the previously fragmented legislative framework covering resource use in New Zealand, and set out, as its primary purpose, to promote the sustainable management of natural and physical resources. It is thus directly concerned with the effects on the environment of changes in land use. One of the responsibilities of regional councils under the Act is the control of land use for maintaining water quality and quantity, and ecosystem health, in rivers and

streams. Councils have also been required to prepare regional policy statements that identify strategies designed to deal with issues arising from proposed changes in land use, such as afforestation.

This chapter discusses the changes likely to accompany forestry land use, with particular reference to annual water yields, storm flows, low flows, sediment yields, water quality, and stream habitats. It also discusses the various approaches adopted by some local government authorities to deal with the question of afforestation effects on water resources.

SOME FUNDAMENTAL CONCEPTS

From a hydrological perspective, the effects of a change in land use on streamflow are best explained within the context of the water balance, which can be expressed as:

$$R = P - E - \Delta S,$$

where R is runoff, P is rainfall, E is evaporation including transpiration, and ΔS is the change in water stored in the soil profile and as groundwater. Evaporation from vegetation has two main components, interception and transpiration.

Because changes in soil moisture and groundwater storage are relatively small on an annual basis, changes in runoff associated with forest management are normally caused by changes in evaporation. Thus the annual water balance equation can be rewritten as:

$$R = P - (E_i + E_t + E_u)$$

where water is returned to the atmosphere through the evaporation of intercepted rainfall, E_i , (wet canopy evaporation), transpiration, E_t , (dry canopy evaporation), and transpiration from the understorey vegetation and evaporation from the soil, E_u . Understorey transpiration can contribute substantially to total water loss from a forest

(Kelliher *et al.* 1986). Where the forest canopy is wetted frequently, or remains wet for long periods, E_i may be 70% of the total evaporative loss for forests. Thus changes in runoff in areas with medium-to-high annual rainfalls are usually dominated by changes in interception loss following afforestation or harvesting. A recent summary of water use by native and plantation forests in New Zealand (Rowe *et al.* 2002) showed that manuka and kanuka scrub may intercept up to 42% of rainfall, beech-podocarp forests 30%, Douglas fir plantations 29%, and radiata pine plantations 23%.

In areas of low-to-moderate rainfall, transpiration losses from forests may exceed that from interception. For example, transpiration losses from young Douglas fir, old Douglas fir, and mature radiata pine stands on the Canterbury plains were 47%, 58% and 42% of gross rainfall respectively, compared with interception losses of between 20 and 29% (Fahey *et al.* 2001).

METHODS OF ASSESSING THE EFFECTS OF CHANGES IN LAND USE

Much of our understanding of the effects of a change in land cover on water resources has come from experimental catchment studies, in which the results of a controlled change or modification in vegetation cover in one or more catchments are compared with an adjacent control catchment (Fig. 33.1), or from direct comparison of nearby catchments with different vegetation cover. An analysis of time trends in a single large catchment can also provide useful insights into the effects of land use on streamflow, water quality and stream biota. However, these time-trend studies require a long-term research commitment to determine the effects of forestry over the whole cycle of plantation establishment, growth, timber harvest and replanting, whereas, for resource management, answers are needed quickly. An alternative approach involves sub-

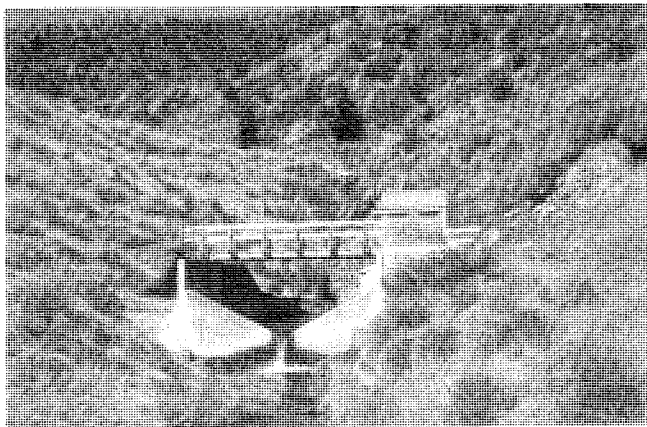


Figure 33.1 View of a weir for measuring streamflow from the forested catchment, Glendhu experimental catchment, upper Waipori River, upland east Otago.

Photo: Landcare Research

stituting space for time—comparing streams with different land use or in different phases of the forestry cycle that have similar catchment characteristics in other respects.

Studies at the catchment scale, coupled with investigations of soil-water-plant relationships have fostered the construction of hydrological models designed to predict the water balance of a catchment by partitioning rainfall (minus evaporation) into soil water storage and runoff. If these models can provide reliable estimates of measured runoff, they should be useful in predicting the effects of forestry management in catchments where data are sparse or absent. Some examples of water balance models developed in New Zealand to assist land and water managers in assessing the hydrological impacts of land cover changes are discussed later in this chapter.

The results of land use comparisons have also been combined with information on the habitat requirements of stream plants and animals to develop conceptual and predictive models of the effects of changes in land use and plantation forest management on the ecological aspects of streams.

EFFECTS OF AFFORESTATION ON WATER YIELD, FLOODS AND LOW FLOWS

Conversion of pasture to forestry

Reforestation of pasture at Mangatu Forest, north of Gisborne, reduced annual runoff by 30% (Pearce *et al.* 1987). At Berwick Forest in east Otago, Smith (1987) compared runoff from two catchments in *P. radiata* and two in pasture, ranging in size from 100 to 300 ha. The average runoff from the forested catchments was 43% lower than that from pasture. At Purukohukohu, in the pumice terrain of the central North Island, over the 23-year period between planting and harvesting, the average annual reduction in water yield was 220 mm, representing a 30% decline (Rowe 2003a). At the Waiwhiu catchment (806 ha) near Wellsford in Northland, the annual decrease in water yield, for the 23 years after 47% of the catchment originally in pasture and scrub had been converted to plantation forestry, was estimated at 300 mm, or 25% (Rowe 2003a). At Kakahu, in the South Canterbury foothills near Geraldine, one catchment (275 ha) planted in pines in the late 1970s was yielding 45% less runoff in 1998 (approximately 20 years into the rotation) than an adjacent control catchment (455 ha) left in pasture and scrub (Rowe 2003a). In the Moutere hills near Nelson, small (2.7–3.4 ha) pasture catchments were converted to plantation forestry and runoff compared to that from adjacent pasture catchments. Once the pine canopy was closed, water yields from the former pasture catchments were 167 mm/y (81%) less than would be expected from pasture (Duncan 1995) (Fig. 33.2).

Annual peak flows for the catchments in pines at

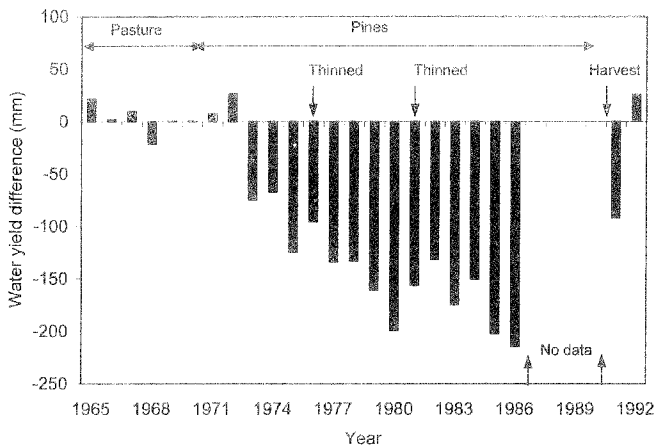


Figure 33.2 Differences in annual water yield between the control catchment in pasture (C5) and the catchment planted in pines (C14) in 1970, Moutere experimental catchments, Nelson, 1965–1992.

Berwick Forest were only a third of those for the catchments in pasture (Smith 1987). Before planting at Purukohukohu, flood peaks were much the same in both catchments, but the presence of a mature forest cover reduced small events by 50%, and larger ones by up to an order of magnitude, compared with the catchment left in pasture (Rowe 2003a). Peak discharges during freshes from the small Moutere gorse and pine catchments were only about 20 % of those from adjacent pasture catchments (Duncan 1980). For more extreme floods at Moutere, Duncan (1995) reports large differences between flood peaks from three pasture and three mature pine catchments: the mean annual floods from pine catchments averaged about 35% of those from pasture catchments and the 0.02 annual exceedance probability peaks averaged about 50% of those from pasture. Figure 33.3 shows differences in peak flows and flood volumes from pasture-

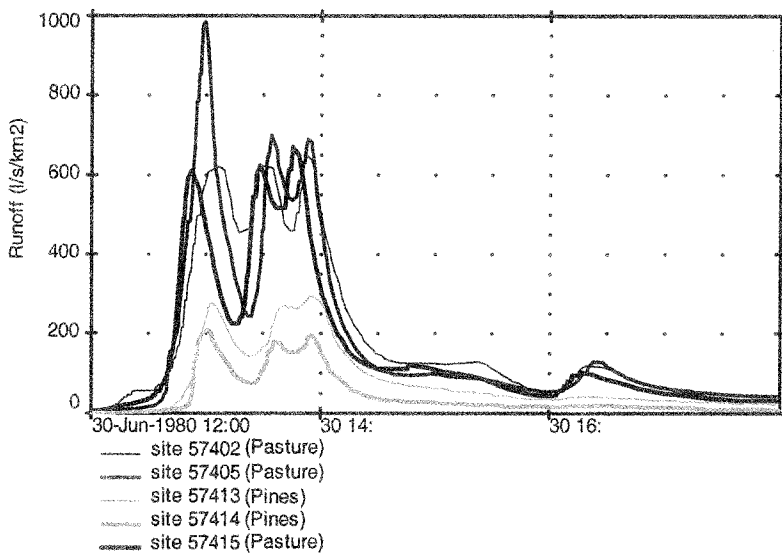


Figure 33.3 Hydrographs from pasture and pine catchments for the flood of 30 June 1980, Moutere catchment study, Nelson.

and pine-covered catchments. It is often assumed that, for large rainstorms, differences in soil moisture under different vegetative covers are small, and easily overwhelmed by heavy rainfall. Duncan (1995) was able to show that, in this instance, the differences in flood volumes from the two cover types were due primarily to soil moisture differences.

Tree canopies intercept rainfall that otherwise would have recharged the soil and groundwater stores, thereby reducing low flows. However, predicting the effects of a change in land use on low flow is difficult because it is very dependent on the amount and distribution of rainfall and other hydro-geological variables. Delayed flow from soil and groundwater stores between storms can form a substantial proportion of total flow, especially in the pumice country of the central North Island and the schist terrain of Otago.

Finding a common measure for low flows is also difficult, especially if, because of small catchment size, low rainfall or porous lithology, flow is intermittent during dry periods. For the small ephemeral Moutere catchments, Duncan (1995) found days without flow increased from 2 to 5 months when pines replaced pasture. When all catchments were flowing, pine or gorse catchments had less than half the flow expected from pasture catchments. For catchments with continuous flow, the lowest average daily flow for seven consecutive days in any given year (the minimum annual 7-day low flow) has been adopted as a useful indicator of the effect of a change in land use on low flows. When averaged over a number of years, this becomes the 7-day mean annual low flow. For Berwick Forest in south Otago, Smith (1987) estimated the reduction in low flows following conversion from pasture to pine forest to be about 20%.

Conversion of tussock grassland to forestry

The hydrological effects of converting tall tussock grassland to plantation forestry have been studied at Glendhu Forest in the upper Waipori basin since 1980 (Fahey and Watson 1991; Fahey and Jackson 1997a). After a 2-year calibration period, one catchment (310 ha) was planted over 67% of its area in 1982, and the other (216 ha) was left in tussock. It was not until 1989 that any effect on water yield was discernible in the record (Fig. 33.4). By 1991 the reduction in runoff from the planted catchment had reached 248 mm or 28%. Over the 10-year period 1993 to 2002, the reduction averaged 257 mm (30%). For a 100% forest cover a 40 to 45% reduction would thus be expected.

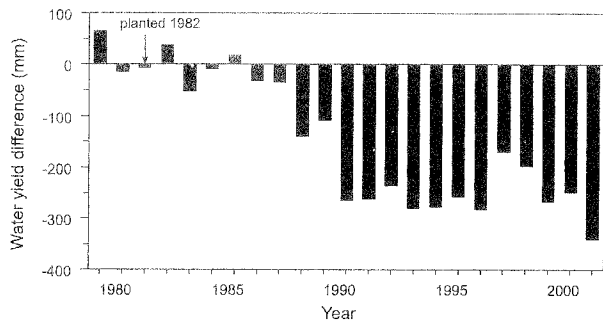


Figure 33.4 Differences in annual water yield between the control catchment in tussock and the catchment planted in pines in 1982, Glendhu experimental catchments, upland east Otago, 1980–2002.

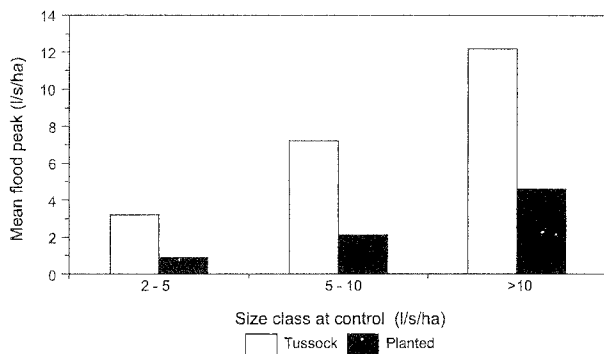


Figure 33.5 Comparisons of mean flood peaks for three size-classes of storms for the post-planting period, 2000–2002, for the tussock and planted catchments at Glendhu.

Mean peak flows for storms in different size classes at Glendhu were similar for the two catchments before planting, but 18–20 years later those for the planted catchment had fallen by between 60 and 70% of those in the tussock catchment. The greatest reduction was associated with the smaller storms (Fig. 33.5).

Pines planted at the Glendhu experimental catchments in 1982 caused no discernible reduction in mean annual low flows until 1987. In the 10 years since canopy closure (1993–2002), the average reduction was 0.15 mm/d or 18%.

Conversion of scrub to forestry

Interception and water storage capacities for tall scrub and forests are similar. Thus, apart from an increase for the first few years after scrub is removed, water yields should not be that different once the forest reaches maturity. For short scrub there may be a difference, however. One of the three Moumakai water-supply

catchments in the Hunua Range south of Auckland was planted in 1970 and the other was retained in bracken and low scrub as a control. The estimated average annual difference between measured and predicted flow (based on the control) for the planted catchment over the 23-year record was 270 mm, which converts to a 37% decrease as a consequence of afforestation (Rowe 2003a). In the Moutere hills near Nelson, two small (4.4 and 7.7 ha) gorse catchments were converted to plantation forestry, and runoff was compared to that from adjacent pasture catchments. Once the pine canopy was closed, water yields from the former gorse catchments were on average 67 mm/y (31%) less than was predicted (Duncan 1995).

At Moumoukai, average peak flows for storms of a range of sizes, 20 years into the rotation, were only half those for the same storms in the adjacent catchment left in scrub.

EFFECTS OF AFFORESTATION ON GROUNDWATER RECHARGE

The results of afforestation studies should also be applicable to areas where water drains to groundwater rather than to a stream. The Moutere valley, south of Motueka, is a case in point. Here horticulturalists are very dependent on groundwater for irrigation. About 14% of the area that serves as the recharge zone is planted in pines, and concerns have been expressed that further plantings will reduce the water supply even further. The Moutere experimental catchments nearby have shown that annual surface runoff can be lowered by as much as 50%, leading Duncan (1993) to conclude that, under a full pine cover, groundwater recharge could be reduced by as much as 70%. A reduction in recharge may also result from the reversion of abandoned pasture to scrubland.

EFFECTS OF STAND MANAGEMENT ON WATER YIELD

There is ample evidence to show that converting pasture or tussock grassland to plantation forestry reduces water yield, and that clearfelling increases water yield, at least in the short term. However, there is less certainty about the effects of thinning and understorey suppression on the availability of water from forests. Radiata pine plantations in New Zealand are normally thinned and pruned after 7 to 8 years. At Longmile, Rotorua, thinning provided an extra 200 mm of soil water, representing a 13% increase in available water (Whitehead and Kelliher 1991). However, thinning can also encourage the growth of a thick understorey, which may contribute up to half of the total forest evaporation in summer (Kelliher *et al.* 1986), but this in turn may be negated by interception if the slash is left in place (Kelliher *et al.* 1992).

EFFECTS OF AFFORESTATION ON WATER QUALITY AND STREAM HABITATS

Conversion of pasture land into pine plantations generally improves stream water quality and biodiversity by reducing contaminant inputs, such as sediment, nutrients, pathogens and agri-chemicals, and providing stream habitat conditions that are more similar to those in the native forests that covered most of New Zealand before humans arrived. Removal of grazing animals and planting trees reduces erosion of steep land, and this typically increases water clarity and reduces the siltation that can clog streambeds. Exceptions to this may occur where stream channels have narrowed during the pasture phase, following the original conversion from native forest to pasture. In the Waikato basin, the channels of many small streams have narrowed by 50% in pasture, due to high sediment inputs and grasses invading the stream channel (Davies-Colley 1997). In these areas, re-establishing shade during reforestation appears to lead to a period of increased bank erosion in small streams after the pasture grasses are shaded out, and the channels widen to the size of a forest stream (Quinn *et al.* 1997). This releases the sediment stored in the banks: it has been predicted to cancel out the benefit of less sediment coming off the hills over the 25–30 year period of the first pine crop rotation at a Waikato hill catchment (Collier *et al.* 2001). However, this is expected to occur only while the channels widen during the first rotation, and streams were predicted to have improved water clarity and export less sediment subsequently. Channel narrowing in pasture decreases as catchment area increases (Davies-Colley 1997) and it does not occur in all parts of New Zealand—e.g., it has not been observed in the Nelson area (Baillie and Davies 2002a).

Change from pasture to forestry also reduces fertiliser inputs and usually reduces nutrient losses from the land. For example, total nitrogen (TN) and total phosphorus (TP) yields from a pine catchment at Purukohukohu, in the central North Island, were 11 and 6%, respectively, of yields from an adjacent pasture site, ten years after the pine catchment was itself converted from pasture (Cooper and Thomsen 1988). Average (geometric mean) concentrations of total nitrogen and total phosphorus in a stream with a pine/regenerating scrub catchment at Whatawhata, Waikato, were 52% and 71% of stream catchments in pasture (Quinn and Stroud 2002). Total phosphorus was also lower (67%) in a stream draining late rotation pine than in a pasture stream at Pakuratahi, coastal Hawkes Bay, but the nitrate concentration was 39% higher (Quinn and Kemp 2001). Afforestation also influences the timing and forms of nitrogen and phosphorus loss, with lesser proportions exported from afforested areas than from pasture during storms, and in particulate, rather than dissolved, forms (Cooper and Thomsen 1988).

Faecal contamination of stream water with pathogenic (disease-causing) micro-organisms is also lower in forest than pasture streams, although the presence of feral animals (e.g., rats, deer, goats, possums) and birds results in some background contamination in forest (McBride *et al.* 2002). Median faecal coliform and enterococci concentrations in Pakuratahi Stream, draining a Hawkes Bay pine catchment, were 55 and 17% of corresponding values in an adjacent pasture stream, where the median enterococci level (57/100 ml) exceeded the guideline for contact recreation (Quinn and Kemp 2001).

Shade from streamside (riparian) trees in pine plantations generally improves stream habitat by controlling the blooms of algae and high water temperatures—conditions that are often associated with an absence of sensitive invertebrate species from many unshaded pasture streams (Davies-Colley and Quinn 1998; Quinn 2000). Leaf litter input to streams in pine plantations is also similar, in quantity and seasonal pattern, to that in native forest streams (Scarsbrook *et al.* 2001) and this helps restore natural forest food webs (Hicks 1997). Afforestation also provides inputs of large wood that provides a variety of habitat functions in natural streams (Baillie and Davies 2002b; Meleason *et al.* 2002; Collier *et al.* 2003). Because of these factors, instream plant biomass, and fish and stream invertebrate communities in late-rotation pine forest streams are similar to those in native forest (Hicks 1997; Rowe *et al.* 1999; Harding *et al.* 2000). However, the total instream biomass and productivity of invertebrates and fish is usually lower in small pine and native forest streams than in pasture, due to lower instream plant growth and cooler temperatures (Hicks 1997; Quinn *et al.* 1997).

In Britain and Europe, conifer afforestation has been associated with acidification of streams (reduced pH and alkalinity), in part due to forests scavenging acid rain from the atmosphere, but there is little evidence of this occurring in New Zealand. Slightly lower average pH (7.0) and alkalinity (14.3 g CaCO₃/m³) were observed in a stream draining a pine and regenerating scrub catchment than in a stream in adjacent native forest (7.3, and 21.1) and pasture (7.4 and 18.1) catchments at Whatawhata, Waikato (Quinn and Stroud 2002). At Pakuratahi, coastal Hawkes Bay, paired streams in pasture and pine had a very similar average pH (8.1–8.2) (Quinn and Kemp 2001).

EFFECTS OF FOREST HARVESTING ON WATER YIELD, FLOODS, SEDIMENT GENERATION AND NUTRIENT LEVELS

Water yield

Few studies in New Zealand have examined the increases in water yield likely to accompany the harvesting of exotic forest. However, research here and overseas

suggests that differences in water use by indigenous and exotic forests should be small. Therefore the results from two studies investigating the hydrological effects of harvesting indigenous forests, one at Maimai near Reefton (Rowe and Pearce 1994), and the other at Big Bush Forest (Donald Creek) in southwest Nelson (Fahey and Jackson 1997a) should be broadly applicable to plantation forests.

Harvesting the original beech-podocarp forest at one of the Maimai experimental catchments (M5) and replacing it with pines caused a maximum increase in water yield of 550 mm (76%) the following year, compared with the water yield from the adjacent control catchment (M6). This difference diminished markedly in the ensuing years as the pines approached canopy closure and, in the eighth year, annual water yields had fallen below those of catchment M6. At Donald Creek, harvesting two catchments (DC1 with skidders and DC4 with haulers) caused a marked and sustained increase in annual water yields compared with the control catchment (DC2) left in beech-podocarp forest (Fig. 33.6). From 1981 to 1984, the difference between DC1 and DC2 averaged 312 mm (61%). Pre-treatment levels were achieved by 1989. The pattern of change at DC4 was much the same. Annual water yields for the 4 years after harvesting averaged 344 mm (68% more than for DC2), and reached pre-treatment levels by 1988.

Based on an analysis of flow records from three small catchments in Glenbervie Forest, north of Whangarei, Rowe (2003a) estimated that streamflow increased by more than 600 mm or 75% in a wet year after pine harvesting, and by 30 to 40% in years with normal rainfall. Data in Rowe (2003a) also show a 290 mm (35%) increase in mean annual streamflow for the Waiwhiu Stream near Wellsford after harvesting. At Moutere, forest harvesting did not immediately increase water yield. However, in the second year after harvest, increases of 220–280 mm/y and 80 mm/y occurred compared to flows under original cover, on former gorse and pasture catchments. The lack of

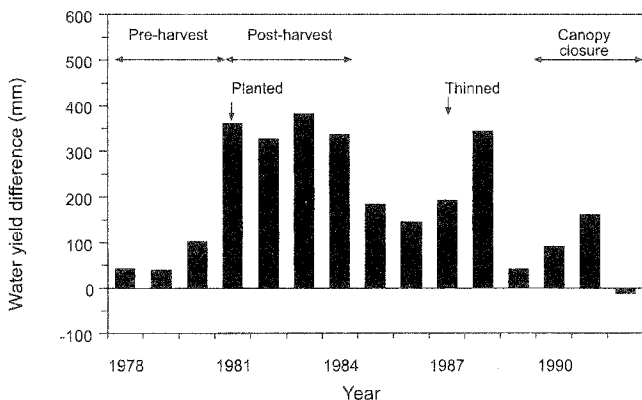


Figure 33.6 Differences in annual water yield between the control catchment in beech-podocarp forest (DC2) and the harvested catchment (DC1), Big Bush experimental catchments, southwest Nelson, 1978–1992.

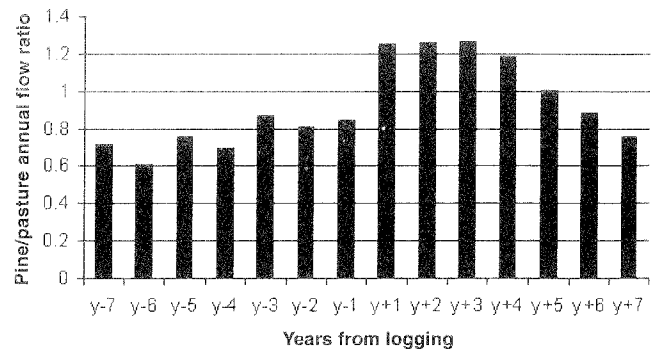


Figure 33.7 Effects of logging and replanting a small pine catchment at Purukohukohu, central North Island, on annual water yield relative to an adjacent pasture stream (from Quinn 2004 in press).

immediate response was attributed to the very low soil moisture levels, which took four months to be replenished. Once moisture levels were replenished, runoff rates increased to be more than those from pasture. In the first and second years after harvest, flows increased 0–60 mm/y and 226–343 mm/y respectively above those expected from pines, thus yielding in the second year flows similar to those expected from pasture. Before logging, water yield from pine forest in a pumice soil catchment at Purukohukohu, Central North Island, was 70–80% of that from an adjacent pasture site. It increased to 130% for the first 3 years after logging, and then declined to 70% in the seventh year after logging (Fig. 33.7).

Floods

At Maimai, mean peak flows from catchment M5 in the 3-year post-harvesting period rose by 60% for small storms and by 30% for larger ones, compared with the adjacent control catchment (M6) (Rowe 2003a).

In the Donald Creek study there was a marked increase in the mean flow peaks after harvesting at DC1 and DC4, compared with the mean for the same storms at the control (DC2). The increase was greatest for small storms (77% and 52% for DC1 and DC4 respectively).

Sediment generation

Forest harvesting operations can not only increase water yields but they can also generate more sediment. This is normally because of accelerated erosion from surface runoff and from mass movement on slopes that have lost their protective forest cover. Most sediment is generated either at the time of vegetation clearance and roading as the forest is being established (Fig. 33.8), or at the time of harvesting. The primary sources of sediment are access roads, contour tracks, log landings, cut-over areas, and unstable channel banks along stream courses. The amount of sediment produced on-site by surface erosion and mass movement can be much greater than that leaving the catchment as sediment yield, especially in large catchments.



Figure 33.8 Sediment mobilised during the land preparation phase of forestry in erodible granite terrain, west of Motueka. Photo: Landcare Research

Sediment production

Forest-wide rates of sediment production from roads in highly erodible granites underlying sections of Golden Downs and Motueka Forests in southwest Nelson were estimated at 37 t/km²/y (Fahey and Coker 1989), but at the time of land preparation and forest establishment, may have been 10 times higher (Table 33.1). This is still much less than the estimated background erosion rate of 500 t/km²/y.

In Queen Charlotte Forest in the eastern Marlborough Sounds, present production rates were estimated at 60 t/km²/y (Fahey and Coker 1993) and projected rates at the time of harvesting (after road up-grading and expansion) at between 100 and 200 t/km²/y (Table 33.1).

Forest roads, with their over-steepened cutbanks and loose unstable sidecast, are very prone to slope failure. Four storms in July and August 1991, including one with a 20-year recurrence interval, triggered a series of cutbank and sidecast failures on access roads in the granite terrain of Golden Downs and Motueka Forests (Coker and Fahey 1993). A total of 2800 t/km² was mobilised over the 2-month period, although only half this amount is thought to have entered local streams, with the remainder being stored mostly on the interfluvies below the sidecast. This is many times greater than that generated by one year of average surface runoff, but it must be kept in mind that mass movements of this magnitude are triggered by infrequent events, whereas surface erosion is continuous.

Sediment yields

At the Maimai experimental catchments near Reefton, sediment yield rates of 264 m³/km²/y for a skidder-logged catchment and 47 m³/km²/y for a hauler-logged catchment were measured (O'Loughlin *et al.* 1980). Assuming a bulk density of 1900 kg/m³, these values convert to 139 and 25 t/km²/y (Table 33.1). The skidder-logged catchment

Table 33.1 Sediment production and yields for different phases of the forest rotation based on plot data from Motueka, Golden Downs, and Queen Charlotte Forests, and catchment data from Maimai and coastal Hawke's Bay. The figures in brackets are suspended-sediment yields from adjacent pasture catchments.

Location	Forest rotation	Sediment yield (t/km ² /year)
Sediment Production	Land preparation	320
SW Nelson	Mid-rotation	37
(Motueka and Golden Downs Forest)	At harvesting	160
	Background rates	500
Marlborough Sounds	Mid-rotation	62
(Queen Charlotte Forest)	At harvesting	218
	Background rates	300–600
Sediment Yields	At harvesting	
West Coast (Maimai)	(skidder)	139 (17)
	(hauler)	25 (17)
East Coast North Island (Napier)	Mature forest (1995–1997)	30 (67)
	At harvesting (1998–1999)	89 (41)
	Post-harvesting (2000–2001)	150 (102)

yielded 8 times more sediment than an adjacent control catchment. The main sediment source was a mid-to-upper slope access track that virtually encircled the small catchment, and 60% of the sediment measured came from one storm.

A comparison of suspended sediment yields from a pasture catchment and one in exotic forest in erodible hill country north of Napier showed that before harvesting, yields from the latter were less than half those from the former (Table 33.1)(Fahey and Marden 2000). After harvesting, the situation was reversed, with annual sediment yields from the logged catchment averaging twice that of the catchment retained in pasture. In the post-harvesting period, suspended sediment yields began to approach those for the pasture catchment within two years, and after three years were substantially less (Fahey *et al.* 2003). A comparison of the 7 years of sediment yield data for both catchments suggested that suspended sediment yields from catchments in plantation forestry over a full rotation of 25 to 30 years may be less than from those catchments retained in pasture.

Nutrient yields

Logging can increase nitrogen and phosphorus losses to streams due to disruption of the forest nutrient cycling, and increases in particulate matter in surface runoff and in overall water yield. For example, total nitrogen yields for a

harvested beech forest catchment at Big Bush in southwest Nelson increased 10 times over the control and were still 3–5 times higher 4 years after logging. The total phosphorus yields went up 2–3 times (Fahey and Jackson 1997b). However, five recent studies of the effects of pine logging and replanting indicate that increases in nutrients after logging can be minor and of short-duration, and in some cases nutrient concentrations decreased after logging.

At Purukohukohu, export of nitrogen and phosphorus increased relative to a nearby native forest site in the first year after logging and replanting of a pine plantation on pumice soils in the Central North Island. (Fig. 33.9). However, nutrient yields declined to below the pre-logging levels in years 2 to 4 after logging. This pattern appeared to be due to a rapid development of groundcover by weeds and soil microbial biomass after logging that helped to retain the nutrients on site (Parfitt *et al.* 2002). Logging of three Rangataiki River tributaries, in the Central North Island, produced increases in concentrations of dissolved reactive phosphorus (up to 370%) and decreases in nitrate in the first 3 months, but these returned to pre-logging levels within 6 months (Collier and Bowman 2003). Similarly, a comparison of nutrient concentrations before, and in the two years after, pine logging and replanting at three sites at Pakuratahi, coastal Hawkes Bay, and at Whatawhata, Waikato, did not show any statistically significant changes in concentrations of dissolved reactive or total phosphorus, ammonium, or total Kjeldahl nitrogen (Quinn and Kemp 2001, and Authors unpublished data, respectively). Nitrate concentrations increased at the Whatawhata site (27% increase in geometric mean), but did not change significantly after logging at Pakuratahi. Furthermore, comparison of stream nutrient levels in Coromandel Peninsula streams over a period of increasing catchment harvest/replanting did not show increases in nitrogen or phosphorus (Quinn and Kemp 2001). In fact, instream concentrations tended to decrease with increases in the percentage of the catchments that had been logged and replanted in pines.

Logging can also increase export of dissolved organic carbon (DOC), due mainly to leaching from felled plant matter on land and within the stream channel. This can give the water a brown colour, affect its taste, and increase the microbial activity that contributes to reduced levels of dissolved oxygen. Dissolved organic carbon increased from background levels of c. 0.5–1.5 g/m³ to between 4 and 9.5 g/m³ immediately after the pine logging of three Rangataiki tributaries, but dropped to within background range within a year (Collier and Bowman 2003). This effect was greatest in the tributary with the highest amount of wood submerged in the channel, where the post-logging reduction in dissolved oxygen was also greatest. The effects of logging on dissolved organic carbon may be more persistent—in a study at Maimai, North Westland, mean concentrations were up to 100% higher in streams that had been logged and replanted in pines eight years previously (Moore 1989). Abundant instream organic debris was identified as the main source of dissolved organic carbon in this study.

EFFECTS OF HARVESTING ON STREAM HABITAT AND BIOTA

Forest harvesting can disturb the stream ecosystem through its effects on catchment runoff, sediment and nutrient supply, and instream habitat changes resulting from the input of logging debris and removal of riparian vegetation. The magnitude and duration of these disturbances varies with site characteristics (e.g., climate, slope, geology and soil type), stream characteristics (e.g., size, channel morphology, streambed type, riparian vegetation), and land management (e.g., the percentage of the catchment logged over a short-time period, and techniques used for managing roading, earthworks, tree felling, extraction, replanting, and riparian areas). Whether or not large rainfall events coincide with the time soon after logging, when groundcover is minimal and the risk

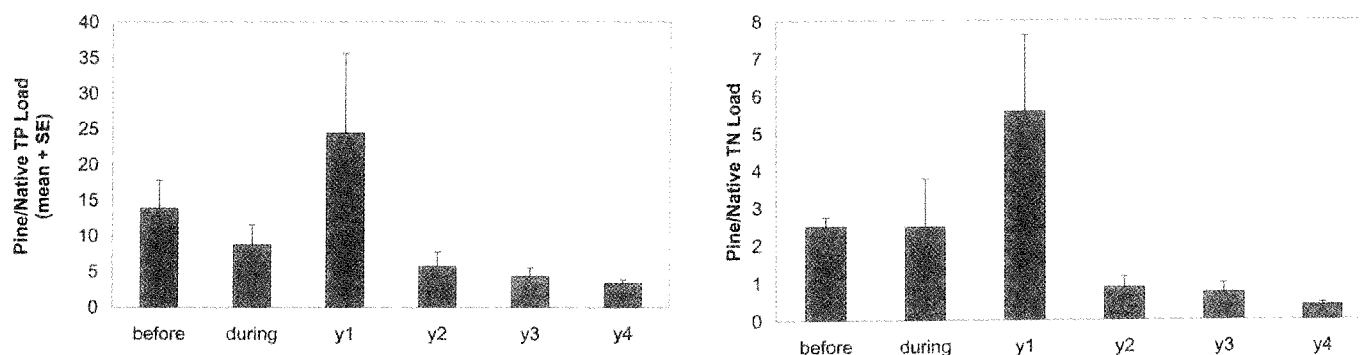


Figure 33.9 Effects of logging and replanting a pine catchment on total phosphorus and nitrogen export relative to a nearby native forest catchment at Purukohukohu, central North Island. A pulse in nutrient loads occurred in the first year after logging, but in years 2 to 4 after logging, nutrient levels dropped to below the level before logging (Quinn 2004).

of sediment mobilisation is greatest, also has a strong influence on streams and downstream aquatic ecosystems (Harding *et al.* 2000).

Figure 33.10 summarises the links between forest harvest activities, stream habitat, periphyton (attached algae) and invertebrates in Coromandel Peninsula streams and Figure 33.11 summarises how stream water quality, periphyton and invertebrates may vary after clear-cutting. Effects on water clarity generally last from the time of logging until groundcover is re-established, and are most severe when intense rain on recently cut-over forests causes landslides (Authors unpublished data). The effects of organic matter (“logging slash”) deposited in streams during logging on dissolved oxygen and food supply are also short-term in Coromandel streams, because high flows

flush this material downstream (see below). Studies in Coromandel, Central North Island, Bay of Plenty, Hawkes Bay, Nelson, and Otago, have shown that removal of riparian vegetation increases water temperature and periphyton, and typically changes invertebrate community composition to a fauna that is more tolerant and able to exploit the altered food resources (e.g., Collier *et al.* 1997; Harding *et al.* 2000; Thompson 2001; Death *et al.* 2003; Quinn *et al.* 2004). Long-term monitoring of Coromandel streams (J. Quinn unpublished data) indicates that these changes persist until regrowth of riparian vegetation restores shade sufficiently to control periphyton blooms (> c. 80% shade, Davies-Colley and Quinn 1998) and to maintain daily maximum temperatures below stressful levels (<c. 20–23°C, Quinn *et al.* 1994).

Harvesting can cause a large input of wood and leaf litter (“harvest slash”), particularly if trees are not pulled away from waterways and/or logs are hauled across streams without suspension. The larger logs and branches are often removed, either mechanically or by hand (“stream cleaning”), to avoid clogging the stream and damage to downstream structures from slash movement in storms. Although stream cleaning does help to address these problems, it often removes pre-harvest wood inputs and sets back the restoration of instream wood that can play a variety of roles in stream habitat formation in plantation and natural forest streams (Baillie and Davies 2002a,b; Meleason *et al.* 2002). This is particularly damaging in streams on pumice and in low-gradient streams that lack other sources of stable structure. Moreover, large amounts of fine organic litter are usually left behind, smothering the streambed, and as it decays, lowering dissolved oxygen concentrations to levels that exclude many sensitive species of invertebrates and fish. For example, one month after clearcut logging of pines and stream cleaning along a small Coromandel tributary (4-m-wide channel, < 0.2 m deep), average litter cover was 4 kg dry mass/m² (75% sticks and 25% leaf litter). This was 50-fold higher than in a downstream reach that was logged leaving a narrow riparian buffer (J. Quinn and A. Wright-Stow, unpublished data). Dissolved oxygen saturation in the water column in the unbuffered reach ranged from 83% (i.e., 8.1 g O₂/m³ at 16°C) in the main flow to 42% in a quiescent backwater on the channel margin. Saturations from 19–52% were measured within dense

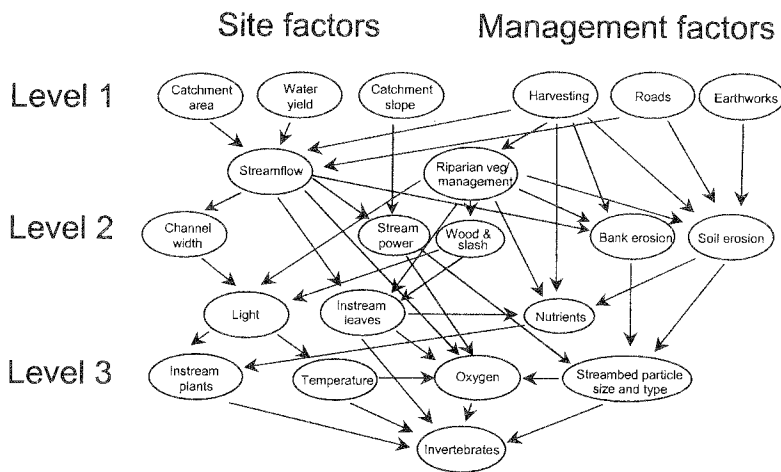


Figure 33.10 Summary of the causal pathways of influences of site factors and forest harvest management on stream water quality, habitat, periphyton and invertebrates.

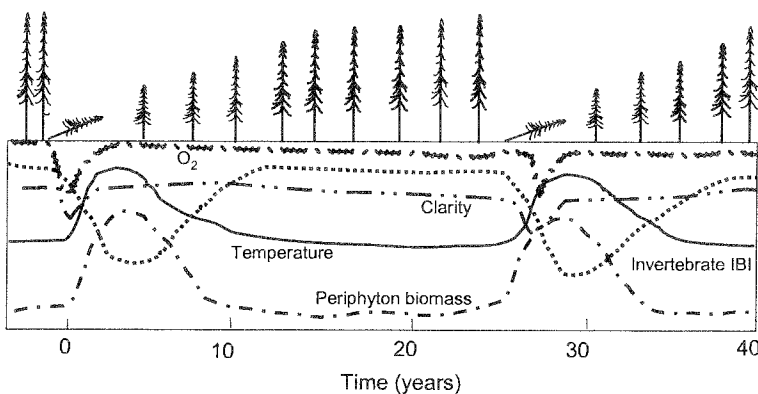


Figure 33.11 Schematic diagram of the magnitude and duration of impacts of pine plantation clearcutting and replanting on stream temperature, clarity, periphyton and invertebrate index of biotic integrity (IBI) values in second and third order Coromandel Peninsula streams.

packs of sticks, needles and leaves that filled the channel to a depth of 0.2 m in parts of the reach. However, high flows reduced the litter to 0.36 kg DM/m² at 2.5 months post-logging, when dissolved oxygen was c. 100% saturated throughout the reach. Whether streams are cleaned or not, large amounts of logging slash do not tend to persist for more than about six months in Coromandel streams that are more than about 1 m wide, due to the frequency of high flows that flush deposits downstream (J. Quinn, personal observation). However, the effects of harvest debris on stream dissolved oxygen may last for greater than two years in streams with stable flows in the pumice catchments of the Central North Island (Collier and Bowman 2003).

Role of riparian buffers

The vegetation on and near the streambanks has a strong influence on the stream habitat. Protecting this area and its vegetation from damage can buffer the stream from much of the effect of logging elsewhere in the catchment. Leaving a buffer of about 10–20 metres width of undisturbed riparian vegetation on either side of the stream does three key things:

- it maintains the forest shade and stabilises the streambanks,
- it reduces the disturbance in the area closest to the stream, where eroded sediment and tree branches are most likely to make it into the stream, and
- undisturbed litter layers within the buffer can filter out sediment carried by runoff from the clear-cut area.

New Zealand studies have shown that retention of forest riparian buffers can greatly reduce the impacts of pine logging on streambank erosion, stream light levels, periphyton, invertebrate communities and native fish (Harding *et al.* 2000; Rowe *et al.* 2002; Quinn *et al.* 2004). Buffers are most protective when they are continuous from headwaters downstream. “Patch” buffers, downstream of unbuffered reaches provide some local benefits (e.g., bank stability, shade and litter input, runoff filtering) but leaving headwater reaches unbuffered can result in high water temperatures and sediment input.

SCALING UP RESULTS TO LARGER CATCHMENTS

The catchment information presented in the preceding sections clearly demonstrates that afforestation reduces water yields and harvesting increases them. However, the question remains as to whether the results from small catchments can be scaled up to larger catchments. Some forest management techniques may, for example, result in changes in water yield that are much less than those quoted

for small-scale experiments. Afforesting 28% of the 900 km² Tarawera catchment in the central North Island caused a 13% reduction in mean annual water yield (Dons 1986), which is comparable, proportionally, to the 30% reduction from a 67% forest cover observed at the much smaller Glendhu experimental catchments. However, forest management practices in large catchments may mean that there will be mixed planting ages, and while the effects on annual water yield may be cumulative over time, they will certainly be less than the 40 to 80% reductions quoted for small experimental catchments.

Data from Maimai and Donald Creek show that gains in water yield after harvesting quickly diminish after replanting. In large forests, only small areas (< 100 ha) may be harvested at any one time, keeping total increases in water yield to a minimum. An increase in runoff of 30% from a small-scale experimental catchment may be lower for an entire forest. For example, harvesting 100 ha in a forest with a total area of 1000 ha may cause only a 2–3% increase in annual runoff, which may be the same as the error of measuring streamflow in the parent stream.

The ecological effects of forestry are also reduced when viewed at the scale of a large catchment or forest. Individual streams within a large forest will have catchments in a range of rotation phases, and the patches of disturbed habitat will move as different areas are logged and regrown. Larger downstream reaches tend to be disturbed over longer periods, but less severely, than headwater reaches.

REGIONAL AUTHORITIES AND THE EFFECTS OF AFFORESTATION ON WATER RESOURCES

The results of studies in New Zealand suggest that forestry in general, and afforestation in particular, could have a profound and lasting effect on annual water yields and low flows. Some regional and district councils have thus considered placing restrictions on forestry as a land use. In many catchments, available water resources may be fully allocated, and in some cases even over-allocated—this situation could be further exacerbated by afforestation. Another reason is a more pragmatic one, relating to the Resource Management Act (RMA). The Act requires Regional Councils to prepare regional plans for air, land, and water, to produce policy statements that identify the main resource management issues for the region, and to establish policies that deal with these issues. The important distinction between the RMA and the resource management legislation that preceded it is that the RMA can be implemented at the regional and district level rather than at the national level. Thus the regions can establish their own policies and regulations within the framework of the RMA. Individual councils have therefore dealt with

the question of the effects of afforestation on water yield in different ways. Three examples are given below.

Tasman District Council

Provision of both potable water supplies and irrigation water in Tasman District has been a concern for decades. It is exacerbated by the low water yield from the cemented Moutere gravels, which lie in a rain shadow. Because of their low fertility, the forest- and scrub-covered Moutere hills were converted to pine forest, which has continued to expand. The resultant declines in water yield have put pressure on surface water and aquifer sources from the Moutere gravels, both of which are fully allocated. Tasman District Council responded to this pressure by attempting to use the RMA and a provisional regional policy statement to limit the extent of new plantation cover to 50 % of each title. This attempt was vigorously opposed by the forest industry. There were appeals (held concurrently) by many factions to the Environment Court. It was at these appeals that the Court and the forest industry recognised that there was sufficient New Zealand scientific evidence that conversion of pasture and scrub to taller vegetation would reduce annual water yield and low flows. The decision from one appeal was that, in a relatively small area called the Moutere Groundwater Recharge Zone, no more than 20% of a title could be planted in new forest (>150 stems per hectare). The decision resulted in a forest company having to cut down a well established, but far from mature, plantation. For the other appeals, the Environment Court decided that the RMA could be used to control land use in order to control water use. Tasman District Council intend to prescribe that no more than 20% of a title may be planted in new forest in a defined Moutere gravel area.

Otago Regional Council

In 1997, the Otago Regional Council began preparing a regional water plan to “provide a framework for the integrated and sustainable management of Otago’s water resources”. Of most concern were the Kakanui, Shag, and Waikouaiti catchments, which are along the comparatively dry north-east coast of Otago between Oamaru and Dunedin. All three catchments had already experienced water shortages, and it was feared that additional afforestation would lead to further reductions in flows, thereby affecting the availability of water for both abstractive and instream uses. Controls on the establishment of exotic forestry in these “water-short” catchments were considered in the consultative draft of the Regional Plan: Water. Numerous submissions were received concerning the provisions proposed for controlling forestry. Some claimed that while afforestation may reduce water yields, the benefits arising from improved water quality and stream habitats were being ignored. Others argued that the Council should also be addressing

the problem of wasteful practices by existing abstractive users, and some stressed the need to identify and map the key runoff-producing areas in each catchment, so they could be targeted for management. After much consideration, the Council decided that it would not include policies designed to control forestry in its regional water plan. However, it did resolve to initiate monitoring that could lead to variation or changes in plan in the future.

Environment Canterbury

In October 2001, Environment Canterbury (ECan) released a discussion draft, as part of the Canterbury Natural Resources Regional Plan, entitled “Impacts of afforestation on water yield” (Environment Canterbury 2001). It identified a total of 118 “forestry-sensitive” catchments in the Canterbury region. The main criteria for selection were that rainfall is the primary source of flow (rather than melting snow or ice), and that each had a limited ability to sustain flows from soil and groundwater sources during summer dry periods. These catchments are found for the most part below 600 m elevation in the Canterbury foothills, the coastal hill country, the intermontane basins, and Banks Peninsula. Many, such as the Waipara catchment in north Canterbury, are locally important for irrigation and domestic water supplies, and available water is often fully allocated. The Discussion Draft accepted that not all areas in these catchments contribute equally to streamflow. Those areas generating the most runoff were referred to as “high-yielding areas”, and were identified by mapping the 7-day mean annual low flow. The sensitivity of each catchment was then ranked as high, medium, or low on the basis of the 7-day mean annual low flow allocated for abstraction. The report concluded that for catchments such as the Waipara, additional afforestation could mean less water being available for out-of-stream users and instream uses. The Council resolved that, where it was likely that afforestation could adversely affect low flows, regional rules would be imposed.

After accepting submissions, ECan prepared an amended document for discussion (Environment Canterbury 2002). In response to concerns expressed by forestry interests, the title was changed to “Impacts of vegetation change on water yield”, and most references to plantation forestry were altered to “forests” or “tall woody vegetation”. The potential benefits of forestry from the viewpoint of water quality and soil stability were also highlighted. ECan are considering ways to reduce the effect of new plantation forests in the most sensitive catchments so as to cause a <5% reduction in the predicted 7-day mean annual low flow, because standard hydrological measurements are unlikely to detect smaller changes. Options include requiring a land use consent for new areas

of plantings. An analysis of the effects of forestry planting shows that afforestation of more than 10–15% of each property is likely to cause a >5% reduction of the 7-day mean annual low flow. Replanting of existing forests, gorse and broom may be allowed as a permitted activity. It is likely that any rules will apply to all of each sensitive catchment, to provide security of supply of water for existing abstractions and to conserve instream values.

Some people consider the options precautionary, because there is little evidence to suggest that, on average, plantings of <20% of a catchment, are likely to have a detectable effect on water yield. While it may not affect large companies in Canterbury, who have scaled back their operations in recent years, it has implications for farm foresters and small companies (Bloomberg 2003).

THE ROLE OF HYDROLOGICAL MODELS

The lack of information on the hydrological effects of forestry over areas measured in hundreds of square kilometres has already been noted. It is simply not practical to operate experimental catchments at these scales. However, an improved understanding of soil-water-plant relationships in recent years, coupled with an appreciation of the storage and movement of water through the soil, has led to the development of a variety of hydrological models to simulate streamflow. These models can also be used to predict the changes in runoff likely to accompany a change in land cover. Two models have been developed in recent years for New Zealand conditions—both of which have the potential to assess the hydrological consequences of converting pasture to forestry.

The WATFIELD model

Landcare Research, and Tasman District Council completed a project “Land-cover effects on water availability”. It was funded primarily by the Ministry for the Environment’s Sustainable Management Fund, with additional contributions from other Councils and City Forests, Dunedin. The aims were to compile background information on water use by different vegetation covers in New Zealand, and to use this information to develop a Decision Support Tool to assist users and managers of water to predict the effects that a land cover change may have on the water balance. A total of six reports were produced, including annotated bibliographies on water use by Radiata pine (Rowe *et al.* 2001a) and Douglas fir (Rowe *et al.* 2001b), as well as hydrology related to land uses (Rowe *et al.* 2001c). The reports included summaries of the hydrological effects of different vegetation covers (Rowe *et al.* 2002) and catchments with data available for use in evaluating the effects of land cover on water resources (Rowe 2003a), plus a synthesis of these data (Rowe 2003b). The basis for the Decision Support Tool is

a water balance model developed by staff at Landcare Research. The model is intended for use where there is a limited amount of data on the climate, soil and vegetation of a given catchment, and is similar to the approach widely used for computing crop water requirements. A graphical user interface provides easy access to the model, and facilitates the input of data, which includes daily rainfall, monthly evapotranspiration based on published summaries, an estimate of interception for the cover type as a percentage of rainfall, an estimate for soil water parameters and crop factors, and two base-flow parameters. From these data and parameters, the model supplies the following outputs: soil water storage, daily, monthly, or annual water yields, and minimum annual 7-day low flows.

This model has been used to successfully predict the annual water yields and minimum 7-day low flows at the Glendhu catchments (3 km²) before and after planting, and likewise for Rocky Gully, a 23 km² catchment in the Hunter Hills near Fairlie in South Canterbury, and the Waipara River catchment above White Gorge (369 km²). The model, with its user guide and accompanying reports, can be downloaded from either the Landcare Research or Ministry for the Environment websites.

The TOPNET model

The National Institute of Water and Atmospheric Research (NIWA) has developed a distributed rainfall runoff model called TOPNET, designed for continuous simulation of catchment water balance and river flow. It can simulate the potential effects of changes in vegetation and climate or forecast floods. The model inputs are rainfall, and temperature time series and GIS maps of elevation, vegetation type, soil type and rainfall patterns. GIS routines divide the catchment into sub-basins that are linked by a branched river network. Flow is routed through the river network using kinematic waves. Each sub-basin is modelled using an adaptation of Topmodel (after Beven and Kirkby 1979) and has three lumped stores: a plant canopy store, a root-zone store and a saturated zone store (Fig. 33.12). Topmodel assumes that available soil water storage can vary within a sub-basin because of topography. The sub-basin model averages vegetation and soil characteristics within a sub-basin. TOPNET can model the effects of land-use change, through the changes in water balance, evaporation, intercepted water from the plant canopy, and, to a lesser extent, changes in the amount of soil water used by plants. When changing from shorter to taller vegetation, TOPNET allows for an increase in aerodynamic roughness, an increase in maximum canopy storage capacity, an increase in maximum root zone storage capacity and a decrease in albedo.

TOPNET is ideally suited to look at the hydrological

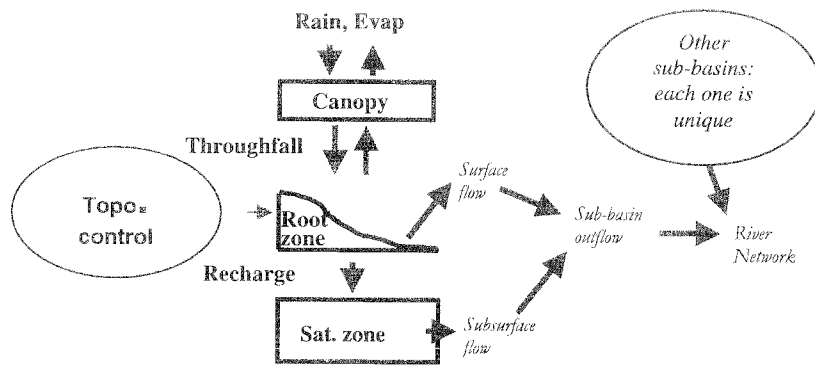


Figure 33.12 Overview of the Topnet model structure showing the canopy, root zone, saturated zone and river net-work components of the model. Arrows indicate flows of water from one model component to another.

effects of land-use change on large catchments, where patchy changes occur over long periods when climate may change. TOPNET modelled the 900 km² Tarawera catchment, which had all these ingredients. Almost 30% of the catchment changed from scrub to coniferous forests over a seventeen-year period beginning in 1964. Comparison of TOPNET results with those presented by Dons (1986) of the hydrological effects of these changes showed the model was able to accurately predict annual changes in flows (Woods and Duncan 1999).

TOPNET was also used to explore scenarios on the probable hydrological effects of changing pasture to pines in the 544 km² Shag catchment in east Otago. Calibration for 7-day low flows gave values similar to those using other methods for estimating low flows. One scenario representing a hypothetical planting in which exotic forests were expanded to cover almost 59% of the catchment, led to a predicted reduction of the mean annual flow and mean annual low flows by 49% and about 45% respectively (He and Woods 2001).

Bayesian belief network models

Bayesian Belief Networks (BBNs) are expert systems in which the cause-effect relationships between variables are defined by a causal network (e.g., Fig. 33.10), and by a set of conditional probabilities for each variable. BBNs have potential for use in forest management to protect aquatic ecosystems because they summarise the complex inter-relationships involved in forest-stream interactions and can predict the effects of various combinations of land management actions that occur in forest operations. They can also be used to help infer the causes of degraded environmental conditions. Each node in the network represents a particular variable and its probabilities of being in each of its possible states (normally 3 or 4 per variable), given the state of "parent" variables. When new information about the state of a variable (e.g., the percentage of the catchment that has been harvested) is

added, predictions on the states of all the other variables are updated. This mimics essential features of human reasoning, such as bi-directional (predictive and diagnostic) inference, dynamic updating of dependencies, and the ability to retract belief in a suspected cause if fresh evidence explains away earlier evidence. A BBN has been developed to relate the state of invertebrate community indicators to forestry practices and other factors, based on experience with Coromandel Peninsula streams and knowledge of their stream ecology (Jowett and Quinn 2001).

CONCLUSIONS

Afforesting close to 100% of small-to medium size catchments that were previously in pasture or tussock grassland may reduce annual water yields by up to 55% and low flows by at least 20%, but the full effects will not be seen until canopy closure 5–10 years after planting. Reversion of pasture to other forms of woody vegetation such as gorse, manuka or bracken will also reduce yields, but not to the same extent. In larger catchments the effects of planned afforestation on water yields and low flows are likely to be less pronounced, because plantings will be at different stages of development throughout the catchment. Excluding high water yielding areas such as riparian zones from planting, coupled with careful management practices, will also help keep reductions in water yield to a minimum. Plantation forests on land previously in pasture or tussock grassland can also reduce flood peaks by a half to a third. After harvesting, water yields may increase by as much as 70%, but if replanting is undertaken right away, yields should return to pre-harvest levels within 6–8 years. After harvest, flood peaks can be expected to increase by between 30 and 70%, with the biggest increases associated with the smaller storms. A brief but substantial increase in water yield can be expected after thinning.

Sediment production increases during forestry operations, especially in the land preparation phase. The main sediment sources are forest roads and log landings, where sediment is mobilized by both surface erosion and, less frequently, by mass movement. The latter has the greatest potential to affect stream ecosystems. Although infrequent, landslides can mobilize much more material in a single event than long periods of surface erosion. Nevertheless, the combined amounts generated by both processes are unlikely to influence the long-term natural erosion rates. Sediment yields from forested catchments at mid-rotation will be only half those from comparable catchments in pasture, but after harvesting may increase

3–4 times. However, when summed over the complete rotation, total sediment yields from pasture catchments are likely to exceed those in forestry.

Pine afforestation of pasture generally improves water quality, by reducing input to waterways of sediment, nutrients and pathogens, and restoring stream habitat towards conditions under native forest. Consequently, stream invertebrate and fish communities in pine plantations are much more similar to those in native forest than in pasture. Clearcut logging can disturb stream ecosystems markedly, through factors such as stream sedimentation, bank erosion, slash input, and shade removal. The duration and severity of effects vary with catchment and stream characteristics, logging practices and climatic factors such as the occurrence of storms soon after logging. In headwater streams, effects on water clarity and sedimentation, and on dissolved oxygen are usually short-term (1–2 years after logging), but marked increases in water temperature and periphyton abundance often persist until a high level of shade is re-established by regrowth of riparian vegetation (c. 7 years in a 3-m-wide stream). Riparian buffers reduce logging disturbance substantially by maintaining forest stream habitat conditions, providing a filtering area of undisturbed vegetation/litter/soil between the clearcut and the stream, and reducing disturbance in the area with the greatest influence on stream habitat.

Territorial authorities responsible for promoting the sustainable management of water resources under the RMA have approached the issue of afforestation effects on water resources in a variety of ways. The Otago Regional Council, for example currently places no restrictions on forestry, whereas Tasman District Council has a 20% limit for new plantings in prescribed areas. Environment Canterbury is presently considering implementing the consent process for new plantings that exceed 10–15% of a property in the most sensitive catchments.

Some of the questions raised about afforestation and harvesting effects on the water balance may be answered through the application of hydrological models such as TOPNET and WATYIELD. The former is particularly suited to assessing land-use change in large catchments.

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Chapter 32

Hyporheic zones

Greg Burrell and Mike Scarsbrook

INTRODUCTION

Hydrologists and geomorphologists have long recognised the importance of interactions between groundwater and surface waters, but ecologists have been relatively slow to grasp the consequences of these interactions (Hynes 1983). In many rivers and streams worldwide, surface water penetrates vertically and laterally down into the substrate, and often mixes there with underlying groundwaters. However, most of our knowledge of the ecology of flowing water systems is confined to surface waters and the benthic sediments visible to us. Surface water carries nutrients, dissolved oxygen, organic matter and other solutes deep into the stream bed, supporting an abundant subsurface microbial and invertebrate community, which in turn may chemically transform waters that return to the surface. This dynamic subsurface environment where surface water mixes with groundwater is known as the hyporheic zone (Orghidan 1959; White 1993). Despite the potential importance of the hyporheic zone for ecosystem processes and biodiversity, our basic understanding of hyporheic ecology lags behind that of surface environments.

The study of hyporheic zones began early last century (Karaman 1935), although detailed ecological studies have been undertaken only in the last 20–30 years (Valett *et al.* 1993), and in New Zealand, ecological studies of the hyporheic zone have been undertaken only in the last decade. Early research focused on describing the fauna, physical environment and chemistry, and defining the boundaries of the hyporheic zone, while more recent research has sought to integrate geomorphology, hydrology, chemistry, and biology into a functional view of the hyporheic zone as an integral component of river ecosystems (Valett *et al.* 1993).

This chapter provides a stepping stone into the hyporheic zone for stream ecologists, groundwater

specialists, resource managers and other students of hyporheic science. The chapter begins with a general discussion of the physical structure of the hyporheic zone and its place in the three-dimensional structure of stream ecosystems. The discussion then focuses on hyporheic ecology, including biodiversity, ecosystem processes and functions. The chapter concludes with an applied discussion of the potential impacts of human activities and management of hyporheic zones.

STRUCTURE OF HYPORHEIC ZONES

What is the hyporheic zone?

Literally translated, hyporheic means “under flow”, which refers to the flow of water beneath the ground surface. Underflow is a term commonly used by hydrologists to describe river flow that is lost to or gained from shallow groundwater, however it does not distinguish the hyporheic zone from groundwater. White (1993) provided a conceptual hydrological definition of the hyporheic zone as the saturated interstitial areas beneath the stream bed and into the stream banks that contain some proportion of channel water or that have been altered by channel water infiltration. White used this definition, as it separates the hyporheic zone from the strict definition of the groundwater zone as subsurface water below the water table that has not yet been influenced by channel process (Freeze and Cherry 1979). Triska *et al.* (1989) used injections of chloride to define the boundaries of the hyporheic zone of Little Man Lost Creek in California from a water chemistry perspective. They recognised a shallow hyporheic zone, comprising >98% surface water and chemically nearly identical to the surface. Beneath the shallow hyporheic zone they named an interactive hyporheic zone, characterised by gradients of dissolved nutrients and gases. This interactive layer they defined as having <98% but >10% channel water.

A biologically-based definition of the hyporheic zone highlights its designation as an ecotone (Gibert *et al.* 1990; Vervier *et al.* 1992), or transition area between distinct ecosystems (e.g., surface water and groundwater), where the ranges of the organisms in each bordering habitat overlap, and where there are organisms unique to the transition area. Surface water-groundwater ecotones are dynamic, as the composition of organisms and habitat vary with variations in sediment composition, hydrology, and individual species requirements. We use the “dynamic ecotone” definition (*sensu* Gibert *et al.* 1990) of the hyporheic zone in this chapter, as it is the most inclusive definition. Stream ecologists commonly accept it, and it highlights the interdependence of physical, chemical and biological factors in understanding hyporheic ecology. Furthermore, the dynamic ecotone definition recognises the functional aspect of the hyporheic zone as an integral component of river-groundwater systems.

Hyporheic flow

Flow through the hyporheic zone is three-dimensional, with vertical, lateral and longitudinal components (Fig. 32.1), varying across a range of spatial scales. At the scale of a valley segment or river reach (10 to 1000 m), the direction of vertical hydrologic exchange varies in response to the water table. High groundwater levels or bedrock-constrained reaches drive groundwater through the hyporheic zone towards the surface channel (zones of “influent” or “gaining” flow), whereas low groundwater levels in extensive alluvium may result in a net outflow of surface channel water into the hyporheic zone (“effluent” or “losing” reaches). At the scale of a riffle-pool sequence (1 to 100 m), water flows down at the head of a riffle and typically wells up at the foot of the riffle, giving a sequence of upwellings and downwellings. At the same scale, channel water travels through gravel or sand bars, parallel to the channel in the “parafluvial zone”, a lateral component of the hyporheic zone. This vertical, longitudinal and lateral exchange occurs as water responds to changes in pressure,

passing down through convex riffle heads or gravel bars, and emerging in pool depressions or eddies at the downstream end of the flowpath (Fig 32.2). At the sediment particle scale (<0.1 m), flow is forced over, under and around sediments.

The direction of hydrologic exchange is often measured at the reach or river segment scale to incorporate into surface and groundwater flow models, which in turn aid with the management of river flows (White *et al.* 2001). Reach-scale gains and losses are often measured using techniques such as multiple flow gaugings along the length of a river during low flow conditions. However, multiple gaugings typically will not detect within-reach variation of hydrological exchange and other methods need to be used, including injection of conservative tracers such as dye or chloride, or measurement of potential vertical hydrologic exchange using manometers (Boulton 1993). Fowler and Scarsbrook (2002) illustrated this point in three Hawkes Bay rivers by measuring positive (upwelling) and negative (downwelling) heads at the heads and tails of riffles within reaches that overall were gaining or losing flow to groundwater.

The rate of flow through the hyporheic zone is primarily influenced by surface flow and by sediment particle size and composition. Interstitial flow rates are lower in fine sediments and slow with depth or distance along a horizontal flowpath, as water is forced over and around sediment particles. For example, in Canterbury, the mean sediment permeability was 460 m/day at 0.3 m depth in the gravels of the Waipara River (Burrell 2001), whereas the mean bed permeability was in the range of 23–36 m/day in 6- to 15-m-deep wells in the Rakaia River (Scott and Thorpe 1986). Quinn *et al.* (1992) found that the deposition of clay-sized sediment from alluvial gold mining reduced bed permeability downstream of the mining operations in some West Coast rivers, illustrating the effect of sediment size on interstitial flow rates. Blocking of sediment interstices by fine sediment, a process known as colmation (Brunke 1999), may also occur naturally during sustained periods of low flow. Experiments in a laboratory flume have shown that during high flows, the fine sediment armour layer is broken up, moving fine sediment downstream and increasing hydraulic conductivity within the bed (Schälchli 1992).

The spatial extent of the hyporheic zone depends on the availability of permeable sediments and the magnitude of surface flow, and may range from being non-existent in bedrock river reaches to several kilometers wide in highly permeable alluvium. Boulton *et al.* (1998) predicted that hyporheic development can be expected to be least in headwater streams, reach a peak in intermediate reaches,

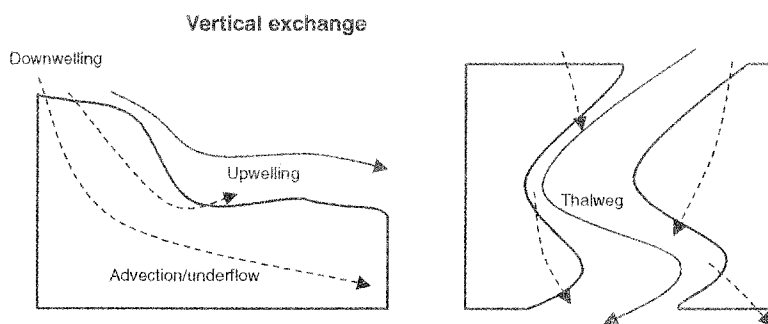


Figure 32.1 Diagrams illustrating vertical, lateral and longitudinal (downstream) hydrologic exchange between surface and subsurface environments. Solid and dashed lines show surface and subsurface flowpaths, respectively.

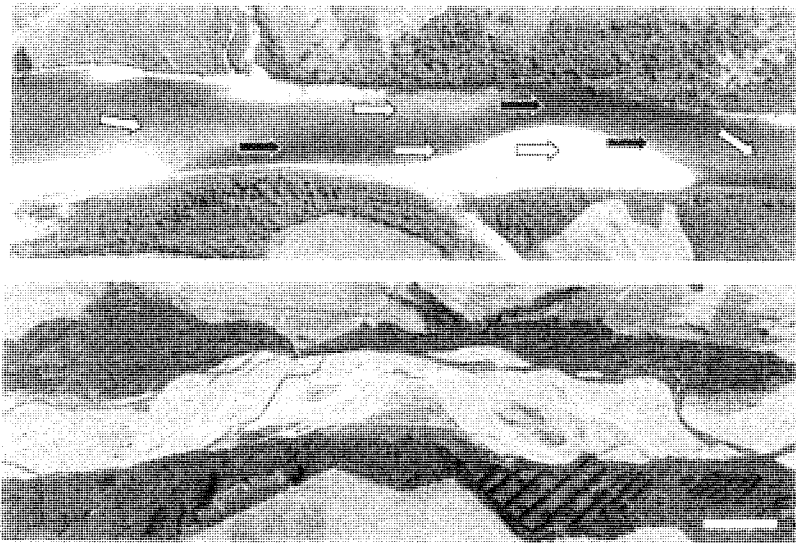


Figure 32.2 The Buller (upper image) and Ashburton South Branch (lower) rivers. Arrows on the Buller River indicate the likely location of downwellings (white fill), upwellings (black fill), and parafluvial flowpaths (grey fill) in relation to point bars. Mean widths of the Buller and Ashburton are 260 and 220 m, respectively. Surface flow dominates in the Buller (62% of mean channel width), whereas surface flow represents only 12% of the total channel width of the Ashburton. The potential for significant hyporheic flow is clearly much greater in the Ashburton, where parafluvial flow most likely predominates. Rivers are the same scale; scale bar is 200 m.

Photos: Map Toaster NZTopo software.

and then decline in lowland rivers, where lower hydraulic connectivity between surface waters and the bed may inhibit hyporheic activity. However, Stanford and Ward (1993) pointed out that alluvium occurs even in bedrock-dominated reaches and therefore the hyporheic zone may be more or less continuous throughout the length of a river, with only its spatial extent or size relative to surface flow varying. This is the basis of the hyporheic corridor concept (Stanford and Ward 1993), which views alluvial deposition and hyporheic zone development as occurring serially, like beads on a string, along the length of a river, with the relative importance of the hyporheic zone at a given location depending on the availability of permeable sediments and variations in water depth, velocity and discharge.

Tracer experiments and computer modelling can be used to quantify the relative proportion or importance of hyporheic flow compared to surface flow (Valett *et al.* 1997). Qualitative “first guess” estimates of the relative importance of hyporheic to surface flow can also be made using aerial photographs, coupled with some knowledge of typical substrate composition. For example, the ratio of the wetted channel width to the total dry channel width (from the edge of well-established vegetation), based on aerial photographs, is 0.62 for the Buller River and 0.12 for the Ashburton River South Branch (Fig. 32.2). Clearly,

the potential for an extensive hyporheic zone is proportionally far greater in the Ashburton than in the Buller, even though mean annual surface flow in the Buller is an order of magnitude greater than in the Ashburton.

HYPORHEIC ECOSYSTEMS

Flow is an important determinant of hyporheic ecosystem processes, and the functional significance of hyporheic processes to the ecosystem of the stream as a whole is related to both the degree of surface-subsurface hydrologic connection and the level of biogeochemical activity within the hyporheic zone (Boulton *et al.* 1998). This section looks at the relationship between physical and biological processes in the hyporheic zone, highlighting the interdependence of these factors in determining the structure of hyporheic ecosystems.

Carbon dynamics

Due to the lack of light beneath the sediment surface, carbon cannot be fixed via photosynthesis, therefore hyporheic biological communities are reliant on external (allochthonous) sources of carbon for energy.

The primary types of carbon entering the hyporheic zone are dissolved and particulate organic matter (DOM and POM), from either surface water or, to a lesser extent, from groundwater. Coarse organic matter is transported and deposited on floodplain sediments and in the hyporheic zone mostly during floods, with the concentration of sediment organic matter being positively related to the frequency of bed-moving flood events (Naegeli *et al.* 1995). Particulate organic matter concentrations typically decline with depth (Burrell 2001; Scarsbrook 1995), and a filtering effect may occur, with the proportion of coarser particles declining with depth (Naegeli *et al.* 1995). Finer particulate and dissolved organic matter may enter the hyporheic zone through downwelling surface water. Groundwater may be enriched by dissolved organic matter, so may also be a source of organic carbon to the hyporheic zone in upwelling areas (Fig. 32.3), depending on the quality of the dissolved matter (see microbiota section below).

As microbial and invertebrate communities break down particulate organic matter through aerobic respiration, interstitial oxygen is used up. Oxygen can therefore rapidly become a limiting factor if there is a sufficient supply of interstitial organic matter and low sediment permeability prevents the replenishment of hyporheic oxygen concentrations from oxygen-rich surface water. Thus, in

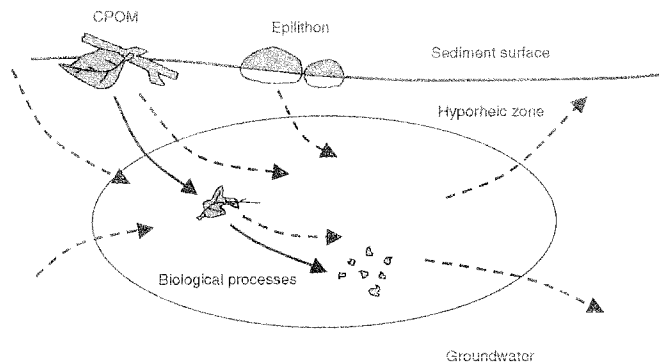


Figure 32.3 Transport of particulate (solid lines) and dissolved (dashed lines) organic matter into and out of the hyporheic zone, showing the role of biological processes in uptake and turnover of organic matter. CPOM is coarse particulate organic matter.

sandy-bottomed streams, oxygen is depleted near the sediment surface by respiration of interstitial organic matter, in turn reducing the rate of aerobic respiration and decay of buried leaf matter (Herbst 1980; Metzler and Smock 1990). Rutherford *et al.* (1995) presented an analytical model for evaluating the effect of interstitial respiration and dissolved oxygen (DO) uptake on surface dissolved oxygen concentrations in two New Zealand rivers affected by gross organic pollution (from pulp mill effluent and dairy factory effluent). The model predicted that in sediments with high organic inputs, porous sediments had a greater effect on surface water deoxygenation than finer sediments. This was attributed to two counteracting processes. Firstly, with reduced particle size, interstitial residence time increases, the rate of dissolved oxygen uptake per unit volume increases, and the drop in dissolved oxygen concentration increases. Secondly, as particle size decreases, interstitial velocity decreases, which reduces the flux of water out of the bed and the overall rate of sediment dissolved oxygen uptake. Therefore, while sediment porosity may enhance hyporheic processes when there is gross organic enrichment, hyporheic respiration may contribute significantly to deoxygenation of surface water, by stripping organic-rich water of oxygen as it passes through. Thus, although fine sediments may result in a rapid uptake of dissolved oxygen over a short distance, the lower overall flow through the hyporheic zone means less overall flux to surface water. The model predicted an optimal particle size of 5 mm, where the contribution of interstitial flow to channel water oxygen depletion was greatest, due to a compromise between interstitial residence time and total interstitial flow.

In coarse sediments, rates of heterotrophic microbial biofilm activity on wood, estimated using the uptake of ^{14}C -glucose, were greater on surface-placed wood than in the hyporheic zone (10 cm depth) in the uncompacted

sediments of Middle Bush, an alpine stream (Tank and Winterbourn 1996). However there was no difference in the microbial community activity between wood buried at 9–15 cm and 19–25 cm in several clear and brown-water West Coast rivers (Tank and Winterbourn 1995) or Middle Bush Stream (Tank and Winterbourn 1996). Oxygen concentrations measured at 25–60 cm depth in gravel-bed rivers in Otago (Olsen and Townsend 2003), Canterbury (Burrell 2001), and Hawke's Bay (Fowler and Scarsbrook 2002) have exceeded 7 mg/l, indicating high rates of interstitial flow, relatively short residence times, and/or low microbial respiration. However, even in relatively permeable stream sediments, reduced hydrological exchange may result in reduced biological breakdown of organic matter in the hyporheic zone compared to that at the sediment surface (Burrell 2001; Rounick and Winterbourn 1983).

Nutrient dynamics

Hyporheic nutrient dynamics are complex, as sediment-scale changes in porosity, organic matter content and oxygen concentration can change the predominant chemical reactions from oxidising to reducing. However, at larger scales, such as a riffle-pool sequence or river reach, trends in hyporheic nutrient concentrations and processing may be more obvious. Howard-Williams (1991) plotted isopleths of redox potentials at grid points in the stream bank of a Whangamata stream at two depths, surface and 2 cm, showing a mosaic of denitrification potential, and an increase in nitrate with depth. Thus, with greater depth, finer sediments and increasing hyporheic residence time, reduced oxygen concentrations result in a shift from nitrate uptake to denitrification, provided there is a sufficient source of organic matter.

There has been substantial New Zealand research undertaken looking at nitrogen cycling in the near-stream environment, particularly focussing on the ability of riparian zones to transform and take up nutrients from groundwater (Howard-Williams 1991). However, there has been no research looking at the role of the hyporheic zone, including lateral and vertical dimensions, in the uptake and turnover of nutrients in New Zealand streams, although current NIWA research in the Waikato, and planned research in the Selwyn catchment (Canterbury) should provide some information in this area.

There are very few published data comparing surface and hyporheic nutrient concentrations in New Zealand. Olsen and Townsend (2003) found higher concentrations of hyporheic (25 cm depth) dissolved oxygen, ammonium and phosphorous in the Kye Burn during winter than in summer, but found no consistent differences in nutrient concentrations between upwelling and downwelling zones. Dissolved oxygen was high (>9 mg/l) in both seasons and zones, indicating high interstitial flow rates and therefore

less potential for gradients in nutrient concentrations between upwelling and downwelling zones. Larned (2002) collected surface and hyporheic (30–50 cm deep) water from 11 minor braids of the gravel-bed Waitaki River during April 2002. Hyporheic water was typically enriched with nutrients, particularly ammonium and phosphorous (Fig. 32.4). The direction of hydrological exchange was not measured in this survey. However, Larned found greater concentrations of hyporheic nitrate and phosphorous in samples taken from the edge, but not underneath the middle of the channel. This was likely due to greater interstitial flow in the channel centre resulting in either greater dilution of nutrients or enhanced microbial activity, and subsequent nutrient uptake. Either way, groundwater nutrient concentrations were higher than in surface water, suggesting microbial uptake in the hyporheic zone before groundwater entered the flowing channel. Larned suggested that under the proposed Project Aqua hydroelectric development, which would involve a reduction in median flow of the Waitaki from 220 to 120 m³/s, there would be a reduced dilution of influent nutrient-rich groundwater. Biggs *et al.* (2003) further suggested that increased groundwater nutrient inputs into the residual Waitaki River could exacerbate proliferations of filamentous algae that may be expected to occur following artificially created low flows following hydroelectric development. Thus, the hyporheic zone has the potential to play a significant role in influencing surface water quality.

Transient storage

Transport through the hyporheic zone increases the length of time and distance that water must travel through a given river reach. The temporary, or transient, storage of water and solutes in the hyporheic zone has important implications for whole-stream processes of nutrient and carbon processing and turnover. Thus, the uptake rate of nitrate (Fellows *et al.* 2001; Valett *et al.* 1996) and phosphorous (Mulholland *et al.* 1997) has been shown to increase within stream reaches in proportion to streambed hydraulic conductivity and the size of the transient storage zone. Similarly, inclusion of the hyporheic zone increases estimates of whole-stream respiration (Grimm and Fisher 1984) and the rate of carbon turnover (Fellows *et al.* 2001). Estimates of hyporheic contributions to streambed respiration have been made for the Waipara River (Burrell 2001), where dissolved oxygen uptake of incubated sediments was used to measure community respiration. Community respiration was lower in an upwelling reach than in downwelling reaches and declined with depth, with the strongest decline occurring between the 0–15 cm and 15–30 cm depth strata (means = 0.65 and 0.40 mg O₂/litre sediment/hr, respectively). However, although hyporheic community respiration rates varied

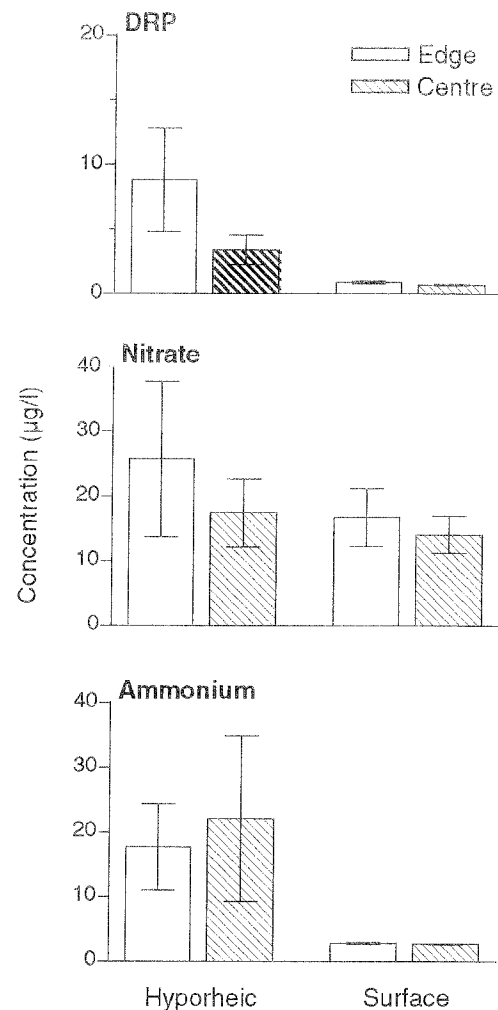


Figure 32.4 Mean (± 1 SE) nutrient concentrations of surface and hyporheic (30–50 cm depth) water taken from the edge and centre of 11 braids of the Waitaki River, South Canterbury, in April 2002. DRP = Dissolved Reactive Phosphorous. Data are from Larned (2002).

substantially among five reaches of the Waipara, the contribution of hyporheic (measured from 15–45 cm depth) to total (0–45 cm) community respiration was fairly consistent, ranging from 50 to 56%. Burrell also indicated that the hyporheic contribution to total respiration was likely to be much higher, given high interstitial dissolved oxygen concentrations (summer mean at 30 cm depth = 7.6 mg/l), and the fact that the extent of the hyporheic zone was likely to have greatly exceeded the maximum sampling depth.

Hyporheic microbiota

The most obvious difference between surface and hyporheic microbial communities is the lack of algae in the hyporheic zone. Some diatoms may be found in the absence of light, even in floodplain wells up to several

kilometers away from the active channel (Ellis *et al.* 1998), however they are never particularly abundant, as the absence of light limits their ability to exist in the hyporheic zone (Fig. 32.5). In contrast, heterotrophic microbes—fungi and bacteria—can be very abundant in the hyporheic zone, and they play fundamental roles in the cycling and uptake of nutrients and organic matter, and as a food source for hyporheic invertebrates.

As heterotrophs are dependent on fixed forms of carbon, they are often abundant where pools of particulate or dissolved organic matter occur in the hyporheic zone. This is particularly true for fungi, which are most frequently associated with particulate organic matter, and are often less common on stone surfaces. Recent research indicates that the relative contribution of fungi and bacteria to total microbial biomass depends on the composition of available particulate organic matter, with fungi dominating microbial biomass on leaves and wood, while bacteria are dominant on fine particulate organic matter, both at the sediment surface and in the hyporheic zone (Findlay *et al.* 2002). Thus, fungi were dominant components of the biofilm coating buried wood in several streams of the South Island's West Coast (Tank and Winterbourn 1995), but were absent from hyporheic stone surface biofilms in the Glentui River in Canterbury (Burrell and Ledger 2003, Fig. 32.5), where fine particulate or dissolved organic matter were the likely carbon sources. Hyporheic bacteria are also limited by the quality of dissolved organic matter, with low molecular weight or simple forms of organic matter being easier to decompose and assimilate than more complex or refractile forms (Marmonier *et al.* 1995).

Most subsurface microbes occur as part of a complex of micro-organisms within biofilms attached to particle surfaces (especially organic matter), rather than floating freely in particle voids (Alfrieder *et al.* 1997). Biofilm microbes excrete mucilage, which binds them to the particle surface and may also be a temporary source of nutrition. Fungi and bacteria excrete extracellular enzymes into the biofilm matrix, facilitating the breakdown of organic carbon into carbohydrates and amino acids that are more readily assimilated by the microbiota. Biofilms may be an important food source for hyporheic invertebrates (Bärlocher and Murdoch 1989). However, depending on its availability, buried particulate organic matter is likely to be a more significant carbon source to the fauna, and microbes most likely "condition" or decompose buried particulate organic matter, converting it into a more digestible condition for invertebrates, as is the case at the sediment surface.

Hyporheic microbial communities are greatly influenced by sediment particle size. Smaller particles, such as fine sand, provide a larger surface area for biofilm attachment per volume of sediment than coarser sediments, however, bacterial abundance in subsurface

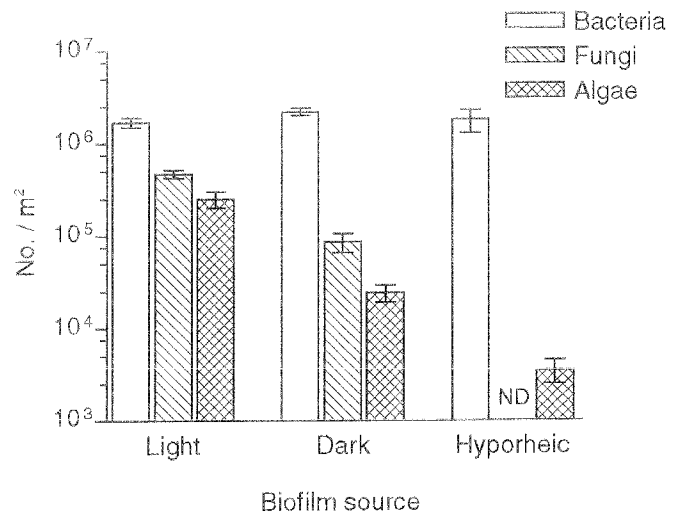


Figure 32.5 Mean (± 1 SE) abundance of bacteria, fungal hyphae, and algal units collected from biofilms grown on stones in the Glentui River under different treatments. ND = no hyphae detected. Reproduced from Burrell and Ledger (2003) with permission from the Journal of the North American Benthological Society.

biofilms is not as high as might be predicted by grain size alone (Alfrieder *et al.* 1997). This is because fine sediments also have a lower hydraulic conductivity and a greater residence time, meaning that the supply of nutrients, organic matter and oxygen may not be sufficiently high to sustain aerobic microbial communities or processes. Increased residence time, as well as the physical and biological filtering effect of streambed sediments, is also responsible for the typical decline in microbial abundance with depth or distance from the surface.

The influence of sediment porosity on microbial processes also depends on the direction of hydraulic exchange. Boulton and Quinn (2000) compared the effect on microbial cellulose decomposers of sedimentation in upwelling and downwelling zones of several tributaries of the Waipa catchment, North Island, by measuring the loss of tensile strength of strips of standard cotton cloth buried in sediments. They found that the deposition of silt reduced rates of cellulose decomposition in upwelling zones, but this relationship was not evident in downwelling zones. The reason for the association of sedimentation and direction of vertical hydrologic exchange was not clear, but may have been due to the greater influence and penetration of surface water and essential solutes (especially oxygen) counteracting the effects of low hydraulic conductivity in downwelling zones, but not upwelling zones, where oxygen-depleted water would reduce aerobic decomposition.

The handful of New Zealand studies of the hyporheic microbiota have included quantitative measures of microbial abundance, community respiration, and several

measures of microbial activity, including cellulolytic activity and uptake of ¹⁴C-labelled glucose by hyporheic biofilms (Tank and Winterbourn 1995; Boulton and Quinn 2000; Burrell 2001; Burrell and Ledger 2003). A great variety of other microbial methods exists for measuring the abundance and activity of aquatic microbes (see Boulton and Boon 1991 for review), and may be applied to the study of hyporheic microbial communities. In addition, hyporheic microbial diversity is an as yet unexplored component of ecosystem biodiversity, although various biochemical methods recently applied to microbial biodiversity in other aquatic systems are available (Boulton and Boon 1991).

Hyporheic invertebrate ecology

Hyporheic invertebrate communities are known as the hyporheos, and include taxa typically associated with surface sediments (epigean fauna) as well as true groundwater taxa (hypogean fauna), which may have a variety of specialisations for subsurface life (see Chapter 29). Gibert *et al.* (1994) provided a classification of subsurface fauna based on their affinity with groundwater and on their life history, distinguishing between members of the hyporheos that are permanent (such as copepods and mites) and occasional, including many insect taxa (Fig. 32.6). In New Zealand studies, a diversity of epigean and hypogean invertebrates have been found in the hyporheic zone (Collier and Scarsbrook 2000), many of them common (Table 32.1), and recent research is beginning to reveal their ecological requirements.

Ecotones, such as the hyporheic zone, are typically areas of high biodiversity, and this has certainly been illustrated in other ecotonal areas such as riparian zones and wetlands.

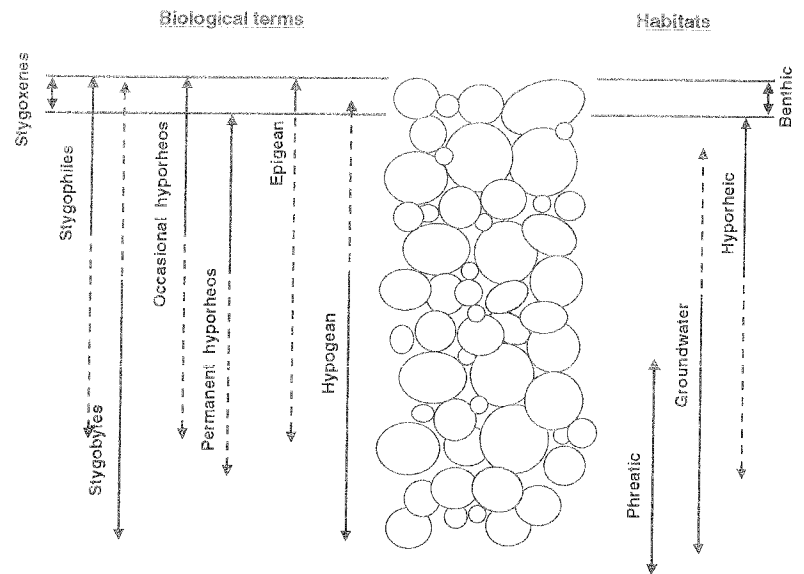


Figure 32.6 Some of the terms used to refer to surface and subsurface habitats and biota. Solid lines indicate where terms are most commonly applied and dashed lines indicate that the terms may extend beyond these limits. For example, epigean fauna spend part of their life cycle at the sediment surface, and most occur in the benthic zone, however some taxa may also be found in the hyporheic zone.

Hyporheic invertebrate biodiversity may also be high, however, biodiversity estimates are often hampered by lack of taxonomic resolution of the permanent hyporheos, particularly smaller taxa, such as microcrustacea, mites and rotifers. Thus, while few New Zealand hyporheic ecological studies have identified mites beyond the “Hydracarina”, Olsen and Townsend (2003) found a diverse hyporheic community of 13 mite genera in the Kye Burn, and made a plea for greater taxonomic effort in order to better appreciate the true diversity of hyporheic communities. As an example of how improvements in taxonomic resolution may influence biodiversity estimates, identification of the meiofauna (predominantly chironomids and rotifers) of Broadstone Stream in Southern England

Table 32.1 Fauna commonly collected in New Zealand hyporheic zones.

Occasional hyporheos “Habitat generalists”	Permanent hyporheos “Interstitial specialists”	Hypogean “Groundwater specialists”
<i>Olinga feredayi</i> (Trichoptera: Conoesucidae)	Acari	<i>Paraleptamphopus subterraneus</i> (Amphipoda: Paraleptamphopidae)
<i>Acanthophlebia cruentata</i> (Ephemeroptera: Leptophlebiidae)	Copepoda (especially Harpacticoida)	<i>Phreatogammarus fragilis</i> (Amphipoda: Phreatogammaridae)
<i>Hydora</i> (Coleoptera: Elmidae)	Ostracoda	Syncarida
Polycentropodidae (Trichoptera)	Tardigrada	<i>Cruregens fontanus</i> (Isopoda: Anthuridae)
Chironomidae	Oligochaeta	

increased the number of benthic taxa present 5-fold, from 24 to 130 (Schmid-Araya and Schmid 2000).

The presence of large numbers of invertebrates within the hyporheic zone suggests an important role of the hyporheic zone in invertebrate productivity. Indeed, several studies in New Zealand have clearly shown that hyporheic habitats make a major contribution to secondary productivity. In particular, Huryn (1996) found that once hyporheic invertebrate production was taken into account, the specter of Allen's paradox, whereby estimates of secondary production fall short of observed fish production, was laid to rest. The importance of incorporating hyporheic components into estimates of secondary production was also stressed by Adkins and Winterbourn (1999). Indeed, population biomass and production of individual taxa may be primarily, or almost solely, driven by activity in the hyporheic zone. Collier *et al.* (in press) found that 66% of reach-scale production of the mayfly *Acanthophlebia cruentata* was in the hyporheic zone of a forested stream in the Waikato. In the same stream the hyporheic zone contributed a remarkable 95% of annual production in the horn-cased caddis *Olinga feredayi* (Wright-Stow, unpublished data).

With increasing depth, invertebrate abundance, especially epigeal taxa, typically declines and community composition changes (see review in Collier and Scarsbrook 2000). Community composition also changes with season—an increase in the abundance of early instar insects may occur in autumn-winter, following the hatching of larvae from eggs laid in summer (Adkins 1997). As the abundance of insects in the hyporheic zone peaks when stream discharge is also greatest, some authors have suggested that the hyporheic zone may serve as a refuge from scouring floods (Marchant 1995; Poole and Stewart 1976). In an experiment in the braided Makaretu River in the North Island, the hyporheic zone was a more important source of post-disturbance colonists than downstream drift or crawling for the mayfly *Deleatidium* (Fowler 2002). Following extensive studies in an Otago stream, however, Olsen (2003) found little evidence that the hyporheic zone was an important refuge for invertebrates during high flows.

Recent research on gravel-bed rivers in Hawke's Bay, Canterbury, and Otago highlight the influence of hydrological exchange on the structure of hyporheic invertebrate communities. Pump-sampling of three Hawke's Bay rivers revealed a hyporheic fauna dominated by epigeal taxa (59% of abundance), but epigeal taxa such as mayflies, dipterans, and beetle larvae were the most abundant animals in downwellings, whereas hypogean invertebrates such as amphipods and copepods were most common in upwellings (Fowler 2002, Fig. 32.7). In the Kye Burn, a tributary of the Taieri River in Otago, the hyporheos, sampled by freeze-coring, was numerically dominated by early instar leptophlebiid mayfly nymphs and unpig-

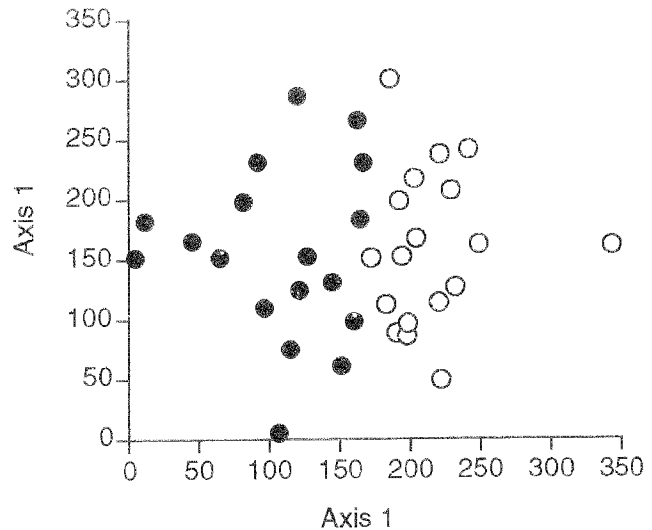


Figure 32.7 Detrended correspondence analysis plot of hyporheic invertebrate communities from upwellings (filled circles) and downwellings (closed circles) in three Hawke Bay rivers. Data reproduced from Fowler and Scarsbrook (2002) with permission from the Royal Society of New Zealand.

mented asellotan isopods (Olsen and Townsend 2003). Community composition in Kye Burn was largely determined by sediment characteristics (particle size and porosity) and the direction of hydrological exchange, and differed between summer and winter. The hyporheos of three Canterbury foothills rivers was numerically dominated by non-insect taxa (85% of all individuals), with harpacticoid copepods being particularly abundant (Burrell 2001). Community composition there differed among reaches differing in the predominant direction of hydrological exchange and also with season. In the Waipara River, invertebrate community composition changed with depth, and varied between upwelling, downwelling and intermittently flowing reaches (Burrell 2001, Fig. 32.8). Dissolved oxygen concentrations in each of these studies were typically high, indicating that other factors, particularly sediment composition and the availability of organic matter, may have limited invertebrate communities.

As the availability of particulate organic matter declines with depth, food may become limiting to the hyporheos, and a positive correlation between the abundance of particulate organic matter and hyporheos is sometimes seen (e.g., Burrell 2001; Scarsbrook 1995). Some hypogean taxa have low metabolic rates, and hence may require lower quantities and qualities of organic matter than epigeal taxa (Spicer 1998), enabling them to survive on thin subsurface biofilms. Although epigeal taxa may be capable of assimilating hyporheic biofilms, buried particulate organic matter is likely to be a more significant source of carbon to them. Thus, Burrell and Ledger (2003) found that while

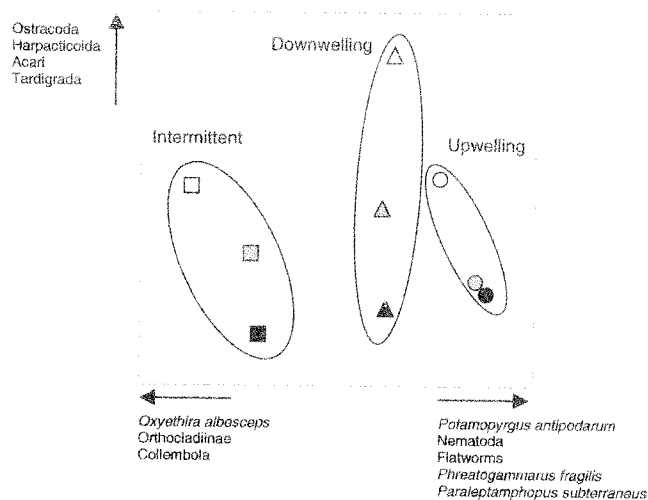


Figure 32.8 Multi-dimensional scaling ordination plot of invertebrate communities from downwelling, upwelling and intermittently flowing reaches of the Waipara River (data from Burrell 2001). Each point is the mean of 5 replicates. Open symbols are 0–15 cm depth; grey fill are 15–30 cm; black fill symbols are 30–45 cm. Taxa correlated with each axis are shown. The intermittently flowing reach was sampled following a 6 week colonisation period when surface flow was present.

the epigeal caddisfly *Olinga feredayi* ingested hyporheic biofilms, significant larval growth occurred only when they were supplied with particulate organic matter. They concluded that, given the lower availability of the particulate matter with depth, the growth of epigeal taxa such as *O. feredayi* in the hyporheic zone depends on their ability to find and move between food-rich patches at the sediment surface and within the hyporheic zone. Winterbourn and Wright-Stow (2002) found a size-specific shift in the distribution of *Olinga* in the hyporheic zone of a Canterbury stream, which they attributed to a combination of life history attributes and feeding requirements. They found that small, early instar larvae were concentrated in surface sediments, presumably close to the area of oviposition. In contrast, they found that larger individuals were found in deeper sediments, which they attributed to the presence of buried coarse particulate organic matter, a preferred food source of late instar individuals of this caddisfly.

The importance of the movement and feeding of sub-surface invertebrates in altering substrate composition, and stimulating sediment microbial activity and organic matter turnover has long been recognised in marine sediments (Powell 1977), and has received some attention in groundwaters (see Chapter 29), but has only recently been examined in the hyporheic zone. Using laboratory mesocosms, Mermillod-Blondin *et al.* (2002) showed that three hyporheic taxa each had unique and measurable effects on substrate composition, microbial activity and

solute dynamics. For example, the burrowing activity of chironomids increased the penetration of oxygen into the sediment, while isopods and tubificid worms stimulated anaerobic microbial activity, presumably due to their uptake of oxygen and excretion of organic matter. Thus, the hyporheos may be both influenced by and have an influence on interstitial flow rates and biogeochemistry.

In addition to the direction of hydrologic exchange and the availability of organic matter, substrate size is an important factor affecting invertebrate community composition. The relative abundance of interstitial fine sediment is particularly important, and may influence the hyporheos in three main ways. Firstly, smaller interstitial spaces naturally favour small, or more cylindrical body forms (Gayraud and Philippe 2001), potentially selecting against larger-bodied taxa, and limiting the penetration of epigeal taxa into the substrate. Secondly, where the ratio of fine inorganic sediment to particulate organic matter is high, food quality for the hyporheos is relatively low. This may be a significant factor affecting the penetration of epigeal taxa into the hyporheic zone, and may be more important than the total quantity of particulate organic matter available (Brunke and Gonser 1999). Thirdly, reduced hydraulic conductivity in fine sediments reduces the supply of oxygen-rich surface water and/or particulate organic matter to the hyporheos, potentially limiting the oxygen or food (Strayer *et al.* 1997).

HUMAN IMPACTS ON THE HYPORHEIC ZONE

Sediment composition and hydrology are fundamental factors influencing the extent and biogeochemical activity of the hyporheic zone (Figs. 32.9 and 32.10). As a result, human activities that alter these factors are likely to affect hyporheic processes significantly (Table 32.2). Impacts on the hyporheic zone can be broken into two categories: impacts on hydrological exchange and impacts on levels of biogeochemical activity (Boulton 2000). Sedimentation is one of the most obvious ways in which the surface-hyporheic exchange can be reduced (Brunke and Gonser 1997), and this process is exacerbated by a wide range of human activities, including roading, mining, forestry, land clearance for agriculture, and river engineering works (e.g., gravel extraction). Industry codes of practice have already been developed to avoid or mitigate the effects of many of these activities (e.g., Vaughan 1993), although the hyporheic zone is usually not explicitly recognised in these codes.

Several studies at Whatawhata, in the Waipa River catchment (Waikato), highlight the effects of land use and sedimentation on hyporheic communities. The effects of land use are primarily related to greater erosion and channel slumping in heavily-grazed pasture catchments, which enhances the transport and downstream deposition

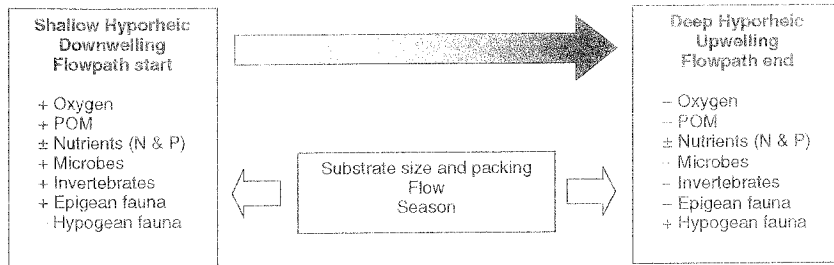


Figure 32.9 The relationship between direction of hydrologic exchange and hyporheic chemistry and biology. Substrate particle size, flow and seasonal factors also influence hydrology-biology interactions.

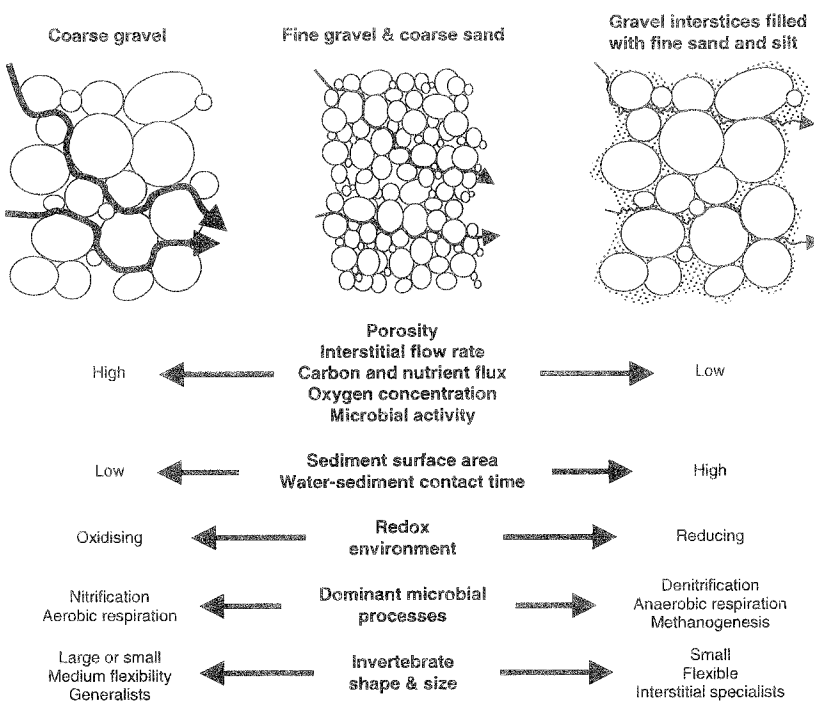


Figure 32.10 The effect of substrate particle size and packing on biogeochemical processes and biological communities.

of fine sediment. This is at least partly responsible for reduced hyporheic dissolved oxygen and an impoverished fauna relative to native forest sites at Whatawhata (Boulton *et al.* 1997). Some hyporheic taxa may be particularly sensitive to sediment deposition, and studies at Whatawhata indicate that siltation of hyporheic habitats is likely to be the principal environmental stressor of *Acanthophlebia* (Leptophlebiidae: Ephemeroptera) populations in modified pasture catchments (Scarsbrook and Halliday 2002; Collier *et al.* in press). In a comparison of streams with pasture or native forest catchments (with high and low levels of sedimentation, respectively), Boulton and Quinn (2000) found that microbial activity (measured as cellulose decomposition potential) was inhibited by

sedimentation in upwelling zones, but not downwelling zones. Thus, the effects of land use on hyporheic processes may be complex, and influenced by within-reach variations in hydrology.

Burrell (2001) provided experimental evidence of the effects of fine sediment on hyporheic communities in a North Canterbury river. Fine sediment of <2 mm diameter added to the gravels of hyporheic colonisation pots (up to 23% of total dry sediment weight) reduced community respiration, invertebrate abundance, and the percent composition of epigeal taxa at depths of up to 45 cm. A novel finding was that invertebrate community composition was influenced more strongly by fine sediment in the hyporheic zone (>15 cm deep) than in shallower sediments. This suggests that an apparent null effect of sedimentation measured by conventional benthic sampling may not necessarily mean an absence of effects on hyporheic communities. In turn, this has important implications for assessing environmental effects, as it may be more appropriate to assess the effects of sedimentation on the hyporheos than on the benthic community.

The clogging of surficial sediments, a process known as colmation (Brunke and Gonser 1997), is caused principally by physical (fine sediment deposition) and biological (formation of algal mats) processes. Although fine sediment deposition and growth of algal mats may be a natural process during periods of sustained low flow, colmation can be

exacerbated by human activities. For example, over-extraction of surface water or hydrologically-connected groundwater, or river regulation by dams may enhance colmation by prolonging periods of low flow and reducing the frequency of flushing flows. Rivers may be very important for groundwater recharge and therefore any reduction in the flow variability or bed heterogeneity that results in clogging of the bed may adversely affect groundwater recharge, as well as subsurface ecosystems.

Low or intermittent flows have direct effects on hyporheic habitat availability, as well as contributing to colmation. The hyporheic zone may provide a refuge to invertebrates from a lack of surface flow, however evidence for a concentration of epigeal invertebrates in the

Table 32.2 Human activities with the potential to impair the functioning of the hyporheic zone. Modified after Hancock (2002).

Activities	Processes that may affect hyporheic zone
River regulation	
Dam construction	Downstream sediment starvation, loss of longitudinal continuity
Regulated flows	Reduced size/frequency of bed-scouring floods, bed armouring
River engineering	
Channelisation	Isolation from floodplain, removal of bends, increased velocities
Regulated flows	Loss of habitat, disruption of flowpaths, sedimentation
Water abstraction (surface and groundwater)	Lowering of water table and river flows, decreased hydraulic pressure at downwelling and upwelling sites
Groundwater contamination (e.g. landfill leachate)	Toxic contaminants in upwelling groundwater
Land use intensification	
Increased nutrient loading	Enhanced periphyton growth
Increased sediment loading	Sedimentation
Stock access	Bed compaction, sedimentation, organic pollution (faeces)
Pesticide use	Toxic contamination
Riparian clearance	Altered flow paths, increased temperature, bank erosion
Forestry activity	Sedimentation
Invasion of exotic plants (e.g. willows, lupins, grasses)	Altered flow paths, enhanced sediment deposition
Mining	Toxic contaminants, sedimentation, pH changes

hyporheic zone in response to low surface flow is lacking. Interestingly, del Rosario and Resh (2000) found a greater diversity of hyporheic taxa in an intermittent Californian stream compared to a nearby perennial stream, and speculated that reduced numbers of surface taxa in the hyporheic zone could release permanent hyporheos from competition. Low flow conditions predominate for much of summer in many foothills-fed rivers of the Canterbury Plains, where water abstraction for irrigation is placing increasing demands on water resources (Morgan *et al.* 2002). In an intermittent reach of the Waipara River, North Canterbury, Burrell (2001) found hyporheic invertebrate abundance was reduced when surface flow was absent, but hyporheic sediments were sufficiently damp to support community respiration at a comparable level to sites with perennial flow (Figs. 32.11 and 32.12). Invertebrate abundance recovered rapidly when continuous surface flow was reestablished, however invertebrate communities were still distinct from those in reaches with perennial flow (Fig. 32.8), indicating that the effects of low flow may be persistent.

Low flow may have further indirect effects on the biota and biogeochemical processes. For example, increased water temperatures and reduced dissolved oxygen concentrations typically occur during low summer flows and may adversely affect the biota if they reach lethal levels.

Flow-dependent nutrient processes such as nitrification may also be disrupted by low flow, potentially affecting the nutrient status of adjacent groundwater and/or surface water systems. The relative importance of direct habitat loss and indirect effects of altered sedimentation, temperature, dissolved oxygen, and nutrients on hyporheic zones are generally poorly understood (Claret and Boulton 2003), but may each contribute to degradation of hyporheic communities. A major collaborative research project is now underway in the Selwyn River catchment that will address some of these knowledge gaps.

The potential human impacts detailed above primarily relate to the connectivity between the surface and the bed, although effects on levels of biogeochemical activity in the hyporheic zone will often follow as a consequence of a reduced exchange. Human activities also may directly affect activity in the hyporheic zone. For example, point-source discharges of nutrient and organic contaminants (e.g., sewage effluent) can significantly affect hyporheic processes. In laboratory microcosm experiments, Ingendahl *et al.* (2002) showed significant differences in hyporheic metabolism and nutrient transformations between sediments taken from sites above and below a German wastewater treatment plant. In particular they found higher rates of nitrification in the organically enriched sediments from below the discharge. They

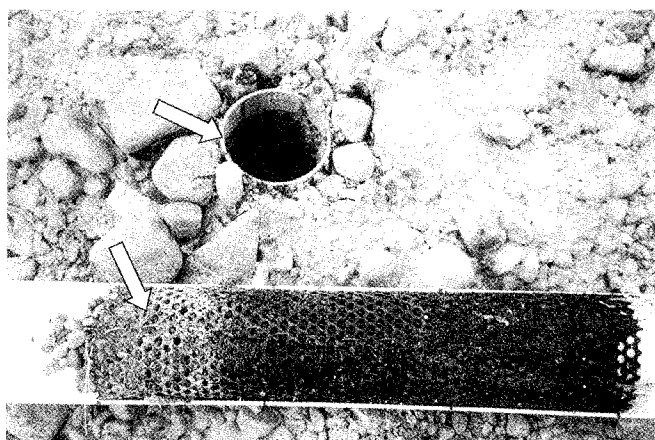


Figure 32.11 Colonisation pot in a dry channel of the lower Waipara River, northern Canterbury. The sediment-filled removable plastic mesh liner (lower arrow), has been withdrawn from the outer shell (upper arrow). A piezometer is attached to the lower left of the colonisation pot outer shell. The colonisation pot is 11 cm in diameter and 45 cm deep. Groundwater was 10 to 15 cm below the sediment surface, so that surface sediments (left end of liner) were only damp, whereas hyporheic sediments (>15 cm depth) were saturated.

Photo: G. Burrell

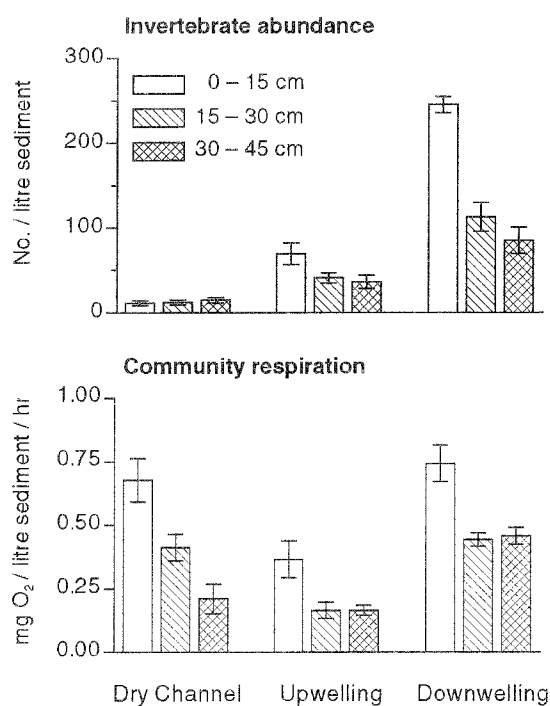


Figure 32.12 Invertebrate abundance and community respiration of colonisation pot sediments measured after a 6-week colonisation period in three reaches with contrasting hydrology of the Waipara River (data from Burrell 2001). Surface flow was absent when the dry channel was sampled, but it received intermittent flow during the colonisation period, and groundwater was 10–15 cm below the sediment surface when sampled (see Fig. 32.12).

concluded that wastewater effluent has a greater potential effect on the hyporheic zone than on the benthic zone. To our knowledge there have been no studies of the effects of point-source discharges on hyporheic processes in New Zealand.

The effects of toxic contaminants (pesticides, heavy metals) on hyporheic biota are very poorly known (Hancock 2002). Some studies have shown that the hyporheic zone is an effective filter for heavy metals originating from mining (Fuller and Harvey 2000), but information on the effects of these concentrated metals on the hyporheos is lacking.

MANAGEMENT OF HYPORHEIC ZONES

The hyporheic zone has a significant impact on both the structure (e.g., biodiversity, fish and invertebrate life-cycle completion) and functioning (e.g. physical, chemical and biological filtration systems) of aquatic ecosystems. Managers should be aware that a fully-functioning hyporheic zone provides an efficient filtration system for stream ecosystems (Hancock 2002), much like that of a filter in a fish tank. Herbst (1980) suggested that the ability of the hyporheic zone to store, transform and release organic matter is an important homeostatic mechanism in stream ecosystems. To maintain the values and functions of the hyporheic zone, river managers must seek to maintain its integrity by ensuring the connectivity between the stream surface and the bed, and by providing conditions whereby the important biogeochemical processes in the hyporheic zone are maintained.

Incorporating the hyporheic zone into river management is only just beginning in New Zealand, but considerable progress has been made in Australia (Hancock 2002; Boulton *et al.* 2003). A first step requires that managers explicitly recognise the importance of interactions and interdependencies of streams and groundwaters. In Australia this has been initiated through the development of Groundwater Dependent Ecosystems (GDEs) management policies (Hatton and Evans 1997; Murray *et al.* 2003). The hyporheic zone is seen as an important groundwater-dependent ecosystem, and as such is included in environmental management policy at both the national (Sinclair Knight Mertz 2001) and state levels in Australia. The management of groundwater-dependent ecosystems in Australia has been prompted by increasing water allocation pressures and the recognition of groundwater/surface water linkages (Boulton 2000). Given that water allocation is also a highly contentious issue in many areas of New Zealand, we suggest that a similar approach to management and protection of groundwater-dependent ecosystems (e.g., springs, wetlands, hyporheic zones, aquifers) could be useful in New Zealand.

We suggest that excessive sedimentation and excessive water abstraction from streams and aquifers are the principal threats to the "health" of the hyporheic zone in many New Zealand rivers. As a first step towards greater protection, managers should continue to seek ways to avoid excessive sedimentation in streams. This may be particularly important in spring-fed streams, where natural rates of sedimentation might be expected to be low, but where intensive land use might increase sediment delivery to streams, particularly where stock have access. Spring-fed streams usually lack the cleansing effects of floods, and are more susceptible to long-term sedimentation problems than streams with more variable flow regimes. A potential indicator of the connectivity between surface and hyporheic zones in these spring-fed streams could be the extent and/or the success of salmonid spawning. Successful salmonid spawning requires substantial exchange of oxygenated surface water with the bed (McDowall 1990), and if this exchange is limited by sedimentation, salmonid recruitment can be reduced.

The effective management of connections between the surface stream, the hyporheic zone, and underlying aquifers requires rigorous scientific methods that help identify the threats to the health of the hyporheic zone. In addition, managers require the means to assess impacts and monitor recovery. Potential measures of hyporheic health have already been suggested (Boulton 2000) and include measures of the extent of hydrological exchange, rates of interstitial activity, and invertebrate patterns of biodiversity and functional structure. Measures of the degree of hydrologic exchange range from relatively complicated tracer methods through to simple measures of differences in surface and hyporheic water temperature. Gradients in dissolved oxygen may provide a simple measure of biochemical activity in the hyporheic zone. Other more powerful measures include the use of cotton strips for measuring cellulose decomposition potential (Boulton and Quinn 2000), and direct measures of hyporheic community respiration (Burrell 2001). The use of invertebrates as indicators of hyporheic "health" is often suggested as a potentially useful tool, but so far little work has been directed at identifying gradients in community structure, or biodiversity patterns in relation to stress gradients.

Where hyporheic connections have been lost, river management strategies should aim, where practicable, to restore exchange processes. In some cases strategies used to meet other management goals could be used or modified for use in managing hyporheic exchange processes. For example, hyporheic exchange processes will often be modified by the encroachment of exotic vegetation into riverbeds. This may occur at relatively small scales (e.g., encroaching pasture grasses in small pasture streams; see

Davies-Colley 1997), or at much larger scales (e.g., encroachment of willows into braided river beds). Hence the maintenance of clear fairways in riverbeds for flood protection (see Chapter 8) may also contribute to the maintenance of good hyporheic and groundwater exchange. The practice of beach-raking (i.e., loosening of armoured gravel bars with a bulldozer and ripper) may be another management strategy that can restore or enhance surface-subsurface exchange in gravel-bed rivers, while also contributing to sediment transport management.

One of the most difficult issues facing managers is how to deal with streams where deposited fine sediments have become a problem. The removal of fine sediments from the bed of streams and rivers can be difficult, particularly in streams where natural flushing flows are absent or reduced by human intervention (e.g., spring-fed or regulated streams). Direct intervention (mechanical dredging and cleaning) will be impracticable in many situations, and managers may have to face the fact that past mismanagement may have irreparably affected many small streams and rivers, and that protection of unaffected waterways will become essential.

New Zealand's river and groundwater resources are facing increasing allocation demands, particularly for irrigation. In a recent keynote address at the Allocation of Freshwater conference in Wellington in July 2002, the Right Honourable Marion Hobbs (Minister for the Environment) noted that the area of irrigated land in New Zealand has doubled since 1985, and water allocation will be a key factor in the management of economic development and environmental management in New Zealand. During this period of increasing demand for out-of-stream water use, managers need to be aware of the strong interactions between groundwaters and surface waters, and the ecosystem consequences of the loss or reduction of these linkages, particularly in the hyporheic zone of our streams and rivers.

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Chapter 31

Karst systems

Paul Williams

INTRODUCTION

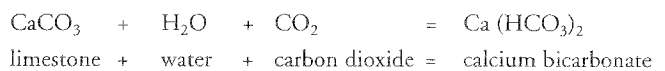
The word *karst* comes from a limestone region at the head of the Adriatic Sea centred on Slovenia and the Istrian Peninsula, which in Roman times was called the Carsus. The landscape is unusual, even bizarre, with large rivers disappearing underground, caves, enclosed depressions, periodic lakes, subterranean rivers (hence the River Styx of classical legend), and large springs. These features became known as “karst phenomena”, and this term is now applied to similar features and landscapes all over the world. New Zealand has a number of regions with karst terrain that are described in Williams (1992a). Leisure activities and adventure sports give New Zealanders first-hand contact with the karst underworld through visits to caves, “black water rafting”, and abseiling into features such as the Lost World and underground Mangapu River near Waitomo.

About 25% of the world’s population depends on water from karst systems for their domestic, agricultural and industrial needs (Ford and Williams 1989). Some small Pacific countries, like Niue, obtain all their water from carbonate rocks. New Zealand is much less dependent on karst waters, although the carbonate rocks in which they are found are quite widespread (Fig. 31.1) and contain potentially important water resources for the future.

KARSTIFICATION

Karst is found in carbonate rocks such as limestone and marble (metamorphosed limestone), because they are highly soluble in rainwater. Other soluble rocks such as gypsum also develop karst (Klimchouk *et al.* 1996), but aren’t common in New Zealand, so will not be considered further here. Rainwater contains dissolved carbon dioxide (CO_2), and is therefore a weak acid, carbonic acid (H_2CO_3). It becomes even more acidic as it percolates

through the soil, where the proportion of CO_2 in the air in soil pores can be a hundred times greater than in the open atmosphere. This makes the percolating water very corrosive towards limestone—as it seeps into fissures such as faults, bedding planes and joints in the rock, it gradually enlarges them by dissolution. In this process, the relatively insoluble calcium carbonate (CaCO_3) rock is transformed into the much more soluble calcium bicarbonate:



Over time, narrow fissures can develop into very large caves and the surface topography can become pocked with sinkholes, a process termed “karstification”. Karst development occurs principally where carbonate rocks are hard and relatively free from impurities (less than about 20% clay). Comparatively soft, marly limestones do not develop karst, mainly because the insoluble residue clogs developing small passages that might otherwise develop into caves. Consequently, karst is not found in the argillaceous limestones near Silverdale just north of Auckland.

Karstification is a fundamental process in developing the water-bearing capacity of most carbonate rocks. Some “primary porosity” is derived from interstices between the accumulating carbonate material (broken shells etc.) as it is deposited in the process of forming the rock. Temporary halts in deposition, sometimes accompanied by minor erosion, produce bedding planes. Bending of the rock during tectonic movement produces cracks known as joints, and stronger movements cause major dislocations or faults. These fissures provide potential pathways for the movement of underground water, and can be thought of as the “secondary porosity” of the rock. Karstification transforms this porosity by chemical enlargement of parts of some fissures into tubes (conduits or caves), producing a very much greater “tertiary porosity”. Circulation of

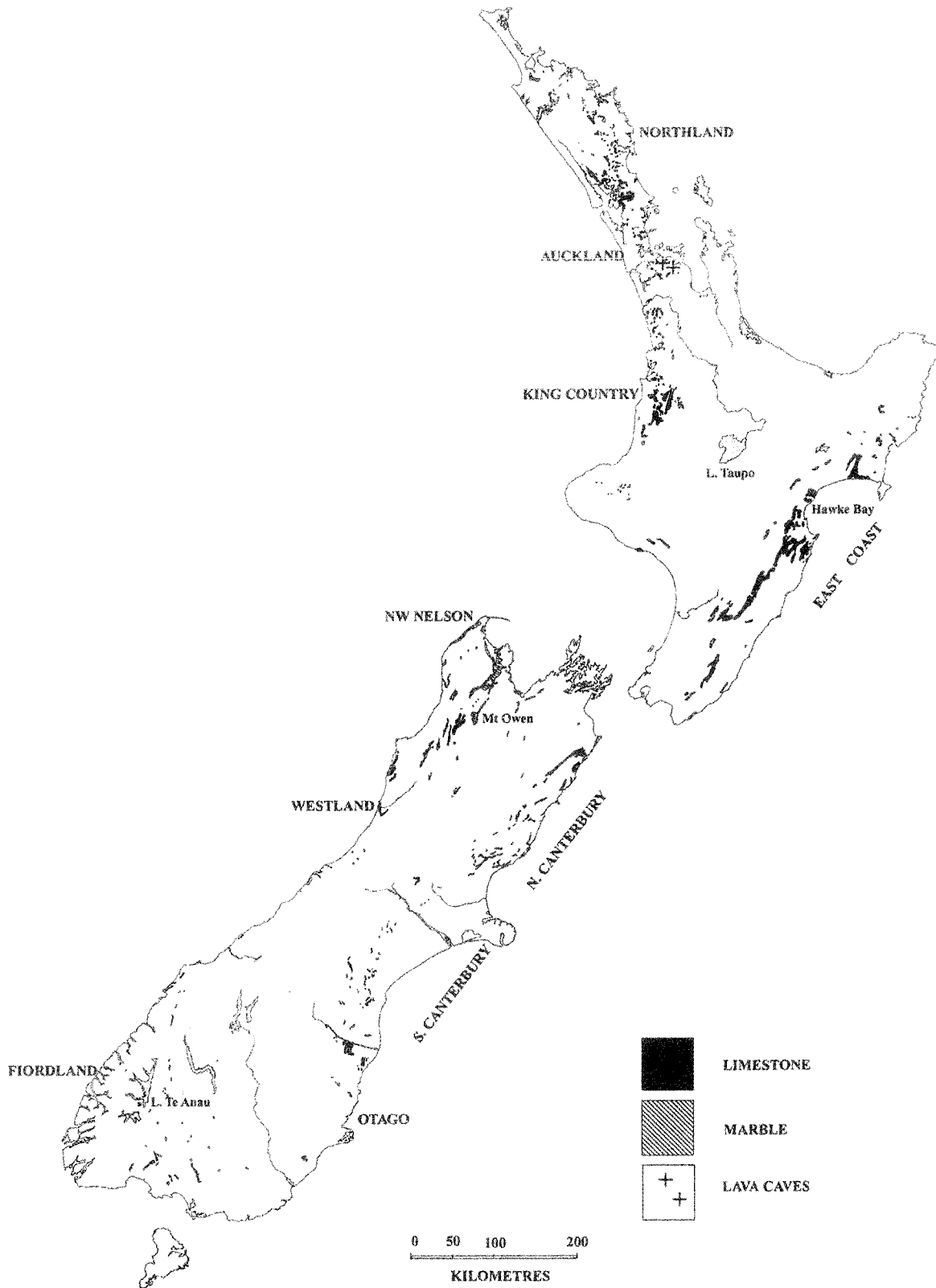


Figure 31.1 Distribution of carbonate rocks in the North Island and South Island of New Zealand (Source: New Zealand Geological Survey; from Williams 1992b).

water through the rock continually flushes the limestone with a dilute acid, and so the rock becomes progressively more and more riddled with interconnected voids of increasing size and complexity. The frequency of fissures that are sufficiently enlarged to permit rapid water flow through them increases with time, and some tertiary passages may become extremely large. The largest natural underground void known is in the Mulu karst of Sarawak in Malaysia. It is a cave chamber with a volume of 20 million cubic metres that could accommodate about 15 football fields on its floor space! The largest chamber in New Zealand, in Hollow Hill Cave at Waitomo, has a volume great enough to contain an ocean liner. Such huge spaces are not produced by chemical solution alone, but by a combination of dissolutional enlargement by underground rivers, followed by repeated roof collapse and removal of the rock debris by river flow.

Karstification does not proceed uniformly throughout a large body of rock, such as a limestone mountain. It tends to decrease exponentially with depth below the surface soil layer, which is the main source of CO_2 and hence of carbonic acid. Percolating water is most corrosive to carbonate rock near the soil and becomes increasingly more saturated with bicarbonate as it percolates downwards. However, because of the unevenness of jointing and faulting, in some places water can penetrate more easily into the rock than in others, and hence move underground more rapidly and reach greater depths before becoming chemically saturated.

The uppermost, heavily corroded layer of rock beneath the soil is known as the epikarst (or subcutaneous zone). Fissures are widest near the soil, but gradually taper or close with depth, usually over a distance of about 10 m. Consequently, infiltrating rainwater penetrates this zone much more easily than it can drain out. As a result, after heavy rain, water accumulates at the base of the epikarst in the form of a suspended aquifer above a leaky capillary barrier (Williams 1983). Such water (and any pollutants it contains) can remain in the epikarst for months, as shown by dye tracing (Bottrell and Atkinson 1992). The waste water from buildings on karst also accumulates in the epikarst—for example, discharge from septic tanks and woolshed effluent. It progressively percolates to the water table.

Because New Zealand is a relatively wet country, there is plenty of water to dissolve karst rocks. In a part of the Waitomo district with an average annual rainfall of 2,350 mm, the limestone is chemically eroded at a rate of about $69 \text{ m}^3/\text{km}^2$ each year (Gunn 1981). On Takaka Hill in Nelson, with an average rainfall of 2,158 mm, the marble is dissolved at a rate of close to $100 \text{ m}^3/\text{km}^2$ per year (Williams and Dowling 1979). At this rate, a kilometre-long cave passage measuring one square metre in cross-

sectional area would take only 10 to 15 years to form, if all the solution within one square kilometre of outcrop were focused in one place. Of course this does not occur, but the simple calculation emphasizes that karstification is rapid in a wet country. In the few million years that New Zealand limestones and marble have been subjected to karstification, there has been ample time to develop the “plumbing”—the cave systems—of large mountains. For example, the uppermost cave passages in Nettlebed Cave beneath Mt. Arthur in northwest Nelson are more than 350 m above the present water table. Some of these passages, which were formed below the water table but are now high and dry, are known from uranium series and paleomagnetism dating to be more than 700,000 years old (Williams 1987). Hence, the many kilometres of intricate passages within the underlying 350 m of marble have been formed in less than about 700,000 years.

CHARACTERISTICS OF ROCKS SUPPORTING KARST AQUIFERS

There are many kinds of carbonate rocks, and their differences are reflected in their varying capacities to store and transmit underground water. Of particular importance are the characteristics of purity, porosity, fissuring and thickness. By definition carbonate rocks contain more than 50% carbonate minerals by weight. There are two common end members: calcite (CaCO_3) and dolomite ($\text{CaMg}(\text{CO}_3)_2$). A rock composed of at least 90% calcite is termed a limestone. If its proportion of dolomite increases up to 50% it is termed a dolomitic limestone. More than 90% $\text{CaMg}(\text{CO}_3)_2$ constitutes dolomite rock. However, in New Zealand, dolomite is restricted to the small area of Mt. Burnett near Collingwood in northwest Nelson. But dolomitic limestone is common on some coral islands in the Pacific such as Niue.

Limestones and dolomites may contain insoluble materials such as clay and quartz, and are considered impure if they contain 10–50% of these materials. Rocks with more than 50% impurities are not defined as limestones or even impure limestones but as, for example, calcareous sandstones or calcareous siltstones. Some limestones, such as the argillaceous limestones just north of Auckland, are pure enough to be quarried for agricultural lime, but are much too impure to develop karst. Nevertheless, most New Zealand limestones are more than 90% calcite and are well karstified. They occur over less than 7% of the country but are widely distributed, although with marked concentrations in the west and east of the North Island and in the northwest South Island (Fig. 31.1).

Groundwater is stored and flows through interconnected voids in the rock provided by the primary,

secondary and tertiary porosity of rock matrix, fissures and conduits respectively. The primary porosity of carbonate rocks varies from about 22% in the coral reefs of Pacific islands to almost zero in the Ordovician Arthur Marble of northwest Nelson. Primary porosity is highest in the youngest rocks, including the Pliocene coquina (shelly) Te Aute limestones of Hawkes Bay and the Wairarapa. In the widespread Miocene-Oligocene limestones, the primary porosity of the rock matrix is usually less than 2%. Jointing usually increases as bed thickness diminishes. Thus the thinly-bedded, platy Miocene-Oligocene limestones that constitute most of the outcrops in New Zealand shown on Figure 31.1 are closely fissured and have a high secondary porosity. The high frequency of fissures also provides many opportunities for groundwater flow, and in New Zealand's wet climate has led to the development of many caves; consequently tertiary porosity is also high. There have been few attempts to measure porosity, but, judging from international experience (for example, see Worthington *et al.* 2000), the combined porosity of matrix and fissures is likely to provide at least 98% of the water storage capacity in New Zealand carbonate aquifers, although the conduit porosity will transmit more than 95% of the groundwater flow. Large volumes of water pass through a network of natural pipes that cover only a small percentage of the plan area. This generalisation also applies to the very massively bedded Arthur Marble of Nelson which, although having very low primary porosity and relatively few bedding-planes and widely-spaced joints, would still have most of its storage capacity in primary and secondary porosity and transmit most water through cave systems. The marble contains by far the longest and deepest caves in New Zealand, with Bulmer Cave on Mt Owen having a surveyed length of 35 km and depth of 850 m, and Nettlebed Cave on Mt Arthur being 27 km long and 980 m deep (with much more unexplored passageway in both caves beneath the water table). On the other hand, on raised coral islands such as Niue and some of the Cook Islands group, the reefs have exceedingly high primary porosity and karstification is not necessary for a large groundwater body to develop. Nevertheless, because of the solubility of the carbonate rock, large caves do form, and a sponge-like network of passages directs the flow of underground water from the interior to springs on the coast. Where fresh and salt waters mix in the coastal zone, the dissolutional potential is enhanced and "mixing solution" increases porosity and leads to the development of flank margin caves (Myroie and Carew 2000).

The thickness of karst rocks partly determines their potential to store groundwater. The Miocene-Oligocene limestones are variable in thickness, ranging from a few metres thick near Port Waikato to about 100 m thick around Waitomo. In the South Island these sediments can

vary from 20–40 m near Collingwood and Paturau to over 200 m in the Punakaiki syncline between Westport and Greymouth. The rocks are also highly variable in composition throughout this thickness, ranging from very pure and crystalline to sandy, shelly and argillaceous. However, even the 550 km of the great Flint-Mammoth Cave system of Kentucky is developed in only about a 100-m thickness of limestone, so the comparative thinness of a carbonate formation need not unduly restrict the development of a major aquifer. Continuity is also an important factor. The Miocene-Oligocene limestones of the Te Kuiti Group occur in patches over about 800 km² of the western North Island, but few really large aquifers can develop because of their discontinuous distribution. Other patches outcrop on mountain summits, for example, the Matiri Tops of the Buller valley; they are freely drained by gravity and so cannot support large underground water bodies.

In contrast with these is the Arthur Marble of Nelson, which in places approaches a kilometre in stratigraphic thickness and extends in a discontinuous belt for about 90 km from Collingwood to Mt. Owen (Fig. 31.1). The outcrop is up to 7 km wide and extends from 1,876 m above sea level on Mt. Owen to well below sea level in the Takaka valley. In places it contains huge groundwater reservoirs, and it sustains the flow of New Zealand's largest springs—the Waikoropupu Springs near Takaka. Smaller patches of marble of similar age outcrop in Fiordland.

THE FUNCTIONING OF KARST HYDROLOGIC SYSTEMS

All the basic concepts and principles of groundwater hydrology apply to karst, but some additional factors give karst its special character. Unlike groundwater systems in other rocks, the circulation of water in a karst aquifer continuously enhances the aquifer's permeability, and thereby progressively modifies the groundwater hydrological system. The capacity of karst rocks to store water increases as the interconnected voids are enlarged by solution. Resistance to water flow within the system simultaneously decreases. Hence, as the storage capacity increases, the water table is lowered and the hydraulic gradient becomes less steep. The water table in karst terrain can consequently lie at a considerable depth below the surface and it can be very nearly flat. For example, under Mt. Arthur the water table lies up to 900 m beneath the upper slopes of the mountain and its level is controlled by the elevation of the main outflow spring, the resurgence of the Pearse River. In the Pearse spring, divers have found flooded passages to penetrate downwards well beyond the limit of exploration at 70 m below the water surface, the marble continuing to an unknown depth.

Three main hydrographic zones are recognised in karst: a) the aerated or vadose zone, b) the permanently waterlogged or phreatic zone, the top of which is the water table, and c) an intermediate epiphreatic zone that is periodically flooded as the water table rises and falls. In New Zealand, the epiphreatic zone can be up to 75 m in thickness in some karsts (Xanadu system, Punakaiki). The uppermost part of the vadose zone beneath the soil is highly weathered because of vigorous carbonate dissolution in proximity to the abundant CO₂ source in the soil. This zone of enhanced porosity is known as the epikarst (also termed the subcutaneous zone) and is typically 10 m thick. All of the hydrographic zones move downwards through the karst as the landscape is lowered by solutational denudation. The elevation of the water table is determined by the level of the outflow spring that drains the aquifer. Hence, as the spring is lowered by downcutting of the river into which it flows, so is the water table. In Nettlebed Cave beneath Mt. Arthur, the water table has been lowered at an average rate of 0.44 m per 1000 years (Williams 1987).

Recharge in karst systems

A fundamental consideration in understanding how a karst groundwater system functions and has developed is the origin of the waters that recharge it. A distinction should be made between the precipitation that falls directly onto the limestone outcrop (termed *autogenic recharge*) and that which falls on neighbouring non-carbonate rocks, from which it flows as streams or rivers onto the karst (termed *allogenic recharge*). These two forms of recharge have major implications for the aquifer. Precipitation falling onto karst is dispersed relatively evenly and has a distinctive chemistry. This type of recharge and associated dissolution results in the development of an extremely intricate network of interconnected fissures and pores, especially in the subcutaneous zone beneath the soil. Hence it is responsible for the generation of most of the diffuse near-surface storage in the groundwater system. By contrast, allogenic recharge is already collected into streams when it flows onto the karst; consequently recharge is focused at the place where stream flow is lost underground. The chemistry of streams flowing onto karst reflects the terrain where they originated, and their erosional capacity may differ from that of rainwater. Incoming streams commonly disappear underground suddenly at stream-sinks, although some streams lose their flow gradually by seepage into their beds. Sinking streams flow through large conduits to springs and are mainly responsible for the development of caves.

Geology and topography determine whether autogenic or allogenic recharge will dominate the development of a karst hydrologic system. Isolated karst plateaus with little or no impermeable coverbeds, such as Mt. Owen in

Nelson, Mt. Kahuranaki on the East Coast, or uplifted coral reefs such as Niue Island, are mainly recharged diffusely by precipitation. But where higher impervious rocks drain onto karst, or where dissected, impervious coverbeds shed water onto underlying limestones, then allogenic recharge results. This is a most common occurrence and is found throughout the King Country where Miocene siltstones and sandstones drain onto the limestones beneath. In Abel Tasman National Park, streams collected on high granite inliers flow onto surrounding but lower-lying marble, and in some of the Cook Islands water is shed radially from interior volcanic uplands towards a ring of uplifted coral reefs. Nevertheless, in practice most karsts are recharged by both direct rainfall and stream flow.

STORAGE AND TRANSFERS IN KARST HYDROLOGIC SYSTEMS

Karst groundwater systems can be visualised as a series of stores of water, such as the saturated (phreatic) zone store and the suspended epikarst store, interconnected by passages through which water flows until it reappears at a spring (Fig. 31.2). Some stores, such as that in the epikarst, are small and can hold comparatively little water at any one time, but transmit a large volume during the course of

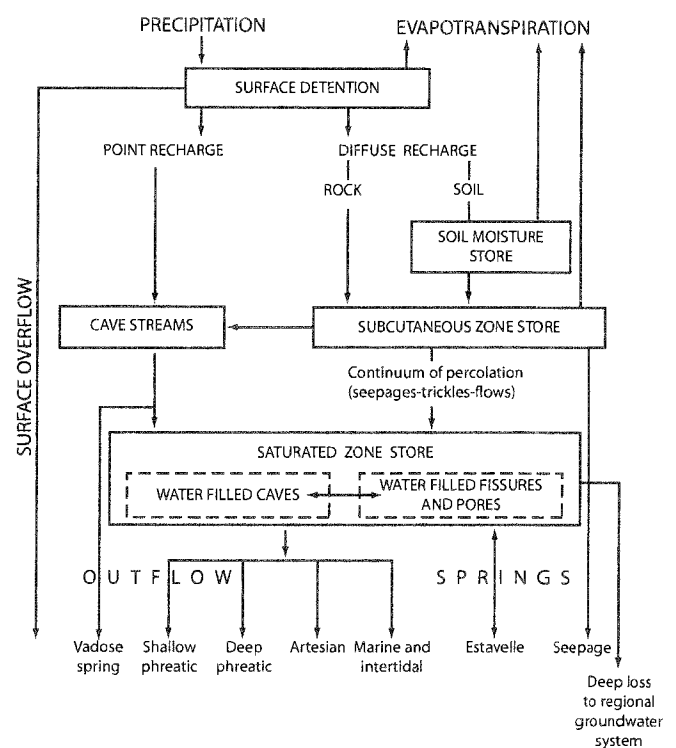


Figure 31.2 Stores and linkages in karst hydrologic systems. The soil moisture store, subcutaneous zone store and cave streams are located in the vadose zone. The saturated zone store is in the phreatic zone (from Ford and Williams 1989).

a year. Others can be very large, such as the porous matrix store, but have relatively little water passing through them because of resistance to flow in the small interstices. In the first case, the turnover time of water in storage is short, perhaps a few weeks or months, whereas in the second it may be many months to a few years.

The connections between the stores are fissures, pipes and pores of various sizes. Some of the largest are caves many square metres in cross-sectional area through which underground rivers flow. In the vadose zone these rivers are similar to those at the surface, except that they have a roof, but within the phreatic zone they completely fill the passage, have no free surface and may flow under pressure. In the phreatic zone, the interstices of the rock are full of water derived both from vertical percolation from the vadose zone and from the back-flooding of phreatic rivers during high discharge events. There is a net movement of water from these fissures into the flooded cave passages and thence to the spring.

Darcy's law describes groundwater movement in the phreatic zone, below the water table, *provided* the movement is slow enough to be laminar, conditions are isotropic and the host rock is homogeneous. In karst these conditions often do not hold, especially because water flow in pipes and caves is frequently turbulent (Ford and Williams, 1989). Nevertheless, in some areas of New Zealand and the south Pacific, Darcy's law can be applied to karst as a reasonable first assumption—for example, in the uplifted coral islands of Niue, the Cook group, and Tonga, and possibly in the thick marble beneath the Takaka valley where storage is huge and turnover time is very slow. Groundwater flow through karst, therefore, tends to vary between the extremes of turbulent flow through pipes on the one hand and Darcian flow through pores and fissures on the other, with most karsts having a mixture of both (Fig. 31.3).

Because karst hydrologic systems are largely subterranean, they are difficult to observe and study. Nevertheless, a great deal can be deduced about the underground system sustaining the flow of a spring from observation of the spring itself. The karst groundwater system is virtually a "black box", but when it rains, the recharge stimulates a response at the spring. The nature of the response depends on the characteristics of the groundwater system through which the recharge has passed. Typically, two simultaneous signals are measured at the spring—the variation in its discharge and the variation in the chemical quality of its water. Electrical conductivity is often used as an index of mineralization of the water. The responses to rainfall of both discharge and conductivity at a spring are rapid when recharge is dominated by sinking streams and groundwater flow passes mainly through caves. But the responses can lag

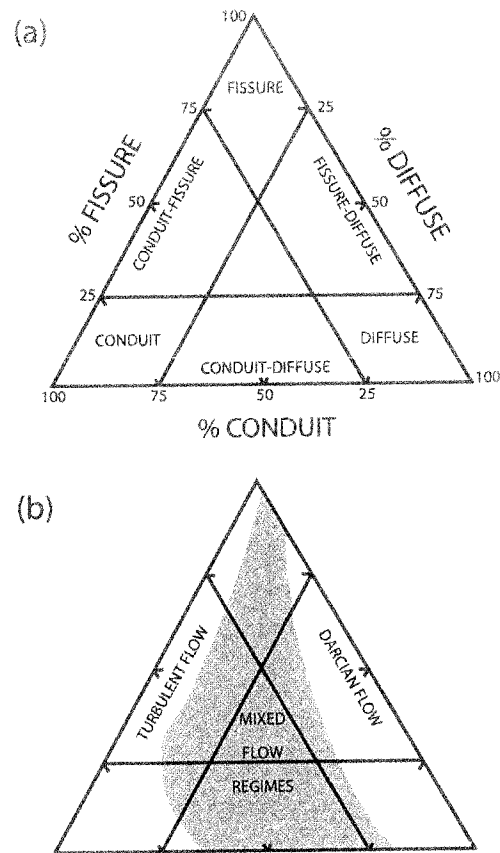


Figure 31.3 Conceptual classification of karst aquifers and their presumed relationship to predominant flow regimes. In (a) a distinction is made between carbonate aquifers dominated by diffuse recharge and flow through interconnected pores, as opposed to point recharge through stream-sinks and strongly directed flow through caves (conduits), or flow through a geometrical network of fissures (joints and faults). As indicated in (b), these styles of groundwater transmission strongly influence whether flow will be dominantly Darcian (laminar), turbulent or a combination of both in different parts of the aquifer (from Ford and Williams 1989 after Atkinson 1985).

appreciably when recharge is autogenic and groundwater flow is Darcian. In the latter case, the frequency distribution of recorded conductivity values is typically unimodal, because of the similar chemical history of the water that has recharged diffusely and has passed through a highly fissured or porous rock. By contrast, in the case of sinking streams and flow through caves, the frequency distribution of conductivity values recorded at the spring is characteristically multimodal (Fig. 31.4). In a well karstified, mixed autogenic/allogenic system, different recharge modes, flow routes and residence times result in contrasts in water quality, and as the water from these various places in the groundwater system emerges at the

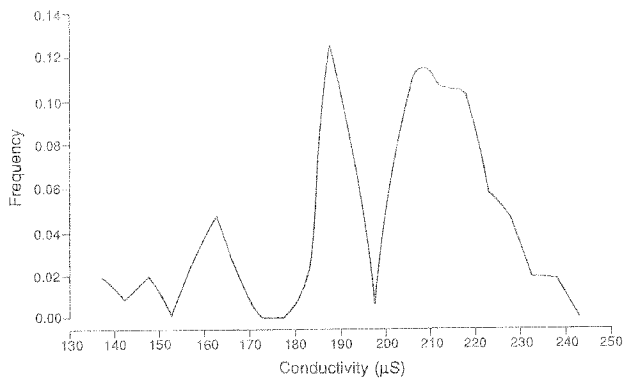


Figure 31.4 Frequency distribution of electrical conductivity values from a karst spring in the Mangakowhai basin near Te Kuiti. The multimodal nature of the curve is typical of well-karstified aquifers with mixed flow regimes (from Williams 1992b).

spring, so the water quality varies. Figure 31.2 illustrates the complexity of some systems. The highest conductivity peak is from water that has passed through the subcutaneous zone, where it dissolved substantial amounts of limestone in open-system contact with the soil CO_2 source. The lowest conductivity peak is the chemical “fingerprint” of relatively unmineralized, rapidly flowing conduit (cave) water from stream-sinks; and the intermediate peaks are from mixed sources with a moderately low residence time that has passed through fissure systems. Most karsts in New Zealand are of this well-developed complex type.

WATER TRACING AND POLLUTION TRACING

Springs are traditional sources of drinking water. People tend to assume that water is purified by filtration through soil and rock, so that when it re-emerges it is essentially free of bacterial and chemical pollutants and is safe to drink. In fact, karst is an extremely bad filter, and in agricultural countries like New Zealand, karst spring water is usually not safe to drink without boiling.

How many tomos (sinkholes) used for rubbish disposal are there on farms in our karst areas? Probably every farm on karst, not just in New Zealand but all round the world, has used naturally occurring sinkholes as convenient places to dump dead stock, farm rubbish and domestic refuse. “Out-of-sight, out-of-mind” is a common philosophy. Yet literally thousands of water-tracing tests by karst hydrologists have shown flow velocities from sink to spring to be typically in the range of 80 m/hour, but to vary by more than four orders of magnitude (Worthington *et al.* 2000). In other words, farm refuse washed by rain from a tomo into an underground stream can often easily travel a

kilometre or more to a neighbour’s spring in a day or so. This is insufficient time to kill bacteria, since most species require more than 100 days to die. Physical filtration in karst conduits is also negligible. Because karst springs in many parts of the world are very important water supplies, considerable effort has been made to develop techniques to trace the source of their waters, and to define the catchment areas from which they are recharged. When the catchment has been defined, land-use practices can be managed to protect the quality of the groundwater.

By far the most effective way of tracing underground waters is by means of fluorescent dyes. The most commonly used dye is fluorescein ($\text{C}_{20}\text{H}_{12}\text{O}_5$), a green dye that was first used as a tracer of water sinking into the limestone bed of the upper River Danube in 1877. This dye is ecologically safe, but is less efficient than Rhodamine WT. The latter is an orange dye designed especially for water tracing, which under good conditions is detectable in concentrations of as little as 1 part per billion. These and other fluorescent dyes can be detected with instruments such as fluorometers and UV spectrophotometers (Ford and Williams 1989; Käss 1998) and results of water tracing experiments can be used to quantify the characteristics of the groundwater system (US Environmental Protection Agency 2002).

MAJOR KARST AQUIFERS

The Takaka valley

The most important karst aquifer in New Zealand in terms of the volume of water in storage is in the Takaka valley, northwest Nelson (Williams 1977; Stewart and Williams 1981; Stewart and Downes 1982; Mueller 1991). It is found in Takaka terrane rocks of the Mount Arthur Group, particularly Arthur Marble 2 of late Ordovician age (Rattenbury *et al.* 1998). These limestone and marble rocks underlie the north-south fault-angle depression of the Takaka valley and outcrop on the surrounding hills to the east and west (Fig. 31.5). The marble extends over about 180 km^2 and has a stratigraphic thickness of well over 500 m, possibly attaining 1,000 m, so its water storage potential is considerable. The downstream 45 km^2 of the marble is capped with Tertiary coal measures, so the outflow zone of the aquifer is confined under artesian conditions.

Three major rivers drain from surrounding highlands into the valley—the upper Takaka River, the Waingaro and the Anatoki (Fig. 31.5). Each river loses flow into its bed when it crosses the marble, providing a major source of recharge. The upper Takaka River has an average flow of 16.1 m^3/s , but loses up to 10 or 11 m^3/s into the aquifer by percolation into the fluvial gravel that veneers the marble. Since the flow of the river is less than 11 m^3/s for

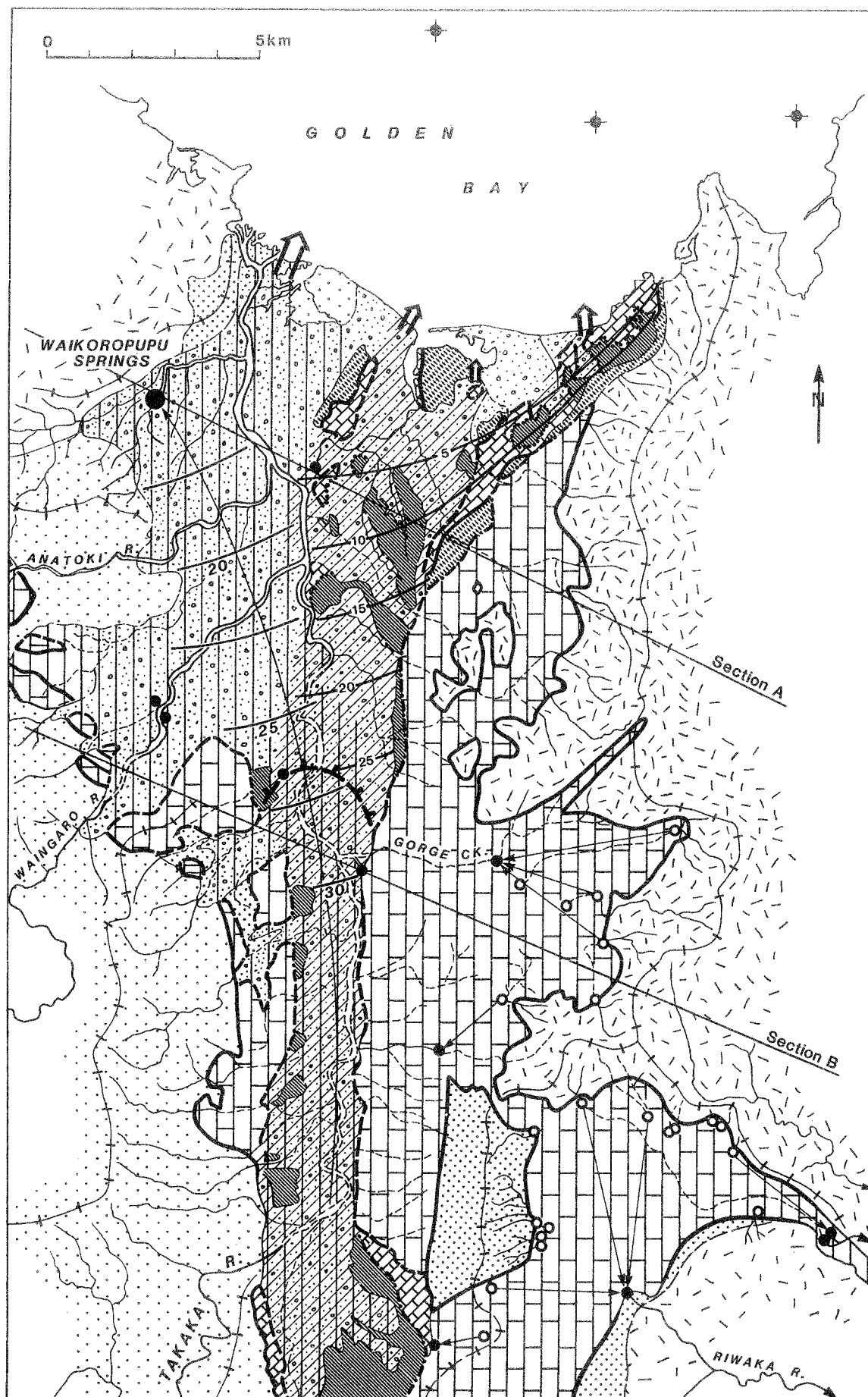
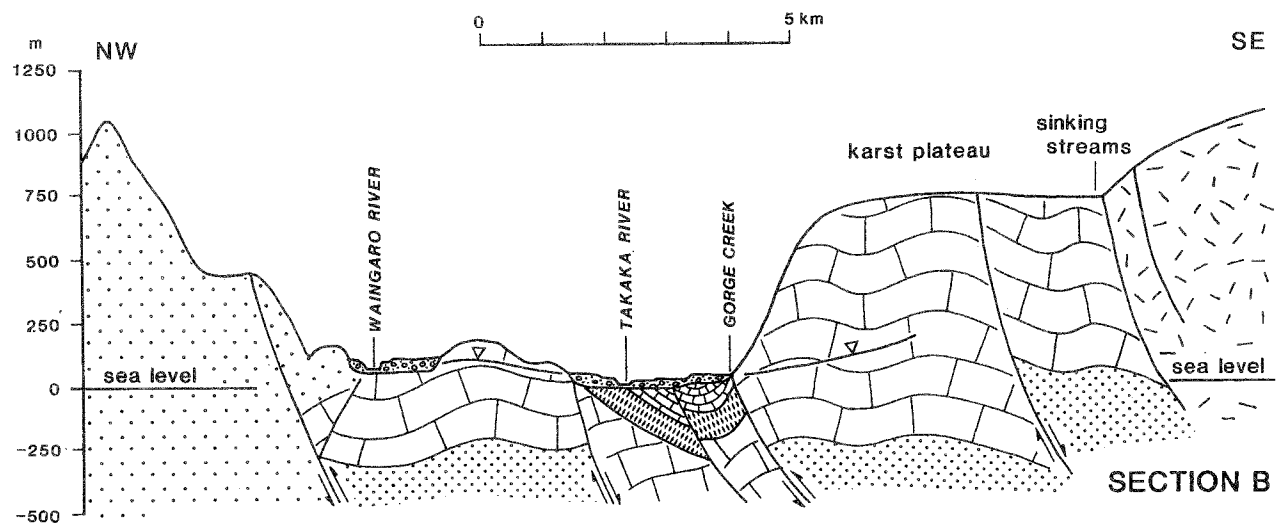
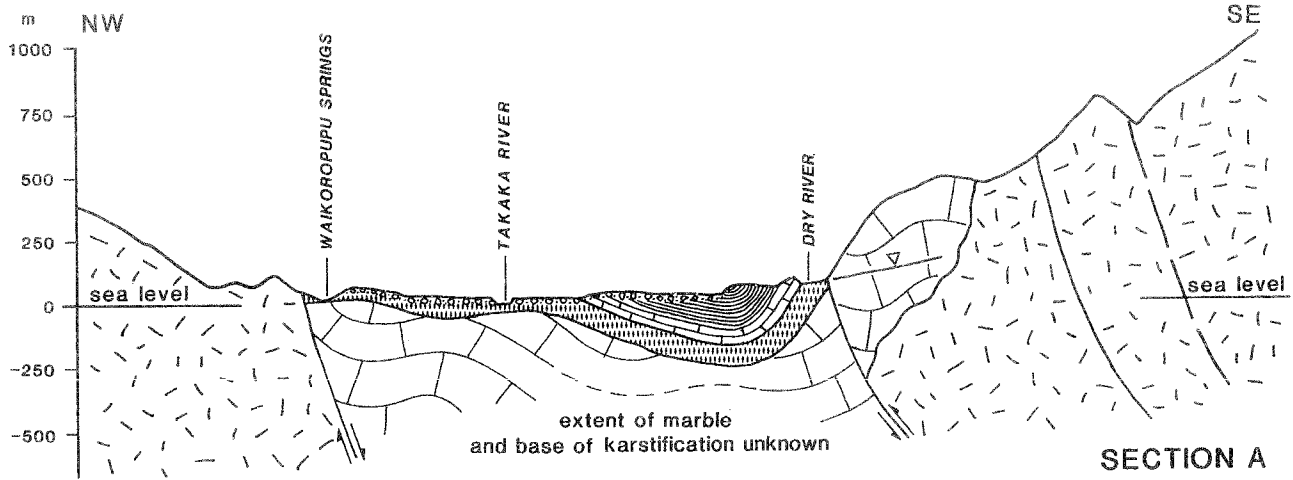
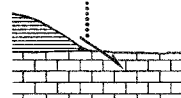
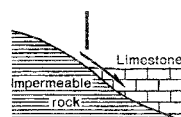


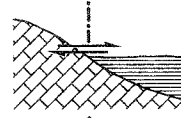
Figure 31.5 A hydrogeologic map and cross-sections of the Takaka valley (from Ford and Williams 1989).



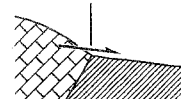
INFLOW BOUNDARY



REVERSIBLE FLUX

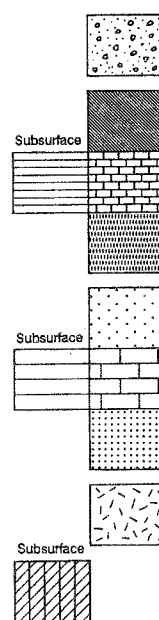


OUTFLOW BOUNDARY



- stream sink
- influent stream
- proven groundwater connection
- groundwater divide in unconfined aquifer
- surface watershed
- limestone 10
- marble 20
- upstream boundary of confined aquifer in marble
- submarine spring
- spring (dry season low flow $>5\text{m}^3\text{s}^{-1}$)
- spring (dry season low flow $<5\text{m}^3\text{s}^{-1}$)
- effluent watercourse, coastline
- presumed groundwater outflow

ROCK TYPE AND AGE



Quaternary sands & gravels <50m

Miocene Tarakohe Mudstone ~600m

Oligocene Takaka Limestone 25 - 62m

Eocene Motupipi Coal Measures ~5 - 330m

Ordovician Onekaka Schist

Ordovician Arthur Marble ~1000m

Ordovician Pikikiruna Schist

Lower Cretaceous to Devonian igneous intrusives

Limestone overlying marble

HYDROLOGICAL CHARACTERISTICS

Unconfined aquifer. Most productive in Holocene floodplain gravels

Aquiclude

Aquifer confined in places

Aquiclude

Aquifuge

Major aquifer, confined in places

Aquifuge

Aquifuge, except in deeply weathered zones

Superimposed carbonate aquifers usually separated by clastics

about 100 days per year, the river bed across the marble is dry for long periods of time (Fig. 31.6). However, the similar-sized Waingaro and Anatoki rivers together only lose about 2 m³/s of their combined flow, because they cross narrower strips of marble. Further allogenic recharge comes from numerous small streams, which drain into the marble from higher non-karst rocks in the surrounding uplands.

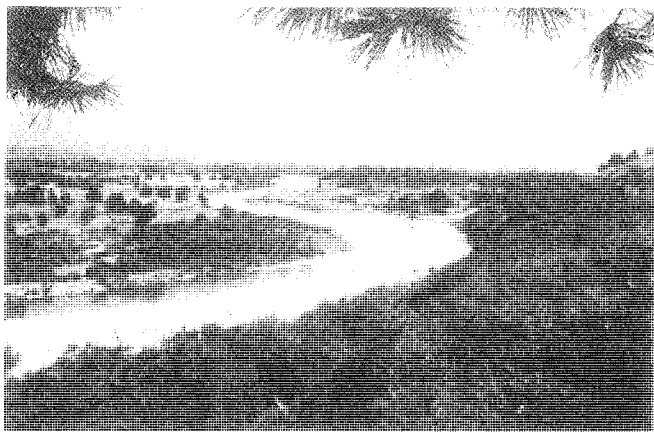


Figure 31.6 The Takaka River in its main influent reach, about 14 km from the Waikoropupu Springs. Up to 11 m³/s of flow is lost over a distance of about 4.5 km, with most of the flow passing through gravels overlying marble at the contact with the Pikikuruna Fault. The view is to the north and shows the final surface water, with the dry river bed further downstream.

Photo: Paul Williams

Autogenic recharge is contributed by rain falling on the 90 km² of marble outcropping on the surrounding hills, much of it within Abel Tasman National Park. Further recharge comes from rain falling onto 45 km² of gravel flats overlying the marble in the central portion of the valley floor. Given evapotranspiration losses of about 700 mm, annual recharge in these areas totals about 1900 mm, which would sustain an outflow of approximately 8 m³/s. However, the total annual recharge to the marble aquifer is estimated to be about 23.4 m³/s. The main outflow from the aquifer, the Waikoropupu Springs (Figs. 31.7 and 31.8), is artesian and has a combined discharge of 15 m³/s, leaving a difference of 8.4 m³/s to be accounted for, some of which may overflow at Spring Brook on the western side of the artesian boundary. Submarine springs are known to exist on the bed of Golden Bay, so this unaccounted sum may be their annual outflow. However, their flows and other characteristics have yet to be measured. These marine resurgences were probably developed during glacio-eustatic low stands of the sea in the Pleistocene, when Golden Bay would have been dry land. A submarine spring near Rangihaeata may be the main outflow.



Figure 31.7 The Main Spring of the Waikoropupu Springs shows a strong artesian boil. The area beyond the person on the far side of the pool is now occupied by a salmon farm.

Photo: Paul Williams

The Waikoropupu Springs are located 2.6 km inland from the head of tide and the main vent is 7.1 m above sea level, but they draw water from marble that extends well below sea level. The spring water contains from 0.4–0.6% salt water, with the proportion increasing as discharge increases, indicating that the more dynamic the outflow the greater the entrainment of saline water (no evaporate beds are known in the region that could provide an alternative source of salt).

Dating of Puppu Springs' water by tritium (H³) and characterisation of recharge and groundwaters by oxygen and hydrogen stable-isotope analyses have shown the Takaka valley hydrogeologic system to be complex (Fig. 31.9). Water emerging at the springs has a spectrum of ages—it is a mixture of water that has come from different sources and been underground for various lengths of time. Taylor (cited in Mueller 1992) postulated a base reservoir of water that has been underground for longer than 10 years, but not more than 20 years, supplemented by a much more recent component, that Stewart (cited in Mueller 1992) identified to be about one year old. Although Stewart and Williams (1981) found the aquifer to be well mixed, it cannot be modelled as a simple exponential reservoir, because it also contains elements of piston flow. It can be envisaged as a cascade of mixing compartments connected by linear conduits (Fig. 31.9), and while the long average turnover time indicates that flow within the aquifer could be laminar, there are clearly turbulent components.

Water emerges at Waikoropupu Springs in a series of vents (Fig. 31.8) that collectively constitute the largest spring in New Zealand. The spring is also amongst the 50 largest known karst springs in the world there being at least 26 with average discharges of >20 m³/s. Their ecology

is recognised as having international significance (Michaelis 1976, 1977) and they are protected within Abel Tasman National Park. Yet despite this status, they are not immune from difficult management problems, because water extraction from the aquifer sustaining the springs reduces their flow and human activity in the recharge zones will contaminate the groundwater and its ecosystem.

A dilemma arises concerning water shortage in the face of plenty when it comes to using some of the Waikoropupu Springs' water resources. The outflow to the Springs River is $15 \text{ m}^3/\text{s}$ on average, with a maximum of about $21 \text{ m}^3/\text{s}$ and, given that the average residence time of water in the system is 3 to 8 years, then the volume of storage in the underground reservoir is at least 1.5 km^3 . This should be more than sufficient water for local needs, but the total amount in storage is less significant for water supply allocation than the dynamic reserves, i.e. the volume involved between the extremes of high water outflow and low water outflow. Since the lowest flow so far measured is $5.3 \text{ m}^3/\text{s}$, and agricultural demand for water is often greatest during droughts, too much pumping from the aquifer or abstraction from the spring may seriously endanger the Springs River ecosystem. Because of existing water allocation for salmon farming beside the Springs, the District Council has set an interim limit of $0.5 \text{ m}^3/\text{s}$ for total abstractions from the recharge zone for the Waikoropupu Springs, and the minimum residual flow in the Springs River is set at $2.9 \text{ m}^3/\text{s}$ (20% of the natural mean discharge). Maintenance of water quality places further constraints, as leaching of materials from agricultural chemicals and stock in the recharge zone and waste-water contaminants from the salmon farm may cause pollution. Increasing suburbanisation of the surrounding karst, which involves septic-tank waste-water disposal, will undoubtedly lead to degradation of epikarstic waters and involve fast flow by-passes to deeper groundwater reserves.

Aquifers in Cenozoic limestones

King Country karst

The distribution of limestones in the King Country in the west-central North Island is shown in Figure 31.10. The stratigraphy of the district has been described by White and Waterhouse (1993) and the geomorphology of the karst by Gunn (1981) and Williams (1992a and *in press*). The principal karst-supporting rocks are the Orahiri Limestone of Oligocene age and Otorohanga Limestone of lower Miocene age. Both formations are pure and crystalline, generally with $>90\%$ calcium carbonate.

The limestones show considerable lateral variability

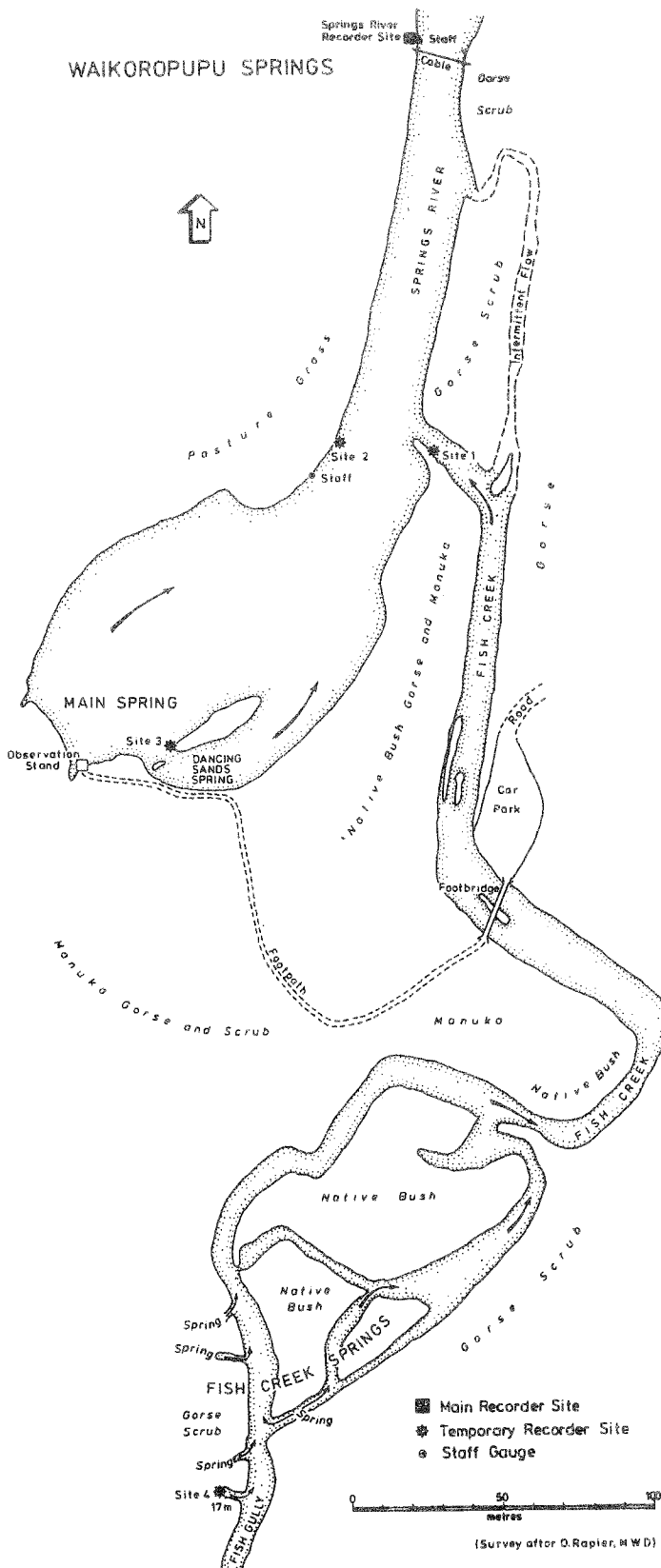


Figure 31.8 Plan of the Waikoropupu Springs and Springs River complex near Takaka (from Williams 1977 after Rapier 1975).

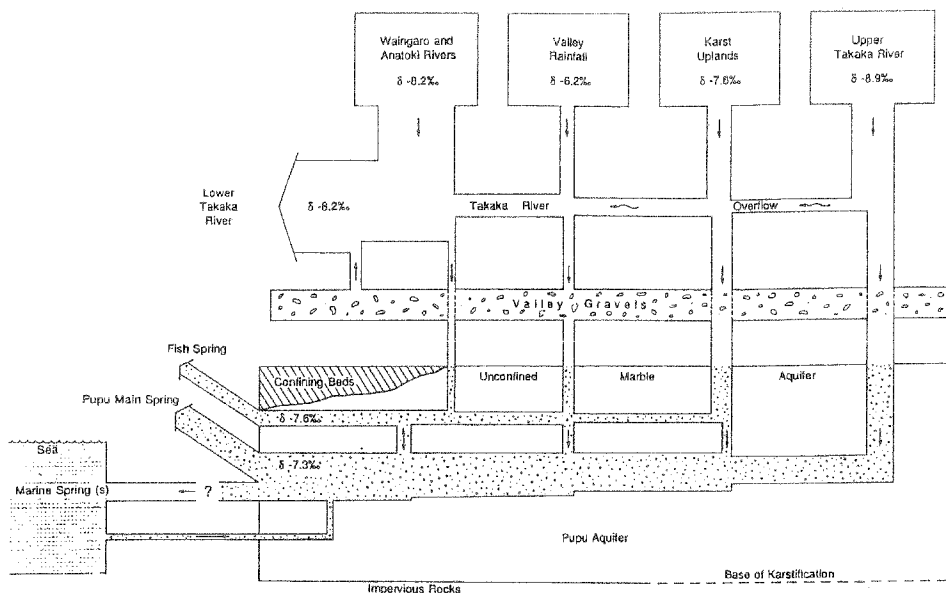


Figure 31.9 A schematic model of the Takaka valley – Waikoropupu groundwater system. Delta (δ) values refer to oxygen isotope $\delta^{18}\text{O}$ in water (from Williams 1992b).

(Fig. 31.11) and are particularly notable for their thin-bedded flagginess in weathered outcrops. Most of the beds are only of the order of decimetres in thickness, although occasionally some exceed a metre, and pseudo-bedding occurs along stylolitic partings arising from pressure solution during diagenesis. Associated with the thin bedding is closely spaced jointing. Thus the limestones have a very high frequency of fissures, which are exploited by hydrogeological processes. In places these limestone formations are separated by a glauconitic calcareous sandstone, the Waitomo Sandstone, but it is lenticular and not always present. Where it occurs it provides an internal barrier to vertical water movement.

The general direction of dip in the limestones is towards the east. Folds are broad and simple and dips are gentle, seldom exceeding $10\text{--}15^\circ$. Prominent N-S faulting results in the limestones ascending from Waitomo towards the west, up onto a structural high underlain by Mesozoic siltstones and sandstones that form the regional basement. Vertical displacements total several hundred metres. As a result of Upper Tertiary and Quaternary tectonic and erosional processes, the karstified limestones in the Waitomo district now present a fragmented mosaic of about 800 km^2 , of which about half is capped by eroded remnants of Miocene mudstones and sandstones (Fig. 31.10). Considerably more essentially unkarstified limestone remains at depth beyond the influence of active groundwater circulation.

An important feature of the karst in the area is its very high density of pitting by enclosed depressions (or dolines). These occupy all the available space on limestone outcrops and produce an “egg-box” pattern of dissection, termed “polygonal karst” (because the topographic divides of the

adjacent dolines form a polygonal mesh pattern in plan). Each cell of the polygonal mesh is part of the recharge system (Gunn 1983) and drains centripetally to underlying conduits that form the distal tributaries of dendritic subterranean drainage networks that eventually emerge at large springs. The Mangakowhai drainage system, an International Hydrological Decade basin, provides a good example (Fig. 31.12), as does the neighbouring Mangapu basin that drains through the Lost World (an abseiling site for cave tourists). Such polygonal karst and its dendritic subterranean systems are very efficient drainage systems, and numerous water-tracing experiments have proven underground water flow to be swift and turbulent. However, not all recharge passes rapidly into the conduit system, because dolines are leaky colanders rather than simple funnels. Much infiltrating water passes through the soil into the underlying subcutaneous zone, where it is temporarily stored prior to further percolation down through the vadose (unsaturated) zone. Some cave passages pass through this zone and provide the opportunity to study percolation processes. Thus from a study comparing oxygen stable isotopes in rain water and percolation water, Williams and Fowler (2002) found the flow in this zone to take of the order of months, although some percolation points responded in a matter of hours (by a translatory pressure pulse effect) to recharge by heavy rainfall.

The largest springs in the district have mean discharges in the range of $1\text{--}3\text{ m}^3/\text{s}$. These include the Mangapu, Mangawhitikau, Mangakowhai and Waitomo springs. All emerge in alluviated valleys. Divers exploring some springs have found drowned stalactites, which provide clear evidence of aggradation of the alluvial lowlands since the caves formed. The hydrological effect of this alluviation

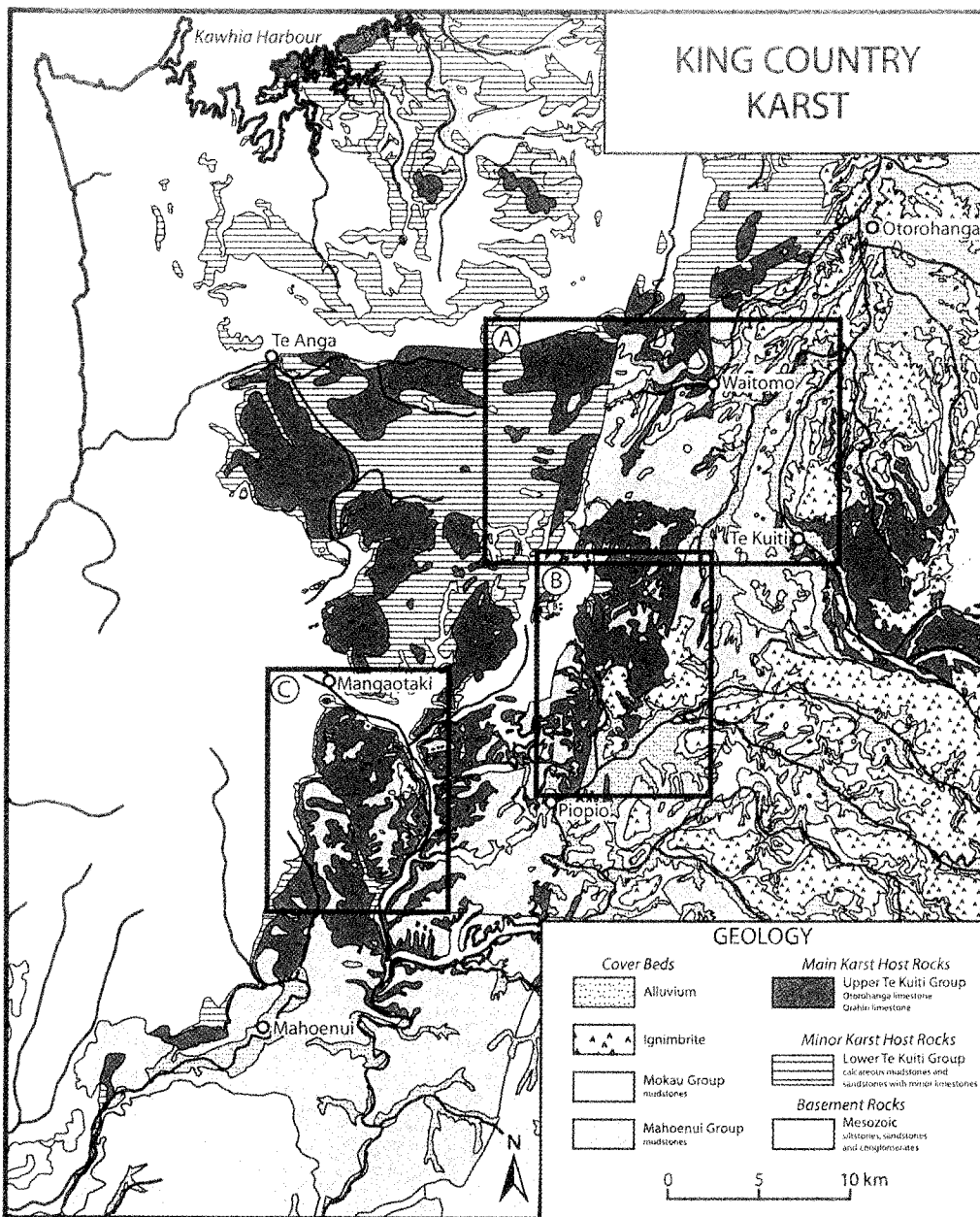


Figure 31.10 Karst occurrences in the King Country, west central North Island. Inset B is the location of Fig. 31.12 (from Williams 2004).

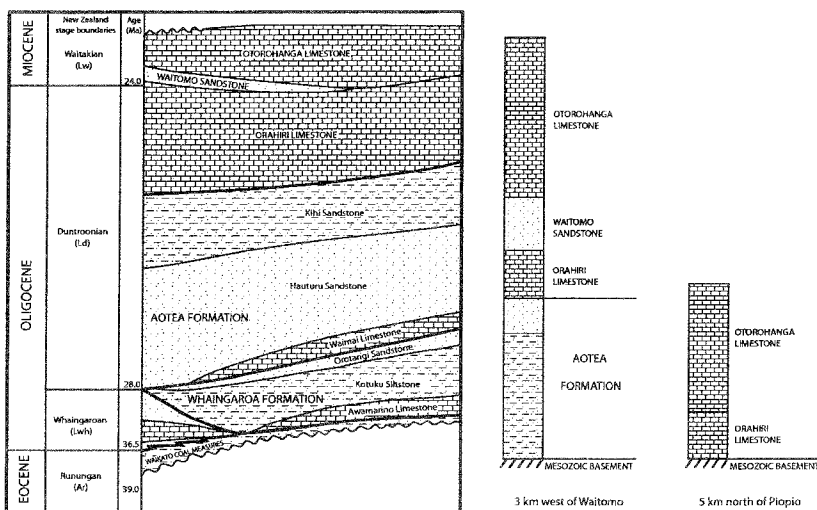


Figure 31.11 Lithological relationships in the King Country karst area around Waitomo (White and Waterhouse 1993).

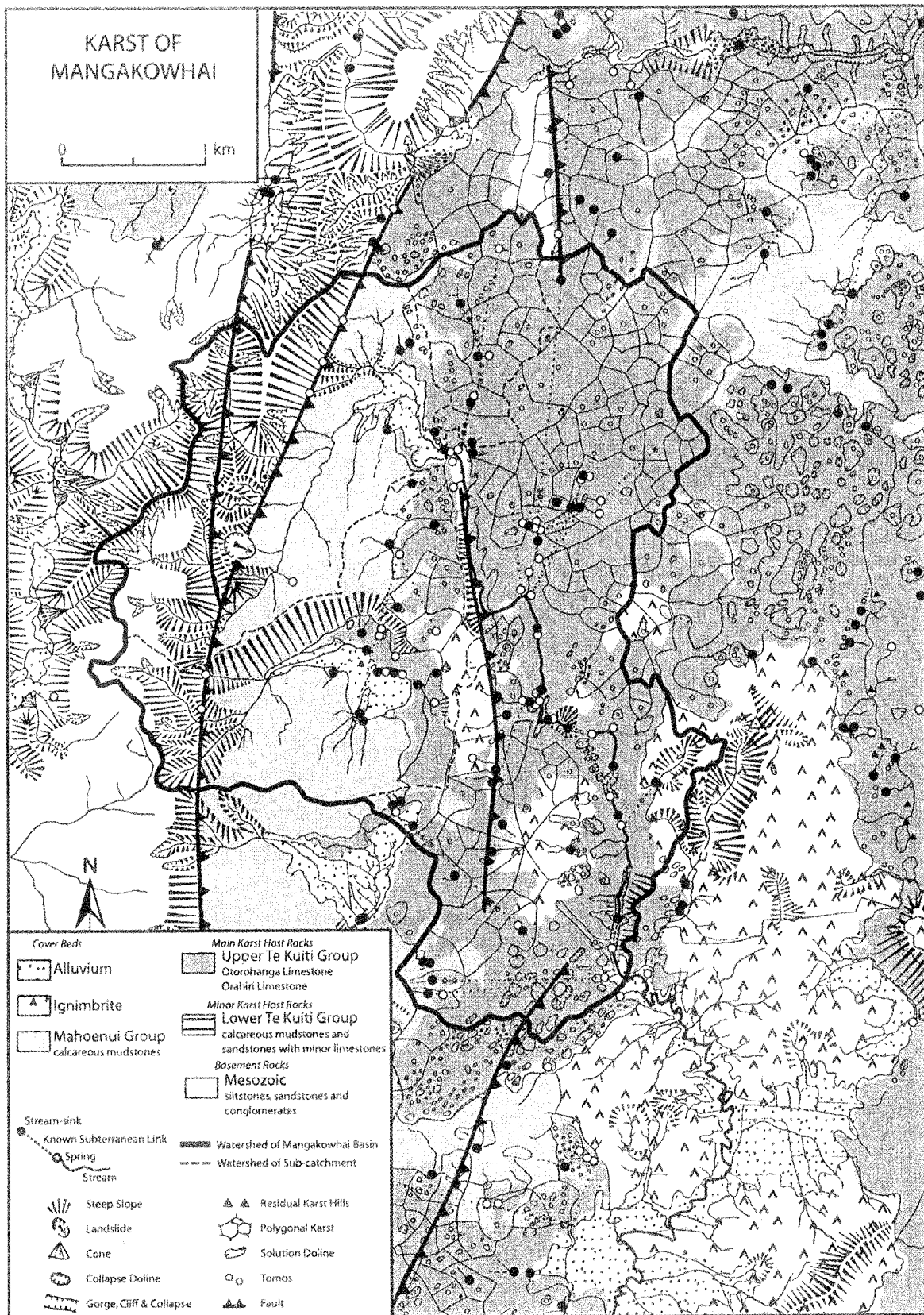


Figure 31.12 Map of karst in the Mangakowhai basin, showing lithological relationships, polygonal pattern of dolines, stream-sinks, springs and traced flow paths (mapping by Xiong Kangning).

has been the natural impoundment of karst groundwaters, although the relatively short flow through time of tracer dyes indicates that storage volumes are not large.

Punakaiki region

Relatively extensive areas of highly cavernous karst occur in the northwest South Island. Tertiary limestones, mainly of Oligocene to earliest Miocene age, are found as a discontinuous strip for over 250 km along the northwest coast, and in various places extend inland, where they have in some places been uplifted to 1000–1500 m. The limestones are quite variable in thickness and lithology, from thin and sandy to thick and crystalline. However, near Charleston and Punakaiki to the south of Westport, the carbonate sequence in places exceeds 700 m in thickness. It has been subdivided into the Waitakere Limestone, Tiropahi Limestone and Potikohuna Limestone (Nathan 1975; Laird 1988), the latter being particularly important for karst development because it is pure and dense (technically a polyzoan biosparite). Its stylobedding results in it having a platy appearance in weathered outcrop, giving rise to the Pancake Rocks at the coast (Williams 1992a). Inland from Charleston and Punakaiki the limestones underlie a plateau at 300–400 m (Fig. 31.13) that is heavily dissected by solution dolines. This gives rise to the southernmost extent of polygonal karst in New Zealand. Rainfall is copious (up to 3 m) and so there is a high density of caves (Main 1993).

The karst hydrology in the region has been studied by Crawford (1994). The area contains an excellent example of underground river capture from Bullock Creek to Cave Creek (Fig. 31.14). Bullock Creek sinks at numerous points that extend in a downstream direction as discharge increases—the combined capacity of the stream-sink zone is about 15–20 m³/s. The waters pass through the Xanadu Cave system and then re-emerge in a limestone inlier in Cave Creek gorge as a succession of overflow and underflow springs (Fig. 31.15). Their combined peak discharge is of the order of 30–40 m³/s, which indicates that the Bullock Creek losses are supplemented by about 30% extra water derived from other sources. The Cave Creek springs yield the largest flood flow from any karst system in New Zealand. During floods, the water table in

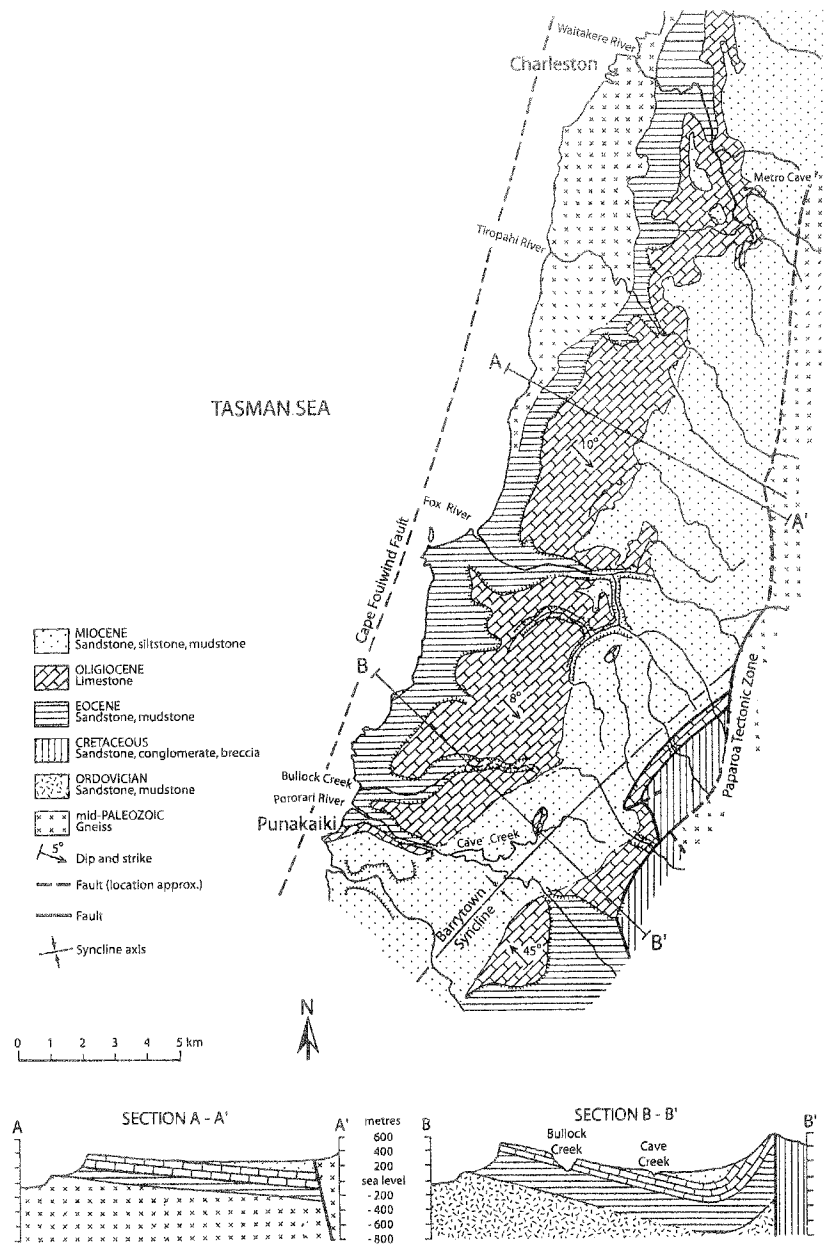


Figure 31.13 Limestone occurrences between Charleston and Punakaiki, west coast South Island (modified after Laird 1988; Nathan 1975).

the karst system can rise by up to 75 m, thus flooding most of Xanadu Cave, which is a floodwater maze cave.

The rate of development of the karst groundwater system in the region is rapid, partly because of the abundant rainfall, and partly because of the rapid uplift, which causes cave streams to incise and forces water tables to fall. The rate of downcutting of the stream flowing through Metro Cave near Charleston is about 0.27 m per thousand years (Williams 1982).

Niue Island

The aquifers found in the uplifted coral reefs of such islands as Niue, Tonga and the Cook Islands group contrast strongly with the karst aquifers described in the preceding

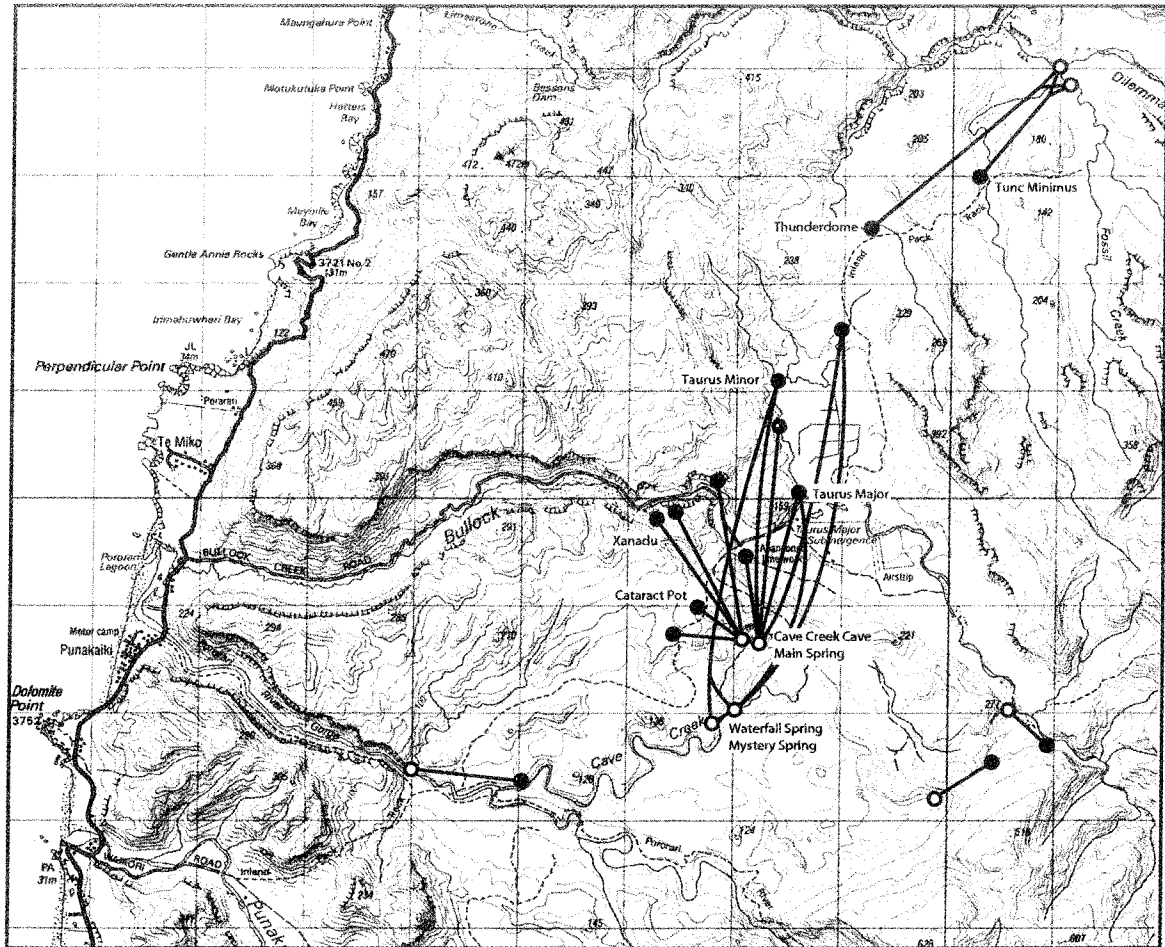


Figure 31.14 Subterranean drainage to Cave Creek springs as indicated by water tracing (from Crawford 1994).

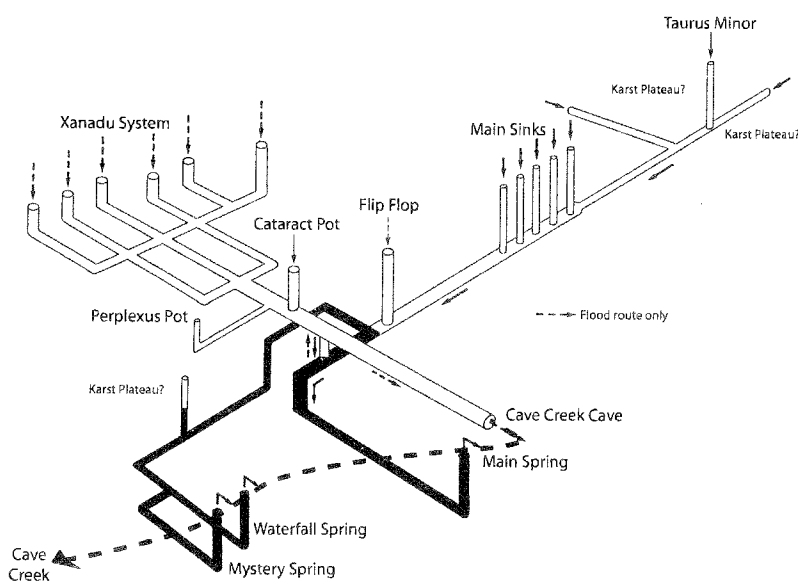


Figure 31.15 Conceptual plumbing diagram of conduits leading to Cave Creek springs. Taurus Major is the principal stream-sink of Bullock Creek. When its capacity is exceeded the excess water travels downstream and progressively overflows into the Main Sinks and then the Xanadu cave system. Still larger flows result in surface overflow down Bullock Creek gorge to the coast. Conduits in the permanently flooded part of the saturated (phreatic) zone are shown in black, although the water table can rise 75 m during floods (measured in Flip Flop Cave). The most downstream outflow point is Mystery Spring and the most upstream outflow point is Cave Creek Cave, which is normally dry. These constitute underflow and overflow parts of the discharge zone respectively. Under dry conditions, Main Spring is the most upstream of the permanent outflow points. The combined flow of the springs sustains Cave Creek, which is a tributary of the Porarari River (from Crawford 1994).

sections. They have a much higher primary porosity, but a lower secondary porosity.

Niue Island has an area of 259 km² and consists only of coral, and so it is an ideal autogenic system, being recharged entirely by rainfall that falls directly onto the carbonate outcrop. The island is an emerged atoll with a former encircling reef that encloses a now dry interior lagoon, the floor of which has been uplifted to about 60 m. The island has a comparatively simple, roughly elliptical shape and is built on a base of volcanic rocks located *ca.* 400 m below sea level. The predominantly dolomitic coral of which the island is composed is late Miocene to late Pliocene in age and the island has been emergent since the early Pleistocene (Wheeler *et al.* 1999). The geomorphic development of the island since that time has been discussed by Terry and Nunn (2003).

Niue illustrates very well the way in which fresh water, with a density of about 1.00 g/cm³, floats on salt water with a density of 1.025 g/cm³. Because of the difference in densities, for every 1 metre that the fresh-water table stands above sea level, the depth to the underlying fresh water/salt water interface is 40 m, provided the conditions are hydrostatic, i.e., there is no circulation. However, recharge generates flow and circulation, so that the depth to the fresh water/salt water interface is rather more than 40 times the height of the fresh water above sea level.

This phenomenon—fresh water lying above salty water—was first investigated at the turn of this century by two European scientists, Ghyben and Herzberg, whose names are now lent to the principle that they discovered. The Ghyben-Herzberg principle of gravitational equilibrium oversimplifies the relationship usually found in nature, because the two fluids are treated as immiscible, while groundwater conditions are normally dynamic rather than static. It was Hubbert (1940) who showed that the interface is deeper under dynamic conditions than under static conditions. The position of the salt water/fresh water interface is now estimated by combining assumptions of essentially horizontal flow (termed Dupuit approximations) with gravitational equilibrium, in so-called Dupuit-Ghyben-Herzberg analysis (Bear 1972).

An important outcome of the hydrodynamic balance of fresh water above underlying sea water is that the water table rises gently towards the interior of an island, thus producing a fresh-water dome. This generates a concomitant effect of the fresh/salt-water interface descending steeply downwards, and reaching its greatest depth beneath the place where the water table is highest. This is usually (but not always) in the middle of the island. The fresh-water body in an ideal circular island is therefore lens-shaped. However, Vacher (1988) has provided an authoritative discussion of the associated hydrodynamics and explains why groundwater lenses beneath islands can vary from their idealised lenticular form. Vacher and Quinn

(1997) provide many case studies of groundwater conditions under carbonate islands.

Niue Island has numerous boreholes drilled to extract groundwater, and it has many caves, especially near the coast, which can be explored down to the water table. This has allowed the elevation of the water table above sea level to be measured at 25 locations (Fig. 31.16). The shape of the water table is not a simple dome rising towards the centre of the island (which is lower than the rim). Instead it consists of a discontinuous string of elliptical domes around the outside of the island, beneath the highest land that was once the atoll that enclosed the lagoon. The water table attains 3 m or so above sea level on these domes, whereas in the centre of the island it reaches only 1.6 m above sea level. There are low spots between the encircling domes that provide avenues for water from the interior to escape to the coast. Springs in the intertidal zone on the coast are sometimes located opposite these gaps (Fig. 31.16).

The fresh/salt-water interface at depth should have a form that is an exaggerated inversion of the topography of the water table. On Niue this form was demonstrated by Jacobson and Hill (1980), who mapped the interface using electrical resistivity. The position of the interface can also be determined directly by lowering an electrical conductivity probe down deep boreholes. The conductivity values show the boundary between the fresh and salt waters to be a zone of transition, not a sharp interface.

A critical point for water extraction on islands (or in any coastal aquifer) is that salt water underlies the fresh water. So if the bottom of the borehole is above sea level, salt water can never enter the bore and contaminate it. Water supply bores may be safely drilled below sea level only if the water table at that point is sufficiently far above sea level that the fresh water/salt water interface remains well below the bottom of the bore. However, if water is pumped from a bore to the extent that the water table is drawn down around the bore, then the underlying fresh/salt water interface will rise by 40 times that fall. As a result of this upward-coning, sea water may invade the bore and contaminate the fresh water supply. Cessation of pumping will allow the water table to return to its natural level and so force the interface downwards again, but in the meantime damage may have been done if salty water was introduced into domestic and industrial supplies or used to irrigate crops.

CONCLUSION

Karst groundwaters in the New Zealand region cover a wide range of types, from highly porous coral aquifers to conduit aquifers in marble with negligible primary porosity. The widespread aquifers in thinly-bedded Oligocene-Miocene limestones are intermediate in nature,

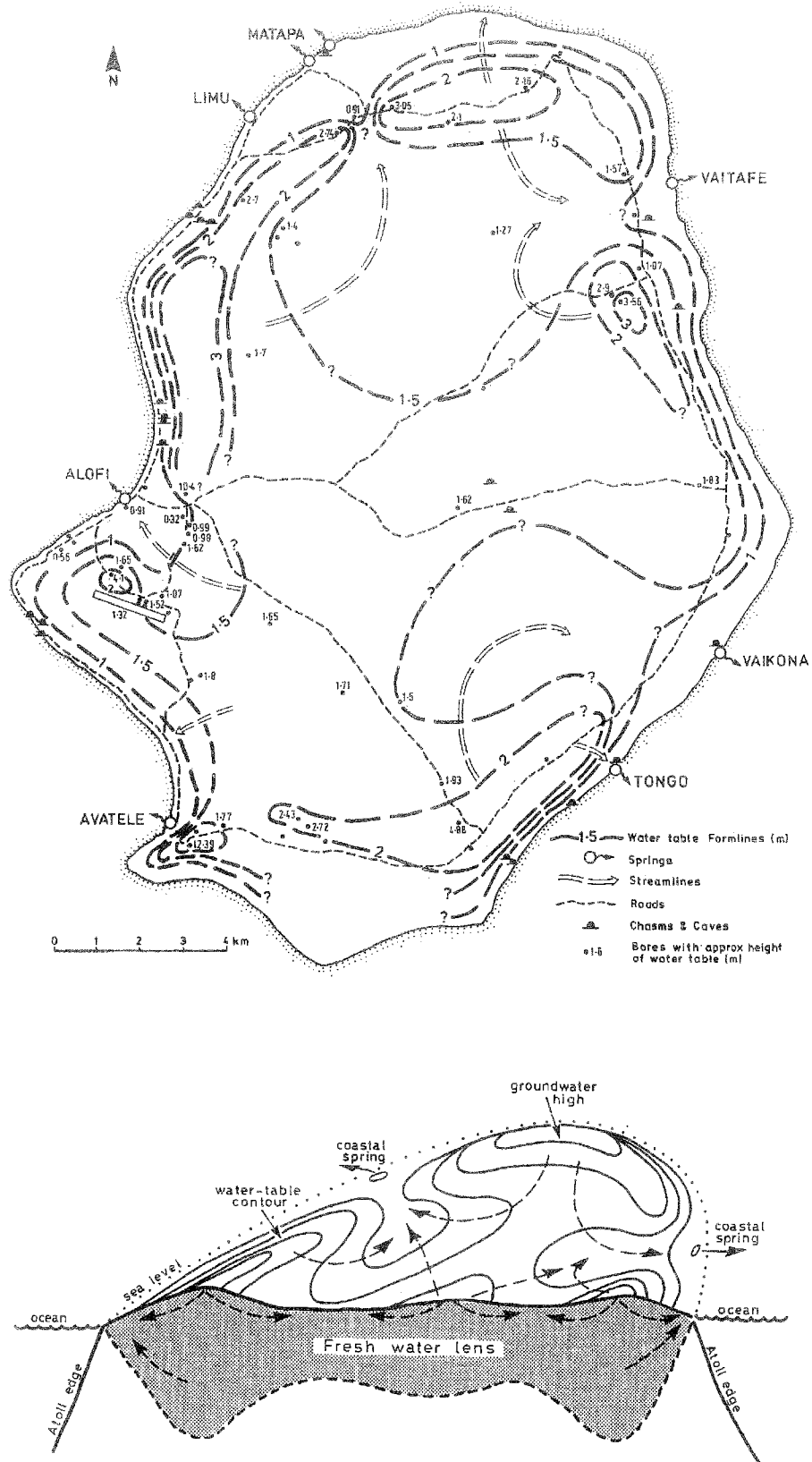


Figure 31.16 Map of the water table and section of the fresh-water lens for Niue Island, south Pacific. Note the relationship of the topography on the water table to the depth of the fresh water/salt water interface (from Williams 1992b).

but tend to be well karstified, with highly developed systems of conduits.

Flow velocities through these aquifers vary by several orders of magnitude. In the coral of Niue, flow rates are usually about 0.5–1 m per day, but exceptionally reach 300 m/day. A similar range of flow rates occurs in the Waikoropupu aquifer of the Takaka valley. But in areas of high relief, where there is greater fall from the recharge zone to springs, flow rates can be much faster, exceeding 2 km/day in the marble karsts of Mt. Arthur and Mt. Owen, and in the limestone karsts of the King Country and Paparoa National Park. Karst springs may have mean discharges of up to 15 m³/s and flood flow discharges of up to about 40 m³/s.

Rapid transit times associated with recharge by sinking streams introduces pollutants very speedily into karst aquifers, which are therefore very vulnerable to pollution. The large size of karst conduits also means that physical filtration is minimal. Hence although karst springs may look attractive, there are few in New Zealand whose waters are safe to drink without first taking the precaution of boiling.

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Chapter 30

Biota of cold-water and geothermal springs

Russell Death, José Barquín and Mike Scarsbrook

INTRODUCTION

Springs are formed where the water table intersects with the earth's surface, or groundwater rises to the surface through rock faults, fractures or depressions. Springs are formally defined as points of natural, concentrated discharge of groundwater at a rate high enough to maintain flow on the surface (Van Everdingen 1991). They vary greatly in morphology and size, ranging from minor seepages in alluvial deposits, to resurgences in karst (e.g., Riwaka River, near Motueka), up to very large vents discharging thousands of litres per second (e.g., Waikoropupu Springs, Takaka). These outcrops of groundwater create semi-insular freshwater habitats that can maintain isolated and unique faunas. The springs of Death Valley in the United States, for example, are home to several species of pupfish found nowhere else (Duvernell and Turner 1998). Organisms restricted to springs are termed crenobionts. In New Zealand, possible crenobionts include the mayfly *Zephlebia nebulosa*, the isopod *Notamphisopus*, several caddisflies (e.g., *Pseudoeconesus*) and an apparently large number of undescribed species of hydrobiid snails. Spring faunas may also include taxa that are typically more groundwater dwellers (e.g., the amphipod *Paraleptamphopus subterraneus*). Biodiversity in New Zealand springs can often be considerably higher than in similar rivers and streams fed by run-off (Death 1995; Death and Winterbourn 1995).

Springs also differ from other water bodies in having a very uniform annual temperature. Slip Spring in the Southern Alps, for example, maintains a water temperature of 8°C throughout the year (Death and Winterbourn 1994) and Waikoropupu Spring, Takaka, a temperature of 11.7°C (Michaelis 1976). Depending on the underlying geology, springs may also have a relatively constant flow regime, a factor that also distinguishes them from other New Zealand running waters. Based on the body of water

into which they flow, there are three broad types of spring:

1. rheocrene—the spring discharges directly into a stream channel,
2. limnocrene—the spring discharges into a pond or lake,
3. helocrene—the spring discharges into a swamp or marsh.

Water from aquifers that penetrate deep into the earth's crust can be heated, and the resulting geothermal springs can have water temperatures that exceed 85°C where they emerge from the ground. At such temperatures the only biota that can survive are thermophilic bacteria and fungi. As temperatures decrease away from the source, however, blue-green algae and invertebrates are able to inhabit the water, and hot springs may exhibit distinctive thermal zonation patterns. Major areas of geothermal activity are found in New Zealand, Iceland, Japan, Russia, India and the Yellowstone region of the United States, although isolated geothermal springs do occur elsewhere in the world.

In this chapter we discuss the characteristics of both cold-water springs (with a mean temperature of less than 20°C) and geothermal springs. However, as their general environments and biotas differ considerably, we will describe them separately. We have tried to focus on springs rather than spring brooks (i.e., streams arising from springs), although exactly where a spring ends and a spring brook begins is unclear. A number of the studies we discuss include both springs and spring brooks, however we focus predominately on information about the springs in these studies. Our focus is on the unique biota of spring habitats, but in attempting to describe and explain observed patterns in the biota, it is first necessary to describe the thermal, chemical, physical and hydrological characteristics that influence the biota of springs. Therefore for each spring type (cold-water and geothermal) we review their physical and chemical characteristics, discuss the

biota observed in New Zealand springs and note how they differ from similar springs elsewhere in the world. We also look at the conservation and biodiversity values of springs and discuss how those values may be threatened by human activities.

COLD-WATER SPRINGS

Introduction

Cold-water springs are widespread in New Zealand. The type of spring (i.e., rheocrene, helocrene, and limnocrene), the volume and constancy of flow, temperature regime and chemistry of the water are all highly dependent on the underlying geology and aquifer type. In contrast to many other countries, there have been comparatively few studies of the biota of cold-water springs in New Zealand. Michaelis (1976, 1977) conducted an extensive study of the flora and fauna of one of the world's largest cold-water springs, Waikoropupu Spring, Takaka, which also has some of the clearest water in the world (Davies-Colley and Smith 1995). Several springs around Cass in Canterbury have also been examined for bryophyte and invertebrate associations (Cowie and Winterbourn 1979), and as part of a wider investigation into the role of habitat stability in determining invertebrate community structure (Death 1995; Death and Winterbourn 1994, 1995). Michaelis (1977) examined the flora and fauna of several other springs throughout New Zealand, although her sampling effort was semi-quantitative and varied among springs. Therefore, as part of our review of cold-water springs in New Zealand we present some new data from a quantitative survey of relatively pristine, predominantly upland springs in four regions: Arthurs Pass, Nelson, Ruapehu and Taranaki (Barquín and Death in prep.) and from a semi-quantitative study of lowland springs in agricultural land in Taranaki, Waikato, Waitaki and Southland (Scarsbrook and Haase 2003) (Fig. 30.1).

Physical and chemical characteristics

Cold-water springs often are characterised by a very constant thermal regime. As mentioned earlier, Slip Spring, Canterbury and Waikoropupu Spring, Takaka have almost constant annual water temperatures of 8 and 11.7°C (varying by less than $\pm 1^\circ\text{C}$), respectively (Death and Winterbourn 1994; Michaelis 1976). Summer water temperatures in our wider survey of some New Zealand springs were between 6°C and 12°C, and were often cooler than similar-sized streams in the area, with a daily fluctuation of no more than 1.6°C (Table 1).

Springs range in discharge from mere trickles to an average of 11 m³/s (8.5–15.0 m³/s range) in Waikoropupu Spring, one of the largest springs in the world (Michaelis 1976, 1977) (for more detail on flow regimes see Chapter 3). The size and type of the underlying aquifer

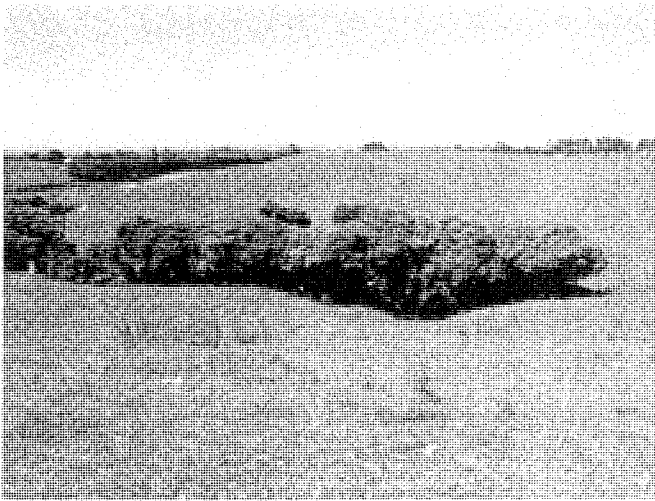
affects the capability of springs to resist drought or experience increases in discharge (Danks and Williams 1991). For example, springs in karst regions, such as Nelson, are more prone to increases in discharge during high rainfall, because of infiltration of surface water close to the source (Bonacci 1987; Barquín and Death in prep.). In contrast, some springs near Arthurs Pass and Cass seem to have relatively constant flows independent of the prevailing weather patterns (Death and Winterbourn 1994). As in rivers and streams differences in the water chemistry of springs can be attributed to differences in their underlying geology and in land use (Table 30.1). For example, springs emanating from carbonate rock generally have a high conductivity (e.g., Fish Creek, Takaka), particularly where the groundwaters have been in long contact with highly soluble rock. Petty (1972) provides considerable detail on the water chemistry of springs in the northern part of New Zealand.

Characteristic flora

As a result of the constant or near constant flow, and to a lesser extent temperature, the substrata of cold-water springs are often heavily overgrown with macrophytes and/or bryophytes. Michaelis (1976, 1977) recorded 5 species of angiosperm and 10 species of bryophyte from Waikoropupu Spring, and Cowie and Winterbourn (1979) recorded 3 species of zone-forming bryophyte from the much smaller Middle Bush Spring. Large open springs (e.g., Waikoropupu), and springs in agricultural land, where the overhanging forest canopy has been removed, are often heavily overgrown with the introduced watercress species *Nasturtium microphyllum* and *N. officinale* (Marshall 1973). This plant is used throughout New Zealand as a food resource. For example, watercress from Waikoropupu Springs is harvested weekly, to both provide a highly desirable product and to control its growth.

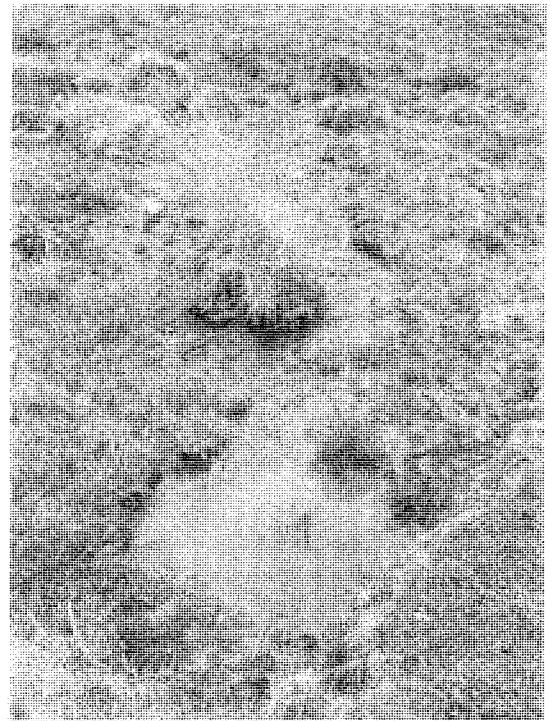
Invertebrate fauna

Many of the invertebrates found in cold-water springs are widely distributed in rivers and streams throughout New Zealand and are generally not restricted to springs. However, some occur mainly on the bryophytes and macrophytes that are common in springs (e.g., some caddisflies such as *Zelotes* spp.—Michaelis 1977; Cowley 1978). In New Zealand, the number of taxa known to be restricted to springs (crenobionts) is low by comparison with world standards, although this is partly a result of the limited number of studies on the biota of these habitats. For example, studies by Dr Martin Haase (NIWA Hamilton) have shown an astonishing diversity of hydrobiid snails in springs and seepages throughout New Zealand. So far more than 30 previously unknown species have been collected.



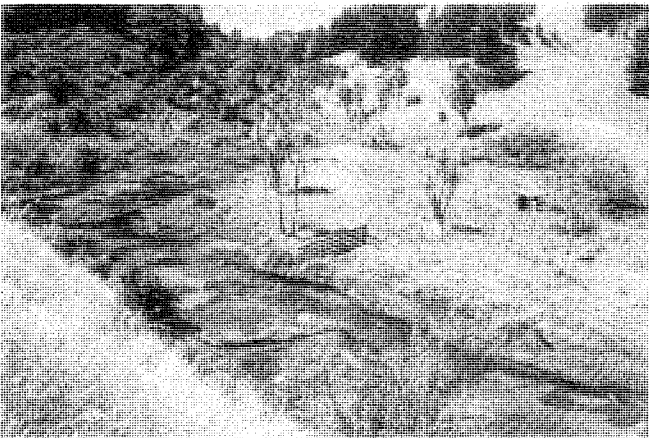
A

Photo: Mike Scarsbrook



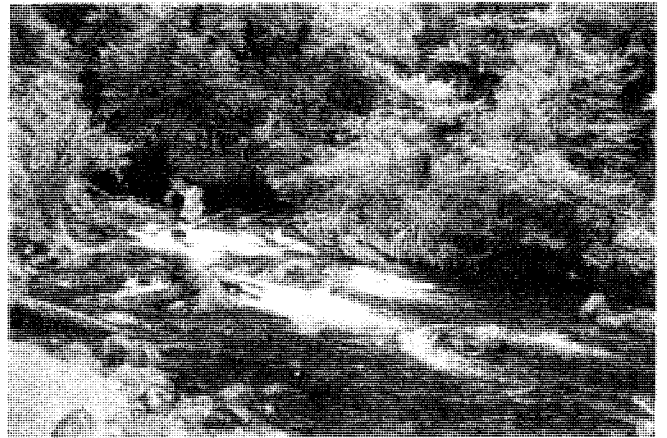
B

Photo: Mike Scarsbrook



C

Photo: Pepe Barquín



D

Photo: Pepe Barquín

Figure 30.1 Coldwater springs. A) lowland spring, Taranaki, B) lowland spring, Southland, C) Blue Spring, Waihou River, Putaruru, D) Spring source, Pearse River, Nelson.

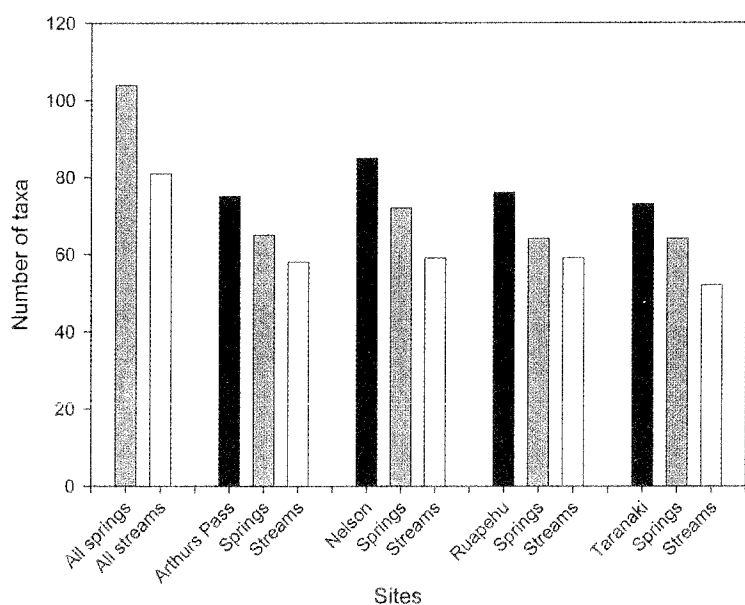
The springs in many upland areas often have a fauna in which insects are the most diverse and abundant group, although Mollusca can dominate numerically in a few (e.g., several springs around Cass). Common insect taxa in springs include Diptera (predominately Chironomidae), Trichoptera (particularly cased caddis) and Ephemeroptera (Michaelis 1977; Cowie and Winterbourn 1979; Barquín and Death in prep). Of 104 taxa collected in a regional survey by Barquín and Death (pers comm) only 12 invertebrate taxa were not insects: these latter included Mollusca, Crustacea and Platyhelminthes. Trichoptera was the most diverse order, with 25 taxa collected in the Nelson

springs, and Diptera the next most diverse, with 18 taxa collected in the Nelson and Ruapehu springs. Coleoptera, Ephemeroptera and Plecoptera had between 4 and 10 taxa per region. Spring habitats as a whole, and in each of the regions, had a higher number of taxa than nearby streams of similar size (Fig. 30.2), although the number of taxa in spring and stream habitats from each of the four regions was similar.

In contrast to many upland springs, non-insects (Mollusca, Amphipoda and Isopoda) can form the dominant groups in lowland springs (Scarsbrook and Haase 2003). Amphipoda and the mollusc *Potamopyrgus*

Table 30.1 Physical and chemical characteristics of 19 New Zealand coldwater springs measured in autumn 2000 and summer 2001. Ck = Creek

Region	Spring Name	Site	Altitude (m)	Nitrates (mg/L)	Phosphates (mg/L)	Conductivity ($\mu\text{S/cm}$)	pH	Temperature ($^{\circ}\text{C}$)	Depth (cm)	Velocity (m/s)	Flow (m^3/s)
Arthurs Pass	Slip	A1	800	0.08	0.09	62	7.3	10.1	17	0.45	0.07
	Slip 2	A2	750	0.03	0.04	43.7	7.3	10.1	12	0.93	0.15
	Cass	A3	570	0.13	0.04	46.1	7.6	11.8	3	0.04	0.01
	Hawdon 1	A4	750	0.06	0.11	64	7.4	11	11	0.22	0.07
	Hawdon 2	A5	750	0.07	0.14	62.5	7.3	10.7	12	0.21	0.16
Nelson	Blue Ck	N1	400	0.1	0.24	114.6	8.6	6.7	18	0.61	0.01
	Pearse	N2	380	0.08	0.12	235	7.9	7.6	18	0.39	0.28
	Riwaka	N3	120	0.22	0.05	249	8.3	10.6	9	0.61	0.29
	Fish Ck	N4	17	0.41	0.06	331	8.3	12.3	15	0.04	0.03
	Gorge Ck	N5	520	0.23	0.12	146.7	8.5	9.8	22	0.78	0.38
	Bulmer Ck	N6	550	0.09	0.25	166	8.6	6.1	14	0.79	0.15
Ruapehu	Waihohonu 1	R1	1100	0.1	0.21	235	6.5	6.9	22	0.8	0.27
	Waihohonu 2	R2	1050	0.09	0.44	181	7.9	7.6	19	0.44	0.17
	Ohinepango	R3	1000	0.1	0.31	64	8.6	6.5	69	0.77	0.93
	Waitaiki	R4	690	0.07	0.29	111	6.8	7.7	23	0.01	0.31
	Waihoruru	R5	700	0.06	0.25	118.6	8.7	10.3	26	0.44	0.24
Taranaki	Taungatara	T1	381	0.16	0.55	112	7.8	7.7	25	0.22	0.43
	Unnamed	T2	353	0.05	0.43	123.8	7.9	7.7	11	0.49	0.06
	Bubling	T3	950	0.07	0.33	93	8.3	5.9	21	0.48	0.08

**Figure 30.2** Number of taxa collected in springs (grey bars), streams (open bars) and combined (black bars) for four regions of New Zealand between 2000 and 2001.

antipodarum can be extremely abundant in some springs, with densities of the latter reaching 180,000 per m^2 in the bryophytes of Waikoropupu Spring (Michaelis 1977). Taxa that are more typically groundwater dwellers, such as the amphipod *Paraleptamphopus subterraneus*, may also be as important a component of the spring fauna as surface forms.

In our regional survey of upland springs we found that the presence or absence of particular taxa had a regional pattern (Barquín and Death in prep). Of the spring-dwelling chironomids, *Maoridiamesa* spp. and *Paratrichocladius pluriserialis* dominated at Arthurs Pass; *Naonella forsythi*, *Eukiefferella* and *Maoridiamesa* dominated in Nelson; *Eukiefferella* and *Maoridiamesa* were most abundant at Ruapehu and no chironomid taxon was abundant in Taranaki springs (Fig. 30.3). Of the cased caddisflies, *Pycnocentria hawdonia* and *Pycnocentroides* sp. were abundant in Arthurs Pass springs; *Pycnocentria mordax*, *Pycnocentria evecta* and *Zelolessica meizon* were common in Nelson springs; *Zelolessica cheira* and *Pycnocentria funerea* were abundant in Ruapehu springs and *Pycnocentria funerea* in Taranaki springs. The ubiquitous *Deleatidium* was also abundant in Arthurs Pass, Nelson and Taranaki springs. The mayflies *Austroclima jollyae* in Nelson and *Coloburiscus humeralis* in Taranaki were equally as abundant as *Deleatidium* at some sites. Finally, Coleoptera, mainly Elmidae and Scirtidae, were common in Taranaki springs. Although many of the common taxa in springs are similar to those in streams in the same region, differences in relative abundance and the numerically dominate taxa do yield distinctly different communities in streams and springs (Death 1995; Barquín and Death in prep.). Among regions, the invertebrate communities

of springs in Arthurs Pass and Ruapehu, and Nelson and Taranaki, are most similar to each other (Fig. 30.4). This may, at least in part, reflect the higher altitude or lack of canopy cover in the springs of Arthurs Pass and Ruapehu.

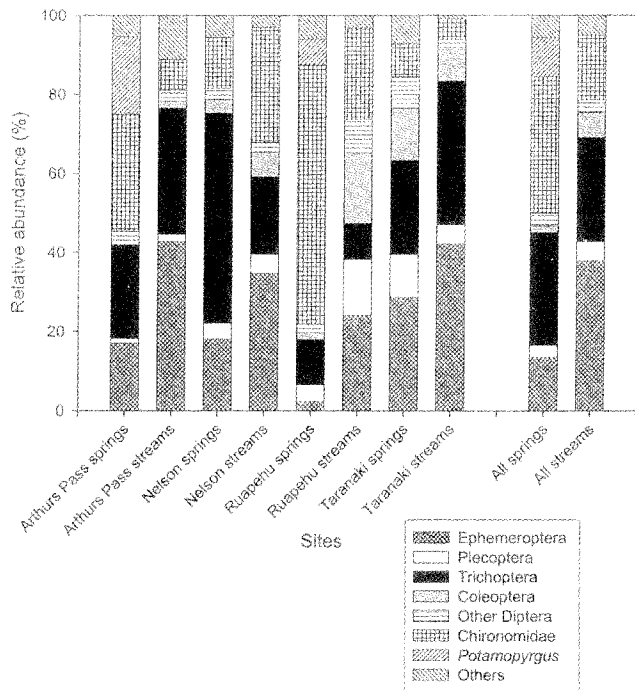


Figure 30.3 Relative abundance of taxa collected from upland spring and stream habitats in four regions of New Zealand, between 2001 and 2002.

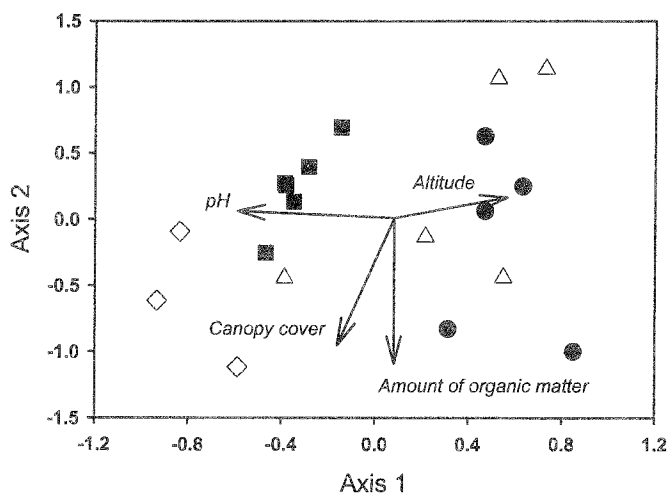


Figure 30.4 Multidimensional scaling ordination plot for New Zealand upland spring invertebrate communities collected in four regions ● = Arthurs Pass, ■ = Nelson, ▲ = Ruapehu and ◊ = Taranaki. Arrows indicate changes in spring characteristics that are associated with the changes in community structure, e.g., low altitude sites on the left and high altitude sites on the right.

New Zealand's spring fauna: a global context

In a review of the literature on North American temperate cold springs, Glazier (1991) identified two broad types of community, those dominated by peracaridans (amphipods and isopods), molluscs and triclads, and those dominated by insects. In general, the former communities occurred in hard-water limestone springs and the latter in acidic soft-water springs. The lack of non-insect taxa in acidic soft-water springs may be explained by the potentially adverse effect of low pH and alkalinity on crustaceans and molluscs. However, the dominance of non-insects in hard-water springs is more difficult to explain. Glazier suggested that physical stability, small habitat area, and lack of large predators in springs may all favour peracaridans, molluscs and triclads. Alternatively, Williams and Williams (1998) suggested that the dominance of non-insect taxa in some spring systems in the Northern Hemisphere is related to glaciation history. Their hypothesis is that more mobile insects predominate in regions that have been subject to recent glacial activity, whereas spring systems not subjected to such disturbance are dominated by organisms with limited dispersal, such as amphipods and molluscs, which may be competitively dominant. We have the opportunity in New Zealand to test this hypothesis with respect to both glaciation and volcanism. The new data presented in this chapter also illustrate a dichotomy in community types between non-insect and insect-dominated faunas (Barquín and Death in prep.; Scarsbrook and Haase 2003) although some differences in community structure in New Zealand may be related to altitude.

The recorded number of taxa in New Zealand springs ranges between about 20 and 50 and is similar to the richness recorded by Sloan (1956) in a Florida spring system (16–32 taxa), Ward and Dufford (1979) in a Colorado spring (36–43 taxa) and Webb *et al.* (1998) in 10 Illinois springs (18–64 taxa). However, the maximum recorded diversity in New Zealand falls well short of that in an Italian helocrene spring where 113 invertebrate taxa were found (Cantonati and Ortler 1998), or a Canadian rheocrene spring with 60 taxa (Williams and Hogg 1988), although the sampling intensity was greater in these studies than in ours. The maximum number of chironomid species recorded in New Zealand springs was seven, much lower than the 38 recorded by Blackwood *et al.* (1995) in springs of the Central Plains of the USA or the 66 recorded by Ferrington *et al.* (1995) in Kansas springs. New Zealand springs had between 4 and 19 trichopteran taxa—higher than the 8 taxa per spring reported by Erman and Erman (1995) in springs in California. As mentioned, New Zealand springs seem to have a higher diversity than similar-sized non-spring-fed streams (Fig. 30.2), a pattern that contrasts with that found

elsewhere in the world, where springs tend to have a lower diversity than similar streams (Meffe and Marsh 1983; Anderson and Anderson 1995).

Conservation and biodiversity values of New Zealand springs

Springs form an important transition zone between groundwaters and surface waters, and also interact strongly with terrestrial environments. Because of this overlap of ecosystems, springs often contain a mix of species from all three environments, as well as obligate (i.e., those able to survive only in a particular habitat) spring species (crenobionts). The diverse and distinctive flora and fauna found in springs make them an important feature of aquatic ecosystems, but a feature that often has little protection.

Permanent cold-water springs are habitats characterised by low disturbance (Williams 1991). This environmental stability, and the position of springs at the interface between several distinct ecosystems, has led ecologists to suggest that springs may be "hot spots" for aquatic biodiversity. A good example of the important contribution of springs to aquatic (and terrestrial) biodiversity is illustrated by work on mound springs in arid areas of inland Australia (Knott and Jasinska 1998). Mound springs are points of groundwater discharge that are elevated above the surrounding landscape through the build-up of calcarenites and/or peat. These small mesic refuges in arid landscapes have been critical to the survival of aboriginal peoples, European explorers and settlers, and their livestock. They have a distinctive flora and fauna (Knott and Jasinska 1998) and have high diversity and endemism among animals with restricted powers of dispersal (e.g., hydrobiid snails, amphipods and fish). The mound springs of western Queensland (Ponder and Clark 1990) and northern South Australia (Ponder *et al.* 1989) have been the sites of an impressive radiation (i.e., evolution of species) in hydrobiid snails, with 22 species described to date. Conservation of these systems depends on two factors: the reliability of the water supply and minimisation of damage to the springs themselves and their associated microhabitats (Knott and Jasinska 1998).

Hydrobiid snails show extensive radiation in some New Zealand spring habitats, particularly in karst areas (M. Haase, pers. comm.).

This group of snails is expected to show a high degree of endemism, as this is a characteristic of the group worldwide (e.g., Ponder and Clark 1990; Haase and Bouchet 1998), and provides a major challenge with respect to the conservation of these snails, since it is impracticable to protect all spring habitats within the landscape.

Another group with a high level of diversity in New Zealand lowland springs is the pericarid Crustacea (amphipods and isopods). Dr Graham Fenwick (NIWA, Christchurch) identified thirteen morphologically distinct forms of "*Paraleptamphopus*" from 34 springs in Waikato, Taranaki, Waitaki Valley and Southland (Fig. 30.5) (Scarsbrook and Haase 2003). These "morphospecies" ranged from those with strong pigmentation and distinct eyespots (epigean, or surface forms), to unpigmented and eyeless forms (hypogean, or subsurface forms). In addition to their high diversity, there may also be a high level of endemism, as six of the 13 taxa were found at single locations (Fig. 30.5). Eight of the paraleptamphopiids were restricted to samples from Southland. These Southland springs also contained specimens of the phreatoicid isopod *Notamphisopus*, which is known only from the southern South Island and Stewart Island (Chapman and Lewis 1976). Initial descriptions of the six known species in the latter genus suggest that they may be spring or seepage specialists (Nicholls 1944). Phreatoicids are also common in some Australian spring habitats (Knott and Jasinska 1998).

Several authors have suggested that springs play a role as refuges in the protection and conservation of aquatic biodiversity. For example, Van Everdingen (1991) suggested that springs provide a thermal refuge in arctic areas, by providing free-flowing water throughout the year.

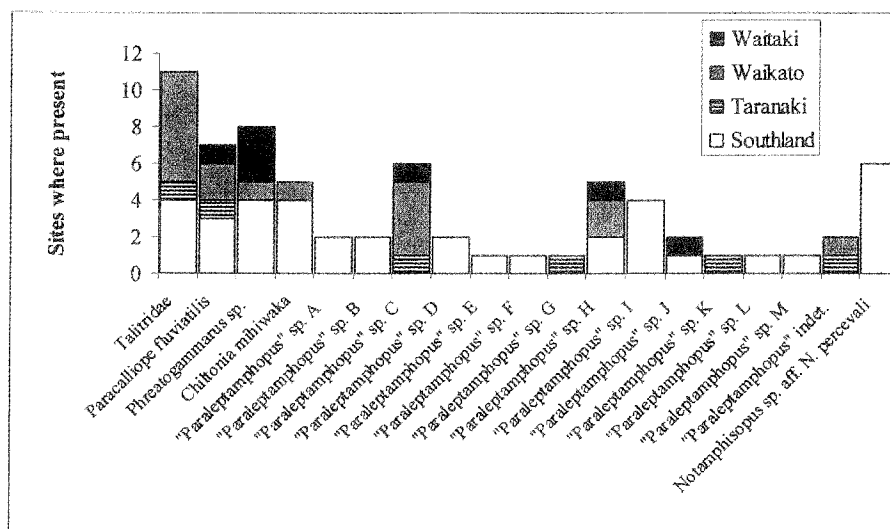


Figure 30.5 Frequency of occurrence of macrocrustacea (Superorder Pericarida) at 34 spring sites from four regions around New Zealand. Reproduced from Scarsbrook and Haase (2003).

In New Zealand Michaelis (1973) attributed the isolated populations of *Rakiura vernale* in Waikoropupu Spring, (a caddisfly generally restricted to the far south of New Zealand) to the cool water temperatures in the spring. Mosley (1983) suggested the cooler temperatures of seepage and spring flow in braided river landscapes provide a thermal refuge for various fish species during periods of high water temperature during summer. The constancy of flow regimes in springs and spring-fed streams may also provide refugia from flood disturbance (Death 1995).

Springs also contribute to functional diversity in aquatic ecosystems. For example, Digby (1999) found that levels of secondary production in seepage habitats in the Rakaia River were an order of magnitude higher than in main braid, or side braid habitats. These seepage habitats contain relatively high numbers of invertebrate species, provide habitat for native fish and are important foraging areas for native wading birds (Hughey *et al.* 1989; Hughey 1998).

Conservation of spring habitats in New Zealand is relatively limited. Many of our larger springs are protected to varying degrees. Some larger springs (e.g., Waikoropupu Springs and Hamurana Spring, Rotorua) and their surrounding land are owned by the New Zealand government, and protected by the Department of Conservation through a strict code of practise for visitors (e.g., recreational divers). Other springs are protected by virtue of their use for water supply (e.g., Blue Spring, Waihou River). Many small springs are located on conservation estate lands, and are protected to some extent, although not necessarily from the effects of feral animals (e.g., deer wallowing). This leaves a large number of spring habitats around the country that are unprotected. In particular, lowland springs are poorly represented in the conservation estate.

Anthropogenic threats to New Zealand springs

Springs are highly localised and thus vulnerable to a range of anthropogenic impacts (Zollhöfer 1999; Smith 2002). Threats from human activity can include reduction in water quantity, changes in water quality, and changes in physical habitat (Table 30.2).

From a biological perspective, the most important requirement of the spring fauna is the persistence of flowing water (Danks and Williams 1991; van der Kamp 1995). In areas of intensive groundwater use, regional lowering of the water table may occur to the point where groundwater discharge occurs only via wells, and all springs and seeps disappear entirely. Urbanisation may also lower the water table—the source springs of the Avon River in Christchurch, for example, have moved steadily downstream as land development has expanded (Marshall 1973). Conversely, new springs may occur if water tables are raised (e.g., by reservoirs, extensive irrigation, or canal

Table 30.2 Potential threats to health and integrity of springs

Threat to spring	Human activities
Reducing water quantity	Groundwater abstraction River regulation Afforestation (increased evapotranspiration) Land drainage and urbanisation Impervious surfaces
Changes in water quality	Fertilisers Agrochemicals Septic tanks Landfill leachate Animal waste Mining Urban/road runoff
Changes in physical habitat	Land drainage Stock trampling/grazing Clearance of riparian vegetation Capture of springs for water supply

construction). Spring habitats may also disappear where the spring source is captured for water supply. Smith (2002) noted that the capture of springs for bottled water operations is a significant threat to spring habitats worldwide and is becoming increasingly common in New Zealand. Problems of flow permanence are likely to be important in lowland areas of New Zealand, particularly where there is intensive use of groundwater for irrigation. Abstraction of water from rivers may also affect spring flows, particularly where aquifer recharge is dominated by rivers. Spring habitats within braided river landscapes may be particularly susceptible to lowering river flows, and this may have important consequences for ecosystem function in braided rivers (Digby 1999). We currently have no detailed information on the effects of flow permanence on spring flora and fauna in New Zealand.

Because the source of springs is groundwater, spring water quality is obviously affected by activities that affect groundwater quality (see Close *et al.* 2001 for review). Contaminants from landfill leachate, pesticide usage on crops and fertiliser application all have the potential to degrade groundwater quality, and springs will reflect this contamination. It has been suggested that groundwater invertebrates (e.g., *Paraleptamphopus subterraneus*) could be used as biological indicators of groundwater quality (Malard *et al.* 1996; Chapter 29 this volume). The difficulty with groundwater biomonitoring is that sampling can be extremely expensive. In contrast, the fauna of springs are relatively cheap and easy to sample. Therefore, the fauna of springs may have potential as

relatively inexpensive indicators of groundwater quality (Williams and Danks 1991). Indeed, the integrity of springs and their fauna may act as indicators of both the sustainability of groundwater abstraction (i.e., assessments of flow permanence) and groundwater quality.

Changes in spring habitat form one of the most pervasive threats to the flora and fauna of springs. Most springs are small, covering no more than a few square meters and are thus highly susceptible to habitat destruction. Few are protected, and all but one of the twelve spring-associated hydrobiid snail species described from artesian springs in Queensland by Ponder and Clark (1990) were considered to be endangered because the springs had no conservation status and were threatened by pastoral activities and water extraction from the parent aquifers. Ponder and Clark (1990) suggested that the greatest threats to springs came from the activities of land managers (e.g., damming, digging out, over-pumping of aquifers), whereas threats posed by access to stock were less serious and more likely to be of a short-term nature. In New Zealand much of our lowland and hill country is used for pastoral farming, a land use that reaches peak intensity in lowland areas, particularly on river floodplains, where spring habitats are directly threatened.

Scarsbrook and Haase (2003) recently completed a survey of 34 lowland spring and seepage habitats in predominantly dairying areas around New Zealand. They focussed on small springs, with the majority having a discharge of <10 l/s. Figure 30.1A shows a small spring arising from limestone hills in central Southland. The damage from stock is obvious upon visiting the spring. Surprisingly this spring contained two new species of hydrobiid snail, and several species of the "*Paraleptamphopus*" group.

A comparison of community composition across the 34 springs indicated that riparian conditions at the spring head may be an important determinant of spring community composition in lowland areas (Fig. 30.6). Shaded sites in all four regions tended to have a different community composition than unshaded sites. This pattern was particularly strong in Taranaki, where Crustacea dominated unshaded sites, but Diptera, Ephemeroptera, Plecoptera and Trichoptera taxa, and molluscs dominated shaded sites.

In contrast to stream biota in agricultural landscapes, spring biota may be buffered to some extent against the effects of land use. They receive a constant supply of clear, cool groundwater, so may not be subject to some of the stress factors present in streams in agricultural catchments (e.g., elevated water temperature and high turbidity). Conversely, the absence of floods may make springs more susceptible to other land-use stress factors, such as the irreversible smothering of habitats by fine sediments. This

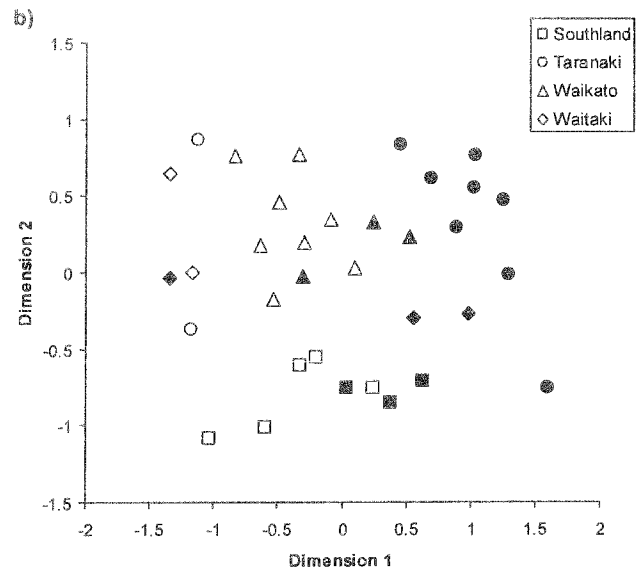


Figure 30.6 Multidimensional scaling ordination plot of presence/absence data from 34 lowland spring sites in four regions. Invertebrate communities differ most as a result of the presence or absence of overhanging vegetation. Open symbols denote unshaded sites to the left of dimension 1 and closed symbols denote shaded spring communities to the right of axis 2. Reproduced from Scarsbrook and Haase (2003).

may be particularly prevalent in areas where stock access may lead to bank collapse. Substratum type may be important in mitigating the effects of stock access. For example, limestone springs with coarse substrata (e.g., boulders) may be less susceptible to trampling damage than alluvial springs, where gravels may be smothered more easily by increased sedimentation. In agricultural landscapes high water temperature may be a major source of stress for aquatic communities (e.g., Quinn and Hickey 1990). Because they have a flow of cool groundwater, spring habitats potentially act as refugia for species that may otherwise be excluded from agricultural landscapes, thereby contributing to the protection of regional biodiversity. Further work is required to assess the extent to which spring habitats act as refugia in modified landscapes.

Given the position of many spring habitats outside the conservation estate, it is clear that the protection of these habitats, and the indigenous flora and fauna they contain, will require education of land managers on the practices required to minimise impacts, and appropriate strategies for restoration.

GEOHERMAL SPRINGS

Introduction

The location of New Zealand at the juncture of two tectonic plates creates upwellings of water heated by volcanic activity. There are also a number of thermal

springs that occur in non-volcanic regions of the South Island (Winterbourn 1973; Stark *et al.* 1976; Mongillo and Clelland 1984). Geothermal areas on both islands give rise to hot springs that can have water temperatures greater than 85°C, low pH, high turbidity and high sulphide concentrations (Vincent and Forsyth 1987; Moss 1998). In the following sections we provide an overview of many of the general characteristics of geothermal springs. Thorough reviews by Brock and Brock (1971), Ellis and Mahon (1977) and Vincent and Forsyth (1987) provide much greater detail on the specific thermal, chemical and biological characteristics of many of the major geothermally influenced lakes and streams in New Zealand, many of which are fed by springs.

Physical and chemical characteristics

There is considerable variation in the chemical, thermal and hydrological characteristics of geothermal springs in different regions of New Zealand (Hunt and Bibby 1992). The water temperature of springs can range from around 30°C to greater than 85°C, declining with increasing distance from the source, and temperature is the dominant factor affecting the aquatic biota of springs. The presence of a marked thermal gradient can give rise to distinct zonation patterns in algae and bacteria downstream and laterally from the spring source. This thermal regime may also be subject to variation created by changes in discharge, weather and season (Brock 1978), and by the draw-off of geothermal waters for human use (James 1985).

Many springs, particularly in the North Island, are also highly acidic (pH 1–4) and the combination of high temperature and low pH can severely limit spring biodiversity. Boiling acid springs are almost exclusively populated by the bacterium *Sulfolobus acidocaldarius*, with other acidophilic bacteria only occurring at temperatures below about 55°C. Cyanobacteria, in contrast, are almost completely absent from acidic springs, although eukaryotic algal species (notably *Cyanidium caldarium*) may be more tolerant (Vincent and Forsyth 1987).

High water temperatures, in association with the underlying geology, can lead to high levels of dissolved solutes in geothermal springs. Total dissolved solids can range from 6000 g/m³ at Tokaanu to 600 g/m³ at Orakeikorako (Ellis and Mahon 1977). Sulphide concentrations in the waters of some North Island springs can also be extremely high (e.g., 6 g/m³ in Ngawha Springs and 1.8 g/m³ at Wairakei (Ellis and Mahon 1977)). These levels of sulphides are toxic to most biota but provide a food source for sulphur-oxidising bacteria such as *S. acidocaldarius*. South Island warm springs tend to have low sulphide levels (Ellis and Mahon 1977). Ammonium concentrations may be considerably higher in geothermal springs than non-geothermal waters, as a result of

hydrolysis and decomposition of organic material in heated sedimentary rocks (Vincent and Forsyth 1987). Phosphorus levels are also high in springs around the Taupo volcanic zone, but no more so than in many non-geothermal waters in the region (Williamson and Cooke 1982). The toxic heavy metals mercury, arsenic and boron may occur at elevated or even lethal levels in geothermal waters in springs around the Taupo volcanic region.

Biodiversity

In contrast to cold-water springs, the extreme environmental conditions of many geothermal springs leads to reduced species diversity in comparison with other aquatic habitats. At the extremes of pH and temperature many organisms are unable to survive. Arthropods appear unable to tolerate temperatures above 45°C (Vincent and Forsyth 1987; Moss 1998), protozoa and algae (*Cyanidium caldarium*) are rare above 55°C (Brock 1978; Brock and Brock 1971), thermophilic fungi occur to 60°C (Brock 1978), cyanobacteria to 70°C (Vincent and Forsyth 1987) and various archaeobacteria (*Desulfurococcus*, *Thermoproteus*, *Sulfolobus*) can occur at temperatures greater than 80°C.

A number of unique archaeobacteria and eubacteria can survive the high temperatures and low pH of geothermal springs. *Sulfolobus acidocaldarius*, for example, is a widespread archaeobacterium in hot acid waters of 55–90°C and pH 1–4. It is a chemosynthetic bacterium that uses hydrogen sulphide to reduce CO₂ to produce organic compounds (Bohlool 1975; Brock 1978). Heterotrophic eubacteria include the aerobes *Bacillus stearothermophilus* and *Thermus aquaticus*, both of which are common in mildly acid to alkaline waters below 85°C, and the obligate anaerobes *Clostridium thermohydrosulfuricum* and species of *Thermoanaerobium*. The enzymes and cell components that allow these organisms to function at such high temperatures have been the focus of considerable research aiming to harness the enzymes for industrial use (e.g., Bergquist and Morgan 1995). For example, the enzyme used to produce high-fructose syrup is produced by a thermophilic bacterium *Bacillus coagulans* (Brock 1985; Vincent and Forsyth 1987).

The Chloroflexaceae is another common group of orange filamentous eubacteria that utilise anoxygenic photosynthesis to survive in geothermal mats, often in close association with cyanobacteria (Castenholz 1984). Below about 70°C, cyanobacteria (blue-green algae) are often the dominant photoautotrophs in the geothermal mats in waters that are neutral to alkaline. They include species of *Synechococcus*, *Phormidium*, *Oscillatoria*, *Calothrix thermalis*, and *Mastigocladus laminosus* (Castenholz 1976; Vincent and Forsyth 1987). Water chemistry often determines what species will dominate, with *M. laminosus* sulphide intolerant and *Oscillatoria*

ampigranulata preferring waters with concentrations of 20–70 g/m³ of sulphide. Eukaryotic algae usually are not as common as cyanobacteria in geothermal springs, although *Cyanidium caldarum* and *Zygonium kumaeense* occur in some New Zealand springs and streams (Brock and Brock 1971; Forsyth 1977). The former can survive in waters with pH 1–5 and temperatures up to 55°C (Brock 1978).

Diptera and Coleoptera are the most common macroinvertebrate inhabitants of geothermal springs and their outflow channels (Fig. 30.7) (Winterbourn 1968). Ephydrid flies, and hydrophilid and dytiscid beetles can tolerate temperatures up to 47°C and pH as low as 2, and several species are obligate thermophiles (e.g., *Ephydrella thermarum* and *Anacaena tepida*) (Winterbourn *et al.* 2000). Several other species in these families also occur in geothermal waters, but are not true thermophiles. Other invertebrate taxa able to tolerate high water temperatures (20–35°C) such as Chironomidae, Oligochaeta and

Mollusca may also be present in geothermal springs at temperatures below their thermal limits. The molluscs *Pseudosuccinea columella*, *Physella acuta* and *Planorbarius corneus* can tolerate temperatures up to 35°C and *Potamopyrgus antipodarum* temperatures up to 28°C, but cannot tolerate low pH (Winterbourn 1969). Several species of Chironomidae, including species of *Chironomus*, *Tanytarsus*, and unidentified Orthoclaadiinae, can be abundant in geothermal springs (Winterbourn and Brown 1967; Winterbourn 1973; Stark *et al.* 1976), while Muscidae (*Limnophora* sp.) and the tipulid *Limonia* have also been recorded in springs at temperatures of 27–32°C and 33–45°C respectively (Vincent and Forsyth 1987). Larvae of the dixid midge *Paradixa fuscinervis* and the mosquito *Culex pervigilans* that are not normally found in geothermal waters have also been found in the Copeland River Springs at temperatures up to 32°C (Winterbourn 1973). Acarina may also be abundant in algal mats; Stark *et al.* (1976) recorded *Hydrozetes lemnae* and *Trimalaconothrus norus* in alkaline waters up to 41°C in the Hurunui River Springs.

The characteristic thermal limits of these taxa often lead to a marked longitudinal change in community composition as the temperature cools downstream from the spring. Spring heads with high temperatures of > 80°C will be dominated by heterotrophic bacteria. In cooler springs or where downstream temperatures drop below 70°C, blue-green algae become the dominant component, often forming quite thick algal mats. Much of this biomass, which is able to build up outside the influence of herbivores, will eventually slough off and float downstream. Finally, as temperatures drop below 45°C, invertebrate grazers and eventually predators, such as beetle larvae, are able to survive. Perhaps more so than in the cold-water springs, the unique biota and their complex interactions with the physical and chemical environment mean that these habitats are highly susceptible to anthropogenic environmental change (James 1985).

CONCLUSION

Unique plants and animals live in New Zealand cold-water and geothermal springs. They represent distinctive aquatic habitats both within New Zealand and worldwide, but they have received little attention compared with more common aquatic systems. Recently there has been an increase in research and management focus on these habitats and this has heightened awareness of their uniqueness and extremely high biodiversity. Their distinctive biodiversity, their thermal and hydrological characteristics, and the potential anthropogenic threats to their integrity, mark them as systems which need considerably more research and management.

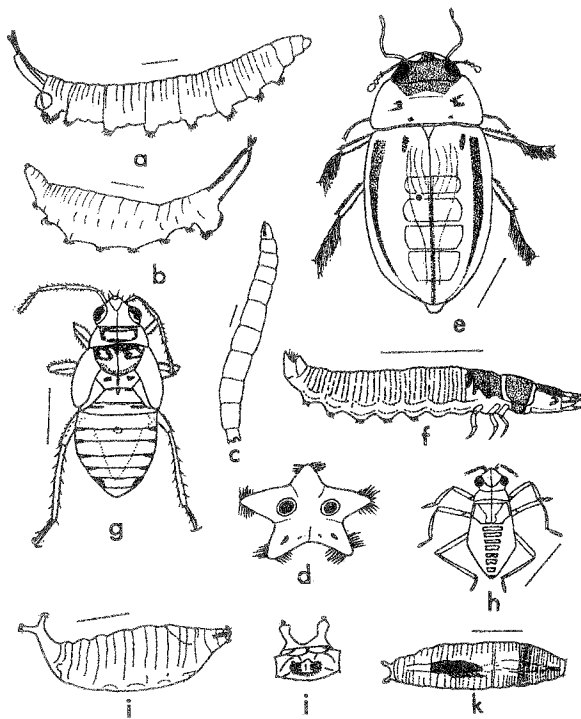


Figure 30.7 Invertebrates that can be present in warm geothermal waters (30–45°C). a) *Ephydrella* larvae (Ephydridae), b) *Ephydrella* pupae, c) Tipulidae larvae, d) Tipulidae larval spiracular disk, e) *Enochrus* adult (Hydrophilidae), f) *Enochrus* larva, g) *Saldula* nymph (Saldidae), h) *Microvelia* nymph (Veliidae), i) *Scatella* larva (Ephydridae), j) *Scatella* pupal respiratory siphons, k) *Scatella* pupa. Length of scale lines represents 1 mm. Figure reproduced from Winterbourn (1968) with kind permission M.J. Winterbourn.

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Chapter 29

Groundwater systems

Graham Fenwick, Hugh Thorpe and Paul White

WHY IS GROUNDWATER IMPORTANT?

About 91% of the liquid fresh water on earth is invisible—it occurs as water within the ground. Groundwater can be drawn on during dry periods when surface waters are inadequate, and in many areas it is often the only source of water available. Groundwater supplies, wholly or in part, more than 50% of New Zealanders with drinking water, and major industries and many irrigators also draw heavily on groundwater.

Groundwater is stored in aquifers and is thus less susceptible than surface water to variations in supply and quality or to pollution. It supports a significant subterranean fauna, maintains wetland ecosystems, provides base (low seepage) flow to most rivers and supplies spring water to many of our clearest and most beautiful lowland streams.

This chapter contains a brief description of groundwater systems for the general reader. A more detailed account of current research on groundwater and regional groundwater summaries is contained in “Groundwaters of New Zealand” (Rosen and White 2001).

WHAT IS AN AQUIFER?

An aquifer is a geological formation that holds water, and which will release this water at a rate fast enough to be useful. To hold water, a geological formation must contain openings or voids (pores) or be fractured. Porosity is defined as the ratio of the volume of voids in the rock to the total volume of rock, usually expressed as a percentage. If the pores are full of water, the formation is saturated. For water to be able to move through the formation, the pores must be interconnected, as in sand or gravel, or linked by fractures, as in hard-rock aquifers. If the pores are microscopic, as in clay, water can move only very slowly,

therefore clay formations are not aquifers. Water-yielding openings can range from barely visible pores, for example in sand aquifers, to the cavernous channels in some limestone areas, such as Waitomo or Takaka.

New Zealand aquifers occur in a variety of geological formations ranging from coarse greywacke alluvial gravel and sand, mostly east of the axial ranges of both islands, through fractured volcanic rock in the central plateau, to fine beach sands in the far north. The aquifers range in depth from a few to more than 700 metres for cold water and considerably more for geothermal water. None of our aquifers is large by international standards: New Zealand is a small country rent by fault lines, which often delineate aquifer boundaries.

Broadly, there are two types of aquifer: confined and unconfined. Confined aquifers are overlain by layers of silt or clay, which prevent water from percolating down into the aquifer from the surface (Fig. 29.1). Confined aquifers are completely full of water, which may be under pressure. An unconfined aquifer is unsaturated (i.e., it contains air) and the level at which the water in the aquifer is at atmospheric pressure is the water table at that point. Water, including pollutants, can percolate down from the surface into an unconfined aquifer (Thorpe 1992).

The energy state of water, per unit weight, within an aquifer is called the potentiometric or piezometric “head” and this changes from position to position as the water moves slowly through the rock. In fact, it is the variation in head from place to place which drives water movement. The piezometric surface (regional water table) is the surface defined by joining the heads at all the points in an unconfined aquifer. One important concept is the hydraulic gradient—it is the change of piezometric head (head loss) between two points divided by the distance between those points (Fig. 29.2). Water always flows in the direction of maximum hydraulic gradient.

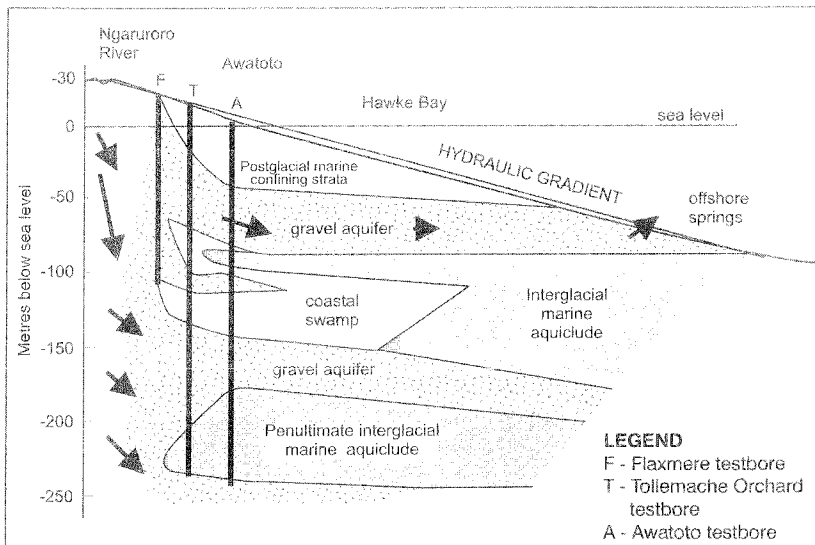


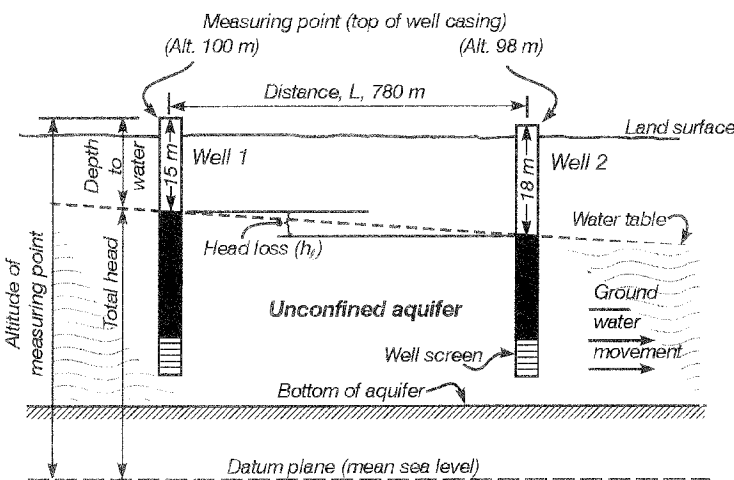
Figure 29.1 Generalised cross-section of the Heretaunga Plains, Hawkes Bay, and the offshore area. F, T and A mark deep exploratory bores. Water leaks from the Ngaruroro River into the unconfined aquifer, which in turn recharge the confined gravel aquifers. Water discharges from the topmost confined aquifer via submarine springs (Luba 2001).

The law describing groundwater flow was formulated in the mid 19th century by Henri Darcy, a young French engineer, in the course of experiments to design a water supply for the city of Dijon. Darcy's law states that the groundwater flow velocity "v" is:

$$v = K \frac{h_1}{L}$$

where h_1/L is the hydraulic gradient, and K is the hydraulic conductivity, a measure of the ease with which water flows through an aquifer

Water enters aquifers from rain falling directly on land overlying an unconfined aquifer, or by seepage from rivers, lakes or the sea. These processes are known as aquifer recharge.



$$\text{Hydraulic gradient} = \frac{\text{Head loss}}{\text{Distance}} = \frac{h_l}{L} = \frac{5}{780}$$

Figure 29.2 Definition of the concept of hydraulic gradient.

SOME NEW ZEALAND AQUIFERS

Aupori Peninsula

The Aupori Peninsula stretches north from Kaitia towards Cape Reinga. Here a basement of volcanic rock is overlain by a sequence of variable but fine marine sands containing silty sands and thin peat layers. This aquifer is important for irrigation for the production of avocados (Thorpe 1992). The aquifer is unusual in that it extends across the entire peninsula near Houhora (Fig. 29.3).

A transect of five deep bores was drilled across the peninsula, and piezometer nests were installed, allowing the pressure distribution in the aquifer to be determined (Fig. 29.3) (Thorpe 1992). The mounding of the water table at the centre of the peninsula confirmed that the only possible source of the groundwater is infiltrating rainfall, and the hydraulic gradients towards both coasts showed

the direction of groundwater flow is from the centre to both the east and west. This resource is useful, but it will need to be carefully managed to ensure that piezometric heads at the coast are not lowered by pumping to below sea level, which would reverse the hydraulic gradient and cause seawater to flow in under the land. (Fig. 29.4).

Heretaunga Plains

The geological structure of the Heretaunga Plains aquifer near Hastings is similar to that of several other coastal areas around New Zealand (e.g., Hutt Valley, Wairau Plains and near Christchurch). In all these places, a series of alluvial gravel aquifers has been laid down during successive ice ages, each separated by a layer of silts or clays deposited during interglacial periods (Fig. 29.1) (Luba 2001).

Most aquifer recharge is by water leaking from rivers that flow across the unconfined aquifers,

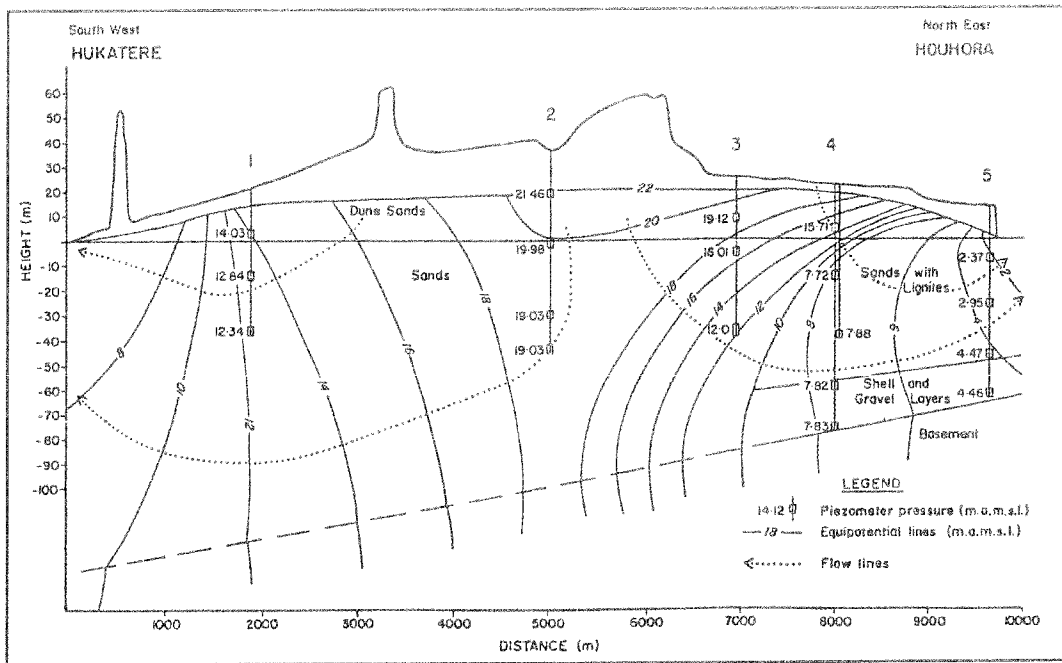


Figure 29.3 Hydrogeological cross-section of the Aupori Peninsula near Houhora, Northland. The Tasman Sea is to the left and the Pacific Ocean is to the right (Groundwater Consultants New Zealand 1987, from Thorpe 1992). Heights are in metres above mean sea level. Numbers 1 to 5 mark exploratory bores.

supplemented by infiltrating rainfall. The natural discharge is to springs near the boundary of the confined aquifers or to submarine springs, which may be many kilometres offshore. In the case of the Heretaunga Plains, recharge is mostly from the Ngaruroro River between Fernhill and Maraekakaho, at an average rate of about 6 m³/sec. Water from this system irrigates some 26,000 ha of New Zealand’s most fertile soils, and supplies domestic and industrial water to Napier and Hastings. There are about 9,000 bores on the Hawkes Bay Regional Council data base.

Waimea Plains

Possibly the most studied aquifer system in New Zealand lies beneath the Waimea Plains near Nelson. It has a relatively small area of around 7,500 ha, but the soils and climate make it one of the most potentially productive agricultural/horticultural areas of New Zealand (Thomas 2001). The productivity is dependent largely on irrigation from bores, which also supply industrial, municipal and rural domestic users. Three aquifers have been delineated beneath the Waimea Plains: the Lower Confined, the Upper Confined and the Unconfined Appleby Gravel aquifers (Fig. 29.5). There are also minor aquifers in the Hope Gravels at the foot of the eastern hills (Thomas 2001). River recharge to this system is mostly from the Wairoa River after it emerges from its gorge. Water passes into the shallow unconfined Appleby Gravel aquifer and then leaks downward into the two confined aquifers.

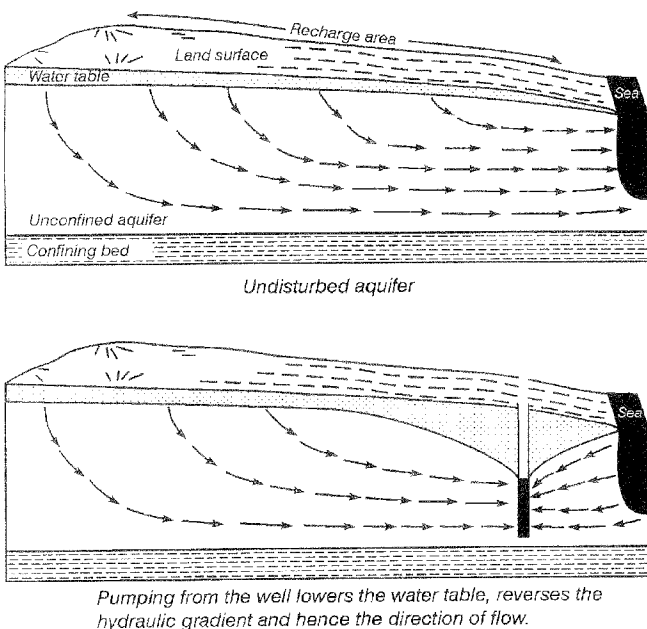


Figure 29.4 The mechanism of sea water intrusion into an aquifer.

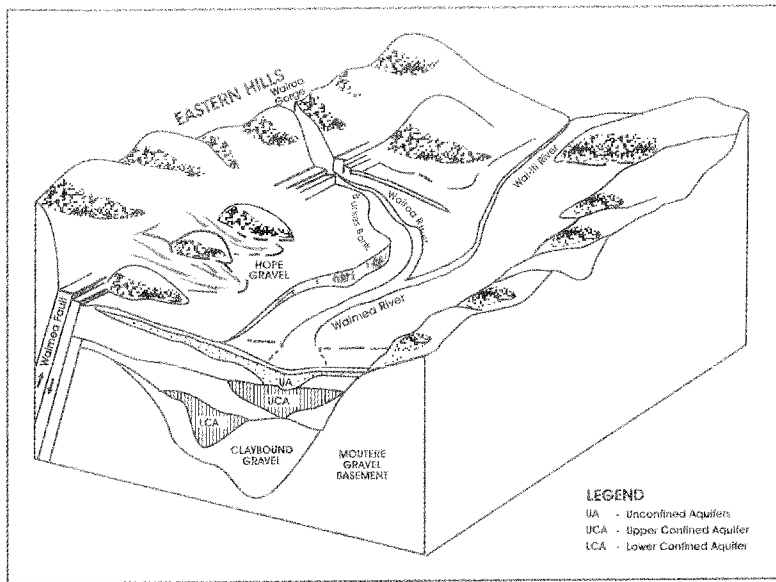


Figure 29.5 Block diagram of the Waimea Plains, Nelson (Thomas 2001).

Rainfall recharge also occurs via the Hope Gravels. Natural discharge is through springs near the Waimea Estuary and also presumably through submarine leakage from the upper confined aquifer into Waimea Inlet.

Intensive horticultural activity with heavy fertiliser application has led to very high nitrate levels in the confined aquifers, but less so in the unconfined aquifers (Thomas 2001). Heavy use of groundwater, beginning in the early 1970s, led to significant seasonal lowering of pressures in the confined aquifers and prompted a major computer modelling study of the surface/groundwater system of the plains (Fenemor 1988). This model has been the basis of management decisions on water allocation limits and is updated as additional data are gathered.

GROUNDWATER-SURFACE WATER INTERACTION

Recharge from rainfall

Generally, water that penetrates below the root zone of surface vegetation (which may be several metres deep) will percolate downward until it reaches the water table (Fig. 29.6). The percentage of rainfall that recharges an aquifer depends on the climate, the type of soil and its thickness, the type of vegetation growing in it and the amount and pattern of rainfall. For a given annual rainfall, more will recharge the aquifers if the falls are heavy and occur

in winter when soils are wet, rather than occurring in the form of light summer showers. Light showers tend to be soaked up by the soil and then transpired back to the air by plants, whereas heavy falls may overflow the moisture-holding capacity of the soil and provide excess water for recharge. Also, soils generally contain worm-holes, root holes or shrinkage cracks, which in heavy rain provide flow paths through which water can bypass the upper zones of the soil and penetrate quickly beyond the level of most active evapo-transpiration.

Estimating the average annual recharge of an aquifer is very important for water managers because it is an upper limit for how much water can be taken from the aquifer on a sustainable basis. In reality, if surface and sub-surface ecosystems are to be maintained, the useable resource is much less than the recharge.

Over a 15-year period at the Winchmore Research Station near Ashburton, 31% of the rainfall passed through the grass-covered soil to recharge the unconfined aquifer (Thorpe 1992). Under bare soil, the yearly average recharge was about 40% of rainfall and this varied from about 10% in January to over 70% in July, even though the rainfall was fairly evenly distributed (Fig. 29.7). This seasonal variation occurs because in summer much of the rainfall is returned quickly to the atmosphere by evaporation or used by actively growing vegetation, and is not available for recharge.

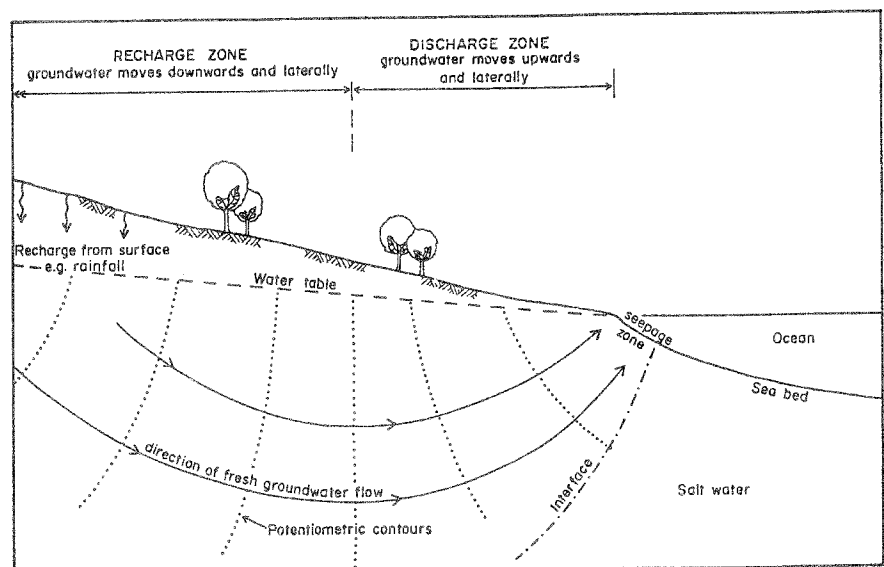


Figure 29.6 The classical concept of recharge to and discharge from an unconfined aquifer (North Canterbury Catchment Board 1986, from Thorpe 1992).

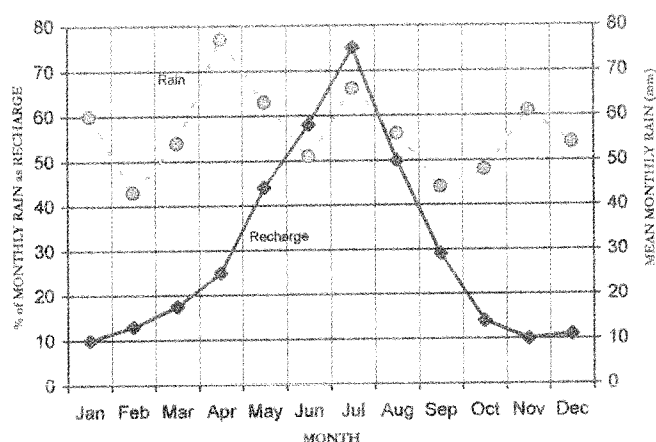


Figure 29.7 Distribution of rainfall recharge throughout the year at Winchmore Research Station, mid-Canterbury (Thorpe 1992, courtesy of AgResearch Winchmore).

Recharge from rivers

It is possible to determine if a river is recharging the groundwater, or if groundwater is discharging into a river by simultaneous flow measurements along the river. Conditions in the river must be low and steady, because fluctuating flows during even a small flood would mask the effects of recharge or discharge from the aquifer. Recharge and discharge may occur in different reaches of the same river as it flows across a plain, usually with recharge to the aquifers in the upper plains and discharge further downstream. Examples are the rivers draining off the Mamaku plateau on to the Tōkorōa plateau, the Wairoa and Wai-iti Rivers near Nelson and the Ashburton River in mid Canterbury.

The low flow in most rivers is supplied by groundwater discharge, as water seeps steadily from the soil and underlying fractured rocks into channels, mostly in the upper valley reaches of rivers.

Springs

The presence of a spring on a plain is an indication of groundwater at some depth, usually in a confined aquifer. Springs appearing on a valley side suggest that there is a layer of impermeable material intersecting the ground at that point and the water has flowed along the upper surface of this layer to emerge as a spring.

Spring flows may range from mere seeps of a few ml/sec to huge flows from karst systems such as the Pupu Springs near Takaka, which produce a mean flow of 13.2 m³/sec (Thomas 2001). In many systems, the springs are concentrated near the boundary of the confined aquifers, where the confining layer is thin and easily penetrated by flowing water. A line of such springs is often the source of a particularly clear stream such as the Avon River in

Christchurch. The outflow from such springs (including submarine outflow) may constitute most of the discharge from a groundwater system (e.g., an estimated 64% from the Heretaunga Plains aquifers) (Luba 2001).

Coastal flows and seawater intrusion

Where an aquifer emerges into an estuary or the sea, the groundwater flows seawards in the direction of the maximum natural hydraulic gradient (i.e., normally the potentiometric head under the land is higher than mean sea level). If the extraction of groundwater from wells near the coast is too great, however, the water table may be drawn down below mean sea level, thus reversing the hydraulic gradient. Saline water may then flow inland, with undesirable effects on irrigated crops and the people who drink the water (Fig. 29.4).

Seawater intrusion has occurred in at least one part of New Zealand, near Motueka, where over pumping led to increased salinity in irrigation wells very near the coast. In this case the salt water appears to be flushed out during winter when irrigation ceases and aquifer heads recover, re-establishing the natural pattern of flow to the sea. The solution in this case was to drill new irrigation well fields further inland so that the head at the coast was less affected by pumping.

GROUNDWATER CHEMISTRY

The chemical composition of groundwater is dependent upon many factors: the chemistry of recharge water, the chemical and biological reactions occurring on the land surface, in the soil, and in the unsaturated and saturated zones, and on the mineral composition of aquifers and confining beds. The chemical quality of groundwater is important because good quality water is vital for human health, animal welfare, irrigation, and most industrial applications. Aquifers can become insidiously polluted—this affects sub-surface ecosystems, and when the groundwater subsequently discharges into surface water bodies such as lakes and streams, their ecosystems may also be affected.

The following discussion of New Zealand groundwater chemistry is based on ionic components—it summarises the “natural”, mainly chemical, make-up of the water. Some of the chemical components of New Zealand groundwater that result from human activity, including nutrients, heavy metals, hydrocarbons and pesticides, are summarised, as well as the occurrence of microbes in the groundwater. Associations between groundwater age and its chemistry are also discussed.

Analysis of the chemistry of groundwater samples taken as part of the National Groundwater Monitoring Programme (NGMP) indicates that the major cations in

groundwater are sodium, calcium and magnesium (Rosen 1997), while the major anions are bicarbonate (HCO_3^-) and chloride (Rosen 2001). Water samples were classified under nine principal water types, using combinations of relative concentrations of these cations and anions. The two most common types of groundwater are Ca-Na- HCO_3^- solutions and Ca- HCO_3^- solutions.

Median sodium concentrations in groundwater sampled in the NGMP are in the range 2–306 g/m^3 , with most median concentrations below 50 g/m^3 (Rosen 2001). Groundwaters probably receive most of their sodium from the ocean via rainfall, since their Na:Cl ratios are close to that of seawater and the ratios fall on the seawater concentration-dilution line. Human and animal waste, landfill leachate, and some types of industrial waste may also contribute sodium to groundwater. Median calcium concentrations are in the range 2–210 g/m^3 and most median concentrations are less than 40 g/m^3 (Rosen 2001). The dissolution of carbonate cement, pebbles and shells is a likely source of calcium in most New Zealand groundwaters, as carbonate rocks are relatively rare. Calcium oxide (CaO), added to pasture to adjust soil pH, is also a possible source of calcium in groundwater.

Magnesium concentrations in NGMP wells are in the range 1–55 g/m^3 , with most samples less than 10 g/m^3 (Rosen 2001). Silicate minerals such as olivine, pyroxene, amphibole and mica are likely to be the main source of magnesium in groundwater. Magnesium may also originate from fertiliser applications.

Bicarbonate is derived from reactions of water and carbon dioxide in the atmosphere, the interaction of dissolved atmospheric carbon dioxide with organic matter in soil, sulphate reduction, and the dissolution of carbonate rocks. For NGMP wells the median bicarbonate concentrations are less than 12 (milli-equivalent), and generally less than 5 (milli-equivalent). Bicarbonate makes the major contribution to groundwater alkalinity, and the median pH in NGMP wells is between 5.8 and 8.5 (Rosen 2001).

Median chloride concentrations in NGMP wells (Rosen 2001) are in the range 1–300 g/m^3 , with most below 50 g/m^3 . Rain is the dominant source of chloride in groundwater in New Zealand. Salt in paleo-seawater trapped in marine-derived sediments is a possible source of both chloride and sodium in groundwater, and this may be influencing groundwater quality in the deep coastal aquifers in Canterbury and Gisborne. Sewage and other human, animal and industrial wastes and leachates are other possible sources of chloride.

In 36% of 10,000 groundwater samples (Daughney 2003) iron concentrations exceed the “aesthetic” guideline value of 0.2 g/m^3 (Ministry of Health 2000). Groundwater from gravel and sand aquifers is more likely than

groundwater from aquifers in most other lithologies to have high iron and/or manganese concentrations when reducing conditions occur. Most New Zealand groundwaters contain concentrations of fluoride, derived from the dissolution of rock minerals, of less than 0.5 g/m^3 (Rosen 2001). No groundwaters of the NGMP exceed the maximum drinking water standard for fluoride of 1.5 g/m^3 (Rosen 2001).

Nutrients

Nutrients in groundwater are of interest for several reasons (Close *et al.* 2001). Groundwater is a sole or partial source of drinking water for approximately 50% of New Zealand’s population and poor groundwater quality may affect human health (Davies 2001). Nutrients in groundwater can cause algal blooms and eutrophication when the water discharges into surface waters, such as streams and lakes. Agriculture is a major source of these nutrients and other contaminants.

Nitrogen is the most-studied nutrient in groundwater because it is the nutrient most commonly occurring at elevated concentrations, and it poses both health (Davies 2001) and environmental risks. In natural groundwater, nitrogen usually occurs as one of three forms: nitrate-nitrogen, ammonium-nitrogen or organic nitrogen (Close *et al.* 2001). A fourth form, nitrite-nitrogen also occurs in groundwater, but usually in significant concentrations only near contaminated sites such as disposal facilities for sewage and organic waste.

Nitrogen enters the land surface from the atmosphere and from sources such as grazing animals, fertilisers, sewage disposal, and from mineralisation of organic matter following tillage. The chemical form and rate of nitrogen leaching to groundwater is controlled by complex reactions of the nitrogen cycle in the soil and vadose (unsaturated) zones, and by the physical properties and thickness of this zone (Close *et al.* 2001). Nitrate-nitrogen is the stable form of nitrogen in aquifers with abundant oxygen, and ammonium-nitrogen is the stable form in aquifers that are depleted in oxygen. Organic nitrogen can enter an aquifer directly if the water table is near the surface; this form of nitrogen is rarely found in significant quantities in uncontaminated New Zealand aquifers (Close *et al.* 2001).

The current New Zealand Drinking Water Standard for nitrogen is 11.3 g/m^3 (as nitrate-nitrogen), and 5% of 111 New Zealand (NGMP) wells are above this level (Close *et al.* 2001). The upper limit (95th percentile) for “pristine” oxidised groundwater in New Zealand is estimated as 3.5 g/m^3 (Daughney and Reeves 2003). Median nitrate concentrations are greater than 3.5 g/m^3 in 19% of the NGMP wells.

Higher nitrate-nitrogen concentrations are generally observed in aquifers within about 40 m of the watertable

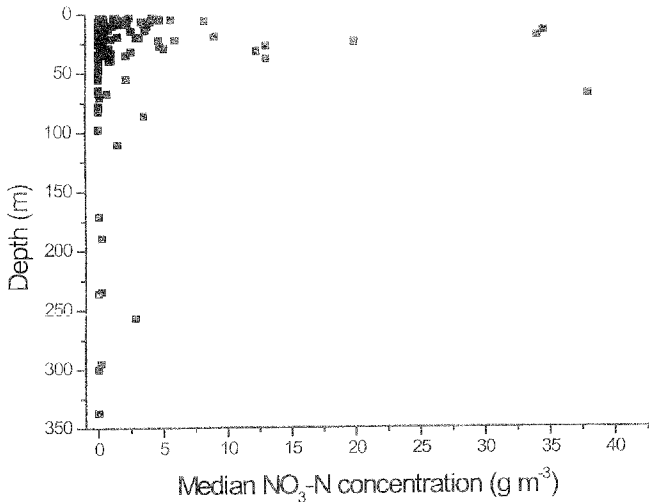


Figure 29.8 Median concentration of nitrate in groundwater versus depth of sample for National Groundwater Monitoring Programme wells (Daughney and Reeves 2003).

(Fig. 29.8), probably because drainage water is likely to mix, denitrify and become diluted in the shallower aquifers.

Nitrate concentrations can show seasonal increases, for instance in response to agricultural practices (Close *et al.* 2001; Rosen 2001). Longer-term changes in nitrate concentrations in groundwater may not be consistent across regions. For example, nitrate-nitrogen concentrations showed no trend in 94 of 129 Canterbury wells (Hanson 2002) over the period 1977 to 2001. Nitrate-nitrogen concentrations increased in 20 wells, and decreased in 15 wells, between 1977 and 2001 (Fig. 29.9).

High ammonium concentrations in groundwater are commonly associated with aquifers where oxygen is absent (Rosen 2001) and with low nitrate concentrations (Fig. 29.10), which is indicative of the chemical reactions controlling the nitrogen species and levels.

New Zealand groundwater generally has low phosphate concentrations, and only three wells in the NGMP have median concentrations greater than 0.2 g/m^3 (Fig. 29.11). No phosphate rock deposits are associated with aquifers in New Zealand, although phosphorous does leach from rhyolitic pumice aquifers in the Taupo Volcanic Zone, causing phosphorous concentrations of up to 0.3 g/m^3 in spring water (Timperley 1983).

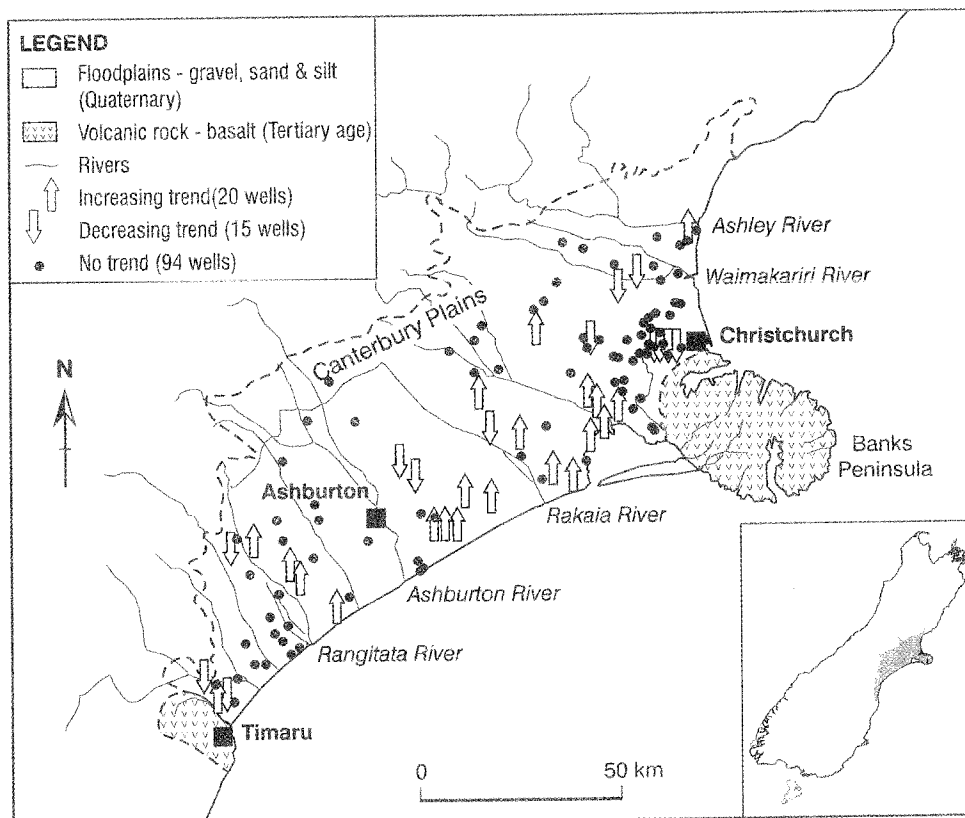


Figure 29.9 Trends in Canterbury groundwater nitrate-nitrogen concentrations, 1974 to 2001 (after Hanson 2002).

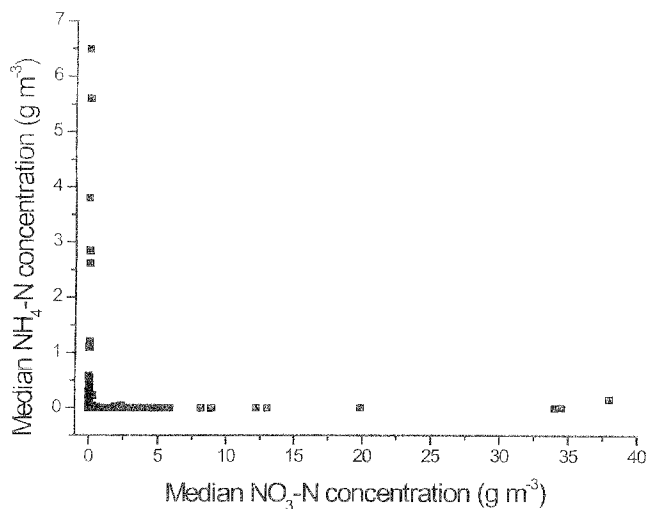


Figure 29.10 Median ammonium and median nitrate concentrations for wells in the National Groundwater Monitoring Programme (Daughney and Reeves 2003).

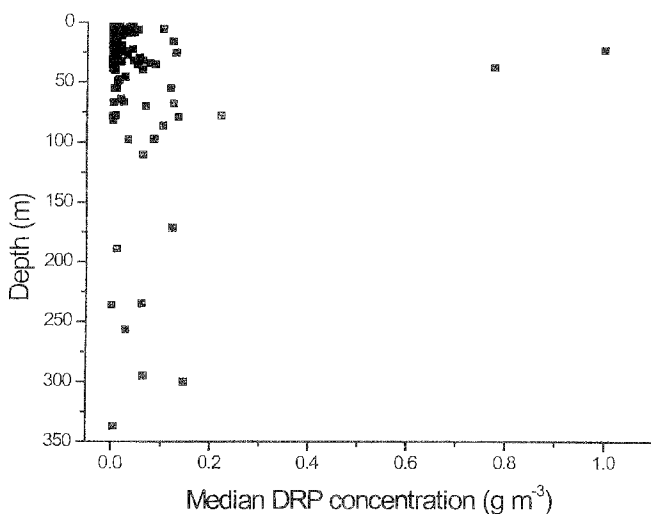


Figure 29.11 Median concentrations of phosphorus in groundwater versus depth of sample in National Groundwater Monitoring Programme wells (Daughney and Reeves 2003).

Heavy metals, hydrocarbons and pesticides

In a survey of NGMP wells in 13 regions, median concentrations of the heavy metals arsenic and chromium were found to be at the detection limits for the analytical methods (Rosen 2001). Three of 87 samples had arsenic concentrations above the drinking water standard of 10 mg/m^3 ; the highest concentration was 54 mg/m^3 , which subsequent investigations found to be associated with contaminated soil around a trough for treating sheep

footrot, up-gradient of the well. All concentrations of chromium, lead and copper were lower than drinking water guidelines. Copper, for example, had a mean concentration in all samples of the survey of 3 mg/m^3 , compared with a drinking water standard of 2000 mg/m^3 .

Analyses of three groups of hydrocarbons (halogenated alkanes and alkenes, aromatics, and polynuclear aromatic hydrocarbons) indicated 22 different species over an 11 year period in Christchurch (Rosen 2001). Concentrations were all well below the New Zealand drinking water standards, indicating that the risk to Canterbury groundwater users is presently low (Rosen 2001).

Pesticides have been detected in shallow groundwater in three national surveys of groundwater between 1990 and 1998/99 (Close *et al.* 2001). A total of 95 wells were sampled in the 1998/99 survey and pesticides were detected in groundwater from 33 wells; groundwater from 18 wells had two or more pesticides present. Seventy-four percent of samples had concentrations of less than 0.1 mg/m^3 ; with only one well having pesticide concentrations greater than the New Zealand drinking water standard. This well was down-gradient of a point source of contamination.

Microbes

Four categories of water-borne pathogens infect humans: viruses, bacteria, protozoans and helminths (Sinton 2001). The first three are micro-organisms and the last is a multicellular organism. Potential sources of microbial contamination of groundwater include grazing animals, septic tank systems, effluent irrigation, land application of sludge, landfills, offal pits and sewer leakages. The viruses include enteroviruses, adenoviruses, Norwalk Virus, Reoviridae and Hepatitis viruses A and E. Water-borne bacteria include *Campylobacter*, *Salmonella*, *Escherichia coli*, *Shigella*, *Vibrio* species, and *Yersinia enterocolitica*. Waterborne Protozoa include *Giardia*, *Cryptosporidium* and *Entamoeba* (Sinton 2001).

Microbial contamination of New Zealand's groundwater (Sinton 2001) is more likely in shallow unconfined aquifers. For example, a survey of wells in Canterbury in 1995 found that 8.8% of wells tested positive for faecal coliforms; generally the contaminated wells were taking groundwater from less than 30 m below ground level (Sinton 2001).

The survival of microbes and their transport to and within groundwater are controlled by factors such as soil moisture, temperature, pH, soil organic matter, pore size, groundwater velocity, adsorption and other factors specific to a given species. In an experiment, Sinton (2001) showed that *E. coli* can travel approximately 900 m down-gradient of an injection well in a gravel aquifer in Canterbury.

Age

The decay of radioactive isotopes, such as tritium and carbon-14, and dissolved gases, such as chlorofluorocarbons (CFCs), are used to identify the sources and residence times of groundwater (Stewart and Morgenstern 2001). These parameters help to improve conceptual flow models and to identify the history of contamination. For example, measurements of nitrate and CFC age of water from wells between the Waimakariri and Rakaia rivers in Canterbury (Fig. 29.12) indicate that nitrate concentrations increased significantly around 1950, possibly due to post-war intensification of farming (Stewart *et al.* 2002). The nitrate-age distribution in Figure 29.12 also may be affected by the deep water being derived from rivers or by conversion of nitrate to ammonia at depth.

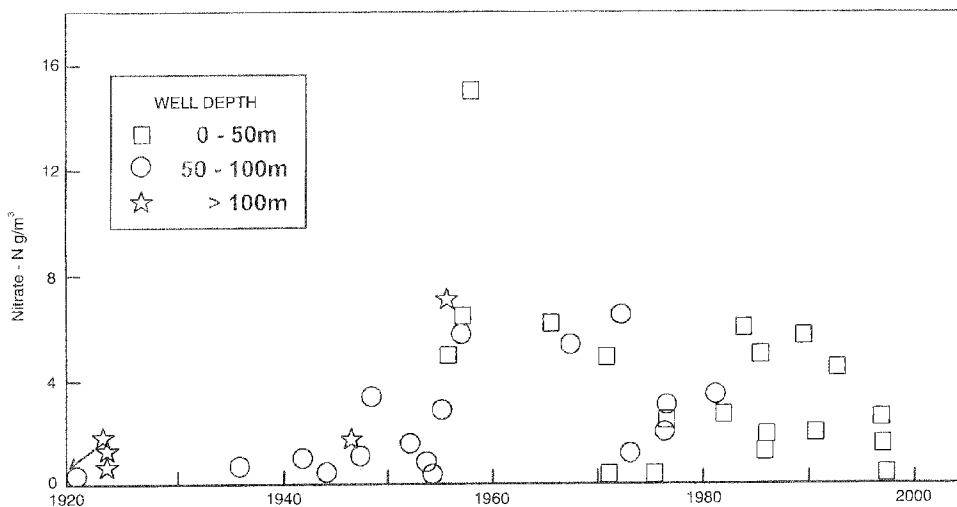


Figure 29.12 Nitrate concentrations and age of groundwater from wells between the Waimakariri and Rakaia rivers, Canterbury (after Stewart *et al.* 2002).

GROUNDWATER ECOLOGY

Groundwater represents the largest freshwater habitat in the world (apart from polar ice), yet very little is known of its biodiversity and ecology. The existence of invertebrates in groundwaters was first reported in 1813-14 by Leach, who found well-shrimps (amphipods) in a well in London. In New Zealand, a remarkable local scientist, Charles Chilton, pioneered the discovery of metazoans (multicellular animals) adapted to complete their entire life histories in deep groundwaters. In 1882, Chilton surprised the local scientific community with his paper describing three blind, unpigmented amphipods (a group of small crustaceans that includes sandhoppers), some up to 20 mm long, in well water from the alluvial gravels beneath his family farm at Eyreton, just north of the lower Waimakariri River in Canterbury. No further

research occurred until, in the early 1970s, Dr W. Kuschel surveyed many wells throughout the country. However, only beetles, snails and mites from these collections were reported in publications. Interest was renewed when Lester Sinton found large numbers of crustaceans in wells at a sewage effluent disposal site in the early 1980s, but further taxonomic work started again only in the late 1990s. Despite this intermittent effort, research shows that the groundwater fauna is diverse and abundant (Scarsbrook *et al.* 2003).

International interest and research in groundwater ecology did not gain momentum until about 1980, when an increasing realisation of the economic and biodiversity value of groundwater faunas in Europe produced sustained investigations (e.g., Danielopol 1989; Gibert *et al.* 1990;

Dole-Olivier *et al.* 1994).

Today, our understanding of the ecology of groundwater systems is sufficient to provide some initial guidelines for their sustainable management, to protect their biodiversity values and their invaluable ecological services. However, resources for researching groundwater biodiversity and ecology lag far behind, considering groundwater's high economic importance and New Zealanders' heavy reliance on it for irrigation, domestic and industrial supplies.

Groundwater habitats and ecosystems

From a biological perspective, aquifers represent a series of subsurface (*hypogean*) habitats variously inter-connected with each other and with surface (*epigeal*) waters. Indeed, an island's (e.g., the North Island's) freshwaters can be regarded as comprising a single, variously inter-connected habitat, with lakes and rivers being but small, albeit highly productive, parts of the entire, mostly subterranean aquatic system. These surface components are dependent on and connected with groundwaters via hyporheic habitats, with complex, dynamic interactions between surface and subsurface waters. Thus, the groundwater realm is increasingly recognised as a distinct ecosystem. The following discussion explores biodiversity, community composition, trophic structure and energy cycling within groundwaters, with special reference to New Zealand.

Habitat types

Groundwaters span a range of habitats that usually are inter-linked:

- parafluvial (beside the surface water) areas of river and lake margins,
- the hyporheic zone (saturated, sub-benthic sediments immediately surrounding the channel of gravel bed streams and rivers) (see Chapter 32),
- groundwaters in limestone or karst (often in caves) (small areas in New Zealand, e.g., Waikato, Hawkes Bay, Nelson, West Coast, Otago) (see Chapter 31),
- groundwaters associated with fractured rocks (e.g., basalt beneath parts of Auckland, greywacke of the Southern Alps), and
- groundwaters flowing through alluvial deposits (e.g., Quaternary sediments of the Heretaunga Plains, Wairarapa Valley, Waimea Plains, Canterbury Plains). Such alluvial aquifers may be either confined or unconfined, each with very different habitat characteristics.

Parafluvial and hyporheic habitats tend to be boundaries or ecotones between surface waters and true groundwaters and are discussed elsewhere (Chapter 32). Spring habitats also differ (Chapter 30). The emphasis in this chapter is on true groundwaters, focusing primarily on groundwaters in alluvial deposits because these are the most extensive and most important economically of New Zealand's groundwater environments. Alluvial aquifers are often remote from surface waters and species inhabiting them are largely restricted to hypogean habitats, whereas cave, hyporheic and parafluvial faunas comprise mixes of epigeic and hypogean species.

Characteristics of groundwater habitats

The main characteristic of groundwater environments is their constant and complete absence of light—this has significant ecological and biological implications. Photosynthesis cannot occur, so that, unless some other pathway is used, the basis of life is *allochthonous* (imported) organic matter. Most groundwater ecosystems, therefore, are limited by the available amount of carbon (Strayer 1994; Poulson and Lavoie 2000). A few highly specialised groundwater communities utilise non-carbon energy sources, for example energy derived from H_2S (Zimmerman and Marston 1994; Sarbu *et al.* 1996) or basalt rock (Stevens and McKinley 1995), but these are unknown in New Zealand.

The second fundamental characteristic is that groundwater habitats comprise a dense, immovable matrix of rock or sediment particles that is relatively inert physically and chemically, with life largely confined to water-filled pores or interstitial spaces within this matrix. Groundwater habitats vary widely, depending on the

substrate particle size and porosity: karst and cave systems that include large underground cavities and tunnels, often with air spaces, are one extreme, whereas compacted clays and poorly fractured bedrock without cavities, both beneath the water table, are two other extremes.

The nature of pore spaces may control the entry and transport of organic carbon, so that food is likely to be a limiting factor in many groundwater environments, especially in fine alluvial aquifers distant from sources of water inflow, where small pore spaces and slow flow rates further constrain the rates of arrival of organic carbon. Additionally, the natural filtering properties of the groundwater matrix may result in a relatively homogeneous source of organic carbon within the ecosystem, constraining the diversity of feeding modes of *stygofauna* (permanent groundwater dwellers), when compared with the highly diverse carbon sources and feeding modes in epigeic aquatic communities.

A third characteristic of groundwater habitats is that they are neither constant nor uniform. Although animals are confined to pore spaces in all groundwater systems, these spaces may be extremely diverse in terms of void size, shape, presence of fines, water velocities, etc., so a wide diversity of fine-scale habitats may be present in groundwater environments (Gibert *et al.* 1994). The Canterbury Plains shallow aquifer, for example, can be imagined as flowing through a highly heterogeneous, 3-dimensional representation of the Waimakariri River bed: a matrix of poorly sorted gravels and sands interwoven with incomplete, random ribbons of clean finer gravels and discontinuous pipes of coarse gravels, cobbles and small boulders; the whole interspersed between irregular lenses of impermeable silty clays.

The absence of light, the inert nature of the groundwater matrix and the constriction of water flows by pores mean that temporal changes in physical and chemical factors are slower in groundwaters than in surface waters. Certainly, changes in water levels, flow rates, temperature and chemical properties can be rapid where pore spaces are large, for example in caves, wide rock fractures and coarse sediments, but changes are generally substantially slower in alluvial aquifers, depending upon pore sizes and water flow rates. Despite this slow change, the physical and chemical properties of groundwater often vary widely. At the Templeton sewage treatment plant in Canterbury, for example, microbial densities, and total organic carbon and dissolved oxygen concentrations in water from wells upstream and up to 300 m downstream of a disposal area differed appreciably at any one time (Fig. 29.13), and within each well over time (Fig. 29.14).

Conceivably, there are finite limits to the depths at which metazoan life occurs in groundwater because oxygen and organic carbon become increasingly scarce with

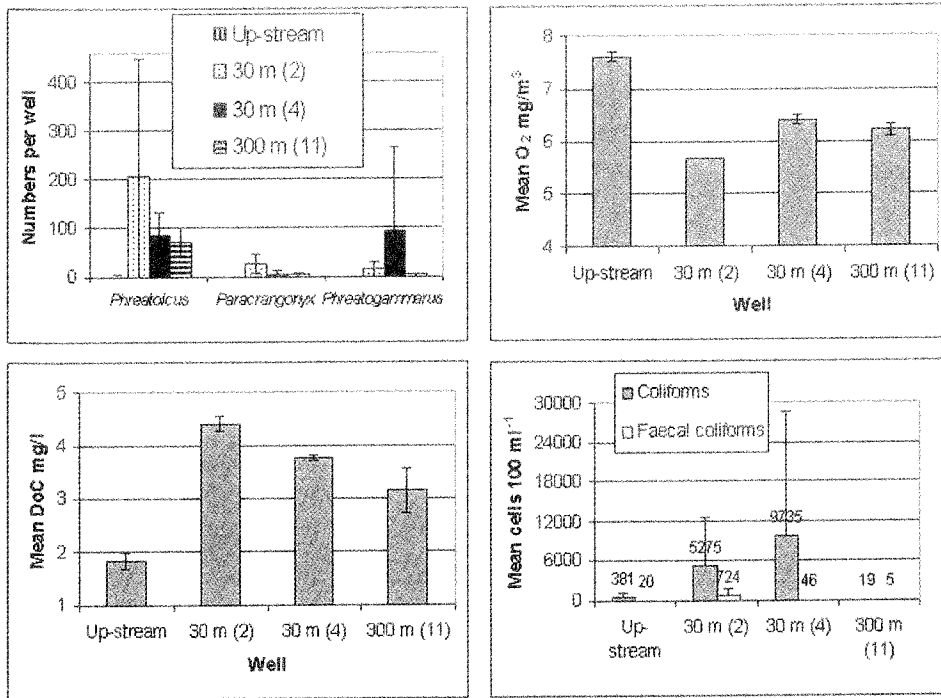


Figure 29.13 Mean faunal abundance, dissolved organic carbon (DOC) concentration, dissolved oxygen concentration and bacterial cell densities (error bars ± 1 SD) in wells upstream and 30 (two wells) and 300 m downstream of an organic enrichment source (Fenwick, unpublished data).

distance from the surface and with greater constriction of water flows. Invertebrates are known to inhabit groundwater in caves more than 100 m below the earth's surface (Strayer 1994) and Protozoa have been recorded from alluvial coastal plain aquifers (i.e., similar to many important aquifers in New Zealand) in the USA to depths of 550 m (Sinclair and Ghiorse 1989). Within the Edwards Aquifer in Texas, two species of catfish are known from wells that have their water source between 400 and 600 m below the ground (Longley 1992), indicating that an entire fauna exists to these depths. Therefore, animals may exist to substantial depths in New Zealand groundwaters, but to date no groundwaters >50 m deep have been sampled.

Biodiversity, structure and functioning of groundwater ecosystems

Animals found in groundwater habitats vary from infrequent, accidental migrants from surface habitats (especially in springs and at cave entrances) to specialised subterranean species restricted to hypogean habitats. Thus, aquatic animals are often classified according to their use of these habitats (Fig. 29.15). *Stygobites* are species specialised to live entirely in true groundwater habitats (Claret *et al.* 1999), which lack primary production by plants and are directly or indirectly reliant on allochthonous energy sources. Benthic (bottom-living) faunas of lakes and rivers are termed *stygoxens* when they show no affinity for interstitial habitats. In contrast, interstitial faunas of lakes and rivers and the hyporheos include species that may be classified as *stygophiles* (occasional or permanent members of the hyporheos) (see Fig. 29.15).

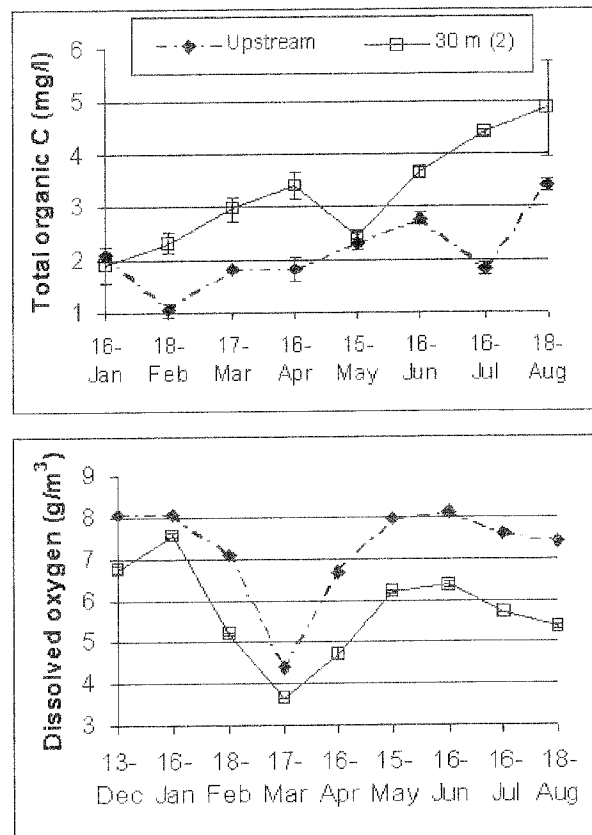


Figure 29.14 Changes in mean total organic carbon and dissolved oxygen concentration (error bars ± 1 SD) in wells upstream and 30 m downstream of organic enrichment in Canterbury (Fenwick, unpublished data).

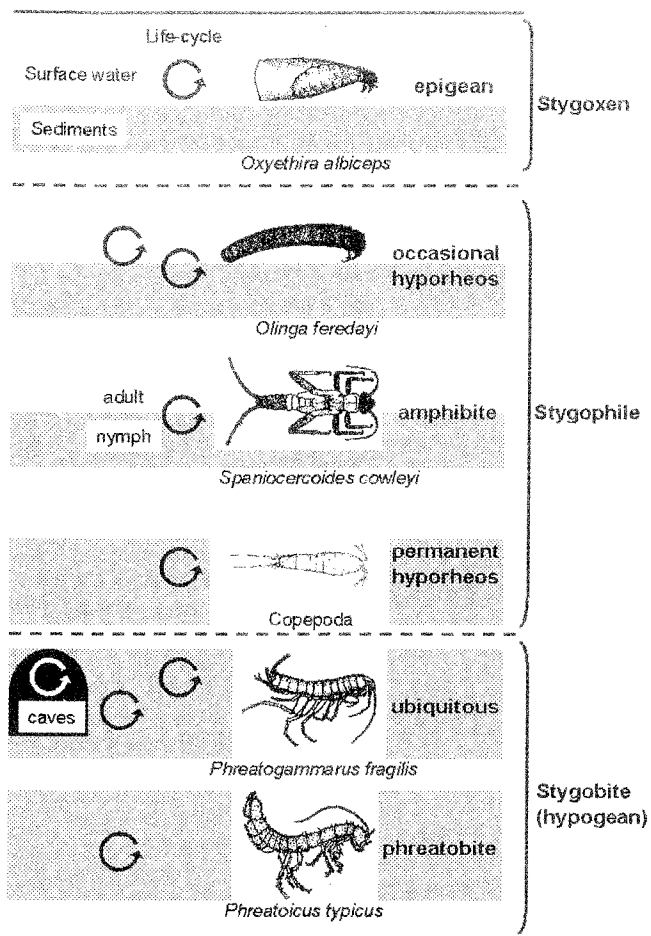


Figure 29.15 Classification of epigeal and hypogean animal habitats showing locations of life histories and some characteristic taxa (modified after Gilbert *et al.* 1994 and Collier and Scarsbrook 2000).

The nature of aquatic life underground

New Zealand's groundwater fauna is poorly known. There are 102 described species, 15 of which are crustaceans (Scarsbrook *et al.* 2003). Several undescribed species occur within the New Zealand stygobitic fauna (Scarsbrook *et al.* 2003), notably smaller crustaceans (copepods, ostracods, amphipods). Given that mites comprise *c.* 17% of groundwater faunas world-wide and that this is one of the better known New Zealand groundwater groups, the total New Zealand groundwater biodiversity probably exceeds several hundred species. Documented, but unpublished, accounts indicate significant additions to this fauna within existing collections (Fenwick 2001; Scarsbrook and Fenwick 2003; Scarsbrook *et al.* 2003).

Some species of most aquatic metazoan groups have adapted to subterranean habitats and evolved adaptations specific to their stygobitic life styles. Loss of body pigmentation is usual because, in the absence of light, it serves no useful purpose. Similarly, true stygobites are

blind, although some may have reduced or unpigmented eyes, indicating that these species forsook epigeal environments more recently than other completely blind, hypogean species. With the loss of eyes, tactile senses apparently take over, so that elongate antennae and greater development of other sensory structures may become important (Gibert *et al.* 2001). More obvious in larger, hypogean crustaceans is a general elongation or slimming of body shape, including a tendency for legs to become more slender and swimming appendages (pleopods) smaller in some. For example, the walking legs (peraeopods) of hypogean phreatoicid isopods (e.g., *Phreatoicus typicus*) tend to be more slender than those of epigeal species (e.g., *Notamphisopus dunedinensis*). Body elongation is not essential for hypogean life, however, and varies, as seen in the contrasting morphologies of *Paracrangonyx compactus* and its close relative, *P. winterbourni* (Fig. 29.16). Conceivably, the slimmer body shape, smaller coxae, thinner peraeopods and much-reduced pleopods of *P. winterbourni* indicate it has adopted a hypogean life earlier, or more completely, than has *P. compactus*.

Variation in body size is regarded as another important adaptation to hypogean life (Gibert *et al.* 2001). Among New Zealand groundwater amphipods and isopods, adult body size varies from < 4 mm to >20 mm, and is probably strongly influenced by habitat, among other factors. For example, animals adapted to Canterbury's coarse-grained alluvial aquifers (e.g., *Phreatogammarus fragilis*) or springs in subalpine shattered rock have access to large interstitial

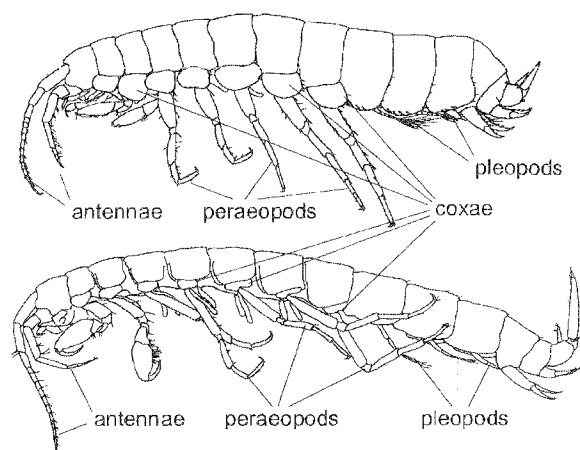


Figure 29.16 Morphological differences between two closely related stygobitic amphipods from Canterbury, New Zealand. *Paracrangonyx compactus* (top, *c.* 9 mm long) has a stouter body and legs, and well-developed coxae and pleopods compared with the more elongated *P. winterbourni* (bottom, *c.* 12 mm long).

spaces with faster water flows, whereas those inhabiting finer-gravel aquifers and springs adjacent to lowland rivers must contend with smaller pore sizes and slower groundwater velocities. Thus, evolutionary pressures differ between the two habitats, one possibly selecting for larger size and the other favouring medium to smaller size. Note, however, that large animals are not always highly mobile—the phreatoicid isopod *Phreatoicus typicus* and the amphipods *Paracrangonyx compactus* and *P. winterbourni* are poor swimmers, but are almost as large as the swift-swimming *Phreatogammarus fragilis*.

Other adaptations to hypogean life include lower metabolic rates, longer ontogenetic development, protracted life histories, increased tactile receptors and responsiveness, and the development of specific behavioural and physiological characteristics.

Groundwater community structure

The number of different species taken from single wells indicates that New Zealand groundwater communities are moderately diverse. Seven metazoan groups (some not identified below class level because of inadequate taxonomy) occurred at Templeton, with 3–6 of these collected from any one well. Attention to smaller invertebrates showed at least nine species present at this site, including six amphipods, one isopod, oligochaetes and copepods. Single wells in South Canterbury and Waimea Plains (Nelson) yielded at least 10 and 6 species respectively (Scarsbrook and Fenwick unpub. data). A lack of taxonomic knowledge of the fauna limits our ability to recognise different species, so that we cannot adequately determine the true biodiversity within an aquifer at present. The recent guide to the New Zealand groundwater fauna (Scarsbrook *et al.* 2003) is a first step, but is far from comprehensive.

A few investigations provide information on the abundances of New Zealand groundwater species or faunas. Mean numbers of up to 250 crustaceans and two other groups (oligochaetes and turbellarians) per well occurred in sewage-contaminated wells, while fewer than two animals per well were found in an uncontaminated well at Templeton in Canterbury (Sinton 1984). The large isopod *Phreatoicus typicus* was the most abundant species present, averaging 207 (SD = 241.7, $n = 4$) individuals per sampling in one well, and as few as 1 (SD = 2, $n = 4$) in others (Sinton 1984; Wilson and Fenwick 1999). In these same wells, four species of amphipods also varied widely in abundance between wells (Fig. 29.13) and sampling occasions (Sinton 1984; Scarsbrook and Fenwick 2003): these variations appear typical of groundwater faunas.

Sampling difficulties present a significant barrier to understanding groundwater biodiversity, and obtaining reliable estimates of the quantitative abundances of species

is even more difficult because of the habitat's heterogeneity and inaccessibility (e.g., see Boulton *et al.* 2003). Indeed, scientists today can sample and photograph abyssal (>3000 m deep) marine faunas more easily and effectively than groundwater faunas below *c.* 5 m depth. Usually, it is impossible to remove or sort representative samples of the groundwater substratum to collect the fauna. Wells provide access to the strata inhabited, but are an artificial situation, possibly trapping individuals entering them and, apparently, excluding others because of the scarcity (<1%) and narrowness (*c.* 1 mm) of openings in casings. For example, very few juvenile or brooding female *Phreatoicus* or amphipods are collected from wells (Wilson and Fenwick 1999). There are also indications that animals avoid wells with steel casings. Quantitative data on groundwater faunas are especially difficult to obtain and difficult to extrapolate to the *in situ* groundwater habitat. At best, abundance data from well sampling may provide indications of the relative abundances of selected species.

Functional ecology of groundwater ecosystems

Energy or organic carbon transformations and flows define the fundamental nature of an ecosystem, making knowledge of these essential to understanding any ecosystem. This is particularly true for groundwater ecosystems, but the nature of the primary organic carbon food sources for the fauna is not obvious. In early studies, including one on the Canterbury Plains, increased abundances of crustaceans and other animals were observed close to sources of organic enrichment, especially sewage effluent. It was speculated that animals were attracted by the potential food source available in the abundant bacteria, bacterial films and organic particles (Husmann 1975; Holsinger 1966; Sinton 1984). Bacteria are ingested and digested by groundwater animals (Sinton 1984; Fenwick, unpublished data), but this is not the full picture.

A series of detailed field and laboratory experiments at Templeton, near Christchurch, including radiotracer work, showed that, although bacteria and fungi were eaten and digested, they were a minor part of the energy sources used by *Phreatoicus* and amphipods. Low concentrations of dissolved and fine particulate organic matter percolated into the groundwater from the overlying oxidation-pond effluent disposal areas, and quickly became incorporated into thin slime layers on all sediment surfaces. Fungi were not present in these layers and bacteria were uncommon. *Phreatoicus* fed by ingesting clay particles and digesting the organic layers from these (Fenwick, unpublished data). Two of the amphipod species examined also ingested larger particles, apparently also digesting their organic coatings. Thus, allochthonous dissolved and particulate organic matter drives groundwater ecosystems, cycling from layers on sediment and gravel surfaces, through grazing animals,

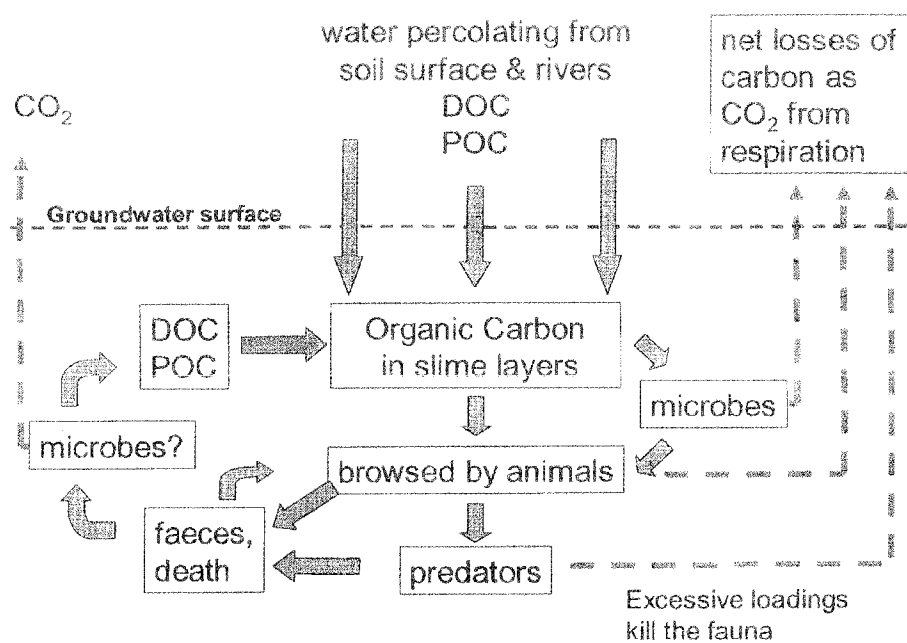


Figure 29.17 Organic carbon pathways in the Canterbury alluvial aquifer system. DOC – dissolved organic carbon; POC – particulate organic carbon (Fenwick, unpublished data).

into scavengers, predators and any microbes, with respiratory losses at every step (Fig. 29.17). Predation, scavenging and other feeding modes are likely also within this trophic web, because most amphipods, probably the most abundant and diverse taxon, scavenge whenever the opportunity is presented (Fenwick in press).

Importance and vulnerability of groundwater ecosystems

Groundwater faunas have significant biodiversity value that must be managed and conserved. They also perform ecological services fundamental to aquifer integrity, helping to maintain the value and usefulness of groundwater resources to humans. Yet, most New Zealand groundwaters are managed as resources with physical and chemical characteristics only, except where there is concern about microbial contamination. Just how important are phreatic faunas to groundwater quality and aquifer integrity? And how vulnerable are these animals to threats from human activities?

Ecological services performed by groundwater faunas

Two New Zealand studies investigating groundwater community responses to pollution from a sewage disposal area in Canterbury found dramatic increases in the abundance of phreatic crustaceans immediately downstream of an effluent disposal site, when compared with an up-stream control site, with attenuation in numbers over about 300 m downstream (Fig. 29.13) (Sinton 1984;

Fenwick, unpublished data). The numbers of animals per well varied appreciably between samplings, but provided general indications of likely abundances within the aquifer itself, and appeared correlated with dissolved organic carbon and dissolved oxygen concentrations, as well as the numbers of faecal and total coliform bacteria in the water at each well (Fig. 29.13). Using a series of explicit assumptions, Sinton (1984) estimated that approximately 20% of the calorific value of the effluent applied to the disposal area would be assimilated by the three principal species in the aquifer below. Another investigation (Fenwick, unpublished data) tracked labelled organic carbon flows to estimate that the larger crustaceans assimilated

>2 t of organic carbon per year over the entire disposal site, and perhaps as much as 1 t/ha/yr (= 100g/m²/yr).

These findings have important implications for groundwater management. The crustaceans clearly are important in maintaining groundwater quality. This occurs in at least three ways. First, potentially harmful microbes are destroyed by animal feeding and digestion. Second, although conversion of large amounts of contaminant material to animal tissue does not remove it from the system, conversion inefficiencies and respiration mean that significant amounts of the contaminant organic carbon are lost from the groundwater ecosystem as carbon dioxide through respiration. Third, by grazing organic layers from particle surfaces, the fauna maintains very fine pore spaces within the aquifer system, conceivably preventing clogging of pores that would slow the rates of groundwater flow and severely degrade groundwater quality through anoxia (lack of oxygen).

Sensitivity of groundwater biota to human activities

The two studies outlined above demonstrate some effects of human activities on groundwater fauna in New Zealand. Periodic massive kills of the fauna observed in some wells, apparently due to unusual effluent irrigation events, were also observed at Templeton (Sinton 1984; Fenwick, unpublished data). Organic enrichment of groundwaters usually increases abundances of phreatic species (Holsinger 1966; Sinton 1984; Notenboom *et al.* 1994; Scarsbrook and Fenwick 2003), but excessive enrichment relative to groundwater flow rates can rapidly

deplete available oxygen, decimating or eliminating invertebrates from caves and groundwater (Holsinger 1966; Sinton 1984; Culver *et al.* 1992). Thus, increased faunal densities tend to be correlated with higher oxygen concentrations and with organic enrichment (Fig. 29.13, see also Husmann 1975), although excess organic loadings can rapidly reduce oxygen availability.

There are very few published accounts of the effects of other pollutants (pesticides, heavy metals, etc.) on groundwater species and nothing is known about the New Zealand fauna in this respect. A few European and North American species of common groundwater groups have been shown to have acute toxicities (adverse health effects from a single exposure) to some common pollutants (e.g., zinc, copper, cadmium, chromium, TRC, Aldicarb), but chronic (repeated exposure) toxicity effects are poorly understood for groundwater animals because they are usually difficult to culture in the laboratory (Notenboom *et al.* 1994). Metals in water and sediments affect animal groups differentially. Increasing concentrations of copper and zinc appear to eliminate amphipods and insects from groundwaters, whereas ostracods and cladocerans (small crustaceans) persist (Notenboom *et al.* 1994), although some hypogean amphipods are capable of regulating their body zinc and copper concentrations (Plenet 1995).

The sublethal effects of pollutants, including endocrine disrupters, on phreatic species and whole communities may be more catastrophic than sudden mass kills in the medium to long term. It is well known that various pollutants interfere with individual growth, development, reproduction and, hence, the ability of species populations to perform their normal ecological and bio-remediation roles. Interference with the normal functioning of these processes is, therefore, likely to disturb the functioning of the ecosystem as a whole.

Groundwater depletion

No information is available on the effects of changes in groundwater level on stygofaunas. However, given that groundwater faunas comprise aquatic organisms adapted for life in water, most species are unlikely to withstand emersion. Most species reported from groundwater are motile; some of the larger crustaceans are highly mobile and probably capable of active retreat during slow natural declines in water levels. More rapid changes, such as those resulting from rapid draw-down caused by pumping, are likely to strand both larger and smaller animals above the receding water table, especially any less mobile species. If water levels rise again, some re-colonisation may be immediate, but the densities of invertebrates, especially the larger ones, will take some time to return to original levels, because groundwater species characteristically have low reproductive rates (Holsinger 1966; Gibert *et al.* 1994; Notenboom *et al.* 1994; Wilson and Fenwick 1999).

THE IMPORTANCE OF GROUNDWATER SYSTEMS IN NEW ZEALAND

Groundwater is of immense economic value to New Zealand, especially because it is effectively stored in aquifers and can be drawn upon in dry periods and allowed to recover when the rains return. It is usually better protected from pollution than surface water although, as noted above, if it does become polluted it is often very difficult to clean up. Throughout New Zealand, some 127 m³/sec of groundwater has been allocated to economic activity (Lincoln Environmental 2000), representing 30% of the total allocated water. This is a peak rate that is unlikely to be reached, because not all water allocated would be used continuously or at the same time, and much less irrigation water would be used in a wet season. Of the allocated total, 17% is for public water supply, 9% is for industrial use and 74% is for irrigation. Further, most rivers derive their low flow from dispersed groundwater seepage and many of our most attractive lowland streams are spring fed. Wetlands are also sustained by groundwater seepage and, being shallow, are particularly vulnerable to lowering of water tables.

It is clear that the economic value of groundwater resources is tied to the healthy functioning of the groundwater ecosystem, which, in turn, is dependent upon groundwater quality and biodiversity. Management of groundwater resources, therefore, requires a significant change in perspective, away from the central focus on groundwater as a physical resource to be managed in terms of supply and demand. Instead, sustainable management depends upon an improved understanding the physical, chemical and biological nature of groundwater systems and the vital role of ecological processes in maintaining groundwater quality.

Vulnerability of groundwater systems

Like all finite resources, groundwater is vulnerable and must be managed sustainably. However, the finite and biological nature of groundwater resources is not obvious, largely because it is invisible. Increasing demand is already leading to over-allocation in some areas, with both adverse economic and environmental consequences. Excessive pumping lowers water tables or potentiometric heads in aquifers, which may cause shallower wells to run dry, spring-fed stream flows to decrease, wetlands to dry out, or seawater to flow inland into coastal aquifers. Streams and wetlands are especially vulnerable because they are fed from the "top" water in an aquifer and may dry out at an early stage of water table lowering.

Intensification of agriculture based on irrigation may have a double environmental effect: the potentiometric heads may be reduced and by-products from the increased

agricultural production may be washed into the aquifer and pollute it.

Unconfined aquifers are particularly vulnerable to pollution because some of their recharge will be by percolation of water directly from the land surface, and this water may transport pollutants such as nitrates, micro-organisms and agrichemicals, throughout much of the system. Nitrate pollution of aquifers has already occurred in many parts of New Zealand, and traces of pesticides have also been detected in a few wells. The problem may be lessened (or postponed) where the soil is thick and the water table lies further below the soil surface. Groundwater moves slowly, often at considerable depth. Pollution is, therefore, insidious, and groundwater will be difficult or even impossible to clean up once it is polluted. Consequently, it is imperative to prevent pollution from occurring in the first place.

Although we now know that groundwater quality and availability are linked to its ecological characteristics, we lack much of the detail. The limited research available from New Zealand and overseas indicates considerable cause for concern, however. Groundwater communities respond in much the same way as epigeal (surface) communities to increased and excess organic loadings, but they are likely to be more vulnerable and take considerably longer times to recover from contamination. Crustaceans, especially amphipods, are among the most sensitive of aquatic organisms to a range of common pollutants (e.g., Thomas 1993) and, because groundwater faunas are dominated by these animals, they are likely to be even more vulnerable to land use and other human activities. Consequently, more attention to protecting the ecological processes in groundwater is vital to their future sustainability.

Future research priorities

Demand for groundwater is increasing rapidly in most parts of New Zealand and, given the resource's finite nature, competition for the resource will intensify. From this it is apparent that management of the resource will become ever more important and effective management must be based on sound science. Priority should continue to be given to methods that quantify groundwater resources and their dynamic behaviour, because both economic and environmental management decisions must be based on an understanding of these characteristics. As agriculture, horticulture and other economic activities intensify, the risk of pollution increases, so the focus must be on prevention, which will depend on better education and planning.

The greatest gap in our scientific knowledge of groundwater is our understanding of this resource as a living ecosystem. Only with a far better understanding of this ecosystem, its functioning and constraints, can we begin to establish better policies for the effective management of

the overall groundwater system that encompass more than simple allocation issues.

Research into groundwater ecology has far lagged behind and further work is urgently required because of the vital ecological services provided by the fauna. Taxonomic research is essential because this defines the units of biodiversity making up the ecosystems. Without the means to consistently recognise and define species, it is very difficult to assess biodiversity values and monitor how these change in response to human activities. It is also very difficult to understand the functioning of ecosystems. This type of research must aim to establish identification methods, so that species distributions can be quickly determined to help establish conservation priorities.

Attention should be given to understanding the ecological tolerances of as many species as practical, with the aim of developing a faunal index of groundwater condition or health. Such an index would greatly facilitate management in the face of increasing human pressures on this invaluable resource.

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Chapter 28

Wetland ecosystems

Brian Sorrell and Philippe Gerbeaux

INTRODUCTION

Wetlands are arguably the least well understood freshwater ecosystems in New Zealand. There has been far less research on the structure of biological communities and processes governing the ecosystem structure of New Zealand's wetlands than on our rivers and lakes, and considerably less is known about them than in many other nations. Although the vegetation of wetlands has been described in detail (Cockayne 1928; Cranwell 1953; Burrows 1969; Dobson 1979; Campbell 1983; Johnson and Brooke 1998; Wardle 1991), little is known about their algal and invertebrate communities. Almost nothing is known about the microbial flora, and our understanding of biotic interactions and abiotic responses is weak. Even larger animals such as fish and birds have been somewhat neglected, with the factors controlling their distribution and abundance in wetlands less well understood than those of rivers or lakes, although some information on fish and bird species in wetlands is available (Best 1979; Ogle and Cheyne 1981; O'Donnell 1985; McDowall 1990; Heather and Robertson 1996). The bird communities of New Zealand wetlands and their ecology are discussed in Chapters 18 and 26, and are not covered in this chapter.

In New Zealand, the formal definition of wetlands is derived from the Resource Management Act (1991), and includes "permanently or intermittently wet areas, shallow water or land/water margins that support a natural ecosystem of plants and animals that are adapted to living in wet conditions". This relatively broad definition includes a range of shallow-water environments with disparate biological communities, but which share a number of common features that distinguish wetlands from other terrestrial and freshwater habitats:

- shallow standing water and/or waterlogged soils (see Chapter 20 for a detailed description of wetland hydrology),

- anoxic conditions (the absence of oxygen) in the soil, and
- dominance by **emergent** aquatic plants (see Chapters 14 and 24). The deeper and faster-flowing water of lakes and rivers excludes emergent vegetation except at the margins, and a wetland landscape is therefore one in which the large bulk of the biomass is present as emergent plants.

Wetlands, as shallow aquatic environments, have often been called **ecotones**, i.e., transitional zones between terrestrial and aquatic ecosystems. The classical feature of an ecotone is that it is an environment that has a mixture of the characteristics of the two adjacent environments, and hence provides habitats for plants and animals from both. Ecotones therefore often have very high species diversity, and this is true of many wetlands. Ecotone environments also tend to be sinks for inorganic nutrients and exporters of organic material, and this is also true for many wetlands. However, the term ecotone also obscures the great diversity of wetland types, as there are also wetlands with naturally low species diversity and wetlands that accumulate rather than export organic matter. In New Zealand, low species diversity may be a feature of wetlands high in native character, with high species diversity sometimes reflecting the invasion of undesirable weeds and pests. New Zealand has a remarkably diverse range of wetland types, and descriptions of these are available from Johnson and Brooke (1998) and Johnson and Gerbeaux (2004). A classification scheme for different wetland types is now available (Ward and Lambie 1999), based primarily on hydrological features and secondarily on chemical and vegetation characteristics. The scheme evolved from an original contribution by Thompson in Stephenson *et al.* (1983) and has been slightly refined by Johnson and Gerbeaux (2004). The major hydrological classes of wetlands are also described in Chapter 20.

Much of the early research in wetland ecology consisted of simple descriptions of plant and animal communities, with little examination of the abiotic factors controlling their distribution or competitive interactions. The amount of wetland research, and its scientific rigour, began to increase dramatically in the 1960s, driven by the environmental politics of the time and the dawning recognition of the values of wetlands in the landscape. This was the time in which modern ecosystem approaches were first applied to wetland ecology, and wetland researchers now need a broad understanding of surface and groundwater hydrology, and water and soil chemistry, as well as plant, animal and microbial ecology. Increasingly, modern wetland ecology stresses the strong links between the biotic structure and the hydrological and chemical environment controlling it, and looks to developing quantitative methods to describe them.

Because of their relatively shallow water and position in depressions in the landscape, wetlands are often highly sensitive to changes in nutrient and sediment input from the catchment due to land management practices. Whereas larger and deeper water bodies may have some resilience to disturbance, wetlands may change more rapidly in response to hydrological and nutrient perturbations. Many have relatively short natural lifespans (hundreds or possibly thousands of years), and the most frequent disturbance factors—drainage, excessive nutrients and sedimentation—accelerate the drying and infilling that remove wetlands from the landscape. The wetland ecology in New Zealand must also be examined in the context of the wholesale post-European destruction of wetlands, which has left small, fragmented remnants comprising less than 10% of the wetland area present in 1850 (Cromarty and Scott 1996). Human modifications to wetlands and wetland management are discussed in Chapter 40.

FACTORS CONTROLLING WETLAND DEVELOPMENT

Wetlands have specific hydrological, physical and chemical features that encourage colonisation by emergent aquatic macrophytes, with these macrophytes then providing the three-dimensional structure that is the habitat for other organisms (algae, animals and microbes). The linkages between these factors are often summarised diagrammatically, as shown in Figure 28.1. The diagram emphasises how the hydrological regime, itself a function

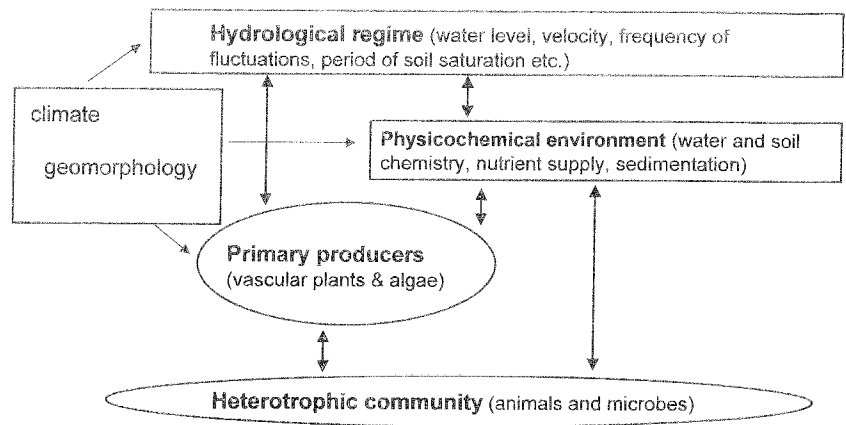


Figure 28.1 Illustration of the links between hydrology, physicochemical environment and biotic structure in wetlands (modified from Mitsch and Gosselink 2000). Double-headed arrows indicate that hydrology controls physicochemical conditions and organisms, but there are also feedback effects of organisms on hydrology (e.g., due to biomass accumulation) and chemistry (e.g., due to photosynthesis in water raising pH).

of climate and geomorphology, is the dominant factor modifying the physicochemical environment (nutrient and sediment inputs and reactions), which in turn determines which organisms can survive in the system. There are also feedback effects—the hydrology is itself modified by the physical environment (e.g., by sediment accumulation) and biota (e.g., by transpiration). Absent from the diagram are some of the direct effects of hydrology on biota (e.g., immersion) which are also important in determining community structure.

Wetland hydrology and hydroperiod

The hydrology of wetlands is described in detail in Chapter 20. In this section we provide a brief overview of the hydrological parameters that are most important in determining biological structure. Hydrology is the single most important factor controlling the establishment and maintenance of wetlands, constraining which organisms grow where, and how productive they are. Wetland hydrology includes the depth of water in the wetland, its fluctuation and the frequency of inundation, and the wetting and drying cycles. The terms **water regime** and **hydroperiod** are frequently used for those aspects of hydrology that control ecosystem structure. Both refer to the frequency and duration of inundation or saturation of an ecosystem, often in terms of the seasonal pattern of water levels in a wetland.

The most important effects of hydrology on biota are indirect. Hydrology affects nutrient availability and soil anaerobiosis (see following section), and the salinity of coastal and estuarine wetlands. Relatively few wetland plants are salt-tolerant and the change in species com-

position along a salinity gradient is dramatic, from relatively diverse freshwater communities to species-poor estuarine communities (Partridge and Wilson 1987; Haacks and Thannheiser 2003). Nutrients are carried in by rainfall, surface streams and rivers, and groundwater, and removed by outflow of water. At one extreme, true **bogs**, which have all their water derived from rain, have very low nutrient concentrations and are dominated by a small number of plants capable of survival in very infertile conditions. Most of the nutrient load in wetlands is derived from groundwater (Raisin and Mitchell 1995), and the most nutrient-rich wetlands are characterised by a small number of highly productive species such as raupo (*Typha orientalis*). **Swamps**, which are affected by surface run-off and groundwater, and **fens**, which have hydrological input from both rainfall and groundwater, often have the greatest species diversity, as their intermediate nutrient levels allow more species to survive than in bogs, but do not allow highly productive species to exclude slower-growing plants. See Chapter 20 and Johnson and Gerbeaux (2004) for more details on differences in hydrology and vegetation in different wetland types.

Aspects of the water regime that directly affect vegetation include mechanical action (waves and currents), and the effects of standing water depth, such as light attenuation, oxygen deprivation and limitation of the surface area of stems and leaves available for photosynthesis. Only a very small proportion of vascular plants can survive flooded conditions, so the first and most important consequence of wetland hydrology is the exclusion of flood-intolerant species and hence a gradient of decreasing species diversity with depth. Wetland plants differ in their degree of **flooding tolerance**, i.e., the depth and duration of flooding they can withstand, and organise themselves along depth and disturbance gradients accordingly. Stable, deep water leads to the dominance of one or two of the most flood-tolerant species—in New Zealand, this will typically be raupo (*T. orientalis*), club rush (*Schoenoplectus tabernaemontani*), or tall spike rush (*Eleocharis sphacelata*). The highest species diversity usually occurs in wetlands with moderate, slow fluctuations in water level, which allows a complex of species differing in flooding tolerance to co-exist. Rapid, large fluctuations in water level are highly damaging to wetland vegetation, because few species can tolerate the associated wetting and drying cycles and shoreline erosion (Mark 1987; Gerbeaux 1993). The degree of water throughflow in the wetland is also important. Stagnant conditions allow organic sediments to accumulate and increase anaerobiosis, providing an environment tolerated by relatively few species, whereas flowing water increases nutrient and oxygen availability, enhancing both species diversity and plant productivity.

Figure 25.1 also indicates that organisms can have feedback effects on hydrology due to their growth and activities. Macrophytes, which develop dense growth in wetlands, greatly reduce the water velocity and increase sedimentation rates and mixing processes, which can change rates of nutrient cycling (Howard-Williams and Downes 1993). Their biomass accumulation and transpiration can also lead to long-term drainage of wetlands or raising of the soil above the water table. Animals may also have effects on hydrology. The larger bird species harvest and transport large amounts of plant material when nesting, which may change directions of water flow. However, New Zealand wetlands lack any large mammals and reptiles, such as beavers and crocodiles, that may have large effects on hydrology elsewhere.

Biogeochemistry and nutrient relations

Wetland soils

Soils are an integral component of biogeochemical processes. They are the site of chemical reactions that dominate wetland nutrient cycling, and the primary source of available nutrients for plant growth. Soil characteristics are important in wetland classification, and are often used to identify wetland classes along with water regime (see Ward and Lambie 1999; Johnson and Gerbeaux 2004). Wetland soil types are often called “hydric” soils due to chemical features resulting from waterlogging (see below), and can be sub-divided into two major types—mineral soils (largely composed of inorganic material) and organic soils (comprising large fractions of plant-derived organic matter).

Soil formation results from the influence of several environmental factors, including soil parent material, topographic position, climate, and biological factors, particularly vegetation (Molloy 1998). The dominant soil type in many New Zealand wetlands is organic soil, the distinctive characteristics of which are described in detail by Hewitt (1998). These soils all contain some organic soil material (i.e., that which has 17% or more organic matter) within their depth profile. The soil parent material may even lack any significant mineral component, as in the case of peat, which accumulates from vegetation in wet hollows and develops into organic soils that can hold up to 20 times their weight in water. Soil material is defined as peat when the organic matter content is 50% or more (Taylor and Pohlen 1979). The most significant peat-forming plants are *Sphagnum* mosses and wire rushes in the family Restionaceae (Davoren *et al.* 1978; Thompson 1987; Clarkson 2002; Johnson and Gerbeaux 2004).

Gley soils are the other key soils of wetlands. They are mineral soils and are often located in lower parts of the landscape affected by a high water table or seepages. Under such conditions iron oxides, which are not readily soluble,

are chemically reduced (see next section) and the ferric iron changes to ferrous iron, which is responsible for the blue-grey colour seen in these soils. Gley soils and organic soils represent the original extent of most New Zealand wetlands (Molloy 1998). However, in the cool to mild and humid climate of the West Coast of the South Island, some wetlands have also developed as “pakihi” on perch-gley podzolised soils that evolved on glacial outwash deposits (Mew 1983). Similar podzolised soils are found in the gumland of the far north of the North Island. These poorly drained soils were degraded through the removal of their original forest cover and are extremely infertile.

Soils can develop over long period of times and so can their properties. The availability of nutrients (fertility) may increase initially, but often over thousand of years it begins to decrease (Molloy 1998). This may in turn cause a shift in the vegetation cover and the type of wetland the soil can support.

Most natural wetland soils are acidic and accumulate organic matter (Table 28.1). This is particularly so for bogs, which have little input of more alkaline groundwater. Nutrient uptake by plant roots, particularly of NH_4^+ , the dominant nitrogen form in wetlands (see below), is accompanied by proton (H^+) excretion from root cells in order to maintain their charge balance (i.e., excretion of one positively charged H^+ ion per positively charged NH_4^+ ion assimilated), and this further lowers pH. Although plant proton excretion is a feature of all wetland soils, it depresses pH more in bogs because of the low buffering capacity of rainwater compared to groundwater. Estuarine soils, which are a mixture of slightly acidic freshwater and alkaline seawater, tend to have a more neutral pH.

The organic matter contents of soil are expressed either directly by combustion as LOI (loss on ignition), or as Total Carbon (TC, Table 28.1). Bogs have high organic matter accumulation, due to the low hydrological exchange and low decomposition rates, which are limited by low nutrient concentrations and low pH. On the other hand,

hydrological throughflow exports much of the organic matter from swamps, and greater nutrient availability also leads to higher rates of organic matter decomposition. The wetlands with the lowest organic matter contents are usually estuaries, which have very high hydrological energy due to tidal action, importing mineral sands from downstream and exporting organic matter.

Bogs are also characterised by very low total nitrogen and phosphorus concentrations relative to other wetlands. Differences in nutrient status between wetlands are therefore often expressed from the ratios of carbon, nitrogen and phosphorus. The higher C:N, C:P and N:P ratios of bogs demonstrate how both plant and microbial growth are more limited by the lower availability of nutrients in these wetlands than in swamps. The nutrient that determines the limits of biological production depends on available (inorganic) fractions in the soil and water, and hence on microbial decomposition rates (see below). Total nutrient concentrations in swamps receiving runoff from human activities can be many times greater than in less disturbed sites (see Table 28.1), and this can have dramatic effects on their community structure (see Chapter 40).

Oxygen and redox potential

Flooding of the soil in wetlands quickly results in anaerobic conditions. Once the soil pores are filled with water, microbial and animal respiration deplete the soil oxygen faster than it can be replenished by diffusion from the overlying water, because gases diffuse 10,000 times slower in water than in air. What follows is a predictable sequence of increasingly anaerobic nutrient transformations, as more energetically efficient bacteria exhaust the electron acceptors they use in respiration and are replaced by increasingly inefficient groups (Fig. 28.2). Aerobic, oxygen-respiring bacteria disappear first, followed by those that respire nitrate, manganese, iron, and sulphate. The least energy-efficient microbes, the meth-

Table 28.1 Mean and range of soil parameters, sampled at 0–10 cm soil depth in bog, swamp and estuarine wetlands in New Zealand (number of sites in brackets in site row, mean value with range in brackets in parameter rows). TC = total carbon, TN = total nitrogen, TP = total phosphorus. Data of Clarkson *et al.* (2003).

	Bogs (6)	Swamps (17)	Estuaries (8)	All sites (31)
Soil pH	4.0 (3.4 – 4.4)	5.2 (4.1 – 5.9)	6.4 (4.9 – 7.1)	5.3 (3.7 – 7.1)
TC (mg/cm ³)	92.7 (24.1 – 239.8)	39.8 (5.2 – 100.6)	18.9 (7.1 – 36.7)	54.6 (5.2 – 239.8)
TN (mg/cm ³)	0.82 (0.02 – 1.83)	2.12 (1.15 – 3.24)	1.22 (0.63 – 2.53)	2.32 (0.02 – 13.3)
TP (mg/cm ³)	0.08 (0.01 – 0.20)	0.28 (0.15 – 0.59)	0.31 (0.15 – 0.64)	0.22 (0.01 – 0.64)
C:N	48.5 (35.9 – 79.7)	18.0 (14.2 – 30.6)	16.6 (11.2 – 23.2)	23.1 (11.2 – 79.7)
C:P	1904 (533 – 4221)	163 (45 – 435)	69 (15 – 137)	447 (69 – 4221)
N:P	39.0 (20.6 – 81.6)	9.1 (4.0 – 20.6)	4.6 (0.7 – 8.1)	13.2 (0.7 – 81.6)

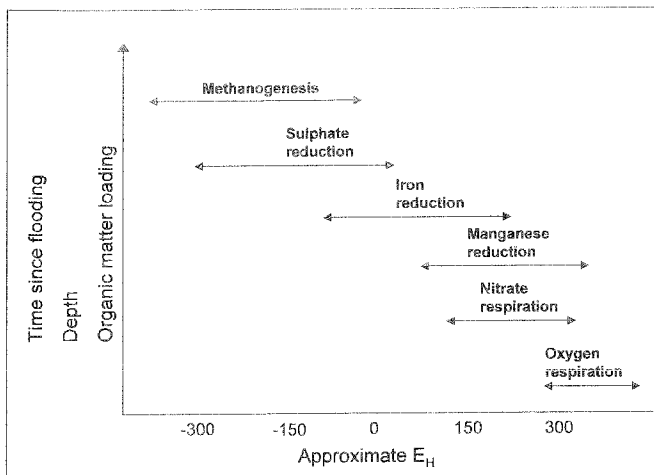


Figure 28.2 Microbially-mediated oxidation-reduction (redox) reactions in soils and their relationship to time since flooding, depth, and levels of organic matter and nutrients. The redox potential (E_H) scale shows the approximate E_H range at which the reactions occur in freshwater wetland soils, compiled from a range of field and laboratory studies from New Zealand and overseas.

anogenic Archaea, dominate the most anaerobic soils, such as those in lowland, nutrient-rich swamps with high levels of organic matter. In most cases, the slow flux of oxygen from the overlying water allows the various groups to co-exist along a depth gradient, with a thin, oxidised aerobic layer at the soil surface.

Soil respiration increases with organic matter levels, so the soil in more productive, nutrient-rich wetlands tends to become more reducing along this gradient. The relative concentrations of the oxidised and reduced forms of the reactions shown in Figure 28.2 are expressed by the **redox potential** (E_H), a measure of the degree of soil oxidation or reduction. Soil oxidation status is one of the most important parameters that structures plant communities in wetlands, because it affects the amount of nitrogen and phosphorus available for plant growth (see following section), and because plants differ in their ability to

maintain live root biomass in anaerobic soils. E_H has therefore been one of the most widely used—and abused—parameters for characterising wetland environments. E_H measurements are strongly affected by differences in pH and temperature in the soils being measured, which need to be carefully corrected for, and are not always comparable in soil types that differ greatly in organic matter content. Measurements of transitions derived under very stable laboratory conditions often may not be applicable in the field. Methanogenesis, in particular, tends to occur at significantly higher E_H values in the field than in the laboratory, especially in freshwater wetlands where concentrations of sulphate are relatively low and limiting for sulphate-reducing bacteria. E_H nevertheless remains a useful coarse approximation of the degree of soil aeration, provided care is taken during measurements, and often correlates well with soil drainage and vegetation patterns. In a relatively pristine freshwater wetland, with little drainage or pollution impact, soil E_H is usually in the range of +100 mV to +300 mV. The lowering of water tables by drainage oxidises the soil and raises E_H above +300 mV, and increases in soil anaerobiosis associated with pollution such as eutrophication lower E_H below 100 mV (Table 28.2). Very negative values (< -100 mV) occur in soils with very high rates of soil respiration and accumulation of products of anaerobic respiration such as volatile fatty acids (e.g., acetate, propionate, butyrate) and sulphide.

Nutrient availability

Nutrient availability is second in importance only to hydrology in controlling ecosystem structure in wetlands. Hydrology itself controls much of the nutrient input and output from wetlands, but most of the nutrient pool present in wetlands at any one time is tied up in either living or dead plant biomass, including soil organic matter. Internal recycling of nutrients (uptake by living organisms and release from dead, decomposing biomass) therefore dominates in most wetlands, unless there is very rapid throughflow or the incoming nutrient load is very high.

Table 28.2 Range of soil redox potentials (E_H) measured at 10 cm depth in some South Island wetlands.

Fluctuating water tables introduce oxygen into the soil, elevating E_H compared to that of similar permanently flooded sites. Increasing nutrient loads increase soil respiration and lower E_H .

Wetland	Soil characteristics and hydroperiod	E_H (mV)
Bullock Creek (Buller District)	Peat, fen, fluctuating water table	+200 – +250
Lake Evelyn (Canterbury)	Mineral, swamp, permanently waterlogged	-90 – +160
Mossburn Swamp (Canterbury)	Organic, swamp, permanently waterlogged	-158 – +140
Styx Mill (Christchurch)	Organic, swamp, fluctuating water table	+160 – +180
Travis Wetland (Christchurch)	Organic, swamp, permanently waterlogged	-100 – +170
Cockayne Reserve (Christchurch)	Organic, swamp, eutrophicated	-200 – +70

The net effect of wetlands on nutrient fluxes in catchments is generally to act as a sink for inorganic nutrients, transforming them into soluble or particulate organic forms that remain within the wetland or are released downstream.

In most freshwater wetlands, the nutrient determining the limit of growth is either nitrogen or phosphorus, although levels of other nutrients such as potassium, iron and manganese can also occasionally limit growth. In the case of phosphorus, only one form, soluble **orthophosphate** (PO_4^{3-}), can be taken up by plants, and phosphorus availability is a function of the difference between their orthophosphate uptake and its rate of microbial regeneration from dead organic matter. Much of the phosphorus in wetland soils is unavailable to primary producers, occurring either as mineral forms or strongly bound to soil particles. Soil oxidation affects phosphorus availability, with PO_4^{3-} precipitating into less available forms in well-oxidised soils, and becoming more available as soils become more reducing. Nitrogen cycling is more complicated, due to the gaseous inputs and outputs driven by denitrifying and nitrogen-fixing bacteria, and the redox-dependence of concentrations of nitrate (NO_3^-) and ammonium (NH_4^+), the two forms available to plants. Little nitrate is available for plant growth in wetland soils because the absence of oxygen inhibits nitrification. This is likely to lead to a limitation of plant growth due to low nitrogen levels, because plants grow best at an $\text{NH}_4^+:\text{NO}_3^-$ supply ratio of approximately 2:1. (There is a trade-off for plants here: NH_4^+ is more energetically efficient as an N form because it can be directly used in metabolism, whereas NO_3^- has to be reduced internally to NH_4^+ by the enzyme **nitrate reductase** before it can be used. However, plants grown solely on NH_4^+ suffer **ammonium toxicity** because the cells in their roots release excess H^+ ions in attempting to maintain charge balance when taking up NH_4^+ , an injury termed **cytoplasmic acidosis**).

ECOSYSTEM STRUCTURE AND FUNCTION

Macrophytes

The various growth forms of aquatic macrophytes are described in detail in Chapter 24 (Littoral communities: algae and macrophytes). The depth range of water in wetlands is such that tall emergent species form much of the plant biomass, although all the other growth forms described in Chapter 24 occur as minor vegetation components, depending on differences in depth and hydroperiod within and between sites. The plant communities respond to the interaction of a range of factors such as elevation, basin morphology and topography, hydrology, nutrients and substrates. A comprehensive description of the vegetation types found in New Zealand

wetlands is provided by Wardle (1991) and by Johnson and Gerbeaux (2004).

Control of community structure by hydroperiod is a striking feature of all wetlands. Only a very small proportion (< 5% worldwide) of vascular plant species are capable of surviving in waterlogged soils and standing water, and there are considerable differences amongst these species in their flooding tolerance (the duration of soil waterlogging, and duration and depth of flooding they can survive). Emergent plants have a range of specialised adaptations that allow them to tolerate flooding. The occurrence of different species at different depths is therefore a consequence of a number of adaptive strategies (Keddy 2000):

- *Differences in sexual recruitment* (germination and seedling establishment). In wetlands seeds germinate mostly in the shallow zones that are occasionally exposed during periods of drought. Even the deeper-growing species, such as *Typha* and *Schoenoplectus* spp., tend to germinate and establish in this zone and then colonise deeper water via the vegetative growth of underground rhizomes. Many wetland plants time their seed production to occur during the non-flood season, and produce large seed banks to take advantage of less predictable wetting and drying cycles. Some also produce buoyant seeds that can be dispersed by water.
- *Differences in vegetative reproduction*. A disproportionately large proportion of wetland plant species have efficient vegetative reproduction by rhizomes, tubers and stolons (Grace 1993). Vegetative growth allows plants to grow in areas where seedlings cannot survive, such as deep water, highly anaerobic soils, and areas with high wave action.
- *Morphological plasticity*. Emergent macrophytes need to keep a significant proportion of their shoot biomass above the water surface for photosynthesis because, unlike submerged macrophytes (Chapter 24), they are very poor at photosynthesising underwater. Many have a **depth accommodation response**, in which leaves or shoots elongate very rapidly when water levels increase (Clevering *et al.* 1996), or will shift their biomass allocation patterns by increasing the ratio of above-ground to below-ground biomass in order to maintain aerial biomass (Fig. 28.3). **Heterophylly** (the production of morphologically distinct leaves above and below the water surface) allows species to exploit both aquatic and atmospheric photosynthesis—the submerged leaves are thin and finely divided to maximise underwater photosynthesis, with the emergent leaves retaining morphological adaptations for aerial photosynthesis, such as cuticles and stomata.
- *Adaptations for oxygen deprivation*. Wetland soils are often anoxic and/or reducing, and wetland plants have internal anatomical adaptations to provide oxygen to

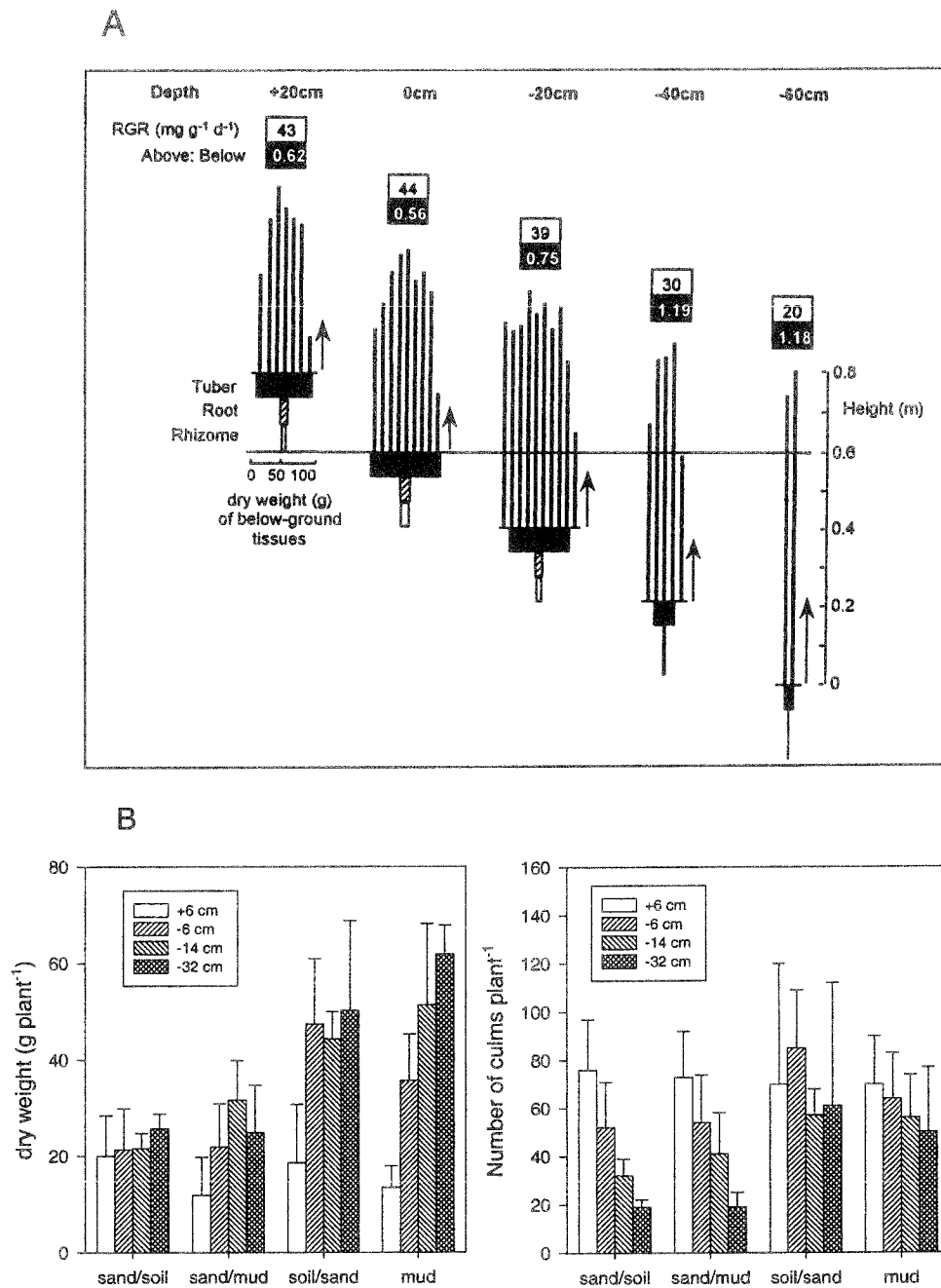


Figure 28.3 A) Changes in the ratio of above-ground to below-ground biomass and shoot structure with water depth for the shallow-growing species *Bolboschoenus medianus*. In deeper water, productivity is lost and a small number of very tall shoots are produced to maintain aerial photosynthesis, with resources being diverted from below-ground to above-ground tissues. RGR = relative growth rate. Reprinted from Blanch *et al.* (1999) with permission from Elsevier. B) Relationship for the more flood-tolerant, deep-growing *Eleocharis sphacelata*, in four soils with increasing nutrient richness and four water depths. In this species there is the same shift to taller shoots, but biomass and number of shoots are maintained, or even increase, with depth, provided there are sufficient nutrients available. Modified from Sorrell *et al.* (2002).

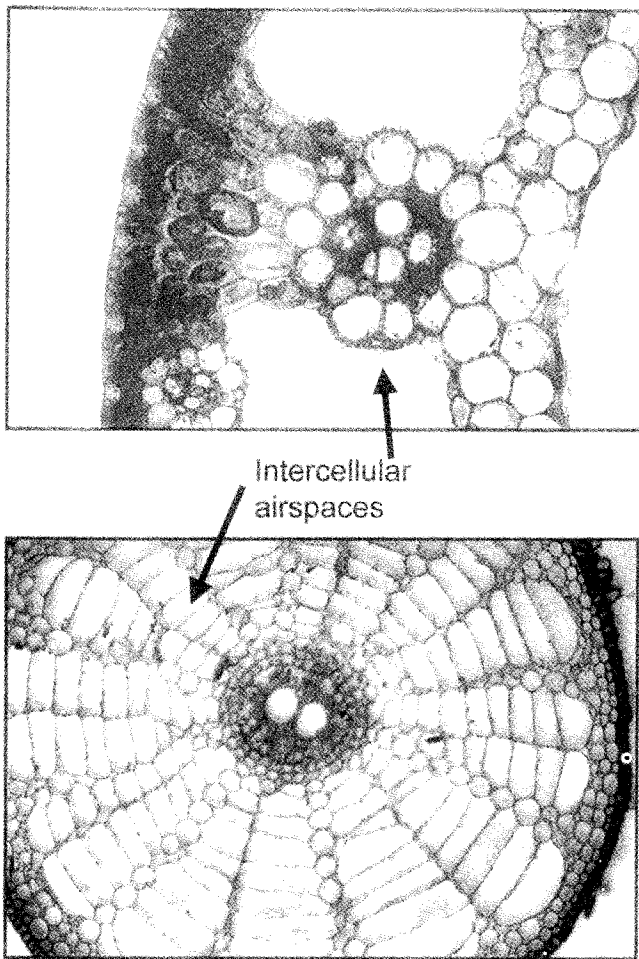


Figure 28.4 Microscopic cross-sections of a shoot (above) and root (below) of the deep-growing emergent macrophyte *Eleocharis sphacelata*, showing the large intercellular airspaces that develop in the tissue.

Photo: B. Sorrell

their below-ground roots and rhizomes. These consist of large airspaces (*lacunae* or *aerenchyma*) in roots and stems that allow the transport of oxygen from the aerial parts of the plant to the underground tissues (Fig. 28.4). Species differ in their degree of airspace development, with species that have the greatest airspace development generally being those that survive best in flooded and reduced soils, and in deeper water. In most species, oxygen is transported through the airspaces by simple diffusion, but a few highly specialised species can generate a pressurised mass flow of gases in the airspaces, which delivers a greater oxygen supply and allows even greater growth underground or in deep water (Brix *et al.* 1992). Depth zonation of emergent macrophytes in wetlands is largely related to the degree of airspace development and ability to generate pressurised flow (Fig. 28.5). The rhizomes of some species also show a remarkable ability to survive for weeks or even months in the complete absence of oxygen, a treatment that would normally be lethal for plants (Crawford 1993).

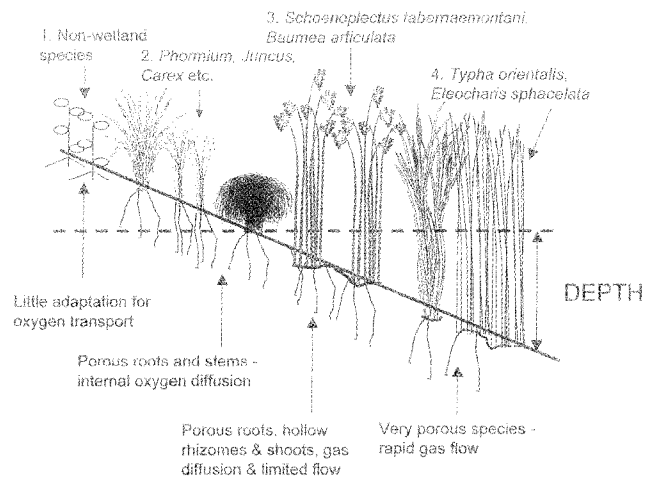


Figure 28.5 Typical zonation of plants along depth gradients in wetlands, with plants grouped according to oxygen transport strategies. After Brix *et al.* (1992) and Sorrell *et al.* (2000).

Other than hydrology and soil aeration, soil and water chemistry are the major factors affecting species composition. Dobson (1979) provided a graphical model to relate New Zealand herbaceous wetland vegetation to mean annual temperature and nutrient status. Thus communities in which *Typha*, *Phormium* and *Carex* are dominant are typical of relatively fertile (mesotrophic to eutrophic) wetlands. *Baumea* spp. are more commonly found in mesotrophic conditions, while the peat-forming species *Empodisma minus* and *Sphagnum* spp. are found at the oligotrophic end of the spectrum, as are cushion-forming species such as *Donatia novae-zelandiae* (but in colder temperatures). Community composition in peatlands (particularly bogs and fens) can also be controlled by the water chemistry (pH and cation base richness), reflected in correlations between the vegetation, pH, conductivity, calcium, magnesium and sodium content of the surface waters (McQueen and Wilson 2000). Autogenic processes (such as gradual elevation of the peat surface above the local water table by plant growth, acidification by *Sphagnum* spp.) also affect the successional pathway by reducing the amount of groundwater and thus the ionic supply into the soil water (Nicholson and Vitt 1994). The importance of base richness and nutrient (especially nitrogen) availability as determinants of community composition varies considerably in different parts of the world, depending on differences in groundwater sources and soil parent material (see Nicholson and Vitt 1994; Wheeler and Proctor 2000; Økland *et al.* 2001; Nakamura *et al.* 2002a). In New Zealand, changes in pH and nutrients (both N and P) over time appear to be important for both bog (Clarkson *et al.* 2004) and swamp communities.

Where nutrients are high, as in marshes and swamps, productivity is high. Bacteria are active in fixing nitrogen and litter is readily decomposed, with often high turnover rates (Mitsch and Gosselink 2000). Here organic matter accumulates because of high productivity, rather than because of low decomposition rates, as in peatlands (bogs and fens). The nutrient that most commonly determines the limits of growth is that which is lowest in supply relative to plant demand. In wetlands this is most likely to be nitrogen, for the reasons discussed earlier and because plants have much greater nitrogen than phosphorus requirements. Hence, limitation by phosphorus availability is most common in the more oligotrophic wetlands, i.e., in bogs, whereas swamp vegetation is more likely to be limited by nitrogen availability. Increasing the nutrient load into wetlands therefore causes not only an increase in primary production, but also a shift from limitation by phosphorus levels to limitation by nitrogen levels, and a shift from bog to swamp vegetation. Few plants can survive in the extremely dilute, phosphorus-deficient environment of bogs, which are often the preserve of very slow-growing species such as *Sphagnum* mosses and carnivorous plants such as sundews (*Drosera* spp.). New Zealand is unusual in having bogs that support relatively tall-growing vascular species such as *Empodisma minus*, which have highly specialised, fine-branching roots that can scavenge the few nutrients that come in via rainwater (Agnew *et al.* 1993), and that exude the enzyme **alkaline phosphatase** to break down organic matter and release PO_4^{3-} .

Understanding the nature of limitation by nutrient levels in wetlands is critical in protecting them from eutrophication, but until recently this could be determined only from laborious fertilisation experiments. Recently, comparisons of the nitrogen and phosphorus concentrations in the tissues of wetland vegetation have shown that the N:P ratio in the vegetation can be a useful indicator of nutrient limitation (Güsewell and Koerselman 2002). In plots of concentrations of tissue N versus tissue P (Fig. 28.6), sites where the vegetation has an N:P ratio < 13:1 are usually limited by the availability of nitrogen, and sites with N:P > 16:1 are usually limited by phosphorus levels. The ratio of the two nutrients tends to be more indicative than individual concentrations because increasing the supply of a limiting nutrient also increases uptake of non-limiting nutrients. Sites between 13:1 and 16:1 may be limited by available levels of either nutrient, and sites with very high concentrations are likely to have become saturated with nutrients, so growth is no longer nutrient-limited. Figure 28.6 shows that bogs are generally very oligotrophic and are limited by the availability of phosphorus, whereas fens and swamps are limited by nitrogen availability. Some fen sites are as equally nutrient-rich as some swamps, but the highest

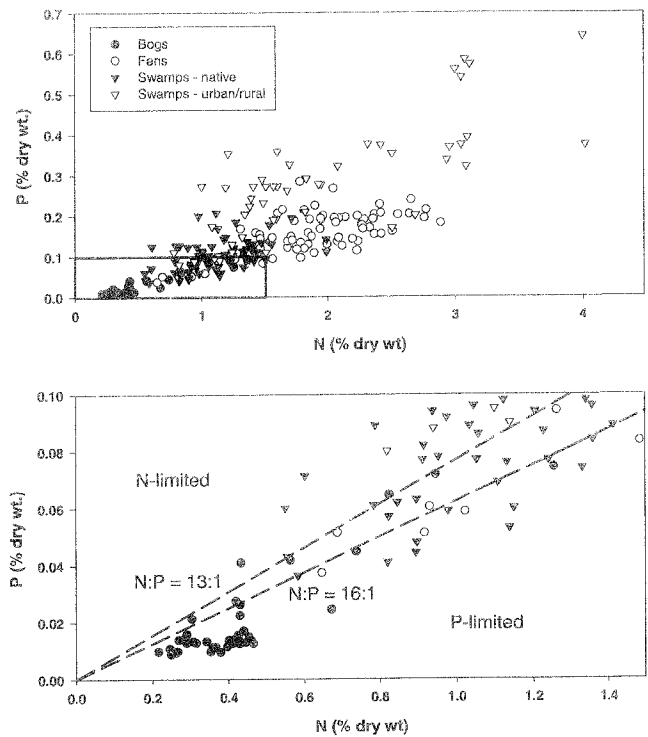


Figure 28.6 Upper graph: Tissue concentration of nitrogen vs. tissue concentration of phosphorus for dominant vegetation from New Zealand bogs, fens, and swamps. The area of the most oligotrophic plots (enclosed by box at lower left) is expanded for clarity in the lower graph. Here the sites are separated into categories of vegetation limited by available nitrogen and vegetation limited by available phosphorus by lines demarcating N-limited and P-limited sites where $\text{N:P} < 13:1$ or $> 16:1$, as defined by Güsewell and Koerselman (2002). Plots between these lines may be limited by the availability of either nutrient.

nutrient levels occur in catchments with rural or urban development.

The changes in net primary production by the vegetation, in the nature of nutrient limitation, and in species composition as nutrient input increases along the hydrological gradient from bogs to fens to swamps are summarised in Table 28.3. Fens are often only slightly more enriched in nutrients than bogs, but their greater groundwater input supplies enough phosphorus to support vegetation that is predominantly limited by the availability of nitrogen, although the community will include a mixture of species limited by the availability of both nitrogen and phosphorus. Vegetation in swamps is almost exclusively limited by nitrogen levels, until very high nutrient inputs lead to a dense biomass that results in productivity being limited by light availability.

Hence, the relationships between fertility, species composition and productivity in wetlands are not simple

Table 28.3 The nature of nutrient limitation, species composition and primary production for bog, fen and swamp communities. Primary production values from Mitsch and Gosselink (2000), species diversity values (mean \pm sd) are authors' unpublished data from a range of New Zealand wetlands.

	Bogs	Fens	Average Swamps	Eutrophic
Main limiting nutrient	P (or occasionally K)	N	N	No nutrient limitation
Component species	Mostly P-limited	N-limited species predominant, some P-limited species present	Mostly N-limited	Mostly light-limited
Species diversity (species/m ²)	0.7 \pm 0.2	1.4 \pm 0.3	1.3 \pm 0.4	1.0 \pm 0.1
Primary production (g dry wt/m ² /yr)	100	340	2,000	6,000

and require careful combinations of field and laboratory studies to be understood. Plant species differ in their strategies for using and conserving nutrients, which is why species composition changes with fertility. In the swamps where growth is limited by low nitrogen levels, survival depends on traits that minimise nitrogen loss from the plant. In nutrient-rich swamps, fast growth is essential for competition, so traits that maximise nitrogen capture and assimilation are more important than those that conserve it. Productive species that dominate nutrient-rich sites are therefore generally found to have a low nitrogen use efficiency (NUE):

$$\text{NUE} = \text{NPP}/\text{N}_{\text{LOSS}}$$

where NPP = net primary productivity, and N_{LOSS} = N loss from the plant in autumn (Nakamura *et al.* 2002b). Nitrogen loss occurs in dead biomass shed as litter from the plant and in soluble organic carbon lost from tissues. Typically, productive species with low nitrogen use efficiency also have a short mean residence time (MRT) of nitrogen in the tissue (Berendse and Aerts 1987), where:

$$\text{MRT} = \text{annual average [N] in plant}/\text{N}_{\text{LOSS}}$$

In contrast, mean residence time is long and nitrogen use efficiency is high in species from habitats with low fertility. These species also generally invest more of their biomass and nutrients in the below-ground parts of the plant that are involved in nutrient acquisition (Aerts *et al.* 1992).

Algae and bryophytes

In wetlands, both macrophytes and sediments offer abundant sites for colonization by bryophytes and algae.

Few data are available for these lower plants in New Zealand freshwater wetlands. Croosdale and Flint (1986, 1988) and Croosdale *et al.* (1994) have described the flora of New Zealand desmids, one of the larger and more important elements of the algae flora, especially in bogs and fens or other acidic wetlands. In a survey of 23 West Coast freshwater palustrine wetlands (including swamps, fens, bogs and pakihi), Gerbeaux (*unpublished data*) found that epipellic and epiphytic diatoms were also an important biodiversity component (over 90 species representing 35 genera) of the microflora. Cyanobacteria are often seen as epipellic in exposed sites subject to desiccation. Dominant algae groups in the epiphyton of more nutrient-rich wetlands include diatoms, filamentous green algae and cyanobacteria.

Algae are a fundamental element in wetland ecosystems. They are the most important primary producers at the bottom of food webs, and their importance to herbivores is clear (Goldsborough and Robinson 1996). As single or small clusters of cells they are easily assimilated. A summary of data on algal production in North American freshwater wetlands is presented by Goldsborough and Robinson (1996), with values ranging from 0.1 to 28 g C/m²/yr (for epipellic) and up to 548 g C/m²/yr (for epiphyton). Besides their contribution to the food web, benthic algae are high in dissolved organic matter, serving as short-term sinks for phosphorus and nitrogen and mediating the nutrient efflux rate from sediment interstitial water.

Many bryophytes, not just *Sphagnum* spp., can be common in peatlands (mainly pakihi, bogs, fens and sometimes swamps). Like the algae, they can be epipellic or epiphytic, growing in a diversity of microhabitats. Their

distribution and abundance on different peatlands within the same climatic zone are related to surface water chemistry along the ombrotrophic-minerotrophic gradient rather than to landforms (Gignac and Vitt 1994). Some species are typical of the more acidic conditions found in bogs and pakihi (e.g., *Campylopus introflexus*, *Dicranoloma* sp., *Trematodon mackayi*). Once the peat surface is no longer subjected to excessive water level fluctuations, bryophytes and shrubs can become established. Gradually hummocks are formed and here the process of acidification and oligotrophism begins. Here, in microtopographic features, bryophytes lower the pH through their cation exchange properties. Amalgamation of the hummocks and channelling of the water away from elevated areas of the mire ultimately leads to the formation of sphagnum bogs (Nicholson and Vitt 1994).

Microfaunal assemblages and invertebrates

Few published studies are available for algae and bryophyte communities in New Zealand wetlands. Ryan (2001) sampled representative areas of vegetation in the Whangamarino wetland (Waikato). She concluded that areas of native vegetation had different community assemblages of testate amoebae, monogont and digomont rotifers, copepods and cladocerans than disturbed areas of vegetation. The species richness was lower during the dry season than the wet season. Modern and fossil testate amoebae of ombrotrophic bogs have also been described by Charman (1997) and Wilmschurst *et al.* (2003) and used to reconstruct palaeohydrological environments.

Odonates are probably the best known of all wetland invertebrates. Comprehensive information on their ecology is provided by Rowe (1987). Among the large Lepidoptera Order, New Zealand's third largest insect Order after Coleoptera and Diptera (Patrick and Dugdale 2000), a reasonable diversity of moths inhabit wetlands and their associations go some way towards describing the diversity of wetland habitats. Patrick and Dugdale (2000) document 114 species or 6.8% of the New Zealand endemic Lepidoptera fauna that show a marked decrease in abundance and/or negative trend in overall distribution. Many of these are geometrids of the sub-family Larentinae. This is probably because of their specialised food-plant preferences, especially their preference for the herbs of damp areas associated with seepages and herb-rich wetlands. These types of wetlands have become uncommon since European settlement. Current surveys carried out by the Department of Conservation as part of pastoral lease reviews confirm the importance of wetlands for moths, with many taxa recorded from a variety of wetland habitats (from lowland to alpine wetlands, from bogs to swamps, from herbfields or sedgeland to shrublands). In addition, our moth fauna includes a

number of regional endemics, and this illustrates the unique identity of wetlands beyond their structural diversity outlined here (E. Edwards, pers.com.). Common moths include the Flax Looper moth (*Orithodydon praefectata*) and the Flax Notcher moth (*Tmetolophota steropastis*); the larvae of the latter attack the foliage of toetoe (*Cortaderia* spp.).

In a survey of Bullock Creek North Swamp (Paparoa National Park, Buller), Watts (2001) found that the number of invertebrate orders and individuals was comparable to that found in Waikato wetlands. Invertebrates from 16 orders were recorded, of which Amphipoda (hoppers) and Collembola (springtails) were the dominant orders caught. Araneae (spiders) and Coleoptera were also common. There was evidence of a difference between native vegetation and disturbed pasture at the site, and the highest diversity (beetles in particular) and abundance of invertebrates were recorded at the site of an old kahikatea stand. The invertebrate communities are highly dependent on detritus, and high densities are often attributed to the abundance of such detritus. The New Zealand freshwater crayfish (*Paraneohrops planifrons*) is well known from certain wetland locations, particularly West Coast pakihi (Grainger 2000).

Fish

McDowall (1975) was one of the first to point out the importance of wetlands for a wide variety of fish, and for the whitebait and eel fisheries. Yet very few wetland publications even today make reference to fish. This may be due in part to the difficulty of sampling fish in palustrine wetland habitats. There is also sometimes a belief that places like peatlands are of low biological productivity and therefore are not attractive to fish. McDowall (1975, 1978) reported on the species of native fish that are found habitually in various types of wetlands, including peatlands. Because many of the freshwater fish species are migratory, access to and from wetlands can be very important, with freshwater habitats close to the coast and at lower elevations commonly more significant than those further inland.

This is particularly true for the shortfin eel (*Anguilla australis*) and for inanga (*Galaxias maculatus*), the major species in the whitebait catch. These two species utilise permanent wetland habitat, but also wetlands periodically flooded and subject to water level fluctuations, a fact that eel fishers often exploit to capture shortfin eels. Shortfin eels can, however, penetrate long distances inland, and the lack of suitable habitat may be what limits their abundance inland. The reputation of the West Coast whitebait fishery is often attributed to the relatively higher percentage of wetland cover remaining in the region, compared to other regions of New Zealand (up to 30% compared to 10% or less elsewhere).

Of the galaxiid family, the Giant kokopu (*Galaxias argenteus*) is the other species associated with swamps. It is found in greater abundance around the margins of open shallow waters that are characteristic of New Zealand swamps, and where there is extensive marginal growth of flax, raupo and other tall semi-emergent vegetation. The Banded kokopu (*Galaxias fasciatus*) and certain species of bullies (particularly the Common bully *Gobomorphus cotidianus*) can also occur but they are less commonly reported from those habitats.

Wetlands are essential habitats for all species of mudfish (Department of Conservation 2003). Mudfish species belong to the genus *Neochanna* in the Galaxiidae family. They tend to prefer habitats that eliminate competition with other fish and predation by other fish. This includes waters that dry up (ephemeral wetlands). This habitat makes them a unique component of New Zealand's freshwater natural heritage and our most specialised group of freshwater fish. *Neochanna helios* (the Northland mudfish) is one of our rarest native freshwater fish. It is restricted to just a few small oligotrophic wetland sites on the Kerikeri volcanic plateau (Ling 2001). *Neochanna diversus* (black mudfish) is known widely from shallow pools of kauri gumlands and from northern peatlands, from Northland to Waikato, where the species is confined mainly to the large protected Whangamarino and Kopouatai Peat Dome sites, which are designated on the Ramsar List of Wetlands of International Importance. These two species appear nowadays to be largely confined to infertile bogs and fens on acidic soils that are dominated by *Baumea* or *Schoenus*, but they are likely to have also once lived in forested wetlands. *Neochanna apoda* (Brown mudfish) is the only species found in both the North and South islands. This species has a wider range of habitats and favours pools in mixed podocarp-broadleaf forest; it is also commonly found in infertile bogs and pakihis or spring-fed streams. *Neochanna burrowsius* (Canterbury mudfish) prefers the slow-flowing, overgrown swampy streams of the Canterbury Plains. Mudfish have recently been reported from the Chatham Islands (McDowall pers. comm.).

Several introduced fish species are also found in wetlands, especially in the North Island. Of those species, Koi carp and Gambusia have been declared unwanted organisms under the Biosecurity Act. Koi carp is also classified as a "Noxious Species" under the Freshwater Fisheries Regulations. To help stop their spread, a containment area for Koi carp has been set up around Auckland and Waikato—here recreational fishing is permitted, but all koi must be killed when caught. Other introduced species that may be found in wetlands are rudd, perch, catfish and goldfish. Wetlands, because of their structural complexity, can serve as refugia for indigenous

fish from predatory fishes. The ecotones between vegetated areas and open shallow water are particularly important in this respect.

CONCLUSIONS AND FUTURE DIRECTIONS

Historically, much of the research in New Zealand wetlands has involved classical vegetation descriptions, and surveys of the use of wetlands by large fauna such as fish and birds. Some progress has been made on relating plant species composition to the hydrochemical environment, usually within individual wetlands rather than amongst wetlands, and often considering only a few of the possible variables at a time. Although factors such as pH, water level fluctuation and nutrients are apparently important, much of the existing database is correlational or descriptive, and little experimental work has been done to test these observations. Previous work on nutrient relations has largely concentrated on measurements of nutrient cycling in relation to pollution control, with relatively little investigation of how nutrient cycling affects species composition and richness.

The development of better species-environment models for our wetlands would be greatly assisted by more whole-ecosystem, process-oriented studies, as seen in much of the European and North American research. Better understanding is required of how recruitment and the growth of plants are affected by subtleties of the hydrological regime, such as the frequency and timing of water level fluctuations, and important abiotic parameters such as E_H . Fertilisation experiments in the field would greatly improve our understanding of the nature of nutrient limitation, and the susceptibility of wetlands to eutrophication.

Our knowledge of the biology of the microflora and microfauna is poor, and needs redressing not only by descriptive studies, but also by further studies of their role in nutrient cycling and food webs. The importance of algae and bacteria in nutrient regeneration and in food webs have recently been emphasised in international research, and they are likely to play similar roles in New Zealand environments. There have been few comprehensive studies of food webs in New Zealand wetlands, and little application of modern techniques such as stable isotope methods that have successfully revealed trophic relationships in lakes. Further study of the complexities of how macrophytes structure habitat for invertebrates, fish and birds would also be valuable.

In addition to these gaps in our understanding of wetland functioning, there are also many information gaps that affect wetland management and restoration. These are discussed in Chapter 40.

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Chapter 27

Aquatic invaders and pest species in lakes

Gerard Closs, Tracie Dean, Paul Champion and Deborah Hofstra

INTRODUCTION

The movement of humans and their goods around the planet has greatly accelerated the rate at which species spread outside the boundaries of their natural range (Kolar and Lodge 2001). Species translocation has been recognised as a major threat to global biodiversity, and has the potential to wreak havoc on ecosystem integrity (Kolar and Lodge 2001). In the U.S.A. it has been estimated that introduced species cause major environmental damage and losses totalling around US\$137 billion per year (Pimental *et al.* 2000). In lakes, the introduction of an exotic species can on occasion have catastrophic consequences. For example, the introduction of Nile perch (*Lates niloticus*) to Lake Victoria, East Africa in the 1950s is estimated to have contributed to major declines in water quality and the extinction of over 200 native species of fish (Verschuren *et al.* 2002).

In New Zealand, the invasion of lakes and wetlands by alien or invasive species is recognised as a major conservation problem. Today, few New Zealand lakes support a biota that is wholly native. In excess of 200 aquatic plant and animal species have been introduced to New Zealand, and in some water bodies invasive species dominate the biota (Collier 1994). Aggressive invasive macrophytes such as hornwort (*Ceratophyllum demersum*) and fish such as European or koi carp (*Cyprinus carpio*) are widely recognised as significant pest species that can threaten the integrity of New Zealand lake ecosystems, and may cost millions of dollars to manage.

In this chapter, we examine the impact and control of invasive species, with a particular focus on lakes. Key processes and features of successful invasions and invasive species are presented, along with case studies that illustrate the ecology, invasion dynamics and management responses to three widespread species.

DEFINING INVASIVE AND PEST SPECIES

Invasion biology or ecology is the field of science that attempts to understand how and why species spread beyond their natural range. It also includes research that seeks to improve the management of invasions (Kolar and Lodge 2001). The management of invasive and pest species is complex and often involves various control options and the resolution of conflicting perspectives. The introduction of species beyond their natural range may be accidental or intentional, but only some have significant ecological and economic impacts. The perception of their impact is a value-based judgment that may vary from positive to negative. It is not surprising that the terminology used to describe species that have spread beyond their natural range is complex (Richardson *et al.* 2000). Using a consistent language to describe and categorise different types of alien, invading or pest organisms is crucial for effective communication, management and understanding of the topic. Hence we need to carefully define the terms “invasive” and “pest”.

What are invasive species?

Many terms have been used to describe an organism that has spread beyond its natural range—exotic, alien, adventive, imported, a novel coloniser, non-indigenous or non-native (Davis and Thompson 2000). Richardson *et al.* (2000) suggest the use of the term *alien* to define a taxon whose presence in an area is due to introduction as a result of human activity. Note, this definition does not suggest any relationship between invasion and its impact. Davis and Thompson (2000) consider that species with a negligible impact on the environment, or those that are indigenous to the region and rapidly expand into new disturbed environments should be termed *colonisers* rather than invaders. Such a definition recognises the fact that many species that have spread beyond the boundary of

their natural range have only a limited impact on ecosystem integrity or biodiversity, or that such a range expansion may occur without human intervention. Cronk and Fuller (1995) define an *invasive plant* as “an alien plant spreading naturally (without the direct assistance of people) in natural or semi natural habitats, to produce a significant change in terms of composition, structure or ecosystem processes”. Such a definition can be extended to describe any *invasive organism*.

What are pest species?

Various terms are used in legislation to describe organisms that might be considered as pests. The Biosecurity Act aims to prevent the introduction of unwanted organisms not already established in New Zealand, i.e., via border control, and to manage unwanted organisms established in New Zealand through the development of pest management strategies. In the Biosecurity Act (1993) pests are defined as “organisms specified as a pest in a Pest Management Strategy”. The criteria used to determine whether an organism is a pest include serious adverse and unintended effects on economic wellbeing; on the survival and distribution of indigenous plants and animals; on the sustainability of natural and developed ecosystems, on ecological processes and biological diversity; or on the relationship of Maori with their ancestral lands, waters, sites, waahi tapu and taonga. Organisms that are recognised as pests can be designated as “unwanted organisms”. An “unwanted organism” includes any organism that a chief technical officer believes is capable or potentially capable of causing harm to any natural and physical resources or human health. A total of 35 aquatic plants are designated “unwanted organisms”, preventing their sale and hence spread to new areas. This also allows for early eradication attempts to be carried out should they become established in a new region. Under the Biosecurity Act, mosquitofish (*Gambusia affinis*) and koi carp are the only fish species designated as “unwanted organisms”.

The term “noxious” also has a legal status in New Zealand, and was used to describe pest plants under the Noxious Plant Act (1978). Noxious fish are those listed in schedule three of the Freshwater Fisheries Regulations (1983) and include walking catfish (*Clarias batrachus*), koi carp, freshwater pikes (Esocidae), a number of piranha species, rudd (*Scardinius erythrophthalmus*) (except within the Auckland Fish and Game Region) and two species of *Tilapia*. The Freshwater Fisheries Regulations restrict the possession of these fish species.

Designation of an organism as a pest, an unwanted organism or as noxious represents an assessment indicating a negative economic or ecological impact from the perspective of society, a sector group or agency. In some instances a species may be perceived as both a pest and an

asset. For example, water net (*Hydrodictyon reticulatum*), an invasive algae, became prevalent in the Rotorua lakes during the mid-1990s. It caused aesthetic problems—forming unsightly floating mats and foul-smelling piles of rotting plants when washed up on lake shorelines—and it affected recreational use (Wells *et al.* 1999). To all those affected parties it was a pest. However, in Lake Aniwhenua water net provided habitat for large numbers of gastropods that were predated by rainbow trout (*Oncorhynchus mykiss*). A marked peak in sport fishing, with large numbers of trophy-sized fish taken, coincided with the water net bloom (Wells and Clayton 2001). In this case the anglers would regard the algae as an asset rather than a pest. Similarly, there is an overwhelming support for the maintenance of valued salmonid fisheries in many parts of New Zealand. However the introduction of salmonids to lakes where they do not occur is now questioned because of their effects on the integrity of native ecosystems (Rowe and Graynoth 2002).

Distinguishing between invasive and pest species

In summary, the terms invasive and pest refer to different attributes. Invasive species are defined as animals and plants that have spread to areas outside of their natural range, whereas pests are species that cause measurable economic or ecological impacts deemed unacceptable to society (or at least elements of it). It is important to note that pest species are not always alien and pest impacts may be very specific. Some valued alien species may be invasive, e.g., rainbow or brown trout, but they are generally not regarded as pests. However, all legally defined pests in New Zealand (animals and plants) are invasive alien species, and any beneficial attributes are outweighed by their detrimental impacts.

WHAT ARE THE IMPACTS OF INVASIVE ORGANISMS IN NEW ZEALAND?

Impacts of invasive aquatic plants

New Zealand lakes seem to be particularly vulnerable to invasion by introduced submerged aquatic plants, and few accessible lakes remain free of these species today. New Zealand lakes have conditions that are suitable for many species, so there are few physical constraints on invasion (Howard-Williams *et al.* 1987). Fifty-eight alien plant species are known to occur in New Zealand water bodies (see Table 27.1 for common species). The most problematic submerged aquatic weeds in New Zealand are the oxygen weeds (*Elodea canadensis*, *Egeria densa*, *Lagarosiphon major*) and hornwort (See Case Study 1). These species grow taller and faster than native species, and can form dense monospecific beds that may extend to the water surface. Most of these species do not reproduce

sexually in New Zealand, often because only one sex of the plant is present. The plants propagate vegetatively (small fragments break off stems and take root independently), so these species can spread rapidly within water bodies (Champion *et al.* 2002).

Invasive plants can have a range of impacts (Fig. 27.1) on native plant species. Introduced aquatic plants can either eliminate native species or severely limit their distribution by displacing them to deeper sites with limited light or to shallower, more exposed sites (Owen 1998). This displacement leads to a loss of habitat for native animals and a reduction in the extent of native seed banks, as native plants become isolated from their original sediments and seed sources (de Winton and Clayton 1996; Champion *et al.* 2002). Dense beds of introduced aquatic plants may also reduce wave action, modify erosion and sediment deposition processes, and change the substrate beneath plant beds. Where introduced aquatic plants form tall, dense, submerged or floating mats, these prevent light from penetrating the water column, cause a build up of low-density sediments with high organic content, and reduce oxygen concentrations within the water body and bottom sediment through respiration and increased litter production (Champion *et al.* 2002). Sediment anoxia may lead to the mobilisation and release of phosphorus into

the water column. In many shallow Waikato lakes, and also Lake Omapere in Northland, invasion of the weed *Egeria* has altered the entire lake ecosystem by causing sediment anoxia and the release of phosphorus, resulting in turbid planktonic algal-dominated water and the collapse of all submerged vegetation (Champion and Burns 2001).

The economic, recreational and aesthetic impacts of introduced plants can also be significant, and are often inter-related. Excessive growths of plants may clog intakes for pumps and power stations, and impede water flow and drainage. It has been estimated that the direct costs of aquatic plant pests due to weed control and lost industrial output (e.g., power generation) are in excess of \$10 M per year (J. Clayton, NIWA, personal communication). Piles of plant debris may also accumulate along shorelines where the material decomposes. The unpleasant odour may affect adjoining property owners and others that live on or use the shores of water bodies. Large surface-reaching weed beds or dense floating mats can interfere with fishing, swimming, water-skiing and boating, by impeding access to open water from boat ramps and jetties, and by clogging or entangling equipment. There are also instances of people becoming entangled in alien invasive weeds and drowning.

Despite their negative impacts, introduced submerged vegetation still provides many beneficial effects on the surrounding environment (see Chapter 24). The total loss of submerged vegetation can result in a shift to a turbid, algal-dominated system under the same nutrient regime. These alternative stable states are discussed by Scheffer (1990) and, in the case of Lake Omapere, by Champion and Burns (2001).

Impacts of invasive freshwater fish

Freshwater fish introductions to New Zealand have occurred since the 1800s and have resulted in the establishment of 21 exotic freshwater fish species (McDowall 2000). Some form highly valued fisheries, while others are considered to be pests (Champion *et al.* 2002). Nine introduced fish species can be considered widespread and well-established: these are shown in Table 27.1 with their distribution and legal pest status. Detailed risk assessments have been undertaken for a number of non-salmonid species introduced to New Zealand waters (e.g., Cadwallader 1977; McDowall 1984; Rowe and Schipper 1985; Hanchet 1990;). European carp has raised particular concern, due to its

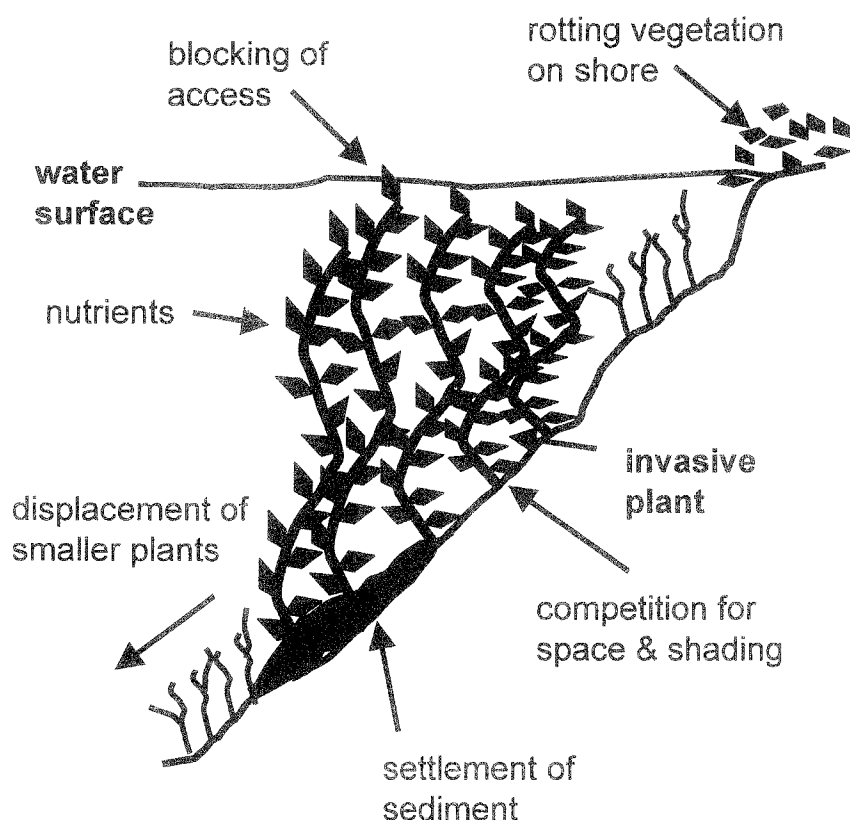


Figure 27.1 Invasive aquatic plants can have a variety of impacts on lake ecosystems.

Table 27.1 Common invasive fish and aquatic plant species in New Zealand

Common name	Scientific name	Distribution	Pest status
Fish			
Rainbow trout	<i>Oncorhynchus mykiss</i>	Most large NI and SI lakes	None
Brown trout	<i>Salmo trutta</i>	Most large NI and SI lakes	None
Goldfish	<i>Carassius auratus</i>	Widespread in NI, scattered in SI	None
Tench	<i>Tinca tinca</i>	Mainly northern NI, scattered elsewhere	None
Rudd	<i>Scardinius erythrophthalmus</i>	Mainly northern NI, very scattered elsewhere	Noxious
Koi carp	<i>Cyprinus carpio</i>	Mainly Waikato, very scattered elsewhere in NI, also Nelson, SI	Unwanted organism
Mosquitofish	<i>Gambusia affinis</i>	Mainly northern NI, scattered in rest of NI, also Nelson, SI	Unwanted organism
Perch	<i>Perca fluviatilis</i>	Small lakes throughout	None
Catfish	<i>Ameiurus nebulosus</i>	Mainly Waikato, very scattered in NI, also Lake Mahinapua, Westland, SI	None
Plants			
Ferny azolla	<i>Azolla pinnata</i>	Northern NI	None
Cape pondweed	<i>Aponogeton distachyus</i>	Scattered throughout	None
Egeria	<i>Egeria densa</i>	Common in NI, few sites in Marlborough and Canterbury, SI	Unwanted organism
Elodea	<i>Elodea canadensis</i>	Common throughout	Local pest
Lagarosiphon	<i>Lagarosiphon major</i>	Common in NI, scattered in most of SI	Unwanted organism
Swamp lily	<i>Ottelia ovalifolia</i>	Common in NI and northern SI	None
Curled pondweed	<i>Potamogeton crispus</i>	Common throughout	Local pest
Purple-backed duckweed	<i>Spirodela punctata</i>	Common in NI, scattered through much of SI	None
Bulbous rush	<i>Juncus bulbosus</i>	Common in peaty waters throughout	None
Yellow flag	<i>Iris pseudacorus</i>	Scattered throughout	Unwanted organism
Floating sweet grass	<i>Glyceria declinata</i> and <i>G. fluitans</i>	Locally common throughout	None
Reed sweet grass	<i>Glyceria maxima</i>	Scattered throughout	Local pest
Mercer grass	<i>Paspalum distichum</i>	Common in NI, local in northern SI	Local pest
Manchurian wild rice	<i>Zizania latifolia</i>	Common near Dargaville, rare in rest of NI	Unwanted organism
Water buttercup	<i>Ranunculus trichophyllus</i>	Common southwards from central NI	None
Waterlily	<i>Nymphaea alba</i>	Scattered throughout	None
Hornwort	<i>Ceratophyllum demersum</i>	Common in much of NI, Moutere, SI	Unwanted organism
Water cress	<i>Nasturtium microphyllum</i> and <i>N. officinale</i>	Common throughout	Local pest / food source
Alligator weed	<i>Alternanthera philoxeroides</i>	Common in northern NI, rare elsewhere	Unwanted organism
Parrot's feather	<i>Myriophyllum aquaticum</i>	Locally common in NI, rare in SI	Unwanted organism
Primrose willow	<i>Ludwigia peploides</i> subsp. <i>montevideensis</i>	Common in Waikato, scattered in Auckland, Manawatu, NI	Unwanted organism
Water purslane	<i>Ludwigia palustris</i>	Common throughout	None
Starwort	<i>Callitriche stagnalis</i>	Common throughout	None
Water forget-me-not	<i>Myosotis laxa</i> and <i>M. scorpioides</i>	Common throughout	None
Monkey musk	<i>Mimulus guttatus</i>	Common throughout	None
Water speedwell	<i>Veronica anagallis-aquatica</i>	Common throughout	None

current rate of spread and its potential to threaten ecosystem integrity (see Case Study 2). The recent designation of mosquitofish as an unwanted species indicates that it is also a problematic species.

Introduced fish also have a wide variety of ecological, economic, recreational and aesthetic effects. Their ecological impacts can include a reduced abundance of indigenous fish and invertebrate species through predation or competition, changes in the distribution of indigenous fish species, hybridisation with indigenous species, the introduction of parasites and diseases, reduced water quality, and habitat degradation and indirect effects on the other species that utilise these habitats (see Fig. 27.2) (Champion *et al.* 2002).

The introduction of exotic fish to New Zealand lakes

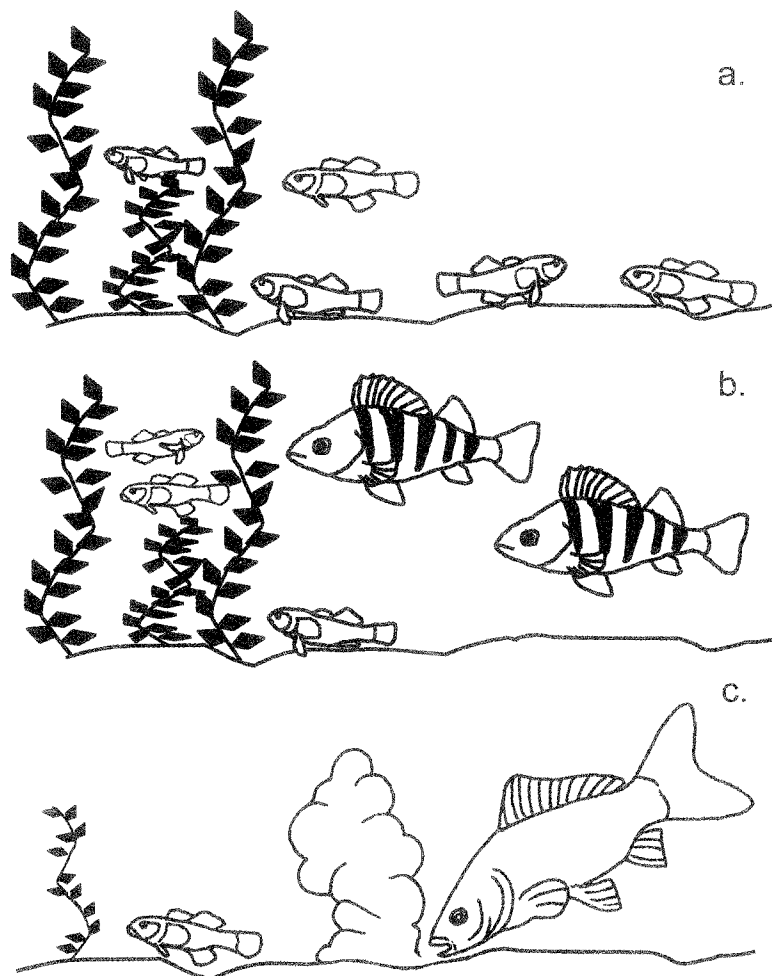


Figure 27.2 a) Common bully (*Gobiomorphus cotidianus*) dominate the littoral fish community in small coastal ponds in many coastal areas throughout New Zealand. b) The introduction of perch (*Perca fluviatilis*) reduces the overall abundance of common bullies and limits their distribution to protected areas. c) The feeding activities of European carp (*Cyprinus carpio*) increase water turbidity and disrupt beds of aquatic plants, further reducing habitats for native fish and invertebrates.

has had major consequences for many native fish species. Predation by rainbow trout caused significant declines in the abundance of native koaro (*Galaxias brevipinnis*) in several North Island lakes (McDowall 1990). Similarly, adult perch (*Perca fluviatilis*) are known to reduce the abundance of common bullies (*Gobiomorphus cotidianus*) in small ponds through a combination of direct predation and competition (Ludgate and Closs 2003). Fish introductions to lakes and wetlands have also been associated with changes in the distribution of native fish species. Mosquitofish attack the sub-adult lifestages of endemic black mudfish (*Neochanna diversus*) and dune lakes galaxias (*Galaxias gracilis*) (Barrier and Hicks 1994; Rowe 1998), which may exclude these species from habitats that mosquitofish utilise.

Introduced freshwater fish can also affect aquatic communities indirectly by altering habitats and reducing water quality. Adult rudd browse preferentially on soft-stemmed native aquatic plant species and can influence the composition of aquatic plant communities and prevent the regeneration of aquatic vegetation (Lake *et al.* 2002). European carp also affect submerged aquatic macrophytes in shallow lakes, as a result of their actions when feeding on organic material in sediments. The carp ingest sediment during feeding, filter food items from it, and expel the remaining material back into the water column. As a consequence, they increase turbidity, mobilise nutrients from sediments into the water column and uproot plants (Hanchet 1990).

The indirect economic costs of reduced water quality and aquatic habitat degradation resulting from invasive exotic fish can be significant. Work on European carp in Australia suggests that agriculture, horticulture and water supply industries are most likely to be affected by carp-mediated sediment disturbance and elevated water turbidity. European carp activity can cause erosion, destabilise channels and banks, and clog water pumping equipment (Koehn *et al.* 2000). Little information is available on the economic impacts of invasive exotic fish in New Zealand, however the costs of operations for control or eradication of exotic fish populations are clearly considerable (Barnes *et al.* 2003).

The economic, recreational and aesthetic impacts of introduced fish are often inter-related and highly variable, and can result in conflicts between different user groups. For example, different groups of anglers may wish

to catch different species of fish. The introduction of rudd and perch for coarse angling can result in a decline in the quality of trout angling if the rudd and perch form dense populations of small fish that will rise more readily to dry flies than trout (McDowall 1984). In Lake Parkinson, trout fishing was abandoned after a large population of stunted rudd became established. Fishing did not resume until rudd were eliminated (Rowe and Champion 1994).

The importation of new fish species to New Zealand also increases the potential of introducing associated parasites and pathogens that may pose a serious threat to native species and valued fisheries. While there are few reported examples of parasites and diseases being introduced to New Zealand through fish introductions, there is no doubt that any introduction carries risks of diseases or parasites. For example, the initial consignment of grass carp into New Zealand was certified as disease free, but was later found to harbour six species of parasite (McDowall 1990). This highlights the importance of border control and quarantine procedures to prevent the introduction of parasites and diseases.

Impacts of invasive freshwater invertebrates

The precise number of introduced invertebrates in New Zealand is unknown. However, several species of mollusc (including the common gastropods *Physa acuta* and *Limnaea stagnalis*), the uncommon medicinal leech (*Hirudo medicinalis*), the freshwater jellyfish (*Craspedacusta sowerbyi*) and at least one rotifer (*Conochilus exiguus*) (Boothroyd *et al.* 2002; Duggan *et al.* 2002) have been documented. Little is known of the range and impacts of introduced invertebrate species in New Zealand. However, a freshwater snail native to New Zealand, *Potamopyrgus antipodarum*, has invaded parts of Australia, Europe and North America and has been widely studied (Zaranko *et al.* 1997; see Case Study 3).

The introduction of two gastropod species to New Zealand has implications for human and animal health. The recently reported *Melanoides tuberculata* (Duggan 2002; Collier and Duggan 2002) is the first intermediate host for the parasitic trematode *Centrocestus formosanus*, which affects humans, herons, rats and dogs as definitive hosts (M. Hine, National Centre for Disease Investigation pers. comm.), and the more widespread *Pseudosuccinea collumela* acts as an intermediate host for liver fluke (*Fasciola hepatica*).

HOW DO ORGANISMS INVADE?

To manage and control invasive species, we need to understand how they colonise new habitats, and why some become invasive or pests. The process of colonisation and invasion can be viewed as a series of steps that must be

completed (Williamson 1996; Kolar and Lodge 2001; Fig. 27.3). Firstly, a species must be transported past the barriers that usually hold it within its natural range. It has to survive the process of transportation and introduction to a new environment. Successful reproduction is required to establish a viable population. Finally, it must be able to disperse from the founding population before it can be considered to be invasive. Identifying these discrete steps in the invasion process may give us clues about how and when we might manage and control invasive organisms. It also enables predictions to be made about potential invasive species and characteristics of species that contribute to successful colonisation and invasion (Kolar and Lodge 2001).

Transport and introduction

A range of factors influence the likelihood of an organism being transported outside of its natural range and being successfully introduced to a new habitat in which it can survive. It must be entrained or picked up by some sort of transport vector, e.g. on the feathers or feet of water birds, in ship ballast water, in inter-catchment water transfers, or as a deliberate introduction by people. For a freshwater organism, transfer overland or across oceans requires transport across inhospitable environments, during which it may be exposed to a variety of ecologically demanding conditions, e.g., removal from water for prolonged periods or extremes of temperature. The longer

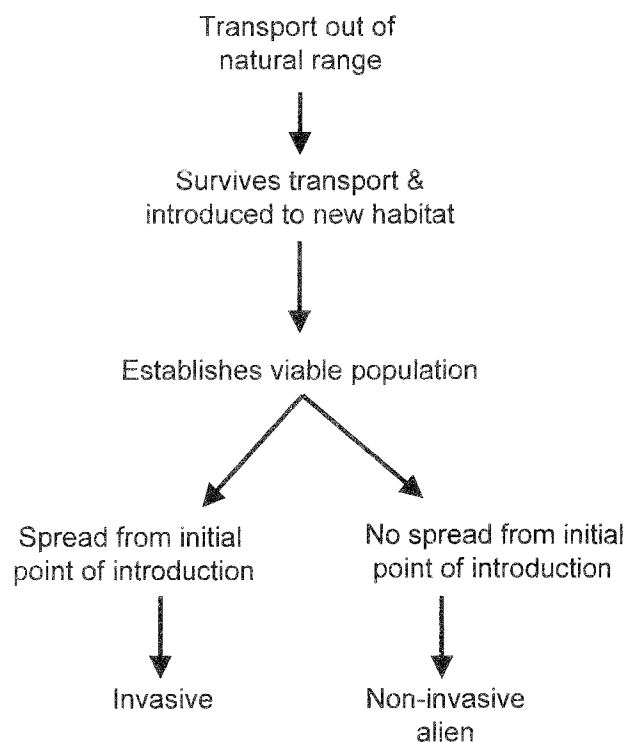


Figure 27.3 Stages to becoming an invasive organism.

such adverse conditions persist, the lower the probability of survival. Consequently, the likelihood of an organism being transferred tends to decline with increasing distance from a potential source population. Finally, the organism must be able to survive the physical, chemical and biotic conditions present in the new environment.

Given New Zealand's geographic isolation, it is not surprising that the majority of freshwater alien species present were introduced deliberately. Most arise from the aquarium and ornamental pond trade (Champion *et al.* 2002), but some were introduced for food, for medicinal purposes or as livestock forage (Champion and Clayton 2000); others have also been legally imported as game animals. The latter process was once encouraged through legislation to acclimatise non-native fish species to New Zealand (Thomson 1922). Unintentional imports have also occurred and include species introduced as contaminants on other water plants (e.g., water net) or perhaps through contaminated ballast water (e.g., dart goby *Parioglossus marginalis*) (Wells *et al.* 1997; McDowall 2000). A small number of fish have also been illegally imported; e.g., rudd and orfe (*Leuciscus idus*) (McDowall 1990).

Establishing a viable population

Generalist rather than specialist species are more likely to survive the wide range of conditions they encounter during transport and successfully become established in a new environment. Such species may be characterised by traits such as broad environmental tolerances and flexible life histories, and may be opportunistic generalist feeders and relatively fecund. To establish viable populations, recently introduced organisms must find a place within the community in which they now find themselves. Species-poor communities may be easier to invade, given that they may have vacant niches and relatively fewer predators and competitors compared with species-rich communities. There is some evidence to suggest that environments that naturally support fewer species, or have been recently disturbed and in which natural populations are depleted, are more likely to be invaded (Moyle and Light 1996). The introduction of a large number of individuals of a potential invader, or perhaps repeated introductions, can swamp local predators and competitors, thus increasing the likelihood of successful establishment (Kolar and Lodge 2001). If the organism reproduces sexually, the introduction of a large number of individuals also increases the chance that both males and females will be released, obviously essential if sexual reproduction is to occur. However, organisms that can reproduce asexually have a significant advantage, as each individual can potentially found a new population. Such generalisations with respect to the traits of organisms and communities

that will favour successful establishment suggest that we may be able to predict which species are likely to establish and invade should they be introduced.

Spreading from the initial point of introduction

The success of an invasive species can be measured by how readily it can spread from its site of introduction. Perhaps the best documented spread of an invasive species is the invasion of North American waters by the zebra mussel (*Dreissena polymorpha*). It was introduced to North America from Europe, and was first recorded in the Great Lakes in 1988. Within 10 years it had spread throughout the Great Lakes and connecting river systems, including the vast Mississippi Basin to its mouth at Louisiana (Kraft and Johnson 2000). On the basis of its initial rapid spread, it was predicted that it would invade all suitable North American waters by the year 2000. However, following the initial rapid invasion of waters connected to the Great Lakes, the rate of new zebra mussel infestation has slowed considerably, as further spread now requires the transfer between river or lake basins or catchments (Kraft and Johnson 2000). The spread of zebra mussel in North America illustrates an important principle in the spread of invasive freshwater organisms, i.e., the spread of an organism through connected water bodies can be rapid, but the rate of spread is usually slower when transfer from catchment to catchment is required.

MANAGEMENT OPTIONS AND STRATEGIES

The series of steps outlined previously in an invasion by an introduced organism provide a management framework for managing invasive species. Each invasion step presents different options for control and management, but as a species becomes more established, management options become progressively limited and usually more expensive to implement (Owen 1998).

Preventing invaders from invading

It is simpler and less expensive to prevent the initial importation and transfer of a potentially invasive organism than it is to control an invasive organism once it has become widespread and abundant. Preventing the importation of an organism into New Zealand represents a key control point for invasive species management. The likely points of entry and transport mechanisms can be readily identified, and detailed inspection and quarantine measures put in place. As most aquatic pests arise from deliberate introductions, the evaluation of the pest potential of candidates for importation is arguably as important as inspection for accidental introductions. Legislation provides for an efficient system of border

control and post-entry quarantine for legally imported organisms (with an obvious exception being the aquarium trade). However, a significant number of new aquatic organisms have entered the country illegally, therefore bypassing this evaluation process (Champion and Clayton 2000). Once the organisms are inside New Zealand, the declaration of unwanted organism status for aquatic animals and plants of high pest potential through the Biosecurity Act (1993) prevents their sale, propagation and distribution. This decreases the probability that they would be spread around the country and reduces the likelihood of invasion and establishment in new water bodies.

Assessment of the risk posed by potential invaders has been undertaken for weeds (Groves *et al.* 2001). The best predictor of potential invasiveness is the history of the species in other countries where it has been introduced. Attributes of the species' biology and ecology, and comparison of environmental variables within the native range and available habitat in the new country is also useful. It is important to recognise that there are many exceptions to such patterns, and that prediction of which species are likely to establish and then invade is still very imprecise (Lodge 1993; Moyle and Light 1996; Kolar and Lodge 2001). Many aquatic pest species in New Zealand have not been reported as pests in other countries (e.g., marshwort *Nymphoides geminata*) (Tanner and Clayton 1985). Risk assessment models are currently used to assess the importation of new plant species into New Zealand and have been used to justify the unwanted organism status of aquatic weeds. One model designed specifically for the assessment of aquatic weeds (Champion and Clayton 2000) was used to rank current and potential aquatic weeds in New Zealand. Rankings in this model agreed with the perceptions of management agencies. The risk posed by prospective introductions of fish has also been evaluated on several occasions. The perceived threat to the environment posed by largemouth bass (*Micropterus salmoides*) and channel catfish (*Ictalurus punctatus*) was great enough to prevent their import or release within New Zealand (McDowall 1968; Townsend and Winterbourn 1992). However, there has been limited assessment of aquarium fish species and McDowall (in prep) has shown that current importation requirements are so generic that some internationally recognised pest fish could be legally introduced.

Identification and management of potential vectors is a key step in preventing the spread of an invasive pest species. In New Zealand, many alien species, including some salmonid fish (e.g., Mackinaw – *Salvelinus namaycush*) and a large number of plant species, are more-or-less restricted to the water bodies where they were first liberated (Champion *et al.* 2002) owing to the absence of any effective dispersal mechanisms.

Where potential invaders exist, management actions should be undertaken to prevent further spread. Dispersal mechanisms range from deliberate introduction, to natural vectors like waterfowl (e.g., vectors for purple-backed duckweed *Spirodela punctata* and the snail *Physa acuta*). Accidental movement due to human activity has been implicated in the spread of many submerged aquatic plants (e.g., hornwort), invertebrates and even eggs and juveniles of fish (Johnstone *et al.* 1985; Champion *et al.* 2002). Common vectors related to human activity include watercraft, mechanical diggers and fishing nets.

Eradication of invasive and pest species

Once an invasive species has become established, the choice of management options will be determined by the potential impact of the species, its current and potential distribution, the effectiveness and feasibility of management (including its costs and benefits), and the values of the sites in which the pest is established (see Owen 1998 for example). Most pest control management objectives stem from a desire to reduce or prevent damage that the pest causes using a cost effective approach that avoids any additional effects on non-target species and communities. Potential management strategies include eradication; control and/or containment; or do nothing. The "do nothing" option is suitable where the level of impact is low, the species is unmanageable, or the risk of further spread is minimal.

Eradication is the permanent removal of an entire pest population, and is an appealing option for managers, as it removes ongoing costs and pest impacts. However, it is often expensive and is rarely achieved in practice. For invasive aquatic plants and fish in New Zealand lakes, eradication options include physical removal, habitat manipulation, chemical control and biological control (Champion *et al.* 2002). In some instances a combination of methods is required. Each approach has advantages and disadvantages that must be assessed carefully before use, and in relation to the benefits of eradication.

Physical removal has been used successfully on two emergent plants, water hyacinth (*Eichhornia crassipes*) and salvinia (*Salvinia molesta*), and may have potential for some submerged plant species in a limited range of environments where invasions are detected at an early stage. However, physical removal is rarely an effective method of eradication, due to the difficulties in removing all individuals. It is generally an option only for small, accessible populations restricted to a limited area, and where the target organism is easily identified.

Manipulation involves modifying a habitat, rendering it unsuitable for the survival and reproduction of the pest species. Eradication of invasive fish and aquatic plant species through drainage has been attempted with mixed

results, with its success being dependent upon complete drainage and on preventing the species from being transferred with the drainage water. Where drainage is not complete, chemical treatment of the remaining water may be necessary. Exotic fish may also be eradicated by removing access to spawning grounds. Aquatic plants may be eradicated by smothering them with material to exclude light. This approach was successfully used to eradicate marshwort from Lake Okareka.

Chemicals have been used to eradicate invasive fish and aquatic plants in a limited number of New Zealand sites. However, the range of habitats in which chemicals can be used is often limited, as few chemicals are specific to the particular pest species, so collateral damage may be high. Only a limited range of compounds is legally available for use in water in New Zealand. Herbicides have been used to eradicate populations of emergent and floating aquatic plants, but not submerged plants. The piscicide rotenone was used to remove grass carp (*Ctenopharyngodon idella*), rudd and tench from two-hectare Lake Parkinson in 1981 (Rowe and Champion 1994). Subsequently it was used in Nelson-Marlborough to remove European carp and mosquitofish from a number of small water bodies (Shaw and Studholme 2001). While chemicals may be effective for eradicating fish from small water bodies, they generally are not practical for larger water bodies, due to the expense and difficulty of achieving full coverage. Where chemicals are used to eradicate species from natural water bodies, it may be desirable to remove valued species prior to treatment for restocking afterwards.

Biological control has rarely been used to eradicate aquatic species in New Zealand. The most frequently cited success is the use of grass carp to eradicate the oxygen weed *Egeria* from Lake Parkinson (Rowe and Champion 1994). The successful use of grass carp to eradicate invasive aquatic weeds appears to depend on the stocking rate of grass carp, the suitability of the environment (e.g., temperature), and the ability to contain the fish within the water body.

Controlling the unwanted impacts of pest and invasive species

Where it is not feasible or desirable to eradicate pests, containment and/or control may be the only way to prevent their further spread, or to minimise their impact. Control and/or containment programs usually carry on-going costs, so it is important to carefully establish clear performance objectives within the management strategy for the pest against which the success of the project can be measured. In many sites, a single method of control is unlikely to be effective, and an integrated range of methods may be required to achieve the desired effect.

Containment is the prevention of the spread of a pest

to new locations. Containment programmes typically involve measures to prevent the transfer of pests from the site, and measures to detect any breaches of the containment area. Containment is difficult to achieve in practice for many water bodies, especially those that are used by the public or connected to other waterways. *Hydrilla verticillata*, an invasive aquatic plant, has been successfully restricted to several lakes in the Hawke's Bay area for several decades. The success of this strategy is attributed to a combination of factors—geographical isolation, public awareness, preventing the access of motor boats to the infested lakes, and controls on the sale and distribution of *Hydrilla*. Further containment measures are planned, including control of the plant at sites that are most accessible to the public.

The aim of pest control is to reduce the size of an invasive species' population, and it is undertaken when the impact of the invasive species compromises the use of the water body in some way. Control options include one-off actions, sustained control to restrict the target species to a threshold density, or sporadic control. The methods for control are similar to those used for eradication, but as removal of every individual is not the aim, a wider range of options may be available. For example, while physical removal of every fish from a lake is unlikely to succeed, it can be used to significantly reduce fish populations. In the Waikato Serpentine lakes, rudd threaten the native plant communities. Suppression of rudd densities by regular netting may be sufficient to reduce their impact (Barnes *et al.* 2003). Around boat ramps and other high-use recreational areas in Lake Dunstan, Central Otago, *Lagarosiphon major* is controlled locally using a combination of herbicide and mechanical removal.

Commercial or recreational fisheries may be established to reduce populations of pest fish. However, the benefits of establishing commercial or recreational fisheries for pest fish control must be carefully weighed against the risk of creating an incentive for people to maintain existing populations or establish new populations. Brown bullhead catfish and European carp continue to be commercially harvested in the Waikato Region. The harvest has generally been small-scale and sporadic due to limited markets and low profit margins. As a result, the quantities of fish harvested do not exert sufficient control on wild populations to significantly reduce their numbers.

Biological control may also offer options for reducing the populations of certain invasive organisms. The use of biological control agents to control invasive aquatic plants is well developed. Two species-specific insects, a beetle (*Agasicles hygrophila*) and a moth (*Vogtia malloi*), have been introduced to New Zealand to control alligator weed (*Alternanthera philoxeroides*), an invasive aquatic plant. These species have become established and are exerting a

moderate degree of control on the plant (Champion *et al.* 2002). Grass carp have proved to be effective in reducing *Hydrilla* to very low densities in one Hawkes Bay lake, but not in eradicating it, and the carp have been proposed as a long-term method of weed control for a number of water bodies. The success of grass carp for partial plant control will depend on the ability of managers to control the stocking density of the fish as they grow in size (Clayton and Wells 1999), and the temperature and water quality of the receiving water body.

CASE STUDIES OF INVASIVE ORGANISMS

The following case studies of three pest organisms illustrate some of the principles covered in the preceding sections, including mechanisms of spread, impacts and options for control. Hornwort (Case Study 1) and European carp (Case Study 2) are both considered as highly invasive pest species in New Zealand. In contrast, little is known of the impacts of the freshwater invertebrates in New Zealand, but experience gained from the spread of the freshwater snail *Potamopyrgus antipodarum* in Australia, Europe and North America is valuable (Case Study 3). A native to New Zealand, it is highly invasive but not necessarily a pest, although its status may change if it is found to have a negative impact on ecosystems valued by humans.

Case study 1: An invasive aquatic plant: hornwort (*Ceratophyllum demersum*)

Hornwort is a perennial, submerged, dicotyledonous plant. Globally, it is probably the most widely distributed aquatic plant, with a native range spanning the tropics to the sub-arctic, in all continents with the exception of Antarctica. However, it is not indigenous to New Zealand and was apparently introduced as an aquarium plant. It was first recorded outside of cultivation in Napier in 1961 and then in Lake Ohakuri, on the Waikato River in 1963 (Chapman 1970). In both cases, it probably became established after the contents of an aquarium or ornamental pond was dumped in the wild. Subsequently, it has spread to lakes throughout the North Island, from Lake Ngakeketa near Cape Reinga (Tanner *et al.* 1986) to Lake Wairarapa near Wellington at the southern end of the North Island (Hofstra and Champion 2002). In 2002 it was found in the South Island for the first time at a few sites near Motueka (Champion *et al.* 2002).

Hornwort reproduces by fragmentation in New Zealand (Coffey and Clayton 1988). Its toothed leaves, which are produced in whorls, can anchor in sediment and it usually grows as submerged beds attached in this way. Hornwort is tolerant of a wide range of environmental conditions,

growing in eutrophic or oligotrophic waters in full sunlight or shade, and it is not sensitive to cold, growing even under ice (Wells *et al.* 1997; Fair and Meeke 1982; Spencer and Wetzel 1993). High dissolved nutrients increase its growth rates (Best 1979), however it grows to greater depths in clear oligotrophic waters (e.g., 15.5 m in Lake Tarawera (Wells *et al.* 1997). To prevent its deliberate spread, the sale and distribution of hornwort and other submerged weeds was prohibited from 1982 onwards under the Noxious Plants Act (1978) and it is now an unwanted organism. However, hornwort readily fragments, and its distribution and abundance continue to increase as it is spread to new water bodies accidentally by contaminated watercraft, drainage machinery and fishing nets.

Hornwort displaces indigenous submerged vegetation and can form dense beds up to 7 m tall. In shallow lakes, beds reaching the surface can form over the entire lake. Hornwort seems to modify the sediment in which it anchors, reducing the ability of indigenous species to re-vegetate once the plant is removed and thereby impeding efforts to restore lake vegetation (NIWA unpublished data). Its economic impacts include reducing the efficiency of hydro-electric power generation (Chapman 1970) and blocking of drains, promoting flooding. It restricts recreational activities like fishing and boating and is a recurrent problem for water skiers and rowers in Lake Karapiro in the Waikato.

Hornwort is now so widespread and present in many relatively large water bodies that management options are limited with current methods. Clearly widespread eradication is not possible, so management is focused on containment and control at specific sites. Habitat manipulation, including lake drawdown, has been attempted in the past but this provided only temporary control, and had major effects on other aquatic biota (Hughes 1976; Johnstone 1982). Power generation authorities currently control hornwort by intercepting accumulations of this plant on screens located near the turbines (Johnstone 1982). Mechanical harvester or cutters have been used, but control is short-lived, and the movement of harvesters (which are extremely difficult to decontaminate) between lakes has been implicated in the further spread of hornwort. Hornwort has been biologically controlled in some drains using grass carp (Wells *et al.* 2002) and this method may be suitable for eradicating this plant in some lakes. Herbicides have recently been used for control in two dune lakes between Wanganui and Kapiti, using diquat (NIWA unpublished data), and in a drain near Lake Wairarapa using endothall (Hofstra and Champion 2002). Diquat is the only herbicide currently registered for use in water in New Zealand, however an application for registration of endothall has recently been sought.

Case study 2: An invasive freshwater fish – European carp (*Cyprinus carpio*)

European carp originated in eastern Asia but have now spread worldwide. They have been present in Europe for over 2000 years, and reached Canada, America and Australia in the 1800s. Although European carp are highly valued in some countries, they have the potential to lower water quality and damage aquatic habitats. They can spread rapidly within a catchment, they tolerate a wide range of conditions, and they have a high reproductive rate. They can also grow to a very large size (in excess of 10 kg) and have a long life span (McDowall 1990). Japanese koi carp (an ornamental strain of European carp) arrived in New Zealand in the 1960s.

In 1980 European carp were declared a noxious species in New Zealand, however in 1983 the first wild European carp in breeding condition was captured in the Waikato River. Illegal transfers and accidental releases resulted in European carp becoming established in the Waikato and Waipa Rivers in the Waikato Region, and in several streams between Auckland and Warkworth. European carp populations have continued to increase in the Auckland-Waikato region, and European carp are now abundant and a conspicuous component of the fauna of the lower Waikato River and its associated lakes and wetlands.

Eradication of European carp from the large and complex habitats of the lower Waikato River is considered impossible with available methods, so management agencies have adopted strategies to prevent further spread. In 1990 the Freshwater Fisheries Regulations were amended and a European carp containment area was established around the feral distribution of European carp. The main purpose of the containment area and the new regulations was to provide for commercial and recreational fishing of European carp within the containment area as a means of controlling the carp, and to formalise the intention of management agencies to restrict European carp populations to within the containment area. However little effort was made to restrict European carp to the containment area or to track its spread. Regional publicity and education initiatives were established to prevent the spread of European carp into specific areas (e.g., Rotorua lakes), but this was undertaken on an *ad hoc* basis. Between 1990 and 2000, European carp expanded their range, with isolated populations becoming established in Northland and the southern North Island (Dean 2003). In 2000 European carp (and mosquitofish) were recorded in the South Island for the first time. In recognition of the significance of these range extensions, and to gain access to management tools and legal powers, the Department of Conservation declared European carp and mosquitofish to be unwanted organisms under the Biosecurity Act 1993.

To assess the management responses required, the

Department of Conservation undertook a survey to establish how widespread the species was in the South Island (Studholme 2003). European carp were found at three sites in Nelson-Marlborough, and eradication by drainage and poisoning was attempted. A similar but less intensive survey program was undertaken in the North Island (Keys *et al.* 2003) to assess the range extension of European carp. In some areas this has resulted in identification of outlier populations, and areas outside of the containment area have been targeted for further assessment and possible eradication. Powdered rotenone was registered for use in New Zealand as a piscicide to provide management agencies with an effective method for control or eradication. In conjunction with the direct management of these European carp populations, a public awareness program was initiated nationally to educate New Zealanders about the risk that European carp and other invasive freshwater fish pose to natural environments, to limit their accidental or spread by ill-informed people.

Case study 3: New Zealand's own global invader – the freshwater snail

Potamopyrgus antipodarum

When we think of New Zealand and introduced species we typically think of the all-too-numerous pest species that have invaded New Zealand. However, a small freshwater snail native to New Zealand, *Potamopyrgus antipodarum*, has been colonising freshwater systems around the world. *Potamopyrgus antipodarum* was first recorded outside of New Zealand in the River Thames, England in 1889, but may have been introduced there as early as 1859 (Ponder 1988). Within 40 years it had spread around most of England and Wales (Zaranko *et al.* 1997), colonising continental Europe by about 1900, and it is now recorded from many parts of Europe (Zaranko *et al.* 1997). It was first recorded from Australia in 1870 and is now present in much of Tasmania and southeast Australia (Ponder 1988; Schreiber *et al.* 2003). It turned up in North America, with the first being recorded from the Snake River system, Idaho in 1987, and more recently in the Missouri River system and the Great Lakes (Zaranko *et al.* 1997).

At present, little is known of the ecological impact of *P. antipodarum* in areas where it has invaded. In southeast Australia, it is most often found at sites that have been affected by agriculture, forestry and urban development, and there is concern that it could out-compete closely related native snails for resources (Schreiber *et al.* 2003). In North America, *P. antipodarum* may achieve very high densities, potentially out-competing native snails for resources and clogging water pipes (Zaranko *et al.* 1997).

Why is *P. antipodarum* such an invasive species? To fully understand that, we need to consider the steps that lead to

successful colonisation and perhaps invasion—entrainment, introduction, survival and establishment (Kolar and Lodge 2001). The spread of *P. antipodarum* around the world has been attributed to the movement of water by ships, either as ballast or drinking water. *Potamopyrgus antipodarum* can survive periods of immersion in water with salinities close to that of seawater for several days (Zaranko *et al.* 1997). It can thus easily survive salinities encountered within a ship's drinking water or ballast tanks, and in the estuarine water into which it is likely to be dumped when tanks are cleaned or ballast water is discharged. The snail is also resistant to short periods of desiccation, hence it may be transported live to new locations in fishing nets, or in macrophytes on boat trailers, or perhaps in mud on the feet of birds.

Once introduced to a new location, a species must locate a habitat in which it can survive and then reproduce. *P. antipodarum* is a habitat generalist—in New Zealand it is one of the most widely distributed freshwater invertebrate species. Hence it is no surprise that it is likely to survive in many of the new environments into which it has been introduced (Ponder 1988; Zaranko *et al.* 1997; Schreiber *et al.* 2003). When it comes to reproducing, *Potamopyrgus antipodarum* is also parthogenetic, i.e., female snails can reproduce asexually. Thus only a single female snail is required to start a new population, a huge advantage for an invasive species.

Finally, the example of *P. antipodarum* illustrates another problem in tracking the spread of invasive invertebrate species. Invertebrates are small and often difficult to identify. *P. antipodarum* has been recognised as a global invader only recently. Although *P. antipodarum* may have invaded Australia as early as the 1870s, Ponder (1988) was the first to recognise that it had originally colonised Australia from New Zealand. Similarly in Europe, it was considered to be a European native by some authors (see Ponder 1988). Therefore, because they may be relatively small and inconspicuous, invertebrate invaders can colonise and become established long before anyone notices, and once established they can be almost impossible to eradicate.

CONCLUSIONS

In this chapter, we have discussed invasive and pest species, identified the stages of an invasion, and how we might use these stages to manage and control invasive species. Finally, we have presented examples of two species that are recognised pest species in New Zealand and one that originated here and is now an invasive species elsewhere. Each of the species presented in the case studies represents an invasive or pest species that is now relatively widespread or abundant in the areas into which they have been introduced. In each case eradication is impossible

with current methods, hence managers are faced with on-going costs for containment and control. Such costs can be considerable, and resources must be re-prioritised as new invasions occur.

New Zealand has the great advantage of being geographically isolated from other countries by the sea, a significant barrier to potential freshwater invaders. However, that is no reason to be complacent. The most cost-effective approach to minimising the impact of invasive species on New Zealand freshwater ecosystems is preventing new invasive organisms from entering New Zealand, restricting the distribution of pest species already present here, and controlling or eradicating the invaders where practical, before populations become well established. However, there are few methods available to control or eradicate many of the organisms that have already established extensive and abundant populations. For some, we must tolerate their presence, and restrict their impacts where control is desirable or practicable.

Future efforts must be directed at the protection of the water bodies that are not yet significantly affected by pest species. We need to identify these sites, determine potential sources of pests, manage vectors for their spread (in nearly all cases human) and increase surveillance to ensure the rapid detection and management of new incursions.

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Chapter 26

Bird communities of lakes and wetlands

Murray Williams

INTRODUCTION

A remarkably small proportion of the world's birds inhabit aquatic environments. Birds adapted primarily to aquatic habitats are largely confined to just two of the 22 orders of birds: Anseriformes (the waterfowl) and Ciconiiformes (waders, gulls, petrels, herons etc.) (Sibley and Ahlquist 1991). The Anseriformes include four families and represent a radiation within a single environment—freshwater; the Ciconiiformes embraces 29 families that have come to typify marine and estuarine, as well as freshwater, environments and, in addition, some grassland and short-turf habitats. Present-day species within the other 19 orders and 65 families of non-passerine birds are exclusively inhabitants of dry land (Sibley and Ahlquist 1991).

Within New Zealand, the family Anatidae, the largest within the Anseriformes, is presently represented by 10 native and introduced breeding species on the main islands and seven species as regular vagrants (Turbott 1990). Within the Ciconiiformes there are representatives of 12 families breeding in New Zealand: those primarily associated with freshwater environments include grebes (Podicipedidae, 3 species), spoonbill (Threskiornithidae, 1 species), herons (Ardeidae, 5 species), shags (Phalacrocoracidae, 3 freshwater species), gulls and terns (Laridae, 5 species) and waders (Charadriidae, 6 freshwater species).

For these birds, lakes and their palustrine margins are habitat islands in an expansive terrestrial sea. Scattered unevenly throughout the landscape, these avian habitats differ markedly in size, and the variability of their water levels confers an unpredictability of resources. Changes in their natural character may progress rapidly to obliterate living opportunity, so wetlands provide fickle avian habitats.

To exploit lakes and other impounded wetlands as places in which to feed, breed or linger, birds have had to evolve

structural adaptations and life history strategies that allow them to respond to the diverse and unpredictable characteristics of wetlands. The structural adaptations are obvious—webbed feet for locomotion on the water; long legs for wading the shallow margins; long bills for probing the substrate, bills with sharp tips or edges for shearing vegetation. However, these adaptations are not unique to lake inhabitants; they are adaptations to aquatic habitats in general and are shown in great diversity by birds frequenting estuarine and marine environments.

One outstanding life history adaptation of wetland birds is their mobility. These birds must move from place to place as seasonal resources diminish or as variations in water levels affects resource availability. Many wetland inhabitants occupy habitat networks rather than single sites: the networks may be national in extent (e.g., Australasian shoveler *Anas rhynchos*) (Fig. 26.1), regional (e.g., bittern *Botaurus poiciloptilus*), or local (e.g., black shag *Phalacrocorax carbo*). Many wetland species are habitat pioneers—for example, rails and ducks find new wetlands uncannily soon after they are created or filled, and both groups are perennial colonists of scattered islands and archipelagos (Ripley 1977; Weller 1980; Roshier *et al.* 2001, 2002). Few true water birds reside year-round at a single site, a distinct contrast with species characteristic of palustrine wetlands or the palustrine margins of lakes.

Freshwater birds may show more flexibility in the timing and extent of their breeding periods than do, for example, marine or forest birds. While all are distinctively spring breeders, most have, from time to time, departed significantly from that norm. For example, any sudden flush in food availability may be converted into a breeding opportunity. This strategy is particularly obvious amongst wetland birds of Australia's arid zones (Braithwaite 1976a,b; Kingsford and Norman 2002), but even in the climatically benign New Zealand environment, some of

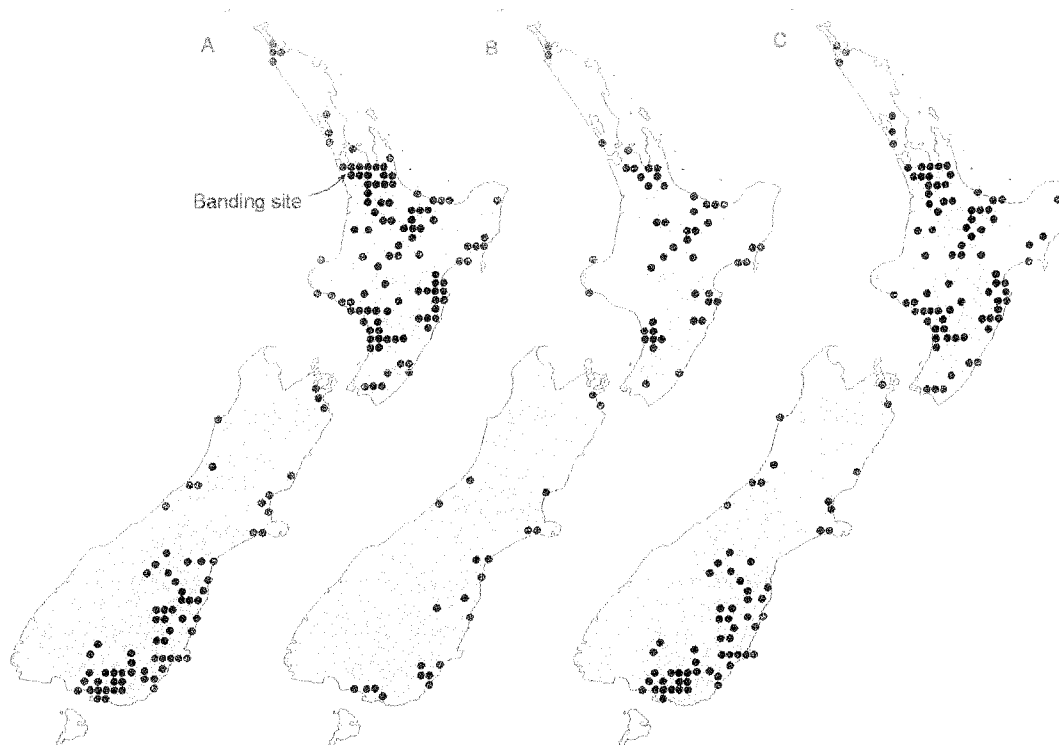


Figure 26.1 Locations at which adult Australasian shoveler, banded at Lake Whangape, Waikato 1978-85, were recovered by duck hunters; (A) all recoveries, (B) recoveries made within 6 months of banding, and (C) recoveries made more than one year after banding (from Caithness *et al.* 2002).

the species common to both Australia and New Zealand demonstrate a behavioural flexibility. For example, black swan (*Cygnus atratus*) breed with strict seasonality and timing on stable wetlands, but their breeding cycle depends on water levels on the large and shallow lakes (e.g., Ellesmere, Whangape) where food suddenly appears in super-abundance (Miers and Williams 1969; Williams 1980). Grey teal (*Anas gracilis*), a species widespread in the Australian arid zone, shows similar variability of response by breeding as water levels rise in the Waikato wetlands in June-July (McFadden 1983), but 2–3 months later during the spring flush in the wetlands of the southern South Island.

Freshwater birds may also demonstrate flexible breeding patterns. Overt defence of food resources, by the establishment of exclusive territories, is not common among truly lacustrine species. For example, freshwater shags (*Phalacrocorax* spp.), black swan, grey teal and scaup (*Aythya novaeseelandiae*) breed in close proximity or colonially, as well as synchronously, on lakes with fluctuating water levels, whereas the waterfowl may show greater nesting dispersion and less synchronicity on wetlands with stable or slowly changing levels (Williams 1980; Marchant and Higgins 1990; Stokes 1991). Pukeko (*Porphyrio p. melanotus*) residing in the palustrine margins of large lakes may have a very labile social system and prolonged breeding seasons compared to the smaller group

structure and more seasonal breeding of those inhabiting smaller wetlands that dry significantly in summer (Marchant and Higgins 1993 and references therein). Whether there is similar variability in the breeding season or breeding patterns of other rails in palustrine environments, or of other open water inhabitants (e.g., dabchick *Poliiocephalus rufopectus*) has yet to be determined. Also unknown is whether there are differences in fecundity and survival among species populations inhabiting contrasting wetland environments. Habitat-induced variability in mating, as demonstrated by pukeko (Marchant and Higgins 1993) and apparent in remnant populations of brown teal (*Anas chlorotis*) (M. Williams, unpub. data), is a largely unresearched topic that has the potential to uncover outstanding examples of life history adaptations.

BIRD COMMUNITIES IN DIFFERENT HABITATS

New Zealand's wetland avian communities are remarkably diverse relative to those of forests. Distance from a continental source—principally Australia—is not too inhibiting for naturally dispersive species that regularly fly over water. However, the diversity of species based on contemporary observations must be judged in light of the traumatic faunal history of New Zealand following the

arrival of rats and humans in the last two millennia. The pre-human avifauna was diverse (Worthy and Holdaway 2002), but species inhabiting wetlands were amongst the greatest casualties of the human onslaught. For example, eight of the 15 waterfowl present in pre-human New Zealand and the Chatham Islands were exterminated, and of the 5 with recent Australian affinities, only one (the black swan) has re-colonised. Nevertheless, New Zealand's wetland avifauna is being constantly augmented: pukeko and white-faced heron (*Ardea novae-hollandiae*) are post-human colonists, and new breeding species during the past century have included Australian coot (*Fulica atra australis*), Australian little grebe (*Tachybaptus novae-hollandiae*), hoary-headed grebe (*Poliiocephalus poliocephalus*), royal spoonbill (*Platalea regia*), nankeen night heron (*Nycticorax caledonicus*), black-fronted dotterel (*Charadrius melanops*), spur-winged plover (*Vanellus miles*) and welcome swallow (*Hirundo tahitica*). Other globally ubiquitous species, e.g., cattle egret (*Bubulcus ibis*) and glossy ibis (*Plegadis falcinellus*), which, like the plover and swallow, exploit a wider range of habitats than just wetlands alone and which are now common vagrants from Australia (Turbott 1990) can be expected to establish a permanent presence eventually.

The composition of an avian community on a lake logically could be expected to reflect the lake's trophic status, the diversity of exploitable resources present, and the present geographic distribution patterns of various species. However, distribution data for various lakes of diverse geographic location, age and origins indicate a high level of community similarity at all sites (Table 26.1). Most New Zealand wetland birds are widely distributed and there are few examples of community species diversity being directly influenced by limnological character, other than the exclusion of macrophyte grazers during phytoplankton-dominated phases (Mitchell *et al.* 1988). Any differences are mostly of relative abundance. Any apparent absences of a species may be more easily explained by historic factors (e.g., contemporary absence of dabchick from South Island), the widespread destruction of the palustrine ecotone (e.g., bittern), or their recent arrival or spread within New Zealand (e.g., Australasian coot, Canada goose *Branta canadensis* in North Island, Australian little grebe). This implies that niche specialisation is not strong within existing freshwater bird communities and opportunities for further colonisation by specialists remain.

Moore *et al.* (1984: Table 5a) provide a detailed list of 84 species recorded at Lake Wairarapa and in its surrounding wetlands—30 (36%) of these are birds whose primary or exclusive habitat is ecotonal vegetation at the wetland periphery. This serves to emphasise that community composition and species diversity at a lake can be significantly influenced by the state and extent of marginal vegetation (Fig. 26.2a,b).



Figure 26.2 Examples of lakes where avian community composition is influenced by marginal vegetation and water levels: a. Pukepuke Lagoon, Manawatu with its extensive ecotone supporting abundant marsh and spotless crake, bittern and pukeko populations; b. Lake Tutira, Hawkes Bay, where the absence of characteristic marginal vegetation limits nesting opportunities for waterfowl and excludes palustrine inhabitants

Photos: Department of Conservation

Table 26.1 Composition of avian communities at New Zealand lakes of differing origin (Lowe and Green 1987) and geographic location. Species are grouped according to their predominant feeding zone. Data in table indicates likelihood of encounter rather than relative abundance and is based on unpublished site records of the Ornithological Society of New Zealand bird mapping scheme (as of 1 August 2003). Key: a = abundant (>75%), c = common (50-75%), p = present (25-50%), r = rare (10-25%), reflecting the proportion of the records for the various lakes containing sightings of each species. The number of records for each lake varies from 5-22.

Species	Scientific names	Glacial Lakes		River Lakes		Dune Lakes			Volcanic Lakes	
		Canterbury	West Coast	Waikato	Otago	Manawatu	Kaipara	Far North	Northland	Rotorua
benthic feeders										
scaup	<i>Aythya novaeseelandiae</i>	a	c		c		c	p	p	a
crested grebe	<i>Podiceps cristatus</i>	p	p							
dabchick	<i>Poliiocephalus rufopectus</i>			a		a	c	c	p	c
Australasian little grebe	<i>Tachybaptus novaehollandiae</i>						p	r	r	
Australasian coot	<i>Fulica atra</i>	c		p	r					p
black swan	<i>Cygnus atratus</i>	c	c	c	a	c	c	p	p	a
water column feeders										
black shag	<i>Phalacrocorax carbo</i>	c	a	c	a	c	c	c	p	c
little black shag	<i>Phalacrocorax sulcirostris</i>			c		c	c	c	p	a
little shag	<i>Phalacrocorax melanoleucos</i>	c	c	c	a	c	c	p	p	a
pieb shag	<i>Phalacrocorax varius</i>			p				r	r	
black-fronted tern	<i>Sterna albobriata</i>	c								
Caspian tern	<i>Sterna caspia</i>			p			r	r	r	
Australasian shoveler	<i>Anas rhynchos</i>	c	r	c	c	c	c	p	p	c
aerial hunters										
Australasian harrier	<i>Circus approximans</i>	a	p	c	a	a	a	a	p	c
welcome swallow	<i>Hirundo tahitica</i>	p	c	c	a	a	c	c	p	c
black-backed gull	<i>Larus dominicanus</i>	c	c	p	c	c	c	p	p	c
shallows omnivores										
grey duck	<i>Anas superciliosa</i>	p	a	p	p	p	c	p	c	c
Mallard	<i>Anas platyrhynchos</i>	a	c	c	a	a	a	c	a	a
grey teal	<i>Anas gracilis</i>	c	r	c	c	c	p			p
paradise shelduck	<i>Tadorna variegata</i>	a	c	c	a	a	c	c	c	c
Canada goose	<i>Branta canadensis</i>	a	c	c	c	a		p		p
white-faced heron	<i>Ardea novaehollandiae</i>	c	r	c	c	c	c	c	p	p
white heron	<i>Egretta alba</i>		p		r		r			
kingfisher	<i>Halcyon sancta</i>	r	r	p	p	r	r	r	r	p
edge probers and gleaners										
red-billed gull	<i>Larus novaehollandiae</i>				p	r	p			r
black-billed gull	<i>Larus bulleri</i>	p			p					p
spur-winged plover	<i>Vanellus miles</i>	p	a	c	c	c	p	p	p	p
pieb stilt	<i>Himantopus himantopus</i>	p	c	c	p	c	c	p	p	r
South Island pieb										
oystercatcher	<i>Haemantopus ostralegus</i>	c								
banded dotterel	<i>Charadrius bicinctus</i>	p								
palustrine margin inhabitants										
fernbird	<i>Bowdleria punctata</i>		p	p	c	r	r	r	r	p
pukeko	<i>Porphyrio p. melanotus</i>		c	c	a	a	c	p	p	p
spotless crake	<i>Porzana tabuensis</i>			p		c	p	p	p	r
marsh crake	<i>Porzana pusilla</i>			r		r				
banded rail	<i>Rallus philippensis</i>						p			
bittern	<i>Botaurus poiciloptilus</i>			c	r	p	p	p		

Lakes included in analysis were:

Glacial lakes – *Canterbury*: Heron, Sarah, Grassmere, Pearson, Hawdon, Lyndon, Marymere, Emma, Clearwater; *West Coast*: Mapourika, Ianthe, Wahapo, Rotokino.

Riverine lakes – *Waikato*: Ngaroto, Waikare, Rotokawau, Rotongaro, Rotomanuka, Maratoto, Rotokauri; *Otago*: Waiholo, Waipori, Tuakitoto.

Dune lakes – *Manawatu*: Ouanuka, Pukepuke, Koputara, Kaikokopu, Foxton 1, Foxton 2; *Kaipara*: Kereta, Karaka, Poua, Ngakaru, Piripoua, Rotopaua, Humuhumu, Waingatu, Rotokawau, Kanono; *Far North*: Shag, Waikere, Taharoa, Kai-iwi, Swan, Half, Waihopo, Ngakapau, Ngatu

Volcanic lakes – *Northland*: Omapere, Owhareiti; *Rotorua*: Rotomahana, Rerewhakaaitu, Rotoma, Rotoehu.

O'Donnell (1985: Table 3) demonstrates that Lake Ellesmere is an even more spectacular avian habitat. A total of 158 species (55% of New Zealand's avifauna) had been, by that time, recorded within its environs, 116 of which exploit the lake proper or its marginal vegetation and, of these, 80 are regular inhabitants. This is a greater diversity than has been recorded on any other New Zealand wetland, and much of this is a consequence of the extensive salt marshes and flats that are a feature of the lake's western margin. It also demonstrates the importance of fluctuating water levels.

SEASONAL CHANGES IN COMMUNITY COMPOSITION

Community composition is rarely static, instead it reflects the responses of differing species to the seasonal availability of exploitable resources. And the abundance of any particular species may follow some well-established variability in its annual cycle. For example, Moore *et al.* (1984) recorded spectacular autumnal increases in waterfowl at Lake Wairarapa, coinciding with seasonal drying of nearby small or ephemeral wetlands, the post-breeding peak in waterfowl numbers, and the use of the lake as a moulting site. Similarly, an autumnal peak in wading bird numbers and species diversity reflected the seasonal use of the lake by northern hemisphere migrants and the enforced concentration of some native waders, e.g., pied stilts, as other shallow wetlands dried up during a phase of post-breeding dispersal. Caithness and Pengelly (1973) reported spectacular seasonal variation in Australasian shoveler numbers at Pukepuke Lagoon, in part a reflection of the birds' mating behaviour, in which large communal courting parties gather at selected wetlands before individual pairs disperse to more remote breeding territories. Hughey *et al.* (1986) recorded this too at Lake Tuakitoto (Fig. 26.3) as well as distinctive seasonal patterns in the presence of other waterfowl and wetland species, which they considered to be a response to varied habitat availability. Other wetland species, e.g., kingfisher (*Halcyon sancta*) and pukeko, may also show seasonal variations in abundance or presence, reflecting their use of differing habitats at differing times of year. Kingfishers are less abundant around lakes during spring and summer, as they retreat to more wooded habitat for breeding. Pukeko tend to be most conspicuous around permanent wetlands during late summer and autumn, using this habitat as a retreat when the more ephemeral wetlands used as spring breeding habitat become dry (Marchant and Higgins 1993).

Many coastal lakes, particularly the large barrier-bar lakes (Low and Green 1987) such as Ellesmere, Forsyth, Grassmere and Onoke, attract northern hemisphere

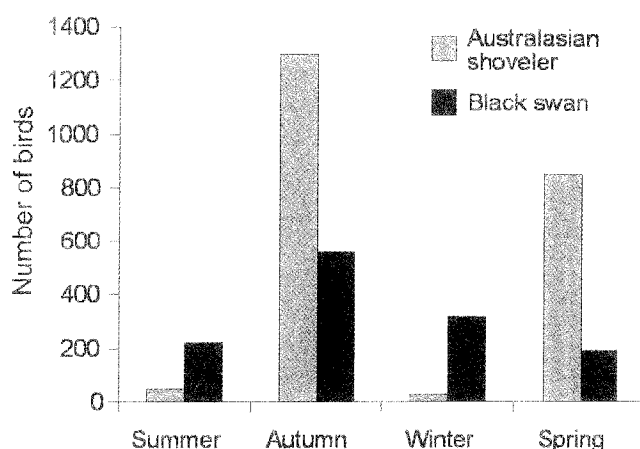


Figure 26.3 Contrasting seasonal presence and abundance of Australasian shoveler and black swan at Lake Tuakitoto, Otago (redrawn from Hughey *et al.* 1986)

wading birds (especially sandpipers and stilts, *Calidris* spp.) during summer as water levels decline and broad flats are exposed (Moore *et al.* 1984; O'Donnell 1985). These seasonal and primarily estuarine species add considerable diversity to the avian communities—Moore *et al.* (1984) recorded 12 species of northern waders at Lake Wairarapa and O'Donnell (1985) 30 species at Lake Ellesmere.

PARTITIONING OF LAKE RESOURCES

The avian communities of lakes or marshlands provide those classic illustrations of habitat zonation so beloved by ecology texts. Ogle and Cheyne (1981) used this illustrative approach (Fig. 26.4) to emphasise the effects of drainage on various biota in the Whangamarino wetlands. Williams (1982) described “zones of dependence” for lake-dwelling waterfowl and Sagar (1982) did likewise for other wetland birds. A more quantitative approach was demonstrated by Moore *et al.* (1984), who used frequency of observation of a range of lake-dwelling birds in various vegetation and landscape zones about Lake Wairarapa to illustrate relative habitat use (Fig. 26.5). This technique, developed by Robertson *et al.* (1983) when surveying wetland bird habitats in the Ahuriri River catchment, was used in other Wildlife Service studies (e.g., Hughey *et al.* 1986) as a crude way of demonstrating niche separation. It also has potential for the statistical analysis of differences between species, although none of the studies employing this technique undertook such analyses.

Remarkably few other New Zealand studies have attempted to quantify avian use of lakes and other wetlands, and in the absence of rigorous statistical analyses in the few studies carried out, there remains considerable uncertainty about how resources are partitioned amongst various species. Even food habit studies of wetland birds in New Zealand are few. Potts (1972, 1977a) provides a

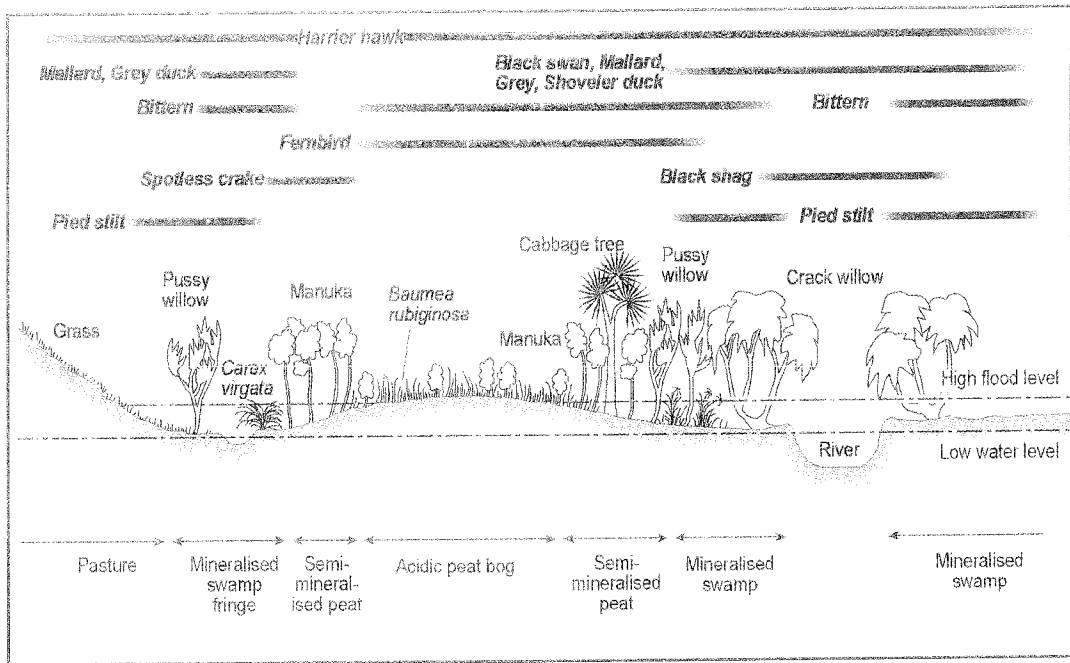


Figure 26.4 Diagrammatic representation of Whangamarino swamp, Waikato, to illustrate habitat zonation of resident wetland birds (redrawn from Ogle and Cheyne 1981).

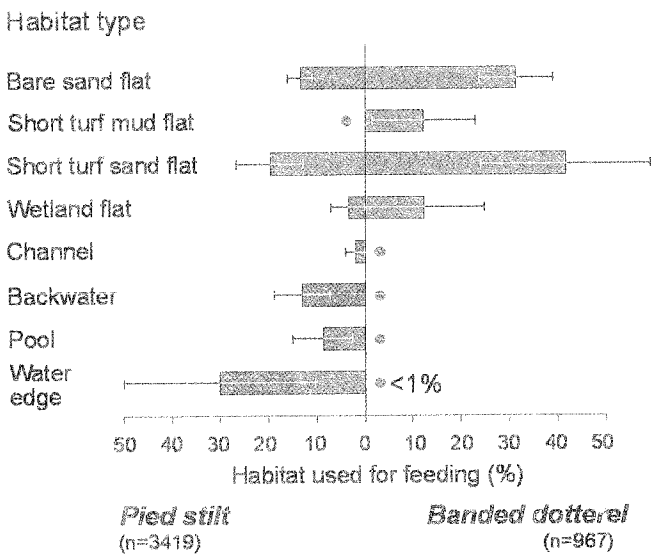


Figure 26.5 Contrasting habitat “blueprints” for banded dotterel and pied stilt, developed to illustrate the different micro-environments exploited by these species at Lake Wairarapa (redrawn from Moore *et al.* 1984).

rare insight by quantifying the dietary similarities of little black shags (*Phalacrocorax sulcirostris*) and little shags (*P. melanoleucos*) feeding in two Rotorua lakes. He demonstrated a significantly greater dependence on freshwater crayfish (*Paranephrops planifrons*) by little shags in both lakes, but otherwise, the two species ate the same size ranges of the small fishes (bully *Gobiomorphus cotidianus*, smelt *Retropinna retropinna*, goldfish *Carassius carassius* and koaro *Galaxias brevipinnis*) in remarkably similar relative proportions (Fig. 26.6). Unfortunately, the study did not extend beyond diet analyses to the timing and location of their feeding effort and thus further elucidate how these two species partitioned what appeared to be the same resource.

A particularly interesting but still largely uninvestigated comparison is that between the native grey duck *Anas superciliosa* and the introduced mallard *A. platyrhynchos*—interesting in that it may highlight a mechanism contributing to the ongoing displacement of grey duck by mallard (Williams 2004). Potts (1977b) asserted the two species “... occupy a similar feeding niche”, largely on the basis of their similar bill morphologies and their being “... commonly seen feeding in association”. Together, Turbott (1946) and Balham (1952) demonstrated that the two species ate similar foods, but no convincing case for competitive dietary overlap has yet been advanced.

Food habit studies, while no longer considered a “trendy” scientific investigation, remain vital to determining how similar species partition resources, and they form the basis for wider community studies which

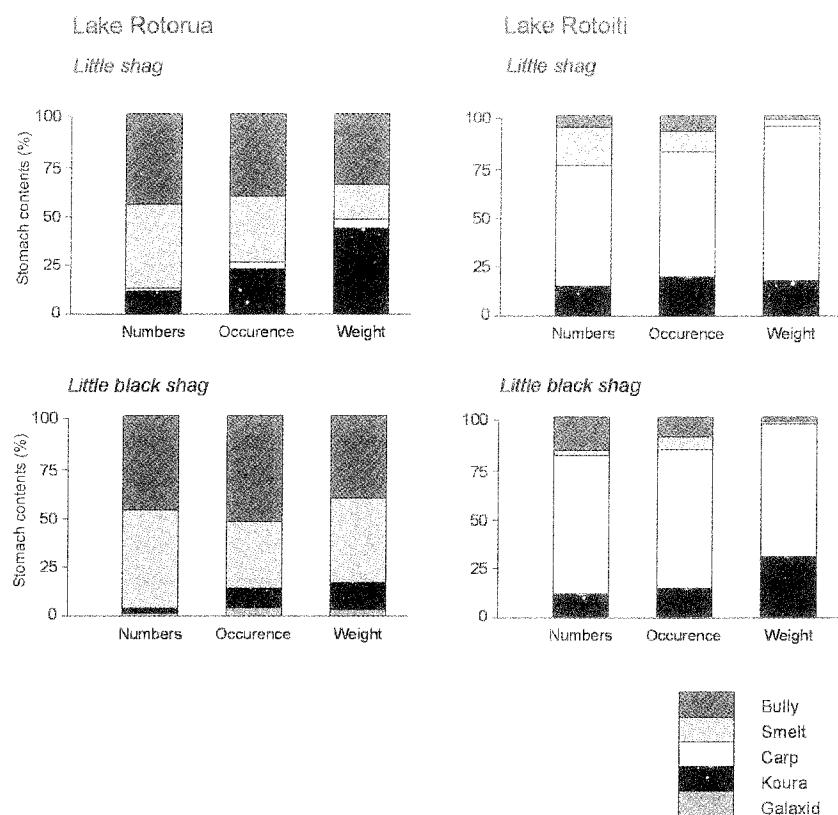


Figure 26.6 Comparative diets of little shags and little black shags at Lake Rotorua and Lake Rotoehu (redrawn from Potts 1972).

seek to identify trophic relationships and energy flows within the wetland ecosystem (see Raffaelli 1997). That there are so few of these fundamental studies is a major oversight and it is likely to prove an impediment to understanding the ecological roles and relationships of birds within wetland environments and managing freshwaters to maintain and maximise their biodiversity.

RELATIONSHIPS OF BIRDS WITH OTHER ELEMENTS OF THE LAKE ECOSYSTEM

Food habit studies lie at the heart of any attempt to evaluate trophic interactions between birds and other biotic components of the lake ecosystem. To quantify energy flow through food webs is a daunting task if it is to be done realistically and thoroughly, and given that few have (Hall and Raffaelli 1991, 1993), it is perhaps unsurprising that there is no single comprehensive example for a New Zealand lake.

It is not as if birds are an unimportant component of the lacustrine ecosystem. Most sit at the upper trophic levels: shoreline invertebrate feeders such as wading birds are at least secondary consumers, and piscivores such as shags and herons are primarily tertiary consumers. Some of the omnivorous species, e.g., mallard and scaup, may transcend at least two trophic levels. Without an under-

standing of these trophic relationships, determining the mechanisms that influence a particular bird's presence and abundance within the lake avian community is an unreachable goal.

Evaluating ecosystem interactions is easier, however, where the food chains involved are short, e.g., for herbivores. Perhaps the best New Zealand example (and one of the better ones internationally) is the work of Stuart Mitchell and colleagues, evaluating the role of black swans in the fluctuating phytoplankton-macrophyte domination of shallow Otago lakes.

Black swans appear to eat almost any vascular aquatic plant, except perhaps *Ceratophyllum emersum* (Sagar *et al.* 1995). In a study of Tomahawk Lagoon, a shallow Otago lake that alternates between being dominated by phytoplankton and by aquatic macrophytes, Mitchell *et al.* (1988) demonstrated that winter swan numbers were directly correlated with the biomass of submerged macrophytes. Because the winter macrophyte biomass was determined by the intensity of algal bloom in the previous summer, so the maximum

winter swan population was also related, inversely, to summer phytoplankton productivity. McKinnon and Mitchell (1994) confirmed that the swan-macrophyte relationship applied to a wider range of other lakes of differing size and trophic status, and that the swan-phytoplankton relationship applied best to lakes with silty sediments rather than those with sand or fine clay. They concluded that sediment rather than the shading effects of phytoplankton may constrain macrophyte biomass and swan populations. But could swan grazing of macrophytes in some way directly contribute to the onset and stabilising of the phytoplankton-dominated state?

By considering the effects of swan grazing within a broader conceptual model (Fig. 26.7), Mitchell and Wass (1995, 1996a,b) were able to address aspects such as grazing rate in relation to macrophyte growth rate and indirect effects through nutrient recycling and bioturbation. They identified a finely-tuned response by the swan population to changes in plant biomass, but showed that the swans had little direct effect upon macrophyte levels through grazing consumption. Swans consumed only 16% of net summer benthic primary production; their grazing rate of 0.007 g consumed/g/day was small relative to plant growth rates of 0.06–0.10 g/g/day and loss rates during periods of macrophyte decline of 0.07–0.18 g/g/day. The swan faecal nutrient input of total

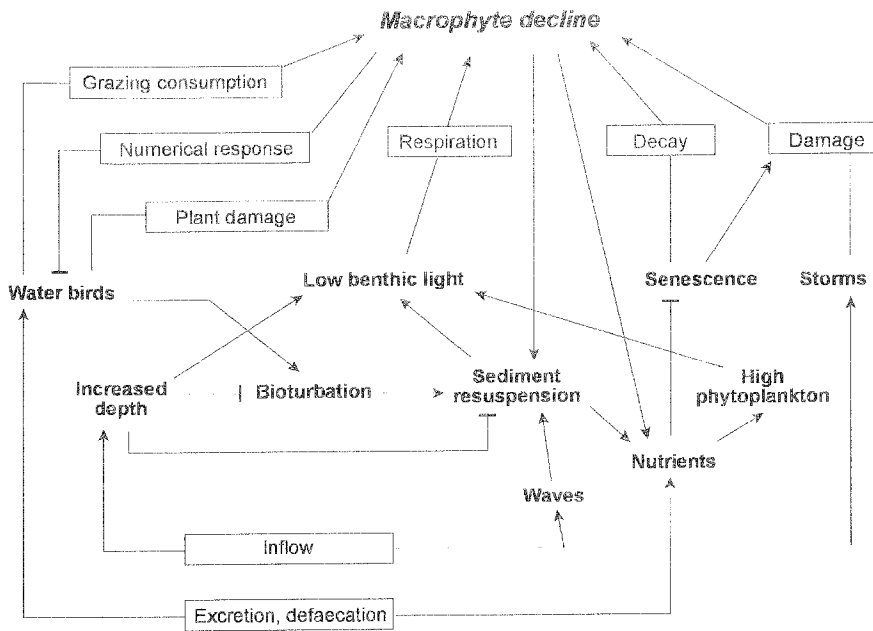


Figure 26.7 Conceptual model for the roles of water birds (e.g., black swans) and other seasonal factors in the decline of macrophyte biomass in shallow lakes. Arrow-tipped lines denote stimulatory effects, lines beginning with bars denote inhibitory effects, while boxes denote confounding factors. Detrital pathways, external nutrient and sediment loads, some components of internal nutrient loads, and food chain interactions affecting phytoplankton all omitted (from Mitchell and Wass 1996b).

and biologically available N and P was shown to be very small in relation to the total nutrient pool in the water and benthic vegetation, leading the researchers to conclude that the waterfowl were insignificant contributors to the nutrient dynamics of lakes. Overall, lack of light was considered the important driver of macrophyte decline. However, Mitchell and Wass (1996b) did opine that despite the cumulative effect of waterfowl grazing consumption being small, under light conditions marginal for macrophyte growth, it could be critical for keeping macrophyte biomass below the threshold for macrophyte dominance.

The beauty, and instruction, of Mitchell's studies is their step-wise focus on elements within a well-constructed model of ecological interaction (Fig. 26.7). In the process, they have provided the only definitive New Zealand data on a long-standing and vexed issue in freshwater management—the relationship between waterfowl grazing and eutrophication (Manny *et al.* 1994; Sagar *et al.* 1995 and references therein).

IMPACT OF SUCCESSION

The successional journey of a wetland—from a shallow lake becoming filled with sediments and organic matter, transforming to a mosaic of open water patches and swamp, and eventually becoming dominated by local climax vegetation—is one measured on a geological time scale. Nevertheless, it is a process that utterly transforms the composition of the resident avian community and profoundly, and differentially, affects species' abundances (Weller 1981 and references therein) (Fig. 26.8).

Classic demonstrations of this process have come mostly from continental studies in the northern hemisphere, where the wetland avifauna is rich in species, each with narrow niches necessitating responses to quite minor changes in vegetation diversity, structure and heterogeneity (Weller 1981; USEPA 2002 and references therein). No New Zealand study has yet demonstrated this effect, most likely because few New Zealand birds are ecologically restricted to palustrine environments and because the widespread obliteration of palustrine wetlands (Howard 1982) and ecotone vegetation around lakes during a century of rampant pastoral development has hidden the relationship.

Only one passerine, the fernbird *Bowdleria punctata*, is an obligate resident of swamp vegetation, although its occurrence there seems dependent on scattered small woody shrubs amongst the reeds and rushes. The presence of other common small passerines, e.g., fantail (*Rhipidura fuliginosa*), grey warbler (*Gerygone igata*), silvereye (*Zosterops lateralis*), yellowhammer (*Emberiza citrinella*) or chaffinch (*Fringilla coelebs*) may require a more advanced seral stage and small patches of woody vegetation (Anderson and Ogden 2003). Present-day ground-dwelling birds in swamp reeds and rushes include marsh crake (*Porzana pusilla*), spotless crake (*P. tabuensis*), banded rail (*Rallus philippensis*), brown teal, bittern and pukeko. Although some of these have patchy or restricted contemporary distributions, and pukeko readily exploits other habitats, a relationship between successional stage, vegetation type and the abundance of some of these species is clearly awaiting detection (Kaufmann 1987).

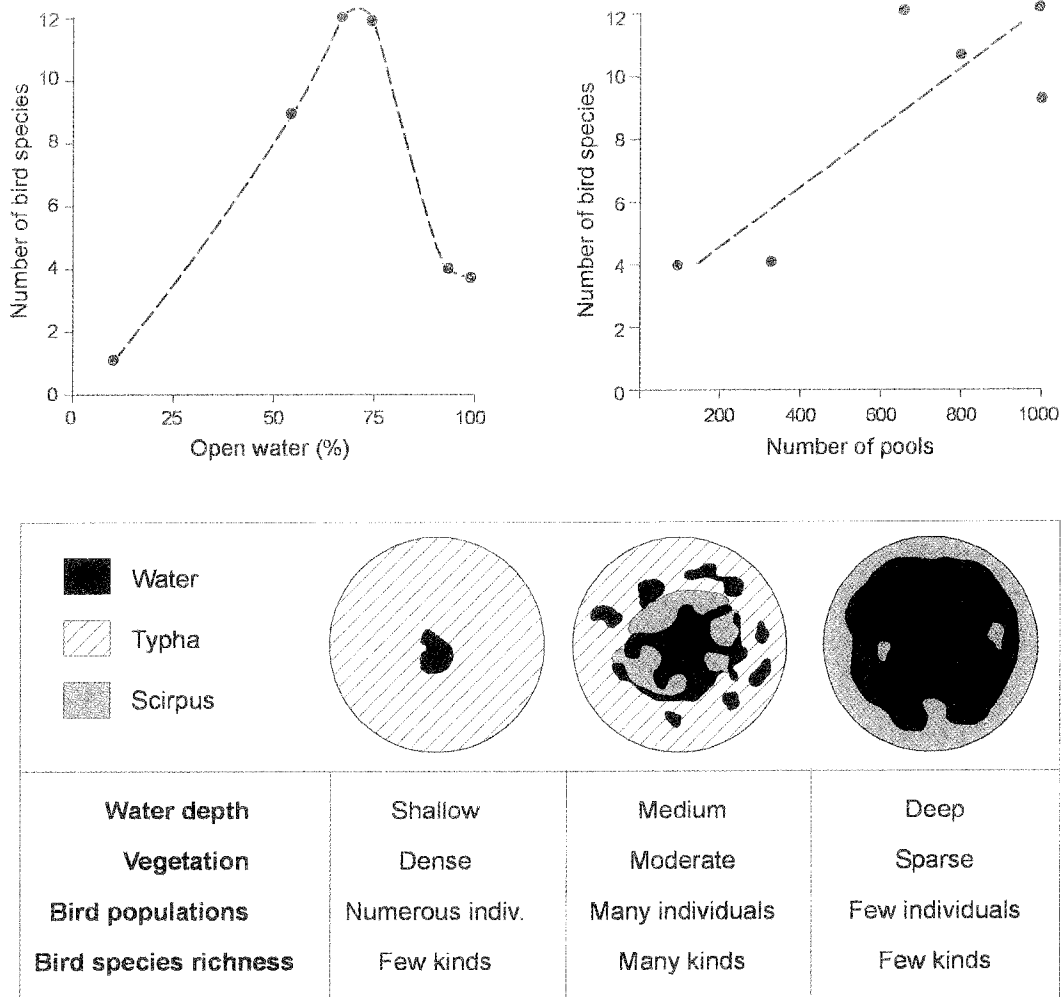


Figure 26.8 A generalised portrayal of the effect of vegetation succession on bird diversity and abundance (redrawn from Weller 1981: figs. 6,7). Bird species richness is related to area of open water and the interspersions of open water amongst dense emergent

GAPS IN INFORMATION

There are four major, indeed glaring, gaps in our knowledge of the avian communities of freshwater wetlands:

- How community composition changes over time
- What birds eat
- Energy flows within lacustrine communities ... where do birds fit in?
- Habitat manipulation experiments to maximise community diversity

If species lists for specific sites and times are the measure, the composition of avian communities of New Zealand's lakes or swamps is well known (e.g., Moore *et al.* 1984). However, studies that have monitored changes in species composition and species abundances over even relatively short time periods (i.e., for more than a year) are rare. As a consequence, a New Zealand perspective on how birds respond to and exploit one of the planet's more productive

environments remains unavailable. A comparison of a species list from one specific survey with that of all other listings for that site (e.g., O'Donnell 1985 for Lake Ellesmere) emphasises just how extensive the lists of "uncommon" species may be and gives a hint of ecological opportunities that are available for new additions to New Zealand's resident avifauna. Long-term studies of wetland birds mirroring those on New Zealand estuaries (Sagar *et al.* 1999; Whelan *et al.* 2003) are sorely needed.

To understand ecological relationships amongst different bird species at lakes, some quite fundamental biological information is lacking. The foods eaten define pathways of ecological interaction, and quantities eaten help define energy flows between trophic levels. Simple though they may be, food habit studies represent the single largest and most important of investigations needed to advance ecological understanding of interactions between birds and wetlands. There is not one published study of foods of a benthic-feeding bird in New Zealand lakes

(e.g., dabchick, coot, scaup), the foods of specialist waterfowl are known only from Australian studies (shoveler, grey teal; Marchant and Higgins 1990) or not at all (brown teal), and Potts' (1972, 1977a) comparison of diets of sympatric small shags (Fig. 26.6) appears to be unique.

Managing wetlands to maintain or enhance biological diversity is part of modern environmental management (e.g., Vant 1987). While there is an abundance of international literature supporting wetland restoration (e.g., Finlayson and Larsson 1991; USEPA undated) and their management for birds (e.g., Scott 1982) and no shortage of local need (e.g., Lake Ellesmere; Taylor 1996), New Zealand examples of wetland management experiments aimed at maximising community diversity are lacking. Being so visible, birds may serve as excellent indicators of wider ecosystem responses to experimental manipulations of nutrient levels, vegetation composition or water-level regimes (USEPA 2002). But for such information to be applied in New Zealand, good empirical data demonstrating such responses by New Zealand birds are needed.

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Chapter 25

Littoral invertebrate and fish communities

David Kelly and Robert McDowall

INTRODUCTION

Highly variable topography, climate and geology in New Zealand contribute to a wide variety of lake types, and thus habitats for biota in lakes. Generally speaking, the littoral zone in lakes is defined as the area of lake bottom contained between the high water mark of the lake and the maximum depth to which rooted (or attached) aquatic plants grow (Wetzel 1983). The fauna of the littoral zone can be remarkably diverse—members of nearly all New Zealand freshwater invertebrate phyla are present in lakes. There is also considerable interaction between the biota that use the pelagic (mid-water) zone of lakes and those that inhabit the littoral zone, with many species utilizing both habitats at various stages in their life histories. Littoral fauna play a key role in the ecology of lakes by controlling the biomass of epiphytic algae and recycling detrital materials in sediments, and they are an important food source for fish. In clear oligotrophic lakes, where the littoral zone can extend to considerable depths, littoral productivity can underpin much of the productivity of lake fisheries and is well correlated with fish yield and biomass (Rasmussen and Kalff 1987; James *et al.* 1998).

Several published reviews provide comprehensive discussions of New Zealand taxonomic groups and their zoogeographic relationships with littoral fauna of the southern hemisphere (Winterbourn and Lewis 1975; Forsyth 1987; McDowall 1987). This chapter will focus more on the ecological relationships between littoral invertebrate and fish communities and their habitats within the littoral zone, and summarize some of the recent work on the ecology of littoral zones. However, a great deal remains unknown of the basic taxonomies and life histories of littoral invertebrates in New Zealand lakes, and even less is known about their ecology. We also outline some methods for sampling littoral invertebrate and fish communities, and describe some of the difficulties and limitations of these methods. Lastly we illustrate some

unique examples of community dynamics and interactions amongst littoral biota and their importance to the ecology of lakes.

PHYSICAL HABITATS OF THE LITTORAL ZONE

Because of gradients in their physical, chemical and biological properties, lake littoral zones provide a diverse array of habitats for invertebrates and fish. These habitats can range from wave-exposed rocky shorelines to soft-bottomed characean meadows extending as deep as 40 m in clear lakes in New Zealand (Schwarz *et al.* 2000). Above the littoral zone are areas influenced by wind-generated spray and waves—they are termed the epilittoral and supralittoral zones. Depending on the degree of wave exposure, as well as on the amount of fluctuation in lake level, this zone may be rich in emergent vegetation, forming a gradual gradient to submersed species, or it may be entirely void of vegetation and be composed of rocky or sandy substrates. Bordering the lower boundary of the littoral zone is the littoriprofundal zone, which forms the transition between the plant-dominated littoral zone and the bare mudflats of the profundal zone (Fig. 25.1).

Sub-zones of the littoral zone

In New Zealand lakes, the littoral zone can be broken into four major sub-zones based on physical characteristics such as substrate composition and wave exposure, as well as on the plant communities that grow there. Large differences in these physical and biological attributes contribute to extremely variable habitats for littoral invertebrate and fish communities.

Eulittoral zone

The uppermost portion of the littoral zone is the eulittoral zone, which lies between the high water mark of the lake and its shore. The habitat can be variable,

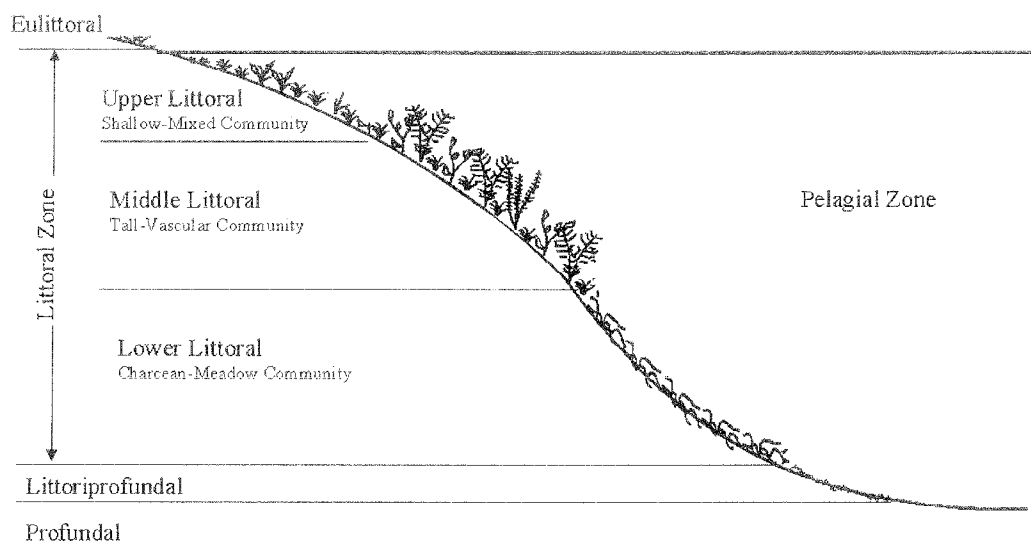


Figure 25.1 Diagram of the littoral zone in most deep (> 40 m) New Zealand lakes, showing the various plant communities.

depending predominantly on the exposure of the shoreline to wind-driven surface waves (Table 25.1). Habitats can range from rocky-bouldery shorelines in exposed areas, to sandy beaches in exposed depositional areas or soft-bottomed mud flats with emergent vegetation in sheltered embayments. The area of habitat can also vary considerably, depending on annual fluctuations in water level, as well as the slope of the lake margins. Many of New Zealand's rivers and lakes are used for hydroelectric power generation. Flows are managed for power use, resulting in substantial water-level fluctuations, and lengthy water

draw-downs can cause extended periods of desiccation, as well as a general increase in the size of the eulittoral. Desiccation, temperature fluctuations and exposure to waves largely dictate the types of biota that can utilize these habitats.

Upper littoral zone

The upper littoral zone forms the first permanently-submersed portion of the littoral zone, and extends typically to around 2 m depth. This portion of the littoral is greatly affected by exposure to waves and by temperature

Table 25.1 Habitat features of differing portions of the littoral zone in New Zealand lakes. Note that these features pertain mainly to deeper clear lakes, with depth ranges being considerably reduced (or absent) in more eutrophic, turbid, or shallow lakes.

Habitat feature	Eulittoral	Upper littoral	Middle littoral	Lower littoral	Littoriprofundal
Depth range	High water mark to 0 m	0 – 2 m	2 – 7 m	Variable 7 – 40 m	Variable 8 – 70 m
Substrate ¹	B/C/S/M	C/S/M	S/M	M	M
Wave exposure	High	High	Medium	Low	Low
Plant cover	Variable 0 – 100%	Medium 20 – 100%	High 50 – 100%	High 50 – 100%	Low 0 – 20%
Canopy height	Low – medium 10 – 50 cm	Low 0 – 20 cm	High 20cm – 3m	Medium 15 – 100 cm	Low 0 – 20 cm
Important physical factors	Wave exposure Desiccation	Wave exposure Temperature	Light Wave exposure	Light	Light
Dominant plant communities ²	EM , LMC	LMC, FLP	TVC	CM	CM, CF

¹ B = boulder, C = cobble, S = sand, M = silt/mud

² EM = Emergent macrophytes, LMC = Low mixed Community, FLP = floating leaved plants, TVC = Tall vascular community, CM = Characean meadows, CF = Cyanobacterial filaments

fluctuations (Hawes *et al.* 2003). The plant communities here can consist of a variety of vegetation types, depending on the degree of exposure, and include emergent vegetation along its upper boundary (e.g., *Carex*, *Typha*), free-floating or attached floating leaved plants (e.g., *Azolla*, *Potamogeton*), and a mix of low-growing submerged plants, e.g., *Lilaeopsis*, *Isoetes* (termed the low-mixed community), extending to depth. The plant cover is often somewhat patchy unless the shoreline is very sheltered, or it can be completely absent, such as in lakes with large fluctuations in water levels (e.g., Lakes Moawhango, Hawea and Tekapo) or lakes that are highly turbid (e.g., Lake Pukaki) (James and Graynoth 2002). The substrates in this zone are also quite variable, depending on the amount of wave exposure, but are usually sandy, with intermixed cobbles and boulders in more exposed shorelines, and a mix of sand and silt in sheltered areas.

Middle littoral zone

The middle littoral zone marks the point in the littoral at which wave exposure diminishes, and it is dominated by taller rooted vascular plants such as milfoils, pondweeds and oxygen weeds (termed the tall vascular community, e.g., *Myriophyllum*, *Elodea*). The plant cover and canopy height are usually considerably greater in this portion of the littoral, extending to 3 m high for some macrophyte species (e.g., *Lagarosiphon*, *Ceratophyllum*), and substrates are typically finer (commonly sand and silt) than in the shallower littoral (Howard-Williams *et al.* 1987). In clear lakes, this portion of the littoral extends to around 7 m depth, beyond which rooted vascular plants are unable to grow, due to low light levels and pressure inhibition of root formation (Coffey and Wah 1988).

Lower littoral zone

The lower littoral zone demarks the transition to plant species adapted to low light, the charophytes (Family Characeae), which are actually delicate macroalgae that form intermediate height (0.2–1 m high) meadows extending to the bottom of the littoral zone. The depth to which charophytes grow has been shown to be closely related to light attenuation (Schwarz *et al.* 2000). In clear glacial and volcanic lakes in New Zealand, charophyte meadows can extend to depths below 40 m (e.g., Lake Wakatipu), while in more productive eutrophic or mesotrophic lowland lakes (e.g., Lake Okaro), turbid lakes (e.g., Lake Pukaki), or humic-stained lakes (e.g., Lake Manapouri) charophytes can be greatly reduced or absent.

Littoriprofundal zone

Forming the transition between the plant-dominated littoral zone and the soft-sediment-based profundal zone is a region that is frequently colonized by non-vascular

plant species adapted to extremely low light, such as filamentous cyanobacteria and bryophytes. In clear lakes these zones can extend up to 70 m, as in Lake Coleridge (James *et al.* 1998). Invertebrate communities within the littoriprofundal tend to be similar to the profundal communities, dominated by sediment-dwelling worms, but more motile species such as crayfish can use this zone for foraging.

Lake type and littoral habitats

The diversity of lakes in New Zealand results in varying development of the littoral zone, providing varied habitats for colonization by invertebrates and fish. This is influenced mainly by the physical and chemical properties of the lake, which are, in turn, determined largely by catchment geology, how the lake was formed, climate, catchment vegetation, and inflows and outflows to the lake (see Chapter 24). Two main factors control the development of littoral zone habitats (e.g., plant growth, substrates)—the bathymetry of the lake, which determines bottom slope and fetch (i.e., wave exposure), and water clarity, which determines photic depth and thermal stratification. These properties vary between lake types, resulting in considerable differences in littoral habitats among lakes.

Glacial lakes are located only in the Southern Alps of the South Island (Irwin 1975). In pristine condition they are commonly deep and clear, with extensively developed littoral zones that often extend to depths >25 m. As glacial lakes often form in mountainous valleys, their shorelines slope steeply and often restrict littoral zone macrophytes. Additionally, they commonly occur in long wind-swept valleys, creating high wave-exposed rocky shorelines over much of their circumference. Several hydro lakes, such as those in the Waitaki Valley, are quite turbid due to suspended glacial fines, and as such have poorly developed littoral flora and fauna. Alternatively, hydro lakes of the Clutha Valley, such as Lake Dunstan, are clear and shallow. Smaller sub-alpine and montane tarns and lakes are also abundant in the Southern Alps and may have extensive littoral plant development over their entire bottom because of their clear waters and shallower depths. Man-made lakes constructed for hydroelectric power generation tend to be located in high-elevation mountain valleys and tend to resemble glacial lakes. Nuisance growths of the adventive macrophyte *Lagarosiphon major* have been a major problem in Lake Dunstan, with surface-growing weed beds along much of its margins (see Chapter 27).

New Zealand also has many volcanic lakes, particularly in the central North Island. The bathymetry and water quality of volcanic lakes can vary considerably—some are clear and deep, with extensive littoral zone development (e.g., Lakes Taupo and Tarawera), whereas others are

shallow and turbid (e.g., Lakes Okaro and Ngapouri). Their shorelines are usually more gently sloping than those of glacial lakes, contributing to well-developed littoral plant communities around much of their circumferences. However, eutrophication has affected many volcanic lakes, such as the Rotorua lakes, leading to phytoplankton blooms, and littoral macrophytes may now extend to only very shallow depths (< 3 m in Lake Rotoiti) because of reduced water clarity.

Lowland forested lakes are present in a few areas of New Zealand, especially the West Coast of the South Island, and these lakes characteristically have humic-stained waters. Because of the lower clarity of these lakes, their littoral plant communities are commonly restricted to shallower depths (< 10 m) than in clear lakes. Lowland coastal lagoons and dune lakes are usually productive and may have dense growths of macrophytes over most of their bottom area. Coastal dune lakes are usually slightly deeper (< 10 m), with low light penetration as measured by secchi depths (usually < 3 m), and macrophyte development to < 4 m (Cunningham *et al.* 1953). Coastal lagoons are usually very shallow (< 4 m), and can vary in their macrophyte coverage, depending upon turbidity. Some are influenced by incursion of seawater at high tides, and are thus brackish, especially close to their mouths, or where there are extensive permeable gravel bars that separate them from the sea. Settlement and development on lowland plains recently in New Zealand's history have resulted in significant flood control problems and eutrophication of many of these lakes, in some cases resulting in collapses of the dense macrophyte beds that once covered them (e.g., Lake Ellesmere) (Gerbeaux and O'Connor 1992).

INVERTEBRATES OF THE LITTORAL ZONE

Littoral zones provide habitat for generally the most diverse portion of the macro-fauna in lakes, especially the invertebrates. One of the unique characteristics of littoral invertebrates in New Zealand lakes is their high degree of endemism and low overall diversity in comparison with faunas in lakes in northern temperate North America and Europe. Several taxonomic groups that are generally abundant in northern hemisphere lakes are poorly represented or completely absent in New Zealand lakes. Certain trophic guilds of invertebrates are thus poorly represented and, specifically, very few species are known to utilize either live macrophyte material or coarse organic matter (Suren and Lake 1989; Winterbourn *et al.* 1981). In New Zealand lakes, littoral invertebrate faunas consist mainly of a variety of larval insects, molluscs, annelids, crustaceans and mites, with about 10–25 species observed in most lakes. A survey of 20 South Island lakes showed a

total of 50 benthic and littoral taxa, with an average species richness of 12.4 taxa per lake (Timms 1982).

The distributions and productivity of littoral invertebrate communities have been related to physical, chemical and biological variables in lakes, both within New Zealand and abroad (Forsyth 1978; Timms 1982; Rasmussen and Kalff 1987; Lalonde and Downing 1992; James *et al.* 1998; Weatherhead and James 2001). Relationships between lake productivity and invertebrate abundances and species composition have, surprisingly, been relatively weak. Forsyth (1978) observed greater abundances of benthic invertebrates in eutrophic lakes; however this occurred only at the most eutrophic end of the spectrum, and mesotrophic and oligotrophic lakes were almost indistinguishable. In a survey of twenty South Island Lakes, Timms (1982) found no relationships between invertebrate community composition and lake trophic state. Stronger relationships have been reported between invertebrate communities and physical and biological habitat variables such as substrate composition, macrophytes, shoreline slope and wave exposure (James *et al.* 1998; Weatherhead and James 2001).

Habitat associations of littoral invertebrates

The composition of invertebrate communities in littoral zones has been clearly related to habitat variables such as substrate composition, organic detritus, and macrophyte species, and zoobenthic communities can generally be classified according to the habitats in which they occur (Fig. 25.2). Based on a survey of littoral invertebrates from nine New Zealand lakes, Weatherhead and James (2001) classified the fauna into four broad categories: rocky shoreline biota, macrophyte-associated biota, sediment-associated biota and sublittoral biota. Similar classifications have been made in earlier reviews (Winterbourn and Lewis 1975; James *et al.* 1998). The following sections give broad descriptions of communities in various habitats of the littoral zone, and the variables that contribute to their distributions.

Invertebrates of rocky shores (upper littoral)

The invertebrate fauna of rocky shorelines can almost be described as a guild similar to those found in flowing waters, due to the similarities in both their composition and related physical characteristics. The fauna is usually dominated by larval insect taxa such as mayflies, caddisflies and, in some cases, stoneflies. Weatherhead and James (2001) observed a significant positive correlation between substrate index (a large substrate index corresponds to large particle size) and mayfly abundances in a multi-lake survey of littoral communities (Fig. 25.2a). In rivers, mayflies typically inhabit swift-flowing riffles and runs with low plant or algal development. In lakes the only habitats where

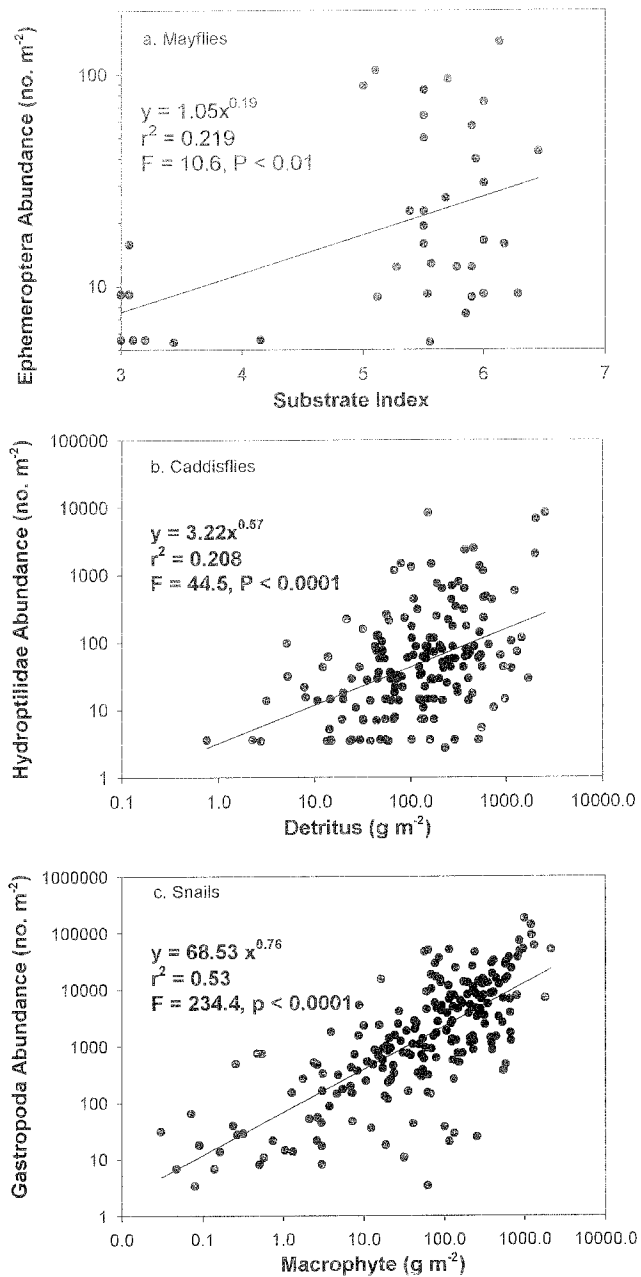


Figure 25.2 Relationships between substrate index, detritus and macrophyte biomass for select taxonomic groups of littoral invertebrates from nine New Zealand lakes. Modified from Weatherhead and James (2001).

mayflies are found are wave-swept rocky shorelines, where substrate movement and wave action prevents significant development of algal biofilms or rooted plants.

James *et al.* (1998) observed mainly hydroptilid caddisflies (*Paroxyethira* spp.) and chironomids in shallow wave-washed areas of Lake Coleridge, but also occasionally found mayflies (*Deleatidium* spp.) and stoneflies (mainly *Zelandobius* spp.). Most of these taxa found along rocky shores are either grazers or generalist browsers that consume either the organic biofilms growing on the surfaces of the stones (epilithon) or organic detritus

deposited between them. Hydroptilidae abundance was positively related to detritus in Lake Coleridge (Fig. 25.2b). Although this community is generally not considered to be the most diverse portion of the littoral, on occasion relatively high numbers of taxa can be found (James *et al.* 1998).

Invertebrates of macrophyte beds (upper-lower littoral)

Invertebrates species associated with macrophytes, also termed epibenthic invertebrates, are generally recognized to be the most diverse and productive component of the lake fauna (Winterbourn and Lewis 1975). Macrophytes provide a complex three-dimensional architecture for colonising invertebrates and can provide a variety of food sources, as well as refuge from predators (Gotceitas 1990; Connolly 1994). Dense vegetation can be sought out specifically by prey species, many of which utilize this single habitat exclusively (Uiblein *et al.* 1996). Despite the large pool of organic carbon stored in the macrophytes, only two groups in New Zealand lakes are generally recognized to directly consume macrophytes, the omnivorous crayfish (*Paranephrops* spp.) and the larval aquatic lepidopteran *Nymphula nitens*, which lives exclusively on its host plant *Elodea canadensis*. Most of the other fauna either consume epiphyton growing on the surfaces of macrophytes (e.g., snails, chironomids, caddisflies) or are predatory on other invertebrates (dragonflies, damselflies, and mites) (Talbot and Ward 1987; James *et al.* 2000b).

A few studies have attempted to characterize the types of fauna associated with different species or types of macrophytes in New Zealand, and in some cases have found reasonable relationships (Biggs and Malthus 1982; Talbot and Ward 1987; James *et al.* 1998). Biggs and Malthus (1982) identified 26 taxa from littoral habitats of the upper Clutha valley, dominated by either the native macrophytes *Myriophyllum propinquoquum*, *Potamogeton cheesmanii* and *Chara* sp., or the adventives *Lagarosiphon major*, *Elodea canadensis* or *Ranunculus fluitans*. In their survey, charophytes were the most productive habitats, although there were differences in the species that inhabited the different macrophyte community types. Similarly, in oligotrophic Lakes Coleridge and Wanaka, the greatest abundance and richness of epibenthic invertebrates was in characean meadow communities, followed by tall vascular communities, and was least in the shallower plant or non-vegetated shorelines (James *et al.* 1998) (Fig. 25.3).

Dominant taxa of the macrophyte beds include the prosobranch snail *Potamopyrgus antipodarum*, which is reported as the most abundant taxon in several studies of invertebrates in New Zealand lakes and rivers (Timms 1982; Talbot and Ward 1987; James *et al.* 1998;

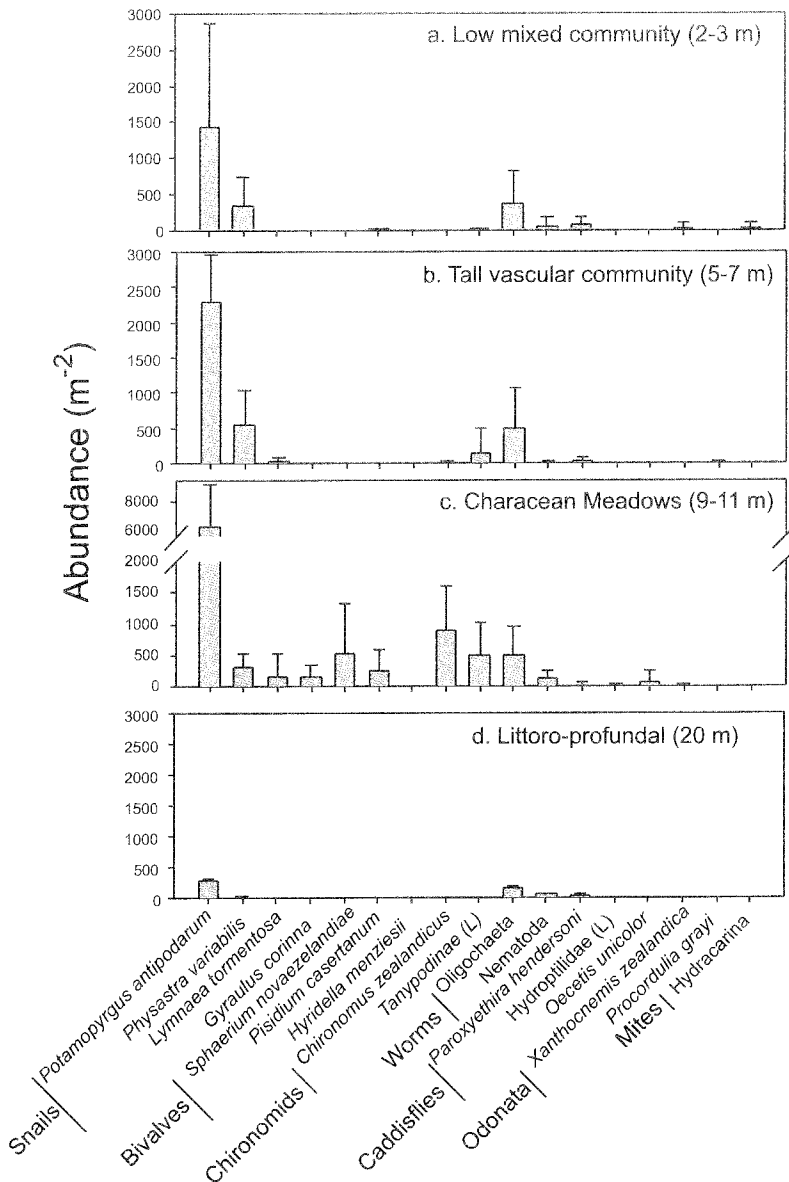


Figure 25.3 Abundances of epibenthic invertebrate taxa in littoral habitats along a depth gradient in Lake Wanaka.

Weatherhead and James 2001). Other common gastropod taxa include *Physa* spp. and *Gyraulus corinna*, which are widely distributed in New Zealand lakes (Winterbourn 1973). Weatherhead and James (2001) reported a significant positive relationship between macrophyte biomass and the abundance of gastropods in nine New Zealand lakes (Fig. 25.2c), presumably reflecting the greater availability of epiphyton, which they consume, growing on macrophytes. More generalist browsers, such as chironomids, are also a dominant component of most epibenthic communities, with *Chironomus zealandicus* and *Macropelopia* spp. being common throughout New Zealand (Forsyth 1978, 1986; Timms 1982). Caddisflies such as *Oecetis unicolor* and *Paroxyethira* spp. are often the dominant browser fauna in macrophyte communities (Fig. 25.2b). The dominant predatory invertebrates associated with macrophytes are mainly damselflies and dragonflies,

with *Procordulia grayi* and *Xanthocnemis zealandica* being fairly ubiquitous (Timms 1982; Weatherhead and James 2001). Mites (Acari), which are predatory on smaller organisms such as protozoans and zooplankton, are usually present in macrophyte-dominated areas, and the arachnid *Dolomedes* has occasionally been observed in North Island lakes (Winterbourn and Lewis 1975). *Dolomedes* usually inhabits the shallow vegetated margins of lakes, and carries down air bubbles on specially adapted abdominal hairs to form an underwater air-filled lair amongst the vegetation. Some dytiscid beetles have been collected, but they are usually at low densities (Winterbourn and Lewis 1975; Talbot and Ward 1987).

Seasonal changes in epibenthic invertebrates in mesotrophic Lake Alexandrina were examined by Talbot and Ward (1987) over a two-year period. They observed a peak in the biomass of invertebrates around mid-summer (Feb-March), although inter-site variability was far greater than seasonal variation. Winter tended to be the time when most taxa were present, possibly because mid-summer periods were more affected by insect emergence. It appears that many of the taxa present in New Zealand lakes are multi-voltine (have more than one generation per year) and thus population fluctuations depend on the timing of emergences. James *et al.* (unpublished report) also examined seasonal changes in the composition of littoral invertebrates in Lake Taupo in relation to water-level fluctuations from hydroelectricity water use. They observed the greatest seasonal changes in shallow littoral communities subject to periodic drying during winter draw-down periods. Most of the species were able to recolonize dried areas in the shallow littoral during a single spring and summer, with the exception of years where fluctuations were of sufficient magnitude to damage macrophyte beds located deeper in the littoral zone.

Invertebrates of soft sediments (middle-lower littoral)

At the base of macrophytes and in non-vegetated areas of the littoral zone is a relatively rich fauna that inhabits fine, organically-rich sediments. Large inputs of material from shoreline vegetation, macrophytes and settling phytoplankton contribute to high organic carbon pools in the sediments, and these are utilized by this predominantly detritivorous community (Kornijow *et al.* 1995; James

Methods for assessing littoral invertebrates and fish

One challenge for limnologists working in littoral zones is obtaining accurate quantitative estimates of the abundances of littoral invertebrates and fish. The littoral zone in most lakes includes a variety of habitats, thus necessitating a large number of replicate samples and sites to encompass spatial variability. Additionally, habitats within the littoral zone can be extremely diverse (e.g., macrophytes, boulders, fine sediment), requiring the use of several sampling methods. This often makes it difficult to quantitatively compare samples collected from differing habitats, sites or lakes, and frequently only estimates of relative abundances can be made.

Littoral invertebrates sampling methods

Benthic grabs: Grab samplers are generally metal-box-type devices with jaws to penetrate and close around the bottom substrate. Common grab types include the Ekman grab, used primarily to sample soft bottoms, and the Petersen grab, which has a slightly heavier construction and can penetrate harder substrates such as sand and gravel. A diver-operated Wisconsin grab, which consists of a set of jaws and a large fine-mesh catch bag, can be used to sample both macrophytes and sediments, but is limited to the depth and conditions at which divers can operate (Fig. 25.4).

Sweep netting: A common method for obtaining macrophyte-associated fauna is by sweeping a fine meshed net through the stands of macrophytes. This has the advantage of sampling a large area of the littoral zone relatively quickly, but does not give a quantitative (per area of lake bottom) estimate of the species present.

Coring: Coring samplers such as gravity corers and freeze corers are used in some instances to sample sediment-associated benthic fauna. All consist of a weighted tube that is driven into the lake sediments, with the sediments retained either in the interior of the core (gravity or piston corer), or frozen to the outside of the core (freeze core). The corer can be a very useful quantitative sampler if only a small area needs to be sampled.

Hess/Surber sampler: On rocky cobbles of the shallow littoral zone, where grabs are unable to penetrate the substrate, a known area of substrata can be brushed into a fine-meshed Surber or Hess sampler.

Fish sampling methods

Quantitative assessments of littoral fish communities pose an even greater challenge, owing to their greater motility and heterogeneous distribution in lakes. Methods include seining, trapping, gillnetting and electric fishing, but catch rates are usually expressed in terms of catch per unit effort (CPUE) rather than actual abundances. Only very labor-intensive statistical methods such as mark-recapture or removal techniques provide a reliable estimate of actual population abundances.

Seining: Seining with nets of different mesh size is usually conducted from the shore, with the net encircled around an area of the particular habitat and then closed and dragged onto the beach. Lead lines and floats keep the net close to the lake bottom and surface.

Fyke netting: Fyke nets of various mesh sizes are placed perpendicularly to the shoreline, with a collecting area at the most offshore end of the net consisting of netted hoops of varying size. The net acts as a barrier to fish moving along the shoreline, which then swim into the collecting area (Fig. 25.4).

Fish trapping: Baited fish traps such a Gee-minnow trap are commonly used to sample smaller benthic fish species, with small conical openings at their ends to guide fish in.

Gillnetting: Gillnetting, using nets of various mesh sizes, is an efficient way to sample a wide array of fish species, however its greatest drawback is that all specimens are usually killed in the sampling.

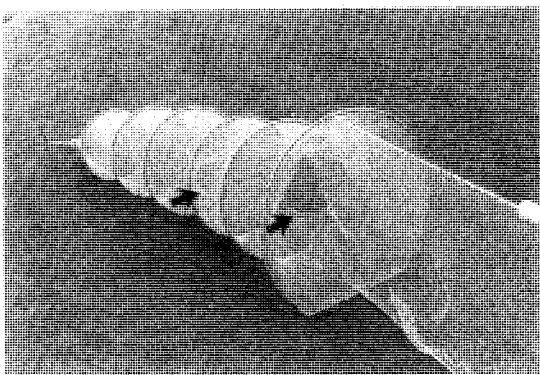


Photo: Robert McDowall



Photo: Rohan Wells

Figure 25.4 Methods for sampling littoral habitats. Left: a set-up fyke net showing the entrances for fish (black arrows). Right: a diver collecting from a bed of *Elodea canadensis* in Lake Wanaka, using a Wisconsin grab.

et al. 2000a). Oligochaetes and some chironomid taxa form the bulk of the species found within soft sediments. Quite a diverse assemblage of lumbriculid (*Lumbricus variegatus*), tubificid and naidid (*Limnodrilus* spp. and *Phreodrilus* spp.) oligochaetes are found in most fine sediments (Timms 1982). However, for this group, taxonomic difficulties often mean that individuals are identified only to their class, Oligochaeta (Forsyth 1978; James *et al.* 1998, 2000a; Weatherhead and James 2001). Similarly there are a number chironomid species that inhabit sediments, but owing to the labour of exact identification (i.e., examination of head capsule antennae and mouth parts) they are often identified only to family (Chironomidae) or subfamily (e.g., Tanytarsinae, Chironominae).

The greatest portion of the total abundance of littoral invertebrates often comprises chironomids and oligochaetes (Timms 1982; James *et al.* 1998). Two species that traditionally form a significant portion of mahinga kai (food) for Maori are the freshwater mussel *Hyridella menziesii* and freshwater crayfish (or koura). Both are quite commonly found in non-vegetated portions of the littoral zone, and also either amongst macrophytes or down in the littoriprofundal.

NATIVE FISH OF NEW ZEALAND LAKES

Among New Zealand's 35 native freshwater fish species, only 13 inhabit lakes (McDowall 2000). This relatively impoverished fish fauna is due largely to New Zealand's geographical isolation and to its long turbulent geological history, marked by marine submergence, massive mountain building, locally intensive volcanism and major

climatic changes. Because of New Zealand's geological history, most New Zealand lakes are young—many no more than 5000 years old (Lowe and Green 1987). As a consequence, there has not been time for the evolution of rich lacustrine fish faunas. There are few strictly lacustrine species, and the fish faunas in most lakes exhibit low species diversity—often only one or two species. The exception to this is in coastal, estuarine or brackish lakes such as Ellesmere and Onoke, which have somewhat richer fish faunas because of their direct connection with sea populations. By contrast, fish communities in temperate regions of North America (north of Mexico) have been estimated at approximately 212 species, composed of a multitude of feeding guilds (Lagler 1952).

Table 25.2 lists the fish species present in different types of New Zealand lakes. Only three fish species are confined to lakes—dune lakes galaxias in a series of coastal dune lakes in Northland, Tarndale bully in a series of small tarns in submontane Marlborough, and Chatham mudfish in two small peaty Chatham Island lakes. These strictly lacustrine species are so atypical of the fish faunas of New Zealand lakes as a whole that they do not merit a significant part of a discussion of the ecology of our lacustrine fish faunas. Other than these three species, virtually all lacustrine indigenous fish are lake-limited derivatives of otherwise diadromous (sea-migratory) species. For some of these species diadromy is obligatory (both eel species, and black flounder), and as a result they are found only in lakes and lagoons into which they can immigrate as small juveniles (apart from a few lakes that eels can reach by migrating across the land, usually on wet nights). Typically they inhabit lowland, often brackish/estuarine tidal lakes,

Table 25.2 Relative occurrence of native fish species in differing lake types in New Zealand. Occurrence is ranked as 0 = never, 1 = rarely, 2 = occasionally, 3 = often, 4 = usually, 5 = exclusively

Lake Type	Common bully	Tarndale bully	Upland bully	Common smelt	Giant kokopu	Banded kokopu	Koaro	Inanga	Dune lake galaxias	Chatham Is. mudfish	Shortfin eel	Longfin eel	Black flounder
Coastal lagoon	4	0	0	3	0	0	0	4	0	0	4	3	4
Coastal dune	4	0	0	3*	2	3	0	3	5	5	3	3	0
Lowland forested	4	0	0	1*	2	3	2	3	0	0	3	2	0
Glacial	3	0	0	1	0	0	4	0	0	0	1	2	0
Volcanic	4	0	0	3*	0	1	4	0	0	0	0	2	0
Submontane tarns	3	5	1	0	0	0	4	0	0	0	0	0	0
Hydro	4	0	0	0	1	0	3	0	0	0	2	2	0

* some by translocation

although longfin eels are very vigorous upstream migrants and populate sub-montane lakes well into the mountains.

Other lake fish species are highly facultative, and some of the lake populations are diadromous and others are land-locked, depending on whether access to and from the sea is available (especially populations of common smelt and common bullies). Low-elevation forested lakes and lagoons may have populations of banded and giant kokopu, but at higher elevations, common smelt, koaro, and common bullies prevail, and they are typically strictly lake-limited. Moreover, some of these populations (especially of common smelt) are translocated from diadromous stocks, especially in lakes of the central North Island, where smelt have been liberated as a forage fish for introduced trout (McDowall 1990b). Translocation may also have contributed to isolated populations of common bullies, which are not known to be good climbers, in higher-elevation lakes. Conversely, koaro have highly adapted pectoral fins that allow them to adhere to the surface of rocks and climb up near-vertical surfaces. Because of this, koaro have been able to populate many high-elevation lakes thought to be isolated from fish migration. One such example is Boulder Lake in the Marlborough ranges near Nelson—here a 60-m waterfall isolates the lake, yet koaro have been able to find their way into the lake.

Habitat use by native fishes

The ecology of lacustrine fish species in New Zealand varies considerably with life stage. Table 25.3 lists the habitats in lake ecosystems used by fish at various stages of their life history (see Table 17.1 and 17.2 for the scientific names of native and introduced fish species in New Zealand). With the exception of common smelt and dune lakes galaxias, the adults tend to be benthic or benthopelagic around the lake shores. Spawning is also often in the littoral zone, with common smelt depositing their eggs on sands in the shallows of the lake shore or around river mouths, and common bullies spawning in crude nests, usually beneath larger cobbles and boulders around the lake shore (though sometimes on upper bedrock surfaces and sometimes on macrophytes). The spawning of lacustrine koaro hasn't been described, but probably occurs mostly in streams flowing into lakes (where these are available—Allibone and Caskey 2000). On hatching, larvae in all species move to the limnetic zone of lakes, where larval and juvenile growth takes place. There are thus ontogenetic habitat shifts during the life of each individual fish (Rowe 1993, 1999; Rowe and Chisnall 1996; Rowe *et al.* 2001, 2002a). Rowe *et al.* (2002b) recently used a GIS-based approach to map habitat areas used by common smelt over their life cycle in Lake Taupo, with particular emphasis on the effects of water-level fluctuations on spawning habitat.

Lacustrine native fish are all carnivores—typically they

Table 25.3 Habitat use at different stages of the life histories of fish species inhabiting New Zealand lakes, and whether the species is capable of diadromy. P = pelagic, L = littoral, B = benthic profundal, T = tributary, ? = unknown, M = marine, and Y/N = yes/no.

Fish Species	Life history stage				
	Spawning	Larva	Juvenile	Growth to maturity	Diadromy
Common bully	L	P	P	L/B	Y
Upland bully	L	?	B	L/B	N
Tarndale bully	B	P	P	L/B	Y
Common smelt	L	P	P	P	Y
Koaro	T	P	P	L/B	Y
Inanga	L	P	P	P/L	Y
Dune lakes galaxias	?	P	P	P/L	N
Giant kokopu	?	P	P	L/B/P	Y
Banded kokopu	?	P	P	L/B/P	Y
Chatham Island mudfish	?	?	?	L/B	N
Shortfin eel	M	M	B	B	Y
Longfin eel	M	M	B	B	Y
Black flounder	M	M	M	B	Y

are plankton-pickers during the pelagic larval and juvenile stages, and become littoral benthic predators with maturity. As such, the diet of most fish species shifts from crustaceans as juveniles, to primarily insects and molluscs as adults. The exception to this is the common smelt, which generally remains a pelagic planktivore through much of its life, although it has been shown to utilize shoreline habitats at night in some lakes such as Lake Okaro (Forsyth and James 1988). Although it was originally thought that fish populations had little top-down effect on zooplankton populations in New Zealand lakes, largely because of the absence of obligate planktivores (Chapman *et al.* 1985), recent evidence suggests that top-down regulation by juveniles may be important (Jeppesen *et al.* 1997, 2000). Also striking is the absence of any native herbivorous fishes, with the exception of the New Zealand grayling, which is now extinct. There are few piscivorous native fish species, with the exception of shortfin and longfin eels (when large), although the introduction of salmonids and other coarse fish species in recent history has greatly increased the abundance of piscivores in most New Zealand lakes.

Impacts of introduced salmonids

Salmonid fishes were introduced to New Zealand during the late 1800s (McDowall 1990a, b) and now are present widely in our lakes. Few New Zealand lakes of any size with suitable habitat for trout currently do not have

populations of either rainbow trout, brown trout or both. Salmonids grow to a large size (often > 500mm) and are predatory; they have clearly had substantial impacts on the indigenous fish faunas of lakes. In the early years of trout introductions, the rapid growth of trout populations was driven by predation by trout largely on lacustrine koaro populations. These soon proved unable to withstand the levels of trout predation, and a serious collapse of koaro populations, followed by trout abundances, took place during the early 1900s. Fisheries managers responded by introducing common smelt into many lakes (McDowall 1990b)—unlike koaro, these have proved able to withstand trout predation and now fuel the growth and abundance of New Zealand's famous lake-based trout fisheries. The introduction of smelt into lakes may, however, not have been without effects on the residual stocks of koaro in lakes—the introduction of smelt into Lake Rotopounamu resulted in a rapid decline and extinction of the koaro populations there (Rowe 1993). If what happened at Lake Rotopounamu has general application, the koaro populations in other lakes may be being depressed by smelt releases into these lakes also. Most attention has been given to the effects of salmonids in lakes of the central North Island, but similar, perhaps less dramatic, and certainly less-recognized changes may also have taken place in other high-elevation lakes, especially glacial lakes along the Southern Alps of the South Island. There were once, for instance, enough koaro in Lake Wanaka to support a small “lake whitebait” fishery in the Matukituki River, which drains into the western shores of the lake. It seems unlikely now, however, that koaro are sufficiently abundant there to provide a fishery as it once did. Whitebait fishing in Lake Wanaka was prohibited many years ago—whitebait were considered more important for feeding trout than for feeding humans—an interesting choice of priorities. Koaro stocks in many South Island lakes seem increasingly sparse, though there are few data to document any changes.

COMMUNITY DYNAMICS AND INTERACTIONS

Some interesting work on littoral ecology in New Zealand lakes in recent years has been aimed at understanding the ecological interactions and foodweb relationships of communities and their importance to the ecology of lakes. Some of this work is addressed in other chapters in this book—Chapter 23 on lake food webs, Chapter 27 on aquatic invaders and

pests, and several of the chapters in Section 5. The following sections illustrate a few examples of interactions between littoral plants and animals, highlighting the interconnectedness of littoral ecosystems and their importance to overall lake ecology.

Macrophytes and fish foraging

Relatively little is known about the interactions between macrophytes (species and/or densities) and fish populations in littoral ecosystems. Studies in estuaries have demonstrated the importance of seagrass beds as a refuge from predation and potentially as foraging habitat for fish (Connolly 1994; Worthington *et al.* 1992; Spitzer *et al.* 2000), but these interactions have been examined for only a handful of freshwater fish species (Engle 1988; Gotceitas 1990). Dense stands of macrophytes can affect fish habitat and foraging in a number of ways (Fig. 25.5). For instance, macrophytes can promote fish predation through their effects on water clarity by attenuating nutrients and reducing wind-driven re-suspension of sediments in the shallow littoral areas (Schriver *et al.* 1995). Macrophytes also provide habitat for diverse assemblages of fish forage in the form of larval insects and molluscs. Conversely, dense plant stands may provide a refuge from predation for invertebrates and small fish (Uiblein *et al.* 1996, Kelly and Hawes in press).

For the New Zealand common bully, Duggan *et al.* (2002) observed that not only does the density of macrophytes affect their ability to forage on invertebrates, but the species of macrophyte was also important. Bully foraging was greater amongst the native charophyte *Nitella* with low stem densities than amongst the adventive hornwort (*Ceratophyllum*). However, dense stands of *Nitella* were far more effective refuge for prey at stem

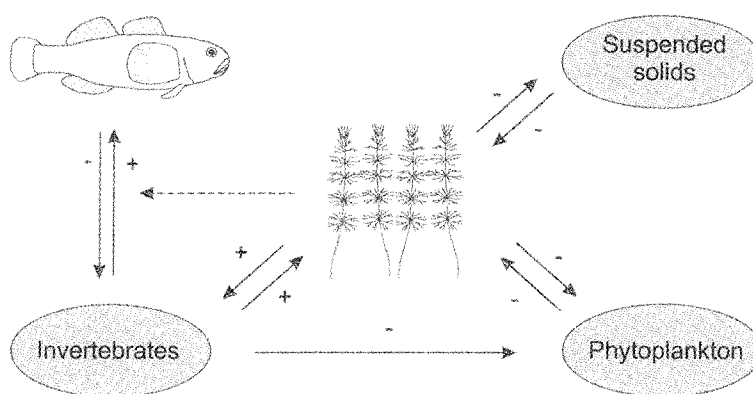


Figure 25.5 Relationships between macrophytes, invertebrates and fish in littoral zones of lakes. Arrows show positive or negative feedback relationships: for example, macrophytes can compete with phytoplankton for nutrients, while conversely phytoplankton can shade macrophytes. From Duggan *et al.* 2002

densities equivalent to that of hornwort. These findings were supported by field measurements of invertebrate densities in the two plant types, with far greater abundances of invertebrates in stands of *Nitella* than hornwort. The distribution of common bullies is positively correlated with littoral macrophytes, with much lower density populations through the unvegetated sublittoral or profundal areas in lakes (Rowe *et al.* 2001). This suggests that although bully foraging may be inhibited by denser stands of macrophytes, benefits to bullies in terms of refuge from predators such as eels and trout, as well as higher densities of prey organisms, may outweigh these costs (Kelly and Hawes in press). Complex three-dimensional habitats in macrophyte beds may be comparable with terrestrial forested ecosystems, and contribute to complex predator-prey dynamics between invertebrate and fish populations through the littoral zone.

Macrophyte grooming

The stability of macrophyte-dominated littoral habitats involves mutually dependent relationships between host plants and grazing invertebrates. While host plants provide a surface on which to live and feed, grazers in turn remove excess growths of epiphytic algae that would otherwise compete with macrophytes for nutrients and light (Kupferberg 1997; James *et al.* 2000b). Littoral communities in New Zealand lakes also appear to be adapted to episodic influxes of high suspended-sediment loads. Because of both large hydrological variations and, on occasion, tectonic activity in the catchment, New Zealand lakes can receive large pulses of sediment from inflowing tributaries. This can, in the short term, reduce light penetration to macrophyte communities, and also result in high rates of deposition of fine sediments on the surface of macrophytes. In deeper portions of the lake, where wave action is insufficient to remove these deposited sediments, plants could potentially be smothered. James *et al.* (2000b) observed that the grazing rates of the dominant grazer *Potamopyrgus* initially increased in response to sediment being incorporated into the epiphyton matrix growing on the macrophytes. This could be partially explained by the reduced nutritional content of the sediment-laden epiphyton, but nevertheless constitutes an interesting positive feedback mechanism by which host plants benefit from grazers.

Littoral biodiversity and species interactions

There is growing concern over the decline of once-abundant populations of kakahi (*Hyridella menziesii*), a freshwater mussel that occurs in the littoral zones of many lakes throughout New Zealand. This species was traditionally harvested as food in pre-European times, and declines in kakahi remain a concern for contemporary Maori. Declines in populations of kakahi in some lakes have been attributed largely to declining water quality and potentially to changes in the productivity and composition of phytoplankton, the main food source for these filter feeders (James 1985).

Less is known about the importance of early life history stages to kakahi population dynamics. Glochidium larvae, released from kakahi, parasitise fish, living on their gill lamellae or attached to the skin, before releasing as juveniles and moving to soft sediments of the littoral zone to filter feed until maturity (Fig. 25.6). It is known that adult koaro (*Galaxias brevipinnis*) are the main host for kakahi glochidia. However, since the introduction of brown and rainbow trout to most lakes in New Zealand, there is strong evidence that present-day populations of koaro are only a fragment of those in pre-salmonid times. For instance, Tuwharetoa historian J. te H. Grace (1959) described how Maori picked adult koaro off beaches of Lake Taupo for food following large storms. Other historical evidence suggests that koaro whitebait runs also used to be considerably greater in proximity to some large lakes such as Taupo. Not only have koaro populations been

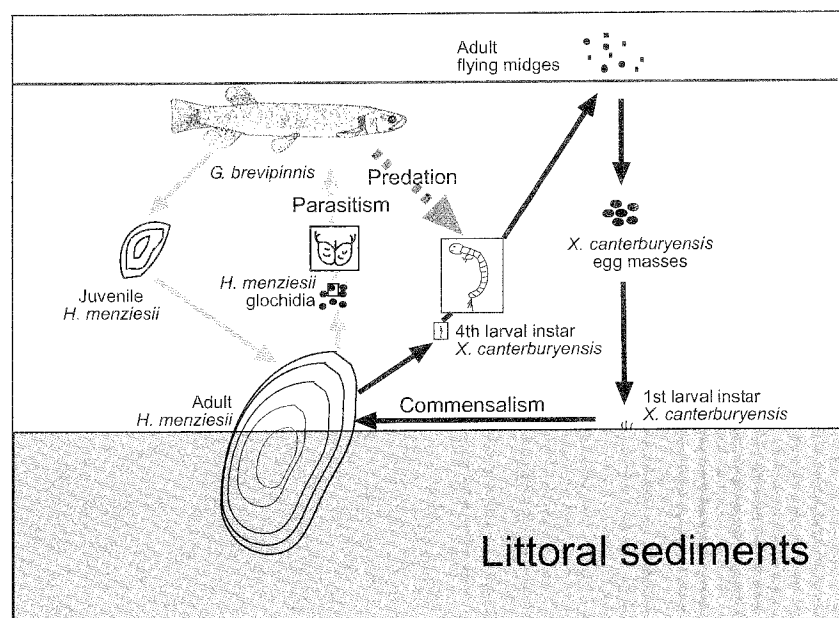


Figure 25.6 Interactions in the life cycle of kakahi (*Hyridella menziesii*), koaro (*Galaxias brevipinnis*) and chironomid larvae (*Xenochironomus canterburyensis*) in the littoral zone, demonstrating the importance of commensal, parasitic and predatory interactions between the three species for the completion of their life history.

affected by the introduction of salmonids, but another native competitor, the common smelt, has been introduced to some lakes. In Lake Taupo, smelt appear to have out-competed koaro populations, and also appear to better elude predation by salmonids than do koaro. Sadly there is little quantitative evidence for populations of koaro in lakes prior to the time salmonids were introduced (McDowall 1990a). There is only limited evidence that kakahi glochidium larvae are able to utilize other fish species, such as smelt or bullies, for this portion of their life history, adding to the uncertainty of the causes of kakahi declines in New Zealand lakes.

Yet another twist in this story is the interaction of kakahi with the early larval stages of the chironomid *Xenochironomus canterburyensis*, which commensally (mutually beneficial) lives inside the shells of adult kakahi. From lake sediments, young chironomid larvae (1st-2nd instar) enter the mantle cavity of kakahi, where they feed on phytoplankton filtered by kakahi until their final larval stage (4th instar). They then exit through the valve of the mussel to pupate and emerge as an adult midge at the lake's surface (Forsyth and McCallum 1978). Chironomid pupae also constitute an important food item for adult koaro, thereby benefitting koaro populations that are, in turn, essential to kakahi larvae. Thus it seems that the success of the three species is at least partially dependant upon the success of their interaction and, maybe more importantly, the demise of one is mutually linked to the other and indeed the biodiversity of the entire system.

CONCLUSIONS AND FUTURE DIRECTIONS

Research on the ecology of littoral zones has contributed a great deal to the understanding of the distributions of biota both within and between lakes, and important habitat characteristics contributing to the distributions of key species. This has provided a framework for managing lakes in New Zealand, defining important habitat characteristics and setting management goals for key target species. However, our understanding of relationships between plant and animal communities in lakes, and how these interact within the framework of ecology, biodiversity and ecosystem functioning is still in its infancy. Some of the first work to examine the dynamics of lake food webs and energy flow through the lake food chain has only recently been published. There is also still a great deal to be understood on the way in which top predators such as trout or eels structure benthic invertebrate, zooplankton and fish communities in lakes. This is of particular concern for New Zealand, where top-predator populations have been extensively manipulated through the introduction of non-native species. Indeed there is still very little

information on invertebrate and fish assemblages from lakes that have not been affected by the introduction of exotic species, so little is known of potential restoration targets should we wish to correct for these introductions in the future. Possibly of even greater significance is how quickly conditions are changing in New Zealand lakes, which, on an international level, are still relatively pristine. The introduction of exotic species, land-use intensification, and water allocation are now viewed by many as major threats to the ecology of lakes. A great deal is still to be understood by researchers, lake managers and industry to effectively protect these sensitive ecosystems.

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Chapter 24

Littoral algal and macrophyte communities

Mary de Winton and Anne-Maree Schwarz

INTRODUCTION

The littoral zone within lakes is defined as the shallow water region where bottom-dwelling plants are found. This region is important as it acts as an interface between land and open water, receiving, processing and modifying inputs of water from the catchment. Moreover, the littoral zone comprises a gradient in water depth that creates diverse physical conditions and hence biological variety.

The lake littoral supports “macrophytes”, referring to the larger plants that are easily discerned by the human eye. Attached and unattached algae (unicellular, colonial or filamentous) are also important components of the littoral zone. Collectively, these plants are important primary producers, from where organic matter originates that enters the lake food-web. Although they also occur in the littoral zone, phytoplankton (suspended unicellular and colonial algae) are the major primary producers in the pelagic or open water habitat (Chapter 22).

The state of knowledge on littoral plants in New Zealand lakes is founded on early descriptive accounts of their composition and structure. In the last 15 years research has moved to determining the major influences on the distribution, abundance and productivity of littoral plants. This chapter focuses primarily on these advances, together with fundamental background information.

LIFE-FORMS OF LITTORAL PLANTS

Littoral macrophytes

Littoral macrophytes are classified into “life-form” groups according to their morphology and habitat, although some species may be represented in more than one life-form group (Sculthorpe 1967). The predominant life-forms change with water depth and the accompanying environmental gradient, especially wave exposure and light. Thus in New Zealand lakes a distinctive zonation,

comprising both native and alien taxa, proceeds from emergent plants at the lake margin through to submerged plants beneath the water surface (Fig. 24.1), although not all life-forms may be represented within a lake littoral.

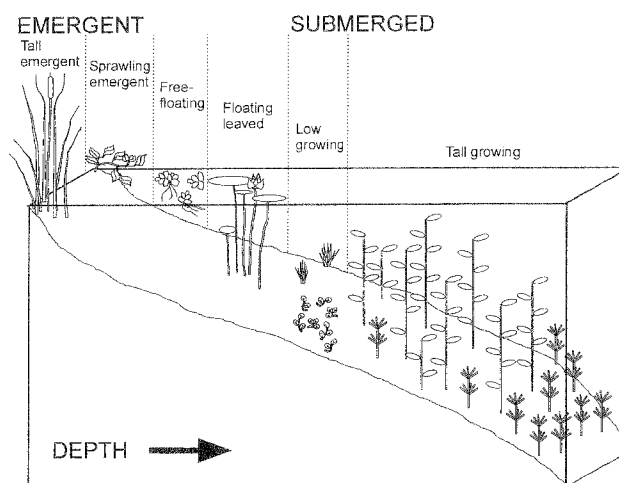


Figure 24.1 Life-forms of littoral macrophytes illustrating their zonation with water depth (plant size not to scale).

- *Tall emergent* plants (e.g. erect reeds and sedges) fringe lake margins to as much as 2 m depth. Tall emergents achieve their greatest development within shallow wetlands (Chapter 28) and will not be considered in detail in this chapter.
- *Sprawling emergent* plants form a horizontal, floating mat at the extreme margins of water bodies. In lakes, this life-form is usually restricted to one metre depth or less and apart from small sheltered lakes (e.g. de Winton and Champion 1993) is more conspicuous as a component of ditch or stream vegetation (Chapter 14) and some wetlands (Chapter 28).
- *Free-floating* plants (e.g. *Lemna*, *Azolla*) occur on the water surface of sheltered shorelines and can completely

cover the surface of small, wind-protected ponds. These small plants have various mechanisms (i.e. buoyancy, water repellence) to maintain their presence at the water surface, but some may sink temporarily in winter.

- *Floating leaved* plants (e.g. *Nymphaea*, *Potamogeton*) remain anchored at the lake-bed, but develop a large proportion of leaf area at the water surface.
- *Submerged low growing* plants refer to often multi-species assemblages of short stature (less than 0.2 m), forming mats or swards (e.g. *Glossostigma*, *Lilaeopsis*, *Elatine*). Elsewhere, these assemblages have been termed the “low mixed community” (Chapman *et al.* 1971), “amphibious sward” (Coffey and Clayton 1988a) or “short shallow water plant association” (Clayton *et al.* 1990). This group includes submerged plants and also amphibious species that may require a period of emersion to complete flowering and fruiting. In addition, quillwort (*Isoetes*) forms a distinctive submerged sward in many South Island lakes.
- *Submerged tall growing* plants comprise the most extensive life-form within the New Zealand lake littoral. This group is further divided into angiosperms (flowering plants), which are typically found at less than 10 m depth (*Elodea*, *Potamogeton*, *Myriophyllum*), and charophytes that extend to greater depths. Most native submerged angiosperms produce aerial, wind-pollinated flowers and so sexual reproduction is limited to depths from which they may attain the surface.

Species of littoral macrophytes can be identified with reference to Coffey and Clayton (1988a).

The special case of deep-water bryophytes

Deep-water bryophytes are a very interesting assemblage that occur in clear water lakes in a few regions of the world. They are relatively well represented in New Zealand. Sparse mosses and liverworts often grow on rocks in the splash zone around lakes and sometimes amongst the low-growing submerged plants. However, bryophytes are also found as a recognisable deep-water plant assemblage, occurring at 10 m or more depth within at least 15 New Zealand lakes and recorded down to 70 m depth (de Winton 1994). These plants form small tangled clumps on silt, rock or wood and grow as a carpet at some sites. They often include members from the genera *Bryum*, *Fissidens*, *Riccardia*, *Symphyogyna* and *Heteroscyphus*, but representatives of a number of other genera are found, including otherwise terrestrial species or plants of wet places. In some ways deep-water bryophytes can be considered “accidental” submerged plants following the introduction of their inoculum from shallow water or terrestrial sources.

Deep-water bryophytes occur in lakes that have exceptionally high water clarity and at sites where they are not subject to disturbance by water movement, siltation

and erosion, or large populations of foraging crayfish (Coffey and Clayton 1988b). The inference is that deep-water bryophytes must maintain a delicate balance between restricted growth under highly light limited conditions and minimal losses.

Littoral algae

In contrast to macrophytes, littoral algal assemblages are categorised firstly according to whether they are unattached or attached:

- *Unattached* algae can form mats over vegetation and substrates. Large mats, termed *metaphyton*, float at or near the water surface due to oxygen trapped within the mats. These may have originated from an attached assemblage, but become detached due to water turbulence.
- *Attached algae*, or *periphyton*, is a collective term for attached algae that grow or accumulate on surfaces in aquatic habitats as slimy films or mats (Chapter 15) and is often used to indicate other microbial components such as bacteria, fungi and microscopic animals as well. Periphyton can be distinguished further according to the type of substrate:
 - *Epiphytic* algae grow attached to the surfaces of macrophytes.
 - *Epipsammic* algae grow on sand.
 - *Epipellic* and *endopellic* algae are associated with mud substrates and may exhibit vertical migration in response to environmental cues such as light.
 - *Epilithic* algae grow attached to rock surfaces.

SPECIAL FEATURES OF LITTORAL PLANTS

Adaptive features of littoral plants

The numerous permanent lakes of New Zealand provide an important habitat for submerged macrophytes, which have special features that differ from terrestrial and wetland plants (Sculthorpe 1967). For example, submerged macrophytes do not require extensive strengthening tissues as internal turgor and buoyant tissues are sufficient to support them within the water column. Moreover, submerged macrophytes do not require the waxy, water impermeable surfaces of terrestrial plants because they need not protect against water loss, while they are bathed in a medium that contains dissolved nutrients and gases. Consequently submerged macrophytes may take up elements across much of their surfaces. Although roots generally remain the most important organ for nutrient uptake because of the higher concentrations present in the substrate, some submerged and free-floating macrophytes lack roots altogether.

Submerged macrophytes have special adaptations to meet the challenge of growing where light levels for photosynthesis are reduced by the overlying water and its

component dissolved and suspended materials (Sculthorpe 1967). They are often considered as “shade plants”, with photosynthetic processes tuned to efficient production at low light. The overlying water also influences the availability of metabolic gases, O_2 and CO_2 , which are required for respiration and as the source of carbon for the photosynthetic process respectively. In the terrestrial environment these gases are available in the atmosphere, but in water the slow rate of dissolved gas diffusion limits their availability. One partial solution for submerged macrophytes is to accumulate O_2 gas from photosynthetic production within tissue spaces during the day, from which it may then be used for respiration at night. Nevertheless, there are likely to be situations when respiration demand is not met. Likewise, the supply of dissolved inorganic carbon may limit plant photosynthesis in some situations. Some submerged plants have overcome this problem by their ability to utilize another form of carbon, bicarbonate ions, a source that has no terrestrial counterpart. Other submerged taxa (e.g. *Isoetes*) are CAM plants (Crassulacean acid metabolism) and can fix carbon from CO_2 released by respiration during the night for use in the photosynthetic process during the day (Keeley 1998). Previously, CAM was believed to be a plant adaptation to drought.

Non-vascular charophytes (Family Characeae) are macroalgae that superficially resemble vascular submerged plants but they do not possess highly differentiated tissues such as a vascular transport network, and instead of roots they possess multicellular rhizoids. Charophytes are able to occupy a wider range of depths within the littoral habitat than vascular submerged plants as a result of such differences. Although charophytes are commonly called “stoneworts” on account of lime encrustation, calcification is rare within the soft waters of New Zealand lakes.

Attached, colonial and filamentous algae are conspicuous within the littoral zone of lakes and other aquatic environments but have a limited presence in the terrestrial environment. A conspicuous feature of submerged habitats is the blanketing algal growths that develop on macrophytes in response to high levels of dissolved plant nutrients.

Emergent, floating and amphibious plants of the lake littoral zone do not differ as fundamentally from terrestrial plants as do submerged macrophytes (Sculthorpe 1967), but they do have important features that enable them to occupy the boundaries between permanent water, temporary water and land. Emergent plants can endure long periods of immersion through an ability to withstand toxic by-products of anaerobic metabolism. They also possess efficient means for supplying O_2 from the atmosphere to roots and other tissues within an anaerobic environment.

Characteristics of New Zealand littoral plants

There are a number of unusual aspects of the native littoral plant assemblages in New Zealand compared to elsewhere in the world. In New Zealand's evolutionary history, maximum native submerged macrophyte diversity and development was associated with lakes as opposed to rivers and streams, which were less suitable for submerged plant growth (see Chapter 14). Some types of macrophyte or growth strategies that are common in other countries are not represented. For example, there are no large native free-floating plants or water lilies amongst the indigenous lake flora. Another feature is the low rate of endemism of just 29% within native aquatic plants (Champion and Clayton 2000), compared to 85% of endemic species within New Zealand's overall native vascular flora.

Most non-endemic native lake plants are also found in Australia and it is likely that a natural pathway for propagule spread exists via wind distribution or migratory water birds (Champion and Clayton 2000). These avenues for sustained gene flow may also explain the low rate of endemism in littoral plants. Such natural pathways for dispersal would be quite species selective, and only a small subset of the aquatic flora of Australia is represented amongst the New Zealand native species. This contributes to a relatively species-poor lake flora, and perhaps one that is particularly vulnerable to invasion by alien introduced weeds (Chapter 27).

Alien macrophytes introduced to New Zealand lakes have had enormous success and there are now few examples of predominantly native littoral vegetation (Wells *et al.* 1998). This success of alien submerged weeds is made all the more remarkable by the fact that many of the most widely distributed weeds are dependent on human activities for their dispersal as they do not produce viable seed in this country (Howard-Williams *et al.* 1987). Their success is at the detriment of native plant communities, which are of lower stature, less competitive, easily replaced and excluded from their littoral habitat (Fig. 24.2). Interestingly, submerged vascular weeds can develop an exceptional biomass in New Zealand lakes, amongst the highest recorded for submerged plants in the world (Howard-Williams *et al.* 1987). The reasons for this abundance are not well understood but may include a lack of herbivores and less pronounced seasonal plant senescence.

Littoral macrophytes in New Zealand lakes are generally perennial, differing from many European and North American lakes where littoral plants often undergo a dramatic biomass decline due to senescence at the end of the summer growth season (Westlake *et al.* 1980). This lack of an annual plant strategy may be due to the mild climate, where minimum winter water temperatures seldom fall below $10^{\circ}C$, and permanence of waters within the littoral of most New Zealand lakes. However, some

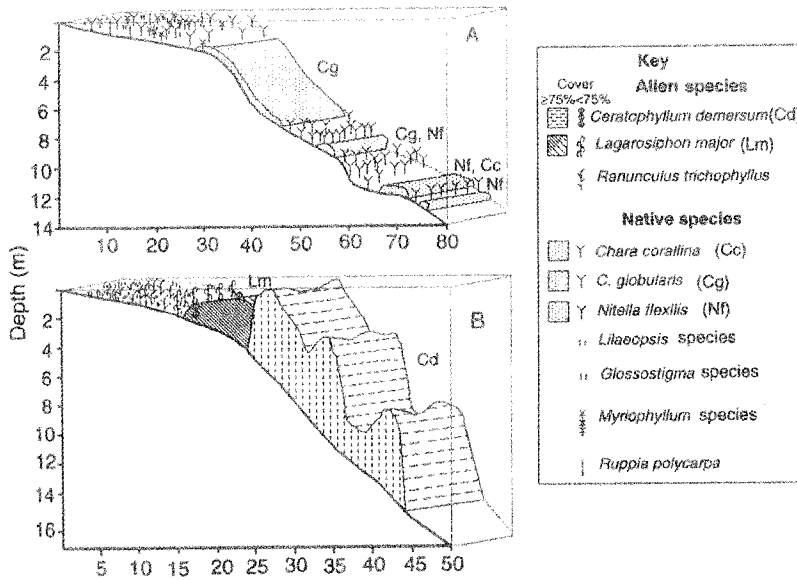


Figure 24.2 Stylised depth profile of A; a predominantly native littoral vegetation, and B; vegetation invaded by alien plants in Lake Tarawera (modified from Wells *et al.* 1997). Height of the shaded plant beds is to scale while the size of individual plant symbols is exaggerated.

shallow lakes show large fluctuations in macrophyte presence, alternating between macrophyte and phytoplankton dominance (e.g. Tomahawk Lagoon, McKinnon and Mitchell 1994).

FACTORS INFLUENCING DISTRIBUTION AND PRODUCTION

Productivity

The production of living biomass by macrophytes and algae in the littoral zone adds organic matter to lake ecosystems, some of which becomes available for consumption by other organisms. Fixation of carbon and conversion to biomass occurs through photosynthesis; the process by which light energy from the sun is converted to chemical energy to fix carbon into carbohydrates. Hence production is commonly expressed as grams of carbon or biomass per unit area per unit time (e.g. g C/m²/yr). Respiration is the means by which the energy of stored foods (e.g. carbohydrates) is made available for the requirements of the plant cell. Photosynthesis and respiration are the basis for all plant growth.

Photosynthesis of submerged plants is dependent on the availability of light and dissolved inorganic carbon through the aquatic medium. The relative availability of light and carbon as well as the quantity and type of other inorganic nutrients determines growth potential, hence a paucity of either light or nutrients can result in growth limitation. For example in a turbid lake, reduced availability of light can limit photosynthesis regardless of the quantity of

nutrients available, while in a clear oligotrophic lake where light is not limiting, nutrients such as nitrogen or phosphorus may be in short supply (Ratray *et al.* 1991a). Limitation of photosynthesis by the availability of CO₂ may occur over short time scales during long periods of photosynthesis within dense weed beds and has been shown to occur at times in dense weed beds in New Zealand lakes (Ratray *et al.* 1991b, Schwarz and Howard-Williams 1993).

The biomass of plant material at a given place and time is the net result of growth and loss processes (Fig. 24.3) over ecologically relevant time-scales. Growth results from the balance between photosynthesis and respiration. Net productivity includes the biomass, or carbon, that results from plant growth and that which may be lost to grazing, senescence and physical disturbance from wave action, activities of biota, erosion or burial. None of these

processes are constant over time, however the strong link between photosynthesis and seasonal variations in solar irradiance and temperature mean that patterns in growth can conveniently be considered over annual time-scales. The maximum water depth at which submerged macrophytes can still carry out photosynthesis is restricted by the seasonal availability of light as well as the

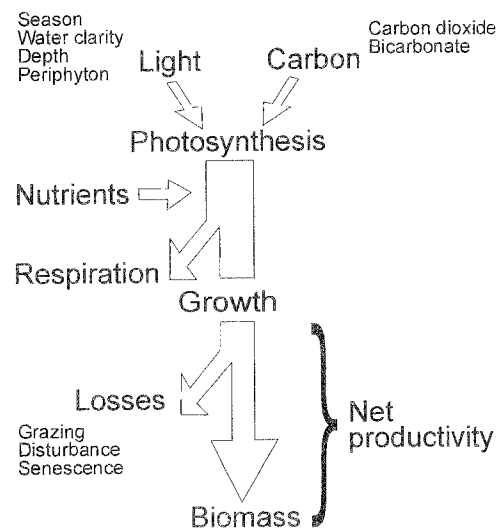


Figure 24.3 Primary components influencing submerged macrophyte productivity and ultimately biomass in the lake littoral. Net productivity is the rate at which biomass is created and is the result of growth and loss processes.

penetration of light through the water column. The latter differs from lake to lake according to water clarity (Chapter 21). The maximum depth to which macrophytes can grow and persist therefore occurs at a resource boundary, for example where there is insufficient light to enable photosynthetic gains to balance loss processes. This means that boundaries can be useful indicators of how submerged plants integrate growth conditions over time. In Lake Waikaremoana for example, while the maximum depth limit for charophytes has been shown to be stable over a period of at least 5 years, during that period no net photosynthesis was recorded over 5 consecutive winter months, suggesting that all growth occurred during the summer months (Howard-Williams *et al.* 1995).

Not only are submerged macrophytes in New Zealand lakes perennial, but they also show apparently weak seasonal patterns in biomass values. A winter minima for biomass of the charophyte *Chara globularis* was shown for Lake Alexandrina (Ward *et al.* 1987) although in that instance the reduction was attributed to wave action rather than a light or temperature effect. In Lakes Taupo and Rotorua there was only weak evidence for seasonal changes in macrophyte biomass over the course of a year with the highest values measured at the end of summer (Schwarz and Howard-Williams 1993), nevertheless the relative growth rate of weed beds dominated by *Ceratophyllum demersum* and *Lagarosiphon major* have been shown to be reduced by as much as 50% during winter (Howard-Williams *et al.* 1996).

The biomass of small algal cells such as phytoplankton and free-living algae on sediments, that have rapid growth and loss cycles in relation to environmental conditions, often show a correlation with the nutrient status of the water assuming other environmental conditions are suitable. The biomass of macrophytes is not correlated with nutrient status to the same degree. Assuming there are sufficient nutrients to enable macrophyte growth to occur then biomass is able to accumulate slowly over time,

relatively independently of nutrient status, by virtue of the stable attached nature of these communities (i.e. if growth ceases for a period, this does not necessarily equate to a loss of biomass) (Westlake *et al.* 1980). For example alien macrophytes such as *L. major* beds can attain high biomass, exceeding 1178 g dry weight (DW)/m² even in oligotrophic New Zealand waters (Howard-Williams and Davies 1988, Howard-Williams *et al.* 1996). Macrophyte biomass values recorded for some New Zealand macrophytes are summarised in Table 24.1.

Unrelated to seasonal patterns in temperature and light availability, fluctuating macrophyte biomass over time has been recorded in New Zealand lakes, in relation to other environmental changes. The effects of changes in water clarity on submerged macrophyte photosynthesis and growth is an aspect of research that has received attention in New Zealand research over the last decade. Annual net carbon balances at the maximum depth for plant growth have now been shown through modelling for charophytes (Howard-Williams *et al.* 1995, Schwarz *et al.* 1996, Sorrell *et al.* 2001) and for vascular plants to be close to zero in years without marked interruption by periods of low water clarity. This illustrates that in the absence of significant grazing, the lower depth boundary can reflect a balance of photosynthetic gains and respiratory losses. However periods of weeks to months of low water clarity due to high suspended sediment concentrations can shift this balance toward a net annual loss. For example in Lake Wanaka a six-week period of very low irradiance during spring reduced productivity sufficiently such that vascular species all had negative annual carbon balances at their depth limits for the year of the event.

Even when littoral plant productivity is suppressed for periods of time, studies have shown that charophytes have a degree of resistance (days to weeks, Fig 24.4) (Howard-Williams *et al.* 1995), which can be explained in part by an ability to acclimate to the lower light environment and to utilise stored carbohydrates (Schwarz *et al.* 1999).

Table 24.1 Examples of maximum biomass values reported for some New Zealand macrophytes.

Plant	Measurement	Biomass (g dry weight/m ²)	References
Charophytes	Range of reported maximum values	50-350	Ward <i>et al.</i> 1987, Howard-Williams <i>et al.</i> 1995, Schwarz and Hawes 1997
<i>Lagarosiphon major</i>	Range of reported maximum values	>1178-2144	Howard-Williams and Davies 1988, Howard-Williams <i>et al.</i> 1996, Wells <i>et al.</i> 1997
<i>Ceratophyllum demersum</i>	Maximum	1489	Wells <i>et al.</i> 1997
<i>Elodea canadensis</i>	Maximum	2167	Wells <i>et al.</i> 1997

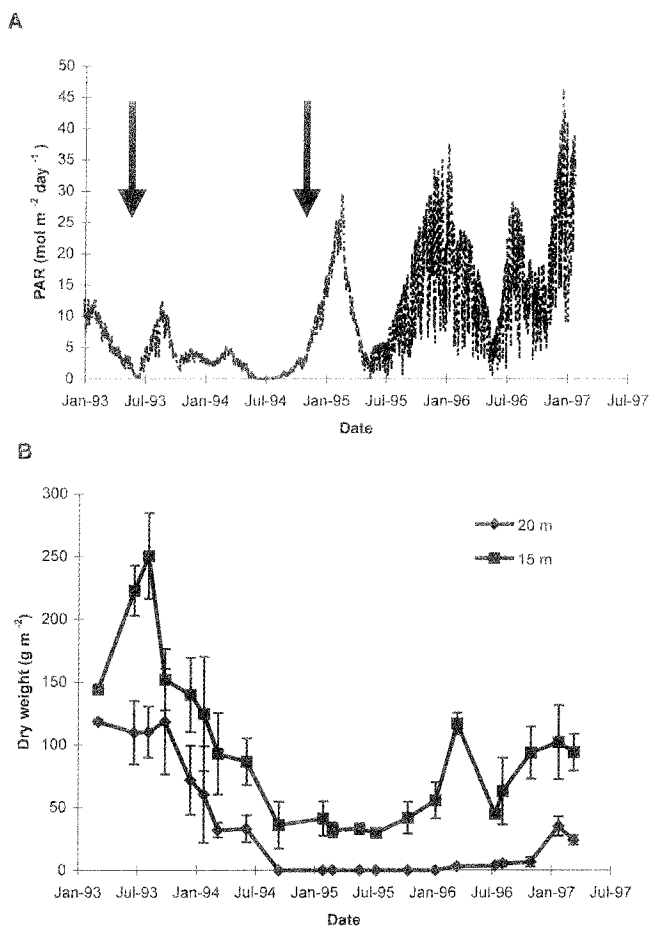


Figure 24.4 A) Seasonal patterns in Photosynthetically Available Radiation (mol (photons) $\text{PAR m}^2/\text{d}$) at a depth of 20 m over a four year period in Lake Coleridge, during which water clarity was very low for a period of almost two years due to turbid inflows to the lake (arrows mark the start and end of the turbid event). B) biomass (g/m^2 dry weight) of charophytes at depths of 15 and 20 m over the same period. Resistance is illustrated by the lag time before a significant reduction in biomass occurred. Despite an improvement in water clarity in 1995 (illustrated by an increase in PAR at depth) recovery of biomass took at least 18 months and up to 2 years at 20 m (modified from Schwarz *et al.* 1999).

Submerged plants in general are adapted to photosynthesise under lower light conditions than their terrestrial counterparts and as “shade” plants show a variety of modifications to physiological, anatomical and biochemical features to suit the available light environment. However if resistance is exceeded complete collapse of macrophyte growth can result (Vant 1987) and then the ability to recover can be small depending on restoration of suitable environmental conditions (Tanner *et al.* 1993), extent of disruption by biota (de Winton *et al.* 2002) and the proximity of suitable vegetative

material (Schwarz and Hawes 1997) or seed banks (de Winton *et al.* 2000).

Water clarity is not the only factor that can affect the amount of light available for photosynthesis by submerged macrophytes. Epiphytic periphyton have been shown to have the potential to markedly reduce the amount of light available to submerged plants (James *et al.* 2000a). A direct relationship has been reported between an increase of periphyton on the leaves of macrophytes and their gradual senescence (Howard-Williams *et al.* 1978).

Assessing littoral plant productivity

Assessing productivity of submerged macrophytes and algae requires measurements of growth and loss processes. An absence of marked seasonal patterns in biomass means that unless there are extreme environmental perturbations, growth and loss processes must be approximately equal. Growth rates of macrophytes can be measured using techniques of marking individual plant stems. If stem density is known, relative growth rates can then be calculated as g/g/unit time (Ward *et al.* 1987). Loss processes such as grazing and mortality are more difficult to quantify and consequently quantitative measurements of biomass as organic (ash-free) dry weight, are commonly used to approximate net production over time (Westlake *et al.* 1980). However, net changes in biomass do not necessarily equate to net photosynthetic production and so do not yield information about the process or mechanism behind changes, i.e. how environmental changes affect photosynthetic gains. Although the same principles of growth and loss apply to algal productivity, measurements of biomass and growth of periphyton have added complications due to their often close association with macrophytes. A useful technique for measuring accumulation of epiphytic biomass is to use artificial plants as colonisation substrates, which can then be destructively sampled over time (e.g. Hawes and Schwarz 1996).

To elucidate the processes determining growth rates it is necessary to understand how photosynthesis and respiration of different plant parts respond to environmental changes over time. There are a number of techniques available to measure such physiological processes in submerged macrophytes that, with the exception of PAM fluorometry, involve measuring changes in carbon uptake and/or oxygen exchange. Some methods for measuring CO_2 and O_2 in an aquatic medium are summarised in Table 24.2. These techniques require the plant of interest to be enclosed in some type of incubation chamber, with options for chamber design and the provision of required light sources being many and varied. While there are some commercially available designs, they are often purpose built by individual researchers.

Table 24.2 Methods used to estimate photosynthetic production of submerged plants, the type of data produced and example of common units, and restrictions to use. Standard methods are described in handbooks such as Strickland and Parsons (1972) or as referenced below for New Zealand studies.

Method	Type	Data obtained	Restrictions
Infra red gas analysis analysis (Rattray <i>et al.</i> 1991b).	CO ₂ in gas/water equilibrium.	CO ₂ uptake (µmol CO ₂ /g DW/h)	Requires enclosure of photosynthetic tissue in an incubation chamber. Assumptions made about the gas/water equilibrium of the chamber.
Carbon isotopes.	Uptake of radioactively labelled inorganic carbon tracer.	¹⁴ C uptake in counts per minute (cpm/mg DW/h)	Requires enclosure of photosynthetic tissue in an incubation chamber. Requires laboratory facilities, incubators and scintillation counter. No estimate of respiration.
Oxygen measurement techniques: oxygen electrodes. Winkler method or coulometry (Schwarz <i>et al.</i> 1996).	Measures photosynthesis in the light and respiration in the dark from O ₂ concentration in the water.	Changes in O ₂ concentration in the incubation medium over time under different conditions (µmol O ₂ /g DW/hr)	Requires enclosure of photosynthetic tissue in an incubation chamber.
PAM fluorometry (Hawes <i>et al.</i> 2003a).	Non-destructive, in situ method, measuring chlorophyll fluorescence of Photosystem II.	Quantum yield or electron transport rate (µmol electrons/cm ² /hr)	No estimate of respiration. Quantitative estimates of production require correlation with gas exchange techniques.

Assessing littoral plant presence and distribution

Several methods are available to describe the presence and distribution of littoral plants in lakes (Table 24.3), which provide data for various research or management purposes. The most suitable method depends on the scale and quality of information sought and time and resources available.

Some methods have been developed or refined following relatively recent advances in technology or increased availability of equipment. For example, depth sensing dive computers have improved the accuracy of plant depth data, while hydro-acoustic/GPS technology now has the flexibility for scientific application. Developments in GIS computer programmes have also opened up possibilities for mapping the extent of littoral vegetation within lakes.

Distribution and productivity of plant life-forms

Tall emergent and sprawling emergent plants

Emergent vegetation is situated at the extreme margin of lakes and so is very susceptible to the action of waves. This means that in large lakes emergent plants are excluded from exposed sites where wave action not only directly disturbs plants, but also removes the fine sediment that is

suitable as a substrate (de Winton *et al.* 1993). Inshore stands of emergent plants can also be removed from the lake littoral by intense cattle grazing but a fringe of tall emergent plants may persist in deeper water that stock cannot access (Tanner 1992).

Free-floating and floating leaved plants

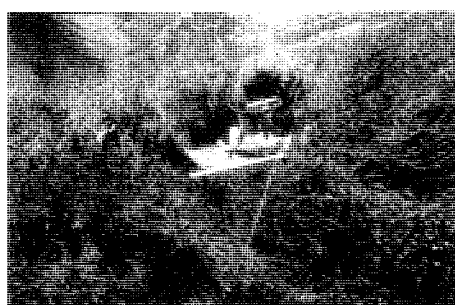
Both free-floating and floating leaved plants are generally restricted to areas that are protected from the wind and hence plants like *Azolla* are most common in small ponds, lakes or swampy areas. Because these growth forms all have their leaves at the water surface photosynthesis is not limited by water clarity and when their roots are bathed or rooted in nutrient rich waters or sediment they are capable of very high production rates. Floating leaved plants are generally restricted to depths from which their leaves can attain the water surface, normally less than 2 m.

Submerged low growing plants

The suitability of a lakeshore habitat for submerged low growing plants depends on the physical action of waves and water level fluctuations to a greater degree than their deeper growing counterparts. Where long fetch and

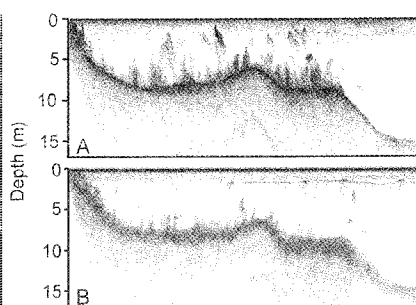
Table 24.3 Common methods used to measure/assess littoral vegetation presence, structure and distribution in New Zealand lakes.

Method	Type	Data obtained	Restrictions
Biomass sampling (Howard-Williams <i>et al.</i> 1996).	Quantitative SCUBA method.	Species biomass composition and depth distribution (m).	Time consuming. Problems sampling tall weed beds. Clarity required for observations.
Line transect (Clayton 1983).	Quantitative SCUBA method.	Species cover (%), height and depth distribution (m).	Time consuming. Training required for scientific method.
Profile "quick survey" (Clayton 1983).	Qualitative SCUBA method.	Species depth (m), cover (%), average, maximum) and height (m, average, maximum).	Training required for scientific method.
LakeSPI (Clayton <i>et al.</i> 2002).	Qualitative SCUBA method.	Extent of native and alien invasion, lake ecological condition indices.	Training required for scientific method.
Hydro-acoustic technology with/without GIS spatial mapping.	Remote sensing.	Vegetation presence (mapping), height and depth range (m).	Ground truthing required for interpretation.
Aerial photographs (or satellite image) improved with infrared imagery.	Remote sensing.	Vegetation presence (mapping).	Vegetation visibility (water clarity, depth). Ground truthing required for interpretation.
Dredging/underwater viewer (Clayton 1983).	Remote sampling.	Vegetation presence, depth range? and species composition?	Vegetation visibility (water clarity, depth). Inaccuracies due to incomplete sampling.

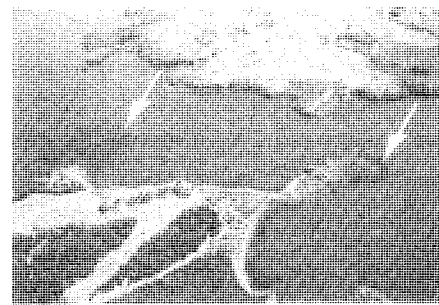


1.

Photo: John Clayton



2.



3.

Photo: Rohan Wells

Figure 24.5 1. Diver carrying out the line transect method of assessing lake littoral vegetation. 2. Sonar profile of changes in an alien weed bed in Lake Wiritoa A; before herbicide treatment and B; 42 days after treatment. 3. Aerial photograph showing the distribution of large weed beds (white arrows) around the western basin of Lake Rotoiti in 1989.

prevailing winds co-occur, wind induced wave exposure can be a key factor influencing the abundance, species richness and depth distribution of these submerged macrophytes (e.g. Spence 1976) and so the shallow littoral can be a habitat where physical processes play an important part in structuring aquatic plant communities. Low growing plants also occur in water depths that are most likely to be subject to periodic dessication and inundat-

ation, or even winter freezing, through water level fluctuations.

On New Zealand lake shorelines subject to periodic physical disturbance, one or more of the taxa *Glossostigma* spp., *Myriophyllum propinquum*, *Lilaeopsis ruthiana* or *Elatine* sp. have been shown to be primary colonists following disturbance (Chapman *et al.* 1971). These taxa form small patches, which then facilitate the accumulation

of sediment thereby enabling secondary colonists to establish. Thus exposure alone does not explain species richness and composition of the low growing community (Riis and Hawes 2003) and these characteristics of the community depend on the successional stage since the last major disturbance. Exposure and the resultant physical disturbance of sediment movement is however a primary determinant of the location of upper depth limit of these low growing plants while the lower limit is likely to be controlled by competition with members of the submerged tall growing community.

Isoetes spp. tend to form largely monospecific beds that extend to greater depths than the other low growing species. The upper depth limit of these swards has been found to be related to wave exposure, while water clarity influenced the maximum depth of colonisation (Hawes *et al.* 2003).

Submerged tall growing vascular plants

The tall growing vascular plants occupy the intermediate depth zones in lakes meaning that they are not affected by wave action or water level fluctuation to the same degree as shallow vegetation. They cannot grow as deeply as charophytes and deep-water bryophytes, generally occurring at less than 10 m. While the effects of increased water pressure at depth have been invoked as a controlling mechanism for some species (Coffey and Chu 1988) there is little experimental evidence for this to date (Sorrell and Dromgoole 1996) and the restriction to shallower depths appears to be primarily due to higher light requirements (Kalff 2002). These higher light requirements can be understood from the balance between gains and losses in production. Vascular species have a greater investment in complex rhizomes and roots than the deeper growing charophytes have in simple rhizoids, hence, while photosynthetic performance of leaf tissue may be equally as shade adapted as the charophytes in a given habitat, photosynthetic requirements on a whole-plant basis may be greater when the carbon investment in rhizomes and roots are accounted for (Zimmerman *et al.* 1994). Measurements of the carbon balance of three New Zealand native vascular taxa, *Isoetes*, *Potamogeton cheesemanii* and *Myriophyllum triphyllum*, provide evidence that the maximum depths to which they can grow can be explained by light attenuation. Respiratory requirements of the below-ground tissues restricted the maximum depths for these species to shallower depths than would be predicted from the balance between photosynthesis and respiration of leaf tissue alone (Sorrell *et al.* 2003).

Charophytes and deep-water bryophytes

Usually well below the influence of wave exposure and desiccation, light is the primary control on the maximum depth attained by deep charophytes in New Zealand's lakes. Of 63 New Zealand lakes, 71% had charophytes at

the maximum depth of growth (excluding those with deep-water bryophytes) (Schwarz *et al.* 2000) with maximum depths up to 34 m making these some of the deepest records in the world. Of these, the availability of light was the main factor explaining the depth to which charophytes grew in South Island lakes, but in some North Island lakes additional factors appeared to be controlling the maximum depth, which is sometimes shallower than predicted on the basis of light availability (Vant *et al.* 1986; Schwarz *et al.* 2000). This apparent anomaly, along with the absence of deep-water bryophytes in the North Island Lake Taupo compared to the South Island Lake Wanaka despite their similar water clarity, has been attributed to browsing and mechanical damage by crayfish (*Parenehraps planifrons*) (Coffey and Clayton 1988b). Increased grazing or disturbance effectively increases the light requirement of submerged plants as these are loss processes which can only be offset by increased production.

Charophyte species often show bands of zonation with depth in the lake littoral, with the species sequence shown to be related to the amount of sub-surface light (Schwarz *et al.* 2002). Individual species differ in their carbon acquisition characteristics and these reflect their different depth distributions, with deeper species having more efficient photosynthetic metabolism (Sorrell *et al.* 2001).

Little is yet known about the eco-physiology of the deep-water bryophyte community of New Zealand's clear lakes although the expectation is that, similar to charophytes, low-light acclimation and exclusion from grazing pressure are primary factors in enabling their persistence where they occur.

Habitat models and predicting submerged plant distribution

Recent research into lake management has moved toward the establishment of predictive models to enable managers to maintain biological diversity. While charophyte depth limits can be predicted with some degree of accuracy on the basis of the light environment, inclusion of physical forcing factors in shallow water requires consideration of a wider range of environmental variables.

Recently an optimum habitat was described for a diverse low growing plant community in relation to water level fluctuations by Riis and Hawes (2002). A decrease in community cover, species richness (Fig 24.6) and depth range was shown to occur as physical disturbance increased. For 21 New Zealand lakes the species richness of the low growing plant community was found to be much lower in lakes with inter-annual variations in water level than those with intra-annual variations.

By combining water level fluctuations with physical disturbance due to wave action, Riis and Hawes (2003) proposed generalised optimum habitat requirements for the low mixed community on a lakeshore. The definition

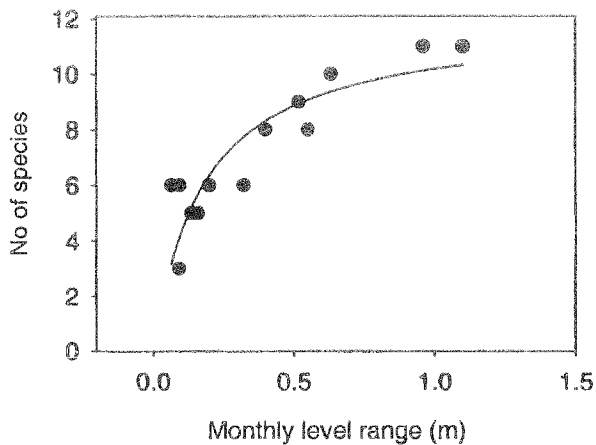


Figure 24.6 Relationship between water level range (mean monthly range) and the number of species of low growing plants per lake (from Riis and Hawes 2002).

was a lake with 1 m monthly water level range and on a shore where mean duration of low level events is less than a month, where maximum exposure the previous year did not exceed 8 m depth of sediment motion or 0.3 m wave run-up, and the shore slope is $<0.1\text{m/m}$. Although generalised for New Zealand lakes, to date there has been insufficient research in a wider range of lakes to accurately assess the relative importance of different physical factors for a given location within a specific lake.

To expand habitat requirements to other aquatic plant life-forms, Hawes *et al.* (2003b) developed statistical relationships between key physical and vegetation variables to define “habitat templates” of key vegetation types. Lake-specific variables were found to provide a good estimate of expected lake vegetation for low growing plants, *Isoetes*, two native, tall growing species and charophytes, however as for the low growing community described above, site specific predictions are still some way off as they require higher resolution models and input of more site-specific data.

Littoral algae

Periphyton consists mostly of “primary producers” at the base of the food chain and can be a major contributor to production in the littoral zones of oligotrophic lakes such as Taupo. In fact, while submerged macrophytes usually dominate the primary production biomass of the littoral zone of oligotrophic lakes, for Lake Taupo it has been estimated that periphyton probably contribute closer to 90% plant production in the littoral zone (Hawes and Smith 1993).

Cliffs are significant habitats for epilithic communities in lakes and where water clarity is high, e.g. in Lake Taupo, periphyton can extend down to below 30 m. The lack of grazers on these surfaces and the unpalatability of blue green algae has been put forward as an explanation for the

high biomass found on cliffs in Lake Taupo (Hawes and Smith 1993). Periphyton also grow on boulders/cobbles within the wave zone, on boulders/cobbles amongst shallow water plants and on the plants themselves and these shallow water algae have been shown to recover quickly after periods of low water levels.

Epiphytic periphyton can comprise a range of algal groups including diatoms, blue-green and green algae that are tightly adhered to the surface of submerged plants as well as filamentous algae that can form conspicuous growths underwater, often overgrowing submerged vascular plants. Examples of the latter are the native taxa *Cladophora glomerata* and *Enteromorpha nana* as well as the alien filamentous green alga, water net (*Hydrodictyon reticulatum*), which bloomed and subsequently declined during the late 1980’s to early 1990’s (Wells *et al.* 1999). *Enteromorpha* can produce unsightly and smelly proliferations at places in Lake Taupo, for example where growths seem to be related to localised nutrient (nitrogen) inputs (Hawes and Smith 1993).

As for submerged macrophytes, the composition and size of a periphyton community varies widely with substrate type, light availability, nutrient supply, grazing by small animals and physical disturbance such as wetting/drying at the water’s edge due to lake level variations, wave-induced sloughing and/or substrate disturbance (see also Chapter 15).

Atypical littoral environments

Littoral plant communities are absent or modified within lakes that are highly dystrophic (e.g. peat or beech catchments), brackish (e.g. coastal lagoons), at extreme altitude, of low alkalinity or highly geothermal, and tarns that are ephemeral or that have extensive water level fluctuations. In these special cases, the factors outlined in the sections above can be over-ridden by other influences on plant distribution and productivity.

LITTORAL PLANT INTERACTIONS

Environmental modification

The presence of plants modifies physical and chemical conditions within the lake littoral (Fig. 24.7), with the extent of environmental modification strongly related to plant cover or biomass, architecture, productivity and also to physiology (Carpenter and Lodge 1986). Plant beds can physically dampen wave action and reduce water movement. Re-suspension of lake sediment is subsequently reduced, but in addition plants promote the settlement of suspended particles (Casanova *et al.* 2003). In tall growing plant beds, light is rapidly extinguished with depth through shading by the plants themselves. Variations through the day in plant photosynthetic and respiratory activity can create sharp fluctuations in O_2 concentrations and pH, which are accentuated due to the lack of water

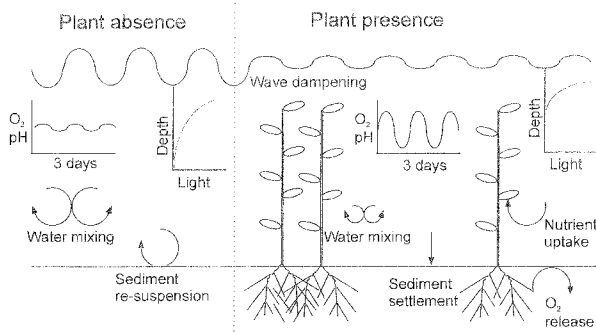


Figure 24.7 Illustration of some major modifying influences of dense macrophytes in the lake littoral environment, where water movement is reduced, daily fluctuations in O_2 and pH increase, light extinction with depth increases, suspended particles decrease, nutrients are sequestered by plants and root O_2 is released to sediments.

movement and replenishment within dense beds. Further environmental modification can occur for algae associated with submerged plant beds. Submerged macrophytes are able to take up nutrients from the water or sediments in excess of their immediate requirements (luxury uptake), thereby sequestering nutrients that would otherwise be available for growth by phytoplankton or periphyton. Vascular plants are capable of O_2 transport to roots and release to the immediate root environment. This can create an oxidised zone, which in turn influences nutrient availability and decay processes.

Plant inter-relationships

Littoral plants compete for available resources, with competition for space, light, carbon and nutrients occurring both intra- and inter-specifically. For instance, many alien submerged plants are able to extend above the sediment to greater heights within the water column than native plants and thereby intercept more light, shading out potential competitors. Macrophytes may limit or prevent photosynthesis of epiphytic algae by modifying conditions of pH (and hence CO_2 availability) and nutrient availability as outlined above.

The corollary of macrophyte competition with epiphytes is that macrophytes provide a renewable substrate of high surface area for the growth of epiphytes (Loeb and Reuter 1981), while excessive quantities of littoral algae can have significantly detrimental effects on submerged macrophytes. This was experienced in New Zealand during the invasion and subsequent decline of water net in the late 1980's early 1990's. In some places smothering by accumulations of this alga caused macrophyte beds to collapse (Wells and Clayton 2001).

Symbiotic relationships are known for the free-floating plant *Azolla*, which harbours the nitrogen-fixing blue-green algae, *Anabaena*, within its fronds (Coffey and

Clayton 1988). Some mutual benefit is also suggested by high infection by root mycorrhizal fungi of plants in New Zealand lakes with evidence that the fungal extension of absorptive root area increased the nutrient status of plants growing in low nutrient substrates (Tanner and Clayton 1985).

Release of chemical substances by littoral plants (allelopathy) to control other taxa is suggested in laboratory experiments, however the complex protocols to prove allelopathy in the field means there is little evidence for these interactions occurring within lakes (Gopal and Goel 1993).

Plant and animal inter-relationships

Littoral plants create a three-dimensional habitat that is essential to many faunal taxa for all or part of their life-cycles. For example, large beds of water net provided an extensive habitat for numerous grazing water snails and a refuge for large numbers of zooplankton in Lake Aniwhenua (Wells *et al.* 1999, Wells and Clayton 2001). Conversely, dense plant beds can exclude fauna such as fish or mussels by creating low O_2 conditions or by presenting a physical barrier to movement.

Epiphytes can be an important food source for grazing invertebrates (Chapter 25). In at least one large, oligotrophic, New Zealand lake where they were not grazed directly macrophytes did not make a major contribution directly to carbon flow to higher trophic levels, even through detrital pathways when decayed (James *et al.* 2000b). Instead it was the epiphytic algae on these macrophytes that were shown to underpin much of the secondary production (James *et al.* 2000b).

Apart from providing an important food source for fish, grazing invertebrates can play another important role in the maintenance of littoral submerged macrophyte communities. Grazers such as the gastropod *Potamopyrgus antipodarum* continuously groom the host plants of periphytic algae and settled sediments (James *et al.* 2000a) thereby providing the plants with better access to light for photosynthesis. Indeed the stability of macrophyte-dominated littoral communities is often viewed as dependent on mutually beneficial interactions between macrophytes, periphyton and grazers (Carpenter and Lodge 1986). Grazers benefit host plants by removing epiphytes that would otherwise compete with them for nutrients and light. Macrophytes in turn benefit epiphytic grazers by providing a refuge and a high surface area for periphyton growth, while epiphytes benefit from the high surface area and, in some epiphyte taxa, by selective removal of potential competitors by grazers.

The illegal introduction of the alien fish, rudd (*Scardinius erythrophthalmus* L.), has widely established reproducing populations of large herbivorous fish in the freshwaters of New Zealand (see also Chapter 27). Consumption trials have found that rudd preferentially

eat a range of native plants over some alien littoral species, at rates of up to 20% of their body weight per day (Lake *et al.* 2002). Not surprisingly, rudd have been implicated in the reduction of submerged plant abundance in some lakes. A fish population, dominated by alien species including rudd, was also found to disrupt charophyte establishment in one shallow lake (de Winton *et al.* 2002).

Where macrophyte beds reach nearly to the water surface, significant waterfowl grazing by black swan (*Cygnus atratus* Latham) can occur (McKinnon and Mitchell 1994).

PROBLEMS WITH LITTORAL PLANTS

Problems associated with littoral macrophytes frequently fall into the category of either insufficient biomass, or excessive biomass. This is illustrated in Figure 24.8, which introduces the concept that a maximum benefit from vegetation presence within a lake occurs at some intermediary plant biomass or abundance level, and divergence away from this optimum leads to management problems.

At very low macrophyte biomass, feed-back mechanisms that act to maintain or enhance habitat suitability for plants tend to be lost. For example, wave dampening capacity and sediment stabilization are directly linked to plant biomass or cover, but in turn these physical processes can detrimentally impact upon plant presence. At some stage, the maintenance of a beneficial level of plant biomass becomes unsustainable (Fig. 24.8). Vegetation decline and loss from the lake littoral is now a growing phenomenon that has wide-reaching impacts on biodiversity and water quality within lakes. For example, at least 60% of shallow lakes in the Lower Waikato region are essentially de-vegetated following years of macrophyte dominance (de Winton and Champion 1993).

At the other end of the biomass scale, problems with

excessive or intolerable macrophyte biomass in lakes are almost always linked to alien weed beds. The reasons for alien weed invasion and dominance, and options for their control are covered in detail in Chapter 27. Problems with littoral algae also relate to their excessive accumulation, such as associated with point sources of nutrient rich waters (e.g. leaching septic tanks), or when smothering growths develop on macrophytes within eutrophic lakes and subsequently lead to vegetation decline.

Loss of biodiversity and native character is one specific problem associated with both ends of the biomass spectrum represented in Fig. 24.8. Conditions that will support a low or unsustainable littoral biomass will be associated with a few, tolerant, plant species, while excessive biomass is almost always associated with large, monospecific weed beds of alien species.

These changes to the composition and abundance of littoral plant communities have large repercussions on the littoral environment and biota. Too often the importance of littoral vegetation in New Zealand freshwaters is not recognised until it is irreversibly modified by alien invasion or disappeared altogether.

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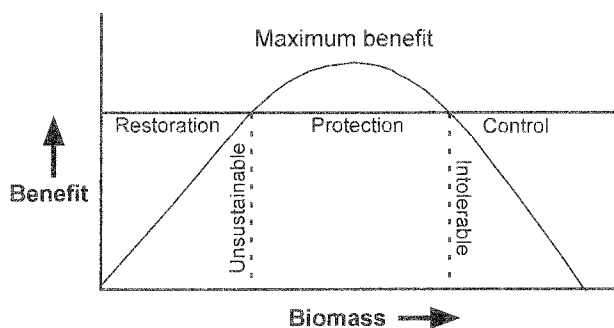


Figure 24.8 Illustration of the concept that littoral plant development has maximum benefit at a particular range of biomass, with restoration, protection or control as appropriate management actions/responses across the gradient of plant biomass.

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Chapter 23

Food webs in lakes

David Rowe and Marc Schallenberg

INTRODUCTION

Food webs are biological systems that encompass the transfer of nutrients, carbon and energy from primary producers and decomposers through to the top predators in ecosystems. They include the main predator-prey and trophic linkages between the plant and animal species present, and the principle energy and carbon-flow pathways in ecosystems. Food webs underpin ecosystem dynamics and stability and they are studied to identify the key species that underpin important resources, such as fisheries. For example, the simple food web underpinning the rainbow trout (*Oncorhynchus mykiss*) fishery in Lake Taupo consists of a few species and a single, limnetic (open-water) food chain linking these species across the four main

trophic levels (Fig. 23.1). The principal food of trout in this lake is smelt (>90%) (Cryer 1991) and the main zooplankton species eaten by the smelt (*Retropinna retropinna*) is the cladoceran *Bosmina meridionalis*. *Bosmina* in turn graze on phytoplanktonic species, including diatoms, green algae, and possibly bacteria (Semenova 1974). The trophic levels in this food web are therefore represented by: (a) the phytoplankton (primary producers), (b) zooplankton (secondary producers), (c) smelt (primary consumers), and (d) trout (secondary consumers). This food web shows that trout production is highly dependent on the limnetic food chain in this lake, and that smelt are the key species in the transfer of limnetic production to the trout.

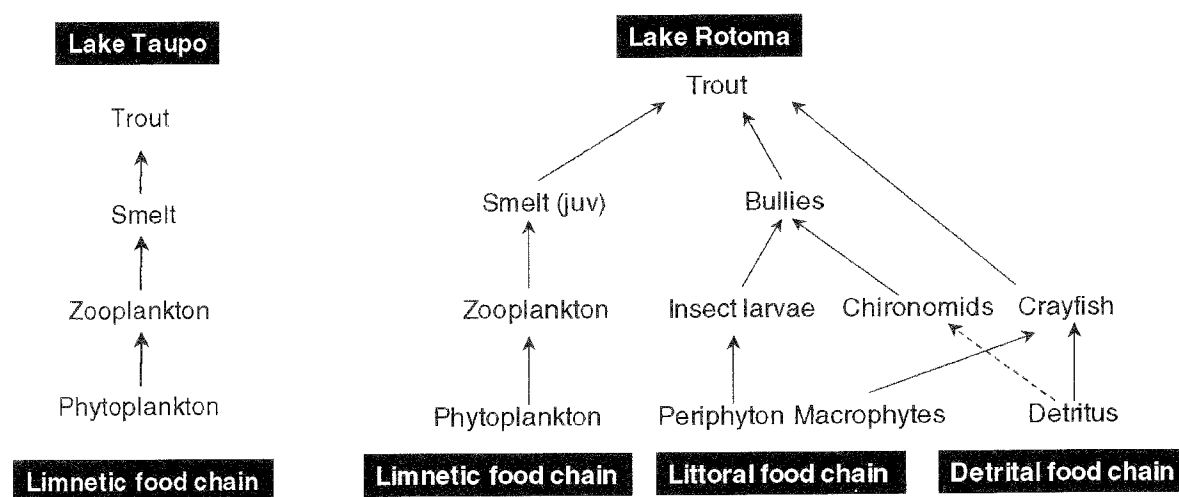


Figure 23.1 The simple, single-chain food web underpinning trout production in Lake Taupo, and the more complex, multi-chain food web in Lake Rotoma.

In other smaller lakes such as Lake Rotoma (which has a surface area of 11.2 km² compared with 622.6 km² for Taupo), limnetic production is less important to trout because the size of the littoral zone is proportionately larger (9.5% for Rotoma versus 2.2% for Taupo), and this zone is also more readily reached by trout. For example, trout are always within 1 km from the nearest shoreline in Lake Rotoma, but can be as far as 9 km away from it in Lake Taupo. The littoral zone is therefore more accessible in Rotoma and the food web underpinning trout production here is more complex than in Taupo, involving both littoral and detrital food chains, as well as a limnetic one (Fig. 23.1). The food web for Lake Rotoma also illustrates the cross-linking of food chains created by omnivory (i.e., predation by a consumer on several species, including species in different food chains). In Lake Rotoma, snails are not an important prey for trout, but they can be a major prey in other lakes.

Such food webs show the trophic links underpinning trout production in lakes, but do not illustrate the real complexity of ecosystem food webs. For example, in large lakes containing many species, the pelagic food web can be highly complex, with extensive omnivory leading to many links across both food chains and trophic levels. Ecosystem food webs can also include the more complex linkages that occur within specific lake habitats and communities such as the littoral zone, the lake bed (e.g., decomposers), or the limnetic zone (e.g., the microbial food web). Furthermore, ecosystem food webs can cross ecosystem boundaries to include links between terrestrial and aquatic, or marine and freshwater ecosystems.

The food webs illustrated in Figure 23.1 are part of an ecosystem web and are termed “structural webs”, as they portray the biotic architecture of the production system in lakes. “Interaction” webs include information on the strength of each connection (e.g., carbon flow, energy flow) and so portray the ecological importance of each link.

Research into the food webs of lakes is growing rapidly now for several reasons. Firstly, stable isotope methods are being increasingly used to elucidate trophic interactions among species (Keough *et al.* 1996; James *et al.* 2000; McCutchan *et al.* 2003). These methods provide better long-term information than diet analysis, but they are expensive and often suffer from too few measurements. Secondly, studies in northern hemisphere lakes have revealed the importance of planktivorous fish for the consumption of zooplankton and hence for indirect effects on phytoplankton density and water clarity (e.g., Carpenter *et al.* 1985; McQueen *et al.* 1986; Reinertsen *et al.* 1990). Other studies have also revealed the importance of herbivorous and benthic fish for macrophyte stability and nutrient cycling within lakes (e.g., Andersson *et al.* 1978; Shapiro and Wright 1984). Knowledge of food webs is

therefore essential for understanding how changes in the trophic status of lakes are mediated by the biota. Thirdly, mathematical models of energy, nutrient and carbon flow through lake food webs are now being developed to help predict the effects of changes in one part of a system on the others (e.g., Jørgensen 1986; Schindler *et al.* 2001). Knowledge of food web structures is essential for these because such models depend on a knowledge of carbon or energy transfers in lakes. Finally, there is a growing interest in the theory of food webs and in the rules that govern their architecture. For example, most food chains within food webs have 3 or 4 links rather than 2 or 5 and the reasons for this are still speculative. Furthermore, there are some important principles that govern the extent of omnivory within a food chain (e.g., Pimm 1982; Pimm and Rice 1987). Knowledge of such rules may provide insights into the effects of introduced species on lakes and show why some species are displaced but not others.

Food web research is therefore expected to expand rapidly in the future and New Zealand lakes provide an ideal “laboratory”. In general, food webs in New Zealand lakes are much simpler than in northern hemisphere lakes because of the relatively few species present. However, the introduction of exotic species into many New Zealand lakes has resulted in food web modification and produced a range of lakes with variations in food web structure. For example, the food web leading to trout in lakes Taupo and Rotoma (Fig. 23.1) differs markedly from that in Lake Otamangakau (Fig. 23.2), because smelt and common bullies have been introduced to the former lakes but not the latter. The food web in Otamangakau is therefore based solely on invertebrates.

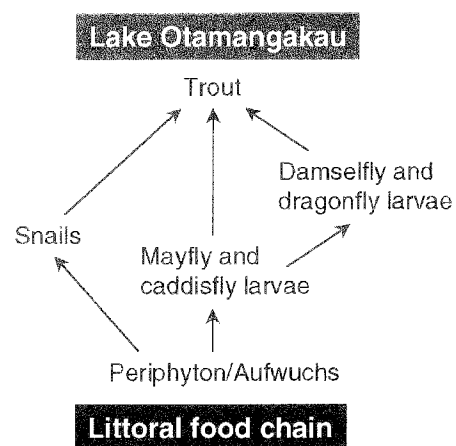


Figure 23.2 The food web underpinning trout production in Lake Otamangakau is based solely on invertebrates.

The few species and the wide range of lakes with similar food webs in New Zealand means that it will be easier to investigate and compare food webs here than in northern hemisphere lakes. To set the scene for such research, this chapter describes the key features of food webs in New Zealand lakes and some of the main factors affecting them. A theme of this chapter is the comparison of food webs and food web dynamics in New Zealand lakes with those of northern hemisphere temperate lakes, where much research has been done on the understanding and theory of food web functioning in lakes.

CHARACTERISTICS OF LAKE FOOD WEBS

Low species diversity

Food webs in New Zealand lakes differ markedly from those in northern hemisphere lakes because of the relatively few species of fish, crustacea and mollusca in New Zealand (Chapman *et al.* 1975; Forsyth 1975; McDowall 1990). New Zealand became separated from Gondwanaland some 70 million years ago (Stevens 1980) and the native fauna developed from the few ancient groups present at that time. Later dispersal of some migratory fish species from other continents, and colonisation by some marine species has since supplemented the freshwater fish fauna (McDowall 1990).

In addition to the few species and the isolation of the fauna, evolution is likely to have been much slower in New Zealand than in the continents, possibly because of the generally colder climatic conditions here and the periodic glaciations that have influenced sea levels and greatly modified species geographic distributions. For example, there has been no rapid speciation and specialisation of fish in New Zealand lakes as has occurred in the more temperate European and North American lakes, and nothing like the spectacular radiation of cichlids in the large, near-tropical lakes of southern Africa. As a consequence, there are few truly lacustrine native fish present in New Zealand lakes.

The native fish in New Zealand are mainly diadromous species that migrate annually between freshwater and the sea. However, landlocked stocks derived from diadromous species have become established in many lakes (McDowall 1990). Some of these stocks were created naturally when they became isolated in the lakes. However, stocking and dam construction have also produced many populations. An exception is the dune lake galaxiid (*Galaxias gracilis*), which evolved from stocks of inanga (*Galaxias maculatus*) (Ling *et al.* 2001) that had been isolated for over 5000 years in small northern dune lakes. This species is now genetically and morphologically distinct from inanga, and its life history traits and morphological adaptations are typical of a fully lacustrine species (Rowe and Chisnall

1996a). In this sense, it is an "obligatory" lacustrine species. However, such populations are rare, and it is apparent that, overall, there has been little evolution of a lacustrine fish fauna in New Zealand lakes. The native fish fauna therefore comprises mainly generalists and there is little specialisation in feeding behaviour, or separation into "guilds", as occurs within northern hemisphere faunas (see also Chapter 25).

The native fish fauna in New Zealand lakes also lacks herbivorous species and true piscivores. The main piscivorous fish are the two species of eel—*Anguilla australis* and *Anguilla dieffenbachii*—which are catadromous. A large (up to 50 cm long), landlocked galaxiid species (giant kokopu, *Galaxias argenteus*) occurs in some coastal lakes, but is not common. Some amphidromous species that have become landlocked are planktivores. For example, the juvenile stage of several species of galaxiid (especially *Galaxias brevipinnis*) and a smelt (*Retropinna retropinna*) are the main planktivorous fish in New Zealand lakes. The larval stage of these species, plus that of the common bully (*Gobiomorphus cotidianus*), are also planktivorous. Detritivores are limited to the grey mullet (*Mugil cephalus*), which occurs only in shallow coastal lakes with access to the sea. Most of the other native fish are primary consumers (e.g., eleotrids, adult galaxiids, juvenile anguillids) and they feed on a wide range of small benthic invertebrates.

Few lakes in New Zealand have no fish, because most can be colonised naturally by small upstream migrant species via lake outlet streams. Some isolated, high-altitude lakes and tarns and some of the smaller, coastal, sand-dune lakes formed within the past 5000 years, and with no outflow, lack fish (Rowe and Chisnall 1997a). However, such lakes are scarce and many have now been stocked with either native or exotic fish species. For example, stocking and inter-lake transfer of native fish by the indigenous Maori population resulted in the establishment of land-locked stocks of galaxiids (at least 2 species), and eleotrids (mainly common bully) in many coastal and inland lakes. During the past 100 years, European settlers created further changes to the fish fauna and food webs of lakes by introducing exotic fish, including salmonids, percids and cyprinids from Europe and North America. Even now, changes are still occurring through the illegal release of perch, catfish, a range of cyprinid species, and *Gambusia*.

The crustacean fauna in New Zealand lakes is also sparse. Non-planktonic, lacustrine species are limited to two freshwater crayfish species (*Paranephrops* spp.), and one of these is usually common in inland lakes. A small freshwater crab (*Amarinus lacustris*), several benthic amphipods (e.g., *Paracorophium* and *Chaetocorophium* sp.), a mysid shrimp (*Tenagomysis* sp.) and a decapod shrimp

(*Paratya curvirostris*) are also present in some lakes. The shrimps and mysids have somewhat restricted habitat requirements, and they are generally confined to coastal lakes where outlet streams provide access to and from the sea. In northern hemisphere lakes, pelagic amphipods and mysids such as *Leptodora kindtii*, *Pontoporeia affinis* and *Mysis relicta* are much more common, but such species do not occur in New Zealand. Furthermore, *Chaoborus* spp. (phantom midge larvae), which are abundant in the limnetic (open water) zone of many northern hemisphere lakes, and which are voracious predators of zooplankton, do not occur in New Zealand.

Molluscan diversity is also limited. The main grazer of periphyton in all lakes is the small benthic gastropod, *Potamopyrgus antipodarum*. Other common gastropods include *Physastra* and *Limnaea* sp. Bivalves are limited to two geographically isolated species of freshwater mussel (*Hyridella* spp.), and to two species of small pea mussel or fingernail clam (*Sphaerium* sp. and *Pisidium* sp.), neither of which are particularly common. Large concentrations of freshwater mussels are a notable feature of some New Zealand lakes. They are confined to shallow waters, occur mainly on sandy substrates, and are the primary filter feeders in some shallow lakes (e.g., Ogilvie and Mitchell 1995).

Limnetic food webs

Because of the limited lacustrine fauna in New Zealand, the limnetic food webs in the larger, inland lakes are generally depauperate in primary and secondary consumer species and so are relatively simple. In general, a diatom-based phytoplankton community is grazed by a zooplankton community of small cladocerans (mainly *Bosmina meridionalis*) and usually a calanoid, or less commonly, a cyclopoid copepod (Chapman *et al.* 1975). In many large South Island lakes, where planktivorous fish such as smelt and juvenile koaro are scarce, the main primary consumers of zooplankton will be the larval stage of the common bully (Rowe and Chisnall 1996b). However, in most of the larger North Island lakes, smelt are abundant and are the main primary consumer. Introduced salmonids (rainbow trout mainly in the North Island, and brown trout in the South Island) dominate the limnetic zone of most large lakes and are the principle piscivores, or secondary consumers. Before trout were introduced, koaro (*Galaxias brevipinnis*) were probably the main limnetic fish in most New Zealand lakes, with juveniles being zooplanktivorous, and adult fish (>50 mm) filling the roles of both benthivore and piscivore (Rowe *et al.* 2002).

Food webs in most of the smaller, shallow coastal lakes in New Zealand differ from those in the larger, inland lakes because these smaller lakes are not suitable for trout. They

are too warm during summer months and most do not contain spawning streams. Food webs in these lakes therefore differ markedly in that their fish fauna is dominated by native species (except where exotic fish have been introduced). Phytoplankton and zooplankton biocoenoses vary more among such lakes (Flint 1975; Chapman *et al.* 1975), and the primary consumers are generally the larvae of common bullies, and the larval and juvenile stages of galaxiids, and/or smelt. There are no limnetic predators in most of these lakes and the only piscivorous fish are eels, which are essentially benthic feeders. However, illegal introductions of exotic species such as perch, rudd, and tench now affect the limnetic food webs of many of these lowland lakes to some extent.

The cladoceran zooplankter, *Daphnia*, is considered a key conduit of nutrients and energy from primary producers to fish in the limnetic ecosystems of many northern hemisphere lakes. *Daphnia* are not as common in New Zealand lakes (Chapman and Green 1987), even in the South Island where larval fish are the only planktivores. This contrasts with many northern hemisphere lakes and has been attributed to heavy predation pressure on *Daphnia* by introduced fish, such as trout (Jeppesen *et al.* 2000). However, *Daphnia* are abundant and are the dominant zooplankton species in Lake Rotoaira (James *et al.* 1999), despite heavy predation by both rainbow trout and juvenile koaro over several decades (Rowe *et al.* 2002). Why *Daphnia* is under-represented in the food webs of New Zealand lakes remains an unresolved question. However, the ability of *Bosmina* to feed on a wider range of prey than *Daphnia* and to feed at much lower cell densities (Semenova 1974) may partly explain its dominance in many New Zealand lakes.

The meta-zooplankton of most New Zealand lakes appears to be dominated by calanoid copepods such as *Calanoeccia* sp. and *Boeckella* sp. Unlike *Daphnia*, which have a high reproductive capacity and are generalist filter-feeders, copepods tend to be slow-growing and to feed selectively on small algae and protozoans. Hence, they are well adapted to the low food densities found in many large glacial lakes. The relative dominance of copepods rather than *Daphnia* has major implications for microbial food webs. Copepods can exert significant grazing pressure on small phytoplankton (Edgar and Green 1994), and they have been shown to be more efficient than cladocerans at consuming heterotrophic nano-flagellates (Burns and Schallenberg 2001a) and, especially, ciliated protozoa (Burns and Schallenberg 1996, 2001a) (Fig. 23.3).

The energy flow from primary producers through the microbial food web to higher trophic levels differs markedly, depending on whether copepods or cladocerans dominate the limnetic zone. When copepods dominate, energy flow to zooplankton appears to be mainly via ciliates

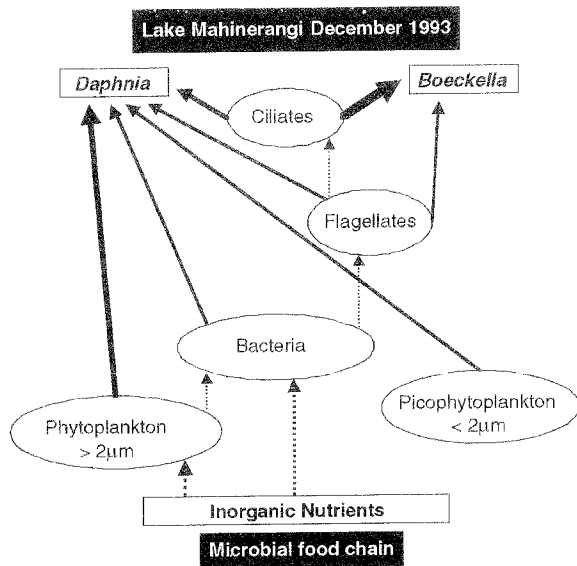


Figure 23.3 Effects of *Daphnia carinata*, *Boeckella hamata* and nutrients on the limnetic food web of Lake Mahinerangi in December 1993, as measured in 4.25 L enclosures. Solid lines indicate consumer effects; broken lines indicate resource effects. The strength of effects is indicated by the thickness of the arrows (modified from Burns and Schallenberg 1996).

at the top of the microbial food web, with respiration losses compounding at each trophic level. However, by feeding across trophic levels, *Daphnia*, “break open” the microbial food web, thereby effecting a more efficient transfer of energy up the food web (Burns and Schallenberg 1996).

It has been suggested that New Zealand limnetic food webs are characterised by relatively low ratios of zooplankton to phytoplankton biomass compared with northern hemisphere lakes (Malthus and Mitchell 1990). Predation on zooplankton by planktivorous fish, as suggested by Jeppesen *et al.* (1997), may potentially be a factor responsible for this. However, the phenomenon also occurs in lakes where planktivorous fish (including larval fish) are scarce. A number of factors, including effective feeding by copepods on both ciliates (Burns and Schallenberg 2001a) and small algae (Edgar and Green 1994), and differences in the edibility of some algal species as suggested by Malthus and Mitchell (1990), might cause this ratio to be relatively low in New Zealand lakes.

Littoral and benthic food webs

Primary production in the littoral zone of many New Zealand lakes is thought to be based primarily on the periphyton, detritus and bacteria (i.e., the aufwuchs) that cover plant surfaces, rather than on the macrophytes, which provide a substrate for periphyton growth (James *et al.* 2000).

The main grazer (i.e., secondary producer) in the littoral

food webs of New Zealand lakes is the snail *Potamopyrgus*, which is an important prey for a number of small fish in lakes. However, a number of lake-dwelling invertebrates, including mayfly and caddis larvae, such as *Triplectides* and *Paroxyethira*, feed directly on macrophyte cells as well as on periphyton, and are in turn consumed by small native fish (Rowe *et al.* 2002). Secondary production in the littoral zone of some lakes also includes filtration of phytoplankton by mussels. Although mussels can form large beds in some lakes, they are not generally consumed in large numbers by either fish or birds, so they form a discrete but important side chain in the littoral food web of many lakes.

Detritus is also an important basis for production in the littoral and benthic zones of lakes. Major detritivores in New Zealand lakes are the chironomid larvae, which, along with *Potamopyrgus*, are the main prey for many small benthic fish. Crayfish are also detritivores, and while present among macrophyte beds in the littoral zone, they are often more common on the weed-free lake bed of North Island lakes, just below the littoral zone (Devcich 1974).

The main primary consumers in the littoral zone are predatory invertebrates such as Odonata (i.e., dragonfly and damselfly larvae), and small fish, mainly common bullies. Below the littoral zone, common bullies are the main primary consumers, and their main prey species are generally snails and chironomid larvae. However, small crustacea (< 5 mm in length), such as amphipods (e.g. *Chaetocorophium lucasi*), copepods and ostracods were the main prey for bullies below the littoral zone at depths down to 70 m in Lake Rotoma (Rowe *et al.* 2001), so crustacea may be the main secondary producers in the benthic food web of other deep, oligotrophic lakes.

Secondary consumers in the littoral zone are mainly native fish such as the koaro and eel. Adult koaro are benthic feeders and occur close to the bottom of some lakes down to depths of at least 70 m (Rowe *et al.* 2002). In the benthic zone of shallow lakes, the main benthivore is generally the eel, which feeds mainly on invertebrates until it attains a length of around 40 cm, after which it increasingly eats more fish.

The production base in the littoral zone of many lakes is varied and substantial, but the overall number of primary and, especially, secondary consumer taxa is relatively low. This will have changed greatly in many small North Island lakes, where a variable range of exotic fish species have been recently introduced and will have added new herbivore, benthivore and detritivore links to the littoral and benthic food webs.

Linkages between food webs

Limnetic and littoral food webs

The diadromous ancestry of many of the native fish species in New Zealand lakes means that the habitat of

adults differs from that of the larval/juvenile stages. As a result of ontogenetic movements between such habitats, there is strong seasonal coupling between the littoral and limnetic food webs. For example, the larval and/or juvenile stage of most of the native fish (e.g., galaxiidae, eleotridae) is limnetic (Rowe 1993a; 1994), whereas the adults are benthic and are found mainly in the littoral zone (Rowe 1993b; Rowe *et al.* 2002). There is, therefore, a large, seasonal movement of fish from the limnetic zone to the littoral zone and a corresponding seasonal change in predation pressure from limnetic zooplankton to benthic invertebrates. Mussels also contribute to such food web coupling by converting limnetic phytoplankton into littoral biomass.

Limnetic and benthic food webs

Malthus and Mitchell (1990) found lower ratios of zooplankton to phytoplankton in New Zealand lakes than in most northern hemisphere temperate lakes and suggested that this results in greater losses of energy and nutrients from the water column to the sediments. Furthermore, the lack of vertically migrating, limnetic mysids, amphipods and *Chaoborus* spp. in New Zealand lakes indicates that vertical energy and nutrient fluxes between the hypolimnetic and epilimnetic waters may be significantly different than in many overseas lakes. Although such species are lacking in New Zealand lakes, the nocturnal vertical migrations of planktivorous fish larvae (Rowe 1994; Rowe and Chisnall 1996b) may fulfill this role in New Zealand lakes to some extent.

Aquatic and terrestrial food webs

A large terrestrial input into trout production occurs in years when beech trees in lake catchments flower. Some South Island lakes are surrounded by beech forest and large numbers of mice (as well as other seed eaters and predators such as stoats) occur in these forests in those years when synchronous flowering of the beech trees occurs. Many of the mice either fall into the lake or enter the water directly and some of these are consumed by large brown trout. Similarly, in some North Island lakes, winged terrestrial insects such as cicadas (*Amphipsalta* sp.) and the green manuka beetle (*Pyronota* sp.) are very abundant during summer months in some years and at these times can contribute heavily to the diet of trout in lakes. There is therefore an ecologically significant transfer of terrestrial production to the secondary consumers of these lakes in some years. Furthermore, terrestrially derived dissolved and particulate organic matter, either deposited into lakes from surrounding vegetation or washed in from streams, rivers and groundwater may also provide a significant source of energy and nutrients to many detritus-based aquatic food webs in lakes, especially in shallow lakes with large catchments.

Birds may also transfer lake-derived biomass to terrestrial ecosystems (and to other lake ecosystems). Shags (*Phalacrocorax* spp.) are the main predators of fish in New Zealand lakes and can feed in one lake while roosting in another. Similarly, ducks, swans and geese feed on littoral invertebrates and aquatic plants (depending on species differences in diet) and can transfer aquatic products to either land, or other waterways.

Marine and freshwater food webs

In many coastal New Zealand lakes where annual immigration and/or emigration of fish such as eels, mullet, inanga, smelt and flounder occurs, there are potentially large transfers of carbon and nutrients from the sea to freshwater and vice versa. Such transfers can be important for ecosystem functioning. For example, Pacific salmon generally die after spawning in freshwater and the decomposition of their carcasses can have significant effects on the productivity of North American west coast outlet streams and downstream lakes. The spring immigration of fish such as whitebait into New Zealand rivers brings an annual influx of marine nutrients and carbon into freshwater ecosystems. However, fish immigration into lakes is likely to be important more for the recruitment of fish than for the import of carbon or nutrients, and to have a greater influence in lowland, coastal lakes such as Lake Ellesmere than in higher altitude, inland lakes. In contrast, the emigration of mature eels to marine spawning grounds can be expected to export carbon and nutrients from lakes, but the extent of such transfer is unknown. Such transfers of nutrients, energy and carbon between ecosystems may be heavily influenced by both fish and bird predation as well as by intensive harvesting (e.g., by humans).

FACTORS AFFECTING LAKE FOOD WEBS

Consumer and resource control of energy flow

Both lake type and how accessible the lakes are to fish have a major bearing on their food web structure and function because these variables influence the composition of the fish fauna. Many of the larger, generally deeper and less-enriched inland lakes of New Zealand are not generally accessible to diadromous fish species because of natural barriers such as falls and rapids. Damming has compounded such effects. As a result these lakes are dominated by salmonids and contain far fewer native fish species (generally limited to bullies, smelt, koaro) than lower altitude, generally shallower and more enriched, coastal lakes. The latter lakes often contain a more diverse native fish fauna comprising eels, bullies, inanga, galaxiids, smelt, mullet, and flounder. This difference in fish fauna, together with the differences in physical attributes among

such lakes (e.g., size, depth, thermal characteristics), can be expected to produce major differences in the food web structure and energy flow between lowland coastal and higher altitude, inland lakes (Rowe and Graynoth 2002).

The extent to which nutrient availability and consumer effects structure food webs in lakes has been studied extensively overseas and more recently in some New Zealand lakes. Although dismissed by Malthus and Mitchell (1990) as an explanation for the low zooplankton to phytoplankton ratios observed in New Zealand lakes, Jeppesen *et al.* (1997; 2000) argued that fish predation on zooplankton could be an important factor structuring zooplankton populations, particularly in shallow oligotrophic to slightly eutrophic lakes. However, Jeppesen *et al.* (2000) concluded that the cascading trophic influence of fish predation on phytoplankton and protists in New Zealand lakes was less pronounced than in northern temperate lakes.

Both consumer and nutrient effects significantly influenced the structure and functioning of microbial food webs in three lakes of different trophic status (Burns and Schallenberg 1996, 2001b). Predation pressure by copepods on ciliated protozoa was found to be stronger than that due to cladocerans in these lakes (e.g., Lake Mahinerangi, Fig. 23.3), suggesting that the relatively greater dominance of copepods, compared with north temperate lakes, may be responsible for the low ciliate abundance characteristic of some New Zealand lakes (James *et al.* 1995; Burns and Schallenberg 1996; James *et al.* 2001)

Food web structure

The length of food chains is another important factor influencing food webs and hence ecosystem functioning. For example, the efficiency of energy transfer from primary production up to fish is reduced as food-chain length increases (Dickie 1976). This implies that energy flow and trout production in Lake Otamangakau (Fig. 23.2) should be much more efficient (though not necessarily greater) than in lakes where forage fish are present, such as Lake Rotoma and Lake Taupo (Fig. 23.1). Furthermore, bioaccumulation of heavy metals and pesticides increases with food-chain length, so the concentration of contaminants in top predators such as salmonids will be higher in lakes with longer food chains (e.g., Berglund *et al.* 2000; Rasmussen *et al.* 1990). This chain-length effect influences concentrations of mercury in top predators (Cabana *et al.* 1994), and it explains the higher mercury levels in the larger and older trout and eels of New Zealand lakes where natural mercury contamination occurs as a result of geothermal springs (Kim 1995; Robinson *et al.* 1995).

The number of species present at each trophic level and

the extent of food web connectivity that arises from species diversity is also important. The low diversity of native species in New Zealand lakes means that food webs are relatively simple. In some ways, this simplifies management of the resource, but this simplicity is also a liability in that a reduction in one species, or the addition of a new species, can dramatically change food webs. Low species diversity means that there is less connectivity and potentially less resilience in food webs. A change in one component of a simple food web can therefore affect not just the chain it is part of, but the entire food web and, hence, the ecosystem (c.f., Vander Zanden *et al.* 1999). Theoretically, simple food webs are inherently more vulnerable to change than are more complex ones, and in this sense they are less resilient. Howard-Williams and Kelly (2003) noted that the paucity of species and interactions reduced the options for restoration by bio-manipulation in New Zealand lakes.

Key species

It is apparent that smelt are a key species in the transfer of limnetic production to rainbow trout in Lake Taupo. The loss of smelt in this lake would be expected to have a major effect on the limnetic food web and trout production here. However, in smaller lakes where littoral food webs are more important, the role of smelt is likely to be lower and it may not be a key species for trout production in such lakes.

Changes in the abundance (as against the loss) of key species can also have major effects on food webs, especially in simple limnetic webs such as that of Lake Taupo. For example, ash from the eruptions of Mount Ruapehu temporarily changed the water chemistry of this lake, resulting in a change in the dominant phytoplankton species, from a diatom, which tends to sink if not re-suspended by water movements (the normal situation), to a more buoyant species of green algae that remained longer in surface waters (pers. comm., M. Gibbs). Trout growth rates in the following year were spectacular and were attributed to the greater availability of the green algae to zooplankton in the surface waters. This is assumed to have resulted in a higher production of both zooplankton and smelt, and ultimately in trout. The following year, the lake returned to its previous state.

Some fish species may also be key species in lakes because they have a major, on-going influence on the abundance of other species, particularly planktivores, and so dominate the structure of the littoral food web. For example trout predation suppresses the abundance of juvenile koaro in many New Zealand lakes (Rowe *et al.* 2003), whereas gambusia (*Gambusia affinis*) suppresses the abundance of the dune lakes galaxias (*Galaxias gracilis*) in several northern dune lakes (Rowe 2003).

Annual variations

Annual variations in fish numbers related to stocking or recruitment success can be expected to temporarily influence food web structures in lakes. A large temporary decline in the abundance of salmonids in Lake Michigan resulted in a decline in planktivorous fish, an increase in *Daphnia*, and a marked increase in water transparency related to reduced phytoplankton (Scavia *et al.* 1986). Such variations in the abundance of top predators can be expected to influence limnetic food chains through top-down effects. In New Zealand lakes, large variations in fish recruitment are likely to be related mainly to the stocking of salmonids or eels. Deliberate overstocking of trout in Lake Okareka (Rotorua) produced some cascading consumer effects, including more *Daphnia* and a reputed increase in water clarity. The stocking of elvers into lakes that they cannot access naturally is often carried out by eel fishermen in the North Island in anticipation of obtaining a harvest in 10 to 15 years time. If the stocked eels are not harvested, they continue to grow and become increasingly piscivorous, thus affecting small native fish (e.g., Rowe and Chisnall 1997b). Year-to-year variations in the recruitment of some fish species into lakes may also be potentially large enough to affect lake food webs. The migrations of sea-run brown trout, salmon, or diadromous native species from the sea into New Zealand lakes are probably too small for this. However, a fourfold variation in the annual abundance of smelt can occur in Lake Taupo, and a sevenfold inter-annual increase in common bullies occurred in a northern dune lake (Rowe *et al.* 1999). Such large inter-annual variations reflect changes in recruitment success between years, with large implications for energy flow in lacustrine food webs.

Seasonal variations

Seasonal changes in water temperature can also have a large influence on some food webs in lakes. Colder seasons resulted in shorter, less complex microbial food chains and, hence, in more efficient transfer of energy to zooplankton in a salmon lake (Sorokin and Paveljevs 1978). Seasonal changes in temperature can also result in predation on different species and, hence, to seasonal variation in food webs. For example, during summer months, rainbow trout in Lake Rotoaira fed primarily on snails, Odonata larvae and on juvenile goldfish in the littoral zone (Fig. 23.4), and the littoral food web dominated trout production at this time. However, in winter, when these prey species were scarce, trout used the limnetic food web and fed on juvenile koaro and *Daphnia*. In winter, invertebrate production is low in many inland, high-altitude, and hence cold lakes. Invertebrate prey availability declines for cold-water fish such as salmonids but small forage fish that overwinter in such lakes provide an alternative food source at this time

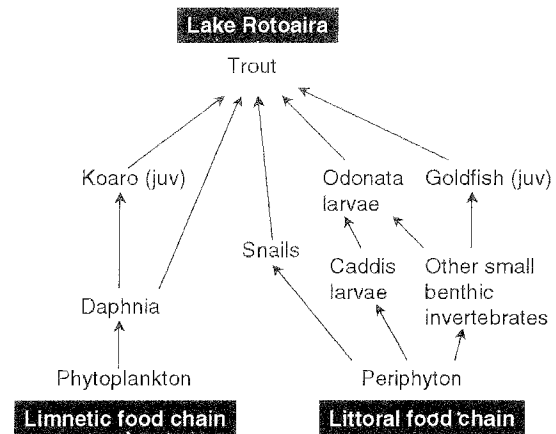


Figure 23.4 Food web for rainbow trout in Lake Rotoaira. The littoral food web is used mainly in summer, and the limnetic web mainly in winter.

and, in Lake Rotoaira, koaro formed the basis for the winter food web. However, in nearby Lake Otamangakau (Fig. 23.2), no forage fish occur so no winter food web exists for trout.

Invasion by exotic species

Food webs can be greatly modified through changes in the abundance of either top predators or primary producers, thus altering top-down or bottom-up processes. A major factor responsible for changes to food webs in many New Zealand lakes is the introduction of new species of plants. Introduced macrophytes, such as the hydrocharitacean oxygen weeds (e.g., *Lagarosiphon major* and the hornwort *Ceratophyllum demersum*) have displaced native plants in the littoral zones of many lakes. A major effect of these invasions was the decrease in diversity of the native macrophyte species contributing to primary production in the littoral zone and the replacement of this community with tall, surface-reaching, mono-specific macrophyte stands. These plant invasions reduced the diversity and stability of primary production in the littoral zone of affected lakes. A further change related to the tall growth form of these exotic plants compared with native species, is likely to have been the increase in area and exposure of plant surfaces for the attachment of periphyton in shallow lakes. A corresponding decline in the open-water, limnetic zone also occurred in many shallow lakes invaded by these exotic plants.

In reviewing the ecological outcomes of over 850 exotic plant and animal introductions in a variety of ecosystems, Simberloff (1981) concluded that predation was the most important cause of extinctions resulting from the invasion of exotic species. In New Zealand lakes, the widespread introduction of piscivorous fish such as salmonids and perch has probably had a great effect on food webs because

these top predators added a new trophic level, while displacing native species and thereby removing existing links. For example trout introductions resulted in a marked decline in koaro in a number of large North and South Island lakes (Rowe 1993b; Rowe *et al.* 2002, Rowe *et al.* 2003). In many of the South Island lakes dominated by brown trout, limnetic koaro virtually disappeared, but they still remained common in North Island lakes dominated by the less piscivorous rainbow trout (Rowe *et al.* 2003). The less extensive decline of koaro following rainbow trout introductions to Lakes Taupo and Rotorua was followed by marked fluctuations in the annual size and condition of trout, indicating variable production via the limnetic food web. This is likely to have been caused by the large annual fluctuations in koaro and trout numbers creating imbalances in the ratio of predator to prey. The later introduction of a more specialised planktivorous fish (i.e., smelt) re-established the limnetic food chain and, together with a reduction in trout density, stabilised annual trout production. Why smelt could withstand trout predation and koaro could not is a still unresolved question, but may be related to either the differing life histories of these species (e.g., semelparity vs iteroparity), to the higher fecundity of smelt (McDowall 1990), or to competition for food between trout and adult koaro, and its absence between trout and adult smelt.

The impacts of piscivorous and planktivorous fish introductions on zooplankton or invertebrates have not been extensively studied in New Zealand lakes to date. The introduction of trout to fishless north temperate lakes has been shown to severely affect large zooplankton, benthic invertebrates and vertebrates. However, the remarkable dispersal characteristics and, hence, widespread distribution of native galaxiids in New Zealand lakes complicates cascading trophic interactions resulting from salmonid introductions. The presence of trout appears to have had little effect on native benthic species, such as snails or crayfish, but the more recent introduction of the brown bullhead catfish (*Ameiurus nebulosus*) to Lake Taupo may reduce the crayfish population here because adult catfish feed heavily on this crustacean. Catfish also feed on a wide range of other benthic invertebrates in lakes, and thus can be expected to compete with the common bully for food. Invasions of catfish may therefore create some large changes in the benthic food webs of lakes.

The effects of other exotic fish species such as goldfish, rudd, tench, koi carp, and gambusia on food webs in New Zealand lakes are not as well known at present, as their geographic spread has, until recently, been generally confined to small northern lakes. However, these species are now being spread further south, and can be expected to add a new level of complexity to lacustrine food webs through the addition of detritivores, benthivores, and

herbivores. Many of the natural predators for these species (e.g., pike) do not occur in New Zealand lakes, so large densities of these species can be expected to occur and food webs will be changed in affected lakes. Large densities of both rudd and tench were a noticeable feature of the fish fauna in Parkinson's Lake and shag (*Phalacrocorax* spp.) predation on these fish was limited, except when all exotic macrophytes were removed (Rowe and Champion 1994).

Introductions of new species of fish to lakes cannot be condoned, but they do provide opportunities to test theoretical concepts that underpin models of food-web structure. For example, Vadas (1990) used food-web models to determine the role of omnivory in structuring freshwater fish assemblages. Modelling predicted that, in theory, a species could not prey on a lower species and simultaneously compete with it for food, because models incorporating such links proved to be highly unstable. This implies that simultaneous predation and competition cannot exist within the same food chain, and that the introduction of a predator with these capabilities will result in the elimination of its prey species. In reality, such simple and constrained food chains are rarely observed in nature because predators are generally omnivorous and switch to other prey when a preferred species becomes rare. However, such a model does help explain the collapse of koaro following the introduction of smelt into Lake Rotopounamu, near Turangi. This lake once contained only koaro, and the introduced smelt are thought to have both competed with the juvenile koaro for planktonic prey in the limnetic zone, while simultaneously preying on their planktonic larvae. As there were no refugia for the koaro (such as deep waters or large inlet streams) they became extinct in this lake (Rowe 1993b).

Climate change

Recent studies have indicated that climate change can also affect lake food webs. For example, it has been suggested that environmental warming differentially affects the risk of extinction of organisms in different trophic levels and that high biodiversity provides a buffer against the effects of environmental variation on food webs (Petchey *et al.* 1999). Ozone depletion may also affect lake food webs, because New Zealand zooplankton species show varying sensitivity to UVB irradiation (Wübben *et al.* 2001). This suggests that UVB may affect the zooplankton community structure and/or zooplankton distributions in New Zealand lakes. Food webs of coastal lakes have been shown to be at risk from both rising sea levels and the predicted future shortages of freshwater on the east coast of New Zealand. A study of Lake Waiholo, a shallow, coastal lake with a connection to the sea, showed that zooplankton abundance and taxonomic richness was strongly reduced by even small increases in salinity induced

by saline intrusions (Schallenberg *et al.* 2003). The authors concluded that climate-induced increases in salinity caused by rising sea levels, lower predicted rainfall (on the east coast), and increasing demands on freshwater for irrigation will strongly alter the limnetic foodwebs of many similar coastal lakes, lagoons, and wetlands in New Zealand.

Eutrophication

Eutrophication can greatly affect the structure of food webs in lakes by altering both the top-down and bottom-up processes that influence biological production. The decline in water quality (e.g., increased turbidity, reduced light penetration, reduced hypolimnetic oxygen) that accompanies eutrophication can also change the depth distribution of many lacustrine organisms (Rowe 1994, Rowe *et al.* 2001). Changes in phytoplankton (e.g., both increased production and more cyanophyceae), in zooplankton (e.g., fewer rotifers), and in benthic invertebrates (e.g., fewer crustacea, more chironomidae) are all symptomatic of increased eutrophication and represent large changes in the structure and functioning of lake food webs. Ultimately, the reduction in light levels in eutrophic lakes results in the disappearance of macrophytes and the domination of primary production by phytoplanktonic algae. Loss of littoral zone diversity resulting from macrophyte collapse is accompanied by an increase in the importance of the detrital food chain.

In New Zealand, such effects of eutrophication are most notable in some of the Rotorua lakes, where increased eutrophication has led to a reduction in pelagic forage fish such as smelt (Rowe and Taumoepeau, in press), an increase in benthic fish such as common bullies (Rowe 1999), and a corresponding change in trout predation from smelt to bullies (Rowe 1984), with a decline in trout growth rates (Fish 1968). These changes suggest a reduction in the limnetic food chain and an increase in the benthic chain, despite increased algal production. In many northern hemisphere lakes, eutrophication has

resulted in a decline in the valued, cold-water fish genera such as salmonids and eels, and a corresponding increase in the smaller, less valued genera such as cyprinids. This shift in the fish fauna no doubt interacts with bottom-up effects to further complicate the changes in food web structures created by eutrophication. At present, cyprinids are not present in many of New Zealand's larger lakes, so the effects of eutrophication on food webs may be less pronounced here.

BIOMANIPULATION OF LIMNETIC FOOD WEBS AND ITS POTENTIAL AS A MANAGEMENT TOOL

Detailed knowledge of the functioning of lake food webs can be useful in biomanipulation—a form of lake management based on the manipulation of limnetic food webs by altering the biomass of piscivorous or planktivorous fish. The stimulation of cascading trophic interactions by manipulating predators or planktivores has been used extensively overseas to control lake water clarity and algal biomass. This is generally accomplished by releasing *Daphnia* (or other efficient zooplankton grazers) from predation pressure. The simplest way to accomplish this is to stock high densities of piscivorous fish. However, a number of limnological criteria must be met for such biomanipulation to have a high likelihood of succeeding (Table 23.1).

Burns (1998) was generally pessimistic about the utility of biomanipulation as a management tool for improving water quality in New Zealand lakes. The differences between limnetic food webs of New Zealand and north temperate lakes—e.g., the scarcity of *Daphnia*, low fish species diversity, scarcity of obligate planktivores, constrained distribution of zooplanktivorous amphipods and mysids, and sustained periods of planktivory by larval fish in New Zealand—indicate that caution should be exercised when attempting such biomanipulation. For example,

Table 23.1 Criteria for water bodies in which biomanipulation may be successful at controlling algal biomass (from Reynolds 1994 and 1997, and modified from Burns 1998).

No.	Criteria
1	Area of water body is < 40 ha
2	Mean depth ≤ 1 m
3	Low hydraulic residence time (< 10 –1 30 days)
4	Macrophytes are dense and cover ≥ 50% of the lake bed
5	Vigorous macrophyte growth is promoted
6	Bloom-forming cyanobacteria do not dominate the phytoplankton
7	Fish are present
8	<i>Daphnia</i> are present and large species are promoted

Jeppesen *et al.* (2000) concluded that “fish, including trout, have a major effect on the zooplankton community structure and biomass in the pelagial of the shallow oligotrophic to slightly eutrophic New Zealand lakes, but that the cascading effects on phytoplankton and protists are apparently [only] modest.” Nevertheless, exotic piscivores and native eels are widespread in New Zealand and their biomass may be manipulated by modifying existing stocking programmes. Alternatively, some planktivorous species may be reduced by disrupting their spawning (e.g., through manipulation of water levels or spawning habitats). Biomanipulation of top predators now underpins much of the northern hemisphere research on restoring lake ecosystems. To date, there have been few attempts to manage water quality using biomanipulation in New Zealand, and, in general, data are insufficient to unequivocally demonstrate positive effects of biomanipulation on lakes where it has been attempted. Howard-Williams and Kelly (2003) noted this deficiency and, in warning of the dangers of applying general models applicable to northern hemisphere lakes to New Zealand, pointed out that successful biomanipulation will often be reliant on local, lake-specific measures. This includes a knowledge of the food web characteristics of lakes targeted for restoration.

The one area where biomanipulation of food webs has proved useful in lake restoration in New Zealand is the use of grass carp stocking to eradicate exotic weeds in lakes (Rowe and Champion 1994; Rowe *et al.* 1999). The grass carp (*Ctenopharyngodon idella*) is an exotic herbivore that does not reproduce naturally in New Zealand waters, so stocking is required to manipulate exotic weed density. Overstocking results in the eradication of a number of the problem weed species, in turn paving the way for removal of exotic fish species. Eventually, native plants regenerate and native fish can be restocked to restore the lake’s original native flora and fauna.

FUTURE DIRECTIONS

Few studies have rigorously documented the impacts of fish introductions on unmodified food webs in New Zealand lakes and, hence, little is known about New Zealand’s indigenous lacustrine food webs. One reason for this is that salmonids have been widely dispersed throughout New Zealand and now other exotic species are being increasingly (and illegally) spread into remaining lakes not suitable for trout. Therefore, the opportunity for studying the structure and functioning of New Zealand’s diverse indigenous lake ecosystems is fast diminishing.

Apart from the successful restoration of Parkinson’s Lake (Pukekohe) through the eradication of all exotic plants and fish (Rowe and Champion 1994), there has been relatively

little research in New Zealand focusing on the potential for biomanipulation as a tool for lake management. Consequently, there have been few attempts to use biomanipulation to manage New Zealand lakes. Those that have occurred are limited to small lakes and to the eradication of exotic plants by grass carp, rather than to the biomanipulation of top predators. Despite a number of important limitations to the likely success of biomanipulation of planktivores in New Zealand (Burns 1998), research should continue to explore such potentially useful tools. For example, the potential for food web manipulation to control the dynamics of toxic cyanobacteria and of human pathogens such as *Campylobacter*, *Giardia* and *Cryptosporidium* in New Zealand drinking water supplies has yet to be explored and may prove feasible. Similarly, manipulation of herbivorous fish, top predators and the spawning habitats of planktivores may also prove useful in the control of exotic fish.

Though limnetic food webs are considered to be relatively simple in structure, their functioning often incorporates biological feedbacks, threshold responses, and adaptive behaviour. Therefore, their food web components and attributes can exhibit non-linear and even chaotic behaviour. The recent widespread recognition of this, and the ongoing development of modelling and statistical tools that can incorporate this complexity, are now leading to an improved understanding of limnetic food web functioning.

In the future, as climate change becomes more pronounced, unknown changes in lacustrine food webs will occur with potentially wide ramifications. Research into the impacts of environmental warming, increased UV radiation, reduced freshwater inflows, and increasing salinisation on lakes is its infancy in New Zealand. It is important that this research continues to develop, as the unique characteristics of New Zealand’s aquatic flora, fauna, and foodwebs necessitates a New Zealand perspective on the potential impacts of climate change. Such research will be necessary if lakes are to be managed to maintain their biodiversity during times of rapid anthropogenic environmental change.

Finally, food web theory holds much promise for understanding the effects of species introductions and removals on lacustrine food webs and, hence, for coping with the problems of exotic species and lake restoration. The recent finding that, in a wide range of complex webs, each species is preyed upon or preys on approximately 10% of all other species, indicates that connectance is limited within food webs (Williams and Martinez 2000). It is apparent that general rules govern the structure of food webs—finding the patterns in food web structures and the rules that govern them will open up exciting possibilities for ecological science, particularly in New Zealand lakes.

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Chapter 22

Primary production in the open water

Marc Schallenberg

INTRODUCTION

Phytoplankton primary production fuels much of the secondary productivity of lake food webs (see Chapter 23), including the production of many aquatic organisms of importance to humans. Phytoplankton are planktonic autotrophic organisms, and include algae and cyanobacteria. They fix inorganic carbon and, in the case of some cyanobacteria, atmospheric nitrogen, converting these to organic forms that can then be used by heterotrophic organisms. In addition to fuelling food webs in lakes, phytoplankton biomass and production are important to humans because nuisance blooms of phytoplankton affect our perception of water quality.

Biomass is an expression of the standing crop of phytoplankton in a lake, usually expressed per unit volume of water. Productivity is the rate of change in biomass per unit time. The biomass and productivity of phytoplankton in New Zealand lakes spans a range typical of other temperate regions and has been discussed by Viner and White (1987). As these authors pointed out, New Zealand is a water-rich nation with relatively high average annual precipitation and a small population. Thus, many New Zealand lakes tend to be in the low end of the range of phytoplankton biomasses and productivities, relative to many other parts of the world.

An excellent review of phytoplankton growth by Viner and White (1987) contains much detailed information about patterns of phytoplankton growth in individual lakes. This chapter will not repeat their detailed descriptions. This chapter focuses on: (1) factors that control or limit phytoplankton primary production and biomass in New Zealand lakes, (2) the applicability of some paradigms concerning phytoplankton, which are largely based on findings from north-temperate lakes, to New Zealand lakes, and (3) how the ongoing problems of anthropogenic eutrophication and climate change affect phytoplankton production in New Zealand lakes.

PLANT NUTRIENTS AND PHYTOPLANKTON IN LAKES

Ever since the 1960s and 1970s, when the links between anthropogenic inputs of nutrients and the development of algal blooms in lakes was firmly established, limnologists have been trying to predict the impacts of human perturbations on nutrient cycles within lake ecosystems. In general, human interest in this issue relates to degraded “water quality”, a term that encompasses characteristics including water transparency, algal biomass, and the dominance of noxious phytoplankton such as blooming cyanobacteria and their often-associated toxicity and unpleasant odour.

Resource limitation

Organisms are dependent on numerous resources obtained from their environment. In 1840, German soil scientist Justus von Liebig proposed what has become known as “Liebig’s Law of the Minimum”, which states that the productivity of an organism at a given time is limited by the resource that is in shortest supply, relative to demands. Liebig’s Law, as it applies to aquatic autotrophs, is now more commonly known as the concept of “resource limitation”.

The phosphorus limitation paradigm

By the mid 1960s, a flood of research, stimulated largely by the accelerating degradation of water quality in the North American Great Lakes, was indicating that the increasing algal biomass in these lakes was driven by the inputs of phosphorus from municipal and industrial effluent (e.g., wastes containing sewage and phosphate detergents). Subsequent research on other North American and European lakes confirmed that the availability of phosphorus generally determines algal biomass development. The remarkably strong positive correlation between

total phosphorus (TP) and chlorophyll *a* (an index of algal biomass) was discovered (Sakamoto 1966; Dillon and Rigler 1974), seemingly confirming the supreme importance of phosphorus in fuelling phytoplankton growth. Consequently, it became a widely accepted paradigm (*sensu* Kuhn 1962) in limnology that algal growth and bloom formation in lakes was related to the availability of phosphorus. As a result, the management of eutrophication became focused on the reduction of phosphorus inputs to lakes.

Though a useful restoration tool in many circumstances, the total phosphorus versus chlorophyll *a* relationship has its shortcomings. One problem is the large statistical error (which is typically \pm one order of magnitude) associated with using the relationship for prediction and management. A second problem is that total phosphorus is not always a good measure of available phosphorus, as it includes phosphorus bound to organic and inorganic suspended particulate matter, as well as that bound to dissolved organic matter, all of which is unavailable to eukaryotic phytoplankton. This is a serious problem in shallow lakes, which commonly experience episodes of wind-induced sediment resuspension and associated high particulate phosphorus concentrations. Another difficulty in assuming that phytoplankton biomass can be controlled by reducing phosphorus input to lakes is that, for some lakes, phosphorus inputs originating outside the lake are a small proportion of the total phosphorus input to the water column. This is typical of lakes that develop anoxic hypolimnia (bottom waters), because under anoxic conditions some phosphorus bound to sediments becomes soluble and diffuses into the water column. In shallow lakes, an extensive contact area between the sediments and the epilimnion (mixed layer) enhances both the diffusion and turbulent entrainment of sediment phosphorus directly into the euphotic zone of the water column (the zone where light is sufficient to stimulate primary production).

Nitrogen limitation in New Zealand lakes

Although the importance of phosphorus availability has been well established for many North American and European lakes (Vollenweider 1968; Schindler 1978), the availability of other macronutrients such as nitrogen (N) and silicon (Si), as well as some micronutrients (see Goldman 1964), can determine phytoplankton growth at times. Phytoplankton communities in New Zealand differ from those of North American and European lakes in that the phytoplankton of many New Zealand lakes is limited by nitrogen availability (White 1983; White *et al.* 1985).

There are a number of possible reasons for the prevalence of nitrogen limitation in New Zealand phytoplankton. Firstly, a substantial part of New Zealand's waters are underlain by volcanic geology and soils. Lakes, rivers and streams situated in volcanic catchments, such as those in the Taupo volcanic zone, receive substantially greater natural phosphorus inputs from rock weathering (Timperley 1983) than lakes in non-volcanic catchments. Secondly, combustion processes (e.g., the combustion of fossil fuels in automobile engines, etc.) convert atmospheric N_2 to NO_x s, which may be dissolved in rainwater. The NO_x s are directly or indirectly (via bacterially-mediated transformation) available to phytoplankton via the hydrological cycle. The atmospheres of the north and south hemispheres undergo little mixing. This, together with the low geographic density of population and industry in the southern hemisphere, and New Zealand's isolation from other large land masses, results in very low NO_x content in precipitation. Thirdly, the typical agricultural soil enrichment practise in New Zealand of planting nitrogen-fixing clover and top-dressing with phosphate fertiliser contrasts with practices in North America and Europe, where more soluble inorganic nitrogen fertilisers are used. The result of this difference may be a lower N:P ratio in agricultural runoff in New Zealand as compared with countries relying heavily on

Biological importance of phosphorus and nitrogen

Phosphorus is important to organisms, as it is a vital component of chemical energy cycling in cells (e.g., in the phosphorylation/dephosphorylation of ATP, ADP, and AMP) and it is a major constituent of nucleic acids. In the environment, phosphorus tends to be bound to particles and readily forms insoluble inorganic oxyhydroxides under oxic conditions. Therefore, its recycling into biologically available forms (e.g., dissolved phosphate) is limited. Under reducing conditions (e.g., when oxygen is depleted), oxyhydroxides are reduced and inorganic phosphorus becomes soluble. Phosphorus is commonly the nutrient that determines the growth of phytoplankton in many north-temperate lakes. Nitrogen is an essential component of proteins (including enzymes) and chlorophyll *a*, and is the nutrient that often determines the growth of terrestrial plants. Phytoplankton growth in many New Zealand lakes is often nitrogen limited.

inorganic nitrogen fertilisers. These differences, in theory, predispose levels of phytoplankton in New Zealand lakes to be limited by nitrogen rather than by phosphorus.

Limitation by availability of other nutrients

Where diatoms are the dominant component of the phytoplankton, silicon availability can limit growth. Diatoms produce siliceous external frustules and thus require dissolved silicon. During diatom blooms (e.g., as occur in Lake Taupo; White *et al.* 1980), dissolved silicon may become severely depleted in the water column. Further growth then depends on the (re)cycling of silicon, which is very slow due to the low solubility of silicate minerals.

It has been suggested that micronutrients such as iron, molybdenum, and cobalt may determine phytoplankton growth at times. For instance, phytoplankton production in the Southern Ocean has been shown to be determined by the availability of iron (Boyd *et al.* 2000). The concentrations of iron in the open ocean are extremely low, whereas concentrations of iron and other trace nutrients tend to be higher in lakes due to crustal weathering and continued micronutrient input from lake catchments and precipitation. This may be why consistent limitation by micronutrient availability in lakes has not been demonstrated, though some researchers have found that the addition of trace quantities of these micronutrients can at times stimulate phytoplankton production. Some bioassay experiments carried out in New Zealand lakes have occasionally shown such responses (Golman 1964; White

and Payne 1977, 1978). However, the potential for micronutrients to limit phytoplankton production in lakes, including New Zealand lakes, has yet to be thoroughly explored.

106:16:1 – The Redfield Ratio

Research on nutrient limitation was greatly assisted by the work of Redfield *et al.* (1963), who showed that nutrient ratios in phytoplankton tend to be relatively constant across taxa in different aquatic environments. Redfield found that the C:N:P ratio in phytoplankton approximated 106:16:1, when growth was not limited by nutrient availability. This finding led researchers to examine nutrient limitation by comparing the ratios (stoichiometry) of available nutrients in the environment with the Redfield ratio. The implication here is that departures from the Redfield ratio would indicate whether C, N or P levels were limiting phytoplankton growth. One limitation of this approach is the difficulty in determining the concentrations of nutrients that are available to phytoplankton. For instance, some phytoplankton, such as cyanobacteria, are able to use dissolved organic nutrients for heterotrophic growth (Paerl 1982), although most phytoplankton are not. In addition, phytoplankton are able to use at least two sources of inorganic nitrogen (nitrate and ammonium), though with different efficiencies and uptake kinetics. Furthermore, many phytoplankton are able to carry out "luxury" uptake of phosphorus when available phosphorus is in excess in the environment. This phosphorus can be stored intra-

Bioassay experiments

A bioassay is a screening test on natural phytoplankton assemblages to determine if their growth is limited by the availability of a particular factor such as carbon (C), nitrogen (N) or phosphorus (P). Bioassays may be conducted *in situ* or in the laboratory, however, in the laboratory some attempt is usually made to mimic *in situ* conditions. An index of phytoplankton growth is measured over time, often with and without the addition of nutrients. Typical indices of growth or primary production measured are: (1) changes in biomass (e.g., chlorophyll *a*, cell counts), (2) radioactive inorganic ^{14}C uptake, (3) uptake of N or P into phytoplankton cells, or (4) oxygen production. In addition, metabolic indicators of phytoplankton nutrient limitation are sometimes used to infer nutrient status. Such bioassays measure indicators such as: (1) alkaline phosphatase activity, (2) dark inorganic carbon uptake in the presence of added ammonium, and (3) the cycling rate of phosphorus as determined by radioactive ^{32}P tracer. As phytoplankton are usually held in bottles for bioassays, incubation times are generally one day or less, though they may be longer. The longer the incubation time, the more likely are shifts in community structure or physiology in response to artificial enclosure and the more likely is the exhaustion of the supply of multiple nutrients. Sometimes the measured rates of primary productivity are standardised by phytoplankton biomass to allow the comparison of biomass- or cell-specific rates between different samples and experiments. It is worth noting that different bioassay methods can yield different interpretations of the nutrient status of the phytoplankton in the same water at the same time (e.g., White *et al.* 1985).

cellularly in the form of polyphosphate molecules, which can provide phosphorus at later times. Extra-cellular nutrient pools are thus not always easy to define, nor are they necessarily indicative of instantaneous nutrient availability or limitation.

New Zealand examples of phytoplankton nutrient limitation

In the New Zealand lakes studied so far, limitation due to nitrogen availability appears to be more common than

limitation by available phosphorus (Table 22.1), with Lake Waikaremoana being the only lake not showing periods of limitation by nitrogen. Ten out of 20 lakes showed limitation due to phosphorus levels at times, with Waikaremoana, Okataina, Tikitapu and Rotoma tending to be mostly limited by phosphorus availability. Out of a total of 10 lake waters in which N, P and both N and P were tested, eight showed an enhanced response when both nutrients were supplied, indicating that the stoichiometry of the supply of available N and P approximated the demand stoichiometry. In other words, the addition of

Table 22.1 Phytoplankton nutrient limitation in New Zealand lakes as determined by measurement of a variety of growth indicators in bioassay experiments. Number of sampling occasions per lake ranged from 1 to > 28. N – limitation by inorganic nitrogen levels alone; P – limitation by inorganic P levels alone; N+P – phytoplankton production when N+P added was enhanced beyond that observed with N or P alone; Other – limitation by micro-nutrient availability; None – nutrient additions not significantly different from control; √ – type of nutrient limitation recorded; × – type of limitation never recorded; blanks indicate that the type of limitation was not investigated.

Lake	Times assayed (references)	Type of nutrient limitation observed				
		N	P	N+P	Other	None
North Island						
Taupo*	16 ^{11,13}	√	×		√ (Fe, Mn, §)	√
Rotorua*	28 ^{1,11,12,10}	√	×		√ §	√
Okaro*	4 ¹³	√	×			√
Rerewhakaaitu*	4 ¹³	√	√			×
Okataina*	6 ^{8,13}	√	√	√		√
Tarawera*	6 ^{8,13}	√	√	√		√
Rotehu*	4 ¹³	√	×			√
Okareka*	4 ¹³	√	√			√
Tikitapu*	4 ¹³	√	√			×
Rotoiti*	6 ^{8,13}	√	×	×		√
Rotoma*	4 ¹³	√	√			×
Rotomahana*	4 ¹³	√	×			√
Horowhenua	8 ¹⁴	√	√			√
Waikaremoana	2 ⁸	×	√	√		×
South Island						
Coleridge	1 ²				√ (Co, Mo, Fe, Mn, Zn, B12)	×
Lyndon	1 ²				√ (Co, Mo, Fe, B12)	×
Mahinerangi	3 ^{3,5}	√	×	√		√
Waipori	5 ⁹	√	√	√		√
Waihola	5 ⁹	√	×	√		√
Tuakitoto	2 ⁷	√		–		√
Johnson	1 ⁴	√	×	√		×
Hayes	1 ⁴	√	×	√		×
Tomahawk Lagoon	12 ^{6,7,9}	√	×	×		√
Hawksbury Lagoon	1 ⁷	√				×

* lakes on the central North Island volcanic plateau § combined trace element mixture B12 – vitamin B12

References: ¹ Burns *et al.* (1997), ² Goldman (1964), ³ Mitchell (1971), ⁴ Mitchell and Burns (1981), ⁵ Mitchell and Malthus (1984), ⁶ Mitchell *et al.* (1988), ⁷ Ogilvie and Mitchell (1998), ⁸ Payne *et al.* (1988), ⁹ Spencer (2003), ¹⁰ Vincent (1981), ¹¹ White and Payne (1977), ¹² White and Payne (1978), ¹³ White *et al.* (1985), ¹⁴ White *et al.* (1991).

only one nutrient soon led to limitation by the other. The majority of lakes also exhibited periods when nutrient additions did not stimulate phytoplankton growth. These periods occurred at any time of year, including the growing season, when light and temperature are generally favourable for phytoplankton growth. This is significant for lake management, as changes to nutrient inputs may not affect phytoplankton dynamics at such times. When availability of nitrogen and phosphorus do not limit phytoplankton growth, the availability of micronutrients may be limiting. Only three studies have examined limitation by micronutrient levels in New Zealand lakes. Goldman (1964) found that micronutrients stimulated phytoplankton growth in Lakes Coleridge and Lyndon, while White and Payne (1977, 1978) found that a mixture of trace elements at times also stimulated phytoplankton growth in waters from Lakes Taupo and Rotorua. Given the known trace-element deficiencies of New Zealand soils and waters (Goldman 1964), the potential for micronutrient limitation of phytoplankton growth warrants further study. A number of non-nutrient factors can also limit phytoplankton productivity and some of these are discussed below.

Some of the variation in the nutrient status of phytoplankton in lakes can be caused by temporal variations in the concentrations and stoichiometry (ratios) of nutrients available in the lake. These changes can be due to floods, discharges of sewage, mixing of bottom waters into the euphotic zone, wind-induced sediment resuspension, etc. The variable nutrient status of phytoplankton between, and within, New Zealand lakes has implications for lake and land-use management. The results in Table 22.1 indicate that changes in nutrient inputs to lakes can have variable impacts (including no immediate impact) on phytoplankton productivity and the potential for bloom formation.

Of a number of different nutrient ratios examined, the ratio of concentrations of dissolved inorganic nitrogen : total phosphorus (DIN:TP) in lake water is the most useful index of the nitrogen and phosphorus status of phytoplankton (Morris and Lewis 1988). The concentration of DIN alone is a good predictor of nitrogen limitation (White *et al.* 1985; Morris and Lewis 1988). Important questions remain as to how to use our current understanding of nutrient limitation to mitigate eutrophication and the formation of phytoplankton blooms. Towards this end, it is important to take

into account nutrient dynamics and the potential for controlling nutrient levels, as well as other factors that limit phytoplankton production when nutrient availability does not limit growth.

Non-nutrient factors that affect phytoplankton growth

Apart from nutrients, a number of other factors have been shown to affect phytoplankton productivity in lakes. Photosynthesis depends on light reaching the photosynthetic apparatus of the cell. Therefore, any factors that influence underwater light levels can potentially affect phytoplankton productivity. In thermally stratified lakes, the thickness of the mixed layer and underwater light attenuation by particles and coloured dissolved substances interact to influence the light exposure of phytoplankton. Phytoplankton circulate via turbulence within the mixed layer of lakes and if they are neutrally buoyant and passively circulating, their mean light exposure can be approximated by calculating the mean irradiance in the mixed layer. If this irradiance is greater than the minimum irradiance necessary for net phytoplankton productivity (i.e., where the rate of photosynthesis is greater than the rate of respiration), then phytoplankton growth is possible. The depth in the water column where irradiance is sufficient to balance respiration losses is known as the compensation depth. In Lake Coleridge, seasonal changes in solar radiation and thickness of the mixed layer, as well as changes in turbidity caused by glacial flour, all interact to affect

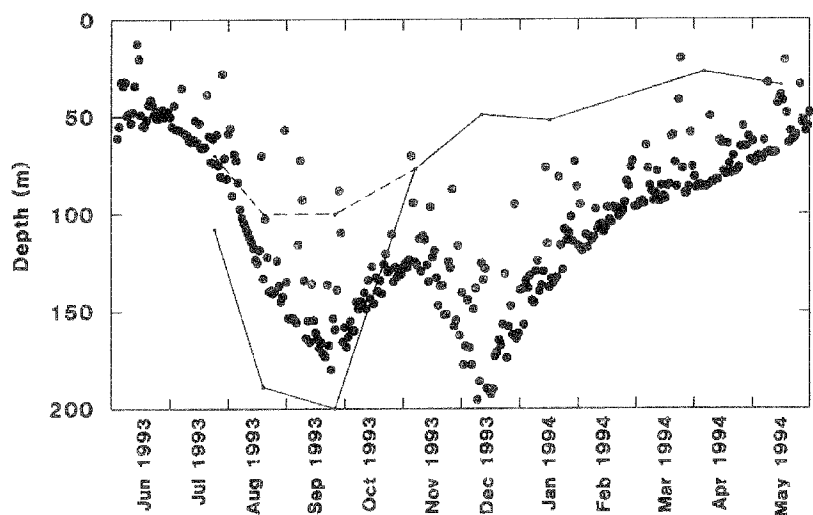


Figure 22.1 The daily compensation depth for phytoplankton net growth (circles) and the water column mixing depth (lines) in Lake Coleridge. The solid line shows the mixing depth at the site of maximum depth (200 m). The dashed line shows the mean mixing depth for the whole lake (mean depth = 98 m). The decrease in compensation depths in October and November was due to a marked reduction in water transparency. From Schallenberg *et al.* (1999).

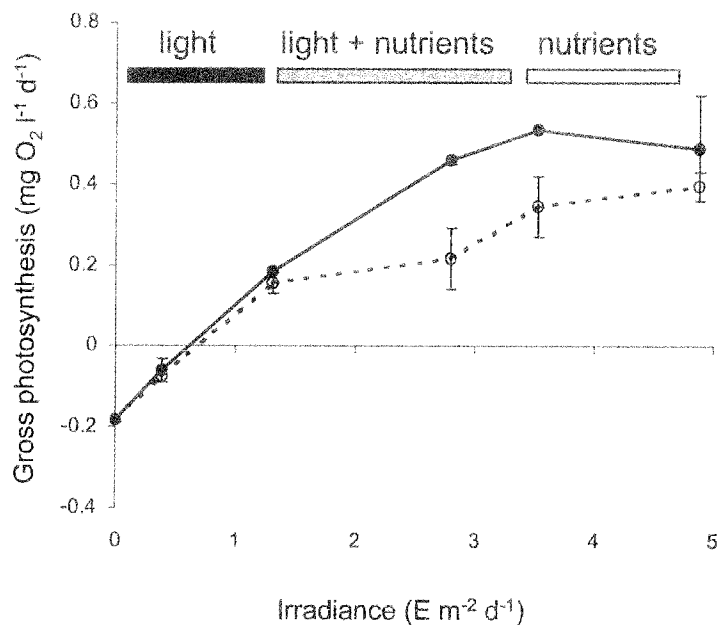


Figure 22.2 Productivity versus irradiance curves for a phytoplankton assemblage from Lake Waihola, showing irradiances where phytoplankton levels are limited by light, by light and nutrients, and by nutrients alone. Closed circles are measurements of oxygen production after N and P were added. Open circles are the controls. Modified from Schallenberg and Burns (2004).

phytoplankton productivity during late winter (Schallenberg *et al.* 1999). Limitation due to levels of available light is likely when the thickness of the mixed layer exceeds the compensation depth, as shown in Figure 22.1.

Generally, nutrient levels will limit growth only when irradiance is sufficient to overcome the light limitation of photosynthesis (usually *c.* 1–6 $E\ m^{-2}\ d^{-1}$, Knowlton and Jones 1996; Carignan and Planas 1994; Schallenberg and Burns *in press*). At light levels within that range, joint limitation by light and nutrients is often observed (Fig. 22.2).

In shallow, polymictic lakes, wind-induced sediment re-suspension causes turbidity that can limit photosynthesis. Strict light limitation due to re-suspension is only likely to occur in Lake Waihola during extreme wind events (Schallenberg and Burns 2004). However, sediment re-suspension also increases concentrations of phytoplankton by entraining benthic algae into the water column, as well as altering nutrient availability by entraining nutrients from the lake bed. Therefore, the effects of sediment re-suspension can be complex.

Another factor that can limit phytoplankton productivity is grazing by zooplankton. If the rate of phytoplankton losses due to zooplankton grazing exceeds the rate of phytoplankton productivity, the grazing may reduce the biomass of phytoplankton. Such effects are

usually caused only by efficient grazers such as the cladoceran *Daphnia carinata* (Mitchell *et al.* 1988; Burns and Dodds 1999), which has high rates of growth and reproduction and can graze plankton of a wide range of sizes (Burns and Schallenberg 1996). While zooplankton can be effective at grazing phytoplankton biomass, they may also, however, stimulate phytoplankton biomass-specific productivity (production rate per unit biomass) by cycling nutrients. This may occur if phytoplankton have taken up all the available nutrients and, as a consequence of this, have a temporarily low ratio of productivity to biomass. By grazing selectively and recycling nutrients, zooplankton can also stimulate both the productivity and biomass of nutrient-limited, inedible phytoplankton taxa.

In theory, phytoplankton productivity in lakes may also be limited by low temperatures. However, acclimation to low temperatures is common in phytoplankton (Davison 1991). For example, the annual peak of phytoplankton productivity in Lakes Taupo and Wakatipu occurs when lake temperatures are near their annual minimum: 10–11°C in Taupo (White *et al.* 1980; Vincent 1983) and 10°C in Lake Wakatipu (Schallenberg and Burns 1997).

However, in lakes where the annual minimum temperature is between 2 and 5°C, productivity tends to be at an annual minimum during winter (Mitchell 1971; Mitchell and Burns 1981).

Phytoplankton populations have been shown to be influenced by viruses, although no such work has been done on New Zealand phytoplankton communities to date. Based on overseas studies, virus infection and mortality has the potential to affect phytoplankton biomass and productivity.

Liebig's Law of the Minimum is a useful starting point for examining the growth of phytoplankton. However, in lakes, potentially growth limiting resources must be considered in broad terms to include light, temperature (heat), and other factors. Furthermore, natural phytoplankton assemblages are composed of numerous taxa, each potentially optimised for different environmental conditions, and adapted to varying environmental conditions. Therefore, the growth responses of phytoplankton communities in lakes, or in bioassays conducted with natural assemblages of phytoplankton, are the integrated responses of a complex biological community. In addition, phytoplankton community growth responses are influenced by grazing pressure. Interpretations of community growth responses in relation to individual taxa or resource availability must therefore be made with some caution.

PHYTOPLANKTON DYNAMICS AND PRODUCTIVITY IN SOME COMMON TYPES OF LAKES

Phytoplankton seasonality and succession

The abundance of phytoplankton as well as the dominance of specific taxa vary over time in lakes. These two factors contribute to successional changes in the community structure of the phytoplankton. Certain patterns in community structure are seasonally driven and recur from year to year in lakes. For instance, diatoms tend to dominate when the stability of the water column is low and turbulent mixing maintains the dense, silicious diatom cells suspended within the water column. For example, the diatom *Aulacoseira granulata* blooms in Lake Taupo during the winter when the water column is isothermal (not thermally stratified) and deep mixing occurs (Vincent 1983). The period of thermal de-stratification and deep mixing also coincided with the annual maxima of the diatoms *Synedra* and *Cyclotella* in Lake Rotongaio (Viner and Kemp 1983). In contrast, some cyanobacteria tend to bloom when the water column is very stable, as they are able to regulate their buoyancy and maintain an optimal depth in the water column when turbulent mixing is low (Burns and Mitchell 1974; Viner and Kemp 1983).

Changes in community structure can have important implications for the lake ecosystem and for water quality. The transfer of energy and nutrients from phytoplankton to the higher trophic levels depends on the consumption of phytoplankton by grazers. Some phytoplankton taxa are particularly nutritious and preferred by grazers, whereas other taxa, such as some colonial cyanobacteria, are not as nutritious (Burns *et al.* 1989; Burns and Xu 1990; Xu and Burns 1991). Furthermore, some cyanobacterial taxa may produce toxins that reduce grazing activity and may harm biota in lakes, and make the water harmful for human ingestion and recreation (Chorus 2001). The ecological and human implications of phytoplankton blooms, therefore, depend on their phytoplankton community structure.

Phytoplankton productivity in large oligotrophic lakes

New Zealand's deep lakes tend to be oligotrophic because of their long hydraulic residence times —e.g., 13 years for Lake Taupo (White and Downes 1977), 4.9 years for Lake Coleridge (Schallenberg *et al.* 1999). The large ratio of volume to inflow of these lakes means that the water typically stays in the lakes for a number of years, and nutrients are stripped out of the water column by plankton, which eventually settle to the bottom sediment, taking the nutrients with them. In lakes with long hydraulic residence times, this self-purification process can

be effective. In such lakes, nutrients may be recycled many times within the epilimnion before they are lost to the hypolimnion via sedimentation. Mineralisation processes in the hypolimnion convert nutrients in sedimenting particulate organic matter into available forms, such as NO_3 . Hence a common feature of deep oligotrophic lakes is that the annual peak in primary productivity occurs after de-stratification of the water column in late autumn or winter, as this process brings regenerated NO_3 into the mixed layer, stimulating phytoplankton production. This pattern is observed in Lake Taupo (Vincent 1983), Lake Coleridge (James *et al.* 2001) and Lake Wakatipu (Schallenberg and Burns 1997). Interestingly, such mixing-driven pulses in productivity are common in many large tropical lakes (Vincent 1983). However, Lake Waikaremoana does not exhibit this pattern and productivity appears to be limited by light levels in winter and by phosphorus levels in general. Productivity in this lake is minimal in winter and is stimulated by the onset of thermal stratification and increasing light levels in spring, a pattern of production typical of temperate warm-monomictic lakes (lakes with one seasonal stratified period) (Vincent 1983).

The deep mixing and low ambient nutrient concentrations that typify many deep oligotrophic lakes tend to encourage phytoplankton dominance by picocyanobacteria (Burns and Stockner 1991; Naismith 1994; Burns and Schallenberg 1998). Picocyanobacteria are extremely small (c. $2\mu\text{m}$ diameter) unicellular cyanobacteria that are more efficient at photosynthesising at low light levels than are eukaryotic algae (Petersen 1991). They are able to utilise dissolved organic nutrients (Paerl 1982) and have a high surface-to-volume ratio, making them effective competitors for nutrients.

Phytoplankton productivity in meso-eutrophic monomictic lakes

A study of phytoplankton productivity in Lakes Hayes and Johnson (Mitchell and Burns 1981) found that seasonal productivity cycles in these lakes fit the classical pattern of productivity in monomictic lakes. Productivity increased from an annual minimum in early-mid winter to reach a maximum in mid summer. Dissolved inorganic nutrients in the water column were already in short supply by the onset of stratification, indicating that nutrient recycling was important in maintaining phytoplankton populations in the epilimnion during the summer stratified period. It appeared that productivity was limited by nitrogen levels in both Lakes Hayes and Johnson, except when the cyanobacterium, *Anabaena*, dominated, as this phytoplankton is able to fix atmospheric nitrogen. Both these lakes had anoxic hypolimnia and accumulated substantial amounts of PO_4 in their bottom waters. As

stratification intensified into late summer and autumn, and nutrients were depleted from the mixed layer, productivity dropped off. In these lakes, productivity was strongly related to mixing processes and to PO_4 accumulation in anoxic hypolimnia, though Mitchell and Burns (1981) noted that zooplankton grazing also had the potential to influence phytoplankton biomass and productivity.

The seasonality of phytoplankton biomass observed in Lakes Hayes and Johnson and the tendency to nitrogen limitation are also common in warm-monomictic North Island lakes (Mitchell and Burns 1981) such as Lake Rotoiti (Fish 1975; Burns *et al.* 1997).

Phytoplankton productivity in shallow, polymictic lakes

Patterns of phytoplankton production in shallow polymictic lakes can be quite different from patterns in monomictic or dimictic lakes. In shallow lakes, there is a large interface between the epilimnion and the lake bed and, therefore, substances and processes in sediments can influence the water column to a much greater extent. This is exacerbated by wind-induced sediment resuspension, which entrains sediments and sediment pore water into the water column when critical shear stresses at the sediment water interface are exceeded (Hamilton and Mitchell 1996). Among substances resuspended during wind events are dissolved and particulate nutrients, organic matter, and algae. Such pulses of sediment-derived materials can result in changes in phytoplankton biomass, productivity, nutrient availability, and light penetration. Therefore, as in the other two lake types discussed here, climatic forcing and resultant mixing greatly influence planktonic production in shallow polymictic lakes.

Shallow lakes tend to be in either a plankton-dominated or macrophyte-dominated state and this state may alternate on time scales of years to decades in some lakes (Mitchell *et al.* 1988; Scheffer *et al.* 1993). It has been suggested that allelopathy (inter-species chemical inhibition) may occur, where macrophytes chemically inhibit phytoplankton production and biomass accumulation, thus permitting sufficient light to penetrate to the macrophyte beds. Much research has been done to elucidate the complex mechanisms underlying alternating states of macrophyte and phytoplankton dominance, as this is an important management problem (Scheffer 1998). Interestingly, from a human perspective, dominance by either phytoplankton (turbid state) or macrophytes (dense macrophyte state) can be perceived as undesirable.

The phytoplankton of shallow lakes in north-temperate regions is often strongly influenced by predator-controlled trophic cascades (Scheffer 1998). The warm summer temperatures and the high productivity of shallow lake

systems appear to promote predator-driven food web interactions. If productive and efficient zooplankton grazers such as *Daphnia* are abundant (i.e., when zooplanktivorous fish are low in abundance), they have the capacity to reduce phytoplankton biomass and productivity (Mitchell *et al.* 1988), increase water clarity, and increase nutrient cycling by grazing heavily on phytoplankton. It has been suggested that such trophic interactions are also important in New Zealand lakes (Jeppesen *et al.* 1997, 2000), although further work is needed to confirm this (see Chapter 23).

Temporal patterns of productivity in shallow lakes are complex, as they depend on some stochastic processes such as wind events and complex food web dynamics. In Tomahawk Lagoon, Mitchell (1971) found that the level of phytoplankton productivity was highly variable from year to year, but exhibited consistent seasonal peaks in summer and periods of low productivity in winter. The year with the lowest phytoplankton productivity corresponded to a year when macrophytes were dominant. The year with the highest productivity occurred when macrophytes were virtually absent but the colonial cyanobacterium *Anabaena* bloomed in the lake. During phytoplankton productivity studies in 2003, when the lagoon was in a turbid state with minimal macrophyte growth, *Anabaena* was again found to be blooming in the summer and phytoplankton growth was not significantly stimulated by nutrient additions (Spencer 2003). Mitchell *et al.* (1988) indicated that *Daphnia* could control phytoplankton at times of high *Daphnia* abundance in Tomahawk Lagoon, however, no *Daphnia* were observed in the 2003 experiments. Nutrient availability did not appear to constrain phytoplankton production in the summer of 2003, indicating that nutrient inputs were balancing, or in excess of, demands of the *Anabaena* bloom.

In tidally influenced Lake Waipori, Mitchell (1971) also found large inter-annual variability in primary productivity. Again the pattern appears to indicate summer productivity peaks, but there was a significant autumn peak in one year and not in the next. Little temporal variability in chlorophyll *a* concentrations was observed over a one-year period in 1997/98 (Schallenberg and Burns 2003). In a series of five bioassay experiments from January to June 2003, phytoplankton were observed to be limited by the availability of N, P, both N and P, and neither nutrient, at different times (Spencer 2003). Phytoplankton productivity in this lake appears to be regulated by a complex set of factors, which vary on different time scales. Phytoplankton dynamics in this lake are complex for a number of reasons. The main non-tidal inflow to the lake is the Waipori River, which flushes the lake on the time scale of days and tends to be low in nutrients, but a highly polluted agricultural drain is intermittently pumped into

the lake. In addition, an average of 72% of the lake volume is displaced in one tidal cycle, indicating a strong tidal influence (Schallenberg *et al.* 2003a).

In Lake Horowhenua, a shallow lake that receives runoff from intensive agriculture and received treated sewage effluent until 1987, phytoplankton showed a clear late summer/autumn peak, with the lake phytoplankton being limited by the levels of available nitrogen in summer and the levels of phosphorus in winter (White *et al.* 1991).

ANTHROPOGENIC IMPACTS ON PHYTOPLANKTON

Eutrophication

The problem of the nutrient enrichment of New Zealand lakes has been studied since the late 1960s. Unlike many countries, New Zealand's population has settled mainly in coastal areas, and only a relatively small human population inhabits lake catchments. Therefore, industrial and domestic sewage inputs tend to be relatively minor contributors to lake eutrophication. Increasingly, sewage discharges to lakes and rivers are being upgraded to secondary or tertiary treatment and being disposed of on land rather than directly into waterways. In New Zealand, agricultural development is the main source of anthropogenic nutrient influx to lakes. Traditionally pastures have been fertilised with phosphorus, as nitrogen-fixing clover was relied upon for nitrogen augmentation of soils. Hence a major source of nutrients to rivers and lakes was via mineral phosphate application to land (Rutherford *et al.* 1987). Recent conversions from sheep stock to dairy farming have altered the traditional scenario and increased the intensity of nutrient flux from land to waters. Dairy-shed effluent disposal directly to waterways was a large problem in many agricultural areas and is now being replaced by effluent disposal on land. However, the success of this procedure in alleviating nutrient inputs to waterways depends on many factors, including soil nutrient retention and exchange capacity, climate, proximity to waterways, and manure application rate (Elliott and Sorrell 2002).

Picoplankton in deep, oligotrophic Lake Wakatipu

The increasing numbers of tourists and tourist development in scenic areas contributes to increased potential for eutrophication, particularly in scenic lakes such as Lakes Taupo and Wakatipu. Such lakes have long water residence times and are therefore slow to respond to increasing nutrient inputs. However, for the same reason, once the lakes have been affected they are slow to respond to restoration efforts such as the reduction of external nutrient inputs. The phytoplankton of such lakes is often dominated by the tiny, non-bloom-forming picocyanobacterium *Synechococcus* sp. Schallenberg and Burns (2001)

studied the potential of these phytoplankters as early-warning indicators of nutrient-related problems in Lake Wakatipu. It was found in bioassays that additions of low concentrations of nutrients did not stimulate picocyanobacterial growth. Rather, low concentrations of phosphorus reduced the abundance of *Synechococcus* cells significantly, a result that had been reported from a few north-temperate lakes. It was suggested that phosphorus additions may have stimulated viral reproduction and lysis (death) of *Synechococcus*, thereby resulting in a decrease in biomass over the two-day experiments. Furthermore, surveys of phytoplankton dynamics in Lake Wakatipu over one year indicated that the abundance and productivity of phytoplankton were only weakly related to total phosphorus concentration, but that the abundance of aggregated *Synechococcus* cells was strongly related to variation in ambient nutrient concentrations and nutrient ratios. While it is not yet clear exactly how nutrient additions influence picoplankton and other algae in deep, glacial lakes, it appears that small changes in nutrient concentrations in Lake Wakatipu affected the dominant phytoplankter in unusual ways, signalling important changes to the lake ecosystem.

Alternative stable states in shallow lakes

Shallow lakes have the potential to oscillate between stable states of macrophyte dominance and plankton dominance (Scheffer 1998). As both states can be stable, there is a theoretical threshold set of conditions that delimit the two states, and a set of feedbacks that operate to buffer against changes in the state (Fig. 22.3). High nutrient inputs to a macrophyte-dominated lake may fuel sufficient phytoplankton production to decrease light penetration, to the extent that macrophytes are unable to grow. This process is enhanced if a shift to cyanobacterial dominance occurs in which the dominant cyanobacterial taxa are able to regulate buoyancy and accumulate in a dense layer just below the lake surface (e.g., *Anabaena* sp.). The death of macrophytes and the mineralisation of macrophyte-derived organic matter adds to nutrient availability, and competition for nutrients decreases once the macrophytes no longer take up nutrients in the system. Mitchell conducted studies on a shallow coastal lagoon (Tomahawk Lagoon), confirming shifts between macrophyte and phytoplankton dominance, with periods of dominance lasting one to five years (Mitchell 1971; Mitchell *et al.* 1988; Mitchell 1988). Both the shading of macrophytes by plankton and allelopathy were found to contribute to the observed dynamics. The water quality of shallow lakes is greatly enhanced during phases of macrophyte dominance. Though the dynamics of alternative stable states in shallow New Zealand lakes are not yet well understood, change in nutrient inputs is potentially an important factor.

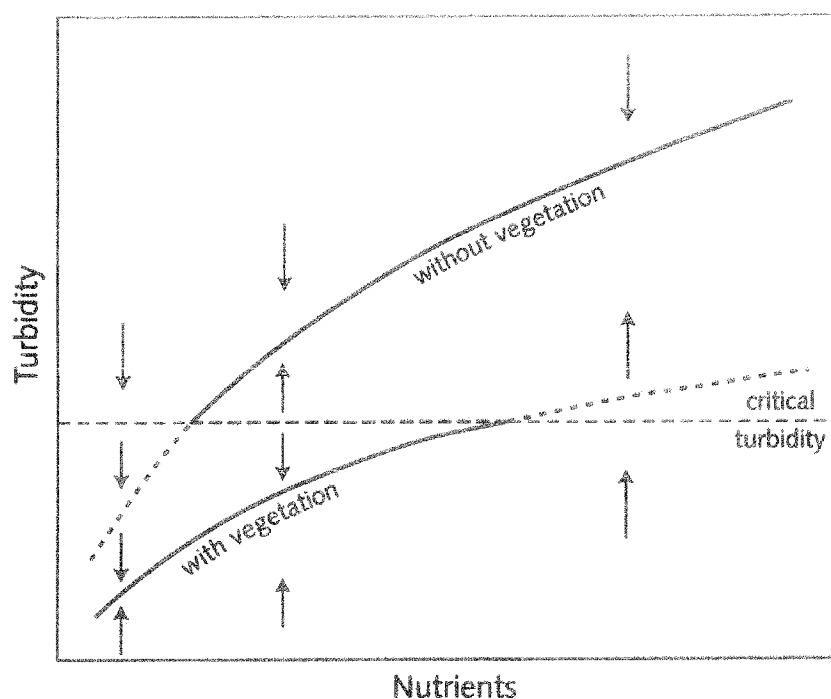


Figure 22.3 Theoretical description of alternative stable states of phytoplankton and macrophyte dominance in a shallow lake, as nutrient levels increase. Macrophyte presence is determined by a critical threshold of algal turbidity, above which light becomes limited and macrophytes are no longer able to survive in the lake. Arrows show the expected responses when the system is not in equilibrium (when it is not tracking the “S” shaped line). From Scheffer (1998)

Cyanobacterial dominance in lakes

Excessive nutrient inputs to lakes can lead to phytoplankton blooms. Although many algal taxa can form blooms (achieve a high biomass of one taxon or a few dominant taxa), cyanobacterial blooms are among the most serious blooms in terms of consequences for lake ecosystems and lake management. Cyanobacteria have a number of characteristics that allow them to compete successfully with algae under certain conditions. They have the ability to control their buoyancy and thereby virtually float on the surface of lakes during the daytime, and many taxa can fix atmospheric nitrogen. Cyanobacteria have a prevalence of colonial forms, which aggregate into colonies too large to be effectively grazed by zooplankton, and they can produce chemical substances that inhibit the grazers (Forsyth *et al.* 1992; Haney *et al.* 1994; Christoffersen and Burns 2001). They have the ability to photosynthesise efficiently at low light levels, due to their production of auxiliary photosynthetic pigments such as phycocyanin, and they can utilise dissolved inorganic carbon at high pH levels.

Certain cyanobacterial taxa can produce potent toxins which may affect wildlife, recreational users of the lake, and people or animals that ingest lake water (Table 22.2).

As yet, little is understood about the triggers that cause cyanobacteria to produce toxins and not all blooms result in toxin production.

Cyanobacterial dominance in lakes has been studied relatively extensively in New Zealand (e.g., Volume 21(3) of the New Zealand Journal of Marine and Freshwater Research was devoted to studies of bloom-forming cyanobacteria in lakes). A number of factors appear to be related to cyanobacterial bloom formation. Blooms tend to occur during warm, calm conditions if nutrients, particularly phosphorus, are also in ample supply. Turbulent mixing interferes with the ability of bloom-forming cyanobacteria to regulate buoyancy and, hence, reduces the likelihood of bloom formation. Many colonial bloom-forming cyanobacteria have the capacity to fix atmospheric nitrogen, though this process is energetically costly. Nitrogen fixation allows cyanobacteria to grow and reproduce under conditions of low NO_3 and NH_4 availability and is carried out in specialised cells within the colonies called

Table 22.2 Cyanobacteria recorded from New Zealand freshwaters that are reported to produce toxins. Asterisks indicate taxa that have formed blooms or scums in New Zealand waters. Data from Ryan *et al.* (2003), Ministry of Health (2001) and Pridmore and Etheredge (1987).

Taxon
<i>Anabaena circinalis</i> *
<i>Anabaena flos-aquae</i> *
<i>Anabaena sp.</i> *
<i>Aphanizomenon flos-aquae</i> *
<i>Aphanizomenon sp.</i>
<i>Cylindrospermopsis raciborskii</i> *
<i>Cylindrospermum sp.</i>
<i>Lyngbia sp.</i>
<i>Microcystis aeruginosa</i> *
<i>Microcystis sp.</i> *
<i>Nodularia spumigena</i> *
<i>Nostoc sp.</i>
<i>Oscillatoria limosa</i>
<i>Oscillatoria sp.</i>
<i>Phormidium sp.</i>

heterocysts. Therefore, it has been suggested and demonstrated in some lakes that low ratios of nitrogen to phosphorus favour cyanobacterial dominance (Smith 1983). N_2 fixation has been shown to be suppressed by additions of inorganic nitrogen in enclosure experiments (Horne and Commins 1987). Burns and Mitchell (1974) found a negative correlation between the density of heterocysts of *Anabaena flos-aquae* and the concentration of inorganic nitrogen in Lake Hayes, supporting previous findings that ambient inorganic nitrogen suppresses the rate of N_2 fixation. Though cyanobacteria tend to dominate under such conditions, they can also dominate when N:P ratios would not seem to indicate limitation due to nitrogen levels.

Nutrient inputs with a low N:P ratio may favour cyanobacteria over algae. Inputs that are likely to contribute to reduced N:P ratios in lakes include domestic sewage (which tends to have a high phosphorus content), runoff from superphosphate-fertilised paddocks, or internal phosphorus sources (sediment re-suspension or diffusion of dissolved inorganic phosphorus following anoxia). Blooms are the results of high rates of phytoplankton production and, therefore, will not occur unless nutrient availability is high, regardless of the N:P ratio of available nutrients. A cyanobacterial bloom may create feedbacks favouring continued cyanobacterial dominance if the senescence and decomposition of cyanobacteria bloom results in anoxic bottom waters, triggering further phosphorus release into the water column. Lakes with such feedbacks may not respond to reductions in external nutrient input, at least until pools of sediment phosphorus are exhausted.

Not only are cyanobacteria potentially toxic to other food web constituents, but nutrient and energy constituents of cyanobacteria may not be exploited effectively by pelagic food webs because of the relative unavailability of many colonial cyanobacteria to grazers. For the many reasons outlined here, cyanobacterial dominance is considered an undesirable state for lake ecosystems.

Climate change

There is now widespread scientific consensus that humans are altering the climate of the Earth. The combustion of carbonaceous fossil fuels, which represent a massive carbon pool sequestered from the biosphere into the lithosphere during the Carboniferous era, is the major contributor to the greenhouse effect, which is warming the planet at an accelerating rate (Intergovernmental Panel on Climate Change 2001).

Changes to the thermal structure of lakes

Records of significant temperature increases over the past decades in epilimnetic and hypolimnetic lake waters are now emerging from around the world. Hypolimnetic

water temperatures appear to be accurate indicators of large-scale and long-term climatic patterns, such as climate warming (Amborsetti and Barbanti 1999). In epilimnetic waters, such trends are more obscured by seasonal and short-term inter-annual variations. Climate warming will tend to increase the stability (resistance to mixing) of thermally stratified water columns. For reasons discussed above, such conditions would be predicted to favour colonial cyanobacteria while being unfavourable to diatoms.

The effects of climate change on the depth of the thermocline and the thickness of the mixed layer will depend on the balance between wind conditions and heating. In addition to a generalised warming trend, predictions of climate change in New Zealand, based on downscaled global circulation models, indicate that global climate change is expected to result in increased El Niño-like weather conditions, entailing dominant westerly and southwesterly weather (Mullan 1996; Ministry for the Environment 2001), generally higher winds, and less precipitation east of the Southern Alps and the North Island Central Plateau. Although substantial uncertainty remains around quantitative predictions of local precipitation and winds, it is likely that global climate change will affect the mixing depths of thermally stratified lakes and therefore affect the light levels experienced by phytoplankton.

Global warming is likely to lengthen the period of thermal stratification in stratified lakes. As hypolimnetic waters are temporarily isolated from the atmosphere during stratification, this will likely result in greater reductions in the oxygen content of hypolimnetic water and higher levels of alkalinity and inorganic carbon accumulation. This may intensify food-web interactions in summer, as the habitat for fish and zooplankton in some lakes will be compressed into a narrow upper layer by the de-oxygenation of bottom waters. Anoxia also results in chemical release of sediment-bound phosphorus, which may contribute to enhanced autumn phytoplankton blooms when thermal stratification breaks down.

There is current concern that a number of factors, including climate change, are contributing to increased stability of the water column of Lake Taupo, such that, if trends continue, Taupo may cease to annually undergo complete mixing in winter (Spigel *et al.* 2003). Among the potential effects of this are reduced mixing depth, higher winter light levels for phytoplankton photosynthesis, de-oxygenation of bottom waters, and higher levels of nutrient accumulation in bottom waters. Such changes may alter the community structure of the phytoplankton, including the winter productivity peak, which has been dominated by *Aulacoseira granulata*. Therefore these changes, if realised, will potentially alter Lake Taupo's water quality, the functioning of its pelagic food web, and its important fishery.

Changes to hydrology and salinity

The predicted decrease in precipitation east of the main ranges, coupled with increasing temperatures and evapotranspiration, will also increase future agricultural demand on water resources. Therefore, less freshwater will be available to recharge some lakes on the east coast of New Zealand. Global climate change is predicted to result in an accelerating rise in global mean sea level due to thermal expansion of the oceans and to the melting of ice sheets. A series of models predicts that by 2100, the global mean sea level should rise between 8 and 88 cm (Intergovernmental Panel on Climate Change 2001). This is potentially substantial, as sea level has risen just 15 cm in the past 100 years at Dunedin, and similar amounts at other New Zealand ports. It is likely then that coastal, tidally-influenced lakes and lagoons will experience increasing salinity from the combination of decreased freshwater inputs and increasing marine influence (Schallenberg *et al.* 2003b). As saline intrusions will become more common and more pronounced, the mean salinity of brackish lakes will increase, and the salinity variations in many of these systems will also increase under this scenario. Salinity is a factor that is overwhelmingly important in structuring the zooplankton community in Lake Waihola, despite the fact that this lake experiences relatively small variations in salinity (e.g., < 5 psu or ppt) and high variations in other water quality variables (Schallenberg and Burns 2003; Schallenberg *et al.* 2003b). For instance, *Daphnia carinata*, a key conduit for energy flow in shallow lake food webs, has a low tolerance for salinity (Hall and Burns 2002; Schallenberg *et al.* 2003b). At the highest range of salinities observed in Lake Waihola, the zooplankton community consists only of the estuarine copepod *Gladioferens pectinatus* and a few rotifer taxa, all in low abundances. This contrasts with the diverse and often abundant zooplankton community (up to 17 taxa) observed during periods of low salinity. Any role that the zooplankton community plays in structuring phytoplankton dynamics in such lakes is strongly influenced by salinity and global climate change. The direct effects of salinity increases on phytoplankton dynamics in New Zealand are not yet well understood, but blooms of toxic cyanobacterium, *Nodularia spumigena*, *Anabaena* sp., and *Microcystis* sp. have been reported from brackish Lakes Ellesmere and Forsyth, indicating that increasing salinity may not reduce the risk of cyanobacterial blooms (Carmichael *et al.* 1988; Christoffersen and Burns 2001).

In regions of New Zealand with a low net water balance, such as in Central Otago where saline lakes already exist (Bayley 1967), evaporative salinity will be expected to increase due to increased evaporation as a result of global climate change.

Stratospheric ozone depletion and UV irradiation

New Zealand, situated at southern mid-latitudes, is susceptible to periods of high solar ultraviolet radiation (UVR), particularly during spring when the Antarctic ozone hole is at its largest. The ecological impacts of UVR are potentially diverse, numerous, and widespread in aquatic ecosystems (Vincent and Roy 1993; Vincent and Neale 2000). Factors related to the exposure of planktonic organisms to UVR include lake altitude, lake depth, and the concentration and composition of dissolved organic matter (DOM). Organisms are protected from UVR damage by the synthesis, or accumulation via food, of UVR-protective pigments such as carotenoids. Copepods of the genus *Boeckella* often show strong carotenoid pigmentation in high-altitude lakes relative to copepods in low-altitude lakes (M. Schallenberg pers. obs.). Though it is not clear whether the copepods synthesise the carotenoids or accumulate them from food, there is likely to be some fitness cost, at least in the form of increased visibility and predation pressure, for copepods that maintain high carotenoid levels.

Dissolved organic matter absorbs UVR strongly and, therefore, reduces the penetration of UVR into aquatic ecosystems in proportion to the concentration of DOM (Rae *et al.* 2001). However, dissolved organic matter is composed of a wide variety of substances (such as fulvic and humic acids, lignins) with varying capacities to absorb UVR and, therefore, the relationship between dissolved organic matter concentrations and relative UVR attenuation is not linear. For example, in humic-stained Lake Brunner, UVR is attenuated almost completely at depths where photosynthetically-active radiation (PAR) is 70% attenuated, whereas in Lake Coleridge, which has low dissolved organic matter concentrations, similar UVR attenuation occurs at depths where PAR is 5% of surface irradiance (Rae *et al.* 2001) (Fig. 22.4). The percent of the lake catchment in forest is positively related to the concentration of dissolved organic carbon in New Zealand lakes (Rae *et al.* 2001).

Increasing UVR creates a feedback in lakes by contributing to the photo-oxidation of dissolved organic matter. In this process, large-molecular-weight dissolved organic matter is photo-oxidised to lower-molecular-weight compounds that are more readily metabolised by aquatic bacteria (Wetzel *et al.* 1995). Hence, increasing UVR reduces UVR attenuation in the water column. Furthermore, it has been demonstrated that global climate change reduces the input of dissolved organic matter from the catchment to some lakes in Canada (Schindler *et al.* 1997), indicating that interactions between global climate change, increasing UVR, and a reduction in UVR attenuation can pose a serious threat to aquatic organisms.

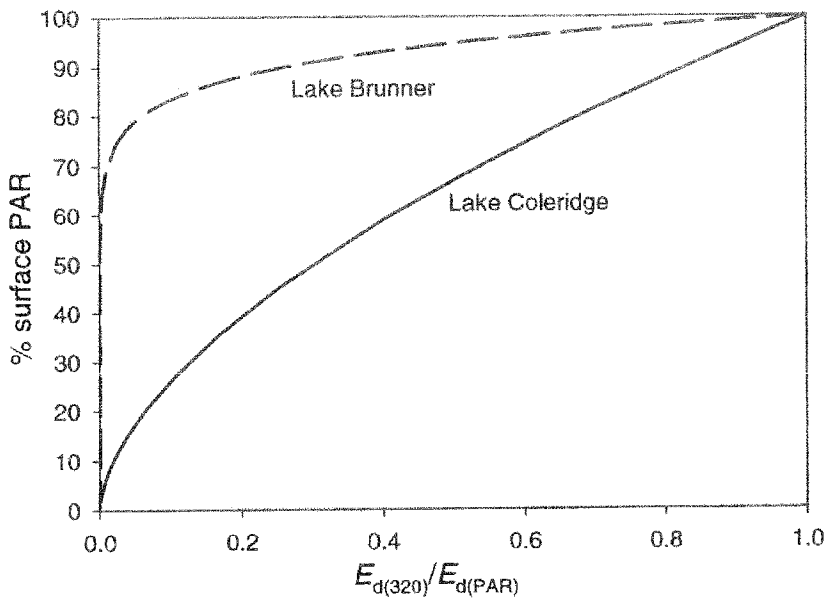


Figure 22.4 The dependence of the ratio of ultraviolet radiation to photosynthetically-active radiation on depth in two lakes with contrasting dissolved organic carbon concentrations: Lake Coleridge (0.4 mg l^{-1}) and Lake Brunner (2.1 mg l^{-1}). Depth is expressed as a percentage of photosynthetically-active radiation at the surface of the lakes to facilitate comparison. From Rae *et al.* (2001).

FUTURE DIRECTIONS

For the most part, the impacts of various stressors on planktonic productivity and nutrient cycling in lakes have been studied individually. Recently, limnologists have been focusing on the interactions between multiple stressors on aquatic ecosystems. In fact most lake ecosystems are simultaneously affected by multiple stressors. Such interactions may induce feedbacks and other non-linear responses in lakes, leading to complex behaviour. For example, recent studies on Lake Taupo are leading to an understanding of how the effects of intensified agricultural that occurred 20 years ago are interacting with climate change, to produce conditions where algal productivity may increase substantially and seriously affect the water quality of the lake (Spigel *et al.* 2003). Furthermore, the intensification of agriculture and the effects of climate change are likely to interact to enhance conditions for cyanobacterial dominance, and possibly cyanobacterial toxin production in many New Zealand lakes. Increasing our understanding of how multiple stressors interact and predicting of the responses of phytoplankton to interacting stressors are two important challenges that lie ahead for limnological research in New Zealand.

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Chapter 21

Physical and chemical characteristics of lake water

David Hamilton, Ian Hawes and Rob Davies-Colley

INTRODUCTION

In this chapter we review the major processes influencing the physical and chemical nature of lake water. Previous reviews of New Zealand lakes (Spigel and Viner 1992) and their physics (Green *et al.* 1987; Hoare and Spigel 1987), chemistry (Timperley 1987) and colour and clarity (Davies-Colley 1987) provide a sound theoretical and practical basis for this chapter. Here we continue the theme of physical and chemical limnology of New Zealand lakes, supplemented by reviews of lake hydrodynamics (Imberger and Patterson 1990; Imberger 1998; Wüest and Lorke 2003) and water clarity (Davies-Colley and Smith 2001).

New Zealand lakes cover an area of c. 3,400 km², or about 1.3% of the total land area. Approximately 70% of these lakes are very small (with an area of less than 0.5 km²), although large lakes dominate volumetrically. The deepest is Lake Hauroko (462 m) and the largest is Lake Taupo (606 km²). A wide variety of processes have formed New Zealand lakes, reflecting the diverse and complex array of geological processes that have shaped the landscape (Lowe and Green 1987). Patterns of lake mixing, however, are more uniform, as a result of the temperate maritime climate that extends over the full latitudinal range of the country, from c. 34°S to 47°S.

LAKE HYDROLOGY AND MORPHOMETRY

Inflows to lakes arise from surface water, groundwater and rainfall, and these transport sediment, nutrients, and both living and non-living organic matter. Rates of water movement (*advection*) in lakes are slow compared with rates in streams and rivers, and this provides opportunities for water composition to change substantially between inflows and outflows, as well as allowing large variations within different parts of a lake. One of the most important variations results from vertical *density stratification*, which

limits vertical circulation for much of the year in all but shallow lakes. Variations in density arise most commonly from gradients in water temperature, with solar heating and convective cooling, though intrusions of brackish water or sediment-laden inflows may occasionally influence the vertical density structure in a lake.

The mean water *residence time* (in days) of a lake may be expressed as:

$$\tau = V/Q \quad 21.1$$

where Q is the outflow discharge (m³/day) and V is the lake volume (m³). τ does not necessarily measure the duration that different inflows are retained within a lake, especially when they are short-circuited to an outflow because of stratification or the proximity of the inflow and outflow (e.g., Lake Rotoiti; Gibbs 1986). τ is inversely proportional to the mean *flushing rate* (1/day). Values of τ may range from several hours to a few days, in the case of reservoirs behind small dams along large river systems, to several years where the lake catchment is small relative to lake volume. For example, Roxburgh Dam, on the Clutha River, has a water residence time of the order of one day, while Lake Taupo has a water residence time of c. 11 years. Calculations of water residence time generally necessitate compilation of a water balance, which may also be used to assess, for example, water level changes due to withdrawal of water for power generation, irrigation or water supply. The water balance is expressed as:

$$\frac{dV}{dt} = \Sigma Q_{in} - \Sigma Q_{out} + Q_{gw,in} - Q_{gw,out} + (P - E)A_s \quad 21.2$$

where t is time, ΣQ_{in} and ΣQ_{out} are the sum of all surface inflows and outflows respectively, $Q_{gw,in}$ and $Q_{gw,out}$ are groundwater inflows and outflows respectively, and P and E are precipitation and evaporation (in m/day), respectively, over the lake surface of area A_s (in m²). The term A_s

varies with water level and can be determined from the area-depth relationship for a lake; the *hypsographic curve*. It follows that lake volume is determined by integration of lake area over successive depth contours, allowing average lake depth to be determined as V/A .

The reduced through-flow and relatively long residence times in lakes, in comparison with flowing waters, means that lakes act as traps for incoming (catchment-derived) inorganic solids and organic solids (*allochthonous production*), as well as organic solids produced in the water column (*autochthonous production*). The rate of sedimentation in New Zealand lakes is highly variable (2 to >200 mm/yr: Lowe and Green 1987) but comparatively high on a global scale, as a result of steep hill slopes, readily eroded and young soils, moderate to high rainfall, and high rates of livestock grazing. High sedimentation rates have sometimes necessitated remedial actions, such as at Roxburgh Dam, where the estimated sedimentation rate is 300 mm/yr (Spigel and Viner 1992). Water levels in the reservoir were temporarily reduced in an attempt to produce river scouring of the lakebed.

LAKE FORMATION

New Zealand lakes have been formed in an active geological environment and most are comparatively young

(< 20,000 years B.P.). The major mechanisms for lake formation have been summarised by Lowe and Green (1987) (Table 21.1). The ten categories of lake formation tend to fall into broad geographic regions (Lowe and Green 1987), but in many cases the formation of individual lakes can be traced to the interaction of one or more processes. The glacial lakes of the South Island are most numerous (c. 38%) and were formed during Pleistocene glaciations. Lakes of volcanic origin occur predominantly in the Taupo Volcanic Zone, but New Zealand also has a large number of coastal systems—around 7.4 lakes, wetlands and lagoons per 100 km of coastline (Schallenberg *et al.* 2003a). Large lakes used for hydroelectric power generation are another important landscape feature and economic resource for New Zealand, and occur predominantly along the flow-regulated Waikato, Clutha and Waitaki Rivers.

LAKE PHYSICS

Climate

New Zealand's climate can be categorised as temperate maritime, with relatively low-amplitude seasonal temperature fluctuations, but higher average wind speeds than in most other temperate areas. According to the classification scheme of Lewis (2000), lakes at >40° latitude should be *dimictic* (two periods of complete vertical cir-

Table 21.1. Classification scheme for New Zealand lakes (adapted from Lowe and Green 1987), mechanisms of formation and examples of each lake type.

Lake type	Formation	Examples
Tectonic lakes	Movements of the earth's crust	Hatuma, Runanga, Brunner
Volcanic lakes	Crater lakes, caldera collapses and lava flow lakes	Taupo, Tarawera, Rotorua
Glacial lakes	Damming by depositional moraines and filling of ice-evacuated depressions	Wakatipu, Manapouri, Coleridge
Landslide lakes	Valley blockage by debris	Waikaremoana, Buller,
Phytogenic basin lakes	Depressions associated with peat	Pools in the Kopuatai Peat Dome and Ahukawakawa Swamp
Riverine lakes	Isolation from river channels	Waikare, Wairarapa, Waipori
Dune lakes	Wind-blown sand deposits creating depressions or blocking valleys	Kai-iwi lakes, Taharoa, Heaton
Barrier-bar lakes	Enclosure of inlets by wave action, tectonic movements or sea-level change	Ellesmere, Wainono Lagoon, Pounui
Karst lakes	Karst landscape depressions or dissolution	Disappear, Koraha
Man-made reservoirs	Hydroelectricity, water supply, irrigation, recreation	Benmore Dam, Clyde Dam, Lake Karapiro

culation per year), yet no such lake has yet been described in New Zealand. This anomaly reflects New Zealand's windy temperate climate, in which lakes mix more deeply than those of continental areas and are not subject to calm cold winters or especially hot summers (Davies-Colley 1988a), although some shallow, high-altitude lakes freeze over in winter. The predominant mixing pattern in deep New Zealand lakes is *monomictic* (one vertical circulation period per year) or permanently mixed to *polymictic* (frequently alternating periods of mixing and stratification) in shallow lakes. Interestingly, however, Lake Taupo has deviated from monomictic behaviour and remained vertically stratified (i.e., *amictic*) in exceptionally warm winters, a phenomenon that is expected to increase with projected global warming (Spigel *et al.* 2003).

Mixing and stratification

Vertical gradients of density in some lakes may be associated with periodic intrusions of seawater (with a salinity ~ 35 practical salinity units, and a density $\rho \sim 1028 \text{ kg/m}^3$) or brackish water beneath buoyant freshwater ($\rho \sim 1000 \text{ kg/m}^3$). These intrusions occur mostly in shallow barrier-bar lakes or occasionally in lakes with an outlet open to the ocean, but vertical density gradients are rapidly destroyed by wind (e.g., Lake Waiholo: Schallenberg *et al.* 2003a). Medium to long-term projected sea-level rises may, however, increase the frequency of saline intrusions and could therefore change some coastal lakes into brackish lagoons or estuaries, while increasing the frequency of saline intrusions in others (Schallenberg *et al.* 2003b).

Seasonal temperature stratification occurs in spring, as surface waters warm due to increased radiation and reduced conductive and convective heat losses to the atmosphere. Stratification is characterised by a surface mixing layer, the *epilimnion*, separated from deep waters of the *hypolimnion* by a region of steep density gradient, the *metalimnion*, the centre of which is the *thermocline*. The following criteria may be used to define the position of the thermocline (e.g., Hoare and Spigel 1987):

$$d\rho/dz = \text{minimum} \quad 21.3$$

$$d^2\rho/dz^2 = 0 \quad 21.4$$

where z is depth (m). For 22 New Zealand lakes, Davies-Colley (1988a) found that the epilimnion depth is closely related ($r^2 = 0.94$) to a power transformation of average *wind fetch*, the distance over water that wind blows, where the fetch, F (m), is approximated as the square root of lake

surface area. The idealised description of lake stratification given above often varies on a daily time scale, due to the formation of a diurnal thermocline at shallower depths in the epilimnion, when solar radiation causes rapid daytime heating of the surface waters. This thin surface layer is subsequently broken down by wind or by night-time cooling.

Figure 21.1 compares the stratification regimes of two Rotorua lakes: Lake Rotoiti (area 34 km^2 , mean depth 33 m) and Lake Rotoehu (area 8.1 km^2 , mean depth 8.3 m). Over the depth range corresponding to the shallower lake, the variations in temperature of the two lakes are very similar, though shallow Lake Rotoehu shows a slightly more rapid and greater magnitude of response to seasonal climatic variations. There is slight warming ($1\text{--}2^\circ\text{C}$) in the bottom waters of Lake Rotoiti through the stratified period, which is contributed by geothermal heating (Gibbs 1986). By comparison, Lake Hayes in Central Otago (area 2 km^2 , mean depth 18.7 m) has similar surface water temperatures in summer to the lakes shown in Figure 21.1, but is $3\text{--}5^\circ\text{C}$ cooler than Lake Rotoiti in winter and also in the hypolimnion during summer (Mitchell and Burns 1979).

Wind acting on the water surface creates a surface shear stress τ (N/m^2), the force per unit area, given by:

$$\tau = \rho_A C_{10} U_{10}^2 \quad 21.5$$

where ρ_A is the density of air (kg/m^3), C_{10} is a dimensionless wind drag coefficient and U_{10} is the wind speed 10 m above the water surface (m/s). The value of C_{10} varies with wind speed because part of τ is directed to a wave stress, to accelerate and maintain surface waves, which vary

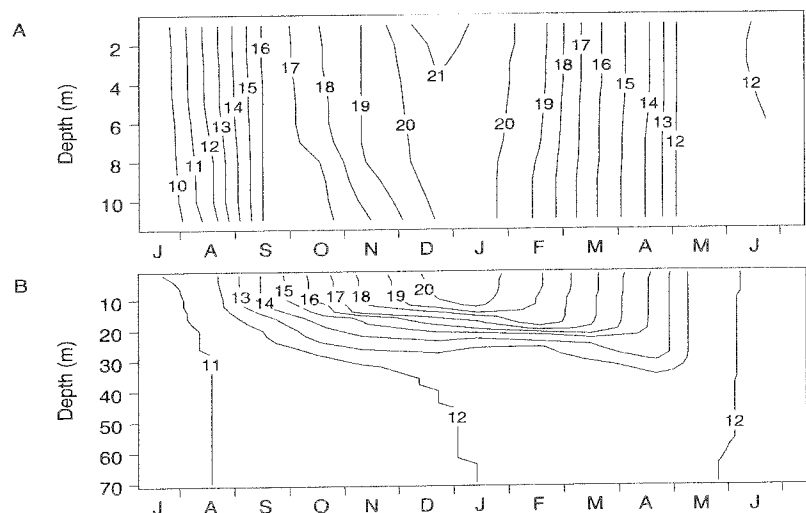


Figure 21.1 Contour plots of temperature isotherms in A) Lake Rotoehu and B) Lake Rotoiti, 2001-2. Note the different vertical (depth) scales in the two plots. Data from Environment Bay of Plenty.

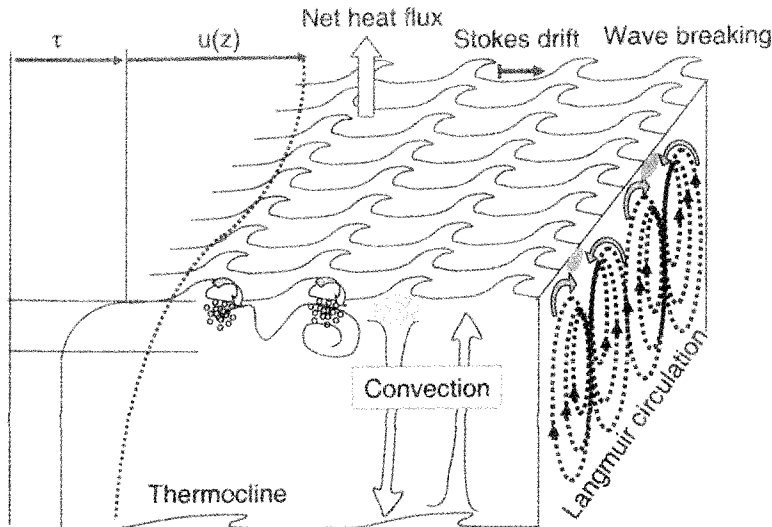


Figure 21.2 Schematic of processes occurring in the surface mixing layer of a lake (modified from Wüest and Lorke 2003). Dissipation of turbulent kinetic energy is represented by the curve associated with surface waves breaking and entraining gases. The net heat flux is negative, representing net loss of heat via convection. τ is the shear stress, which consists of an applied stress from air at the water surface, a non-linear wave component in the upper water layer, and a linear momentum component in the lower layer. $u(z)$ is the depth-varying air or water (wave orbital) velocity. Both τ and $u(z)$ are referenced vertically (x - y axis).

in magnitude with the wind, and part is directed into generating currents and associated momentum beneath the waves. An interesting and complex feature of wind acting on a water surface is Langmuir circulation (Fig. 21.2), a coherent, near-surface circular motion oriented parallel to the wind direction (Langmuir 1938). This circulation pattern is often observed on windy days at the water surface when the paired elliptical water motions converge to form parallel streaks of foam, algae and other floating material.

Inputs of momentum from the wind, as well as convective cooling, create turbulent kinetic energy that eventually destroys the seasonal stratification, in early winter. This transition may be usefully quantified by changes in the *Schmidt stability* S (see Idso 1973), a value that measures the resistance of a lake to mixing:

$$S_t = \frac{g}{A} \int_0^{h_m} (h - h_g) A(h) \rho(h) dh \quad 21.6$$

where $A(h)$ is the area of the lake at elevation h (m), A is the surface area, h_m is the surface water elevation, and h_g is the elevation of the centre of volume of the lake, defined by:

$$h_g = \frac{\int_0^{h_m} h A(h) dh}{\int_0^{h_m} A(h) dh} \quad 21.7$$

Prior to complete mixing (turnover), there may be large downward displacements of the metalimnion at the leeward end of a lake. This creates internal waves in the metalimnion to counter-balance the arrest of water at the leeward boundary. These waves are initiated at the basin scale and may have frequencies of a day or more, but they also interact and degenerate to produce smaller internal wave motions of progressively increasing frequency (i.e., hours to minutes). In deep Lake Coleridge (mean depth 98 m), internal wave excursions exceed 40 m in height near the middle of the lake and may be twice as large near the lake boundaries (Schallenberg *et al.* 1999). The Coriolis effect may also play a role in deflecting internal waves, as well as other basin-scale water motions, in a counter-clockwise direction in the southern hemisphere. The region where internal waves interact with the lake bed is the *benthic boundary layer*, where rates of vertical mass flux may be an order of magnitude or more higher than in the central lake basin (MacIntyre *et al.* 1999). The mixing and displacement of the metalimnion in the

near-shore region are important in the resupply of nutrients and substrates to the phytoplankton and bacteria that are often concentrated in the metalimnion (i.e., a “deep chlorophyll maximum”). The mixing regime in a stratified lake may therefore be partitioned into turbulent surface and benthic boundary layers, and a relatively quiescent interior part of the lake.

The extent of displacement of the metalimnion can be quantified by the Wedderburn number W (Spigel and Imberger 1987):

$$W = \frac{\Delta \rho g h^2}{\rho_0 u_*^2 L} \quad 21.8$$

where ρ_0 is a reference density for water, $\Delta \rho$ is the density difference between the surface mixing layer and the metalimnion, L (m) is lake length at the metalimnion in the direction of the wind, u_* (m/s) is the water friction velocity due to wind stress, and h (m) is surface layer thickness. For $W \leq 1$ the uppermost thermocline will reach the surface at the upwind end of the lake and metalimnetic water will be entrained into the surface mixing layer, causing a rapid increase in mixing layer depth. Otherwise ($W > 1$), the mixing layer will deepen only very slowly.

In a stratified fluid the vertical mass flux is highly important, e.g., in supplying nutrient-rich deep water to the surface (sunlit) layer where nutrients may be in short

supply for primary production. This flux is defined through the rate of *turbulent diffusion* K_d (m^2/s), which depends on the mixing efficiency γ_{mix} , the rate of dissipation of the turbulent kinetic energy ε (W/kg) and the local water column stability given by the buoyancy frequency N ($1/\text{s}$):

$$K_d = \frac{\gamma_{mix}\varepsilon}{N^2} \quad 21.9$$

where N^2 is defined as:

$$N^2 = \left(\frac{-g}{\Delta\rho} \right) \left(\frac{\partial\rho}{\partial z} \right) \quad 21.10$$

where g (m/s^2) is gravitational acceleration. Values of N^2 increase with increasing stratification, and may vary substantially with depth in a stratified lake, or by as much as nine orders of magnitude between lakes. N^2 may be measured with CTD profiles (conductivity, temperature and depth) and γ_{mix} may be estimated from physical processes (Ivey and Imberger 1991) and from lake properties such as morphometry and stratification (Gloor *et al.* 2000). It has been found that around 90% of ε is dissipated in the benthic boundary layer, whilst most of the rest is dissipated in the lake interior (Wüest and Lorke 2003).

Surface heat fluxes and momentum

The factors that influence the heating, cooling and vertical stratification of lakes include surface and bottom heat fluxes, wind, water clarity, inflows, outflows and *morphometry* (lake size and shape). Electro-magnetic radiation energy fluxes can be divided into shortwave (0.3–3 μm) solar radiation inputs, and long-wave emission through thermal infra-red (3–100 μm) back-radiation. The near-infra-red waveband of short-wave solar radiation (0.7–3 μm), about 50% of total solar radiation, is absorbed by water molecules within the top few millimetres of the water surface. Visible and ultra-violet (UV) radiation (0.3–0.7 μm) penetrates more deeply, to an extent depending on the concentration and nature of dissolved and particulate material in the water column (see section below, “Light Penetration”).

Other surface heat fluxes include conductive (sensible) heat exchange and evaporative (latent) heat loss. A report by the Tennessee Valley Authority (1972) still remains one of the most detailed and authoritative accounts of heat fluxes in lakes and is used as a basis for prescribing heat fluxes in detailed models of lake hydrodynamics such as DYRESM (Hamilton and Schladow 1997), which has been applied to lakes Rotorua (Rutherford *et al.* 1996) and Taupo (Spigel *et al.* 2003).

Inflows

A river that enters a lake will initially push lake water ahead of itself. If the river inflow is cold, and denser than the surface lake water, it will plunge beneath the surface, or if it is warm and buoyant, it forms a thin surface overflow. Inflows into well-mixed lakes or overflows into stratified lakes are rapidly entrained into the water column or surface mixing layer, respectively. Nutrient-laden surface overflows may, however, be very important to the primary production of stratified lakes as they insert nutrients directly into the surface mixing layer.

Plunging river inflows entrain lake water as they flow downward along the bed of a lake. When the Ohau Channel flow from Lake Rotorua to Lake Rotoiti is colder than surface waters of Lake Rotoiti, it plunges and can eventually be more than three times larger in volume than where the inflow first enters the lake (Gibbs 1986). The plunging inflow is also an important source of oxygen for the hypolimnion (Vincent *et al.* 1991) and is one of the factors that complicates estimates of oxygen deficit in the hypolimnion (Vant 1987; Burns 1995). Plunging inflows propagate to the bottom of a lake or intrude horizontally at a point of neutral buoyancy. These intrusions may occur as an interflow at the thermocline, where large vertical density gradients retain the inflow within a relatively discrete depth range, though diurnal variations in inflow temperature also act to disperse it.

Outflows

Most natural lakes have surface outflows, but in a large number of man-made lakes, water is withdrawn from one or more discrete depths. In stratified lakes with subsurface outlets, most of the outflow comes from horizontal transfer of water to the outlet. The strength of stratification and the discharge rate, however, also influence the amount of water transferred from elevations above a subsurface outlet (Hocking *et al.* 1988).

Wave action and sediment re-suspension

Surface waves result from wind blowing over a lake surface. Hamilton and Mitchell (1996, 1997) showed that these waves increase water column concentrations of suspended sediments and nutrients in shallow South Island lakes by re-suspending material from the lake bed, particularly when the sheltering effects of macrophytes are reduced or absent. In very shallow Hawksbury Lagoon (Otago), the large variations in light attenuation resulting from re-suspended sediments are closely related to wind speed (Fig. 21.3). Lake Ellesmere (area 186 km^2 , mean depth 1.7 m) lost its dense macrophyte bed in the Wahine storm of April 1968 and has since had permanently high concentrations of inorganic suspended solids, while

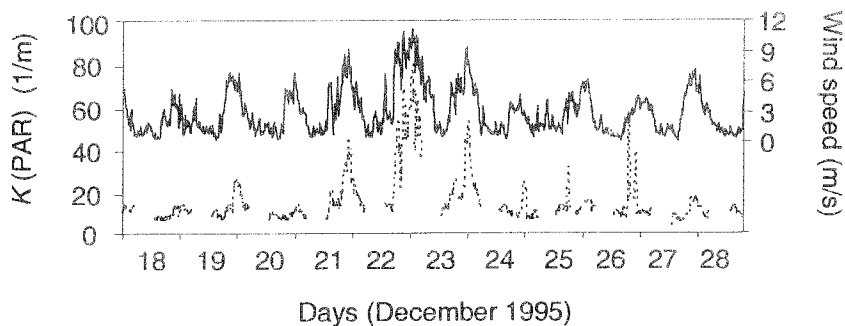


Figure 21.3 Time series of the attenuation coefficient (dashed line) for photosynthetically available radiation, $K(\text{PAR})$, (left-hand vertical axis) and wind speed (bold line, right-hand vertical axis) for Hawksbury Lagoon, Otago. (Authors' unpublished data.)

Tomahawk Lagoon No. 2 alternates between high and low water clarity (Mitchell *et al.* 1988). Wave action prevents the re-establishment of macrophytes in Lake Ellesmere, either by direct erosive effects, or indirectly, through unfavourable light levels (Gerbeaux 1993). Permanent or temporary switches to low water clarity have occurred in several other shallow New Zealand lakes (e.g., Waahi, Waikare, Omāpere).

A brief outline is given here of the dominant processes that affect sediment re-suspension by waves. The net effect of sedimentation and re-suspension on the concentration of sediment particles in the water column is given by:

$$\frac{dC}{dt} = \frac{R}{h} - C \frac{v}{h} \quad 21.11$$

where dC/dt is the rate of change of suspended-sediment concentration C (g/m^3) with time t (s), h is water depth (m), v is the sedimentation rate (m/s) and R is the re-suspension rate ($\text{g}/\text{m}^2/\text{s}$).

The sedimentation rate of a spherical particle at steady state is given by Stokes Law:

$$v = \frac{g(\rho - \rho_0)D^2}{18\eta} \quad 21.12$$

where η is the dynamic viscosity of water ($\text{N}\cdot\text{s}/\text{m}^2$), D is particle diameter (m), g is gravitational acceleration ($9.81 \text{ m}/\text{s}^2$), and ρ and ρ_0 are the densities of the particle and water (kg/m^3), respectively. An additional parameter, the form resistance, may be included in Eq. 21.12 to account for particle shape (Reynolds 1992). The re-suspension rate is commonly given as (Hamilton and Mitchell 1996):

$$R = \alpha \left[\frac{\tau_{bed} - \tau_c}{\tau_{ref}} \right] \quad 21.13$$

where α is the re-suspension rate constant ($\text{g}/\text{m}^2/\text{s}$) and τ_{bed} , τ_c and τ_{ref} are the measured or estimated bottom shear stress (N/m^2), the critical shear stress for initiation of sediment re-suspension (N/m^2), and a reference shear stress set to $1 \text{ N}/\text{m}^2$, respectively. Surface waves interact directly with bottom sediments in the littoral zone of deep lakes, or across shallow lakes when the depth of water is less than half the wavelength (Carper and Bachmann 1984). Surface wave-induced currents also affect bottom shear stress, but are generally too small in magnitude to entrain sediment in most lakes.

Acoustic doppler current meters may be used to take vertical profiles of water velocity or to measure it at a discrete point, so that bottom shear stress can then be estimated. Most commonly, however, bottom velocity is determined indirectly from estimates of the bottom orbital amplitude A_{bed} (m) and velocity U_{bed} (m/s) of water under the influence of wave action. Both A_{bed} and U_{bed} can be calculated from surface wave characteristics, applying linear (Airy) wave theory. The relevant surface wave characteristics, determined from the wind speed, fetch and water depth, include the significant wave height, the average height of the highest one-third of the waves (crest to trough), wave period and wavelength (CERC 1984). Figure 21.4 shows the critical roles of lake size (as it affects fetch) and water depth in generating bottom shear stress for two wind speeds, where the shear stress was estimated from a simple computer model based on empirical wave formulae and linear wave theory (CERC 1984).

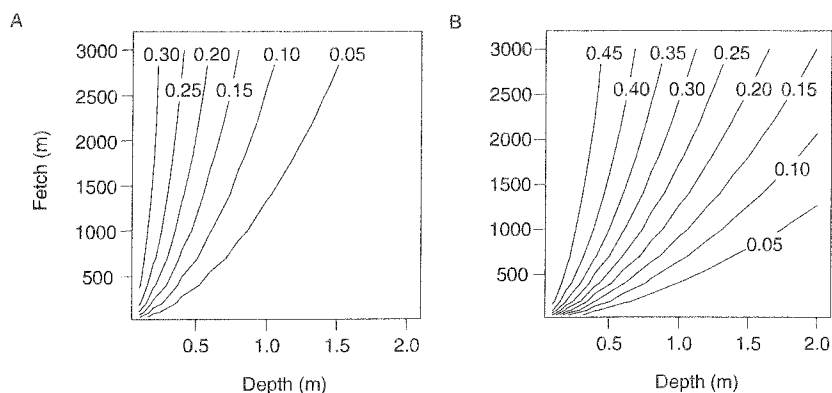


Figure 21.4 Contour lines of bottom shear stress (N/m^2) estimated from water depth, wind fetch and wind speeds of A) $4 \text{ m}/\text{s}$ and B) $8 \text{ m}/\text{s}$. (Authors' unpublished data.)

LIGHT IN LAKES

The science of light behaviour is called optics and here we give a very brief introduction to the optics of lake waters. The optical character of lake waters is of great importance to their ecology because it determines the amount of light present at different depths for plant growth and for vision of aquatic animals (Kirk 1994a). The distance animals (and people) can see through water is also very important, and the visual clarity and colour also affect aesthetic quality (and thus the amenity value) of lakes (Davies-Colley *et al.* 1993). An understanding of the visual characteristics of water is deceptively simple—while the distance to which light penetrates in water and how far one can see an object are affected by similar variables, they are only weakly related, and then through very complex mechanisms.

The two most important optical processes in water are absorption, involving the conversion of light energy to another form, ultimately heat or chemical energy through photosynthesis, and scattering, in which light photons change direction without a change in energy (Davies-Colley *et al.* 1993; Kirk 1994a). The processes of absorption and scattering together attenuate light, and constituents of water that absorb or scatter light are referred to as light-attenuating constituents. Absorption is quantified by an *absorption coefficient* a (1/m), which may be conceptualised as the probability of absorption of a photon per unit length of light path. Scattering is similarly quantified by a *scattering coefficient* b (1/m). These two *inherent optical properties* sum to a third inherent optical property, the *beam attenuation coefficient* c ($= a + b$) (Kirk 1994a).

Light-attenuating constituents

There are five main light-attenuating constituents of lake water, in common with most other natural surface waters (Table 21.2). Pure water is not really the colourless compound of elementary chemistry, but is pale blue-green coloured because it selectively absorbs red wavelengths of light. In deep water with little attenuation by substances

other than water—for example in Lake Wakatipu or Lake Coleridge—the blue-green colour due to red light absorption by water itself can be readily observed. Besides absorbing light, water also scatters light, rather weakly and with strong wavelength selectivity. In very pure waters the wavelength selectivity of scattering by molecular water contributes noticeably to colour, and accounts for the blue-violet hue of tropical seawater, spring waters (notably Waikoropupu Springs in Nelson; Davies-Colley and Smith, 1995), and a few remarkably clear lake waters such as Crater Lake, Oregon (Davies-Colley *et al.* 1993).

The weak scattering of pure water is swamped by the scattering by suspended particles in all but the very clearest lake water, however the influence of the spectral absorption of pure water on water colour is only masked by that of other constituents in highly light-attenuating waters.

Typically only one dissolved constituent of lake waters contributes appreciably to light attenuation. A large fraction of the dissolved organic matter in natural waters is chromophoric (coloured) dissolved organic matter (CDOM), mostly “humic” in character, similar to a water extract of soil humus. This material is often referred to as “yellow substance” (Gelbstoffe in German, or gilvin; Kirk 1976) when its optical character is the main concern. It absorbs light strongly in an approximately exponential spectral pattern with declining wavelength (Davies-Colley and Vant 1987):

$$g(\lambda) = g_L \exp[S(L - \lambda)] \quad 21.14$$

where g is the absorption coefficient (1/m) (denoting gilvin; Kirk 1976), λ (m) is wavelength, L is a reference wavelength (m), g_L is the value of $g(\lambda)$ at $\lambda = L$, and S (1/nm) is a parameter (the spectral slope) characterising the steepness of the absorption spectrum. Yellow substance is strongly light-absorbing in the blue region of the spectrum and so tends to impart a yellow hue to waters—hence its name. So-called “dystrophic” lakes (those rich in humic material) have a very high concentration of yellow substance and correspondingly high light attenuation that restricts light penetration and may therefore inhibit algal

Table 21.2 Light-attenuating constituents of lake waters.

Constituent	General characteristics	Absorption	Scattering
Water itself	Pure water	Strong in red region	Weak, often negligible
Yellow substance	CDOM, mostly humic	Strong in blue region	Negligible
Phytoplankton	2–100 μm in size, heavily pigmented	Strong due to pigments (blue and red regions)	Strong
Detritus	Particulate organic matter	Strong in blue region	Strong
Inorganic solids	Clay minerals, silt particles	Low	Strong

growth. The scattering of light by yellow substance is negligible.

All other light-attenuating constituents of importance in lake waters are particulate materials (Table 21.2). Phytoplankton cells are typically strongly light-scattering and are generally close to the size of 5 μm , where light scattering by organic particles is maximal (Davies-Colley and Smith 2001). This explains why eutrophic lakes laden with algae have low visual clarity. Phytoplankton are also noteworthy for their highly spectrally-selective absorption of blue and red light by chlorophyll-*a* and other pigments, which accounts for the green hue of eutrophic waters. Smaller organic particles, notably bacterial cells, are often present at much higher concentrations than phytoplankton in lakes, but contribute comparatively weakly to light attenuation, owing to their small size and lack of pigmentation.

Non-living organic particulate material, known as detritus (tripton: Kirk 1994a), is more heterogeneous than phytoplankton, and may also attenuate light strongly by both scattering and absorption. The absorption spectrum of detritus is often similar to that of yellow substance, in being featureless except for an exponential rise with declining wavelength (Eq. 21.14). However, unlike yellow substance, detritus also scatters light appreciably.

Finally, inorganic particulate material suspended in waters is strongly light-scattering, but weakly light-absorbing. Mineral particles must be of small size in order to remain suspended in quiescent waters, thus only silt-sized and smaller (<60 μm diameter) particles typically contribute to light attenuation in lakes. Plate-shaped clay minerals settle very slowly unless aggregated into larger particles, and may contribute appreciable cloudiness (turbidity) in shallow lakes in which wind-waves frequently disturb the bottom (Gerbeaux and Ward 1988). Maximal light scattering of mineral particles occurs around 1 μm , within the size range of some clay minerals.

Water clarity

There are two main aspects of water clarity—light penetration and visual clarity (Davies-Colley and Smith 2001)—and both strongly influence the ecology and amenity value of lakes. Light penetration refers to the rate at which light of different wavelengths is reduced (attenuated) with depth into the lake by absorption and scattering. Visual clarity refers to the distance (range) at which submerged objects can just be resolved by eye, which is also reduced by absorption and scattering. Visual clarity and light penetration are both “optical” properties of lake waters, in as much as they both depend on light absorption and scattering, but there is no unique relationship between them. That is to say, light penetration into a lake cannot

be predicted from measurements of visual clarity or *vice versa* (Davies-Colley and Vant 1988).

Light penetration

Photosynthetically-available radiation (PAR; light in the 400–700 nm band) coincides with visible light and is often measured in lakes by taking vertical PAR profiles with submersible PAR sensors. Figure 21.5 shows an example of the typical exponential decline of PAR in a depth profile. The light penetration can be quantified by the slope of the $\ln(\text{PAR})$ vs depth profile, known as the irradiance attenuation coefficient, K . A more “intuitive” index of light penetration is the depth of the 1% PAR level, which is an approximation of the maximum depth of plant growth in lakes and has been termed the *euphotic depth*. For example Vant *et al.* (1986) found that macrophytes in North Island lakes grew to a depth just slightly above the euphotic depth, and Hawes *et al.* (2003) defined minimal light requirements for a range of aquatic plant species in terms of % penetration of light. Light penetration turns out to be more strongly related to light absorption than scattering, because it is absorption that actually removes light from the water column. The main effect of scattering is to force light photons to take a tortuous path through the water column, which increases their effective path length and thus the probability of their absorption over a given depth interval (Kirk 1994b). A small fraction of incident light “escapes” the water body due to back-scattering (typically “reflectance” is about 3%).

The slight non-log-linearity of the PAR depth profile (e.g., Fig 21.5) is mainly due to the progressive optical filtration of the light with depth due to selective absorption by certain water constituents. Typically water itself dominates absorption of red light, and, as we saw in Table 21.2, organic constituents of lakes often selectively absorb blue light, with the result that light of intermediate wavelengths

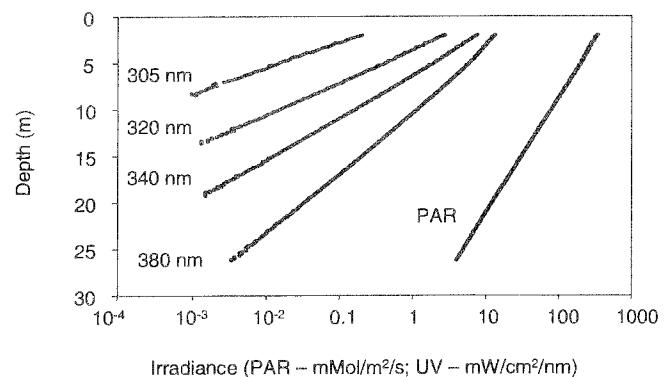


Figure 21.5 Depth profile of photosynthetically available radiation (PAR) and of spectral irradiance at several UV wavelengths measured in Lake Taupo as part of the TAUPEX experiment. (Authors’ unpublished data.)

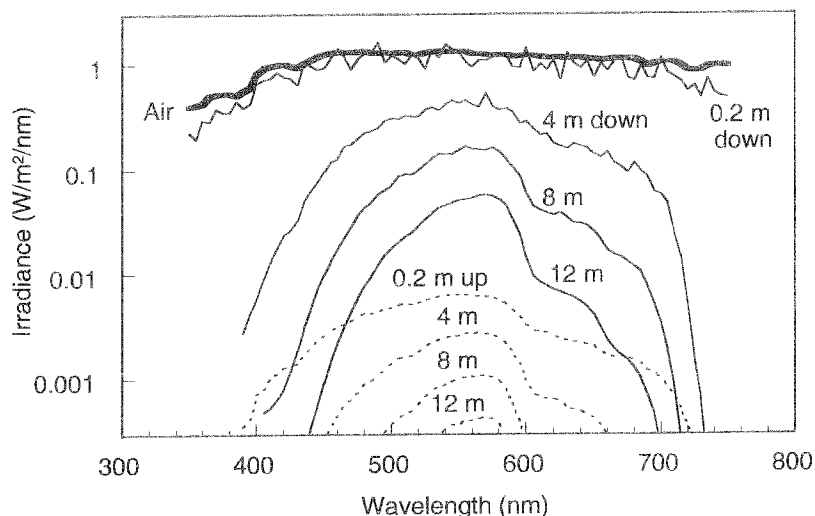


Figure 21.6 Spectral irradiance in Lake Manapouri near the boom at West Arm Power Station. The upper values (down) correspond to different depths of downwelled irradiance and the lower values (up) to the corresponding depths of upwelled irradiance. (Authors' unpublished data.)

in the green region is most strongly penetrating. Figure 21.6 shows an example of spectral distributions of light at different depths in a green-coloured lake. The filtration of the light is shown by an increased concentration of the light energy in the centre of the visible spectrum with increasing depth.

Typically, optical trends in the blue part of the spectrum extend into the ultraviolet wavelength range, which is more strongly attenuated than blue light, and chromophoric dissolved organic matter (CDOM) is particularly effective at reducing the penetration of ultraviolet (UV) (Rae *et al.* 2001). This is of increasing concern for potential damage to aquatic life now that UVB irradiance has been increased by ozone thinning in the stratosphere.

Visual clarity

The distance submerged objects can be seen through water depends on the contrast of the image of the object versus the background light in the water. When this contrast, which can be mathematically defined and measured, has fallen to a certain threshold, the eye, whether that of human or fish, can no longer detect the target. Because the threshold contrast is very nearly the same for different individuals, the visual range in a particular lake water is highly reproducible by different observers (e.g., Smith (2001) reported a coefficient of variation of 5%). Therefore, the observation of visual clarity in waters is not the "highly subjective" procedure often decried in the literature (Davies-Colley and Smith 2001).

It turns out that this "contrast attenuation" process occurs at the same rate, at least for horizontal viewing, as the beam attenuation, c , and indeed, measures of visual

clarity such as the Secchi depth (see below) can be used to estimate c . Because the scattering coefficient is usually much larger than the absorption coefficient (typically $b \sim 10a$, although varying appreciably between lakes), and because c is the sum of a and b (see above), visual clarity depends more strongly on scattering than absorption of light, unlike light penetration. The strongly humic-stained waters of Lake Brunner, Westland, are visually clear, despite the high humic absorption. Conversely, Lake Tekapo water has a very low light absorption coefficient (little greater than that of water itself), but its high mineral sediment ("glacial flour") content results in a high scattering coefficient and this reduces the visual range to a metre or so.

Water colour

The colour of light at depth in a lake is determined by the spectral properties of the light-attenuating constituents and can be measured using a submersible spectroradiometer of the type used to obtain the data plotted in Figure 21.6. Colour has relevance to the light field for vision of aquatic animals and photosynthesis of aquatic plants (Davies-Colley *et al.* 1993). For humans, other than when diving, the spectral distribution of light of most interest is that emanating from the water due to backscattering, and which can be measured by pointing a spectro-radiometer to look downwards at the water surface.

Water colour has three main aspects. Most important is the *hue*, which is related to the part of the spectrum where there is most energy. For example, Lake Manapouri appears green in hue because the "dominant" wavelength is in the green region of the spectrum (Fig. 21.6). The *brightness* of the lake water is related to the amount of energy reaching the observer's eye—Lake Manapouri returns little energy and hence the water appears dark. Finally the *colour purity* (or saturation) is related to the dispersion of light energy around the dominant wavelength (Davies-Colley *et al.* 1993)—Lake Manapouri has a broad peak of light energy and is therefore of low colour purity.

Downward-looking radiometers mounted on satellites or aircraft are being used increasingly to obtain information on water colour in inland, coastal and ocean waters (with appropriate corrections for atmospheric effects). As we have seen, the constituents of water determine the characteristics of backscattered light, and one of the main aims of remote sensing of water colour is to calculate the concentrations of key constituents.

Measurement of the optical characteristics of lake waters

Table 21.3 lists optical measurements that may be made as part of an investigation of lake waters, in order of increasing complexity and sophistication. The optical properties listed in Table 21.3 are mainly *apparent optical properties* (Davies-Colley *et al.* 1993; Kirk 1994a) because they are weakly dependent on the light conditions, as well as being strongly dependent on the inherent optical properties (*a, b, c*).

The most common and simple optical measurement on lake waters is that of visual clarity, traditionally made with a Secchi disc, a white (or black-and-white) disc that is observed vertically and the depth of “extinction” noted (average of depth of disappearance and re-appearance) (Davies-Colley 1987). It is important to use a viewer for the Secchi observations, so as to minimise water surface glare (Smith 2001). Although visual clarity measurement with a black target is to be preferred on theoretical grounds (Davies-Colley 1988b), many very valuable datasets have been assembled with Secchi discs on New Zealand lakes. Furthermore, the vertical observation is usually more

convenient for one person from a boat in deep water.

Visual clarity may also be measured as black body range (hydrological range: Duntley 1962), conveniently using a black disc viewed horizontally. Black disc visibility is inversely related to beam attenuation coefficient (an inherent property) and is independent of lighting conditions (because black objects ideally reflect no light), so black disc visibility may be regarded as an inherent optical property. Steel and Neuhauser (2002) showed that black disc visibility is empirically identical to *horizontal* (rather than vertical) Secchi disc range, so we may expect that black disc range is very similar in magnitude to horizontal sighting ranges of practical importance, as was inferred theoretically by Duntley (1962). The vertical Secchi depth is appreciably greater in magnitude than the black body range, however. Empirically it is found that (vertical) Secchi depth, $z_{SD} \sim 6.4/c$ (Kirk 1994a), while the (horizontal) black disc visibility, $y_{BD} = 4.8/c$ (Davies-Colley 1988b). From this, we may infer that Secchi depth is typically about one-third greater than black disc visibility ($z_{SD} \sim y_{BD} 6.4/4.8 = 1.33y_{BD}$) (Davies-Colley and Close 1990).

Another measurement that is sometimes used to indicate water clarity is nephelometric turbidity. We

Table 21.3 Optical measurements and observations for lake waters

Optical property	Measurement	Instrument	Measurement protocol (reference)
<i>Field measurements</i>			
Visual clarity	Secchi depth	Secchi disc, viewer	Extinction depth of Secchi disc observed vertically (Smith 2001)
	Black disc visibility	Black disc, viewer	Extinction depth of black disc observed horizontally (Davies-Colley 1988b)
Light penetration	PAR attenuation coefficient, $K(\text{PAR})$ euphotic depth	Submersible PAR sensors	Measure PAR at different depths down to at least the euphotic depth (Davies-Colley <i>et al.</i> 1993)
‘Brightness’	PAR reflectance coefficient, $R(\text{PAR})$	Submersible PAR sensors	Measure ratio of upwelling to down-welling PAR at different depths (Davies-Colley <i>et al.</i> 1993)
Spectral measurements	Attenuation coefficient, $K(\lambda)$; Reflectance coefficient, $R(\lambda)$	Submersible narrow-band light sensors OR submersible spectro-radiometer	Measure spectral irradiance at different depths and upwards- <i>versus</i> downwards-directed (Rae <i>et al.</i> 2001)
Colour matching	Hue, saturation (colour purity)	Underwater viewer, colour standards	Visual match of colour standards to water colour viewed simultaneously (Davies-Colley <i>et al.</i> 1997)
<i>Laboratory measurement</i>			
Yellow substance	Absorption coefficient of a membrane filtrate at 440 nm	Double-beam spectro-photometer	Measure absorbance of a membrane-filtrate. Correct for residual scattering. (Kirk 1976; Davies-Colley and Vant 1987)

recommend against the use of turbidity because it is instrument-specific and is merely a relative measure that has no meaning until calibrated to, for example, visual clarity (Davies-Colley and Smith 2001).

As highlighted above, visual clarity, measured for example, by Secchi disc, does not provide a useful index of light penetration into lakes, which can only be measured by deploying light sensors in the water. Measurement of the irradiance attenuation coefficient, K , using a submersible PAR sensor is recommended in order to characterise light penetration for photosynthesis of aquatic plants. Also useful, but much less often measured, is the PAR reflectance coefficient, $R = \text{PAR}_u/\text{PAR}_d$ where the subscripts indicate upwards- and downwards-directed PAR. The reflectance coefficient quantifies the “brightness” of the water (one of the attributes of water colour) and Kirk (1994b) discusses how R and K measured in tandem can be used with the results of his Monte Carlo modeling (Kirk 1994c) to provide very useful optical characterisations of waters, including estimation of the absorption and scattering coefficients.

For specialist purposes, spectral irradiance measurements in water are required in order to estimate $R(\lambda)$ and $K(\lambda)$ at the particular wavelengths of interest or over the whole visible and near-visible spectrum. If measurements at relatively few specific wavelengths suffice, narrow-band sensors may be used, such as the PUV-500 instrument used by Rae *et al.* (2001) to characterise penetration of ultraviolet wavelengths into lakewaters. More generally, submersible spectro-radiometers are used to estimate $R(\lambda)$ and $K(\lambda)$ as functions of wavelength throughout the ultraviolet to near infra-red spectrum (Davies-Colley *et al.* 1997).

For some purposes it may be sufficient to optically characterise the lake water by recording its colour rather than taking spectral measurements. Description of water colour is very subjective, but colour can be reproducibly matched by eye to stable colour standards. Davies-Colley *et al.* (1997) reported accurate hue matching and less accurate, but still useful, matching of colour purity (“saturation”) using Munsell Standards in a diverse range of New Zealand lake waters. Brightness is best measured instrumentally as reflectance rather than by colour-matching.

All of the optical measurements introduced above are best made *in situ*, but there is one optical measurement we would recommend in routine lake survey that must necessarily be done in the laboratory; the measurement of yellow substance by spectrophotometer on membrane-filtered water samples (Davies-Colley and Vant 1987; Kirk 1976). More complex spectrophotometric measurements may be used as an alternative to *in situ* spectral irradiance measurements, but require very high-quality laboratory spectrophotometers and procedures such as the use of an

integrating sphere or other specialised accessories (Davies-Colley *et al.* 1993).

LAKE CHEMISTRY

Chemical constituents of lake waters

Chemistry is a fundamental attribute of aquatic ecosystems; the availability of material integral to biota can set limits on the type and abundance of organisms that can develop and many other dissolved or particulate materials can indirectly affect the performance of groups of organisms. The chemical constituents of water can be categorised based on their provenance, concentrations, whether dissolved or particulate, and their degree of interaction with biological processes.

Traditionally, the *major ions* are those that are present at relatively high concentration (typically measured in g/m^3). These constituents are mainly derived from rock weathering and are weakly involved in biological processes, thus concentrations change slowly. Such constituents are termed “conservative”. Trace elements are present at low concentrations (typically measured in mg/m^3 or lower), and are also derived from rock weathering, but they are often required in biological systems, though again at low concentrations. Nutrients (predominantly carbon, nitrogen and phosphorus) are elements that form a major part of living organisms, and their concentration can have profound impacts on lake ecosystems. Concentrations range upwards from mg/m^3 and the close interaction with biological processes means that these can be very dynamic constituents. Dissolved gases fall into two classes: biologically active (e.g., oxygen, carbon dioxide, sulphur gases, methane, nitrogen, nitrous oxide) and biologically inert gases, such as argon and helium. Dissolved organic matter (DOM) is a group of compounds that, as its name suggests, is ultimately derived from biological processes, though the longevity of these compounds in water varies considerably. One of the most important components of dissolved organic matter is chromophoric dissolved organic matter (CDOM—see section above: “Light attenuating constituents”). Finally, particulate constituents are those that are insoluble, but are present as suspended solids. In the following section we will consider each of these groups of chemical constituents in turn.

Major ions

Globally, it is recognised that the major ion composition (Na^+ , K^+ , Mg^{2+} , Ca^{2+} , Cl^- , SO_4^{2-} , HCO_3^-) of water is largely determined by the lithology of the catchment (see also Chapter 11). Hard granitic or metamorphic rocks tend to yield relatively few, mostly monovalent ions, whereas softer sedimentary rocks are more easily weathered and some, particularly limestones, yield more divalent ions and

carbonates. Proximity to the sea and its aerosol and spray, as well as the influence of geothermal waters in some parts of New Zealand, also influence major ion composition.

The regional pattern of major ion distribution in New Zealand lakes was reviewed by Timperley (1987). He noted that the chemistries of South Island lakes were mostly influenced by the predominant metamorphic schists of the Southern Alps. The Nelson lakes group lying in solely schist catchments had low concentrations of major ions, with a slight dominance of Ca over Na and SO_4 over Cl. The presence of sedimentary rocks, including limestone, in the catchments of some lakes on the eastern side of the Alps, led to an increase in Mg, Ca and HCO_3 compared with ions in the Nelson lakes. Timperley noted a small increase in Cl on the western side of the Alps that might be attributed to proximity to the ocean and prevailing westerly winds. Similarly, proximity and occasional connection to the ocean affects the major ion chemistry of coastal water bodies on the east coast of New Zealand (e.g., Schallenberg and Burns 2003).

Timperley (1987) summarised four major groups of lakes within the North Island. In the dune lakes of the west coast, sea salts dominate, giving high concentrations of Na and Cl, with concentrations of other ions dependent on local lithology. The major ion content of Lake Waikaremoana reflects the presence of Tertiary calcareous marine sediments in the lake's catchment, and anions are thus dominated by Ca. The lakes of river floodplains formed a group with few coherent chemical features. Within the unusual waters of the Taupo Volcanic Zone, major ionic composition is affected by both the underlying volcanic lithology and geothermal fluids. Many of the Taupo Volcanic Zone lakes have very unusual chemistries, with high concentrations of Na, Cl and SO_4 . Deep geothermal fluids are moderately saline, providing high Na and Cl, and the fluids can contain high concentrations of CO_2 and H_2S (Ellis and Mahon 1977; Giggenbach and Glover 1992). H_2S readily oxidises to H_2SO_4 on contact with oxygen, and this can introduce very high concentrations of SO_4 into lake waters. Timperley and Vigor-Brown (1986) have used the relative concentrations of the various ions associated with geothermal water, steam and rock weathering to derive a budget for the relative contributions of different mechanisms to the ionic composition of the waters of the Taupo Volcanic Zone.

Trace elements

With the exception of the geothermally-influenced waters of the Taupo Volcanic Zone and a study on Lakes Manapouri and Hayes by Reid *et al.* (1999), there is relatively little information on trace elements in New Zealand lakes. This reflects firstly the analytical difficulties of reliable determinations of trace elements in natural

waters, secondly their apparent unimportance with respect to problems of eutrophication (which have focussed on nitrogen and phosphorus) and the overwhelming interest in the highly unusual geothermic waters (e.g., Giggenbach and Glover 1992). Studies of geothermal waters have tended to focus on potentially toxic elements, notably mercury (transferred in geothermal steam) and arsenic (from geothermal water). For example, Timperley (1987) describes increases in arsenic, boron, lithium and mercury in the Waikato River as it passes the Wairakei geothermal power station discharge, and Aggett and Aspell (1980) note that arsenic is concentrated in organisms within the Waikato River. Further studies (Kim 1995; Kim and Burggraaf 1999) have found bioaccumulation of mercury associated with the trout food web in the Rotorua lakes, several of which are geothermic. By contrast, amongst lakes of glacial origin, Lake Manapouri has trace metal concentrations that are extremely low on a global scale, while concentrations in Lake Hayes are intermediate.

Dissolved gases

Many of the gases dissolved in lake water are derived from the atmosphere, with a small but important number from biological and geothermal sources. We have already seen how CO_2 and H_2S enter with geothermal steam and water, but both of these gases have other sources.

In a freely-circulating water column, the amount of dissolved gas tends to approach that expected at equilibrium ("saturation") with the atmosphere (see also Chapter 11). This equilibrium concentration is dependent on temperature and salinity, with solubility decreasing with increasing temperature and salinity. Pressure also affects gas solubility—water at 10 m depth (two atmospheres of pressure) could contain twice as much gas as water at the surface (one atmosphere), but as the equilibration takes place at the surface this pressure effect is normally not important. However, whenever an atmospheric gas is present at concentrations other than air equilibrium at one atmosphere, it indicates a source or sink other than the air-water interface.

Oxygen

In considering dissolved gases, it is useful to examine how a body of water initially equilibrated with the atmosphere can begin to deviate from saturation. For oxygen, this is initially quite simple. Oxygen is produced by photosynthesis in plants and is consumed by respiration by all aerobic organisms and by chemical oxygen demand. As described above, physical stratification can isolate the lower lake waters from the upper, and under such conditions photosynthesis can dominate in the epilimnion, and respiration in the hypolimnion. Depending on the rate of oxygen consumption in the hypolimnion, there is a gradual

decline of dissolved oxygen (Fig. 21.7). In Lake Pupuke the rate of oxygen consumption is not uniform within the hypolimnion. In this lake, there are two main sites of oxygen consumption—at the sediment surface and in the thermocline. Such complexities are not uncommon in lakes with stably stratified deep waters. These zones are likely to be associated with decomposition of organic particulate material deposited on the bed sediment, or else trapped or slowed down as it settles through the density gradient of the metalimnion.

Depletion of oxygen in the hypolimnion is termed *hypolimnetic oxygen depletion* (HOD). The rate of hypolimnetic oxygen depletion varies with a number of factors: the temperature, the amount and rate of supply of decomposable material and the size of the oxygen reservoir. The amount of decomposable material in the hypolimnion (and associated sediments) is a useful proxy for time-integrated production of the lake, particularly when a high proportion of organic matter reaching the hypolimnion is in the form of sedimenting phytoplankton. Because of this, hypolimnetic oxygen depletion has been proposed as an indicator of lake trophic status (Strom 1931; Hutchinson 1938; Vant 1987). In order to correct for the effect of the size of the oxygen reservoir at the start of stratification (so as to allow for inter-annual comparisons), the *volumetric hypolimnetic oxygen depletion rate* (kg oxygen/m³/day) is used. However, inter-lake comparisons, particularly where bathymetry differs between lakes, are not reliable, as variation in the volume of the hypolimnion that is in contact with the surface area of sediment can significantly affect the volumetric hypolimnetic oxygen depletion (Charlton 1980). Attempts to compensate for this by using the *areal hypolimnetic oxygen depletion rate* are only partly successful (Burns 1995), though Schallenberg and Burns (1999) explained 96% of variation in areal hypolimnetic oxygen depletion in Lakes Coleridge, Moke, Hayes and Johnson with an existing model that included chlorophyll *a*, mean depth and euphotic depth. Overall, volumetric hypolimnetic oxygen depletion and areal hypolimnetic oxygen depletion are useful integrative measures of biological functioning and trophic status within a lake, for they facilitate year-to-year comparisons, but they need to be used cautiously when comparing different lakes (Burns 1995). In Lake Pupuke, for example, there has been little change in volumetric hypolimnetic oxygen depletion over the past 20 years, suggesting that this lake is in a reasonably stable state.

A second feature of the dissolved oxygen concentration in Lake Pupuke (Fig. 21.7) is that the concentration of

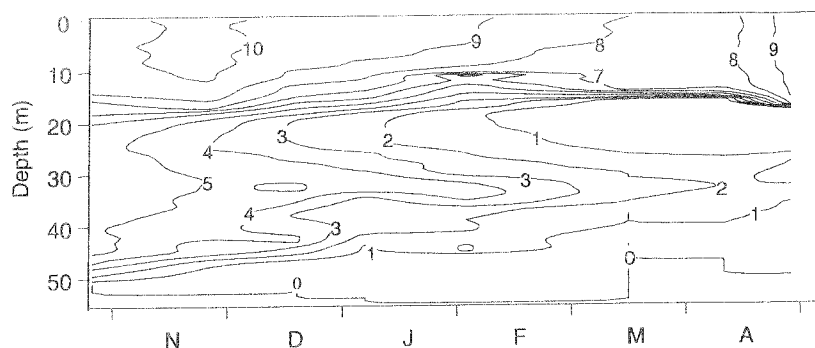


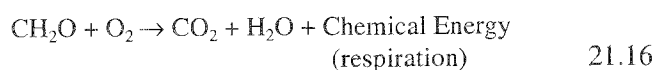
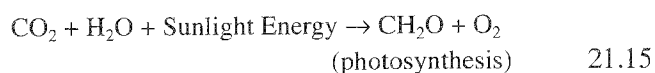
Figure 21.7 Time-depth plot of dissolved oxygen concentration (g/m³) in Lake Pupuke, summer 2002-03 (data from Auckland Regional Council).

oxygen declines in the epilimnion over summer. As the temperature of the water slowly rises, the solubility of oxygen in water declines. Over the stratification period, the temperature of the epilimnion rose from 14 to 24°C, which would translate to a decline in oxygen solubility from 10.3 to 8.4 g/m³, before temperatures declined again to 14°C before overturn. Clearly, in the freely circulating upper layer, sufficient contact with the atmosphere can occur for oxygen re-equilibration.

Changes in oxygen solubility with temperature underlie the use of % saturation to describe a water's oxygen status with respect to the atmosphere. This is simply the observed concentration referenced to that which would prevail were the water saturated with oxygen at the same temperature and at one standard atmosphere pressure. For example, at 15°C, water is saturated with 10.07 g/m³ oxygen and at 5°C with 12.77 g/m³. Samples of these waters would therefore both be 100% saturated, despite having different concentrations. Conversely, water at 15°C with 12.77 g/m³ of oxygen would be 127% saturated (100 x 12.77/10.07) and water at 5°C with only 10.07 g/m³ would be 79% saturated (100 x 10.07/12.77).

Carbon dioxide and pH

Uptake and release of CO₂ accompany changes in dissolved oxygen related to photosynthesis and respiration, respectively. Photosynthesis is the assimilation of carbon into organic matter (denoted below as CH₂O), using sunlight as an energy source and releasing oxygen as a byproduct, whereas respiration involves releasing that chemical energy by oxidising the organic material using oxygen, and releasing CO₂ as a byproduct:



CO₂ is difficult to measure in lake water, and the measurement is complicated because, unlike oxygen, CO₂ dis-

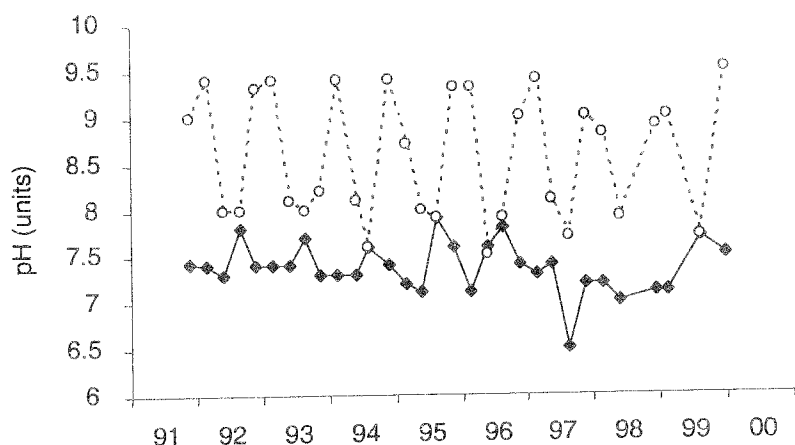
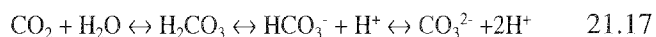


Figure 21.8 pH in the epilimnion at 5 m (open circles) and within the hypolimnion at 50 m (closed diamonds) in Lake Pupuke from 1991 to 2001 (data from Auckland Regional Council).

solved in lake waters participates in a series of equilibrium reactions with the water itself (Stumm and Morgan 1995). Changes in the concentration of CO_2 in water can, however, often be detected as shifts in pH. Waters with high rates of photosynthesis, and thus depleted CO_2 , normally show an increase in pH, while deep waters with accumulating CO_2 show a tendency for pH to decrease (Fig. 21.8). This occurs because CO_2 and pH are intimately linked through the following equilibrium reactions, involving dissolved free CO_2 , carbonic acid, bicarbonate and carbonate:



It is clear that adding CO_2 to the left side of the equation drives the equilibrium to the right to produce more H^+ and drive down pH, and *vice-versa*. This process is often most significant when high concentrations of plant biomass are present, for example during algal blooms in eutrophic lakes or within macrophyte beds, and it is discussed more fully in Chapter 11. Such pH shifts can affect not only the speciation of dissolved forms of inorganic carbon, but also that of ammoniacal nitrogen and the solubility of many metallic elements. In general, H_2CO_3 dominates equilibria below pH 4.5, HCO_3^- above pH 6, and CO_3^{2-} above pH 10.5.

Alkalinity is closely related to elements of the dissolved inorganic carbon (DIC) equilibrium. It is defined as the concentration of anions of weak acids, which the carbonic acid derivatives usually dominate, but also includes contributions from other weak acids such as H_3SiO_4^- . It is determined by titration of water to an end point of pH 4.5, at which point virtually all dissolved inorganic carbon will have been converted to H_2CO_3 or dissolved CO_2 . Alkalinity is measured in “bicarbonate equivalents”, with the amount of acid needed to reach the pH 4.5 endpoint approximating the equivalent amount of bicarbonate.

More generally alkalinity (in equivalents) is given by:



where A^- is the anions of weak acids other than H_2CO_3 . Waters with a pH below 4.5 will obviously have negative alkalinity.

Hydrogen sulphide

CO_2 is not the only gas linked to pH in New Zealand waters. As we have seen above, H_2S from geothermal sources oxidises rapidly to H_2SO_4 (sulphuric acid) in the presence of oxygen, and dissolution of this gas in water reduces pH. Many geothermally influenced waters are therefore acidic, for example, the pH in lakes Rotokawa and Rotowhero in the Taupo Volcanic Zone is around 2–3,

associated with high SO_4^{2-} concentrations from S^{2-} oxidation (Timperley and Vigor-Brown 1986)

In addition to geothermal sources, H_2S is produced through anaerobic metabolism of SO_4^{2-} . Once oxygen has been removed from waters (i.e., when they become anoxic) or lake sediments through respiration, microbes are forced to turn to alternative electron acceptors and to anaerobic pathways of energy production from organic materials. Sulphate-reducing bacteria reduce SO_4^{2-} in the process of oxidising organic material, thus releasing energy, CO_2 and S^{2-} . S^{2-} can occur in a range of compounds, and can combine with hydrogen to form H_2S , the pungent “rotting egg” smell associated with many strongly reducing environments. An example in recent years in New Zealand was the lake formed by the Opuha Dam in South Canterbury (authors’ observations). Here H_2S was produced during the decomposition of drowned soils and vegetation and, as the dam released hypolimnetic water, H_2S was also released to the air. The extent to which sulphide can accumulate in anoxic water and sediment is dependent on the availability of SO_4^{2-} . It is a much more important process in marine environments rich in SO_4^{2-} than in freshwater. Sulphide is rapidly oxidised when exposed to oxygen, however, H_2S is highly toxic to many aquatic organisms at low concentrations, and at times sufficient concentrations accumulate in lake hypolimnia to become toxic during turnover.

Nitrogen gas

Nitrogen gas is a principal component of the atmosphere and is usually the dominant dissolved gas in natural waters. However, nitrogen is also a key nutrient for organisms: it occurs in a wide variety of compounds and various microbial processes have evolved to allow nitrogen to be transformed into a number of forms. For this reason, the dynamics of nitrogen are described in the following section.

Nutrients

In this section we deal with nitrogen (N) and phosphorus (P). Carbon (C), along with a variety of trace metals (see above), is also essential to aquatic biota. At times the low availability of trace elements and inorganic C can be limiting to growth, in the case of inorganic C particularly for organisms restricted to a sub-set of the forms of inorganic C present in water (see Chapter 22). In most freshwaters, however, it is N and P that are often supplied at such low rates that algal growth is limited. While in most north-temperate lakes P levels determine the limits of growth, in many New Zealand lakes N availability can limit growth (White *et al.* 1985). Enrichment with nutrients often follows human intervention in landscapes, and the consequences of enrichment—eutrophication, particularly the proliferation of phytoplankton—have received a great deal of attention in New Zealand and internationally in the latter part of the 20th century (Burns 1991; see Chapter 22).

Nitrogen

Nitrogen is a key element within biological systems, not least because it is a constituent of amino acids that form the building blocks of all proteins. Nitrogen can form more than 10% of the dry weight of organisms. It occurs in freshwaters in a variety of forms: as nitrogen gas, ammoniacal-N ($\text{NH}_3 + \text{NH}_4^+$), N_2O , NO_2^- , NO_3^- and various organic forms. The total amount of N available is largely determined by inputs from the catchment (N is found as NO_3^- in high concentrations in run-off from many agricultural areas, particularly intensive dairying—see Chapters 11 and 34), but the speciation of N is dependent on a range of biogeochemical processes.

Nitrate uptake

Nitrogen normally enters lakes in its most oxidised form, as NO_3^- . This nitrate is available for uptake by plants but, where N is in plentiful supply relative to growth, NO_3^- remains in the water. Most New Zealand lakes tend to have rather low concentrations of NO_3^- (Fig. 21.9) and concentrations $>20 \text{ mg m}^{-3}$ occur only rarely, in lakes with agricultural catchments.

Nitrogen fixation

When fixed nitrogen is in short supply, nitrogen gas can be converted to ammoniacal nitrogen ($\text{NH}_3 + \text{NH}_4^+$) through the microbial process of N_2 fixation. This ammoniacal nitrogen is then available for growth. In

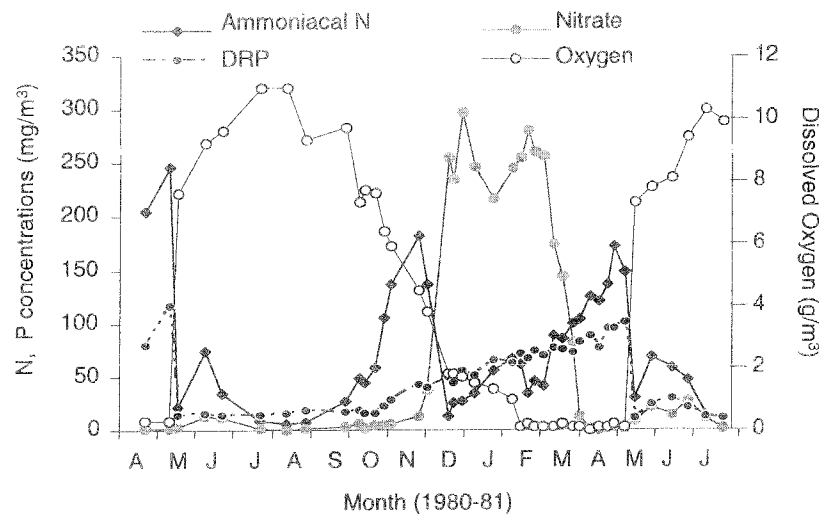


Figure 21.9 Time series of dissolved oxygen, ammoniacal N, nitrate-N and dissolved reactive phosphorus (DRP) from 60 m in Lake Rotoiti.

freshwater, N is fixed predominantly by heterocystous cyanobacteria (planktonic forms such as *Anabaena* and benthic forms such as *Tolypothrix*), using energy derived from photosynthesis to reduce N_2 to NH_4^+ . N-fixation is a strictly anaerobic process and the heterocyst is a specialised oxygen-free cell that protects oxygen-sensitive enzymes (Fogg *et al.* 1973). N-fixation is, however, a very energy-demanding process, and it competes with carbon assimilation for energy in these cyanobacteria. Thus organisms grow more slowly when they rely on N-fixation than when an alternative source of fixed N is available. When N availability is limiting growth, however, as in Lake Taupo, then, if sufficient light energy is available, N-fixation potentially confers a selective advantage to heterocystous cyanobacteria.

Nitrogen and decomposition

Organic nitrogen is recycled to inorganic forms through microbial action and through digestion. The primary product of decomposition is NH_4^+ . Although NH_4^+ is readily available to algae for uptake, when decomposition is spatially separated from the photosynthetic zone, for example in the hypolimnion, microbes can obtain energy for fixation of carbon through oxidation of NH_4^+ to NO_2^- and NO_3^- . This process, known as nitrification, requires oxygen and, while it is energetically favourable, is poorly so, and the bacteria performing this operation are slow-growing.

Once oxygen runs out in the decomposition zone, nitrification is no longer possible, and at this time denitrification may occur. In this process nitrate replaces oxygen as an electron-acceptor and nitrate is in turn reduced, to yield NO_2^- , N_2O and N_2 (Priscu *et al.* 1986). This process is very important, as it can lead to the loss of fixed nitrogen from a lake system, in the form of the gases

N_2O and N_2 . Lake Rotoiti provides a classic example of this sequence of processes (Fig. 21.9): first NH_4^+ is generated in the hypolimnion, which is then oxidised to NO_3^- , which, once oxygen is depleted, is lost as gaseous N_2 , while NH_4^+ begins to reappear in the water column. Of course within a given water body these processes may be spatially as well as temporally separated. Where oxic-anoxic boundaries are sharp, as in sediments or at sediment-water interfaces, nitrification and denitrification may occur within a few centimetres of each other and sharp gradients of nitrogen species can then occur. In contrast to Lake Rotoiti, a more oligotrophic lake in which hypolimnetic depletion of oxygen does not proceed to anoxia will not show this full range of processes. Lake Taupo, for example, does not reach anoxia during stratification, and in this lake NO_3^- continues to accumulate in the hypolimnion throughout the stratification period (Fig. 21.10).

Dissolved organic nitrogen

Dissolved organic nitrogen (DON) refers to a poorly-defined group of compounds that may represent an unusually high proportion of fixed N in New Zealand lakes. For example, in Lake Taupo, where inorganic forms of N are virtually undetectable during the summer stratification period, dissolved organic nitrogen can be present at almost invariant concentrations of 40–60 mg/m^3 throughout the year. A substantial proportion of dissolved organic N can therefore be hypothesised to be a reservoir of N that sits largely outside the biological cycle (*refractory N*)—though of course many other compounds within the overall dissolved organic N pool (e.g., amino acids) are very rapidly consumed (*labile N*). An important consequence is that measurements of total N, commonly interpreted as an index of total available N, may

substantially overestimate the pool of N actually in circulation within the lake ecosystem (White *et al.* 1985).

Phosphorus dynamics

Like nitrogen, phosphorus (P) is a fundamental component of living organisms, forming part of nucleic acids and adenosine triphosphate (the basis of energy management within cells). Like nitrogen, phosphorus that is taken up in the *trophogenic* (organic matter-producing) regions of lakes often ends up being re-mineralised in aphotic regions. Here inorganic P is steadily released into the water column, as is evident in Lake Taupo (Fig. 21.10). In Lake Taupo, the ratio of N:P release is close to 10, reflecting the expected stoichiometry of mineralisation of organic material. Once oxygen reaches very low concentrations, however, dissolved reactive phosphorus (DRP) is released from sediments by a chemical process that does not follow such a stoichiometry. This process relates to the scavenging of PO_4^{3-} from water during the formation of particles of insoluble Fe^{3+} and Mn^{4+} oxides and oxyhydroxides that occurs under aerobic conditions. These insoluble compounds settle to the lake floor and would ultimately form minerals such as apatite that lock up phosphorus. However, under anoxic conditions, Fe^{3+} and Mn^{4+} are reduced to Fe^{2+} and Mn^{2+} , which are much more soluble. As these forms dissolve, any bound PO_4^{3-} is released. Rapid release of dissolved reactive phosphorus into anoxic hypolimnia can accompany such transitions. Once the reduced forms of Fe and Mn reach oxygenated water, chemical and microbial oxidation can occur, and re-precipitation of P will generally ensue, though this P may also be available for biological uptake.

Note that inorganic phosphorus is frequently referred to as DRP or FRP—dissolved or filterable reactive phosphorus. This convention acknowledges that the standard techniques used for phosphorus analysis do not distinguish between a group of inorganic forms of P, all of which are closely related to PO_4^{3-} , but which are not all equally available to algae. The term filterable reactive phosphorus (FRP) is rarely used but is technically more correct as dissolved reactive phosphorus (DRP) can include colloidal forms that pass through a filter but are not dissolved.

Particulates

Particulates fall into two categories, predominantly mineral or organic (see also Chapter 11). Mineral particles are denser than water and so sink rapidly (Eq. 21.12) unless they are very fine-grained. In most cases, mineral particles are therefore deposited close to the point at which they

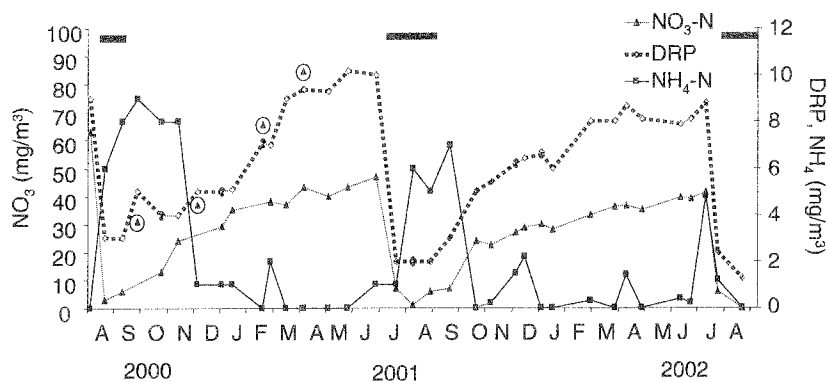


Figure 21.10 Time-series of bottom water (150 m) nutrient concentrations at a mid-lake site in Lake Taupo. Ringed data are regarded as anomalous because of geothermal activity. Black bars indicate duration of the winter mixed period. (Data from Environment Waikato.)

enter a lake. In lakes, the finest mineral particles can stay suspended for weeks (Schallenberg *et al.* 1999) and account for the exceptional optical properties of lakes with high loads of fine glacial "flour", such as lakes Tekapo and Pukaki (Davies-Colley and Smith 2001). Shallow lakes too can maintain a very high suspended mineral load when wind-induced wave action repeatedly re-suspends material from the lake bed.

Mineral materials are mostly finely-divided silicates or oxides, or clay minerals (layer silicates) derived from weathering, which may interact with organic and inorganic fractions through surface adsorption. Dissolved reactive phosphorus in particular has a high affinity for particulates, which may be plentiful in turbid waters. For example, Lake Ellesmere water has a very high total P content (up to almost 1000 mg m⁻³) and this is very highly correlated with suspended sediment, as most is sediment-bound (Ward *et al.* 1996; Hamilton and Mitchell 1997). This particle-bound P is likely to be in dynamic equilibrium with the dissolved fraction and thus the particulates may act as a buffer for dissolved reactive phosphorus. Other chemicals that have high affinities for particulates include heavy metals and some hydrophobic organic contaminants.

As discussed above, organic particulates, including algae, bacteria and organic detritus, play a major role in the cycling of C, N and P (see also Chapters 11 and 22) and the light climate of lakes. Organic particulates can be degraded back to inorganic forms (mineralised) either in the water column, where there is often a zone of intense mineralisation within the thermocline (see Fig. 21.7), or after they become part of the sediment. The processes that lead to these transformations are not regulated only by biota. Interactions with geological, physical and chemical processes are also responsible for determining concentrations of inorganic and organic material.

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Chapter 20

Hydrology of wetlands

Dave Campbell and Rick Jackson

INTRODUCTION

Wetlands are generally a minor component of New Zealand's modern-day landscapes; however this was not always the case. More so than almost any other natural ecosystem, wetlands have been reduced to a fraction of their former extent, with only 10% of their original area, largely because they occupied low-lying, often fertile land prized for agriculture. The decline in the extent of wetlands in New Zealand since the mid-nineteenth century is one of the most dramatic examples in the world (Park 2002).

Despite this decline there remains an enormous diversity of wetland type, form, and function to be found across a wide spectrum of landscape and climate types in New Zealand. Wetland diversity is driven, above all else, by the diversity of key hydrological parameters that dictate the amount and type of water, material and nutrient inflows and outflows. Because of the intimate relationships that exist between water, nutrients and plants within wetlands, wetland hydrology is often referred to as "ecohydrology", since it is impossible to separate out the purely physical from biological processes in these dynamic, living ecosystems.

Wetlands are increasingly valued for their ecosystem services and as refuges for the threatened plants and animals they contain. They have also been valued for their food and fibre resources, for flood protection, recreation, water storage, erosion control, and retention and transformation of nutrients. As the awareness of wetland values increase, so there is an increasing demand for monitoring wetland condition (Clarkson *et al.* 2003) and restoring wetland habitat (Chapter 40). Both of these require knowledge of wetland hydrology.

There have been relatively few published studies of wetland hydrology carried out in New Zealand compared to other parts of the world, which means that any attempts to manage hydrological factors for conservation or restoration purposes will be fraught with difficulty.

Jackson (1984, 1987) studied the soil water regimes and hydrological processes of "pakihi" wetlands in Westland in relation to drainage for afforestation. Evaporation and peat moisture regimes in Waikato peat bogs have been reported by Campbell and Williamson (1997), Thompson *et al.* (1999) and Campbell *et al.* (2002), while Maggs (1997) carried out some groundwater investigations on the edges of the large Kopouatai bog, Hauraki Plains. The groundwater regimes and the effects of lake-level fluctuations, geothermal inputs and effluent discharge on the hydrology and chemistry of the South Taupo wetlands have been reported by Eser and Rosen (1999, 2000) and Chagué-Goff *et al.* (1999). Fahey *et al.* (1998) and Bowden *et al.* (2001) describe the role of small headwater peat wetlands in hillslope runoff processes at Glendhu, Otago. Dickinson *et al.* (2002) described the structure and ecology of unusual patterned peat wetlands in Otago in relation to water chemistry and hydrology. Schallenberg *et al.* (2003) presented a water balance model for the large Waihola-Waipori lake/wetland complex in Otago.

While these studies have examined aspects of wetland hydrology for specific sites, given the great variety of wetland types in New Zealand, there are major gaps in our understanding of hydrological processes. This chapter describes the main types of wetlands found in New Zealand, and the hydrological processes involved in their formation and functioning. The state of knowledge of the hydrological regimes of particular wetland types are then described based on the results of research studies.

WETLAND CLASSIFICATION

Wetland definition

Wetlands are places where surface water, ground water and "dry" land meet. As Johnson and Gerbeaux (2004) put it, "wetlands are precisely that: wet lands". Wetlands

occur in a wide variety of locations within the landscape, ranging from estuaries to mountain tops. Because of the incredible diversity of wetlands, whose study straddles the biological and physical sciences, there is a vast, confusing, and often contradictory terminology used to describe them. Even defining what constitutes a wetland is fraught with difficulty, and different definitions have been formulated by ecologists, limnologists, hydrologists, soil scientists, and even lawyers and politicians. For instance, according to the Ramsar Convention, established in 1971 to foster international cooperation in the conservation and wise use of wetlands and their resources, “wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres.”

For New Zealand purposes, the Resource Management Act (1991) defines wetlands to include “permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions”.

Wetland classification can be based on several levels: the hydrological system within which the wetland occurs (or hydrosystem), nutrient (or trophic) status, substrate (or soil) type, landform, and vegetation type. At the highest level, that of the hydrosystem, Johnson and Gerbeaux (2004) recognise marine, estuarine, riverine, lacustrine, palustrine, inland saline, geothermal, underground and nival (snow) wetlands. In this chapter, we limit our scope to palustrine wetlands—those wetlands which are fed by rain, groundwater, or surface water but do not occur within the normal boundaries of estuaries, lakes or rivers.

In practice, classification of wetlands into particular types is often extremely difficult, partly because definite boundaries are rare in nature and distinct wetland types may grade into each other. A classification scheme has been developed for New Zealand wetlands (Ward and Lambie 1999), and Johnson and Gerbeaux (2004) provide a guide to its application.

Wetland types

Palustrine wetlands may be crudely subdivided according to their nutrient (or trophic) status in a manner similar to lakes (Chapter 21). While perhaps not immediately obvious, the nutrient status of a wetland is

intimately linked to the primary hydrological factor within it: that of water supply. Wetlands supplied with water, nutrients and mineral sediments from flooding rivers are relatively fertile, supporting a rich diversity of plant types, with high rates of primary productivity and rapid decay of dead plant material. These are termed *eutrophic* wetlands. *Mesotrophic* wetlands are those partly supplied from groundwater seeps and runoff from mineral soils—they have intermediate nutrient status. Those supplied solely from rainwater have extremely low levels of nutrients, and are termed *oligotrophic*, or *ombrotrophic*, the latter term referring to the meteoric water source. Along this spectrum of trophic status we can recognise three main wetland types—*swamp*, *fen*, and *bog*—ranging from high to extremely low nutrient availability. Figure 20.1 indicates typical landscape positions and water flows affecting these wetland types, and Figure 20.2 shows New Zealand

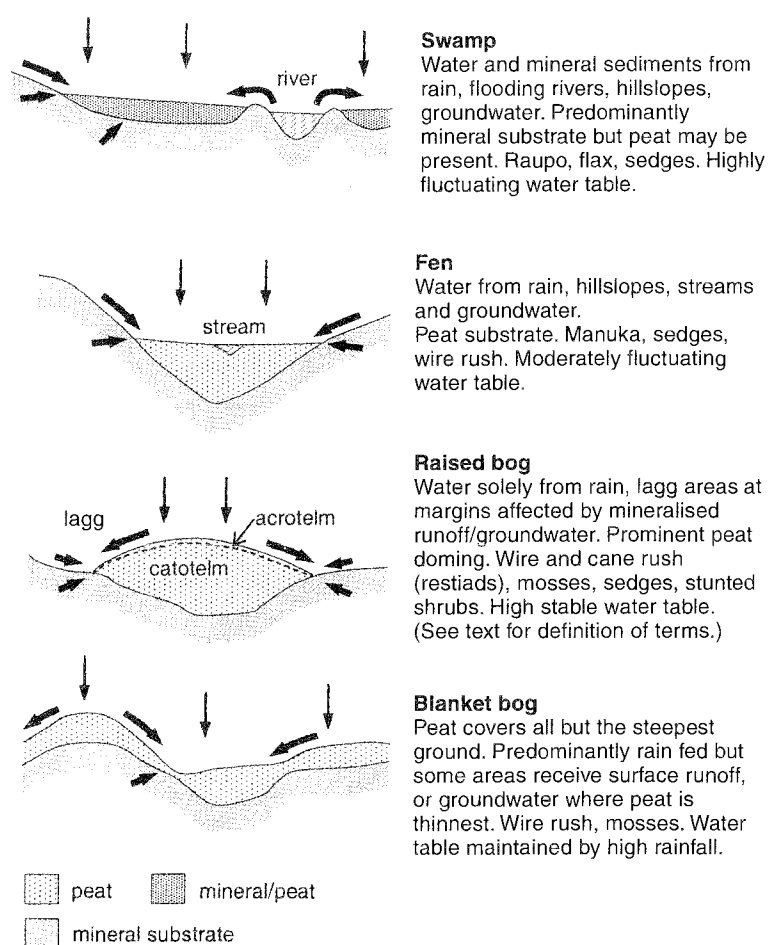


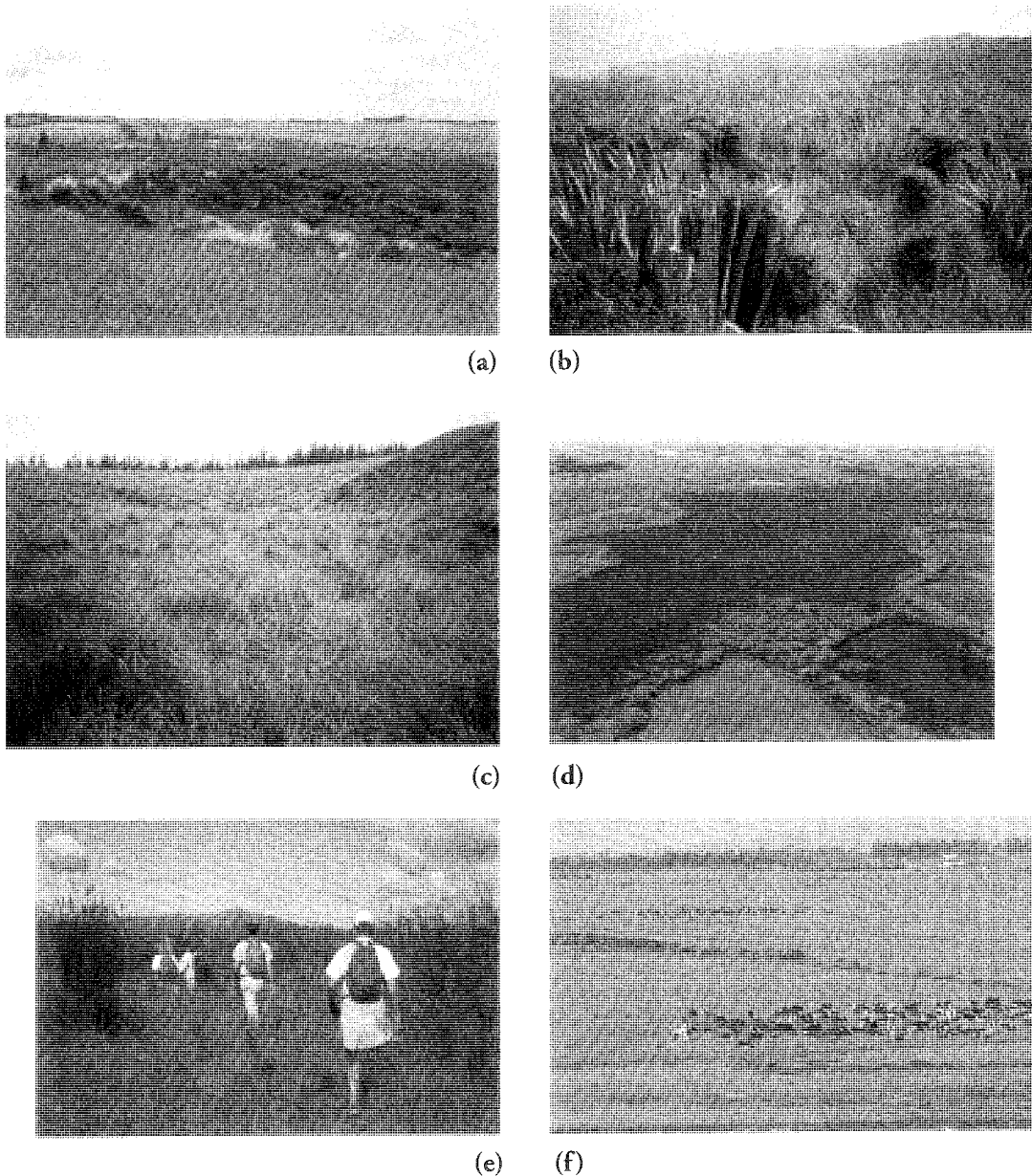
Figure 20.1 Typical landscape positions and morphology of the major palustrine wetland types, showing directions and sources of water flows, and indicating typical New Zealand vegetation associations. See text for definition of terms. Adapted from Charman (2002).

examples. The chemistry of wetland waters and soils is described in detail in Chapter 28.

Wetlands evolve over time in response to variations in water and nutrient inputs, often driven by the gradual accumulation of organic matter as *peat*, which can alter wetland morphology and hydrology at the landscape scale. Hence what is a swamp or a fen today, may in future become a bog as external nutrient inputs decline, ultimately driving dramatic changes in vegetation assemblages and soils.

Peat wetlands

Many wetlands (but by no means all) contain peat (Fig. 20.3). Peat is the preserved remains of plant material accumulated over long periods of time, and it can form deposits many metres in thickness and cover vast areas of the land surface. Peat forms the substrate upon which successive generations of wetland plants grow, and is the medium within which water is stored and moves within the wetland.



Photos: Dave Campbell

Figure 20.2 Examples of New Zealand wetlands: (a) Valley-bottom swamp at College Stream, Southland. (b) Maher Swamp, Westland, with vegetation of flax (*Phormium tenax*), sedge (*Carex* sp.), raupo (*Typha orientalis*), and shrubs (mainly *Coprosma* sp.). (c) Headwater fen at Glendhu, Otago, with vegetation of red tussock (*Chionochloa rubra*), wire rush (*Empodisma minus*) and *Sphagnum* mosses. (d) Opuatia wetland complex, Waikato, showing fen/young bog dominated by *E. minus* with grey willow (*Salix cinerea*) invading mineralised margins. (e) Vegetation of Kopouatai raised peat bog, Waikato, dominated by *E. minus* (ground cover) and the cane rush *Sporadanthus ferrugineus*. (f) The fate of many of New Zealand's wetlands: present-day view across Rukuhia peatland, near Hamilton.

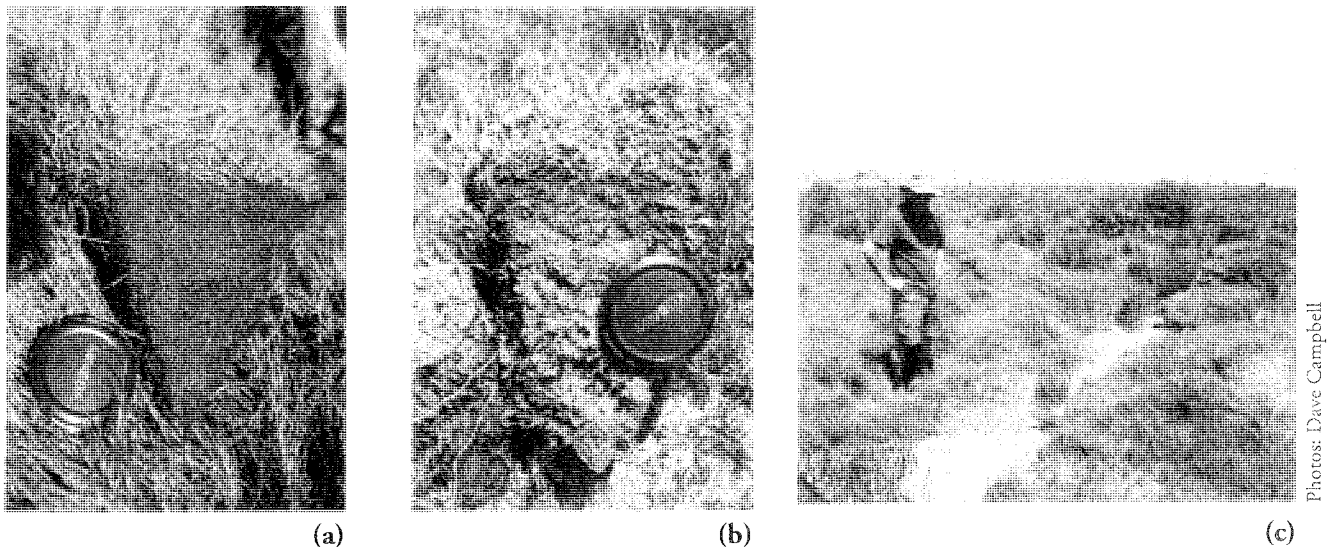


Figure 20.3 Two of the main types of peat in New Zealand bogs: examples from Southland (a) wire rush (*Empodisma minus*) peat (b) *Sphagnum* peat (c) a peat-forming environment at Centreburn Bog, with *Sphagnum* mosses filling a wet flush surrounded by a dense canopy of wire rush.

Peat accumulation requires certain hydrological conditions, in particular the maintenance of saturation, leading to low oxygen levels (anaerobic conditions) that prevent rapid rates of decomposition. The low trophic status and acidity of fen and bog environments also contribute to low rates of decomposition, leading to peat accumulation. The term *peatland* is often used to describe peat deposits, whether or not they are presently wetland environments. For example, large areas of peatland have been drained and developed for agriculture in Northland, Waikato, Bay of Plenty, Otago and Southland; these were formerly extensive peat bogs (Davoren 1978). Peatlands have considerable economic value, which has led to widespread destruction of wetlands, since peat is mined as an energy source and as a horticultural raw material, and peatlands, once drained and modified, form the basis for highly productive agricultural soils.

The term *pakihi* is loosely applied to common wetland landform/vegetation assemblages that occur in high rainfall regions in the west of the South Island from Nelson to Fiordland, on a range of ultra-infertile, impervious soil types, occasionally including peat (Mew 1983; Johnson and Gerbeaux 2004).

WETLAND HYDROLOGICAL PROCESSES

Wetland hydrology determines nutrient flows and water chemistry, and affects plant assemblages, primary productivity and the buildup of organic matter, which in certain situations leads to peat formation.

The hydrological regime, or *hydroperiod*, of a wetland encompasses the patterns of water storage and movement within and across its boundaries and may have strong cyclic

patterns at daily, seasonal, annual or even longer timescales, driven largely by external factors such as the seasonality of precipitation and energy cycles.

The wetland water balance

Wetlands form where topographic, drainage or climate factors lead to water surpluses, so that the wetland substrate is saturated for all or much of the time. A wetland is a hydrological system that has water inputs (from rain, groundwater and surface water inflows) and water outputs (evaporation, groundwater and surface water outflows). A wetland also stores water in the soil, saturated subsoil and in living plant tissues—changes in the amount of stored water over time depend on the relative magnitudes of water inputs and outputs. Hydrologists express the balance between these terms using the water balance equation:

$$\text{input} - \text{output} = \text{change in storage} \quad 20.1$$

or, by including symbols for all terms and grouping inputs and outputs, for a wetland this becomes:

$$(P + Q_{\text{in}} + G_{\text{in}}) - (E + Q_{\text{out}} + G_{\text{out}}) = \Delta S \quad 20.2$$

Symbols are: ΔS =change in stored water within the wetland, P =precipitation, Q_{in} =surface water inflows, Q_{out} =surface water outflows, G_{in} =groundwater inflows, G_{out} =groundwater outflows, E =evaporation. For a given wetland the water balance may not include all of these terms, and it may be appropriate to partition an individual term into its components. For instance surface water outflows may be partitioned into baseflow and quickflow terms (Chapter 4). The wetland water balance is shown diagrammatically in Figure 20.4, and three examples of water balances are also shown. Bogs have relatively simple

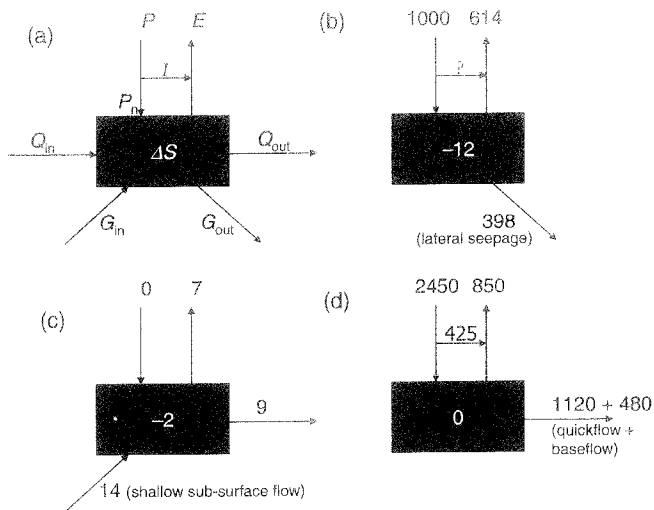


Figure 20.4 (a) Generalised water balance model for a freshwater wetland (adapted from Heathwaite 1995). See text for symbol definitions. Additional symbols are: P_n =net precipitation, I =interception loss. Example water balance models for New Zealand wetlands: (b) Moanatuatua peat bog, mean annual water balance 1999–2000, units mm/year (Smith 2003) (c) head-water fen, Glendhu Otago, average daily water balance during a 10-day rainless period in January 1995, units mm/day (Bowden *et al.* 2001) (d) Larry River pakihī, Westland, average annual water balance for 1983–1986, units mm/yr (Jackson 1987).

water balances because there are usually no surface water or groundwater inflows, while fens are linked to other hydrological systems, which provide groundwater and surface water inflows. The pakihī example is a simple catchment with impermeable soils, allowing only evaporation and surface water outflows.

When ΔS is positive, the total amount of water storage within the wetland increases—usually indicated by a rise in the water table and an increase in soil moisture content. Conversely, a negative ΔS indicates a lowering water table and total decrease in stored water. Water balance terms are usually expressed as depths of water, and for conversion purposes these are obtained from volumetric flows of water (m^3) divided by the wetland area (m^2).

Quantifying the terms of the wetland water balance poses challenges to the hydrologist because it is often difficult to completely define wetland catchment areas in low-lying areas, and surface and groundwater inflows and outflows may be difficult to assess.

Most water balance studies of wetlands estimate some terms of Equation 20.2 as residuals; this can lead to unacceptable errors in, for example, wetland nutrient budgets (La Baugh 1986).

Water inputs

Water inputs to wetlands always include precipitation, and in bogs this represents the only significant source of water. Surface water inflows are important for wetlands affected by periodic inundation from flooding rivers, or where a wetland is intimately connected to a stream, such as for riparian wetlands. Wetlands formed at the base of slopes may also receive inflows from overland or subsurface flows. Groundwater inputs may be locally important, particularly for fens and swamps in low rainfall environments, or completely absent where wetlands are formed on impermeable materials such as clay, or where peat has accumulated to sufficient depths, because of its low permeability.

Precipitation is routinely measured by hydrologists using storage and automatic rain gauges. Placement of rain gauges in wetlands requires care, with due regard for proximity to tall or overhanging vegetation, surface flooding, and any exposure to wind that might lead to under-catch. For sub-alpine and alpine wetlands, snow may provide significant sources of water, with snowmelt delaying wetland recharge into spring.

Water outputs

Evaporation and interception loss

Evaporation may constitute the largest loss of water from a wetland, and it includes water transpired by living vascular plant tissues, evaporation of precipitation intercepted by the plant canopy, evaporation from open water surfaces, the moist substrate or soil, and evaporation from non-vascular plants, especially mosses.

There has been considerable confusion about the magnitude of evaporation rates from wetlands, due partly to the great diversity of wetland types and environments, as well as the complexity of the evaporation process. Also leading to confusion has been the wide range of techniques used to measure and estimate evaporation, many of them inappropriate (Ingram 1983). Evaporation from peat wetlands has been extensively reviewed by Ingram (1983) and Eggesmann *et al.* (1993).

The evaporation rate is highly dependent on several variables—energy supply (primarily solar radiation), saturation vapour pressure deficit (a measure of the “drying power” of the air) and wind (Chapter 3). Evaporation rates vary diurnally and seasonally largely because of the variation in solar radiation. The key factors limiting evaporation rates are those affecting water supply. While soil moisture deficits are uncommon in wetland soils, there are often strong surface constraints on water loss, with plant physiology and canopy structure being paramount among them.

Evaporation is rarely measured in hydrological studies, because the instruments are expensive and require

specialized knowledge to operate; the use of most techniques is restricted to extensive areas with homogenous surface conditions, which is a limitation in many wetlands. Methods for measuring actual evaporation include lysimeters and the Bowen ratio and eddy covariance techniques, both the latter being referred to as micrometeorological methods, because they rely on measurements made in the turbulent atmospheric boundary layer above the vegetation canopy. Eddy covariance is now used worldwide to make short- and long-term measurements of evaporation, energy balances and fluxes of trace gases, particularly carbon dioxide, in wetlands and other ecosystems. Wetland evaporation is commonly estimated from climate data using “potential evaporation” formulae (Chapter 3); however caution is required since measurements of evaporation in New Zealand wetlands have demonstrated that actual rates are generally much lower than the estimates obtained from these models (Campbell and Williamson 1997; Thompson *et al.* 1999).

Interception loss is difficult to measure for the dense vegetation present in many wetlands, hence its role in the hydrology of wetlands is little known. In northern New Zealand bogs the extremely dense canopies formed by the jointed rushes have received some attention, by Agnew *et al.* (1993), who attempted to measure stemflow, and by Campbell and Williamson (1997), who measured net precipitation under a dense wire rush canopy and estimated canopy storage capacity to be approximately 2 mm.

Surface water

For hydrologically isolated wetlands such as raised peat bogs, where water inputs are solely from precipitation, surface outflows may be difficult or impossible to measure because outflows are radial and flowing water may be intermittent or difficult to identify at the surface. In other types of wetland with channelised surface water inflows and/or outflows, such as fens, swamps, and pakihi, traditional hydrological measurements using stage-discharge relationships and weirs or flumes may be useful.

Surface water movement within wetlands in the form of permanent or temporary channels, flushes or pools can lead to variations in nutrient and sediment concentrations, and be a key factor in vegetation zonation.

Groundwater

In general wetlands are unlikely to act as significant recharge areas for groundwater systems. The reasons for this include the fact that fens and swamps often occur at the base of slopes or in valley bottoms where groundwater is emergent, and the deep peats of bogs act as confining layers separating these wetlands from underlying groundwater systems. Groundwater discharge from wetlands is most likely to be into adjacent water courses.

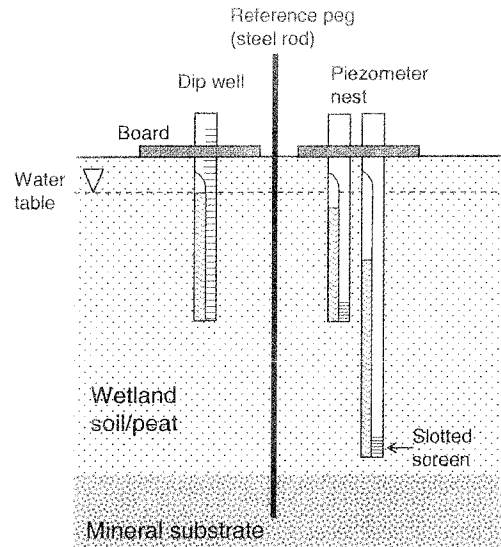


Figure 20.5 Construction of dip wells and piezometer nests for observing wetland water table position and hydraulic head gradients respectively. Cutaways show water levels inside tubes, with piezometers in this case indicating a downward head gradient, hence downward groundwater movement.

Groundwater interactions between wetlands and their surroundings are assessed using *piezometer nests* (Fig. 20.5). The directions of vertical groundwater movement may be determined at single sites using pairs of piezometers screened over discrete intervals at different depths. Three-dimensional flow directions must be determined by constructing flow nets based on equipotential contours (Chapter 29) constructed with measurements from multiple piezometer nests. Groundwater discharge is often estimated using numerical models; however these rely on assumptions that may not be valid in wetlands (Burt 1995). Because of the difficulty of measuring groundwater contributions to wetland water balances, they are often estimated as a residual term (e.g., Schallenberg *et al.* 2003).

Water storage and the water table

Dig a hole in a wetland soil, and it will almost invariably fill with water. The level to which the water rises defines the water table. In some cases the water table may lie above the ground surface for much of the year, or it will intersect depressions to form pools.

For many wetlands the most obvious hydrological factor is the position of the water table in relation to the ground surface. Fluctuations in the water table are more immediately apparent in wetlands than in other terrestrial ecosystems and water-table measurements often form the basis for environmental monitoring in wetlands, as a key indicator of the hydrologic regime in wetlands, or as a factor assumed to dictate other ecosystem processes, such as the distribution of vegetation types.

Variations in water storage in wetlands are usually indicated by changes in the water-table position. The water table also denotes a significant boundary within wetland soils. The zone beneath the lowest level reached by the water table during dry times is permanently lacking in oxygen, and decomposition of organic material occurs only slowly. The zone through which the water table fluctuates is periodically oxygenated and aerobic decomposition can occur.

The water table may be simply measured by digging a hole in the wetland substrate and observing the level to which it fills with water, or by installing *dip wells* and measuring the height of the water surface below the ground surface (Fig. 20.5). A dip well must be perforated or slotted along its entire length, whereas a piezometer is designed to measure pore water pressure, or *hydraulic head* at a specific depth below the water table (Fig. 20.5). Dip wells and piezometers may be simply constructed from PVC tubing and wrapped in mesh (such as shade cloth obtained from hardware stores) to prevent them clogging with fine sediments or peat. Wetland soils, particularly peats, are usually soft enough to allow dip wells and piezometers to be inserted by hand, often to depths of several metres. Water-table and hydraulic head measurements may be made manually with a tape measure or water contact sensor, or automatically by a range of modern sensors and dataloggers, including pressure transducers and capacitance probes.

Depending on the application for which measurements are made, water levels may be simply referenced to the ground surface, or to a local or regional datum. In general, for ecological purposes, for example to aid in the interpretation of vegetation patterns, it is usually sufficient to reference the water-table position to the ground surface. In order to map groundwater and surface flows, water levels must be measured relative to a local or regional datum, involving the use of very accurate differential GPS or leveling surveys. Pegs made from steel tube or rod may be pushed through soft wetland soils and hammered into underlying firm materials to provide a fixed reference (Fig. 20.5).

The measurement of water levels in wetlands is complicated in situations where the wetland substrate is unstable. Ingram (1983) describes the phenomenon of “bog breathing”, in which the surface of peat wetlands can rise and fall seasonally. At Whangamarino wetland in the Waikato, Hodge (2002) measured variations in the peat surface of up to 163 mm over an eight-month period.

The wetland water table responds to several variables, illustrated by 30-minute measurements obtained within Maher Swamp, a coastal-dune peat wetland on the Barrytown Flats near Punakaiki, Westland (Fig. 20.6). The water table was close to, or above, the surface during this period, and responded very rapidly to inputs of rainfall. When the water table was beneath the surface, its response was magnified compared to rainfall—particularly evident on days 28, 40 and 44 in Figure 20.6. Since wetland soils above the water table are close to saturation, small additions of water from rain result in magnified water-table rises. The large rainfall event on days 47–49 caused floodwaters from nearby streams to spread into the wetland. The water table declined rapidly after rain, especially when surface runoff occurred around day 50. During rainless periods (days 35–40 and 42–44) the rate of water-table decline below the ground surface was characteristically stepped, with the most rapid rates of decline occurring during daytime. This behaviour is caused by evaporation during the day resulting in more rapid drawdown of the water table. Bowden *et al.* (2001) also observed this behaviour in a fen at Glendhu, Otago, and used it to estimate daily evaporation rates (Fig. 20.4c).

Soil moisture

There are very strong linkages between the water table and soil moisture content when the water table is shallow because water lost to evaporation is rapidly replaced by capillary rise. Shallow water tables sustain peat volumetric moisture contents up to 90% (Thompson *et al.* 1999; Campbell *et al.* 2002). Once the water table drops to low levels, for example during times of drought or when artificial drainage of adjacent land affects the wetland water

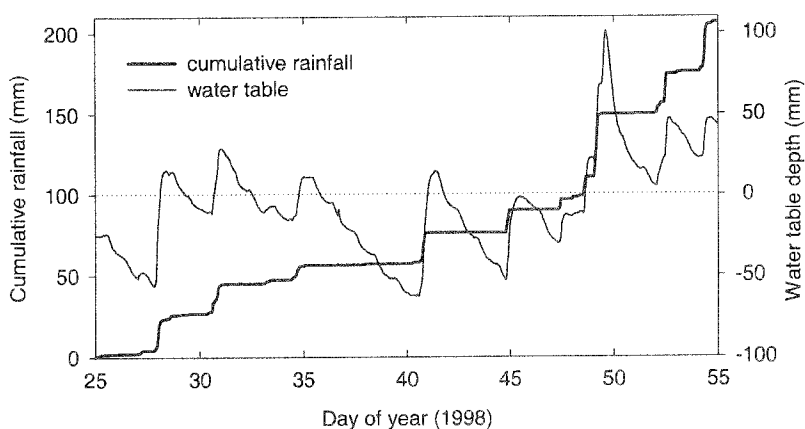


Figure 20.6 Water table regime and cumulative rainfall for a 30-day period during summer 1998 at Maher Swamp, Westland. Both rainfall and water table depth are plotted to the same scale. The horizontal dotted line represents the approximate ground surface. (Unpublished data, University of Waikato).

table, surface soils may become very dry and matric potentials decrease dramatically once the larger soil pores are emptied. At the Moanatuatua peat bog south of Hamilton, Campbell *et al.* (2002) found near-surface volumetric moisture contents between 14–30%, which remained low despite large rainfall events that caused rapid rises in the 0.3–0.4 m deep water table.

Measurements of soil moisture content may be obtained by destructive sampling using the gravimetric technique, or by indirect methods such as Time Domain Reflectometry (TDR) and heat pulse probes. Care should be exercised using reflectometry-type instruments, because factory-supplied calibration coefficients may not be valid for organic soils. Wetland soils have extremely variable dry bulk densities, with peat in particular having very low density (Waikato peats range from 30–70 kg/m³, Campbell *et al.* 2002), thus moisture content should always be expressed on a volumetric basis.

Peat soils shrink and swell with water losses and gains, leading to seasonal variations in ground surface levels in wetlands, and to problems with irreversible shrinkage and compaction when peat is drained for agriculture (McLay *et al.* 1992).

Hydraulic properties of wetland soils

The hydraulic properties of wetland soils are important because they determine the relationship between changes in water storage (ΔS) and water-table fluctuations via the *specific yield*, as well as rates of subsurface flow and interactions between the wetland and regional groundwater systems. Specific yield is defined as the yield of water, expressed as a depth, which results from a unit lowering of the water table (Ingram 1983). When the water table is above the ground surface, the specific yield will be close to unity (depending on standing vegetation density); the yield will be less than unity below the surface and typically reduce with depth as soil pore sizes decrease.

The properties of wetland soils change with depth, particularly in peatland systems, because peat at depth has undergone greater amounts of decay and compaction. The

state of decay of peat is commonly assessed in the field using the 10-point von Post scale (Clarkson *et al.* 2003), which has often been related to the hydraulic properties of peat (Charman 2002). As peat degrades, pores decrease in size and the permeability, or hydraulic conductivity, typically declines. King (1999) studied the variation in hydraulic conductivity with depth at the Kopouatai peat bog in the Hauraki Plains, and concluded that most water movement occurred in the near-surface peat (Fig. 20.7).

Most groundwater models rely on assumptions about water movement in saturated materials, given by Darcy's Law (Chapter 29). Peat has been attributed non-Darcian properties, because pores may be blocked by gas bubbles generated by microbial activity, which retard water flow (Burt 1995).

The role of wetlands in runoff generation and flood control

The role of wetlands in attenuating floods and as source areas for surface runoff has been keenly debated (Fahey *et al.* 1998).

Wetlands may be viewed as either source areas or sinks for runoff (Burt 1995). Extensive lowland wetlands may act (by nature or by engineering design) as spill areas capable of temporarily storing flood water, thereby reducing possible serious threats of large floods to developed landscapes and infrastructure. In this regard wetlands are little different to lakes, as their ability to attenuate floods depends on the area available for inundation.

Historically, headwater wetlands were regarded as hydrologically important for attenuating floods and sustaining baseflows through dry periods, as if they were sponges capable of soaking up a large volume of water then releasing it slowly (Burt 1995). In reality, wetlands remain wet because their soils generally have such low hydraulic conductivity that they release water only slowly, or because they receive water inputs from surrounding hillslopes or aquifers. Because they remain close to saturation for much of the time, wetland soils have a very

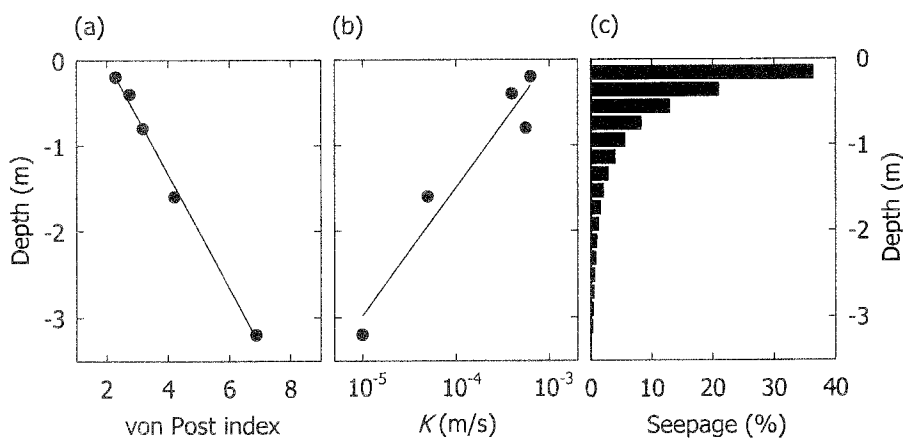


Figure 20.7 Variation in peat physical and hydraulic properties with depth for Kopouatai bog, Hauraki: (a) peat decomposition states given by the von Post scale (b) hydraulic conductivity measured *in situ* using a piezometer method (c) percentage of lateral flow occurring within 0.2 m increments of depth, calculated using Darcy's law. Modified from King (1999).

limited capacity to store additional water from rainfall, let alone flood waters flowing in from outside, as indicated by the magnified reaction of the water table to rainfall in Maher Swamp (Fig. 20.6). Hydrographs for headwater catchments dominated by wetlands, whether swamps or peatland, tend to be flashy compared to catchments with deep mineral soils or extensive aquifers.

Fahey *et al.* (1998) and Bowden *et al.* (2001) reported on hillslope runoff mechanisms for a headwater catchment that included a small peat wetland at Glendhu, Otago. They concluded that the wetland did not contribute enough water to sustain the high rates of baseflow observed in streams; rather the wetland was a pathway for water released slowly from the unsaturated soil horizons of the surrounding hillslopes. This wetland had little regulatory effect on total storm runoff, peak flows or lag times, and during large storms the water table quickly rose to the surface, so that the wetland became a source area for saturation overland flow (Fahey *et al.* 1998). Where high rates of baseflow are recorded from wetlands, they are invariably acting as conduits for runoff from larger hydrological systems further upslope (Burt 1995; Fahey *et al.* 1998).

Jackson (1987) found that runoff from pakihi wetlands at Larry River near Reefton was dominated by quickflow, contributing 70% of the 1600 mm annual runoff (Fig. 20.4d). Drainage construction for forest planting increased the frequency of higher flood peaks. The water table in these wetlands was observed to rise to the surface very rapidly during rainfall, leading to widespread surface ponding and overland flow. Pakihi is also incapable of sustaining substantial baseflows, with only 10 mm of water yield being measured over a 19-day rain-free winter period.

HYDROLOGICAL REGIMES OF NEW ZEALAND WETLANDS

Rain-fed wetlands

Bogs represent an extreme end member in the spectrum of wetland types. A bog is a peat wetland that has evolved to a point where the sole input of water is from precipitation. Since meteoric water contains very low concentrations of key nutrients such as nitrogen and phosphorus, peat bogs are oligotrophic.

Bogs occur as raised bogs or as blanket bogs (Fig. 20.1). Raised bogs generally form in depressions or on flat land, where peat accumulates in sufficient thicknesses to raise the bog surface above its surroundings. Most of New Zealand's remaining raised bogs are in Waikato, Westland and Southland. Blanket bogs form in cool, moist environments so there is an excess of water available from precipitation. In these climates peat may even accumulate on hillslopes. Good examples of blanket bogs are found in

the southern South Island, on the Chatham Islands and the subantarctic islands. The Awarua Plains blanket bog south east of Invercargill covers an area of approximately 120 km².

Raised peat bogs have precipitation as the sole water input, and water outputs are via evaporation and lateral seepage and occasional surface flows. Figure 20.4b shows the annual water balance for the remnant Moanatuatua bog in the Waikato, where there are no surface water outflows because the water table is abnormally low for a bog, partly because of the drainage of surrounding farmland.

Northern New Zealand peat bogs are unusual because they have formed under relatively warm and dry climate conditions, outside the climate zones generally considered conducive to the formation of deep peat deposits. Their vegetation is dominated by the indigenous jointed rushes, including *Empodisma minus*, the major peat-forming plant in New Zealand bogs (Fig. 20.3). Its dense mat of roots with fine hairs growing above the peat surface forms the bulk of the peat. The dominance of vascular plants in northern New Zealand bogs sets them apart from most of the world's raised bogs, which are dominated by non-vascular mosses, and allows a much greater influence of plant physiological controls on water loss via transpiration.

Bog morphology

Peat accumulates within bogs to such an extent that the water table is raised above the influence of both the regional water table and mineralized runoff (Fig. 20.1). In this situation the water becomes acidic due to the high cation exchange capacity of peat and peatland plants, particularly *Sphagnum* mosses, and in New Zealand the roots of *E. minus* (Agnew *et al.* 1993), removes cations from solution and releases hydrogen ions. Organic acids also contribute to lowering pH (Charman 2002).

In situ peat consists almost entirely of water. Campbell *et al.* (2002) measured Kopouatai bog peat—the unsaturated peat had a volumetric moisture content of up to 88%, and the saturated peat up to 97%, with an organic volume fraction of 3% and a negligible mineral fraction. A raised peat bog, then, has more in common with a lake than a terrestrial soil system, except that it is a lake with its surface sloping downward away from the centre, and which may be comfortably walked upon (Fig. 20.2e).

It was once believed that the raised water table in bogs was caused by capillary rise within the small pores of peat; however this does not explain the domed shape of many bogs, and calls for unreasonably small pore sizes (Charman 2002). The elliptical cross-sectional shape of many raised bogs is now thought of as a groundwater mound, the shape of which has been predicted from the water balance and estimates of hydraulic conductivity (Ingram 1983). Mark

et al. (1979) presented surface topography and water table data for the Borland bog in Southland, which clearly showed this doming.

Raised peat bogs are conventionally divided into two hydrological zones (Fig. 20.1). The bulk of the water is stored within the underlying, more highly decomposed peat of the *catotelm*, a relatively inert zone characterized by low hydraulic conductivity and very slow rates of water and nutrient movement. The *acrotelm* is the layer of peat near to the bog surface through which the water table fluctuates on an annual basis. This zone is periodically aerated and decomposition rates are greater than within the *catotelm*. The younger peat within the *acrotelm* has a lower bulk density, larger pore spaces and a relatively high hydraulic conductivity, so that most water transmission occurs close to the surface, where lateral flows predominate (Fig. 20.7c).

The outside edge of a raised bog, referred to as the *lagg* (Fig. 20.1), is affected by mineralised runoff or groundwater, so that peat type and vegetation associations are often more typical of fen or swamp environments.

Evaporation from bogs

Unusually strong controls on water loss by evaporation have been found for northern New Zealand peat bogs dominated by the jointed rushes *Empodisma minus* and *Sporadanthus ferrugineus*, which are members of the restiad family (Clarkson 1997; de Lange *et al.* 1999). These unusual plants have xerophytic (water-conserving) properties, owing to their arid-land ancestry in southern Africa and Australia. Campbell and Williamson (1997) found that evaporation rates from a bog surface covered in a dense canopy of *E. minus* were highly conservative and much lower than rates found for Northern Hemisphere bogs. The dense canopy effectively prevents solar radiation from reaching the underlying peat surface, and acts like a thick mulch preventing water loss from the moist peat. The plants themselves have extremely low transpiration rates and the ecosystem has been paradoxically described as a “wet desert” because the low evaporation rates cause the microclimate to resemble that of an arid environment (Campbell and Williamson 1997).

Figure 20.8 shows a comparison of measured daily evaporation for three New Zealand wetland vegetation types found in swamps and bogs. Evaporation rates from *E. minus* are extremely low compared to other bog and swamp vegetation, however, even sedge/flax vegetation in Maher Swamp, Westland, had evaporation rates well below what is expected of well-watered grassland, given by the Priestley-Taylor evaporation rate on Figure 20.8 (McAnaney and Judd 1983).

It is highly likely that the water-conservative adaptations of *E. minus*, in particular, are key factors in providing a

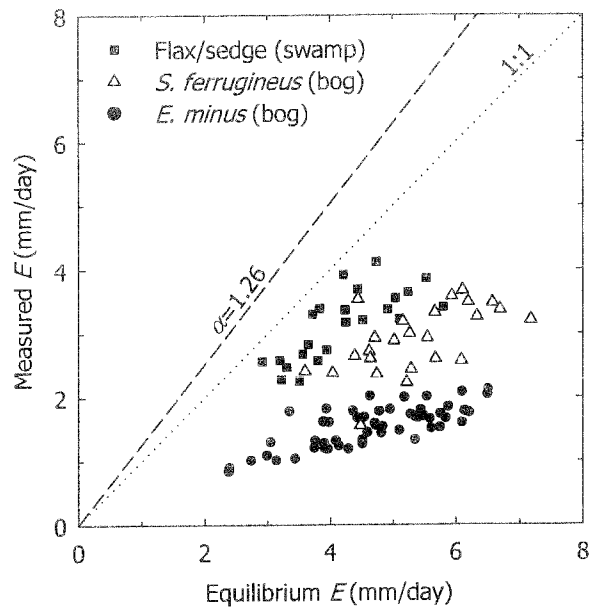


Figure 20.8 Measured summertime daily evaporation rates for three New Zealand wetland vegetation types, compared to calculated equilibrium evaporation. Flax/sedge in Maher Swamp, Westland (Unpublished data, University of Waikato); *S. ferrugineus* and *E. minus* in Waikato peat bogs (data from Campbell and Williamson 1997 and Thompson *et al.* 1999). Dashed line indicates Priestley-Taylor evaporation rate, with $\alpha = 1.26$.

high, stable water-table regime in northern New Zealand bogs, thereby encouraging peat accumulation (Campbell and Williamson 1997).

Water table and groundwater regimes

The water table regime of peatlands has long been linked to the processes of peat formation, whereby it is assumed that a high stable water table minimizes aerobic decomposition rates, encouraging the preservation of organic material (Clymo *et al.* 1998). Several overseas studies have confirmed that lowering the water table, either via artificial drainage or climate change, reduces rates of carbon accumulation (e.g., Oechel *et al.* 1998; Roulet *et al.* 1998). This simple scenario cannot always be demonstrated, however, because a lowered water table can also lead to higher rates of primary productivity. Hilbert *et al.* (2000) presented a more complicated model of peat formation where, for example, rates of peat accumulation may actually increase as the water table lowers.

The water table regimes for three Waikato peat wetlands during 2001 are shown in Figure 20.9. Kopouatai, a relatively pristine bog, has a water table that remains close to the peat surface throughout the year. At Whangamarino, a fen or young bog, there is even less variation in the water table, especially during late winter and early summer, when

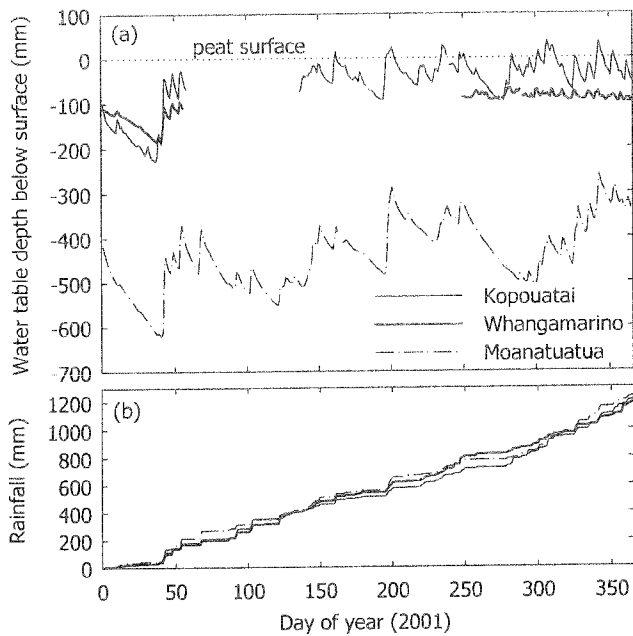


Figure 20.9 Annual time series of half-hourly water-table depth below the peat surface and cumulative rainfall totals at three Waikato peat wetlands in 2001 (Hodge 2002). Missing data are caused by equipment failure.

measurements of surface elevation relative to a datum suggested that the peat was actually floating. Compared to these high stable regimes, at Moanatuatua bog the water table is unusually deep, possibly as a result of the intensive drainage of adjacent farmland or successional changes leading to vegetation types having greater evaporation rates (Thompson *et al.* 1999).

Also evident from Figure 20.9 are the magnified reactions of the water table to inputs of rainfall, followed by relatively rapid declines during dry periods. The deep water table at Moanatuatua reacts most to rain inputs and drainage outputs, because the deeper, more degraded peat at this site (Shearer 1997) has a lower specific yield. This illustrates a key influence of wetland soil factors on the hydrological regime.

Vertical head gradients in Waikato peat bogs have been investigated by Maggs (1997), King (1999), Grimshaw (2000) and Hodge (2002). Within the Whangamarino and Kopouatai bogs, which are close to the level of major rivers and the sea, vertical head gradients appear to be negligible. At Moanatuatua bog, elevated 40–50 m above the Waikato River and 5 km distant, a downward head gradient exists in parts of the bog, possibly where thinner sealing layers at the base of the peat allow the regional head gradient in the underlying mineral sediments to be expressed within the bog (Grimshaw 2000).

Shearer (1997) suggested that differences in the hydrogeological setting among the Waikato bogs could be responsible for marked differences in the stratigraphy of peat degradation, because it may affect the water-table

regime. Shearer and Clarkson (1998) hypothesised that recent peat degradation at Whangamarino wetland could have been driven by water-table declines caused by historic lowering of bed levels in the nearby Waikato River.

Pakihi wetlands

Johnson and Gerbeaux (2004) introduced a wetland class comprising pakihi and gumlands. Pakihi land is widespread on the west coast of the South Island (Mew 1983), and often has a secondary vegetation cover of manuka, ferns (especially *Gleichenia dicarpa*), restiads and mosses that has replaced a previous podocarp-hardwood forest because of disturbance, including fire. Many sites referred to as pakihi can be classified as bog or fen (Johnson and Gerbeaux 2004), but others are on mineral soils with peaty topsoils 10 to 30 cm thick. Pakihi wetlands occur on flat to rolling land with poorly drained and acid (*ca.* pH 4) soils of very low fertility. Like bogs, pakihi wetlands receive water only from rainfall, but the topography and the very low permeability (*ca.* 10^{-8} m/s) of the mineral soil horizons result in outflow almost entirely to streams (Jackson 1984, 1987). The rainfall is high (>2000 mm/year) and frequent throughout the year, so that a water table is present 10 to 30 cm below the surface most of the time. It requires only 10 to 20 mm of rainfall to bring the water table to the surface and rapid runoff occurs overland and through the upper peaty topsoil. Over 70% of runoff is quickflow, but flow rates decline to less than 1 mm/day within a day or two after flood events, reaching <0.1 mm/day after two weeks without rainfall. The stream waters are typically brown, with high concentrations of dissolved organic carbon and a pH of less than 4.0 (Collier *et al.* 1990). A catchment water balance for three years at Larry River showed an average rainfall of 2450 mm and a stream outflow of 1600 mm (Fig. 20.4d). Assuming other losses to be negligible, the annual evaporation was 850 mm, of which interception and transpiration were estimated to each contribute about half. Whitehead *et al.* (2002), working on a similar wet and infertile soil in south Westland, but in pristine, mature podocarp/hardwood forest dominated by rimu, obtained a value of 450 mm for the combined tree transpiration and understorey evaporation losses. Thus although frequent rainfall and low permeability of the mineral subsoil are the main causes of persistent wet conditions at these sites in Westland, the low transpiration rates and low fertility resemble those of bogs.

Surface water and groundwater-fed wetlands

While bogs form distinctive landforms in their own right, wetlands dominated by surface water and groundwater inputs have diverse characters, with gradations in nutrients, substrates, vegetation and landscape positions. In this section key aspects of the

hydrological regimes of fens and swamps are described, illustrated by the results of the relatively few hydrological studies conducted in these environments.

Fens

Fens are peat-dominated wetlands that occur in a wide range of environments, ranging from bog margins and swamp perimeters, to hillslope toes and alluvial fans (Johnson and Gerbeaux 2004). They may also exist on level ground where peat accumulation is insufficient to hydrologically isolate them from the underlying mineral substrate. Fens may therefore be considered transitional environments, with a range of water sources and a nutrient status ranging from oligotrophic to mesotrophic. Their hydrological conditions are thus extremely variable, but more dependent on the influence of external hydrological systems than bogs. New Zealand fens have vegetation composed of sedges, restiads, ferns, tall herbs, tussock grasses and scrub (Johnson and Gerbeaux 2004).

Groundwater is often a key water input to fens, and this can be a factor leading to a relatively stable, high water table. Eser and Rosen (1999) described the groundwater hydrology of the Stump Bay wetland complex along the southern shores of Lake Taupo, an 8 km² wetland with peat deposits amongst relict beach ridges. The wetland is characterized by through-flowing or discharging groundwater with high concentrations of iron and, based on water chemistry, can be subdivided into two parts, a western section affected by seepage water from the Tongariro River, and an eastern section fed by ground water discharging from its source on the Turangi Plains. Nutrient concentrations are generally low, indicating a fen environment. Eser and Rosen (2000) studied the effect of the artificial control of Lake Taupo levels on the wetland. Significantly higher summer lake levels have led to an increase in wetland area and a higher water table, which in turn has led to changes in vegetation communities, possibly encouraging the invasion of grey willow (*Salix cinerea*).

The small headwater fen at Glendhu, Otago, studied by Fahey *et al.* (1998) and Bowden *et al.* (2001) (Fig. 20.2c), acts as a conduit for shallow subsurface water flows from deep mineral soils on the surrounding hillslopes, and on its own is not a significant source of baseflow for the first-order stream draining it. The water balance of this wetland over the course of a 10-day rainless period is shown in Figure 20.4c, and indicates that surface flows out of the wetland were provided by the difference between subsurface inflows and evaporation, with the wetland water storage declining only slightly. With a total catchment area nine times larger than its own surface area, this wetland highlights a key difference between bogs and fens, in that the volume of water throughput can be dominated by inflows rather than solely by precipitation.

Dickinson *et al.* (2002) describe the hydrology and water chemistry of some unusual patterned fen complexes in the Garvie Mountains, Otago, and used these to explain the distribution of vegetation types. They hypothesised that the relatively recent genesis of these *Sphagnum*-peat wetlands was caused by an increased water yield from the catchment once the original forest cover was burned and replaced by tussock, *ca.* 700 years ago. Water availability (expressed as water table elevation or peat moisture content) was significantly related to vegetation patterns. Ca/Mg ratios indicated a mesotrophic to marginally ombrotrophic nutrient status, suggesting a mixture of ground/soil water contributions in addition to meteoric water. These fens have pools that appear to be linked by networks of naturally occurring pipes, with the result that water levels in the pools fluctuate wildly, with some pools filling and others emptying. Dickinson *et al.* (2002) hypothesised that runoff from surrounding land may be diverted underneath the wetlands via these pipes.

Some fens eventually evolve into raised bogs, once sufficient peat has accumulated to isolate them from inputs of mineralised surface runoff and groundwater. Examples of such transitional fens are to be found in the north Waikato: Whangamarino and Opuatia wetlands are prime examples (Fig. 20.2d).

Swamps

Swamps occur in low-lying areas such as valley floors, deltas, plains, and the margins of lakes and rivers. Both mineral and peat substrates can occur, with flood waters providing a rich source of sediments and nutrients. Their vegetation can be extremely variable, ranging from sedge, rush, reed and flax, to scrub and forest (Fig. 20.2b).

Swamps generally receive only a small portion of their water inputs from rain; hence their hydrological regimes are heavily influenced by external runoff processes, especially flooding, as was illustrated for Maher Swamp on days 47–50 in Figure 20.6. The Bullock Creek swamp near Punakaiki, North Westland, is flooded several times each year by streams draining from adjacent hill land. In a 12-month period in 1999–2000 six floods covered the central part of the swamp to depths ranging from 200 to 1500 mm, providing a total of 5000 mm of water input in addition to the 3000 mm of annual rainfall (Landcare Research, unpublished data).

The water table within swamps may be permanently or periodically above the surface, with leads of standing water allowing internal flows of surface water. The water-table regime tends to be far more variable than those of bogs and fens, owing to the often large influxes of flood waters.

Riparian swamps are fairly common in New Zealand, but in general they are heavily modified by additions of nutrients from agricultural runoff, and are often infested

with weeds such as willow. Their role in nutrient processing is highly valued, particularly denitrification, the efficiency of which is highly dependent on water pathways and residence times. Peacock (1998) and Tomer *et al.* (1999) studied the role of small riparian swamps at Whakarewarewa Forest near Rotorua in the removal of nitrate from municipal wastewater effluent applied to land within its catchment. Groundwater inflows form the largest proportion of water inputs to these wetlands. While wetland organic substrates have a large capacity for nitrate removal, actual denitrification was hindered by the short residence time of water flowing through the wetlands.

Chagué-Goff *et al.* (1999) reported on changes to part of the South Taupo wetland in response to sewage effluent discharges, and increased flooding frequency from the Tongariro River caused by lake level changes and increased erosion in the upper catchment. The wetland acts as a sink for nutrients and metals derived from wastewater, which has resulted in significant changes to the vegetation, water quality and sediment chemistry. Inputs of mineral sediments from floods in recent decades have led to the cessation of peat accumulation, illustrating how changes to hydrology and land use may cause a shift in wetland type. Cores taken through wetland substrates can be used to infer changes in hydrological conditions throughout a wetland's history.

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Chapter 19

Geomorphology and hydrology of lakes

Paul Mosley

INTRODUCTION

This chapter, which introduces the geomorphology and hydrology of New Zealand's lakes, has drawn on a number of valuable sources of information, including textbook chapters by Lowe and Green (1987, 1992) and Spigel and Viner (1992), the *Inventory of New Zealand Lakes* (Livingston *et al.* 1986), and the *Lake Managers Handbook* (Vant 1987; James *et al.* 2002). Subsequent chapters consider the chemical and physical characteristics of lake water, and the biota and ecosystems that are characteristic of lakes.

Lakes may be defined as “bodies of standing water occupying a basin and lacking continuity with the sea” (Lowe and Green 1992). Although they cover a total area of less than 3,500 km² (1.3% of the country's land area) they are a distinctive element of the landscape in many parts of New Zealand. They have immense intrinsic value as natural features, with characteristic physical processes, ecosystems and biota. They have incalculable social and economic value, as places that people choose to live beside and visit, as tourist attractions, and as a source of water for domestic and industrial use, irrigation, and hydropower. Lakes may mitigate the severity of flooding along rivers downstream, as they store and release floodwaters coming from the headwater catchments, and they modify sediment and solute concentrations in the rivers downstream.

In New Zealand, lakes are managed within the provisions of the Resource Management Act, in which they usually are referred to in conjunction with the coastal marine area, wetlands, and rivers. Hence, Section 6 of the Act requires that all persons exercising powers and functions under it:

“...shall recognise and provide for the following matters of national importance:

(a) The preservation of the natural character of the coastal environment (including the coastal marine area),

wetlands, and lakes and rivers and their margins, and the protection of them from inappropriate subdivision, use, and development.”

This clause is the first to specify a “matter of national importance” in Part II (Purpose and Principles) of the Act, which is a clear signal of the importance accorded by Parliament to preserving and protecting lakes, among other water bodies. A sound knowledge and understanding of lakes and their characteristics is essential for achieving the purpose of the Act.

ORIGINS OF LAKES

Lakes form when a geomorphological process or processes create a depression and surrounding rim that catches and retains water (Lowe and Green 1992). The processes may be “constructive”, when the rim is built up, “destructive” when the basin is excavated, or “obstructive” when a pre-existing landform is blocked. Lakes are, in geological terms, transitory features of the landscape. In New Zealand, most lakes were formed as recently as the late Pleistocene or Holocene, and in their present form are less than 15,000 years old. Hence, for example, most of the lakes of glacial origin in the South Island date from the period of de-glaciation during the last 10,000 to 12,000 years. Lakes of volcanic origin in the North Island have a range of ages, depending on the dates of eruptions—modern Lakes Okataina and Tarawera are about 7,000 and 5,000 years old, respectively. Ultimately, all lakes are destined to disappear, as they fill in with sediment, or their outlet is cut down to the extent that the lake is drained. Holocene shorelines tens of metres above the present level of many glacial lakes, e.g., along Lake Ohau in the Waitaki catchment, attest to the speed with which lakes can be modified. Similarly, the extensive deltas at the head of many of these lakes, e.g., Lake Pukaki (Fig. 19.1), are

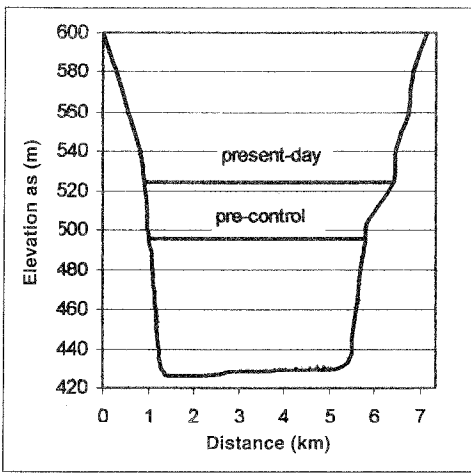
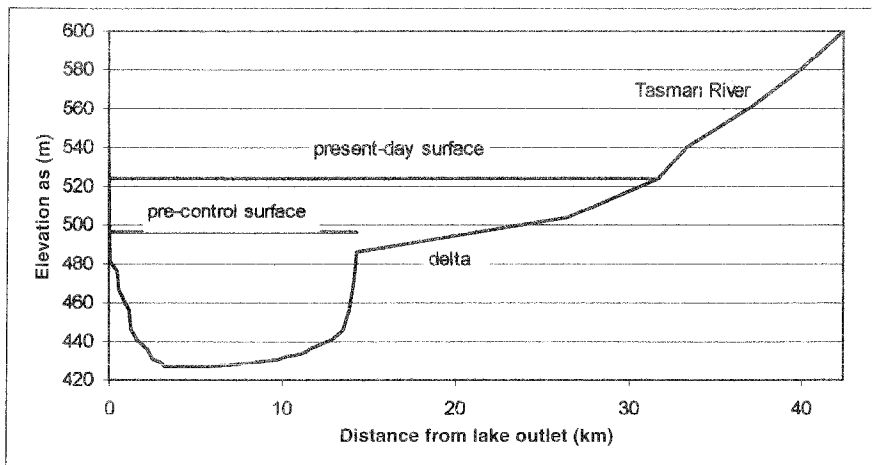


Figure 19.1 Longitudinal and transverse profiles of Lake Pukaki, showing the flat-bottomed glaciated cross-section and the advancing delta at the head of the lake, which is a submerged extension of the Tasman River. Construction of a barrage and control gates for hydro-electric power generation has raised lake level by over 33 m and drowned the delta. Depth data are taken from the Lake Pukaki bathymetric chart (Irwin 1972b).

evidence also of rapid infilling. However, on the human time scale such a fate awaits only a few of New Zealand's lakes and reservoirs, principally those whose catchments produce large volumes of sediment, or which have been dammed by easily eroded landslide debris.

Various classifications of lakes have been proposed, perhaps the most comprehensive being the 76 modes of origin identified by Hutchinson (1957). Lowe and Green (1987) developed a simplified classification suitable for New Zealand's lakes, many of which have been created by a combination of processes (Table 19.1). Particular lake-forming processes tend to be characteristic of different areas of the country, so distinct "lake districts" can be identified (Fig. 19.2). Thus, for example, volcanic lakes are found principally in the Central Volcanic District of the North Island, the majority of glacial lakes are found in the axial ranges of the South Island, while landslide lakes are found in hilly or mountainous areas, particularly near active fault lines. Overall, glacial lakes are most common, 38% of the total, followed by riverine lakes (16%) and dune lakes (15%). Nearly 10% of New Zealand's lakes are artificial, but this figure does not include the thousands of farm ponds that have been constructed throughout the country in recent years.

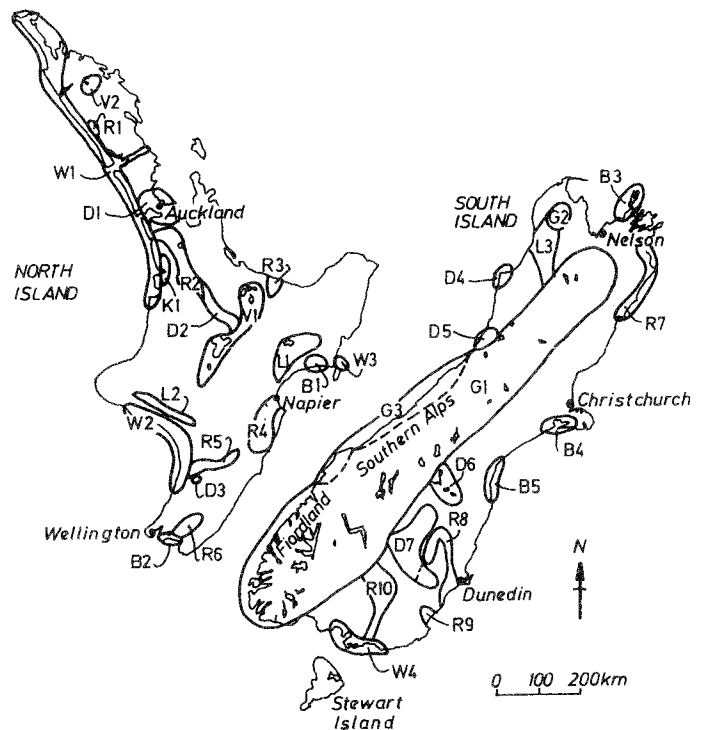


Figure 19.2 The distribution of major lake types, classified by geomorphological process (from Spigel and Viner 1992; after Lowe and Green 1987). Letters indicate lake types and numbers indicate particular localities: B: Barrier; D: Reservoirs; G: Glacial; K: Karst; L: Landslide; R: Riverine; V: Volcanic; W: Sand-dune.

Table 19.1 Classification of New Zealand lakes based on mode of formation (after Lowe and Green 1987).

Lake type	Number ¹	Mode of formation
Glacial (G)	289	Glacial scouring and/or fluvio-glacial deposition
Riverine (R)	121	Creation of depressions by channel migration and abandonment
Dune (D)	119	Deposition of barriers by wind-blown sediment
Landslide (L)	40	Blockage of valleys by landslide debris
Barrier bar (B)	32	Deposition of barriers by longshore coastal sediment transport
Volcanic (V)	30	Volcanic eruptions and deposition of barriers by lava flows
Tectonic (T)	6	Creation of depressions or barriers by faulting or folding
Solution (S)	1	Dissolution of carbonate rocks and ground collapse/subsidence
Peat (P)	0	Subsidence in peat areas or peat accumulation
Artificial (A)	65	Human activity, normally damming or excavation
Undetermined	72	
Total	775	

¹ Note: includes only lakes with a length >0.5 km

Glacial lakes

Glacial lakes are found only in the South Island, particularly in and at the margins of the formerly or presently glaciated axial ranges. Most are “fossils”, but some, such as Tasman Lake at the terminus of the Tasman Glacier, are active pro-glacial lakes. Several different sub-types of glacial lakes have been identified in New Zealand, including:

- Lakes in ice-excavated rock depressions, of which there are two main groups: *cirque lakes* found in the rock amphitheatres gouged out by mountain glaciers (e.g., Ivory Lake, Waitaha River catchment, Westland), and *piedmont lakes*, which occupy the U-shaped valleys created by valley glaciers (e.g., Lake Rotorua, Nelson).
- Lakes held back by glacial moraine (e.g., Lake Brunner, Westland), fluvio-glacial outwash (e.g., Diamond Lake at the head of Lake Wakatipu), or alluvial fans (e.g., Lake Grasmere, in the Waimakariri River catchment).
- *Kettle lakes*, formed in depressions created by the melting of ice blocks left stranded in moraine by retreating glaciers (e.g., Lake Marymere, upper Waimakariri catchment).
- *Pro-glacial lakes* at the toe or margin of a currently active glacier (e.g., Douglas Glacier lake, Karangarua River catchment, Westland).

Glacial lakes may be created by combinations of processes, including non-glacial processes such as the deposition of fluvial sediments. For example, Lake Wakatipu occupies an ice-gouged valley, but the rock rim of its rock depression is capped by 75 m of glacial moraine, to provide a total maximum lake depth of 380 m.

“Actively glacial” lakes such as Ivory Lake have distinctive hydrological regimes that reflect the seasonal progression through winter snow accumulation, spring snow-melt, and summer rainfall (Chapter 6, Fig. 19.3). The regimes of “fossil glacial” lakes reflect glacierisation to the degree that their catchments presently are snow- and ice-covered. For example, the regimes of Lakes Tekapo, Pukaki and Ohau in the Waitaki basin would reflect precipitation, snow and ice cover under natural conditions, but their levels and outflows now are controlled for hydro-electricity generation.

Volcanic lakes

Most of New Zealand’s lakes with a volcanic or tectonic origin are found in the three main areas of recent volcanism, the Taupo Volcanic Zone, the Auckland Volcanic District, and the Bay of Islands-Kaikohe Volcanic District (Fig. 19.2). Although the larger lakes, such as Taupo and Rotorua, have existed and have been evolving for more than 100,000 years, most are only a few thousand years to a few tens of thousands of years old, and the youngest, Lake Rotomahana, a crater lake near Rotorua, dates from the Tarawera eruption of 1886. Lowe and Green (1987, 1992) provide detailed surveys of the history, geologic setting, and mode of formation of the volcanic/tectonic lakes. There are three principal types of volcanic/tectonic lake, with the geological processes sometimes acting together:

- *Caldera lakes*, which occupy caldera basins created by ground collapse and faulting following magma withdrawal during massive eruptions (e.g., Lake Taupo,

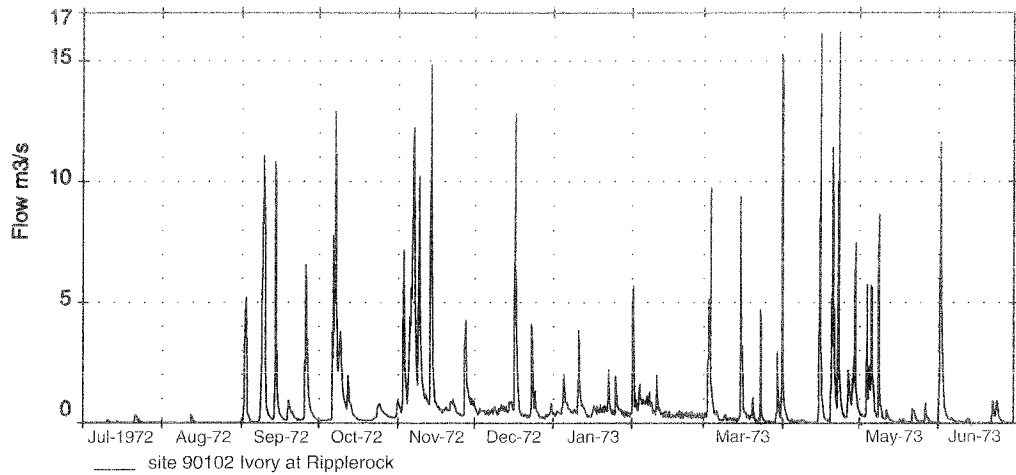


Figure 19.3 The outflow hydrograph for Ivory Lake typifies the hydrologic regime of a glacial lake. (Plot provided by Kathy Walter, NIWA). There is virtually no outflow during the winter months of June-August, then outflow slowly rises to a maximum in November-January under the influence of snowmelt and rain-on-snow storms. Baseflow declines thereafter, with rainstorms causing fashes through into May.

created by a series of eruptions over the last 100,000 years, and particularly during the last 22,000 years; Fig. 19.4).

- *Crater lakes*, which occupy the craters created by explosive eruptions (e.g., Crater Lake on Mt Ruapehu, which is located in an active vent and is 1,800–2,000 years old).
- Lakes created when lava flows or pyroclastic deposition dams a valley (e.g., Lake Rotoehu, near Rotorua, created about 8,500 years ago during the Rotoma eruptive

episode, and Lake Pupuke, on the North Shore of Auckland, which is probably over 40,000 years old).

The hydrological regime of a volcanic/tectonic lake will reflect the climatic and geological characteristics of its catchment (Chapter 6). Groundwater and geothermal water flows are particularly significant in active volcanic zones, because of the highly fractured and faulted nature of the bedrock, and the extensive deposits of highly permeable volcanoclastic sediments and lava

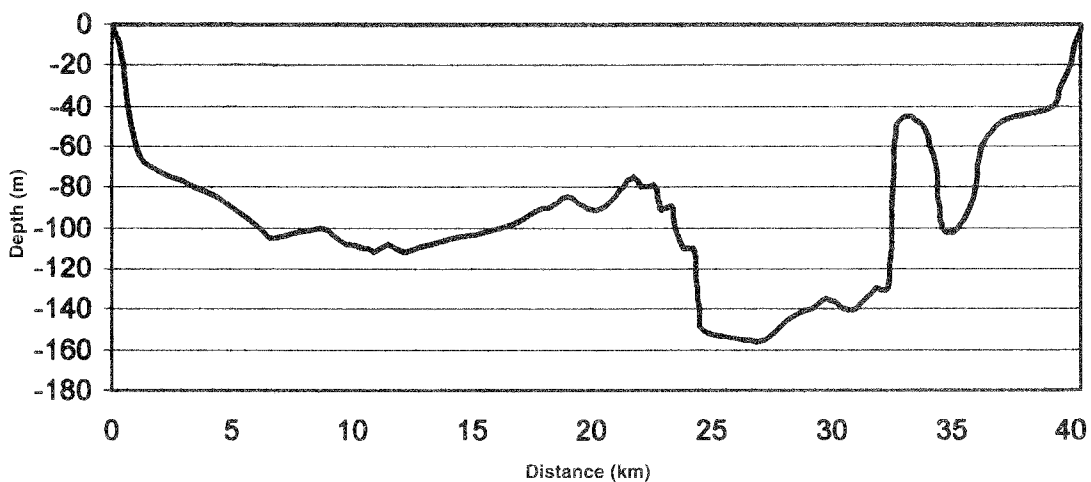


Figure 19.4 Long-profile of Lake Taupo, a “caldera lake” that is New Zealand’s largest lake. The profile extends across the line of maximum lake length, from the SW shore at Waihi to the NE shore at Taupo. Depth data were taken from the Lake Taupo Bathymetry chart (Irwin 1972a).

Other geological processes

Most other lakes are formed in depressions created—either by excavation or by construction of a barrier—by rivers, shoreline processes, aeolian (wind) action, landslides, solution, or peat development.

Small, shallow *riverine lakes* commonly form on river floodplains as river channels migrate and avulse, leaving water-filled depressions along abandoned channels and *oxbow lakes* in meander cutoffs. The Taieri River on the Upper Taieri Plain (Otago) provides one of the best examples of oxbow lakes anywhere in the world (Fig. 19.5).

Lateral lakes are common in Waikato, where deposition of alluvium, particularly as levees adjacent to a river channel, blocks drainage and creates lake basins along the margins of the flood plain (e.g., Lake Whangape, near Huntly). The hydrological regimes of oxbow lakes generally reflect their location on a river flood plain, in that lake level is controlled by the groundwater level in the flood plain sediments, by direct precipitation onto the lake surface, and by periodic inflows from the river during times of flood. Lateral lakes are more likely to have additional surface water inflows from tributary valleys and to be hydrologically disconnected from the main river.

Barrier-bar lakes result from the formation by long-shore drift of a barrier-bar or spit across a coastal embayment (or, in some cases such as Lake Rotongaio, across an embayment in the shoreline of a larger lake). They are, therefore, near the sea, their surface water level is within a few metres of sea level, and their water is commonly brackish. Most are thought to have existed in their present form for less than about 6,500 years, since sea level reached its current position (Lowe and Green 1987). Most are less than 1 km² in area, and less than 5 m deep; the largest (but still shallow) barrier-bar lake is Lake Ellesmere, near Christchurch, with a present-day area of 182 km². Lake Ellesmere's pre-settlement extent was considerably greater, but its level has been managed by periodic artificial opening of its outlet to the sea to increase the frequency of natural breaching, to allow the cultivation of former lake bed. The hydrological regime of a barrier-bar lake is fundamentally controlled by its proximity to sea level, inflows from rivers and springs, and the cyclic process of barrier breaching and reconstruction (Fig. 19.6).



Figure 19.5 Oxbow lakes on the scroll plain of the Taieri River. Photo: Neville Peat

Dune lakes formed by aeolian (wind-blown) deposits and sand dunes are common on the west coast of the North Island, and are found at various other locations around the coastline of the North Island, South Island, and Chatham Islands. Lowe and Green (1992) identified two basic types:

- *Dune barrage lakes*, created by blockage of a valley draining towards the coast (e.g., Lakes Ototoa and Kuwakatai on the South Kaipara Peninsula), or formed

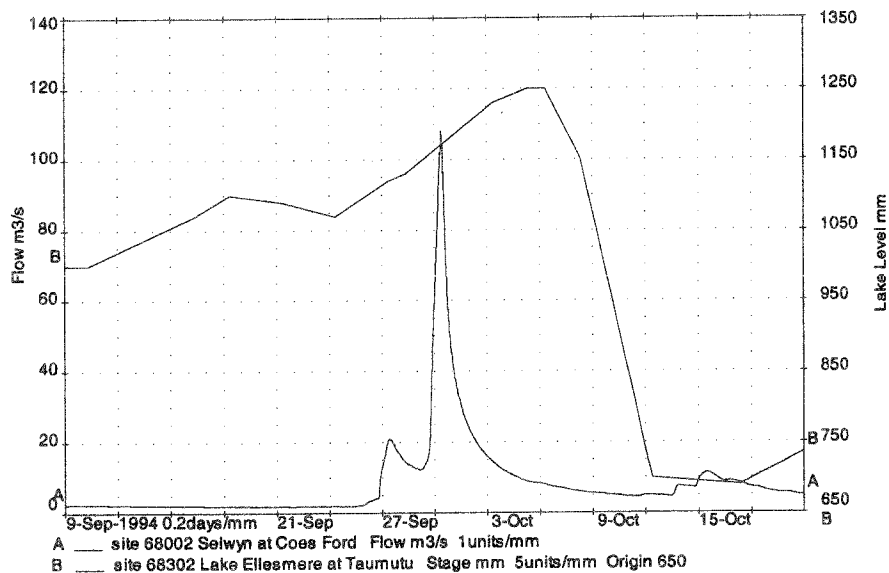


Figure 19.6 Lake Ellesmere water level and Selwyn River inflows, September–October 1994. The lake level is rising progressively due to tributary and groundwater inflows; the 110 m³/s Selwyn River fresh on 30 September has no discernible effect on lake level. It appears that an unsuccessful attempt may have been made on 18 September to breach the lake outlet to lower the water level. However, the outlet was breached on 6 October, causing lake level to decline from 1.25 m above mean sea level to 0.7 m amsl. (Plot provided by Esther Smith, Environment Canterbury).

in depressions between two or more dunes (e.g., a number of lakes along the Manawatu coast in the vicinity of Lake Horowhenua). The hydrological regime of such lakes is related principally to groundwater levels and sea level, and to inflows from streams draining to the coast.

- *Deflation lakes*, which occupy hollows excavated by wind erosion. The sub-type designated “perched dune lake” contains water derived largely from rainfall, and is held by an underlying aquiclude created by processes such as iron-pan formation; the sub-type “water-table-window lake” forms when the groundwater level rises above the bottom of a deflation hollow.

Lowe and Green (1992) suggested that most dune lakes are less than 6,500 years old (since the present-day sea level was established), although many are younger, or have been drained and refilled as a result of migrating coastlines and fluctuating sea levels. Most dune lakes are less than 0.5 km² in area, and shallow—although the deepest are over 30 m deep (e.g., Lake Taharoa, North Kaipara Peninsula, is 37 m deep and also substantially older than most, over 50,000 years old).

Landslide lakes are created when valleys are blocked by rockfalls or rock avalanches; they are restricted to areas of steep terrain, and are particularly common in seismically active parts of the country with high rainfalls (Adams 1981; Fig. 19.2). Many landslide lakes are only temporary. A 6-km-long lake created in the Buller River by a rockfall during the Inangahua Earthquake of 1968 was drained by failure of the dam only a day after the earthquake. The failure released 1.5 million m³ of water, causing a flood wave that moved down valley at about 5.4 m/s (Sutherland 1969). Many “permanent” landslide lakes survive for much longer periods than this; Lake Tutira (Hawke’s Bay) is estimated to be about 6,500 years old, for example. However, because of the high erosion rates characteristic of the terrain in which landslide lakes are found, they are prone to both rapid infilling and rapid downcutting of their outlets. Lake Matiri, near Murchison, is about 300 years old, and has an estimated remaining lifetime of 400 years before delta sedimentation and outlet downcutting eliminate it (Adams 1981).

The largest landslide lake is Lake Waikaremoana, with an area of 54 km², a maximum depth of 248 m, and an estimated age of 2,200 years. Before it was controlled for hydroelectricity generation, the lake’s normal outflow of 14 m³/s was by leakage underground, with overflow at the surface outlet only in very wet seasons (Adams 1981). Water emerged at twelve or more springs, but during construction for hydropower development the underwater leakage points in the lake were located and blocked by dumping rock fill onto them from barges. Waikaremoana typifies the hydrology of landslide lakes, in that their

behaviour is controlled by catchment hydrology and the permeability of the landslide deposits. Many, such as Lake Challice, drain entirely underground; others, such as Lake Christabel, predominantly drain underground, while others discharge predominantly via a surface outlet.

Solution lakes are rare in New Zealand, because karst terrain on carbonate rocks is restricted in area (Chapter 32). The largest is Lake Disappear in the King Country, which appears in a polje (a large, flat-floored solution depression) after heavy rainfall, when inflows are greater than drainage from the polje can discharge. There are other small, usually temporary ponds in the King Country, as well as in limestone and marble terrain in Hawke’s Bay, Wairarapa, Nelson and Canterbury.

True *peat lakes* (phytogenic lakes) are created as a result of the accumulation of peat; in New Zealand there are few phytogenic waterbodies large enough to be called lakes, although many ponds and pools can be seen in localities such as the Kopouatai peat dome (Hauraki), the Ahukawakawa Swamp (Taranaki), the Lagoon Saddle mire (Canterbury), or peat bogs in the Chatham Islands. Many lakes created by other processes subsequently have been modified by peat growth (e.g., the riverine Lake Maratoto: Selby and Lowe 1992).

Artificial lakes

Artificial lakes—reservoirs—generally are created by a dam or barrage across a river valley, and tend (like landslide-dammed lakes) to be long, narrow and dendritic in plan. Their purpose is to store water that otherwise would flow unused to the sea, allowing people to use water at times of their choosing, rather than when it would be discharged naturally. Reservoirs in New Zealand are predominantly for hydroelectric power generation (e.g., the sequences of lakes along the Waikato and Waitaki Rivers) or urban water supply (e.g., Auckland’s water supply reservoirs in the Hunua and Waitakere Ranges). In other countries, many large reservoirs have been constructed principally for agricultural irrigation; in New Zealand, there are few such reservoirs: Lake Opuha in Canterbury is a noteworthy example. However, there are vast numbers of small “farm dams”, which store water for stock drinking water and other purposes, and there is a move towards installing “off-stream storages” such as those in the Glenmark Irrigation Scheme of North Canterbury, which are used to supply spray irrigation (Fig. 19.7).

Structures now control the levels of a number of natural lakes, to manage storage and outflow for power generation and other purposes. Thus, for example, the Waitaki system now includes the managed natural lakes Tekapo, Pukaki and Ohau, and the artificial lakes Ruataniwha, Benmore, Aviemore, and Waitaki, as well as a system of canals that allows the water to be used to generate electricity in a sequence of eight power stations.



Figure 19.7 Off-stream storage reservoir in the Glenmark Irrigation Scheme near Waipara, Canterbury. The water is held in the nearer reservoir by a constructed embankment about 4 m high on three sides; the further reservoir is held by a barrage across a topographic depression, but both are fed by inter-basin transfer from Weka Stream. Other reservoirs in this scheme are contained entirely by embankments. Photo: Paul Mosley

CHARACTERISTICS OF LAKES

The preceding section classified lakes in terms of their geological origin, but there are many other characteristics of lakes that may be of more significance to water resources managers. The Inventory of New Zealand Lakes (Livingston *et al.* 1986) compiled much of the information available at the time it was prepared, and Lowe and Green (1987) summarised key morphometric data for major lakes.

Characterisation of lakes

Chapter 8 reviews the characteristics—including “natural character”—specified in resource management law that may be relevant for managing a river, and these also are applicable to lakes. Vant (1987) provided a checklist of information needs related to lake management, and Mosley (1999) summarised the characteristics of lakes that may need to be considered by people “exercising powers and functions” under the Resource Management Act (Table 19.2). The characteristics listed in Table 19.2 could be used as a starting point for appraising, for example, the likely environmental effects of a proposal to modify or use a lake in some way. They would need to be considered in much more detail, and in terms of measurable dimensions and parameters. The factors related to the character of the lake water itself are dealt with more fully in Chapter 21. The morphometry of the lake is dealt with in the following section.

Lake bathymetry

Bathymetry is the science of measuring the depths of water bodies; bathymetric charts are available for almost all of New Zealand’s largest lakes and many of its smaller ones (Spigel and Viner 1992). A bathymetric chart presents

Table 19.2 Characteristics of a lake relevant to resource management requirements.

NATURAL CHARACTER AND SOCIO-ECONOMIC MODIFICATION

- A. LANDSCAPE SETTING – LANDSCAPE BEYOND THE IMMEDIATE MARGIN OF THE LAKE** Valley type (if any) and dimensions; other significant landforms Lithology Vegetation, land use, erosion types
- B. ALLUVIAL PLAIN SETTING** Presence, extent and dimensions of alluvial plain (if any) Alluvial plain surficial material and soil type Alluvial plain vegetation and land use
- C. CHARACTER OF THE LAKE MARGIN** Lake edge profile and long section. Lakeshore materials. Near-shore water/littoral zone. Vegetation and land use along the lake margin. Degree of human modification, including bed disturbance and resulting water turbidity; structures; introduction or removal of vegetation; deposition of substances; reclamation of the bed.
- D. PHYSICAL CHARACTER OF THE LAKE** *Dimensions:* altitude, surface area, volume, length of major axes, maximum and mean depth, length of shoreline, residence time, catchment area. *Geographical:* origin, out-flowing rivers, bathymetry, relief, topography, regularity of shoreline. *Enclosure:* containment of the lake within its basin. *Climatic:* air temperature seasonality, sunshine hours, precipitation
- E. CHARACTER OF THE WATER** *Hydrological:* water level variations. *Physical:* water column temperature seasonality, transparency. *Chemical:* pH, suspended sediment, DO, COD, chlorophyll *a*, total phosphorus, total nitrogen, historical trends. *Biological:* main plant and animal species, primary production rate, biomass or population density, fish catch, historical trends/changes.
- F. SOCIO-ECONOMIC CONDITIONS IN CATCHMENT** *Catchment land use:* vegetation types, land use types, soil erosion types. *Human population:* numbers, distribution, settlements, waste disposal facilities. *Industries:* type, products, output, waste disposal facilities.

USES OF THE LAKE WATER RESOURCE USES AMENITY VALUE

- G. VISUAL QUALITY** (the following can be ranked on a scale 1 to 5, for the lake region, lake basin, and lake or lake sector)
 - Unity:* the degree to which the visual elements coherently fit together as a whole
 - Identity:* the extent to which a particular landscape has its own identifiable character.
 - Diversity:* the degree to which variation in form, texture and colour provides contrast.
 - Mystery:* the extent to which patterns in the landscape are partially hidden and “mysterious”.
 - Uniqueness:* the degree to which the landscape is unlike any other in the region.
 - Sensitivity:* presence of areas that are susceptible to undesirable modification.
 - Visibility:* the extent to which a landscape can be seen.
 - Naturalness:* the extent to which the landscape has been modified by people.
- H. RECREATIONAL POTENTIAL**
 - Litter and other human waste
 - Presence of beaches, islands and campsites suitable for recreation
 - Presence of facilities such as picnic grounds, toilets, play areas, interpretive/visitor centres, etc.
 - Accessibility of banks for vehicles and walkers
 - Ease of access to water for trailer-mounted craft, and individuals
 - Points of interest, including archeological, historical and cultural features, nature walks, etc.
 - Floatability and obstructions to navigation for different craft.

the results of a survey of the subsurface contours of the lake bed, usually involving hundreds of depth measurements or soundings carried out along pre-set survey lines or transects. Details of the lakes for which charts are available may be obtained from the National Institute of Water and Atmospheric Research.

Spigel and Viner (1992) used the bathymetric chart of Lake Rotongaio, a barrier-bar lake separated from Lake Taupo by a sand-bar, to demonstrate basic aspects of lake bathymetry. The areas measured between each pair of contours (Fig. 19.8, top) are used to prepare a hypsographic curve, or cumulative area-depth plot, and a cumulative volume-depth plot is prepared by multiplying the successive areas by the relevant depths (Fig. 19.8, bottom). The maximum depth of the lake is 21 m (Fig. 19.8, top), and the total lake surface area is 34.84 ha (cumulative area at zero depth on the hypsographic chart, Fig. 19.8, bottom left). Mean depth is calculated by dividing total lake volume by lake surface area, in this case $3,631,200/348,400=10.42$ m. Mean depth may be visualised as the depth of a flat-bottomed, vertical-sided basin with the same surface area and volume as the natural lake. Maximum lake length, the distance (not necessarily in a straight line) between the two most remote points on the shoreline (in this case, the segmented line along the line of greatest depth: Fig. 19.8 top), is 1,200 m. The maximum effective length is the length of a straight line connecting the two most distant points on the shoreline over which wind and waves may act without interruptions from land and islands. Effective fetch at a particular point is a measure of the uninterrupted distance available to the wind to generate waves over a range of directions from that point. This characteristic is used in formulae to predict the height or energy of wind-generated waves. The maximum fetch at a point is the largest value and direction of effective fetch, and is of particular significance when considering the likely effects of waves on beach erosion, aquatic vegetation, shoreline structures, boat moorings, etc.

Lake shorelines

The shoreline is the interface between the body of the lake and the surrounding land surface, a zone where physical and biological processes are dynamic, the lake ecosystem is especially diverse and productive, and human activity can have particularly rapid and marked adverse effects. Chapters 23-25 consider the biological and ecological aspects of the shoreline and littoral zone.

New Zealand's lake shores vary in composition from erosion-resistant bedrock through to deposits of silt and mud, and lake beaches have a wide range of sediment types, from boulders and cobbles to mud. The physical processes of erosion, sediment transport and deposition along a lake

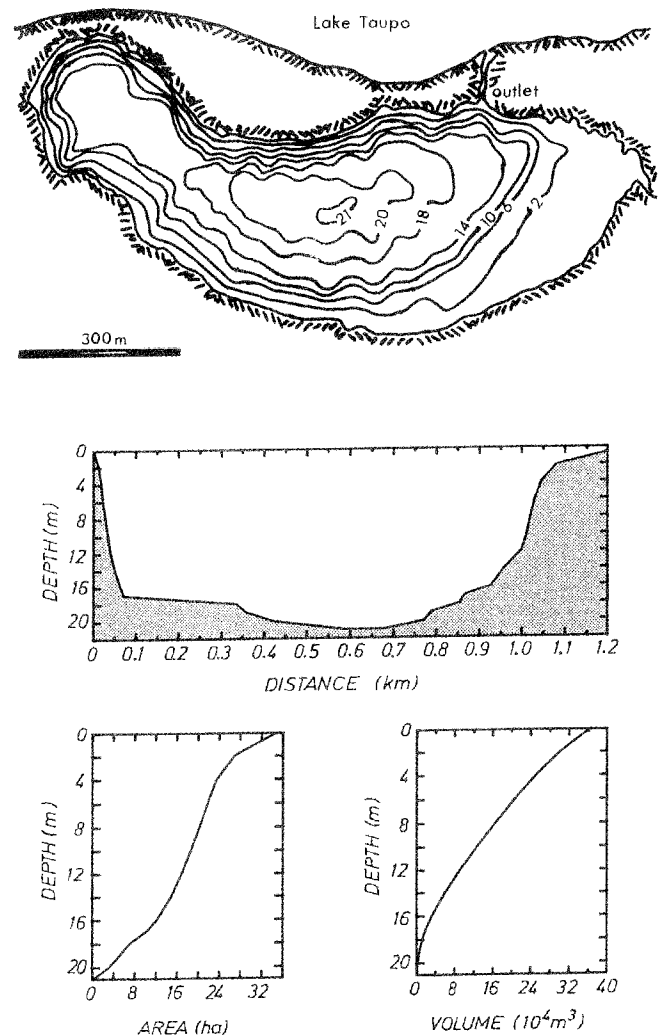


Figure 19.8 Bathymetry for Lake Rotongaio. Top: depth contours in metres (simplified from Irwin 1978). Middle: profile along the transect along the line of greatest depths in the lake. Bottom: hypsographic curve and cumulative volume curve of the lake. (from Spigel and Viner 1992).

shore or beach are similar to those along a sea coast, and their effectiveness depends largely on wave energy, principally during high-energy storms that occur a few times a year. Wave energy is a function of wind velocity and duration, fetch, and water depth; on many lakes it is less than along the sea coast, because the effective fetch is short.

The physical characteristics of a particular section of lake shore are in a dynamic equilibrium with the local wave climate and with the water level regime, defined in terms of the average level, range of levels, indices of variability, etc. A condition of dynamic equilibrium implies that a lake shore that is composed of erodible and transportable material can at some times be eroding and at other times be accreting, but that the long-term location and condition

of the shore remain approximately the same. Wave shape determines whether erosion or accretion occurs at a particular time. Short, steep waves tend to mobilise sediment and transport it away from the shore towards deeper water; they may cause rapid shoreline erosion during a storm. Waves that transport sediment towards and along the shore tend to be associated with low-energy wind conditions, and therefore cause accretion much more slowly, over periods of months and years.

Beaches on New Zealand lakeshores typically consist of three elements, the foreshore, the nearshore shelf, and the offshore face (Pickrill 1976; James *et al.* 2002; Fig. 19.9). The foreshore, extending from mean water level to the upper limit of wave action, generally has a slope of about 8° . It grades down to the nearshore shelf, which extends, at a slope of about 6° , from mean water level out to a distinct break in slope. Below this, the lake bed drops steeply, at approximately the angle of repose of the sediments of which it is composed. Studies of Lakes Manapouri and Te Anau (Pickrill 1976) indicated that the characteristics of the nearshore shelf are related to effective fetch length and wave energy: the greater the wave energy, the wider the shelf and the greater the water depth at the outer edge. This appears to be the case whether the shelf is a result of progradation (building outwards) or erosional retreat of the shoreline (Allan 1998).

As lake level rises and falls, the location changes at which the energy of breaking waves is dissipated, and therefore at which erosion, transport and deposition are focussed (James *et al.* 2002). Extremely high or low levels may cause events such as the undercutting and collapse of marginal hillsides that normally are beyond the reach of wave action, or slumping of the outer shelf if the lake is lowered rapidly, causing the formerly submerged, saturated sediments to become unstable. Entire beaches have been lost as a result of such destabilisation.

If a lake's water level regime is altered by changing the average, range, or variability of levels, for example to meet scheduling requirements for hydropower generation, a period of shoreline adjustment will follow. A new regime will alter the location on the beach profile (Fig. 19.9) at which wave energy acts, and will produce a new profile form and/or location (James *et al.* 2002). Hence, a lowered lake level would remove the upper part of the profile from wave action, cause active reworking of the shelf and offshore face as wave energy is concentrated on a narrower shelf, cause sediment to be displaced down the offshore face into deeper water, and result in slow progradation of the shoreline. A raised lake level would move the zone of wave action further up the profile, perhaps causing erosion of the foreshore and backshore behind it, and aggradation of the offshore shelf.

Changes in lake level regime may have a significant

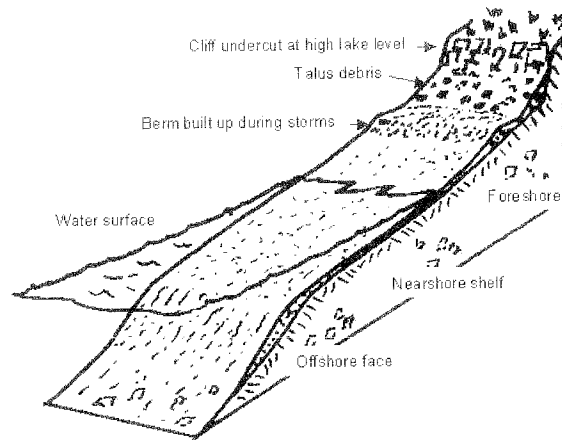


Figure 19.9 Typical lakeshore profile developed on glacial moraine at Lakes Manapouri and Te Anau (after Pickrill 1976).

impact on water levels in marginal wetlands and on the morphology of deltas constructed by streams and rivers that flow into the lake. An extreme case is Lake Pukaki, in which the delta of the Tasman River has been drowned by the raised lake level (Fig. 19.1). Lowering of levels can cause significant downcutting by inflowing rivers, as nickpoints retreat headwards along the channel; for example, the lowering of Lake Benmore by 4 m for a 3-month period induced up to 0.8 m degradation of the Ohau River for a distance of 1.3 km upstream (Mosley 1984). Generally, lake level lowering is temporary, so such effects are likely to be short-lived; however, even a short period of degradation may undermine riverbank structures and bridge piles.

THE WATER BALANCE OF LAKES

Water balance calculations are essential for many aspects of lake management, and for design of water resources projects that use lakes and reservoirs for storage. McMahon (1992) provides a useful introduction to the computations needed to analyse the capacity of a lake or reservoir, and to route flows through it. In New Zealand, a particularly important use of water balance calculations is to estimate lake inflows, because direct measurement in the unstable rivers of many headwater catchments is not feasible.

None of the elements of the lake water balance are measured directly, but are computed from measurements of water level, precipitation, and weather conditions (Chapters 2 and 7). Hoare and Spigel (1987) considered some of the problems relating to the use of such measurements in calculating flows and volumes.

The relationship between water level and lake volume

The volume of water contained in a lake basin, the lake storage, increases during rainfall, because the outflow channel of most lakes cannot discharge the inflow from the lake's tributaries. As a result, the water level rises. Commonly, water level (stage) recorders are located near lake outlets, and are used not only to provide a record of lake level and volume, but also to provide (with the help of a stage-discharge rating curve) a record of lake outflow.

The volume of water in a lake is not the only factor that determines water level, however. Other factors may be important for real-time management of the water resource and when designing lake-shore structures such as flood walls. Other factors that, without a change in lake volume, may increase water level above that measured in a stilling well at a stage recorder include:

- *Wind waves and waves generated by boats.* On a large lake at the point of maximum fetch, wind waves that are running up onto a lake shore may have an amplitude of 0.5 m and more. Wave height H (m), the vertical distance between the trough and the crest, can be estimated from the equation:

$$H = 0.02 W^{1.06} F^{0.47} \quad 19.1$$

in which W is wind speed (m/s) and F is fetch (km). Boat-generated waves may be of a similar magnitude, but their characteristics depend on factors such as the type and speed of boat and its distance from the shore.

- *Wave set-up.* The drag exerted on the water surface of a lake by the wind also causes "tilting" of the water surface, or set-up, as water level is elevated on the down-wind shore and depressed on the up-wind shore. Set-up may be a few centimetres or tens of centimetres in a deep lake (e.g., it is commonly 1–3 cm in Lake Taupo, although a 30 cm set-up has been measured there: Hoare and Spigel 1987), and well over a metre in windy conditions in extensive, shallow lakes such as Ellesmere (Spigel and Viner 1992). It may be calculated using the equation (Hoare and Spigel 1987):

$$\delta D = W^2 (L/D) (1 + 0.07W) 10^{-4} \quad 19.2$$

in which δD is the difference in water level (m), L is lake length (km), and D is average basin depth (m).

- *Changes in atmospheric pressure.* As weather systems—depressions and high pressure areas—pass, the water surface in a lake respond by rising during the passage of a depression and falling during the passage of a high-pressure system. A 1 millibar change in atmospheric pressure is equivalent to a water level change of 10 mm.
- *Seiching.* Seiching is a rhythmic oscillation of water levels in a lake basin, following set-up by wind or rapid

change in atmospheric pressure. It is observable at a particular point on the shoreline as a gradual rise and fall of water level, commonly on the order of 5 cm, and on a time scale of a few tens of minutes. Seiches have been recorded in Lake Taupo with a period of 36 minutes, in Lake Wanaka (39 minutes), and Wakatipu (52 minutes; Thompson and Ibbitt 1978; Heath 1975).

The range over which water level varies differs widely from lake to lake. The most extreme ranges are experienced in lakes that are controlled for water supply or hydropower generation. Lake Moawhango (a reservoir near Waiouru) and Lake Hawea have experienced maximum fluctuations of 15 m and 22 m respectively; before control, water levels in Hawea varied by less than 3 m. Uncontrolled lakes with a surface outlet generally have a much more restricted range of levels, depending on the geometry of the lake outlet and the "stage-discharge relationship" between water level and outflow, and on the area of the lake surface relative to the catchment area contributing inflows. For example, Lake Wakatipu (catchment area of 3,134 km² and lake area of 289 km²) has an extreme range of 3.48 m. The highest level was recorded in a flood in 1999 which inundated central Queenstown for many days, and which had an estimated average recurrence interval of once in 150 years. Although outflows from Lake Wakatipu are not managed, a now-disused outlet structure that was installed in the 1920s raised minimum lake levels by about 0.4 m, and probably modified the stage-discharge relationship significantly.

The water levels in closed lake basins (i.e., lakes without a surface outlet) tend to vary more widely than those in open lakes. Their rates of inflow are proportional to catchment precipitation, and generally are more variable than rates of sub-surface outflow, which respond relatively little to changes in lake level. For example, during 1970–75, the level in Lake Tarawera, which has a surface outlet, varied over 0.48 m, whereas that in the nearby Lake Rotoma, which has only subsurface outflow, varied by 3.01 m in response to long-term variations in precipitation (Spigel and Viner 1992; Fig. 19.10).

The water balance equation

The water balance of a lake during a specified time interval may be summarised by the equation:

$$\Delta S/\Delta t = Q_{\text{surface, in}} - Q_{\text{surface, out}} + Q_{\text{groundwater, in}} - Q_{\text{groundwater, out}} + A(P - E) + \epsilon \quad 19.3$$

in which $\Delta S/\Delta t$ is the rate of change in lake storage (volume/time), $Q_{\text{surface, in}}$ is surface water inflows (streams and rivers, overland flow), $Q_{\text{surface, out}}$ is surface water outflow, $Q_{\text{groundwater, in}}$ is groundwater inflow, $Q_{\text{groundwater, out}}$ is groundwater outflow, A is lake area, P is precipitation onto

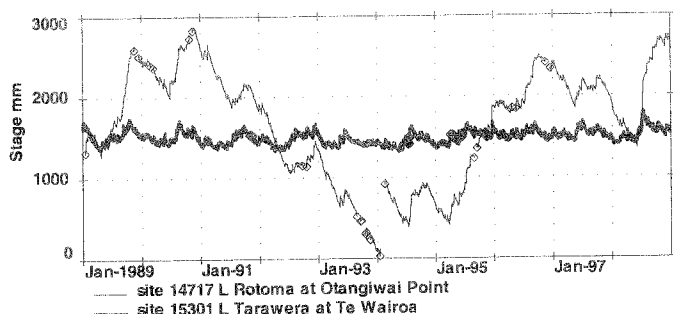


Figure 19.10 Water surface elevations (relative to an arbitrary datum) in Lake Tarawera and Lake Rotoma, 1989–98. (Plot provided by Kathy Walter, NIWA, using data provided by Environment Bay of Plenty). There is an annual cycle in water level in both lakes, but Lake Rotoma also shows a larger, longer-term variation in water level that reflects the slow response of subsurface outflow to variations in precipitation inputs.

the lake surface, E is evaporation from the lake surface, and ϵ is an error term. For a reservoir, there may be additional terms for inter-basin transfers of water from other catchments ($Q_{\text{transfer, in}}$), and withdrawals for purposes such as urban water supply ($Q_{\text{withdrawal}}$). If all of the flow rates (Q terms) are measured in m^3/s , then P and E should be specified in units of m/s and A as m^2 , giving $\Delta S/\Delta t$ as m^3/s .

Of the terms in Equation 19.3, only $Q_{\text{surface, out}}$ is easily measured; P somewhat less so. Evaporation is best estimated using techniques described in Chapter 6, and $Q_{\text{surface, in}}$ is normally calculated from lake outflow, change in lake level (proportional to change in storage), and precipitation data. The two terms related to groundwater are almost impossible to measure directly, but can be calculated using numerical groundwater models. Very good data on geological conditions and aquifer characteristics are required for models to provide reliable results. For example, groundwater inflows to a lake in an alluvial basin might be at sub-surface springs that are associated with outcrops of buried paleo-channels (the courses of ancient rivers), and outflows similarly may be at particular locations where there are highly permeable sediments. Use of data that describe average conditions is unlikely to produce accurate estimates. The difficulties of estimating the groundwater component of a lake water balance are well shown by Brown's (2001) discussion of groundwater movement under the lower Canterbury Plains in the vicinity of Lake Ellesmere.

The relative importance of the water balance terms varies between lakes, and over the year. Thus, for example, Scotter and Kelliher (Chapter 6) have shown that evaporation rate from Lake Tekapo varies five-fold over the year, with a maximum rate in winter, in response to

weather conditions (e.g., windiness, vapour pressure differences). The groundwater components of the lake water balance are particularly prone to vary among lakes. The water balance of Lake Ellesmere, located on alluvial sediments at the foot of the Canterbury Plain and separated from the ocean by a permeable alluvial barrier, can be expected to be much more influenced by groundwater inflows and outflows than the balance of a lake such as Wakatipu, whose basin is scoured deep into bedrock by glacial action. Some lakes, such as the landslide-dammed Lake Chalice or several closed-basin lakes of volcanic origin in the Rotorua Lakes district, have no surface outflow, and drain entirely underground.

Examples of lake water balance

There are relatively few published data on the water balances of New Zealand lakes, although of course calculations have been made for many lakes for water resource management projects. The long-term water balance of Lake Rotoiti, near Rotorua, calculated by Spigel (1989) using data for 1957–1984, shows the relative magnitudes of the components in Equation 19.1, for a lake in which groundwater inflows and outflows are relatively large because of the volcanic soils and sediments in its catchment. Over the long-term, ΔS was not significant; the relative magnitudes of the other terms were:

$Q_{\text{surface, in}}$ (Ohau Channel, as the dominant inflow from Lake Rotorua): $22.8 \text{ m}^3/\text{s}$

$Q_{\text{surface, out}}$: $18.0 \text{ m}^3/\text{s}$ (Kaituna River)

P : $2.19 \text{ m}^3/\text{s}$ ($2,010 \text{ mm}/\text{y}$, over a lake area of 34.35 km^2)

E : $0.96 \text{ m}^3/\text{s}$ ($880 \text{ mm}/\text{y}$, over a lake area of 34.35 km^2)

Other terms ($Q_{\text{groundwater, in}}$, $Q_{\text{groundwater, out}}$, inflow from small streams and overland flow, ϵ): $3.55 \text{ m}^3/\text{s}$

Within the "other terms", Spigel was interested in one not included in Equation 19.3, Q_{sto} , which he took to be the surface water flow directly from the principal lake inflow (Ohau Channel) to the nearby lake outlet (Kaituna River), principally during storms. Spigel estimated that ϵ was $\pm 2.5 \text{ m}^3/\text{s}$, and concluded (based on fragmentary well records) that there is almost always net groundwater inflow into the lake. The magnitude of the "other terms" gives no information on the magnitudes of $Q_{\text{groundwater, in}}$ and $Q_{\text{groundwater, out}}$, but only the net difference between them.

The relationship between lake inflows, lake outflows, and storage is of particular interest. The outflow from a lake varies in direct proportion to the height of the water surface above the outlet; in a large lake, a considerable

volume of inflow during a storm may raise the lake level only a few millimetres, so that the rate of outflow increases relatively little in comparison with inflow. Accordingly, a lake outflow hydrograph usually is very “damped” in comparison with the inflow hydrograph. This is well illustrated in hydrographs for two floods into Lakes Wanaka and Hawea, in July and August 1990 (Fig. 19.11). The first flood into Lake Wanaka peaked at nearly $1,800 \text{ m}^3/\text{s}$, whereas the outflow rose only slightly, to almost $200 \text{ m}^3/\text{s}$ (Fig. 19.11, top). There was a small increase in water level—about 0.4 m (Fig. 19.11, bottom)—but because the area of Lake Wanaka is 193 km^2 , this represents a substantial volume of water, approximately 80 million m^3 . The second flood peak was smaller, $1,240 \text{ m}^3/\text{s}$, but because lake level had been raised by the preceding flood, the outflow rose further, to $240 \text{ m}^3/\text{s}$. It may be seen, on Figure 19.11 (top), that the outflow peaks at the point that the inflow and outflow hydrographs intersect.

The behaviour of the nearby Lake Hawea was completely different to Lake Wanaka, because it is equipped with a control structure to manage the lake water for hydro-electricity generation downstream. The shape of the inflow hydrograph was similar, but inflows were less, because the catchment area is smaller and the catchment is further away from the zone of maximum rainfall along the Main Divide. The outflow hydrograph was not at all similar, however, because the control structure was used to release an almost continuous flow of $200 \text{ m}^3/\text{s}$ for six weeks following the first flood (Fig. 19.11, middle). For much of this time, the outflow from Hawea was greater than the inflow, and also greater than from Wanaka. As a result the level of Hawea fell rapidly, by 3.3 m over the 6-week period, whereas the level of Wanaka dropped much more slowly, by only 1 m (Fig. 19.11, bottom). The two “steps” on the Hawea water level graph look similar, but the one on 6 August was caused by the second flood, and that on 25 August by a brief closure of the outlet gates.

SEDIMENT IN LAKES

Any lake will be filled eventually by sediment—if it is not drained beforehand by downcutting of its outlet or eliminated by tectonic movements or other geological processes. For example, Lake Tutira (Hawke’s Bay), which was created by landsliding about 6,500 years ago, is estimated to have a remaining lifetime of only 600 years (Page and Trustrum 1997) because of today’s very high rates of erosion in its catchment.

The lake sediment budget

Sediment is transported into a lake, deposited, and transported from a lake by a variety of processes (Chapter 12;

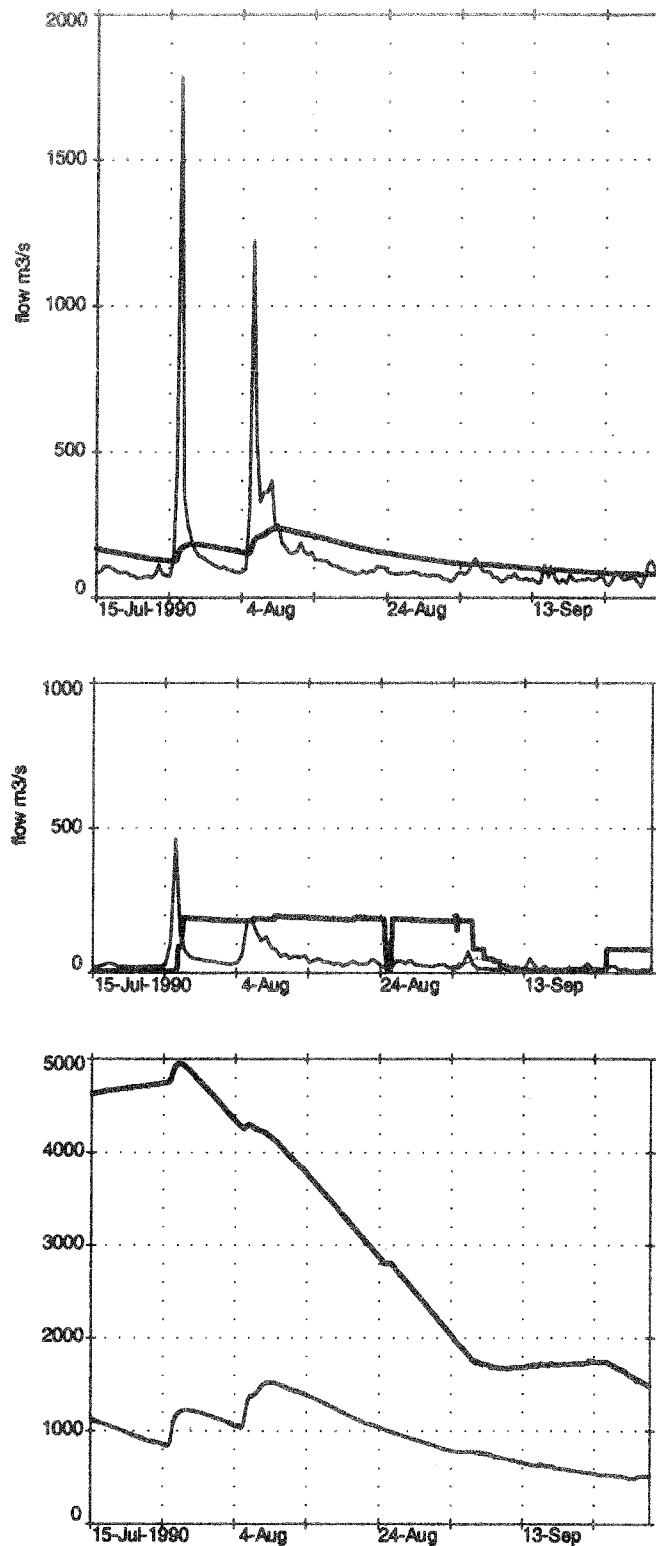


Figure 19.11 Inflow and outflow hydrographs (narrow and thick lines respectively) for Lake Wanaka (top) and Lake Hawea (middle), July–September 1990. The bottom plot shows water levels for Lake Wanaka (narrow line) and Lake Hawea (thick line) for the same period, relative to arbitrary datums. (Plots provided by Roddy Henderson, NIWA. The permission of M-Co and Contact Energy to use hydrological data is acknowledged).

Fig. 19.12). Sediment inputs are predominantly by rivers, streams and, in the case of glacial lakes, glaciers that carry debris into the lake basin. Erosion of the shoreline by wave action and mass movement of adjacent hillsides may introduce large volumes of sediment, and wind-blown “airfall” sediment may also be a significant input into lakes such as Lake Tekapo, which is downwind of a large sediment source, a pro-glacial braided river. Wind-blown volcanic ash forms distinctive layers on the beds of many North Island lakes, and has provided an invaluable means of estimating ages and rates of infilling of the lakes.

In addition to the inorganic sediments that are the products of erosion around the lake shore, in the lake catchment, and further afield, organic deposits are a significant component of the sediment budget of a lake. When the plankton—floating microscopic plants and animals—that inhabit the lake waters die, they sink to the bottom. As they do so, they undergo a process of mineralisation, in which bacteria and fungi break down the organic compounds of which they are composed into inorganic compounds and trace elements. However, this process often is not complete before the “plankton rain” reaches the lake bed, and a layer of organic ooze builds up, which provides a reservoir of the carbon, nutrients and trace elements required for biological production (Spigel and Viner 1992; Chapters 22 and 23).

An important element of the lake sediment budget is the trap efficiency of a lake (Brune 1953). The trap efficiency of a lake is the proportion of incoming sediment that is held by the lake, rather than being carried out through the outlet channel, largely as fine suspended load and wash load. There are few direct measurements of trap efficiency in New Zealand, but Page and Trustrum (1997) estimated a trap efficiency for Lake Tutira of 97%, Jowett (1984) listed a trap efficiency of 80% for Lake Roxburgh, and we can assume a trap efficiency approaching 100% for a lake like Waikaremoana that rarely has surface

outflow. Brune (1953) concluded that trap efficiency is most strongly determined by the capacity of a lake relative to the total inflow, and the size of sediment supplied. A lake whose capacity is one tenth of the total annual inflow is likely to retain 90% of the sediment that enters it—up to 95% when the catchment supplies predominantly coarse sediments, and perhaps 80% when the catchment supplies fine material (Fig. 19.13).

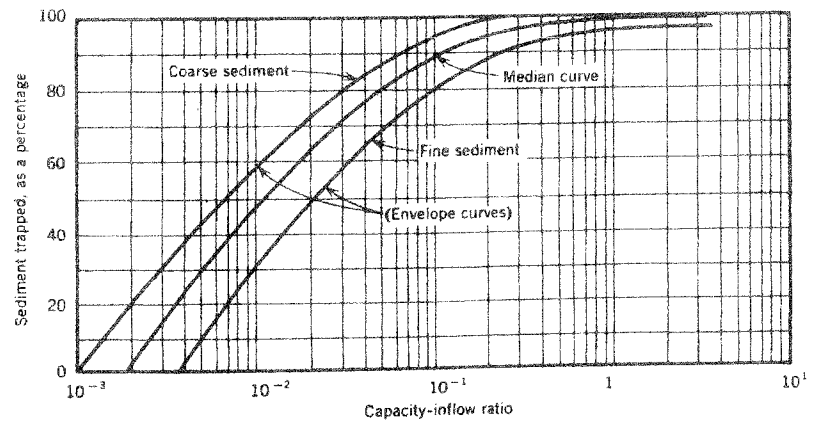


Figure 19.13 The trap efficiency curve developed by Brune (1953).

Processes and rates of sedimentation

When a river or stream enters a lake, its ability to transport sediment decreases abruptly. Coarse bed load and suspended load tend to be deposited close to the river mouth, as “foreset beds” and “topset beds” on an advancing delta (Fig. 19.12). Finer suspended sediment and wash load remain in suspension for longer before being deposited; indeed, they may be carried all the way through the lake. The fate of the suspended load depends largely on the difference in density between the river water and lake water, which is controlled by differences in water temperature and suspended sediment concentration. If, because it is warmer, the river water is less dense than the lake water, it will spread across the surface of the lake as an “overflow”, and suspended sediment eventually will drop out of suspension and be spread over a wide area of the lake bed. If the densities are exactly the same, the river water mixes with the lake water over the entire depth of the lake. If the river water is colder and/or has a higher suspended sediment concentration than the lake water, it will be denser and the inflow will flow down the bed of the lake as an “underflow”. An underflow may carry coarse sediment well beyond the river mouth. Sub-surface channels have been observed in some lakes where underflows are common; underflows from the Dart and Rees Rivers can

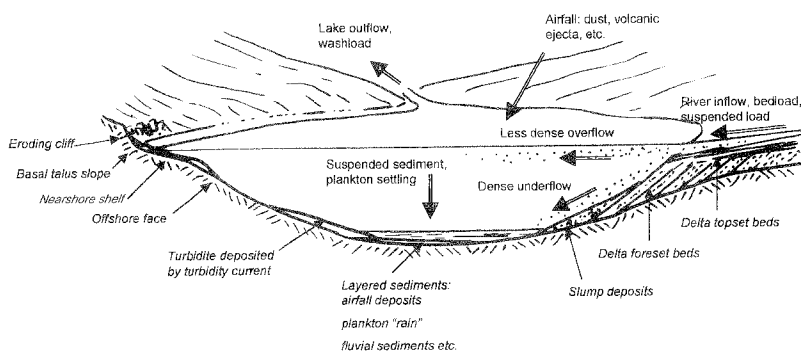


Figure 19.12 Sedimentation processes in lakes.

carry sediment over 60 km along the bed of Lake Wakatipu (Pickrill and Irwin 1982).

Sediment that is deposited on the lake bed may be remobilised by wave action, earthquakes, slumping caused by oversteepening of a depositional slope, erosion by turbidity currents (rapidly moving, very turbid underflows that may be initiated by a slump), underflows, and other processes. As a result, sediment tends to move downslope towards the deepest part of the lake, a phenomenon called "sediment focussing". Over a period, sedimentation on a lake bed progressively may build up considerable depths of layered deposits. Such deposits, built up over long periods, reflect particular events such as major floods, volcanic eruptions and landslides, as well as seasonal/annual cycles of deposition. Their characteristics—such as chemistry, particle sizes and pollen content—can tell us a great deal about the history of a lake and its catchment. For example, lake bed cores taken from Lake Tutira enabled Page and Trustrum (1997) to correlate erosion rates with changes in catchment vegetation and with Polynesian and European settlement. Using the known dates of identifiable volcanic tephra layers, and the depths of sediment above them, they concluded that sedimentation rates under pasture (13.0 mm/year) have been 8–17 times the rate under natural forest (1.47 mm/year), and 5–6 times the rate under the fern-scrub cover of Polynesian times (2.43 mm/year).

Rates of lake sedimentation vary widely, in response to rates of catchment erosion and fluvial sediment transport (Chapter 12). Lake Tutira is a rather extreme example, because of the very eroded nature of its catchment. Sedimentation rates in the nearby Lake Waikaremoana, whose catchment has a largely undisturbed forest cover, presently are less than 1 mm/year, although rates of up to 2.3 mm/year were inferred by Matthews (1992) during Polynesian times, apparently in response to the Kaharoa Tephra eruption.

Because of the importance of sedimentation rates for the management of hydroelectricity reservoirs, surveys are available for a number of lakes (Jowett 1984). For example, surveys of the bed of Lake Roxburgh during the period 1956–1989 indicated that the volume of the lake had decreased from 117 million m³ to 59 million m³, a 50% reduction in 33 years, at a rate of 1.8 million m³ (sediment volume) or 300 mm (average sediment depth) per year (Spigel and Viner 1992). Construction of Lake Dunstan upstream will prolong the life of Lake Roxburgh beyond the 66 years implied by such high rates of sedimentation.

Recent surveys of the Kawerau Arm of Lake Dunstan are particularly interesting, because the heavy sediment load contributed by the Shotover River and other tributaries to the Kawerau River are causing rapid infilling (Otago Regional Council 2000). Increases in bed level

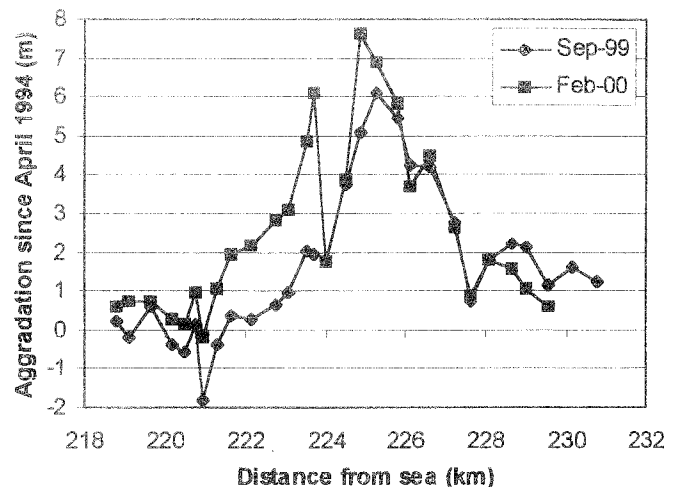


Figure 19.14 Sedimentation along the Kawerau Arm of Lake Dunstan, 1994–2000 (data from Otago Regional Council 2000).

during 1994 to 2000, averaged across surveyed cross-sections, are as great as 7.65 m (1,320 mm/year), with sedimentation advancing progressively downstream from the head of the lake (Fig. 19.14). Up to September 1999, most sedimentation was upstream of Bannockburn Bridge (river km 224.5). There was a major flood in November 1999; during and subsequent to the flood sedimentation was predominantly downstream of the bridge, and there was some re-mobilisation of sediment at the head of the lake. This sedimentation is causing substantial changes to the geomorphology of the river and lake, with the creation of extensive gravel bars, silt banks and wetlands, with attendant implications for recreational users, riparian landowners, and hydroelectric power generation.

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Chapter 18

River bird communities

Colin O'Donnell

INTRODUCTION

Freshwater rivers form a major habitat for over 160 species of water birds, many of which are unique to New Zealand (O'Donnell and Moore 1983; Maloney *et al.* 1997; Heather and Robertson 2000). Rivers can be classed into four broad habitat types that support markedly different water bird communities—braided rivers, single-channel lowland rivers, mountain torrents and the estuarine mouths of rivers on the coast.

The inhabitants of rivers range from those with generalised needs to those with more specialised requirements. Many water birds use rivers extensively, but also feed and breed successfully in other aquatic systems such as lakes, ponds and swamps, or occur at sea and along the coast. Some are not restricted to aquatic habitats, feeding or breeding in terrestrial zones as well. A range of species have specialised habitat requirements that restrict them to certain river habitats. Five endemic species have evolved on rivers and spend all, or crucial parts of their life cycle on them—blue duck¹, wrybill, black stilt, black-billed gull and black-fronted tern. Their specific adaptations for living on rivers include set migratory patterns, specialised foraging behaviours and the ability to breed in the unstable river environment (O'Donnell and Moore 1983).

This chapter describes the general characteristics and composition of river bird communities in New Zealand, the special adaptations of some bird species for living on rivers and their habitat requirements. The chapter concludes by describing the significance of rivers nationally and internationally for the conservation of threatened species. The major threats to the viability of many river bird species and the management needed to ensure their survival in the long term are described. If populations of

indigenous birds are to be sustained or restored, then conservation efforts need to focus on developing an integrated approach to management of rivers for birdlife, focusing on predicting the impacts of modifying river flows and developing techniques for maintaining suitable flow environments, managing exotic weeds that encroach on breeding habitats, undertaking river works at a time of year that does not threaten breeding birds, and controlling introduced predators and recreational impacts.

WATER BIRDS OF RIVERS

Eight guilds, or communities of water birds with similar requirements, use rivers in New Zealand (O'Donnell 2000). These guilds have similar foraging and nesting requirements, so they can be grouped by their use of habitat types (Verner 1984). Habitats are simply the places that birds live, and microhabitat is a term used to define a sub-component of a broader habitat type. For rivers, guilds are identified primarily by characterising the main substrates, vegetation types and the depth and velocity of water that species use for feeding and then grouping the species with similar characteristics together.

I have listed 47 freshwater species characteristic of river habitats that are present on the rivers themselves or at their mouths (Table 18.1). These are not all the species that use these habitats. Over 160 bird species have been recorded using rivers and their mouths in New Zealand (e.g., O'Donnell and Moore 1983; O'Donnell 1985; Sagar *et al.* 1999; Heather and Robertson 2000). The additional freshwater birds are classed as rare species, mainly originating from the Arctic, Asia or Australia. Some have small populations of less than 200 individuals occurring in New Zealand (e.g., Australian little grebe *Tachybaptus novaehollandiae*, sharp-tailed sandpiper *Calidris acuminata*),

¹ For scientific names see Table 18.1

Table 18.1 Guilds of indigenous water birds characteristic of the New Zealand rivers. River types are B = braided rivers, S = single-channel rivers, T = torrent rivers, M = river mouths and estuaries. # = threatened species (after Hitchmough 2002), † = arctic migrants.

Guild	Characteristic species Common name	Scientific name	River types
1. Open water divers	black cormorant #	<i>Phalacrocorax carbo</i>	B, S, T, M
	little cormorant	<i>P. melanoleucos</i>	B, S, M
	little black cormorant	<i>P. sulcirostris</i>	S, M
	piebald cormorant	<i>P. varius</i>	M
	spotted shag	<i>Stictocarbo punctatus</i>	M
	NZ scaup	<i>Aythya novaeseelandiae</i>	S
2. Deep water waders	white-faced heron	<i>Ardea novaehollandiae</i>	B, S, M
	white heron #	<i>Egretta alba</i>	S, M
	royal spoonbill #	<i>Platylea regia</i>	M
	black stilt #	<i>Himantopus novaeseelandiae</i>	B
	piebald stilt	<i>H. himantopus</i>	B, S, M
	spur-winged plover	<i>Vanellus miles</i>	B, M
	South Island pied oystercatcher	<i>Haematopus ostralegus finschi</i>	B, S, M
	variable oystercatcher	<i>H. uniclor</i>	M
	bar-tailed godwit †	<i>Limosa lapponica</i>	M
golden plover †	<i>Pluvialis fulva</i>	M	
3. Shallow water waders	wrybill #	<i>Anarhynchus frontalis</i>	B, M
	banded dotterel #	<i>Charadrius bicinctus</i>	B, M
	New Zealand dotterel #	<i>C. obscurus</i>	M
	black-fronted dotterel	<i>C. melanops</i>	B, M
	knot †	<i>Calidris canutus</i>	M
	turnstone †	<i>Arenaria interpres</i>	M
	red-necked stint †	<i>Calidris ruficollis</i>	M
4. Dabbling waterfowl	black swan ¹	<i>Cygnus atratus</i>	S, M
	paradise shelduck	<i>Tadorna variegata</i>	B, S, M
	grey duck #	<i>Anas superciliosa</i>	B, S, M
	NZ shoveler	<i>A. rhynchotis</i>	B, S, M
	grey teal	<i>A. gracilis</i>	B, S, M
	brown teal #	<i>A. aucklandica</i>	S, M
5. Torrent specialists	blue duck #	<i>Hymenolaimus malacorhynchos</i>	T
6. Aerial gulls and terns	black-fronted tern #	<i>Sterna albostrata</i>	B, M
	white-fronted tern #	<i>S. striata</i>	B, M
	Caspian tern #	<i>S. caspia</i>	B, S, M
	white-winged black tern	<i>Clidonias leucopterus</i>	M
	black-billed gull #	<i>Larus bulleri</i>	B, M
	red-billed gull	<i>L. novaehollandiae</i>	M
black-backed gull	<i>L. dominicanus</i>	B, S, M	
7. Swamp specialists	Australasian bittern #	<i>Botaurus poiciloptilis</i>	B, S, M
	marsh crake #	<i>Porzana pusilla</i>	B, S, M
	spotless crake #	<i>P. tabuensis</i>	S
	pukeko	<i>Porphyrio porphyrio</i>	S, M
	banded rail	<i>Rallus phillipensis</i>	M
	fernbird #	<i>Bowdleria punctata</i>	S, M
8. Riparian species	NZ pipit	<i>Anthus novaeseelandiae</i>	B, M
	welcome swallow	<i>Hirundo tahitica</i>	B, S, M
	NZ kingfisher	<i>Halcyon sancta</i>	B, S, M
	harrier hawk	<i>Circus approximans</i>	B, S, M

¹ Black swans are generally considered an introduced species. However, Worthy (1998) found that bones of the "extinct" New Zealand swan were identical to those of the introduced Australian species. Thus, it appears that black swan is an indigenous species that was inadvertently re-introduced to New Zealand.

or occur as accidental vagrants from overseas (e.g., white ibis *Threskiornis molucca*, Mongolian dotterel *Charadrius mongolus*), or as species that primarily use other habitats and are not dependent on rivers (e.g., the reef heron *Egretta sacra* occurs mainly along the coast and cattle egret *Bulbulcus ibis* usually feeds on farmland).

Open water divers

On rivers, this guild comprises shags and cormorants and diving ducks (Figs. 18.1 and 18.2). These species usually forage in open, deep waters and channels on both lakes and rivers, throughout New Zealand. They hunt by diving for fish and aquatic invertebrates taken in the water column or the bottom substrate. Scaup nest in vegetation overhanging the waters edge at water level. Cormorants usually nest high in overhanging trees (especially willows) and on overhanging rock outcrops.



Figure 18.1 Black cormorants on nest: an open water diver characteristic of most rivers in New Zealand.

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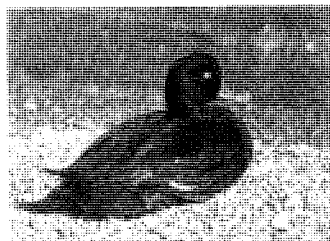


Figure 18.2 Male New Zealand scaup, a diving duck characteristic of single channel rivers.

Photo: Peter Morrison.
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Deep water waders

Deep water waders are birds with medium-long legs that allow them to forage in water depths of > 200 mm as well as in shallow water (e.g., stilts, herons, oystercatchers; Fig 18.3). They breed on the ground in open areas, especially shingle or sand free of emergent vegetation (Fig 18.4). They usually roost in flocks in similar habitats. This guild includes species that breed in New Zealand, as well as species that migrate here from the arctic once they finish breeding there (e.g., godwits; Fig 18.5). Thus, arctic migrants do not breed in New Zealand but use habitats here as their wintering grounds. Stilts, herons and oystercatchers occur widely on river habitats throughout New Zealand, but the majority of arctic migrants inhabit estuarine and coastal water bodies. However, they feed in microhabitats similar to those of resident waders.

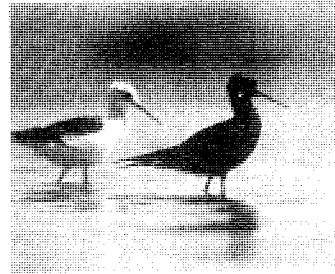


Figure 18.3 An adult and an immature black stilt wading in deep water.

Photo: Rod Morris.
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Figure 18.4 South Island pied oystercatcher nesting on bare shingle.

Photo: Gordon Watson.
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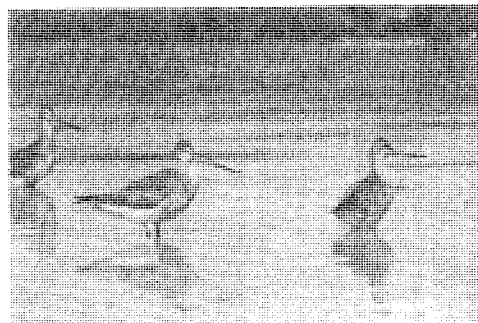


Figure 18.5 Bar-tailed godwits feeding in deep water at high tide, Manukau Harbour.

Photo: Dick Veitch.
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Shallow water waders

In contrast, shallow water waders have much shorter legs that restrict them to feeding in water less than 80 mm in depth, and most use is of water less than 40 mm deep (e.g., plovers, dotterels, sandpipers; Fig 18.6). They breed on the ground in open areas, especially shingle (Fig. 18.7), wetland turf, or sand, and usually roost in flocks in similar habitats. This guild includes species that breed in New Zealand and arctic migrant waders.



Figure 18.6 Wrybill feeding in shallow water along the edge of a braided river riffle

Photo: Aalbert Rebergen.

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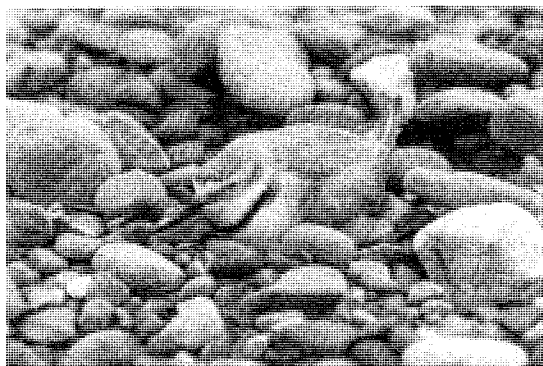


Figure 18.7 Wrybill on nest on clean shingle, Rakaia River

Photo: Dick Veitch.

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Dabbling waterfowl

Ducks and swans feed predominately by dabbling while floating on the open water of rivers, streams or lakes (Fig. 18.8) or they graze on wetland turf, saltmarsh and pasture. Most species nest within dense cover in swamps or riparian vegetation along watercourses and roost by floating on open water or by sitting on banks and shingle bars.

Torrent specialists

This guild contains species that specialise in foraging in the white, turbulent waters of fast flowing streams and rivers where they feed on aquatic insects (Fig. 18.9).



Figure 18.8 Dabbling ducks. From left to right: grey duck (1), grey teal (1) and New Zealand shoveler (3)

Photo: J. L. Kendrick.

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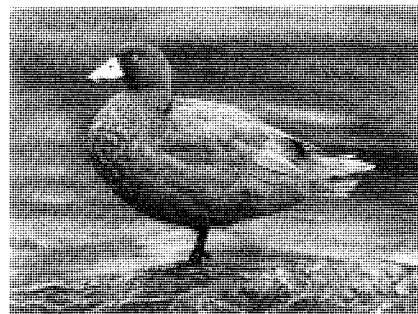


Figure 18.9 Blue duck, New Zealand's only torrent specialist.

Photo: Alan Reith.

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Aerial hunting gulls and terns

Gulls and terns are characteristic of aquatic and terrestrial habitats throughout New Zealand. While both groups of birds are generally associated with coastal habitats and even urban areas, several species are aerial hunters, characteristic of rivers. Black-fronted terns, and black-billed gulls in particular (Figs. 18.10 and 18.11), can be seen flying over open water or river channels and diving for invertebrates and small fish. Although they also feed at times over farmland, the black-fronted tern usually does so only in the breeding season when floods force them to do so, or in the winter when energy demands are not so high (Lalas 1977; Roberston *et al.* 1983). They nest almost exclusively on open shingle bars and islands in rivers.

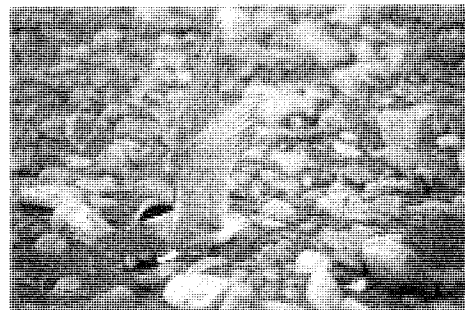


Figure 18.10 Black-fronted tern on nest with two eggs, Rakaia River.

Photo: Dick Veitch.

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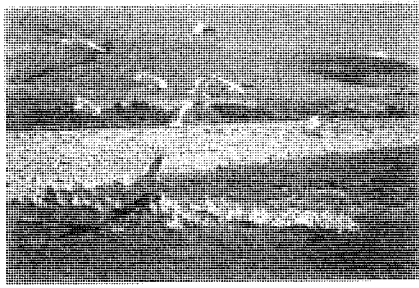


Figure 18.11 Black-billed gulls guard a crèche of fledglings, Cardrona Valley. Photo: P. Bush. Crown Copyright. Department of Conservation 2004

Swamp specialists

This guild comprises rails, such as marsh crake and pukeko, and bittern that dwell in dense swamp vegetation associated with wetlands (Fig. 18.12). Although usually associated with palustrine wetlands, they frequently occur along riparian margins of rivers where swamp vegetation occurs. Their diet consists of seeds and invertebrates gleaned from vegetation or surface water with good vegetative cover. They also consume vegetation itself, and, in the case of bittern, fish and amphibians. They generally nest within *Carex secta* or *Typha orientalis* and other rushes (e.g., Kaufmann and Lavers 1987; O'Donnell 1994).



Figure 18.12 The marsh crake is a common inhabitant of riparian wetlands along rivers of the South Island. Photo: Peter J. Moore. Crown Copyright. Department of Conservation 2004

Riparian wetland species

This group includes species that do not exclusively depend on either terrestrial or aquatic habitats (e.g., swallows, pipits, kingfishers). They do not depend on wetlands, though are often associated with them as much as any other habitat for either breeding or feeding.

BIRD COMMUNITIES OF BRAIDED RIVERS

The braided rivers commonly found in New Zealand are habitat types that are rare internationally (O'Donnell and Moore 1983). They represent some of the largest habitats for aquatic birdlife in New Zealand. Most shingle

ivers overseas form by different geological processes (Miall 1977). Canterbury has 60% of all braided river habitat in New Zealand, with other braided rivers also found mainly in the South Island, although a few small ones occur in Wellington, Manawatu, the Bay of Plenty and the East Coast of the North Island (Wilson 2001). Braided rivers are highly unstable, being characterised by high spring-summer flows, and rapid and frequent flooding.

Braided rivers are extensive habitats that provide a much wider variety of microhabitats than the single-channel rivers that are typical of other parts of New Zealand. The largest is the Rakaia River, which covers well over 30,000 ha. Such sheer size means that these habitats are extensive enough to support viable populations of a large number of bird species. The very wide range of channel sizes, shapes and flow rates, and extensive areas of shingle, islands and terraces provide diverse habitat that supports breeding and feeding populations of over 80 species of birds, many of which are unique to New Zealand (O'Donnell and Moore 1983). Individual rivers can support many thousands of birds. For example >20,000 wetland birds have been recorded on the Ashburton River at one time (O'Donnell 1992). Each braided river, while superficially similar to the others, is sufficiently distinctive in its habitat characteristics to provide for a unique combination of birdlife, each with different requirements (O'Donnell and Moore 1983).

Specialist adaptations

Specialist adaptations for living on braided rivers include migratory patterns, specialised morphological features of the birds, specialised foraging behaviours, narrowly defined ranges of preferred habitats, and the ability to breed in the unstable river environment, with rapid re-nesting and short intervals between egg-laying (O'Donnell and Moore 1983).

Many braided river species are migratory. They have a set annual cycle, and so their requirements change throughout the year. They arrive on the rivers in late winter and early spring, to take advantage of the increase in food supplies so they can breed there. They migrate away again in mid-summer to their wintering sites in areas with less harsh climates (usually coastal estuaries in New Zealand, particularly in the north, although some migrate to Australia).

The arrival of birds to breed coincides with the onset of increased flows in the river in spring and the associated rapid increase in the availability of aquatic invertebrates as previously dry channels become inundated again (Sagar 1983). The birds use this increase in food availability to improve their body condition after migration and to sustain themselves during periods when energy demand is high (when establishing territories, producing eggs, and then raising chicks). If this period of increased flow is

threatened, then many bird species (particularly waders and terns) would not have sufficient food supplies to sustain themselves through the breeding season.

The arrival of birds varies by several weeks, depending on the species and the location of the river. The first birds to arrive are South Island pied oystercatchers in late June and early July. The first riverbed specialists, wrybill, arrive in mid August. On the Rakaia River, nesting commences in the last week of August, with peak laying of the first clutch in the first week of September. Most first clutches are complete by the first week of October, but second clutches are laid through November and early December (Hay 1984; Hughey 1985). Most wrybill begin migration away from the rivers again in late December and January (Hay 1984).

Black-fronted terns appear to nest about a month later than wrybill. For example, on the Ohau River, the earliest nest for black-fronted terns was 26 September and the peak laying began in the second week of October. Black-fronted terns are present and courting at potential colony sites from early September (Lalas 1977; Keedwell 2002). The earliest banded dotterel nests on the Ohau are in the first week of September or last few days of August and peak laying begins in the second week of September. Near the coast, banded dotterels start breeding in late July and peak in September, then are finished by the end of December. However, at higher altitude on rivers in the Mackenzie Basin they don't start breeding until September (on the Tekapo and Ahuriri Rivers) or October (on the Tasman and Godley Rivers). In the high country banded dotterels finish nesting at the end of January (Bomford 1978; Pierce 1983).

Braided-river birds thus have relatively short, highly synchronous breeding seasons compared to most bird species. They are able to begin nesting rapidly on arriving on the breeding grounds and can re-nest quickly if spring flooding causes nest failure (Beer 1966).

Specialist feeding behaviour is at its extreme in the wrybill. This is the only bird in the world with its beak turned to one side (the right). This adaptation helps the bird sweep mayfly larvae from the undersides of flattened shingle along the margins of river channels (Pierce 1979). Other adaptations are less obvious, such as the dependence of black-fronted terns on aquatic insects drifting in the water columns of major channels (Lalas 1977) or the complex communication black-billed gulls use to identify patchy food sources (Beer 1966).

BIRD COMMUNITIES OF SINGLE-CHANNEL RIVERS

The bird communities of single-channel rivers throughout New Zealand are typically simpler, and total populations smaller, than in braided rivers. This is

generally because habitat areas, in terms of breeding and feeding, are substantially smaller on single-channel rivers. Compare for example, the braided Rangitata River, which covers > 18,000 ha and supports > 20,000 wetland birds, with a relatively good single-channel river for birds, the Styx River in Christchurch. Good habitat on the Styx covers <200 ha and usually supports a similar number of indigenous water birds (O'Donnell 2000).

There are exceptions. Some of the largest single-channel rivers such as the Waikato and Clutha support large wetland bird populations by virtue of their size. The only one studied in detail, the Clutha River, supported >11,000 wetland birds at one time in 1986 (Hughey *et al.* 1986). The composition of the bird community on the Clutha is typical of other single-channel rivers in that it has a higher proportion of open water divers and dabbling ducks and far fewer waders than braided rivers. In summer and autumn, waterfowl were at their peak on the Clutha with >7000 birds present, and, as is typical, they were dominated by the introduced mallard duck (*Anas platyrhynchos*), which at times composed 95% of the ducks present. Gulls were also common, but their numbers peaked in spring when black-backed gulls and black-billed gulls were nesting. However, terns were rare, with a maximum count of only 14 black-fronted terns in winter (Hughey *et al.* 1986).

BIRDS OF MOUNTAIN TORRENTS

Only one bird species, the blue duck, is characteristic of this habitat type in New Zealand (Fig 18.9), although cormorants also occur frequently. Blue ducks are rare and very cryptic, being partly nocturnal, and now occur only in the remotest mountain streams (Heather and Robertson 2000). In many catchments, populations on individual streams are very small, often numbering only one or two pairs. Most are confined to mountainous headwaters in the South Island and central North Island, but a small lowland population in the South Canterbury foothills is unique. It is thought that most populations were once linked via their low altitude reaches, but habitat modification and the impact of predators have isolated blue ducks, restricting them to headwater rivers. The blue duck chooses river sections that have high water quality, stable banks and low transport of sediments, riparian native vegetation, and an abundance of aquatic invertebrates. They nest in riverside caves, hollow logs, and beneath flax bushes (Adams *et al.* 1997).

BIRD COMMUNITIES OF RIVER MOUTHS

Bar-type lagoons and estuaries characterise river mouths along the whole New Zealand coast. These habitats support diverse bird communities, including all the guilds

mentioned above except torrent specialists. Lagoons and estuaries are influenced by both fluvial factors such as flows, flood rate and sediment load, but also by coastal processes. They supply a range of additional feeding, resting and breeding microhabitats not found upstream. These include the open water areas of lagoons and estuaries at high tide, open areas of mud or sand that are usually saturated or covered in a surface water film after being exposed following the receding tide, and saturated wetland turf or vegetated saltmarsh flats covered in prostrate wetland vegetation (e.g., *Mimulus repens*, *Selliera radicans*, *Cotula coronopifolia*, *Sarcocornia quinqueflora*). Emergent wetland vegetation, often in standing shallow water, or periodically inundated by the tide, is usually more extensive than on the river-beds proper. Such communities are often dominated by sedges and rushes (e.g., *Juncus* spp., *Schoenoplectus* spp., *Typha orientalis*, *Carex* spp., *Apodasmia similis*). In the north of New Zealand, mangroves form a characteristic part of the community.

These estuarine systems are usually dominated by shallow and deep water waders, some of which occur in vast numbers and tend to comprise mainly migrant species such as knots and bar-tailed godwits from the arctic and pied oystercatchers from the inland South Island. New Zealand estuaries support over 160,000 waders of local breeding species and a similar number of migrants from the Northern Hemisphere (Sagar *et al.* 1999). The largest estuaries, the Manukau and Firth of Thames, at times collectively support over 50,000 waders, with >35,000 godwits and >10,000 knots recorded (Veitch 1978). Add to these the less well-documented populations of waterfowl, gulls, terns and swamp birds (e.g., Bellingham and Davis 1983), and the importance of estuaries to freshwater birds is clear. Braided-river species such as wrybill and black-fronted terns also migrate to estuaries in the north of the North Island and to coastal South Island in the winter (Veitch 1978; Latham 1981).

There is a general positive relationship between estuary size and the numbers and diversity of water birds present, while catchments with larger runoff have lower species diversity (Whelan *et al.* 2003). Waters from the most productive estuaries often originate from hundreds of small catchments (e.g., Manukau Harbour), rather than from major rivers.

IMPORTANT MICROHABITATS FOR FEEDING AND BREEDING

All habitats can be subdivided into smaller components that members of each bird guild generally use for foraging, breeding or roosting (microhabitats). For example, on a braided river, the habitat can be divided into terrestrial or aquatic habitats, into zones with different substrates, or

levels with different flooding frequency. Channels can be divided into types with different flow characteristics. In reality, individual species have very complex habitat requirements that vary seasonally. Many studies have characterised patterns of microhabitat use for numerous freshwater species by describing in detail the types of wetland used and the microhabitat, including flow characteristics such as water colour, depth, speed, and channel width, and the type of substrates where feeding occurs (e.g., Robertson *et al.* 1983; Moore *et al.* 1984; Robertson *et al.* 1984; Hughey *et al.* 1986). Data within these studies are very detailed, and provide valuable information for resource managers when assessing the potential impacts of water level modification on freshwater birds.

Each guild of birds requires quite different microhabitats to sustain their breeding and feeding needs (Tables 18.2 and 18.3). However, requirements of individual species appear to be consistent across different rivers, largely because the foraging behaviour of most species is relatively fixed and different wading birds are limited to specific maximum water depths by the different lengths of their legs (e.g., Fig. 18.3 and 18.6). Thus, the different composition of bird guilds on a particular river and the population size of each freshwater species present reflect the types of microhabitat present and their relative abundance. Individual river systems complement each other. If conditions change from year to year, or season to season, bird usage may also change from one river to another, depending on the conditions on each river. However, with habitat degradation continuing, particularly in the form of flow modification, predation from exotic mammals and encroachment by introduced weeds, birds may lose their versatility of habitat choice.

The importance of different microhabitats can be seen by examining the different habitat types and use birds make of them across a typical river profile. Highest on the river profile on the margins of the river are small sheltered ponds and backwaters, often shaded by willows. These microhabitats usually result from subsurface water inflow or waters trapped during floods and freshes. Studies have consistently shown the importance of ponds and backwaters for deep water waders, open water divers and dabbling waterfowl. For example, on the Ahuriri River, 5–20% of feeding observations of black cormorants, 10% of paradise duck feeding and 15–30% of grey duck feeding were in this habitat type (depending on the season) (Robertson *et al.* 1983). On the Waitaki River, 5% of feeding observations of black cormorants and 5–10% of paradise duck feeding were in this habitat type (Robertson *et al.* 1984). Most of the use that bittern and marsh crake make of braided rivers is in this habitat type (up to 40% of observations on the Ahuriri, Tekapo, Waitaki and Opihi Rivers).

Table 18.3 Main microhabitats used for breeding by river bird species. Key to the microhabitat types: 1. Riparian vegetation; 2. River terraces; 3. Shingle bars and flats; 4. Major/deep channels and open water; 5. Shallow channels; 6. Mud and sand flats; 7. Wetland turf and saltmarsh; 8. Riparian swamps. * = habitat used for >10% of the time. Adapted from O'Donnell (2000).

	Main Breeding Microhabitats							
	1	2	3	4	5	6	7	8
1. Open water divers								
black cormorant	*	*						
little cormorant	*	*						*
little black cormorant	*							
piebald cormorant	*							
spotted shag ¹								*
NZ scaup								
2. Deep water waders								
white-faced heron	*	*						
white heron	*							
royal spoonbill	*							
black stilt		*	*		*			*
piebald stilt	*	*	*		*		*	*
spur-winged plover	*	*	*				*	
South Island piebald oystercatcher	*	*	*				*	
bar-tailed godwit ¹							*	
golden plover ¹							*	
3. Shallow water waders								
wrybill			*					
banded dotterel	*	*	*				*	
NZ dotterel						*		
black-fronted dotterel		*	*					
knot ¹								
turnstone ¹								
red-necked stint ¹								
4. Dabbling waterfowl								
black swan		*					*	*
paradise shelduck	*	*						*
grey duck	*	*						*
NZ shoveler		*						*
grey teal		*						*
brown teal	*							*
5. Torrent specialists								
blue duck	*	*						
6. Aerial gulls and terns								
black-fronted tern		*	*					
white-fronted tern			*					
Caspian tern			*			*		
white-winged black tern			*					
black-billed gull			*					
red-billed gull			*					
black-backed gull	*	*	*			*	*	*
7. Swamp specialists								
Australasian bittern								*
marsh crake								*
spotless crake								*
banded rail	*							*
pukeko	*	*						*
fernbird	*							*
8. Riparian species								
NZ pipit	*	*	*					
welcome swallow	*							
NZ kingfisher	*							*
harrier hawk	*	*						*

¹ Do not breed on New Zealand rivers or estuaries

Where the backwaters are out in the open, and not overhung with vegetation, they provide habitat for both short and long-legged waders. For example, on the Ahuriri River, 10–35% of white-faced heron feeding (depending on season of the year), 5% of wrybill feeding, 5–20% of feeding by stilt species, and 10% of South Island pied oystercatcher feeding was in this habitat type. Black-fronted terns hawked for insects for 5–15% of their time over backwaters (Robertson *et al.* 1983). On the Waitaki River 5–10% of white-faced heron feeding, 5% of oystercatcher feeding and 15–20% of pied stilt feeding was in backwaters (Robertson *et al.* 1984).

Smaller channels with riffles, runs and seepage zones often occur relatively high on the river profile. These channels, being very shallow, are prone to dry up if flows are reduced. They are particularly important for short-legged wading birds such as wrybill and banded dotterel, and to a lesser extent, long-legged waders (e.g., Bomford 1978; Pierce 1979, 1982; Hughey 1985, 1997). For example, on the Ahuriri River, up to 40% of feeding observations of banded dotterel, 30% of wrybill, 35% of pied stilt, and 15% of oystercatcher feeding were in minor channels (Robertson *et al.* 1983). Black-fronted terns and black-billed gulls hawked over small channels for up to 30% of the time. On the lower Rakaia, over 80% of wrybill feeding and 50% of banded dotterel feeding was in these minor channels (Hughey 1985).

Bars, flats and spits composed of shingle, cobbles, mud or wet sand comprise much of the terrestrial habitat within the riverbed and at the estuarine mouths. These microhabitats are important for breeding in the majority of braided-river birds, but also for foraging of some. For example, on the Ahuriri River, up to 30% of foraging time of pied oystercatchers and wrybill was spent on saturated or wet sand flats in summer. Banded dotterels spent up to 45% of their time feeding on wet sand in winter and 20% in summer, and they often spent more time in terrestrial areas in spring, with 50% of feeding on shingle and cobble bars.

Major channels (both riffles and runs) with large flows are very important for black-fronted terns. For example, on the Ahuriri River, 50% of spring-time feeding, 35% of late summer feeding, and 70% of autumn feeding were in this microhabitat (Robertson *et al.* 1983). On the Waitaki River, 55% of black-fronted tern spring-time feeding, 35% of late summer feeding, and 45% of autumn feeding was on major channels (Robertson *et al.* 1984). Lalas (1977) did a detailed study of black-fronted tern foraging behaviour on the Tasman, Ahuriri, Ohau and Waitaki Rivers. He found that black-fronted terns were specialists at feeding in major channels. They are mainly insectivorous and feed swallow-like on emerging aquatic insects on or above the water surface or by catching insects drifting below the surface by dipping their beaks under water while

in flight. The black-fronted tern is possibly the largest day-feeding, non-soaring bird that feeds solely while flying (Lalas 1977), therefore its energy costs of feeding are relatively high compared with other river birds. When the availability of aquatic insects is reduced in major channels (e.g., during freshes or inclement weather), black-fronted terns find alternative prey along river banks and in small channels. However, handling and search times, and competition for food increase significantly, thus raising the energy costs of hunting. Major channels are also important for black-billed gulls (Ahuriri River: 15% of feeding; Waitaki River: 10%), Caspian tern (Ahuriri River: 80% of feeding; Waitaki River: 90% of feeding) and white-fronted tern (Waitaki River: 90% of feeding). On the lower Rakaia River 40% of pied stilt foraging and 15% of oystercatcher feeding was in this habitat (Hughey 1985).

Riparian specialists such as swallows, pipits and kingfisher forage over a variety of microhabitats. Swallows forage across the river profile but particularly over backwaters, pools and runs. Pipits concentrate on shingle bars and the edges of both major and minor channels, whereas kingfishers generally hunt over backwaters (Robertson *et al.* 1983, 1984). Many braided-river birds also feed on farmland adjacent to rivers. However, use by all—except oystercatchers, spur-winged plover, black-backed and black-billed gulls—is generally very low. Activity by black-fronted terns over farmland is mostly limited to the non-breeding season (Lalas 1977; Robertson *et al.* 1983, 1984; Hughey 1985; Hughey *et al.* 1986).

Seasonal changes in habitat use can be subtle or quite noticeable. For example, black stilts used similar habitats between seasons, but in winter made proportionately more use of wetland areas and terrestrial habitats and less use of minor river channels (Robertson *et al.* 1983).

THE SIGNIFICANCE OF RIVERS FOR CONSERVATION

International significance

High levels of endemism among freshwater bird species occurring on rivers in New Zealand highlight their importance internationally. For many species this means that only here can their conservation be assured. Maintenance of self-sustaining populations of these endemic species is absolutely dependent on maintaining river habitats and reducing the impacts of threats implicated in causing their decline.

Internationally, braided rivers are rare habitat types. Most such rivers referred to in the literature from North America are river deltas or a type of braided river formed by different geological processes (Miall 1977). Elsewhere in the world, few bird species are specifically adapted to the type of braided rivers common in New Zealand (O'Donnell and Moore 1983).

Many New Zealand rivers fulfil criteria for designation as Wetlands of International Importance as laid out by the Ramsar Convention to which New Zealand is a signatory. The Convention includes criteria pertaining to sites containing representative, rare or unique wetland types. A wetland should be considered internationally important if it supports vulnerable, endangered, or critically endangered species or threatened ecological communities, if it supports populations of plant and/or animal species important for maintaining the biological diversity of a particular biogeographic region, or if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions. The Convention also considers a wetland to be internationally important if it regularly supports 1% or more of the individuals in a population of one species or subspecies of waterbird, a criterion often fulfilled when considering New Zealand rivers.

National and regional significance

Many rivers also fulfil criteria used to determine 'national importance', 'outstanding character' and 'significance' in the context of the Resource Management Act 1991 (RMA).

During the late 1970s and early 1980s the New Zealand Wildlife Service undertook the first national inventory of habitats of significance to wildlife, termed Sites of Special Wildlife Interest (SSWIs). Sites were assessed according to 16 criteria (Imboden 1978). The criteria used were based on those used internationally for identifying habitats of importance. Many braided rivers were classed as being of 'Outstanding' or 'High' value to aquatic birds using these criteria (O'Donnell and Moore 1983). High rankings were based on the presence of breeding populations of rare and endangered endemic species and their associated habitat. Many rivers appear to be of vital importance to threatened species such as black stilt, black-fronted tern, black-billed gull and wrybills, which have very limited breeding distributions. Braided rivers provide an extensive habitat type not represented elsewhere in the country. They are generally of large size and contain viable populations of many of the freshwater species that were typical of the habitat type and a diverse range of microhabitats for breeding, feeding and roosting.

Regional and district councils are required to recognise and provide for protection of significant habitats for indigenous fauna under Section 6 of the RMA. Definitions of 'significance' of sites in the context of the Act are being debated widely in the community. However, several nature conservation principles are generally accepted as being relevant to sustaining wildlife (e.g., Soulé 1986; Myers *et al.* 1987; O'Connor *et al.* 1990; Norton and Roper-Lindsay 1999). That is, significant habitats should contain representative fauna communities, and/or natural diversity,

and/or rare and distinctive species. They should be relatively intact and well connected (spatially or temporally) with other habitats, contain wildlife that is important in maintaining ecological processes, and be viable in the long term. A large number of New Zealand rivers are significant in the context of these criteria. Representativeness indicates how typical a habitat is of a particular ecological region and the quality of the site at representing the habitat type. Implicit in assessing representativeness in terms of bird communities includes determining the proportion of indigenous water bird guilds represented within a habitat, overall species diversity, and whether the habitat supports taxonomically distinct species. Generally, size of habitat is considered a measure of its potential carrying capacity (total numbers of birds) and of the diversity of species, guilds, communities and microhabitats likely to be present, and thus its overall life support capacity (e.g., Preston 1962; MacArthur and Wilson 1967; Gilpin and Hanski 1991; Whelan *et al.* 2003). Rivers and estuaries that would score highly in terms of life support capacity are often those that are large (thousands of hectares), support thousands of birds and have the ability to provide basic life functions of species guilds (breeding, roosting and feeding habitat). High natural diversity can be measured in terms of how many species are present and the number of key microhabitats present. Areas with greater diversity tend to have the greatest potential for maintaining all life support functions throughout much of the annual or lifetime cycle of a species.

Significant habitats are also those that have special features that make them distinct from all or most other sites. They are distinctive because they provide a specific life history function not provided by other sites. Sites can be distinctive because they are (a) special examples of particular habitat types, (b) important over-wintering sites away from the breeding grounds, (c) significant stopover points where migratory species can replace fat reserves lost during migration and build up condition to help survival through breeding or over winter, (d) one of few breeding areas for a species, (e) one of the largest breeding sites in the region, (f) important moulting sites, or (g) provide for some other specialist requirements for a particular species.

Threatened species, by definition, are declining at such a rate that they will become extinct in the short to medium term if nothing is done to reverse the declines in their populations. As a consequence, all sites that have threatened species (even though not necessarily every year) are of national significance because of the role they play in recovering populations. However, the relative significance of sites depends on how rapidly species are declining, their population size, and recovery potential.

The role of networks

Not all rivers are significant in themselves, especially small single-channel rivers and small estuaries. However, when they form part of a network that collectively provides crucial habitat for a species, they can become significant. For example, collectively the small rivers, streams and ponds in the Christchurch area support over 5000 New Zealand scaup, approximately 25% of the world population of this species. Networks of stop-over points visited by birds during migration when they need to replenish diminishing fat reserves can also be critical.

THREATS TO RIVER BIRDS

River bird communities and their habitats are subject to a wide variety of threats, which affect the viability of these aquatic bird communities throughout New Zealand. The threats are largely human-induced, and affect all river types, though their magnitude will vary from site to site. Some threats originate from direct management of aquatic habitats and activities within them, while others result from land use in the surrounding catchment. Overall, the relationships among different threatening processes are very complex and they often interact with each other (Fig. 18.13).

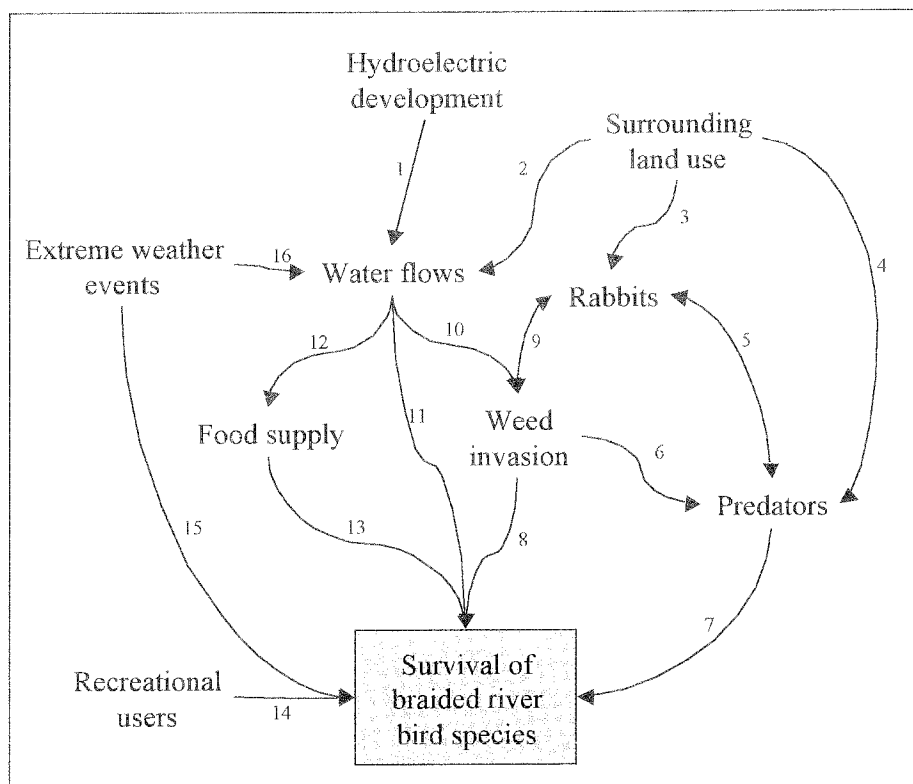


Figure 18.13 Interrelationships of some factors influencing the survival of braided-river birds. Examples of hypothesised interactions between the different factors include: 1 – water is redirected into canals for power generation, lowering river flows and flood frequencies; 2 – water is abstracted for irrigation of farmland, lowering river flows; 3 – rabbit control on farmland lowers rabbit abundance; 4 – differing land-use practices changes habitat availability for predators; 5 – predator abundance is altered by changes to rabbit abundance but also helps control rabbit abundance; 6 – vegetation on riverbeds provide cover for predators; 7 – predators prey on eggs, chicks and adults; 8 – weeds clog up breeding habitat and alter feeding habitat; 9 – vegetation provides cover and food for rabbits, but some weed species are controlled by rabbit grazing; 10 – lowered water flows and floods allow vegetation to establish on riverbed; 11 – floods destroy nests; 12 – lowered water flows can alter abundance of aquatic insects; 13 – food abundance can influence survival of young or condition of breeding adults; 14 – fishers, campers and four-wheel drivers can destroy nests or disturb breeding birds; 15 – extreme cold spells can kill eggs and chicks; 16 – high rainfalls can cause floods.

Source: R. Keedwell, Science & Research Unit, Department of Conservation, Christchurch.

Introduced predators

It is now well established that introduced mammalian predators (Fig. 18.14) have a major impact on the productivity and survival of water birds, particularly those on braided rivers. Black stilts, wrybill, banded dotterels and black-fronted terns are especially vulnerable to predation, with eggs, chicks and adult birds all being preyed upon. Although the best data on predator densities come from the upper reaches of braided rivers, we know that introduced predators are also very common in lowland areas of New Zealand as well (e.g., Fitzgerald 1964, Hughey 1985; Ragg *et al.* 1994; Alterio and Moller 1997; Alterio *et al.* 1998; O'Donnell 2000; Gillies 2001; Moss and Sanders 2001). Considerable research on this problem is being undertaken in New Zealand so that effective predator control programmes can be developed.



Figure 18.14 Infrared video image of a feral cat killing a banded dotterel on its nest at night. Photo: Mark Sanders. Crown Copyright. Department of Conservation 2004

The impacts of predation depend not only on numbers of predators and their prey present (Brown and Keedwell 1998), but also on the flow characteristics of the river (number and speed of channels) and the density of exotic weeds (Fig. 18.13; Taylor 1984; Hughey 1985; Pierce 1987; Pascoe 1995; Rebergen *et al.* 1998). Any reductions in flows of rivers that reduce the number of channels or their depth appear to reduce the availability of predator-free islands for birds to nest on safely (Bomford 1978; Pierce 1987). Lower flows would not only increase access by predators but also by people, their vehicles and stock—all of which crush or trample nests. Although it is well known that predators can swim across river channels, it appears that the frequency with which they do so is limited considerably by the river flow. Pascoe (1995) radio-tracked predators and compared trapping rates and the rates of predator footprints in river banks to that of islands isolated by major river channels on the Tekapo and Ahuriri Rivers. Pascoe found that on islands in mid-channel, the densities

of cats (the main river bird predator) were 11 times lower, and those of ferrets 3 times lower than on adjacent river banks. The only mustelid tracks recorded and all radio-tagged hedgehogs, were on mainland sites with none on islands.

On the lower Rakaia River, 95% of wrybill nests and 90% of banded dotterel nests were on islands separated from the riverbanks by flowing channels, generally with flows of >10 cumecs of water. There, flooding caused most of the failures in breeding and only 10% of wrybill nests and 12% of dotterel nests were lost to predators. However, on the much smaller Ashley River, where nests were on banks or islands separated by channels with flows of only 1 or 2 cumecs at most, predation was more likely to affect breeding success (33% of nests lost to predators, Hughey 1985).

Encroachment of exotic weeds on riverbeds

Introduced weeds, particularly broom, yellow lupin, Russell lupin and gorse, are particularly invasive and are a serious threat to habitats of braided-river birds (Stead 1932; O'Donnell and Moore 1983; O'Donnell 1992; Maloney 1993; Brown 1999a, 1999b; Maloney *et al.* 1999). Exotic weeds cover large areas of many braided riverbeds that were formerly bare shingle or covered in indigenous mat plants. Areas clear of emergent plants are key breeding and foraging habitats for many birds (Fig. 18.7). Apart from their direct impacts, exotic weeds are thought to stabilise shingle islands, increase deep channelisation of the riverbed, decrease the availability of shallow water foraging areas, and increase risk of predation (O'Donnell and Moore 1983; Robertson *et al.* 1983; Balneaves and Hughey 1990; O'Donnell 1992; Pascoe 1995; Hughey and Warren 1997; Rebergen *et al.* 1998). When indigenous birds are forced to nest closer to river channels because higher terraces are now covered in vegetation, they are subject to an increased frequency of flooding from freshes and floods. Natural patterns of floods and freshes are the main way that weeds are removed from river beds, although not all vegetation is ever cleared this way.

Abstraction and diversion of water for irrigation and other uses

Maintenance of flow regimes on rivers is essential for maintaining accessible food supplies, and the feeding and nesting habitat of river birds dependent on freshwater, particularly threatened species (Hughey 1985, 1997, 1998; Hughey *et al.* 1989). Braided-river bird communities in particular are highly attuned to the natural flow regimes and instability of rivers (O'Donnell and Moore 1983; Robertson *et al.* 1983).

The precise effects of water abstraction will depend on

how much water is taken. The impacts are likely to be incremental as more and more water is abstracted. Taking very small amounts of water (by say one small irrigation scheme) might not have a significant effect in itself (depending on the amount to be taken), but when combined with other abstractions over time would significantly reduce aquatic habitat for some species. A diversity of shallow- and deep-water habitats is needed to maintain the variety of bird guilds present on a river, but abstraction will affect different species in different ways. Abstraction of water has the potential impact of reducing critical feeding habitat. Reduction in flows firstly reduces the size and number of small channels, as well as the overall useable area of aquatic habitat for the bird community. Shallow water waders are dependent on small channels, seepage zones and backwaters, so would be directly affected by channel reduction if the water table on the river were lowered. However, the aerial feeders will be affected more by reductions in larger channels. Abstraction has indirect impacts on breeding habitats as well. For example, there may be greater magnitudes of fluctuations in river flow. This means that birds like black-fronted terns that like to nest close to water will nest lower within the channel, and be subject to increased flooding because freshes would still be of normal size.

Periods of prolonged low flow result in a greater accumulation of sediment, with detrimental effects—reduced availability of invertebrate food sources, increased algal growth, lower oxygen concentrations causing invertebrate deaths, and decreased velocity (affecting composition of invertebrates, particularly invertebrate drift). For example, any flow modifications that would reduce the drift of mayflies in larger channels may reduce food availability to black-fronted terns.

The different requirements of the full range of bird guilds all need to be taken into account when trying to understand their habitat and flow requirements. One flow regime is not going to be optimal for all bird species—short-legged waders need very shallow water for feeding, medium-legged waders need moderately shallow waters, and aerial feeders feed over deeper channels. Most species require bare shingle areas for nesting and roosting. Thus, maintenance of natural flow characteristics and variability of flows on the rivers is more likely to cater for the full diversity of bird groups than much reduced or stable flows.

Flood control works and channelisation

A variety of flood control works (construction of stop banks, channelisation, clearing floodways with bulldozers, spraying exotic weeds) can disturb birds directly, reduce their breeding success and modify riverbed habitats. Machinery can immediately destroy feeding areas and nest sites, and river control works can be potentially

catastrophic if undertaken during the spring-summer breeding season. Stopbanks and channelisation leads to confinement of the river channels, a reduction in the number of active channels, a decreased area of habitat, and a reduction in the ability of channels to change direction and renew areas of bare shingle.

Dams

Dams directly threaten riverbed birds by flooding significant nesting and feeding habitats (e.g., creation of the dams on the upper Waitaki River). Discharges from dams influence siltation down stream, changing the character of the water and thus food availability for birds. Untimely discharges from dams upstream can flood nesting sites of birds, and reductions in freshes and floods can lead to channel stabilisation, reduction in food availability and increased encroachment of weeds. If flows are reduced so much that channels dry up, there will be an almost complete loss of aquatic birds (e.g., Pukaki River, Maloney 1999).

THREATENED SPECIES

Rivers support populations of 18 threatened species that are characteristic of the habitat type (Table 18.1). Of these, black stilt, blue duck, wrybill, black-fronted tern and black-billed gull, and to a lesser extent banded dotterel and brown teal, are dependent on rivers and their associated wetlands for their food, breeding sites, and shelter during at least one phase of their life cycle. They have become rare because they no longer have enough suitable feeding and breeding sites, or adequate shelter to sustain their numbers. Individuals are preyed upon by introduced pests at a rate higher than the animals can replace themselves naturally, or introduced species out-compete them for the resources that they require. These species have been declining steadily since at least the time when Europeans arrived in New Zealand, and all habitats where they occur should be considered key sites for their recovery.

Ensuring the continued viability, recovery and health of these populations, and the flow regimes that sustain them, should be paramount goals for the management of their habitats. Threatened species, with their specialised requirements, tend to be those least able to cope with the changes brought about by a modification or loss of habitat, so they can be good indicators of ecosystem health and of the impacts of management practices.

The threatened species include the black stilt, which is by far the rarest bird species inhabiting rivers (Fig. 18.3). It is critically endangered—it will soon be extinct if management cannot reverse its decline. The total population of black stilts numbers less than 200 individuals, with only a few breeding pairs left, all in the

Mackenzie Basin. Black stilts rarely attempt breeding elsewhere in Canterbury now (O'Donnell 1988). However, they were once common on the lowland reaches of rivers in New Zealand. The Department of Conservation is developing techniques for captive rearing and release of fledglings, as well as for protection of birds nesting in the wild in the Mackenzie Basin. The intention is to restore populations in Canterbury in the future, and the current work programme for the species involves evaluating future release sites for establishing other populations outside the Mackenzie Basin (Maloney and Murray 2001). Black stilts are highly dependent on shallow channels on braided rivers for both breeding and feeding, and on the shallow margins of lakes and ponds in winter (Budgeon 1977; Robertson *et al.* 1983). A small proportion of the population leaves the Waitaki Basin in winter and can be seen on estuaries across the country.

Also included is the black-fronted tern (Fig. 18.10), which is restricted to New Zealand, where it breeds almost entirely on eastern South Island braided riverbeds (O'Donnell and Moore 1983). It depends on large channels for feeding, and on safe bare areas of shingle for breeding (Lalas 1977; Robertson *et al.* 1983). Populations are far below carrying capacity and have been slowly but steadily declining on both breeding and wintering grounds. Some populations that have been monitored have declined by 25–75% over the last ca. 20 years (the Opihi, Orari, Ashburton, and Ahuriri breeding rivers and Bay of Plenty wintering grounds; O'Donnell 1992; Maloney 1999; Keedwell 2002). The national population is thought to number between 6,000 and 10,000 birds (Keedwell 2002). Black-fronted terns now appear to be thriving on only five rivers that have relatively high flows—the Rangitata, Waitaki, Rakaia, Waimakariri and Wairau Rivers. The Waitaki and Wairau populations are now threatened by major power-generation developments, and the Rangitata by major irrigation schemes. Black-fronted terns nest almost exclusively on shingle on islands within braided rivers and specialise in feeding on mayflies drifting in larger river channels (Lalas 1977). They form small colonies, widely scattered on braided rivers in the spring. At this time virtually all their feeding is on the river, although after breeding they forage more frequently over farmland.

The wrybill is a New Zealand endemic, with the majority breeding on braided riverbeds of Canterbury and a small number in Otago (Fig. 18.7). Wrybills are specialised for foraging in shallow riffles and runs on braided rivers (Pierce 1979; Hughey 1997) and they breed on bare shingle flats (Hay 1984). The carrying capacity of riverbed areas dominated by small channels is high because the wrybill's territories can be significantly smaller than in areas dominated by major channels (Hughey 1998). This

reflects the greater biomass of aquatic invertebrates accessible to wrybill in shallow channels (Sagar 1983; Hughey *et al.* 1989). Wrybill migrate to the coast in late summer following breeding, and then to the North Island, particularly the Auckland harbours and the Firth of Thames (Veitch 1978, 1979). Nationally, wrybill populations have been declining significantly, from a maximum population of ca. 7000 birds in the 1960s and 1970s (Sibson 1963, Hay 1984), to 5111 in 1994 (Davies 1997), and 4100–4200 birds in 2001 (Riegen and Dowding 2001).

CONSERVATION MANAGEMENT OF RIVER BIRDS

If significant populations of freshwater bird species are to be sustained or restored in the long term, then management programmes need to integrate a combination of techniques appropriate to each river—predator and weed control, the retention of sufficient environmental flows to maintain feeding and breeding habitats, and sensitive day-to-day management of rivers and water bodies.

Land-use activities that adversely affect important habitats of freshwater bird species within the beds of rivers and their margins should be avoided. Managing rivers need not be detrimental to indigenous birds, at least in some situations. Integrated management will need to involve managers from regional and district territorial authorities, the Department of Conservation, affected parties, iwi and the community.

The Department of Conservation (DoC) is a lead agency in implementing the Government's 'New Zealand Biodiversity Strategy' (Department of Conservation and Ministry for the Environment 2000). Goal 3 of the strategy has become a major focus of the Department's efforts over the last two years ('Halt the decline in New Zealand's Biodiversity'). This goal requires that we 'maintain and restore a full range of remaining natural habitats and ecosystems to a healthy functioning state' and 'maintain and restore viable populations of all indigenous species and subspecies across their natural range.' This includes protecting a range of populations of each species so that their genetic diversity is also protected. Responding to this goal is leading to an expansion of the Department's efforts. Specifically, the Department's Key Step 1 is to 'Protect and restore New Zealand's Natural Heritage' (Department of Conservation 2003, Page 25).

A priority for improvement is freshwater sites. The Department recently reviewed the status of New Zealand's threatened species and identified 2373 species as threatened, many of which occur mainly in freshwater habitats (Hitchmough 2002). The Department is also

responsible for issues pertinent to the Wildlife Act (1953). Virtually all indigenous species that occur on rivers are protected under this Act. Therefore, the Department is concerned with preventing extinctions in the first instance and then restoring native populations so that they are sustained in perpetuity with a minimum of management. As a move towards this, the Department has recently begun setting up a Braided Rivers Recovery Group. The Department, in partnership with the community, is strengthening its efforts to reverse declines in the blue duck through the Blue Duck Recovery Programme. The threat category to this species has recently been revised to a more critical level—nationally endangered. Two management techniques are being applied experimentally—predator control at key sites, and captive breeding and release into the wild at these sites. The first experiments were conducted in Taranaki and they are now being applied in Fiordland and Nelson. The Nelson restoration effort involves integrated pest control, led by the community. If successful, these efforts will serve as models for restoring populations elsewhere in the country.

Regional and district councils are also beginning to play key roles in the management of the biodiversity of rivers. When carrying out their functions, they are required to take into account a range of natural and cultural values (RMA: s6, s7, s8). They are required to recognise and provide for the protection of areas of significant vegetation and significant habitats of indigenous fauna from inappropriate subdivision, use and development (s 6(c)), and the relationship of Maori and their culture with taonga (6(e)). There is also a duty to have a particular regard to the intrinsic values of ecosystems (7(d)). With the Government beginning to implement its national biodiversity strategy, there is an expectation that local authorities will play a more active role in safeguarding indigenous biodiversity. The development of regional water plans provides a mechanism to recognise the significant values of rivers for freshwater birds and to manage those values appropriately.

CONCLUSIONS

Rivers should be managed to sustain the outstanding wildlife communities that occur in these habitats. Sustainability, for bird populations, is defined in terms of ensuring the persistence of sufficient amounts of foraging, roosting and breeding sites for wildlife to maintain viable populations in perpetuity. There needs to be sufficient habitat available so that birds can disperse, and still thrive, if unfavourable conditions develop in particular parts of the habitat. There also needs to be sufficient habitat left to restore habitats of threatened species in the future. The different requirements of the full range of bird guilds all

need to be taken into account when trying to understand and maintain their habitat. One flow regime is not going to be optimal for all bird species, and maintenance of foraging habitat would be meaningless without protection of a viable amount of preferred nesting, roosting or breeding sites as well—and *vice versa*.

Maintenance of environmentally sensitive flow regimes in braided rivers is essential for maintaining feeding and nesting habitat of river birds, particularly threatened species. Maintenance of flows as close as possible to natural regimes during the critical breeding months (August to January) will ensure that the full range of channel types required for birds are present—backwaters and pools, seepage zones, saturated sand banks, minor and major channel riffles and runs; shallow channels <40 mm deep for shallow water waders, channels up to 200 mm deep for moderate-sized deep water waders, and very deep, fast channels for black-fronted terns.

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Chapter 17

Fish communities in rivers and streams

Angus McIntosh and Robert McDowall

INTRODUCTION

As a result of its long slender shape and steep topography, the New Zealand archipelago is richly endowed with rivers, many of them quite short and swift-flowing with coarse gravel/cobble substrates (see Chapter 8). Early Maori made extensive use of fish inhabiting these rivers, harvesting primarily shortfin and longfin eel, grayling, lamprey, smelt and whitebait. It is likely that these fisheries were critical sources of protein for Maori in inland areas (McDowall 1990). Although early European settlers were disparaging about the fish fauna of New Zealand rivers (Hursthouse 1861; Spackman 1892), this stemmed largely from ignorance of the fauna owing to the fish tending to be secretive and often nocturnal. Partly, however, it was also because the sorts of species that the settlers knew in British rivers were lacking. To remedy this, they sought to introduce exotic species, especially salmonids (McDowall 1994). Interestingly, while these introductions initially added to the diversity, they also have resulted in widespread local extirpations of the native fauna (McDowall 1990, 2003; Crowl *et al.* 1992). Taking a contemporary, overall view of the riverine fish fauna we now recognise about:

- 38 native “freshwater” species (Table 17.1)—though the number is uncertain because we are still discovering new species (7 species have been described in the past 7 years) (McDowall & Wallis 1996; McDowall 1997a; McDowall & Chadderton 1999; Ling & Gleeson 2001; McDowall & Waters 2002, 2003);
- 21 exotic species that have been introduced to our waters from overseas (Table 17.2)—though the number of exotic species is in a state of flux because some formerly listed species may no longer be present in natural habitats, while other species are becoming established here; and
- 7 primarily marine species that invade well into lower

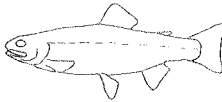

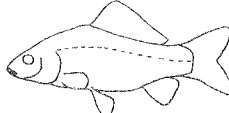


river systems—though the number of marine invaders varies according to opinions about how much a marine species needs to live in fresh water to be included in the fauna. In general we will not discuss the marine invaders.

An interesting aspect of the fauna is that nearly all of our freshwater fish species are found largely, if not exclusively, in riverine habitats—only three of the native species and none of the exotic species can be described as primarily lake-dwellers (though some have both riverine and lacustrine populations, and some exotics such as rudd and goldfish may be common in weedy ponds—see Chapter 25 for discussion of lake fishes).

By far the most speciose family in the fauna is Galaxiidae, with 22 described species (56% of the fauna); New Zealand is certainly one of the centres of diversity for this southern cool-temperate family. In addition to their broad southern cool-temperate relationships, parts of the fauna have affinities with that in Australia. Many species are in decline and of conservation concern, especially many of the galaxiids, which are adversely affected by habitat modification as well as the spread of exotic species, especially salmonids (Hitchmough 2002; McDowall 2003; Barner 2003; Table 17.1).

In this chapter we explore influences and processes that affect distribution patterns of our freshwater fishes, and which contribute to formation of fish communities in our rivers. These influences are a complex and fascinating amalgam of historical and ecological processes that are hard to tease apart, though we attempt to do so here. Before doing so, it is important to note a distinctive aspect of the fauna’s ecology. Probably more than in any other country, many of our freshwater fishes exhibit regularly timed migrations to and from the sea, a phenomenon known as “diadromy” (McDowall 1988), which comes in three distinct forms: anadromy, catadromy and amphidromy (Fig. 17.1). Nearly half our species are diadromous and, as

Table 17.2 The exotic freshwater fishes of New Zealand.

Family	Species	Status/comments
Salmonidae (trout and salmon & char) 	<i>Salmo trutta</i> (brown trout)	sports fish – widespread
	<i>Salmo salar</i> (Atlantic salmon)	sports fish – Fiordland lakes*
	<i>Oncorhynchus mykiss</i> (rainbow trout)	sports fish – widespread
	<i>Oncorhynchus tshawytscha</i> (chinook salmon)	sports fish – Eastern South Is.
	<i>Oncorhynchus nerka</i> (sockeye salmon)	sports fish – Waitaki Valley*
	<i>Salvelinus fontinalis</i> (brook char)	sports fish – scattered populations
Ameiuridae (catfishes) 	<i>Ameiurus nebulosus</i> (brown bullhead catfish)	pest fish – Central Nth Is. & West Coast
Cyprinidae (carps) 	<i>Carassius auratus</i> (goldfish)	ornamental fish
	<i>Cyprinus carpio</i> (European or koi carp)	pest fish
	<i>Tinca tinca</i> (tench)	sports fish†
	<i>Scardinius erythrophthalmus</i> (rudd)	pest fish – noxious
	<i>Leuciscus idus</i> (orfe)	illegal introduction
	<i>Ctenopharyngodon idella</i> (grass carp)	used in weed control
	<i>Hypophthalmichthys molitrix</i> (silver carp)	used in weed control
Poeciliidae (live bearers) 	<i>Gambusia affinis</i> (gambusia)	pest fish
	<i>Poecilia reticulata</i> (guppy)	released aquarium fish
	<i>Poecilia latipinna</i> (sailfin molly)	released aquarium fish
	<i>Xiphophorus helleri</i> (swordtail)	released aquarium fish
	<i>Phalloceras caudimaculatus</i> (caudo)	released aquarium fish
Percidae (perch) 	<i>Perca fluviatilis</i> (perch)	sports fish†

* Status of populations precarious † Regarded by some as pest fish

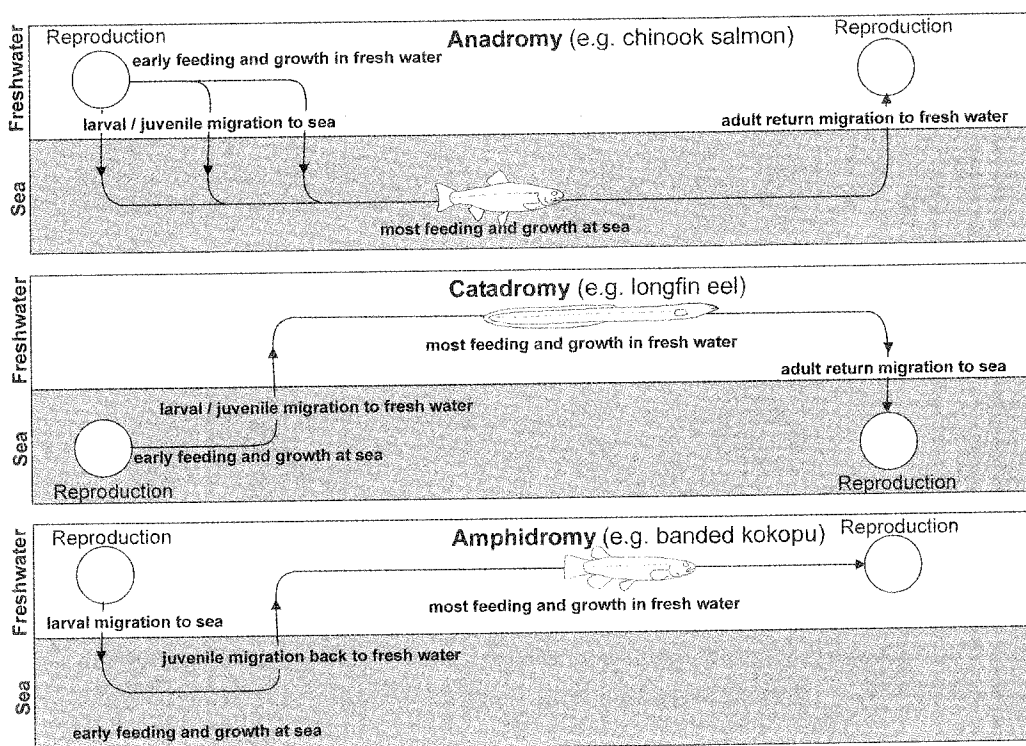


Figure 17.1 The three forms of diadromy (redrawn from McDowall 1998).

we discuss below, diadromy has profound implications for the distributions of species, the assembly of fish communities, and the ability of fish communities to recover from adverse impacts.

WHAT CONTROLS FRESHWATER FISH DISTRIBUTIONS?

The fish species present at any locality vary widely, and their diversity is controlled by a series of interacting influences. Firstly, habitat suitability is important. Species will generally be present only where there is suitable habitat (though that does not necessarily mean that they will always be present where there are suitable habitats). For several reasons, the species present at any site may not reflect the full range of species in our fauna for which that site is apparently suitable.

The role of history

Species evolve in restricted geographical areas and, especially for freshwater fishes, their ability to disperse more widely across the landscape is severely constrained by the lack of freshwater habitat continuity and the linear shapes of river systems. So, some species are present at certain localities but absent from others simply because of 1) the vagaries of their evolutionary history, and 2) changes in the geological landscape that have influenced the characteristics and connectivities of the rivers in which they live. Species that evolved, for instance, on the old Otago Peneplain, inland from Dunedin, are absent from Northland simply because they have been unable to reach Northland, or even Marlborough freshwater habitats. They tend to be “stuck” where they evolved, or have spread only marginally into a wider area contiguous with the area where they evolved.

In some closely-related species groups, for which we understand genealogical relationships, the distributions of species reflect aspects of New Zealand’s turbulent geological history. The distributions of species of mudfish (*Neochanna*) reflect this as, for example, in the northern and central South Island where the brown mudfish, is found only along the West Coast (west of the Southern Alps) and north of about Okarito, whereas the Canterbury mudfish is found only east and south of the Southern Alps, and south as far as the Waitaki River

in North Otago (Fig. 17.2A). On the face of it, this clearly looks like a speciation event in which a former continuous ancestral mudfish stock in the northern South Island was sub-divided by uplift of the Southern Alps beginning in the Pliocene, about 5 million years ago (and it is still going on). An observant naturalist might ask, however, why mudfish spread south along the West Coast only as far as Okarito. One such naturalist was Malcolm Main (1989), who suggested that absence of brown mudfish south of Okarito could reflect the impacts of Pleistocene glaciation, when extensive ice sheets covered the land further to sea than the present sea shore. If he is right, then we would expect other faunal elements to demonstrate the same restricted southern distribution—and so they do: freshwater fish examples include dwarf galaxias and upland bully (Fig. 17.2B,C). Their southern distributional limits down the West Coast vary somewhat, but they are concordant with the absence of *Nothofagus* beech trees in South Westland, south of about the Taramakau River (Wardle 1991), so there is an explainable pattern. But why no Canterbury mudfish south of the Waitaki River valley in the east? Probably, mudfish became established in proto-Canterbury and spread south across the formative plains but, it seems, have been unable to invade the Otago Peneplain—maybe there have been no effective riverine connections, or perhaps when there have been, mudfish have been unable to use them to disperse southwards into suitable habitats in Otago and Southland.

The same observant naturalists might also see that the brown mudfish (again, also dwarf galaxias and upland bully) are present on both sides of Cook Strait (Fig 17.2A), and might wonder how this can be so: It is simply that,

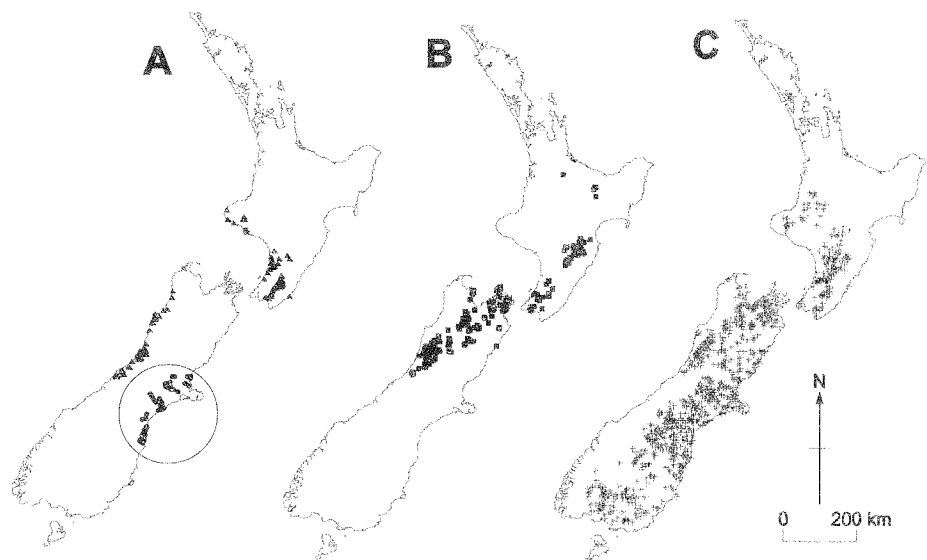


Figure 17.2 Distribution of non-migratory fish species using records from the NZ Freshwater Fish Database: (A) brown mudfish (triangles) and Canterbury mudfish (circled); (B) dwarf galaxias; and (C) upland bully.

also in the Pleistocene, there was a land connection across Cook Strait at which time some South and North Island rivers became interconnected, facilitating spread of aquatic fauna across what is now Cook Strait (Te Punga 1953; Fleming 1979; Lewis & Carter 1994). So there are patterns in the distributions of species and species groups that, if we look carefully and insightfully, still reflect quite old events in the geological and climatic history of New Zealand.

The role of diadromy

As mentioned above, migration of diadromous species into riverine populations has strong influences on distribution. Their distributions provide biogeographical scenarios that are quite different from those driven by history. Diadromous species tend to have overlapping distributions, are very widespread across New Zealand's latitudinal range, and none of them has a distribution that seems to reflect New Zealand's geological history (Fig. 17.3). Nor do related species have complementary (allopatric) distributions. Nearly all of them are found from north to south, and may be found on offshore islands (even occasionally in Australia and/or Patagonian South America—lamprey, inanga). This means that diadromous species are, in theory, available to contribute to fish communities throughout the length of the country. Why is this so? Fairly clearly, because these species spend significant parts of their lives at sea, they can spread widely around the coasts, and invade rivers almost everywhere.

Notably, diadromous species are present in the rivers of South Westland (Fig 17.3), from which fish faunas were extirpated during Pleistocene glaciations (discussed above), and this can also be attributed to their ability to reinvade

from the sea, when Pleistocene ice sheets retreated and rivers started to flow again. So, having marine life stages has the clear advantages of allowing diadromous species to reinvade perturbed rivers once the effects of the perturbation have dissipated.

A corollary of this relates to the fact that diadromous species enter fish communities by migrating upstream from the sea. There will tend to be more species, and each species may be more abundant, at low elevations and near to the coast. Thus, both species diversity and abundance tend to decline with distance inland and altitude (Fig. 17.4) and this is due to hindrances to upstream migration and variation from species to species in both their ability and their instinct to penetrate long distances upstream. We find that fish like inanga, common smelt and giant bully are found only short distances upstream, whereas similar habitats well inland and at higher elevations may have none of these species. In contrast, longfin eels, shortjaw kokopu and koaro move long distances inland and are present from the lowlands to substantial elevations. Analyses of national and regional fish distribution records, indicate that altitude and distance inland are among the most important variables in distinguishing between fish communities (Hayes *et al.* 1989; Minns 1990; McDowall 1998; Joy *et al.* 2000; Jowett & Richardson 2003). These strong associations between community structure and distance from the coast have led to migratory behaviour and migratory passage being labelled as one of the most important factors influencing native fish distributions in New Zealand (Joy & Death 2000, 2001). Models have even been constructed for regional faunas that predict community composition with 70–80% accuracy, based on large-scale catchment variables like eco-region, altitude, and distance upstream from the coast (Joy & Death 2002).

Thus, dispersal ecology and migratory behaviour in migratory species have the potential to obscure patterns generated by history—historical determinants provide the background against which an analysis of ecological determinants can be undertaken. Understanding these principles and processes, combined with understanding the historical processes that have contributed to patterns of evolutionary divergence and speciation, are the keys to meaningful interpretation of distribution patterns and community structuring (which are, in a sense, different sides of the same coin).

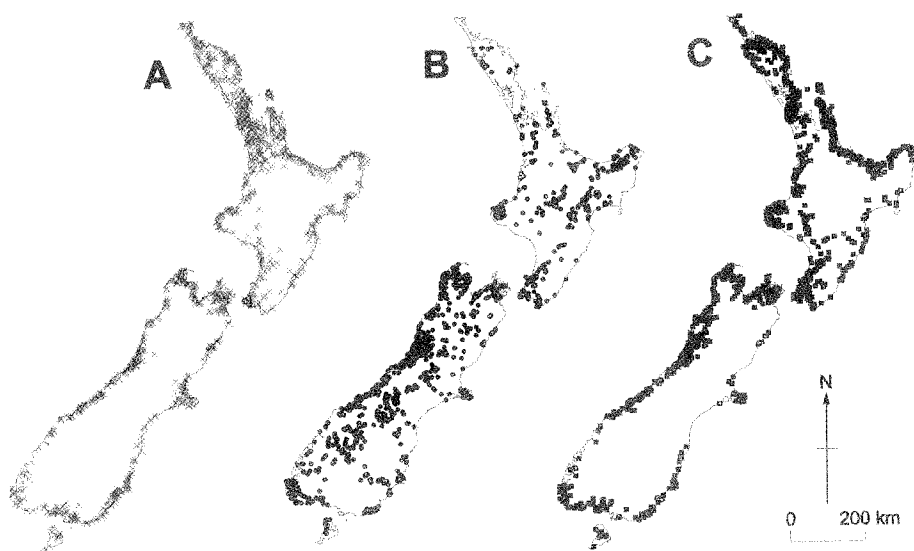


Figure 17.3 Distribution of migratory fish species using records from the NZ Freshwater Fish Database: (A) inanga; (B) koaro; and (C) redfin bully.

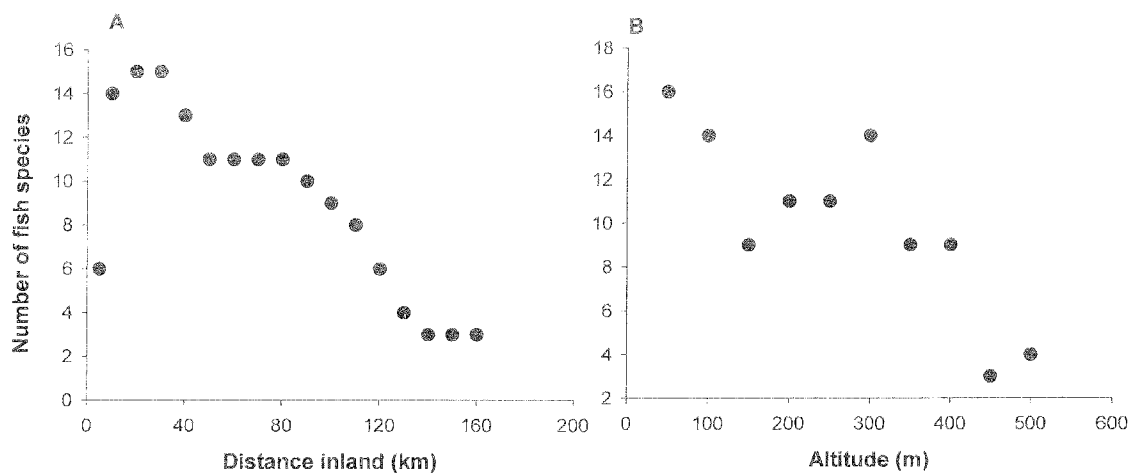


Figure 17.4 Relationship between fish species richness and variables associated with distance from the coast for (A) Westcoast of the South Island and (B) Taranaki (from McDowall 1998 and Joy *et al.* 2000).

Establishment of landlocked populations

Several of the typically diadromous species, discussed above (and see Table 17.1) can establish landlocked populations; usually this is in lakes whose outlets become blocked, though not always (Closs *et al.* 2003). Not much is known about the life histories of lake populations of diadromous species, but it seems likely that larval and juvenile life stages are found in the pelagial areas of the lakes, and that the adults are either littoral/benthic or are to be found in the lake tributaries (see Chapter 25). To some extent, the patterns of movements mirror what happens in diadromous populations, with the lake pelagial replacing the sea.

The presence of these lake populations has several implications for understanding fish distributions and communities. Some lakes are further inland and/or at higher elevations than their more typical diadromous conspecifics, and this simply means that, sometimes, there has been greater than usual inland penetration. Furthermore, some of the principles governing distribution, discussed above, may break down, though they are not negated. Why some species are facultatively diadromous and others are not (Table 17.1), is unknown but may be related to the suitability of larval rearing habitat (Closs *et al.* 2003).

Origins of some non-diadromous species

Phylogenetic studies are showing that some non-diadromous species share common ancestries with diadromous species, e.g., the dune lakes galaxias is regarded as a derivative of the inanga (McDowall 1990; Ling *et al.* 2001), and roundhead, flathead, dusky, Eldon's and Gollum galaxias are all probably non-migratory lineages derived from the diadromous koaro (Waters & Wallis 2001). Our mudfish species form a lineage that shares a

common ancestry with the Tasmanian mudfish, *N. cleaveri* (McDowall 1997b), and as the Tasmanian species is diadromous, New Zealand's mudfishes can be viewed as non-diadromous derivatives of the Tasmanian species—the same speciation process, but involving a broader geographical range.

Why migrate?

The prominence of diadromy amongst the New Zealand freshwater fishes prompts the question: Why do these fish undertake such migrations? Again, we are looking at a complex mix of processes. Part of the answer lies in the evolutionary history of our fauna. The ancestral freshwater fish colonists of the infant New Zealand land mass were almost certainly diadromous—they came to New Zealand by oceanic dispersal (McDowall 2002), and many present day inhabitants have inherited that trait. The anguillid eels are a particularly good example of this. The adults leave fresh water to spawn in the western subtropical Pacific Ocean. One of the three eel species now known from New Zealand (the Australian spotted eel) only began arriving here around 30 years ago, and this is an example of derivation /biogeography in action (McDowall *et al.* 1998). The navigation processes that allow eels to return to these spawning grounds, many 100s of km away from New Zealand, and which enable the progeny to return to New Zealand (and in two species also return to eastern Australia and Tasmania), are very poorly understood, but there must be a particularly strong navigation/guidance system that enables them to all synchronise the time that they reach the same general location so far away (McCleave & Jellyman 2002).

It is hard to fathom why these fish would undergo such apparently hazardous and energetically expensive journeys

between fresh and salt water if there were not strong selection pressure to do so. These traits must continue to be, in some sense, “beneficial”. Part of the cause may be that it enables eels to inhabit river systems that would otherwise have no fish in the “eel niche”. Part of the answer may lie in the fact that anguillid eels have, globally, been unable to establish landlocked populations—perhaps there is some physiological limitation related to survival of the eggs and sperm, or connected with eels’ peculiar leptocephalus larva. To maintain a presence on a land mass with such a “disturbed” history as New Zealand probably required great population resilience in the face of dramatic changes in the amount and location of available habitat, associated with sea level changes, mountain building and volcanism that has characterised New Zealand’s geological history (Fleming 1979; McDowall 1996b). Thus, the dominance and persistence of migratory life cycles that characterise the present day fish fauna may be a product of strong evolutionary, selection pressures for a life history strategy that facilitates population recovery after habitat disturbance. The fact that non-migratory species are absent from areas of relatively recent volcanism, glaciation and disturbance, whereas diadromous species are widespread (Main 1989; McDowall 1996a, 1998) is evidence that these migratory strategies continue to be important for supporting fish populations in some areas.

Examining the migratory behaviour of our indigenous freshwater fish fauna, in comparison to that of the exotics, also offers some insights into the evolution of these migratory strategies. The chinook salmon, stocked in South Island rivers in the early 1900s, is one of very few examples of successful establishment of a migratory fish, globally. The failure of this species in other parts of the world, and of sockeye salmon, Atlantic salmon and rainbow trout to establish migratory stocks in New Zealand, despite much effort (McDowall 1994), implies that they have been unable to adapt their movements and/or that some very explicit navigational cues and processes are involved in permitting their return from the sea.

Some aspects of the migratory behaviour of chinook salmon are in stark contrast with the migrations of native species. Chinook salmon show extreme “homing” behaviour that allows adult fish to return to their natal stream to spawn. This leads to patterns of strong, local genetic differentiation, whereby salmon in relatively neighbouring streams may have quite distinctive genetic make up. This may be reflected in morphological and behavioural differences, e.g. size of adult fish at maturity, the size and number of their eggs, timing of departure of juveniles to sea, etc., are all fine-tuned to local conditions and, especially, the distance of spawning grounds from the sea. These sorts of life-history strategies seem to have evolved in more stable continental conditions like those found in North America

where homing is very strong (Dittman & Quinn 1996). Elegant studies of chinook salmon indicate that they have undergone intense life history selection in New Zealand, resulting in considerable genetic differentiation between stocks in different New Zealand rivers (Quinn & Unwin 1993; Quinn *et al.* 1996; Unwin *et al.* 2003), even though they have been here for only about 100 years (25–30 generations).

In contrast, three New Zealand species of migratory galaxiids, when examined from widespread locations around New Zealand, exhibited very little genetic differentiation among populations (Allibone & Wallis 1993), implying that there is gene flow among the populations and thus a lack of homing. The absence of homing in these diadromous galaxiids may be adaptive because migratory behaviours that saw galaxiids returning to only their natal site would be disadvantageous in view of our long history of geological (and thus riverine) disturbance. Recent work on stream selection by whitebait returning from the sea suggests that they are choosing the rivers based on a variety of water characteristics, including temperature, pH and turbidity (McDowall & Eldon, 1980). In addition, however, return may be influenced by odours released by conspecifics living in upstream habitats (Baker & Montgomery 2001; Baker & Hicks 2003). Selection based on odour may be more important than catchment vegetation type (Rowe *et al.* 1992), even though vegetation may be an important factor influencing habitat selection at the local level (McDowall 1990; McDowall 1997c).

While differences between salmon and galaxias migratory strategies may underlie the different environments that these fish evolved in, the commonality is that they all migrate at some stage to sea and back to fresh water. Analysis of general patterns of migration support the idea that these migrations are adaptive because they enable the fish to exploit habitats where food production is high. The movement of early life stages to sea in cool and cold temperate latitudes in New Zealand may be because marine productivity is higher at sea than in fresh water (Gross *et al.* 1988). Analysis of the timing of migrations by New Zealand fishes indicates some may be synchronised with pulses of marine productivity whereas others may have more to do with the physiological constraints of movement between salt water and fresh water, but no one factor explains the timing and direction of movement (McDowall 1995).

The role of local factors

When we shift our focus, and begin to examine how fish species form communities in particular reaches of river and streams, we find that species’ distributions vary spatially and temporally with respect to local abiotic and

biotic conditions. Preferences for water quality and temperatures, flow variables, substrate characteristics, catchment and riparian vegetation, and so on, can all influence the distribution of fish (Richardson *et al.* 1994; Jowett & Richardson 1995; Allibone & Townsend 1997a; McDowall 1997c; Rowe *et al.* 1999). Extremes in local conditions like floods and droughts can also restrict the use of local habitats (Jowett & Richardson 1994; McIntosh 2000). These processes result in establishment of recognisable and variously consistent stream fish assemblages across New Zealand (McDowall 1990; Richardson & Jowett 2003). For example, we find that in slow-flowing, meandering streams with sandy substrates, inanga, common smelt and common bullies often are found together (though only at low elevations—none of them penetrates far inland). In contrast steeper, tumbling, bouldery streams in the bush tend to have koaro, shortjaw kokopu, longfin eels and redfin bullies. These species may move inland > 100 km and, being skilled climbers, even penetrate upstream past steep falls, though because redfins are less skilful upstream migrants, they tend to decline and disappear from fish communities at sites at higher elevations and further inland. So, there may be a “sorting process” that goes on among the species that relates to their upstream migration. The result is that there will be inland habitats suited to species with low penetration, but these species will not be present because they are unable to move that far inland.

McDowall (1996b) suggested that the mechanisms structuring communities dominated by diadromous fishes could be partly understood by viewing them as a continuum from accessible or “open” fish communities that are structured by the supply of colonists from outside at one extreme, to inaccessible, or “closed” communities that are structured by local biotic and abiotic influences at the other extreme. However, the extent to which fish communities at any particular reach are an outcome of migratory access or are influenced by local abiotic/biotic factors is still unclear. The identification of variables related to migratory passage in classifications of communities certainly supports the view that the supply of recruits from remote locations is important in determining structure. However, interactions between species, such as competition and predation and the physically disturbed nature of some New Zealand stream environments seem likely also to influence fish distribution at a local scale, but their role is not revealed by community analyses.

In Taranaki, fish community structure is strongly correlated with structures, like dams that are thought to hinder upstream movement of migratory species (Joy & Death 2001). This supports the idea that access is a key determinant of community structure and that, contingent upon access, communities will either be structured by

“open” or “closed” dynamics. Further investigations of climbing abilities (Baker 2003) and the drive to migrate inland will help to reveal what proportion of the patterns of community assemblage can be attributed to species-specific restrictions on inland penetration by barriers that limit the presence of particular species at a site. However, some caution is needed when interpreting these large-scale patterns as some abiotic conditions may also correlate with distance from the coast or altitude (e.g., temperature). If community composition correlates with distance inland/elevation, it is not always obvious whether the correlation is driven by upstream access or by the effects of temperature differences (or by substrate coarseness that relates to steep terrain and swifter flows). Alpine galaxias, for example, appear to be limited to cooler waters found at higher altitudes in South Island rivers (Dunn 2003).

Banded kokopu appear to exclude koaro from certain habitats in unmodified Stewart Island streams dominated by migratory species, so biotic interactions may still influence fish community structure in “open” systems (Chadderton & Allibone 2000). Furthermore, because fish habitat-use patterns reported from Stewart Island were quite different from those elsewhere, this study also indicates that contemporary studies of species’ distributions, and thus of species’ interactions, may also be coloured by the presence of trout or habitat modifications to mainland streams. These confounding issues hinder the unravelling of the nature of interactive process among the native species.

The apparent exclusion of non-migratory galaxiids from tributaries accessible to koaro in the Lake Mahinerangi/Waipori River system (McDowall & Allibone 1994; Allibone 1999), reveals a hierarchy of interspecific effects. Non-migratory galaxiid species were present only where koaro derived from the lake were unable to penetrate. Similarly, koaro were generally present only where introduced brown trout were excluded by barriers to upstream penetration. These patterns indicate that access can be a significant determinant of community structure, but also that strong local ecological interactions can lead to one group of species replacing another. In this situation biotic interactions are probably strongest in the “open” communities downstream, whereas “closed” upstream areas may form refuges for competitively inferior species. Interactions like these are important for both understanding the processes controlling distributions, as well as for the conservation of rare non-migratory galaxiids that are vulnerable to displacement by koaro and/or trout (Allibone 1999).

Additional interactions between non-migratory galaxiids and exotic brown trout in the South Island also illustrate the potential for both “open” and “closed” community dynamics to be important. In tributaries of

the Taieri River in Otago, the dominant factor determining the presence of non-migratory galaxiids at any site is the existence of downstream barriers that prevent invasion by brown trout (Townsend & Crowl 1991). When they have access, brown trout typically exclude non-migratory galaxiids by either competition for food and space (McIntosh *et al.* 1992; Edge *et al.* 1993) or predation (Crowl *et al.* 1992). In contrast, barriers to upstream trout access do not explain the distribution of non-migratory galaxias in headwaters of the Waimakariri River. The presence of galaxias there is determined by whether trout > 150 mm long were present, which is most likely related to stream size (McIntosh 2000). The size of the trout was important because only trout > 150 mm long were able to prey on all size classes of Canterbury galaxias.

Another important axis in determining local community structure is disturbance. In the Waimakariri River headwaters, flood disturbance creates a refuge for non-migratory galaxiids from trout predation. Although the galaxiids are considerably less abundant in flood-prone streams, trout appear to be excluded from these streams by the harsh physical conditions (McIntosh 2000). Disturbance can also create windows of opportunity for some fish species in much the same way as barriers to migratory species do. Rare Canterbury mudfish, for example, are probably prevented from occupying many otherwise suitable habitats by the presence of predatory fish such as trout and eels, but they survive in habitats subject to intermittent drying and low dissolved oxygen levels (Eldon 1979) through cutaneous oxygen uptake, a low metabolic rate (Meredith *et al.* 1982) and occupying damp locations (Cadwallader 1975; Fig. 17.5).

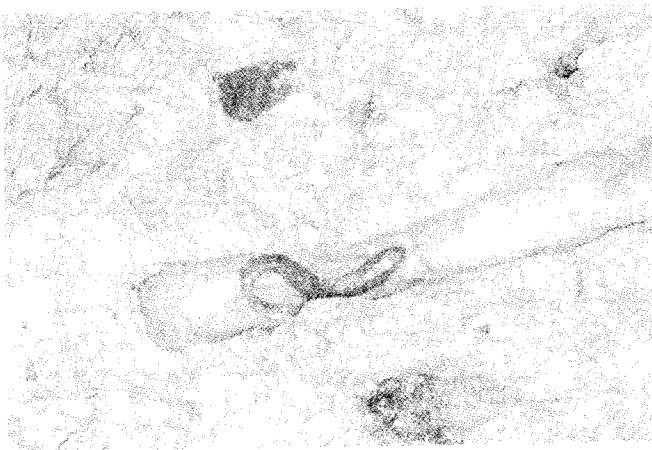


Figure 17.5 A Canterbury mudfish in a damp “burrow-like” refuge located in a clay stream bank in which it had survived a period of drought. Whether these fish construct (or alternatively how they find) these refuges is unknown.

Photo: Jeanne O'Brien

Alternative life history strategies?

It is clear that as in most ecological systems, control of community structure in rivers and streams represents an amalgam of mechanisms. In New Zealand, access to a reach for diadromous species is a primary determinant of community composition, but subsequent patterns can be modified by local interactions and behaviour. For example, recolonisation by diadromous species after a disturbance means that their populations are resilient, whereas resistance to disturbance is important for non-migratory galaxiids.

This highlights alternative types of life history strategy adopted by New Zealand freshwater fishes. Migratory species, which predominate in low elevation sites: often live longer and grow larger; have many small eggs and larvae; these hatch into an environment that lacks abundant suitable small food organisms; the fish larvae emigrate to sea where there is abundant food, though there is also marine predation; they have high dispersion rates (which may jeopardise return to freshwater habitats); undertake a return migration that has costs and exposes them to estuarine predators; and they need to search for suitable habitats for growth to maturity. Non-migratory species are often distributed further inland; they may be short-lived and not reach such large size (Fig 17.6); may have relatively larger eggs and fewer, larger larvae (Fig. 17.7); these larger larvae have an advantage in habitats with impoverished small food supplies; they do not migrate to sea; may be swept downstream but dispersion is not high and the only migrations are compensatory for this downstream dispersion; costs of migration are low and predation may not be heavy; they rear close to congenial habitats for growth to maturity. Although the characteristics in each of these two groups overlap quite broadly, there is evidence here of a series of tradeoffs that optimise each of the alternative strategies. For example, the energy costs and risks of migration and the need to relocate to freshwater habitats for maturation in migratory species are offset by the competitive advantage gained from capturing abundant marine food resources early in the life history, rapid growth. Migration may also reduce mortality caused by predation by the abundant adult fish in lowland locations, but does expose them to broad dispersion and predation at sea. For migratory species, the further inland the adult habitat is, the higher the energy expenditure on migration and the longer the fish must spend moving between habitats. Movement between habitats might be associated with the highest risk, and this is consistent with migratory species that move inland long distances (longfin eel, koaro, shortjaw kokopu) being amongst the most long-lived. This is consistent also with the non-migratory strategy being favoured amongst fish of inland locations.

A further implication of all the above is that for

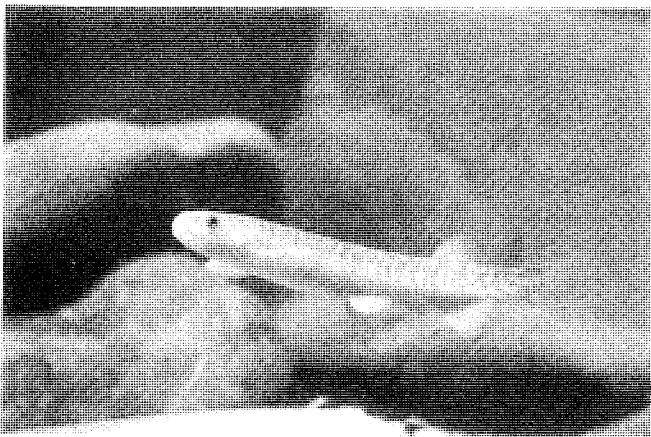


Figure 17.6 An adult Canterbury galaxias about 80 mm long.

Photo: Angus McIntosh

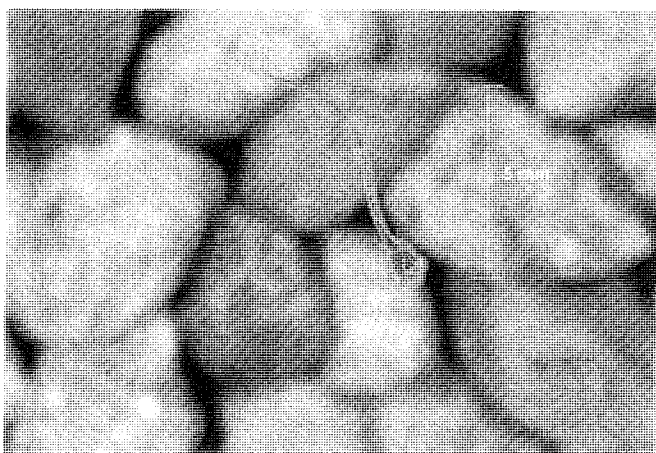


Figure 17.7 A Canterbury galaxias fry about two weeks old.

Photo: Angus McIntosh

migratory species, events long distances from their habitats in fresh water, as well as conditions in their chosen freshwater rearing habits, may be highly important in controlling survival, growth rates, abundance and other life history metrics. At its most simple, survival at sea may be a major determinant of how many inanga are present in the lower reaches of a stream, or how many lampreys penetrate our river systems. Alternatively, the abundance of non-migratory species at any site is likely to be determined largely by proximate environmental conditions—since they tend to spend their entire lives in one relatively restricted area.

Taking all the above points, together, it becomes apparent that there is a whole series of historical and ecological processes that are “sifting” the fish faunas and can be regarded as determinants of what is present and what is not, and which result in the fish communities in our rivers (Fig. 17.8). Each of these determinants must be taken into account when we seek to interpret what processes and causations are contributing to the communities we observe at any location.

POPULATION DYNAMICS

The question of what controls the abundance of fish in our streams and rivers is also complex, especially for migratory galaxiids. Many New Zealanders are interested in the answer because the size of their whitebait catches may depend on it. As outlined previously, the presence of adult migratory galaxiids is certainly determined by there being suitable migratory up- and downstream passage. But the adult fish also have freshwater habitat requirements, e.g. banded kokopu are restricted largely to forested

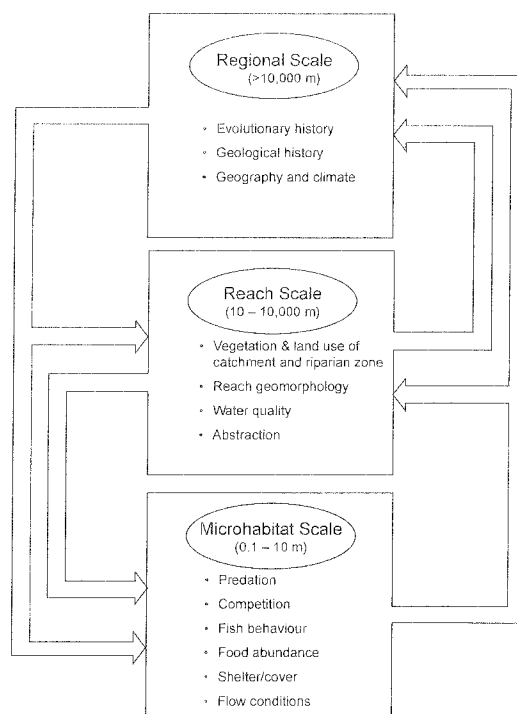


Figure 17.8 The interaction of factors affecting fish abundance and composition in New Zealand rivers and streams over a hierarchy of spatial scales. The diagram emphasises that although processes affecting fish communities are controlled at particular spatial scales, those processes can influence community development at other spatial scales. For example, climatic conditions affecting inanga whitebait growth at sea (a regional effect) will likely influence the abundance of inanga (and whitebait catches) in reaches of lowland streams. Similarly the presence of forest in a reach will affect the availability of wood in the stream that could be used for cover by banded kokopu at the microhabitat scale. Furthermore, processes operating at smaller scales also influence communities at larger scales. For example, a waterfall that restricts trout access to upstream habitat will prevent trout predation on non migratory galaxiids in reaches upstream. Likewise, odour from adult banded kokopu in a headwater stream may well attract banded kokopu whitebait to the catchment.

streams, shortjaw kokopu favour podocarp forest and avoid beech forest (McDowall 1997c), whereas inanga are dependent on low-elevation, gently flowing streams often associated with dense bank-side vegetation that they use for spawning. If access to these habitats is blocked, the species simply will not be present.

The extent to which the size of adult populations is controlled by features of habitats is unknown, but in large measure depends on the supply of juveniles, e.g. if insufficient whitebait arrive in the adult habitat to fill all potential places, then adult population size will be limited by the availability of recruits. In that case, factors affecting the migratory stages, like conditions along the migratory passage, the abundance of predators (like trout and whitebaiters), predation by marine organisms, conditions at sea affecting the availability of planktonic prey, and dispersion at sea by oceanic current systems, will all have critical influence.

Determining what controls the abundance of whitebait is even more difficult because it rests on understanding not only how conditions during migration and at sea affect numbers of whitebait, but also on whether the production of larvae is affected by the number of adults in the first place. Unravelling these issues will require some cunning science and much hard work. It seems intuitively likely that there is high mortality of the migratory species like whitebait and eels after the often huge migrations of juveniles back into river systems, but little is known about these aspects of freshwater fish population dynamics.

The question of what limits populations of non-migratory species has been investigated more thoroughly, but in many ways is just as complicated. A study by Alex Huryn (1996) in headwater streams of the Taieri River throws some light on the types of factors that limit non-migratory fish populations. He found that brown trout consumed virtually all the food being produced (including both aquatic and terrestrial invertebrates), which implies that the trout populations were limited by food availability (see also Chapter 13). This result mirrored an earlier study of trout in the Horokiwi Stream, which similarly calculated that trout consumed all of the available food (Allen 1951, and called the "Allen paradox, by Huryn). Moreover, models of trout foraging suggest that trout populations are controlled by the availability of invertebrate foods (Hayes *et al.* 2000). Huryn (1998) also investigated a population of the non-migratory Eldon's galaxias in a stream neighbouring the trout stream, and he found that the galaxias consumed less than 20% of the available macroinvertebrates. This means that either: prey are much better at avoiding predation by the native galaxias, the energy demands of the galaxiid population are much lower, or some other factor limits galaxias population size. Dunn (2003) showed that in inland Canterbury streams both

Canterbury and alpine galaxias recruitment into populations was closely related to flow and habitat conditions affecting survival of the small fry. This suggests that for non-migratory galaxiids the retention of small fry (which swim in the water column) in the adult habitat (Fig. 17. 7), and their survival, could be strongly affected by habitat conditions like low flows and floods.

Native fish seem particularly well suited to the fluctuations in flow found in New Zealand rivers, but their populations are still strongly impacted by floods and droughts. For example, Dunn (2003) observed fluctuations in the abundance of alpine galaxias over three orders of magnitude in a single year in intermittent reaches of streams in the headwaters of the Waimakariri River. In this case, some alpine galaxias cut off by receding flow appeared to tolerate cessation of surface flow by burrowing in the substrate, but populations as a whole still experienced high mortality. Furthermore, although some Canterbury mudfish individuals seem to find suitable refuge habitats to survive drought (Fig. 17.5), their populations still experience considerable mortality during such events (Eldon *et al.* 1978).

FISH BEHAVIOUR

The burrowing behaviour of alpine galaxias and migrations of diadromous galaxias described above are examples of the important links between fish behaviour and population dynamics. Two additional aspects of fish behaviour that have potentially important influences on fish populations are (1) their nocturnal activity and (2) their spawning behaviour. The reclusive habit of native fish resulted in early naturalists being "...very much surprised to find a country with such fine rivers, lakes and streams, but with no fish of any value to them" (Ayson 1910); there was comment that "every one of these mighty rivers, every one of the thousand creeks and streams that flow into them...were tenantless and profitless to the sportsman" (Spackman 1892).

Nocturnal behaviour

Two ecological factors may explain the nocturnal behaviour of native fish. Firstly, the risk of predation by birds may have forced them to curtail daylight activity by day (Glova *et al.* 1987). Elsewhere, avian predators like herons and kingfishers strongly influence the behaviour and abundance of fish (Steinmetz *et al.* 2003). This "predation by birds" argument seems unlikely to completely explain the New Zealand situation given the perception that there are relatively low levels of avian predation on native fish today. However, it is possible that the nocturnal behaviour of galaxiids may be a product of historically more diverse and abundant bird populations

(see Chapter 18). Another possible explanation for nocturnal native fish activity may be that they exploit food supplies that are night-active. For example, the foraging activity of koaro in some streams is well synchronised with the nocturnal activity of invertebrates (Glova & Sagar 1989), though this begs the question of why the invertebrates are nocturnal. Koaro have been observed drift-feeding by day in a troutless stream (Hayes 1996), raising the prospect that nocturnal behaviour of native fish might be a recently acquired behaviour connected with introduction of trout. Presence of trout in streams is known to induce nocturnal behaviour in many invertebrates (McIntosh & Townsend 1994; 1995a) and native fish may be driven to increasingly nocturnal foraging both by competitively dominant/predatory trout and/or trout-induced changes to invertebrate behaviour (McIntosh & Townsend 1995b; McDowall 2003).

This “trout-induced changes” explanation may apply to smaller fish like the non-migratory galaxias which frequently occupy riffle areas where visual distortion provided by water turbulence should make them less susceptible to predation by birds. However, trout-induced changes cannot explain all nocturnal behaviour amongst the natives because many native fish are nocturnal in the absence of trout, and are in fact superbly equipped to capture prey in total darkness. Most native freshwater fish have a well-developed cephalic laterosensory system of pores and canals, and rows of superficial neuromasts, which add to the sensory capacity provided by neuromasts in their lateral lines, and these enable fish to detect and capture prey by sensing prey vibrations in the water (Montgomery *et al.* 1995; McDowall 1997d; Fig. 17.7). Torrentfish lost their ability to capture prey at night when their lateral line sensory system was chemically disabled (Montgomery & Milton 1993) demonstrating the importance of the lateral line system in locating prey. These sensory systems are less effective in fast water, and especially suit fish which often occupy quiet back-water areas like banded kokopu and shortjaws that take terrestrial insects falling into the water (Main & Lyon 1988; McDowall *et al.* 1996).

It is interesting, in this context, that bullies tend in lake populations to lose the covering of the canals, so that the neuromasts become superficial. Galaxiids typically have additional neuromasts that form an “accessory lateral line” along the back, which may be associated with feeding on terrestrial prey (Main & Winterbourn 1987; McDowall 1997d; Fig. 17.9).

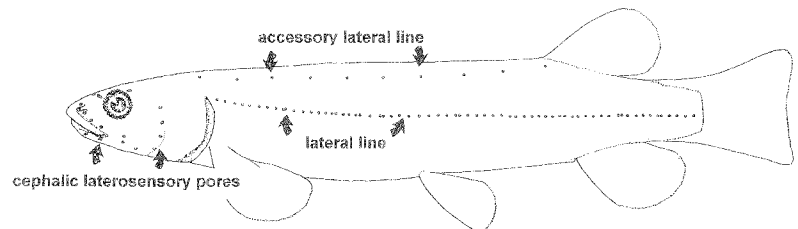


Figure 17.9 The location of the lateral line, accessory lateral line and cephalic laterosensory pores on a koaro (note the dots indicating the neuromasts associated with these sensory structures are shown larger than life) (From McDowall 1997d).

Only through intensive studies are further aspects of native fish behaviour likely to be unravelled. For example, detailed observations and the use of radio tracking allowed Bruno David and co-workers to follow the behaviour of individual giant kokopu in Otago streams (David & Closs 2001). They showed giant kokopu to be creatures of habit, consistently foraging in the same pools, and returning to the same hiding places when not active (David & Closs 2003; Figure 17.10). Giant kokopu were consistently active only during the night in winter, but sometimes became day-active during summer (David & Closs 2003). By day they even appeared to form a linear size-related

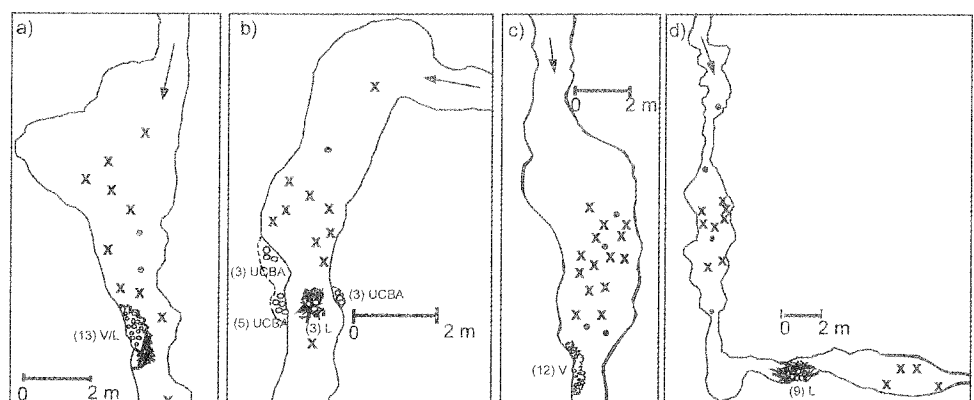


Figure 17.10 Locations of four different individual giant kokopu occupying four pools in an Otago stream during summer indicating the direction of flow (arrows); locations of night foraging positions (crosses), day foraging positions (black circles) and day cover positions (open circles); and the different types of cover used (V = aquatic vegetation, L = log, V/L = vegetation/log combination and UCBA = undercut bank) (modified from David & Closs 2003).

dominance hierarchy, similar to that observed in many Northern Hemisphere drift-feeding salmonids (David & Stoffels 2003). They observed some interesting changes in behaviour during increased stream flows. Depending on the magnitude of a flood event, some giant kokopu shifted to nearby refuge habitat, some occasionally moved considerable distances in the stream, but others may not have shifted at all. When the flood abated, sometimes fish returned to their original location, and sometimes they occupied a new location (David & Closs 2003). These movements seemed to be associated with exploring the habitat and may even have been associated with reproduction or finding mates (David & Closs 2003). This is the first comprehensive study of behavioural interactions involving native fish and highlights how much there is to learn about their habits and behaviours.

Spawning behaviour

Little is known about the breeding behaviour of some native species and intensive studies have also been required to unravel the mysteries of native fish spawning. Although the downstream movement of larvae of migratory galaxiids, or the appearance in backwaters of non-migratory galaxiids, respectively, give a clues to the timing of spawning (Ots & Eldon 1975; McDowall & Suren 1995), pinpointing the habitats where galaxiids spawn, and the cues associated with spawning behaviour, have proved difficult. Several migratory galaxiids have been shown to deposit their eggs on sections of stream bank that are not normally immersed in water. Inanga do this in river estuaries, when high spring tides flood the bank vegetation (McDowall 1990); koaro, banded and shortjaw kokopu deposit their eggs amongst stream bank litter and gravels during elevated stream flows, around the level of bankful flow, 0.3–4 m laterally from the wetted stream edge at base flow and 0.15–0.7 m above the level of baseflow (Mitchell & Penlington 1982; O'Connor & Koehn 1998; Allibone & Caskey 2000; Charteris *et al.* 2003; Fig. 17.11). For shortjaw kokopu, numbers of eggs found have tended to be considerably less than estimated fecundity, suggesting that many eggs are washed downstream or lost to predation or, alternatively that the females use multiple spawning sites. In all of these galaxiids the eggs develop out of water, dehydration being prevented by either the retention of humid atmosphere among the estuarine vegetation (inanga), or by spawning taking place in streams with heavy riparian plant cover (other galaxiids; Fig. 17.11). Larvae appear to hatch and be transported by high flows, which inundate the spawning sites around one month after spawning (Charteris *et al.* 2003).

Painstaking searches have frequently been required to find spawning sites (e.g., Allibone & Townsend 1997b; Allibone & Caskey 2000). There is considerable variation

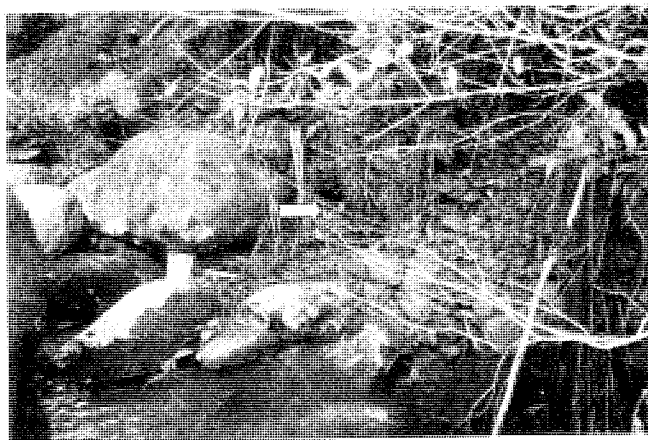


Figure 17.11 The location (indicated by arrow) of a shortjaw kokopu spawning site amongst leaf litter on the banks of a Taranaki stream. Photo: Sjaan Charteris

among species in spawning behaviour and habitat use, and the habitats involved may be different from those occupied at other times, indicating small-scale reproductive migrations and the use of specialised habitats. Non-migratory galaxiid species such as Canterbury flathead, and Eldon's galaxias spawn under or between coarse substrate gravels in the water that provides cover for eggs (Allibone & Townsend 1997b; Moore *et al.* 1999), although some species, such as dwarf and roundhead galaxias, appear to scatter sticky eggs (Hopkins 1979; Allibone & Townsend 1997b), and in dusky galaxias they are found amongst rootwads of riparian plants. An interesting, and as yet unexplained dichotomy in galaxiid reproduction is that migratory (diadromous) species typically spawn in autumn and winter, whereas non-migratory species spawn in spring. These differences may be related to selection that optimises timing of larval hatch in relation to food availability at sea (for diadromous species) and in fresh water (for non-migratory). It could also have something to do with the need for recruits of non-migratory species to have long enough to reach a minimum size that allows them to overwinter in often very cold upland streams in their first year of life.

Reproduction in bullies is quite different. Males establish territories, most often beneath large rocks, and entice females into the territory. The females deposit their eggs, usually, on the underside of the rock, and these are fertilised by the male, which continues to guard the nest while the eggs develop. Successful males may mate with several females beneath the same rock; and similarly females may produce repeated batches of eggs and spawn several to many times in one spring and summer (McDowall 1990; McDowall & Eldon 1997). Spawning of both diadromous and nondiadromous bullies takes place in spring and early summer. But the differences in egg size

discussed above for galaxiids apply also to bullies, with diadromous species having much smaller, but relatively greater numbers of eggs than non-diadromous species.

CONSERVATION

The fact that over 50% of the native fish species currently recognised are considered threatened with extinction (Table 17.1) and more rare species are yet to be described, makes the job of arresting declines in their populations urgent and understanding of the mechanisms structuring their populations vital. Although large scale catchment variables appear to be useful in predicting community structure, identifying mechanisms that cause those patterns in fish community structure is important for management. Diadromy is certainly important, but understanding how it interacts with other potential influences like local habitat conditions, catchment land use and species interactions is essential if we are to understand factors that contribute to community assembly, and to enable their effective management. We need to know the extent to which distributions are determined by the migratory drive of species, local habitat conditions, and whether there is some type of sorting process that involves interactions between species.

The solutions to problems facing native fish populations will vary depending on the mechanism controlling fish distributions. When upstream access controls community structure, factors remote from the site are obviously implicated, and unoccupied habitats cannot be assumed to be unsuitable (McDowall 1993). In sea-migratory species, freshwater populations are being replenished from the sea, year after year. Therefore local perturbations in fresh water, such as a fish kill caused by pollution, may be compensated for by recolonisations into that habitat from downstream, once the perturbation is no longer influencing survival and abundance. This difference may have been crucial in the apparently variable impacts of brown trout on native non-migratory galaxiids versus migratory ones, for example. The continual supply of migratory galaxiid recruits from other locations may offset the local effects of trout predation, although trout may still impact populations of migratory galaxiids like shortjaw kokopu and koaro (Goodman 2002; Crowl *et al.* 1992). The widespread distribution of inanga larvae at sea has probably facilitated restoration by allowing repopulation of restored habitat. In many cases inanga populations have been increased by improving bankside vegetation suitable for spawning (Taylor 2002).

One corollary of the migratory life history strategy, however, is that it is crucially important that migratory species are able efficiently to locate upstream habitats that are suitable for their rearing. Human impacts on river

channels pose particular problems for migratory fish. These impacts largely relate to upstream passage, and include: river mouth closures preventing migration from the sea; changes in flood frequency and size; increased sedimentation as a consequence of changed catchment management and vegetation that increases turbidity inhibiting feeding and movement (Rowe & Dean 1998); elevated temperatures that follow reduction in flow volumes and reduced water depth associated with abstraction; construction of dams and weirs that both interfere directly with migration (McDowall 1992, 1999) as well as changing habitat from swift flows and coarse substrates, to slow or nil flow velocities and fine substrates; wetland drainage; and also impacts of exotic species. Furthermore, it is likely that the "sources and sinks" mechanism (Pulliam 1988) may result in high production of recruits from favourable habitats being spread widely, but thinly, across diverse habitats, some of which may be unfavourable. Any situation where the supply of recruits is remote from the destination of the migrants makes migratory populations vulnerable, and this includes fisheries exploitation, since during migrations fish are highly concentrated both spatially and temporally, e.g. when whitebait are migrating upstream or adult eels migrating downstream. In the latter case there is clear evidence for exploitation severely altering the age structure of populations, with large females being particularly vulnerable to exploitation and disappearing from the population (McCleave & Jellyman 2003). The long term population impacts of such harvesting are obvious.

These situations do not arise in the same way in non-migratory species. If some event such as pollution, or drought, or whatever, extirpates them from a river system, the species is not easily restored to that system. The non-migratory galaxiids evolved substantially through the loss of migratory behaviour (Wallis & Waters 2001) and their populations are subdivided. This means they are prone to local extinctions and the damaging effects of human impacts will persist for a long time. Thus, where various salmonid species are invasive there is high potential for adverse effects and non-migratory galaxiid populations have been severely fragmented (Crowl *et al.* 1992; McDowall 2003; Simon & Townsend 2003). Thus, it is not surprising that the non-migratory galaxiids are the most seriously endangered group of fish in New Zealand.

The characteristics of galaxiid movement and spawning outlined above indicate that understanding the behaviour of native fish is essential for their management and conservation. Many of the galaxiids seem not only well adapted to the variable flows characteristic of New Zealand rivers (see Chapter 10), but this may also require flow variability to facilitate important aspects of their behaviour. For example, fluctuations in discharge appear essential for banded and shortjaw kokopu spawning and provide

opportunities for giant kokopu to move between habitats that are restricted at base flow. Use of forested riparian areas, which maintain high humidity around eggs, could also explain the restriction of banded and shortjaw kokopu to forested streams. Continued harvesting, especially of indigenous forest, may have serious implications for the future welfare of species that favour streams with forested riparian margins. In addition, the implications of reforestation with exotic tree species, for stream ecologies and the welfare of indigenous fishes, are poorly understood (but see Chapter 33).

The nocturnal behaviour of giant kokopu and the terrestrial spawning of banded and shortjaw kokopu appear to be quite different from the habits of most fish elsewhere, adding to the uniqueness of these New Zealand galaxiids. The behaviour of newly described non-migratory galaxiid species with even higher threat classifications (Table 17.1) is much less well known. Many of these rare non-migratory galaxiids are found in the eastern and southern South Island where increased frequency and severity of droughts and high demand for water abstraction put the remaining populations at risk. Current information indicates that hydrological disruptions, especially during fry development, can impact on non-migratory galaxiid populations (Dunn 2003), abstraction can facilitate invasion by non-native species and limit spawning (Allibone 1999), and the availability of suitable spawning habitat could limit the distribution of some species (Moore *et al.* 1999). There are widespread serious impacts from water abstraction and reduced flows, and virtually nothing is known about the dimensions of these effects on threatened native fish. Moreover, continued agricultural development, urbanisation and mining pose considerable threats to native fish but their influences on native fish communities are also poorly understood (but see Chapters 34-36).

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Chapter 16

Stream invertebrates

Mike Winterbourn

INTRODUCTION

Streams and rivers are inhabited by a diverse fauna of invertebrates. They provide important links in running water food webs by feeding on living and decomposing plant material as well as other animals, and being eaten in turn by higher predators notably fish, but also birds. The adults of aquatic insects may also be eaten by predatory terrestrial invertebrates such as spiders, thereby providing a food subsidy to terrestrial ecosystems. Some provide food for humans (e.g., crayfish and mussels) and others are the hosts of parasites, or may be parasites themselves. The larvae of freshwater mussels attach to the skin and gills of some freshwater fish, many larval water mites are ectoparasites of larval insects, and the first-stage larvae of gordian worms develop inside caddisflies and mayflies. From an ecosystem perspective, invertebrates play important roles in organic matter processing and energy flow. In addition to their direct feeding activities they remove particles from the water column, reduce the size of organic matter, and act as habitat modifiers. Freshwater crayfish, for example, disperse fine sediments over the stream bed, thereby making habitat available for algae and other smaller invertebrates.

Because running water invertebrates are such a diverse group, and demonstrate a multitude of habitat and environmental relationships, they are used widely for monitoring and evaluating water quality, and more broadly, "stream health". Freshwater crayfish are being farmed in several commercial aquaculture ventures, and many of the trout flies tied by anglers mimic aquatic insects. Other species, notably molluscs, have invaded aquatic ecosystems outside their natural range, often assisted unwittingly by humans, and become nuisance species. For example, the European zebra mussel (*Dreissena polymorpha*) out-competes local bivalves in many North American rivers and has driven some of them locally

extinct. Lastly, stream invertebrates are of scientific interest in their own right, and several groups including the chironomid midges and hyriid mussels provide evidence for biogeographic relationships between New Zealand, Australia and South America (Walker *et al.* 1999; Boothroyd 2000).

Stream invertebrates were described from New Zealand for the first time in the early 1800s. Since then numerous taxonomists have contributed to the documentation of our fauna and continue to do so. Many taxonomic papers contain keys to groups of invertebrates that include running water species, but for other taxa, especially lower invertebrates (e.g., sponges, hydras, flatworms and leeches), no comprehensive guides are available. The most complete identification works are to Crustacea (Chapman and Lewis 1976), Mollusca (Winterbourn 1973) and Insecta (Winterbourn *et al.* 2000), but they are outdated to varying degrees. The recent overview article by Winterbourn (2000) includes references to many of the more important papers, which describe or revise New Zealand stream invertebrates and use their currently accepted names. The main groups comprising our fauna are listed in Table 16.1.

Special features of our stream invertebrate fauna are the high degree of endemism at the specific and generic levels, a strong Gondwanan element, and a paucity of introduced species. Many insect families that dominate stream faunas in the northern hemisphere are not found in New Zealand, and others are represented by single, widespread species. Examples are the alderfly *Archichauliodes diversus* (Corydalidae), the mayfly *Coloburiscus humeralis* (Coloburiscidae), and the stonefly *Austroperla cyrene* (Austroperlidae). Furthermore, *Potamopyrgus antipodarum*, is the only hydrobiid snail described from surface freshwaters (several estuarine and subterranean, and spring-dwelling species are also known), where it can be

Table 16.1 Summary of the composition of the New Zealand stream invertebrate fauna.

Major taxa	Common names	Principal families
Insecta		
Ephemeroptera	Mayflies	Leptophlebiidae, Coloburiscidae
Plecoptera	Stoneflies	Gripopterygidae, Notonemouridae
Trichoptera	Caddisflies	Hydrobiosidae, Conoesucidae
Hemiptera	Bugs	Corixidae
Coleoptera	Beetles	Elmidae
Diptera	True flies	Chironomidae, Tipulidae
Mecoptera	Scorpionflies	Nannochoristidae
Megaloptera	Dobsonflies	Corydalidae
Odonata	Dragonflies	Corduliidae
Crustacea		
Decapoda	Crayfish, shrimps	Parastacidae, Atyidae
Amphipoda	Scuds	Paracalliopidae
Mollusca		
Gastropoda	Snails	Hydrobiidae
Bivalvia	Mussels, clams	Hyriidae, Sphaeriidae
Annelida		
Oligochaeta	"Earthworms"	Tubificidae, Naididae
Polychaeta	Bristle worms	Nereididae
Hirudinea	Leeches	Glossiphoniidae
Others		
Porifera	Sponges	Spongillidae
Hydrozoa	Hydras	Hydridae
Turbellaria	Flatworms	Dugesiidae
Nemertea	Proboscis worms	Prosorhochmidae
Nematoda	Roundworms	?
Nematomorpha	Gordian worms	Gordiidae
Bryozoa	Bryozoans	?

the numerically dominant macroinvertebrate. Also notable is the trend towards terrestriality in gripopterygid stonefly larvae (see below), the presence of a scorpionfly (Mecoptera: Nannochoristidae) with a stream-dwelling larva, and a single freshwater polychaete known also from Fiji (Glasby 1999).

This chapter describes some of the main features of invertebrate groups found in New Zealand rivers and streams with particular emphasis on the more abundant ones. Taxonomic and ecological diversity, life history characteristics, feeding biology, distribution and habitat are discussed. Community composition is described briefly, but community dynamics and the roles of invertebrates in running water ecosystems are included in Chapter 13.

ECOLOGY OF INVERTEBRATE GROUPS

Ephemeroptera (mayflies)

The mayflies are a primitive group of insects with large, delicate fore-wings that are held unfolded above the abdomen at rest. The eggs of most species are laid in water

and develop into aquatic nymphs, which moult an indeterminate number of times before attaining their full size. The short-lived, winged adult is terrestrial and is preceded by a winged subimaginal stage that lives for about a day before the final moult. Because the non-feeding adults are so short-lived, reproductive maturity is attained in the final larval instar, which stops feeding, and if female, becomes filled with eggs. The functions of the adults are to mate, lay eggs and disperse, whereas the larva is primarily a feeding and developmental stage.

The New Zealand mayfly fauna comprises over 40 species in 19 genera and 8 families. All our mayflies have phyletic affinities with other fragments of Gondwana, and all genera are endemic. The largest family is the Leptophlebiidae, which includes 30 species in 12 genera, including the ubiquitous *Deleatidium*. In terms of nymphal morphology and behaviour leptophlebiids can be described as crawlers, that feed by grazing hard surfaces and by sweeping up loose detritus and silt with their brush-like maxillae. Leptophlebiids occupy a wide range of habitats in forest and grassland streams. However, more species are known from the North than South Island

(21 v 6). Towns (1987) found that the 24 species on Great Barrier Island had a variety of flow and habitat preferences, with distinctive assemblages of species being found on wet rock faces, cobbles, wood and leaves, and among bryophytes. In general, *Zephlebia* species inhabit slow-flowing streams, whereas *Deleatidium* species live in faster flowing water and on unstable substrata (Towns and Peters 1996). The latter are very widely distributed, and often the numerically dominant insects in rivers with cobble or gravel beds, including those prone to flooding. Although nymphal populations are reduced in abundance by floods, oviposition, egg-hatching and immigration from refuge areas such as minor braids and river margins result in the rapid re-establishment of populations.

In contrast to leptophlebiids, the nymphs of *Nesameletus*, *Rallidens*, *Siphlaenigma* and *Oniscigaster* are swimmers, although they are also grazers and collectors of fine detritus. Nymphs of the five known *Nesameletus* species tend to be found in smallish streams and trickles where flow rates are moderate to slow (Hitchings and Staniczek 2003), and in Canterbury *N. ornatus* and *N. austrinus* generally occupy open and forested streams, respectively, with little overlap in their distributions (Winterbourn 2003). Nymphs of *Siphlaenigma* are usually found on terrestrial vegetation hanging in the water, whereas *Oniscigaster* is most common in pools where it may lie partly buried.

The deposit-feeding nymphs of *Ichthybotus* burrow in fine stream sediments with their legs and tusk-like mandibles, whereas the heavily-built nymphs of *Coloburiscus* are filter feeders that trap particles with their setose legs and mouthparts (Fig. 16.1). The nymphs of *C. humeralis* offer weak resistance to flood flows, and most live on the undersides of stones in moderately stable, stony streams. In some forest streams where nymphal densities

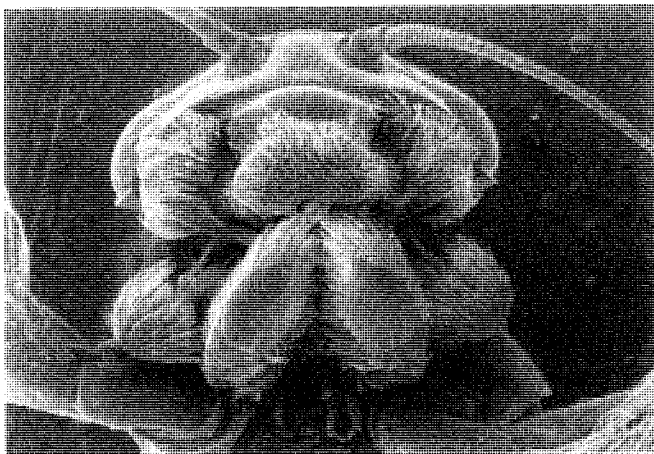


Figure 16.1 Mouthparts of the filter-feeding mayfly *Coloburiscus humeralis*. The hairy palps are used to trap particles and transfer them to the mouth.

Photo: Rachel McNabb

are very high they are likely to be significant processors of fine detritus. Lastly, the flattened, swimming nymphs of *Ameletopsis* are the only predatory mayflies in the New Zealand fauna. They inhabit stony streams in many parts of the country and have a distinctive, bright yellow subimago and adult.

Life histories

Knowledge of the life histories of New Zealand mayflies has been drawn together by Scarsbrook (2000). Most life cycles appear to be of about one-year's duration, although some species can have a rapidly developing summer generation in addition to a longer, slower one (Fig. 16.2). The poorly synchronised nature of many life cycles is reflected in the long flight and egg-laying periods of females. The presence of very small nymphs in many months may also indicate the occurrence of resting eggs that hatch after varying periods in the water. Studies on the life cycles of *Oniscigaster*, *Coloburiscus* and *Nesameletus* suggest that developmental synchrony increases with altitude and from north to south. Consistent with this scenario, flight periods are longest in the north where some species emerge year-round, and more restricted further south.

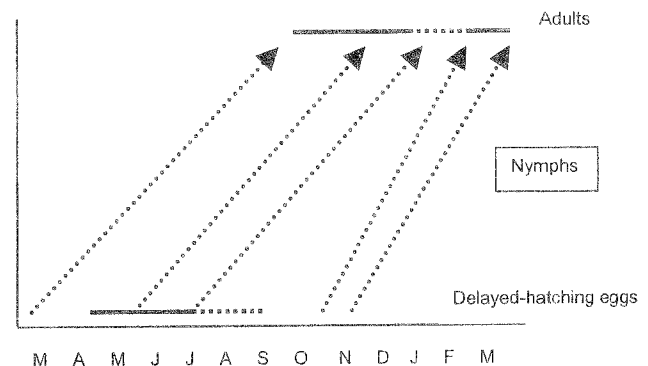


Figure 16.2 Diagrammatic representation of the life cycle of a *Deleatidium* species showing flight period, presence of delayed-hatching eggs, and winter and summer generations of nymphs with incomplete separation.

Plecoptera (stoneflies)

Most stoneflies are small insects whose two pairs of wings are folded over the abdomen in a tubular manner at rest. Adults are often found under stones close to the nymphal habitat, and when disturbed they usually run, rather than fly away. In general, adults are terrestrial, whereas most nymphs and eggs are aquatic. However, the nymphs of some alpine stoneflies in New Zealand live out of water and the eggs of *Megaleptoperla diminuta* are not

laid until the developing embryo is almost a fully formed nymph. The nymphs and adults of many species are brown or black, although the large *Stenoperla* species are predominantly green, (or sometimes bluish or yellow) and *Austroperla cyrene* has distinct yellow bands on its otherwise black legs. Hydrogen cyanide is produced by *A. cyrene* so this “wasp-like” pattern may warn potential predators that it is distasteful (McLellan 1996).

Four families of Plecoptera (Eustheniidae, Austroperlidae, Gripopterygidae and Notonemouridae) containing 20+ genera and over 100 described species are known from New Zealand. The large family Gripopterygidae (Fig. 16.3) is a classic Gondwanan group present also in Australia and South America. Stoneflies are often considered to be “cold-adapted” with many species restricted to cool mountain streams and forest where water quality, including dissolved oxygen concentration, is high. The geographic distribution of the order within New Zealand river systems appears to be limited largely by temperature, and Quinn and Hickey (1990) found that plecopteran biomass was minimal at river sites having mean annual temperatures $>13^{\circ}\text{C}$. This requirement for low temperatures probably restrict some species to forested localities, especially in the north of the country and at low altitudes.

Most New Zealand stonefly nymphs are detritivores that ingest particles of decaying plant material and their associated microflora. Many also ingest algae, and the large *Stenoperla* (and possibly *Megaleptoperla*) species are carnivorous, at least in their later instars. Common prey of *Stenoperla* are small chironomid larvae and mayfly nymphs, which are often the most abundant invertebrates available on and under stones.

Habitat

Although stonefly nymphs are typically aquatic, a feature of the New Zealand Gripopterygidae is a trend



Figure 16.3 The larva of the gripopterygid stonefly *Zelandoperla*.
Photo: Angus McIntosh

towards terrestrialism. This differs in extent among genera and species, from being semi-terrestrial alongside stony streams in the final instar of *Acroperla trivacuata* to living a fully terrestrial nymphal life in several wingless alpine species. Some *Taraperla* species are fully aquatic only in their earliest instars, whereas the nymphs of *Nesoperla fulvescens* occur under stones away from the active channel in dry river beds and are submerged only during spates and floods. One species, *Taraperla howesi* that is usually found in the water by day, has been seen feeding on lichen-encrusted boulders beside the stream at night. This suggests that by extending their habitat onto land, gripopterygid nymphs in general may increase their feeding opportunities. However, in doing so they expose themselves to predation by terrestrial invertebrates, including both web-building and free-living spiders, which feed on them in the riparian zone (Collier *et al.* 2002).

Fully aquatic stonefly nymphs are most commonly found in stony streams among the gravels and cobbles of riffles. Typically, they occur on the undersides of stones, although some *Zelandoperla* species (Fig. 16.3) are also found on bedrock, the faces of boulders, and mossy waterfalls. *Halticoperla viridans* lives in films of seepage water, whereas wood is an important habitat of *Austroperla cyrene*. The nymphs of various species forage on dead leaves and twigs, and several species of *Spaniocercoides* inhabit the hyporheic zone, at least for parts of their larval lives (McLellan 1991). Although typically found in clear, unpolluted streams, some stonefly species are tolerant of very low pH and high concentrations of metals such as iron and aluminium. Thus, at least seven species occur in brown-water streams at pH 4.5 or less, and *Spaniocercoides philpotti* inhabits streams severely impacted by acid mine drainage (Harding 2000).

Life histories

The life histories of New Zealand stoneflies were reviewed by Scarsbrook (2000). Most species probably have a 1-year life cycle, although those of the larger species of *Stenoperla* and *Austroperla* may be 2 or 3-years. Some gripopterygid species have moderately well synchronised growth, with nymphal size increasing through the winter and emergence in spring or early summer. However, other species have poorly synchronised life cycles with adults emerging in many months, even at the same locality. Collier and Smith (2000) found that adults of three gripopterygid species lived 8-19 days in humid conditions at 10°C , and that sooty mould fungi, pollen and diatoms were their principal foods (Smith and Collier 2000). Terrestrial feeding, especially by females, may be a prerequisite for the completion of reproductive development and mating.

Trichoptera (caddisflies)

The Trichoptera is a highly diverse order both taxonomically and ecologically. The New Zealand fauna comprises 48 genera in 16 families, and over 230 described species. A high proportion of these live in streams and rivers, but lakes, ponds and wetlands are also inhabited by some species. The success of the group has been attributed in part to the production of silk by larvae, which has enabled a diverse array of cases, retreats, and food-capture nets to evolve.

The eggs, larvae and pupae of caddisflies are usually aquatic, although in a few species one or other of them can occur on land. *Triplectides cephalotes* is ovoviviparous and first stage larvae rather than eggs are released into the water by the adult female. The somewhat dome-like, stony pupal chambers of hydrobiosid and hydropsychid species can be seen attached to the undersides of large stones in many streams, whereas numerous other species pupate in the final larval case, which is usually foreshortened and glued to a rock. Caddisfly adults are moth-like, terrestrial insects, but have hairy not scaly wings. Adults live from days to months, but their mouthparts are reduced, so although they drink they probably do not feed. Many are strong fliers, and the primary dispersal stage in the life cycle. However, Collier and Smith (1998) found that the main area of activity for adult caddisflies was within 30 m of the stream edge.

Most of our caddisfly species probably have 1-year life cycles. Adult flight periods of some species decline in length from north to south and with increasing altitude, and also differ among families (Fig. 16.4). Thus, many hydrobiosids have flight periods that can encompass the entire year at some sites, whereas conoesucids are on the wing for only a few months.

The Hydropsychidae, Hydrobiosidae and Conoesucidae are the commonest caddisfly families in New Zealand running waters, and among the best known. Some features of their life histories and ecology are outlined below.

Hydropsychidae

The larvae of Hydropsychidae occupy retreats attached to stones or bedrock and construct a silken capture net a centimetre or two wide close to its entrance. Detritus, algal cells and small invertebrates that are caught in the nets are ingested by the larvae, sometimes along with the net, which is then replaced. The silk threads are secreted by modified salivary glands, which open on the tip of the labium. Nets may be strung

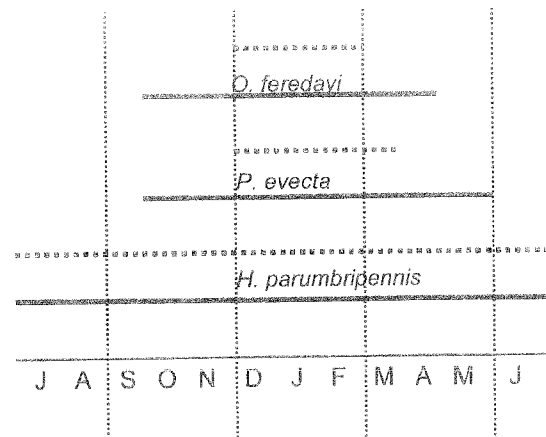


Figure 16.4 Flight periods of 3 caddisfly species at Cass, inland Canterbury (600 m a.s.l.) (dashed lines) and in the Waitakere Ranges, Auckland (45 m a.s.l.) (solid lines). Vertical lines delimit seasons. Data from Shearer (1996), Ward *et al.* (1996) and Winterbourn and Crowe (2001).

between stones, or attached to “posts” stuck to the substratum by the larva (Fig. 16.5). In the Kaniere River two *Aoteapsyche* species can occur together at densities of up to 10,000/m², but they avoid direct competition by having different microdistributions. Thus, most *A. colonica* larvae attach to the undersides of stones, whereas *A. rarururu* inhabits their tops and sides (Harding 1997). Hydropsychid larvae can also make sounds by scraping a projection on the foreleg across a series of ridges on the underside of the head. This stridulatory mechanism may be used by larvae to communicate with each other, although this has yet to be demonstrated, convincingly.

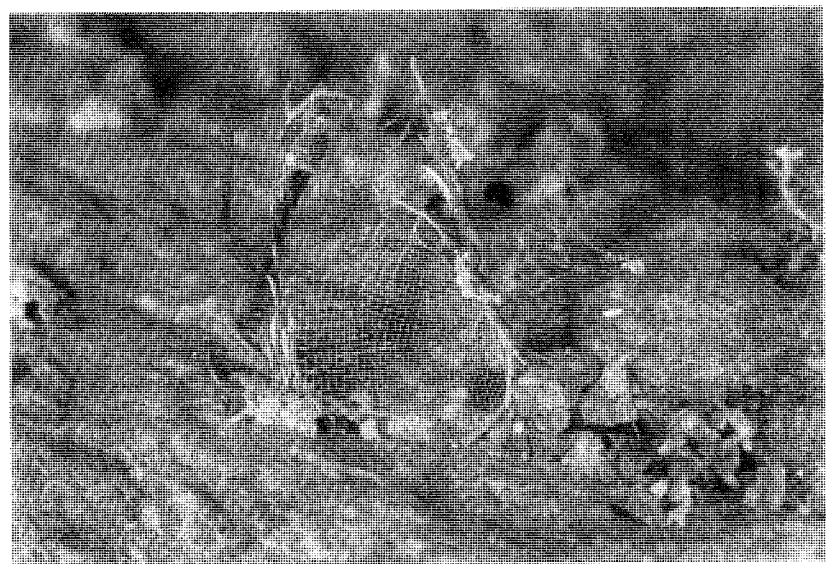


Figure 16.5 Net of the hydropsychid caddis *Aoteapsyche rarururu*.

Photo: Simon Pollard

Hydrobiosidae

The Hydrobiosidae is the most diverse family of Trichoptera in New Zealand with over 100 species in nine genera. Larvae lack cases in all instars (Fig. 16.6) and have chelate forelimbs, which they use to capture prey. Early instar larvae are detritivorous but older ones are carnivorous with chironomid and sandfly larvae being common prey. Habitats of hydrobiosid larvae are typically the undersurfaces of stony substrata in fast-flowing rivers and streams, although some *Tiphobiosis* species commonly inhabit seepages. Tolerance to organic enrichment differs among species, with the more tolerant ones such as *Hydrobiosis parumbripennis* often being common in moderately enriched lowland streams draining farmland. In contrast, species such as *H. styx*, a hydrobiosid endemic to Banks Peninsula, are restricted to forested headwater streams where water quality is high (Harding 2003).

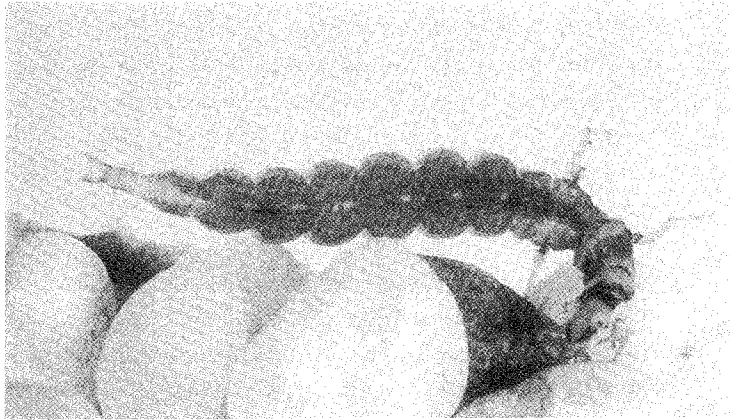


Figure 16.6 The free-living (caseless) larva of *Hydrobiosis clavigera*.

Photo: Phil Pointing

Conoesucidae

The Conoesucidae is a family of small caddisflies whose larvae have portable cases consisting of a matrix of silk to which sand grains are often attached. As caddisfly larvae grow larger the anterior end of the case is extended and broadened to fit the enclosed individual, while a posterior section is nipped off. Furthermore, if the case is lost a new one is built from scratch as described for *Pycnocentroides aeris* by Prestidge (1977). Larvae of conoesucid species appear to be primarily collector-browsers that remove and ingest algae and fine detrital particles from the surfaces of stones and plants. Some, like the horny-cased *Olinga feredayi* may also eat decaying leaves, and *Pycnocentria funerea* eats wood (Collier and Halliday 2000).

Species of *Pycnocentria*, *Pycnocentroides* and *Olinga* are some of our most widely distributed and abundant caddisflies, all three being found on stones and submerged vegetation, in hill-country and lowland streams. *O. feredayi*

also inhabits the hyporheic zone of streams down to at least 45 cm. In an intensive study of conoesucid life cycles in four small streams within the Waimakariri basin, Shearer (1997) found *O. feredayi*, *Pycnocentria evecta* and *Pycnocentroides aureola* in the two stable streams with large macrophyte beds, but only *O. feredayi* in the two streams that lacked macrophytes and were more prone to bed movement and scouring. Annual production of the three species combined was also much higher in the stable streams (Fig. 16.7), although production of *O. feredayi* was greatest where it occurred alone.

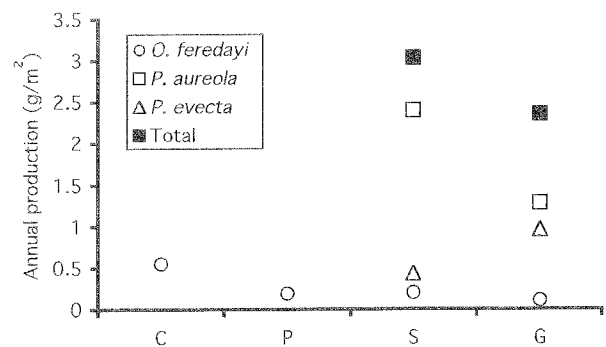


Figure 16.7 Annual production of 3 cased conoesucid caddisflies (*Olinga feredayi*, *Pycnocentroides aureola*, *Pycnocentria evecta*) and their combined production where applicable in 4 inland Canterbury streams. C=Coach, P=Porters Pass, S=Slip, G=Grasmere streams.

Other insects

Eighteen families of Diptera have representatives in New Zealand freshwaters, including lakes, ponds, rivers and streams. They include many cosmopolitan genera, although most species are endemic to this country. Some dipteran groups (e.g., the chironomid subfamilies Podonominae, Aphroteniinae and Diamesinae) show strong southern affinities, especially with Chile and Patagonia. As in many other countries, the most abundant dipteran family in New Zealand streams and rivers is the Chironomidae. Most chironomid midge larvae feed on fine detritus and diatoms, although many Tanypodinae are predators of smaller chironomids, at least in the final instar. Larvae of *Xenochironomus canterburyensis* develop in the mantle cavity of the mussel *Hyridella menziesi*, whereas those of a species of *Eukiefferiella* attach to the bodies of *Deleatidium* nymphs where they also pupate (Fig. 16.8). Another species of *Eukiefferiella* is the most abundant

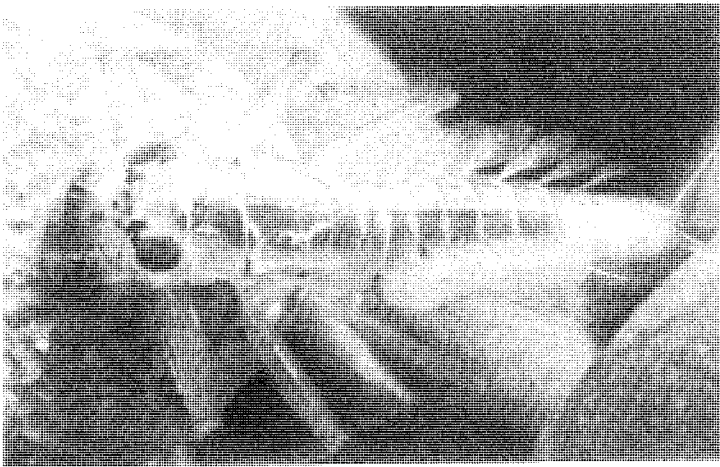


Figure 16.8 The tube-dwelling commensal chironomid *Eukiefferiella* attached to the abdomen and gills of a *Deleatidium* mayfly nymph. Tube length 2 mm.

Photo: Angus McIntosh

invertebrate living near the source of glacial rivers at temperatures $<2^{\circ}\text{C}$ (Milner *et al.* 2001), and yet another is one of the few insects to inhabit acid mine drainages with pH less than 3. The life cycles of New Zealand midges are not well known but the longevity of the larval stage probably ranges from several weeks to the best part of a year, depending on species and temperature. Boothroyd (1987, 1999) found that most of the 15 species he studied in a North Island stream had poorly synchronised, non-seasonal life cycles, with some emerging throughout the year.

The Simuliidae (blackflies or sandflies) are probably best known in New Zealand for their nuisance value. In fact, only the females of two species bite people regularly: *Austrosimulium australense*, which occurs throughout the country and *A. unguatum*, which inhabits the South Island and Stewart Island (Crosby 2000). Both males and females may also feed on nectar from flowers, although little is known about adult diets or habitats in New Zealand. Blackfly larvae often occur in dense aggregations on the upper surfaces of stones in fast-flowing streams and rivers. There they filter bacteria and fine organic particles from the water using elaborate head fans. The larvae of another dipteran family, the Blephariceridae, or net-wing midges, sometimes compete with blackfly larvae for space on stones in swift-flowing mountain streams, but unlike the simuliids they are algal grazers not filter feeders. Blepharicerids attach to the substratum with six ventral, suctorial disks whose mode of action resembles that of a piston pump (Frutiger 2002). Commensal chironomid larvae also live on the ventral surfaces of some larvae, their bodies wrapped sinuously around the suckers.

The other diverse dipteran family encountered frequently in our streams is the Tipulidae (craneflies). Although many larvae are prominent members of the

stream benthos, few can be identified to species or even genus. Larvae occur amongst stony substrata (various species of Eriopterini and Hexatomini), in seepages (*Zelandotipula* spp.), on bryophytes (*Limonia hudsoni*), and in rotting logs (*L. nigrescens*). They include detrital feeders, predators, wood-eaters and omnivores like *Aphrophila zelandica*. *A. zelandica* has a well-synchronised, univoltine life cycle with a short emergence period in early summer (Winterbourn 1996), and is the only aquatic cranefly whose life cycle has been studied in this country.

Other insects sometimes found in New Zealand running waters are species of Odonata (dragonflies and damselflies), Hemiptera (waterboatmen and backswimmers), Mecoptera (a scorpionfly), Coleoptera (beetles) and Megaloptera (dobsonflies). Tiny, black and brown hydraenid beetles can be found in forest streams throughout the country,

whereas the larvae of elmids are often abundant in more open rivers and streams, especially where fine sediment is present. Adult elmids are also aquatic and appear in the same habitats as their larvae for a few weeks in summer. Two other beetle families characteristic of forest streams are the Scirtidae and Hydrophilidae. The larvae of some scirtid species leave the water at night to graze on encrusting riparian plants, whereas others rasp and graze the surfaces of submerged leaves and wood. Their adults are not aquatic, and their biology is poorly known. The large family Hydrophilidae includes many terrestrial species as well as others with aquatic larvae and adults. Some have been found in decaying logs in streams, whereas others inhabit lowland streams and ditches as well as thermal springs and their outflow channels.

New Zealand's only megalopteran, the corydalid *Archichauliodes diversus* is one of our largest stream insects, and one of the most widely distributed. Recent research indicates that North Island populations are genetically more similar to each other than to South Island insects, although reported differences are small (Hogg *et al.* 2002). Larvae are voracious predators in stony streams and rivers where they feed on a wide range of smaller invertebrates, especially chironomids and mayflies. Larval development may take up to 3 years before pupation, which occurs beneath stones alongside the stream channel. Adult life-span is short (6–10 days, Scarsbrook 2000) and anecdotal evidence indicates that emergence of adults may be highly synchronised.

Mollusca

New Zealand has a relatively small described fauna of freshwater molluscs (Winterbourn 1973), although a large number of small hydrobiid species have been discovered, recently, in springs and seepages (see Chapter 30).

The hydrobiid *Potamopyrgus antipodarum* is widely distributed and often abundant throughout the country in lakes, ditches and ponds as well as running waters. This highly successful species exhibits substantial variability in shell size, shape and ornamentation, is ovoviviparous (young develop in a brood pouch), and many populations consist solely of parthenogenetic females (Fig. 16.9). Populations at any one locality may include sexually reproducing individuals and numerous genetically distinct, parthenogenetic clones, apparently derived from them (Jokela *et al.* 2003). Sexual reproduction in these snails may be maintained, in part, by locally adapted trematode parasites, while some clones may be adapted to particular habitats. *P. antipodarum* is also a highly successful invader of freshwaters in Europe, Australia, and North America. It reproduces and disperses rapidly in new environments in the absence of parasites and possibly predators, sometimes building up local populations with densities of over 100,000/m².

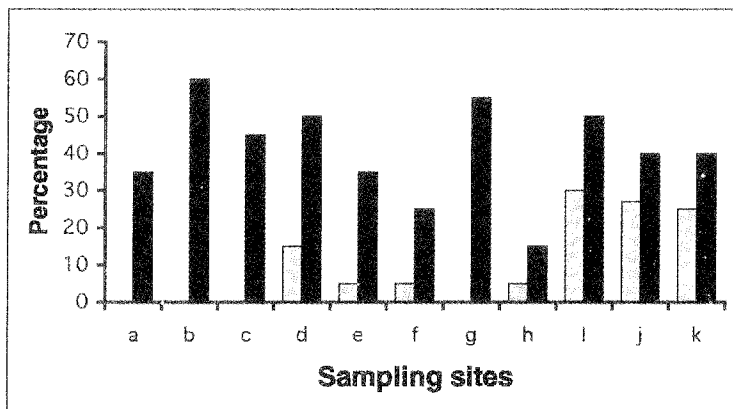


Figure 16.9 Frequencies of *Potamopyrgus antipodarum* males (black bars) and snails with spiny shells (stippled bars) at 11 sites 100 m apart along a Waikato stream, showing the large variation in shell form and sex ratio that can occur locally, in this common species (after Haase 2003).

Other features of the New Zealand molluscan fauna are the presence of the limpet-like *Latia neritoides*, which occurs in stony streams and rivers in the North Island and fluoresces when disturbed, and the introduced *Physella acuta* (formerly *Physa*), which is widely distributed and often abundant, especially in the vicinity of towns and cities. The recent finding of a population of *Melanoides tuberculata*, a tropical thiarid snail, in thermal waters near Taupo (Duggan 2002) is of concern, since it is a strong competitor whose introduction has led to reductions in native snail populations elsewhere. At least some of the lymnaeid species found here can act as intermediate hosts of the liver fluke *Fasciola hepatica*, and a second fluke whose cercariae cause “swimmers itch”. New Zealand’s

largest bivalves are the widely distributed mussel *Hyridella menziesi* and the poorly known *Cucumerunio websteri*. However, the most abundant bivalve in freshwaters is the fingernail clam *Musculium novaezelandiae* (formerly *Sphaerium*), which can disperse when young by attaching to the legs of flying insects.

All our gastropod species are grazers, which ingest algae, fine detritus and microbes that develop on stones, dead leaves and wood. In contrast, bivalves are primarily filter feeders that remove algae, bacteria and other fine particles from the surrounding water. Because of its sensitivity to chemical contaminants *H. menziesi* has been used to measure the effects of geothermal and agricultural contaminants in the Waikato River (Hickey 2000). Most gastropods probably have one or more generations per year, and most lay eggs in jelly-like egg masses or capsules on vegetation, stones and other hard surfaces. Initial development of young sphaeriid bivalves is in a brood chamber at the base of the gills, whereas *H. menziesi* has glochidia larvae that attach to the skin of native fish, including eels, bullies and koaro (Walker *et al.* 2001). Estimates of mussel longevity obtained by counting shell annual rings indicate that some individuals can live for at least 33 years.

Annelida

Oligochaetes, polychaetes and leeches are all found in New Zealand streams and rivers. The sole polychaete species *Namanereis tiriteae* (formerly *Namalycastis*) is known only from stony streams near Palmerston North and east of the Manawatu Gorge. Nothing has been reported of its biology, but its lack of eyes and body pigment suggest it maybe primarily a hyporheic species (Glasby 1999). Similarly, little is known about the biology and ecology of our small leech fauna, which includes several species of Glossiphoniidae. They are most common in low-gradient streams and lakes where some (e.g., *Placobdelloides maorica*) are known to feed on the blood of ducks, and others (*Alboglossiphonia* spp.) are assumed to eat snails.

The Oligochaeta is the most diverse and best known of our segmented worm groups, and is represented by seven families, Lumbricidae, Lumbriculidae, Haplotaxidae, Tubificidae, Naididae, Phreodrilidae and Enchytraeidae. They are primarily, but not exclusively, inhabitants of soft sediments, which also provide their main source of food. Oligochaete communities in streams can be quite diverse with 15 species recorded by Marshall (1974) from the Leeston Drain in Canterbury, and 13 from Devils Creek near Reefton, West Coast, by Cowie (1980). In both these environments Naididae and Tubificidae were the most diverse families with 4-6 species identified. Enchytraeidae

have rarely been reported in New Zealand stream studies, but it is likely that some records of the naidid *Breviatría multiprostatus* (= *Telmatodrilus multiprostatus*) are actually of an enchytraeid species (Pinder and Brinkhurst, 2000).

Oligochaetes are most commonly associated with organically polluted streams and rivers where they may attain enormous densities. In New Zealand the most abundant species in polluted waters are usually a tubificid *Limnodrilus hoffmeisteri* and a lumbricid *Lumbriculus variegatus*. Living specimens of the latter can be recognised easily because they swim in an undulating manner like an eel, whereas tubificids typically coil up like a spring when disturbed. *L. variegatus* reproduces asexually, by breaking in half and regenerating the missing segments, whereas many others deposit eggs in roughly spherical cocoons.

Two other interesting species are the lumbricid "earthworm" *Eiseniella tetraedra*, and the naidid *Chaetogaster limnaei*. The former is an introduced species, now widespread in New Zealand, and recognizable by the square cross section of the posterior third or so of its body. *C. limnaei* is a very small worm that lives in the mantle cavities of snails, including *Potamopyrgus antipodarum*, *Gyraulus corinna* and *Lymnaea stagnalis*. It feeds on diatoms and other particles, and has been known to ingest juveniles of *P. antipodarum* that may have been being expelled from the snail's brood pouch.

Crustacea

The Crustacea include the largest stream invertebrates (crayfish) and also many of the smallest ones. The latter include copepods, ostracods and cladocerans, many of which inhabit the hyporheic zone and are discussed in Chapter 32. In addition to crayfish, the larger crustaceans that inhabit running waters are species of Amphipoda (Fenwick 2001), Isopoda, mysid shrimps (in the lower reaches of some rivers) and an atyid shrimp. A tanaid shrimp is known from the Tarawera River (Chapman and Lewis 1976), and a hymenosomatid crab, *Amarinus lacustris* occurs in parts of the Waikato River, as well as some coastal lakes in the North Island.

Three species of *Austridotea* (Isopoda: Idoteidae) are found in New Zealand rivers and streams (Chadderton *et al.* 2003). None occur north of Banks Peninsula and one (*A. benhami*) is restricted to streams on and close to the Otago Peninsula. All three species may be exposed to mildly saline water at times, but the commonest species, *A. lacustris* is found most frequently in freshwater just above the estuarine zone. The slater-like *Austridotea* species are bottom dwellers, and on Stewart Island *A. lacustris* can be abundant in leaf packs, on submerged logs and on the undersides of stones. They are all omnivores, although

primarily detritus feeders. *A. annectens* has a well synchronised one-year life cycle with young being released in spring.

Our most widely distributed freshwater amphipod is *Paracalliope fluviatilis*, which is abundant in many weedy and urban streams. Several white and purple *Paraleptamphopus* species that are difficult to identify, occur commonly among stones and plant debris in springs and forested streams, including acidic, tannin-stained waters.

New Zealand's only freshwater shrimp *Paratya curvirostris* (Fig. 16.10) is widely distributed in lowland rivers and streams. Larvae develop in estuaries and the young shrimps migrate upstream developing first into males and then females. This protandric mode of development results in all the large shrimps being females. Shrimps are often found at the margins of rivers amongst overhanging grasses, or in submerged weed beds where they eat decaying plant material. The mysid shrimps (*Tenagomysis* spp.), which are most common in estuarine water feed on detritus, algae and small crustaceans such as copepods and cladocerans.



Figure 16.10 The atyid shrimp *Paratya curvirostris*. Body length 25 mm.

Photo: Alan Carpenter

Lastly, two parastacid crayfish (koura), *Paranephrops planifrons* and *P. zealandicus* inhabit New Zealand streams, and lakes. They have disjunct geographical distributions with *P. planifrons* in the North Island and on the west of the South Island, and *P. zealandicus* on Stewart Island and in the east of the South Island. However, few crayfish have been found in Canterbury and they are absent from the Canterbury hill-country and mountains, except at Lake Georgina where *P. zealandicus* was introduced. Unlike freshwater crayfish in the northern hemisphere (Astacoidea), both New Zealand species are unusually acid-tolerant and commonly inhabit brown water streams where the pH is about 4. They are nocturnally active, and during the day live in weed beds or beneath stones, logs and overhanging banks. Both species ingest large amounts of plant detritus, as well as invertebrates, including snails, insects, flatworms, and crustaceans. Water temperature has a strong influence on crayfish longevity, which is greater

in the south than the north, and also on the attainment of sexual maturity, which can take 6 years in Otago but only 2 years in the Waikato (Whitmore and Huryn 1999; Parkyn *et al.* 2002). Crayfish have been described as “ecosystem engineers”, because their activity helps redistribute fine sediments and may keep stone surfaces free of deposits. Their bioturbatory activity may also stimulate the growth of epilithic algae, which are eaten by smaller grazing species, but conversely reduce the amount of habitat available in the form of coarse detritus (Usio and Townsend 2002).

STREAM COMMUNITIES

Stream communities in New Zealand are highly diverse at the species level and vary greatly in terms of the numbers and identities of species found together. A wide range of environmental and biotic factors influence the composition of communities and are discussed in Chapter 13. Here I make some very general comments on the nature of invertebrate communities characteristic of contrasting stream environments within the country. As a rule, stream faunas include a small number of abundant and common species and a large number of rare ones. Accordingly, it is important to appreciate that the observed species richness in streams is influenced in part by sampling effort, such that more (rare) species will be found in larger samples. Also, abundance at a particular location can vary considerably from year to year due to the effects of a variety of factors including disturbance frequency and intensity, and colonization processes. However, Scarsbrook (2002) found that persistence in terms of species occurrence was promoted by constancy of habitat conditions, and that over a 9-year period the communities at 26 river sites appeared to fluctuate in abundance around a relatively stable state.

Similarly, Winterbourn (1997) found that persistence of species was high in three South Island, mountain beech forest streams over a 5-year period, whereas abundance varied considerably.

Because abundances of individual taxa can fluctuate markedly at a particular site, it is unrealistic to define communities explicitly in terms of absolute or relative abundance of taxa. Nevertheless, it is possible to predict what major taxonomic groups of invertebrates should dominate communities in different kinds of streams and rivers. To this end, Figs 16.11 and 16.12 show the relative abundance of major taxa in a range of contrasting running waters in New Zealand. Their locations and settings are indicated in Table 16.2.

Things to note are the strong numerical dominance of Ephemeroptera and/or Plecoptera in mountain streams and in streams and rivers in high rainfall areas like the West Coast where discharge is highly variable and floods are common. Where it flows over the Canterbury Plains the Rakaia River provides a particularly harsh physical environment for invertebrates and in major braids the mayfly *Deleatidium* may be the only abundant insect. In contrast, Trichoptera attain greater significance in hill-country streams with lower rainfall, and in low-gradient streams fed by lakes, which moderate flow fluctuations. Springbrooks also provide more “benign” conditions where abundant chironomids, snails (mainly *Potamopyrgus*), and crustaceans, especially amphipods (the main contributor to the “others” category in Slip Spring; Fig. 16.11), typically are found. Chironomid larvae also tend to dominate rivers like Charming Creek, a West Coast forest stream contaminated by acid, coalmine drainage.

The examples included in Fig. 16.12 are lowland streams with varying degrees of habitat and water quality degradation. Kaniwhaniwha Stream in the Waikato is in

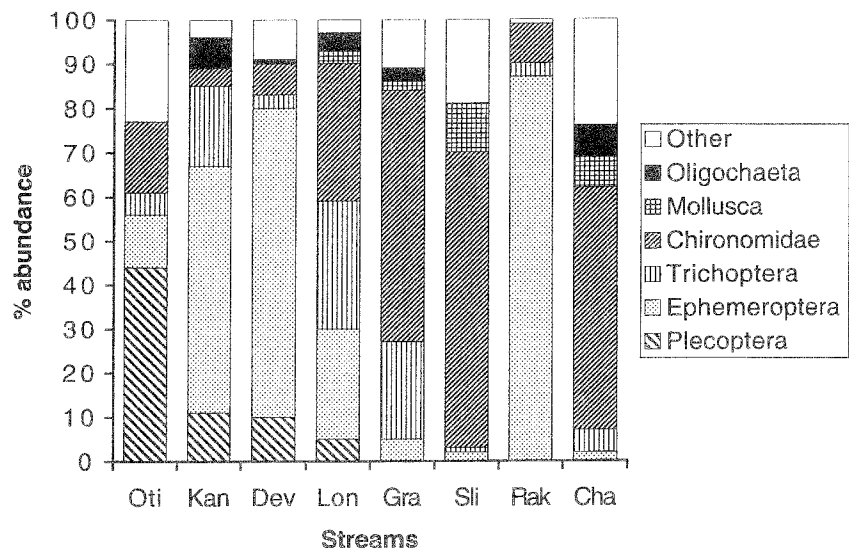


Figure 16.11 Relative abundance of major invertebrate groups reported in 8 contrasting stony streams and rivers. See Table 2 for details.

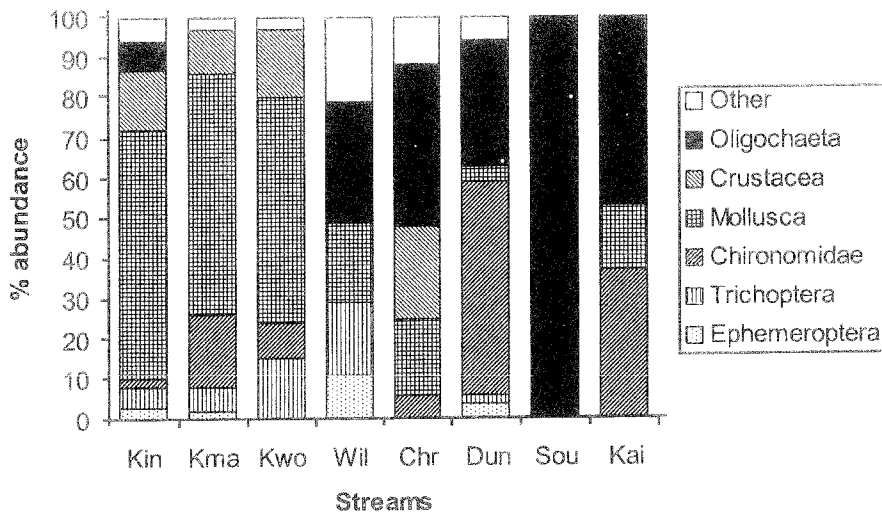


Figure 16.12 Relative abundance of major invertebrate groups reported in 8 lowland, urban and organically polluted streams. See Table 2 for details.

relatively good condition, and the three histograms show that invertebrate assemblages are very similar on sand and gravel, macrophytes, and wood, with snails (*Potamopyrgus*) and crustaceans (amphipods) dominant. Wildon Drain a small, stony stream in rural Canterbury has a more diverse fauna dominated by Trichoptera, Mollusca and Oligochaeta, but also includes Ephemeroptera, an indicator of reasonably good water and habitat quality. In contrast, urbanization has negatively affected many stream faunas through reducing the quantity and quality of water, increasing sediment inputs, and by divorcing streams from their surroundings. Urban stream faunas can differ considerably depending on locality-specific factors (Suren

2000), as indicated in Fig. 16.11, which shows the average faunal composition of urban streams in Christchurch and Dunedin. Although stream faunas in both cities are dominated by Chironomidae, Mollusca, Oligochaeta and Crustacea their relative abundances are very different.

The last two histograms show the invertebrate faunas in 1970 of two streams then polluted by organic wastes from a freezing works (South Branch) and a fellmongery (Kaiapoi River). The benthos of both streams was dominated by oligochaetes (almost exclusively in the South Branch), but the presence of chironomids and particularly molluscs in the Kaiapoi indicate a lower level of contamination.

Table 16.2 Identities, locations and settings of the streams included in Figures 16.11 and 16.12.

Name	Location	Setting/condition	Data source	
(a) Figure 16.11				
Oti	Otira River tributary	Southern Alps	Alpine zone	Harding 1994
Kan	Lake Kaniere tributary	Westland	Rain forest	Harding 1994
Dev	Devils Creek	Westland	Rough pasture	Cowie 1980
Lon	Longslip Creek	Lindis Pass	Tussock grassland	Harding 1994
Gra	Grasmere Stream	Inland Canterbury	Lake-fed; grassland	Death 1991
Sli	Slip Stream	Porters Pass	Stable springbrook	Death 1991
Rak	Rakaia River	Canterbury Plains	Large shingle river	Digby 1999
Cha	Charming Creek	North Westland	Mine-polluted; forest	P.A.Ryan (unpubl.)
(b) Figure 16.12				
Kin	Kaniwhaniwha River	Waikato	Pasture: stony bed	Collier et al. 1998
Kma	"	"	Pasture: macrophytes	"
Kwo	"	"	Pasture: wood	"
Wil	Wildon Drain	Canterbury Plains	Pasture: stony bed	Harding 1994
Chr	Christchurch streams	Christchurch	Urban, suburban	Suren 2000
Dun	Dunedin streams	Dunedin	Urban, suburban	"
Sou	South Branch	North Canterbury	Polluted (1970)	Winterbourn et al. 1970
Kai	Kaiapoi River	North Canterbury	Polluted (1970)	"

CONCLUDING COMMENTS

Although this chapter has focussed principally on the three major macroinvertebrate phyla found in New Zealand's running waters, representatives of at least six other animal phyla (excluding those represented only as meiofauna) are also present. Furthermore, I have not commented on the water mites (Acari) a highly diverse, but poorly understood group of Arthropoda with at least 93 species in New Zealand (Cook 1983). Notes on each of these groups are given by Winterbourn (2000) whose summary indicates that much remains to be learnt about them. In particular, our freshwater sponge, hydrozoan, nematode and bryozoan faunas are very poorly understood, and even though flatworms are common in many streams and rivers their reproductive biology and ecology are essentially unstudied. Much fundamental research remains to be done in order to better understand the systematics, physiology, and ecology of New Zealand running water invertebrates, and to provide a sound basis for their conservation and management.

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Chapter 15

Periphyton

Barry Biggs and Cathy Kilroy

INTRODUCTION

Periphyton is the “slime” coating the surfaces of stone or wood in streams, rivers, lakes and other waterways—indeed, this community is likely to be found on the surface of most stable objects in freshwaters. It can be difficult to see with the naked eye, except that it gives submerged objects a brown or brown-green colouring. However, in some places periphyton can proliferate and form thick slimy layers or streaming filaments in running water, and clouds or mats of green or brown filaments in pools.

Periphyton is the most common descriptor for this community in freshwater ecology. However, *benthic algae* and *phytobenthos* refer to essentially the same thing, since the community is composed mainly of algae, including Cyanobacteria (previously called “blue-green algae”). Periphyton may comprise filamentous green algae (Chlorophyta), red algae (Rhodophyta) and yellow-green algae (Ochrophyta: Tribophyceae, Chrysophyceae), along with diatoms (single-celled algae; Ochrophyta: Bacillariophyceae), Cyanobacteria, a range of other algae including single-celled and colonial chlorophytes, and non-algal material including bacteria and fungi. In New Zealand freshwaters, diatoms are widespread and diverse and certainly the best-studied component of periphyton communities.

Periphyton has several important roles in freshwater ecosystems. The algal species making up the community are primary producers (that is, they photosynthesise). Diatoms store energy as oil droplets and this makes them a rich food source for many aquatic invertebrates. In flowing water, the periphyton community quickly assimilates dissolved nutrients and other material and thus can purify stream water (Meiring *et al.* 1994).

Large growths of periphyton can accumulate in streams and along lake shores during warm, settled summer weather. Some common instream values may be com-

promised by periphyton proliferations (Table 15.1), though assessment of the effects can be very subjective. The effects of these growths on water quality and ecosystem degradation have not been extensively documented, and a number of cause-effect assumptions in this linkage need further research.

In this chapter, we first discuss our current understanding of the factors that influence periphyton biomass accrual and loss, including factors that may lead to nuisance proliferations in some rivers. This is followed by brief descriptions of the periphyton communities typically found in different freshwater habitats in New Zealand. Methods for sampling and analysis of periphyton are not discussed in this chapter, but are summarised in Biggs and Kilroy (2002).

FACTORS CONTROLLING PERIPHYTON BIOMASS

Local factors controlling biomass and type of periphyton are the result of what can best be described as a “cascade of controllers”. At the top of the cascade are the ultimate controlling variables—climate (including precipitation and temperature) and catchment geology. Together with land use (a human-influenced component of the stream environment), these variables control the physical and biological features of the wider landscape setting of stream ecosystems (Biggs *et al.* 1990). Lower down the cascade are the reach-scale variables, which are of more immediate importance to periphyton. These include light, flow regime, wave action, nutrients, temperature and other biota that may interact with periphyton, such as grazing invertebrates. It is these local, or proximate, variables that can often be influenced and managed by humans.

Broad-scale spatial patterns of periphyton are a result of

Table 15.1 Instream values that can be compromised and associated problems that may arise as a result of periphyton proliferations (from Biggs 2000a).

Instream Value	Problem
Aesthetics	Degradation of scenery, odour problems
Biodiversity	Reduction in diversity of invertebrate communities (including loss of pollution-sensitive taxa), dislocation of fish communities (see Water Quality)
Contact recreation	Impairment of swimming, odour problems
Fishing	Reduction in fish activity/populations, fouling lures
Industrial use	Taste and odour problems, clogging intakes
Irrigation	Clogging intakes
Monitoring structures	Fouling of sensor surfaces
Potable supply	Taste and odour problems, clogging intakes
Native fish conservation	Impairment of spawning and living habitat
Stock health	Toxic blooms of cyanobacteria
Waste assimilation	Reduces streamflow, reduces ability to absorb ammonia, reduces ability to process organics without excessive DO depletion
Water Quality	Increased suspended detritus, interstitial anoxia in stream bed, increased DO and pH fluctuations, increased ammonia toxicity, very high pH
Whitebait fishing	Clogging nets

environmental conditions, and natural variability in the life cycle of communities. In general, it is a battle between periphyton growth and losses that determines the extent of community development at any given point (Fig. 15.1).

The rate of periphyton biomass accrual is dictated by the rate of division of cells on the substrate. This, in turn, is controlled by the supply of resources and temperature. Resources refer to the nutrients phosphorus, nitrogen and

carbon (usually from dissolved carbon dioxide in the water), and light. Together these provide the basic building blocks and the energy necessary for photosynthesis and cell growth.

In general, temperature sets the upper limit for growth rates in algae (DeNicola 1996), given that adequate nutrients and light are available (termed *nutrient saturation* and *light saturation*). This is because the rates of many

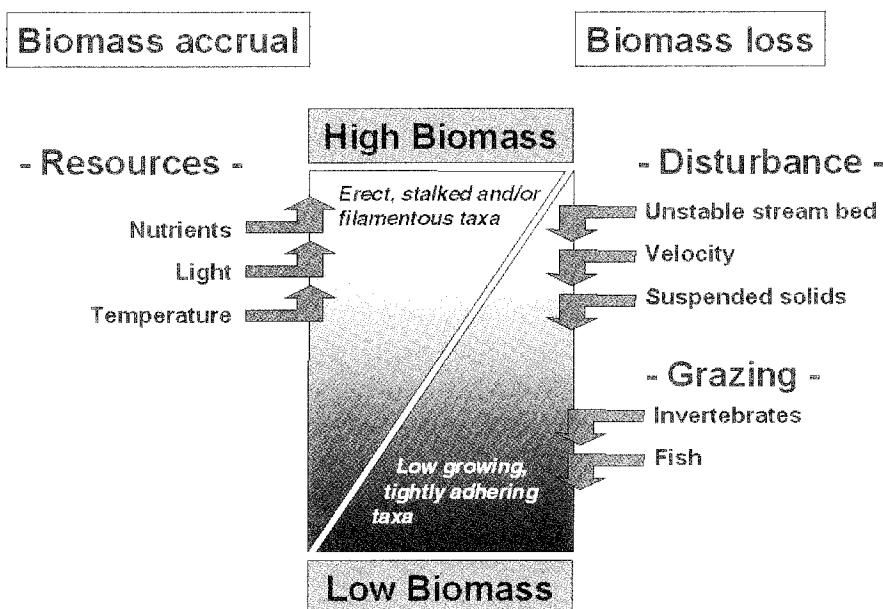


Figure 15.1 Summary of factors controlling periphyton biomass accrual and losses. The relative balance of “biomass accrual” and “biomass loss” processes is depicted by the width of the triangles that make up the central rectangle. The physiognomy of the community likely to dominate each end of the gradient is also shown (reproduced from Biggs 1996, with permission from Academic Press).

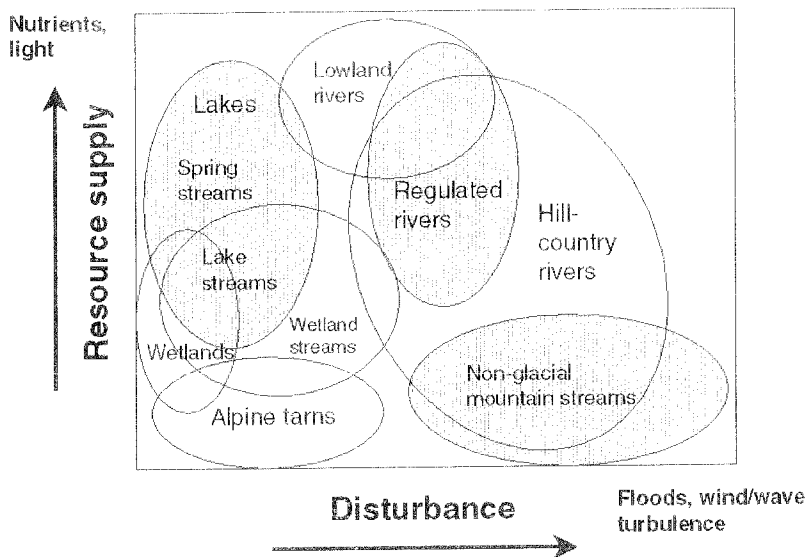


Figure 15.2 General habitats for periphyton in New Zealand as a function of the interaction between physical disturbance and resource supply (mainly nutrients). Hatched areas are included for clarity only, to indicate which habitat is associated with which encircled area (after Biggs 2000a).

metabolic processes (such as the production of photosynthetic enzymes, light harvesting and respiration) depend on temperature. Temperature also interacts with nutrient supply at levels below nutrient saturation through an alteration in metabolic and energy pathways (DeNicola 1996). Low temperatures may cause cell membranes to stiffen, thus impairing the ability of proteins to actively transport nutrients into the cell (Nedwell 1999).

Different combinations of physical disturbance and nutrient supply occur in New Zealand's diverse freshwater systems and create a "habitat matrix" (Fig. 15.2). The different zones in this matrix encompass characteristics that are found in specific habitat types, or parts of habitats. Different communities have evolved to exploit the various habitats. In New Zealand most ecological work on periphyton has been in flowing water. As in other parts of the world, periphyton communities in New Zealand's lakes and wetlands have received comparatively little attention (Lowe 1996).

Factors controlling periphyton growth

Light

Light is a fundamental factor affecting periphyton production in freshwaters (Hill 1996). However, light levels need to be very low before they limit growth, and high light levels, particularly in the short-wave (UV) range may also inhibit or otherwise affect growth (Hill 1996). It seems likely that the growth of some periphytic red algae is inhibited by high light intensities. For example, *Batrachospermum* is generally found in shaded streams and

Bostrychia can proliferate under bridges (P. Novis, pers. comm.).

The relationship between light intensity and photosynthetic rates has been studied in some common New Zealand stream periphyton communities (Dodds *et al.* 1999). The onset of light saturation of photosynthesis for whole mats of mucilaginous diatoms was usually less than 500 $\mu\text{mol photon/m}^2/\text{s}$ (bankside summer light intensities are commonly $>1500 \mu\text{mol photon/m}^2/\text{s}$). Similarly, Davies-Colley *et al.* (1992) in several West Coast streams, and Young and Huryn (1996) at multiple sites in the Taieri River, Otago, found that where the river was fairly shallow and clear, photosynthesis was generally light-saturated at bankside intensities $>250 \mu\text{mol photon/m}^2/\text{s}$. Hawes and Smith (1994) demonstrated a major increase in the rate of carbon fixation in Lake Taupo littoral periphyton at light intensities up to 300 $\mu\text{mol photon/m}^2/\text{s}$, which suggests that the lotic and lentic periphyton communities

are responding similarly to changes in light intensity. In turbid waters (e.g., >7 NTU (Nephelometric Turbidity Units), equivalent to a black disk clarity ~ 1 m), reductions in productivity can occur as incident light levels drop below $\sim 1000 \mu\text{mol photon/m}^2/\text{s}$ (Davies-Colley *et al.* 1992; Young and Huryn 1996). Shading by riparian vegetation is a possible way to reduce excess growth of periphyton in enriched streams. However, in a series of experiments in stream-side channels at Whatawhata, Waikato, Quinn *et al.* (1997) found that periphyton communities adapted to low light conditions and were light-saturated at $<100 \mu\text{mol photon/m}^2/\text{s}$. The biomass of periphyton declined when shade was increased to over 60%. In a subsequent study, Davies-Colley and Quinn (1998) surveyed stream sites in the North Island and found that levels of periphyton chlorophyll *a* of $>100 \text{ mg/m}^2$ were mainly restricted to reaches where the light levels exceeded 3% of unobstructed sky light. Light levels of less than this could occur with stream widths of ≤ 4.5 m if riparian zones were covered in native forest species and ≤ 5.5 m if riparian zones were covered in tall pine trees.

Inorganic nutrients

With sufficient light, periphyton growth may be controlled by nitrogen (N) and/or phosphorus (P) concentrations. Levels of these nutrients may be linked to an intensification in land use, but they can also originate from soft Tertiary sedimentary rocks and recent volcanic rocks in the catchment (Close and Davies-Colley 1990; Biggs 1995).

An important question, particularly for water management, is: Which nutrient is in shortest supply and so might be limiting rates of periphyton production? In general, if soluble inorganic concentrations of either nutrient are very high (e.g., $P > 50 \text{ mg/m}^3$ and $N > 500 \text{ mg/m}^3$), then nutrient availability is unlikely to be limiting productivity. At lower concentrations either nutrient could be important. Despite moderately extensive research (e.g., Biggs and Close 1989; Winterbourn 1990; Biggs *et al.* 1998a; Francoeur *et al.* 1999), few generalisations are possible about the habitats most likely to be limited by low levels of nitrogen or phosphorus for periphyton growth. Exceptions are streams draining alluvial outwash plains that have been developed for agriculture, and streams draining recent volcanic areas. The former usually have high nitrogen because of enriched groundwater inputs. As a result phosphorus is likely to be the nutrient whose levels limit growth (e.g., Biggs and Close 1989). In contrast, streams draining recent volcanic areas tend to be rich in phosphorus, and levels of nitrogen are likely to be limiting growth (White 1982). The limiting nutrient can change during the year, depending on factors such as the physiological status of the algae, community composition, biomass and light/temperature (Francoeur *et al.* 1999). In a study of West Coast and Canterbury foothills streams, Biggs and Smith (2002) demonstrated that the proportion of nitrogen-fixing Cyanobacteria in the mat increased sharply when the mean monthly soluble inorganic nitrogen concentration dropped below 25 mg/m^3 , and the ratio of soluble inorganic N:P was < 10 . This latter ratio has often been used to indicate if the availability of nitrogen or phosphorus is limiting periphyton growth, but the ratio is often not reliable (Francoeur *et al.* 1999; Francoeur 2001).

There is some discussion in the international literature as to the most appropriate nutrient measurements for characterising periphyton nutrient supplies (Biggs 2000b; Dodds 2003). We will focus on the soluble inorganic nutrients in streams, as these have yielded the strongest (though still weak) relationships between nutrients and periphyton biomass in New Zealand streams (Biggs 2000b).

It is important to understand that nutrient–biomass relationships are closely related to hydrological disturbance and invertebrate grazing. Extended periods of low flow with minimal grazing are required before a given nutrient supply will be converted into biomass. In other words, the communities need time to grow. Biomass can be quantified as chlorophyll *a*, either mean monthly (i.e., the average state of the stream) or as maximum monthly (this is of greater interest for water management).

It is also important to understand that “snapshots” of nutrient concentrations are usually not adequate to link to, or predict, biomass. This is because soluble nutrients

in the water at one instant do not indicate the longer term supply or flux. Indeed, extremely low phosphorus may be recorded in streams at times of periphyton blooms as a result of uptake by the algae. To characterise the nutrient status of a stream requires data to be collected at least monthly for over a year (to obtain mean monthly concentrations). The high stream-power of many New Zealand waterways means that accumulated algae are usually washed from the systems, thus probably contributing little to longer-term nutrient recycling.

While the growth of cells on the surface of periphyton mats does not appear to be limited by low concentrations of nutrients (because flowing water enhances nutrient mass transfer; Bothwell 1989), overall biomass is related to nutrient supply concentrations in the water (Biggs 2000a,b). Studies carried out to date on New Zealand streams have shown weak relationships between mean monthly periphyton biomass and mean monthly soluble inorganic nutrient concentrations (e.g., $r^2 = 0.122$ for nitrogen and $r^2 = 0.226$ for phosphorus, based on data from 30 stream sites: Biggs 2000b). Correlations are slightly improved for maximum monthly biomass (Fig. 15.3), but there is still considerable unexplained variance. Much of this is related to the duration of stable flows: for a given mean monthly nutrient concentration, peak biomass is higher in streams with long periods without floods (e.g., > 50 days). The duration of stable flows has been incorporated into a statistical model of maximum monthly periphyton biomass in streams, and together with nutrients explains more than 70% of the variance in biomass (see Biggs 2000a,b for more details). New Zealand appears to be the only country where reasonable quantitative links between periphyton biomass, and nutrient and disturbance levels, have been established for natural community growth in streams.

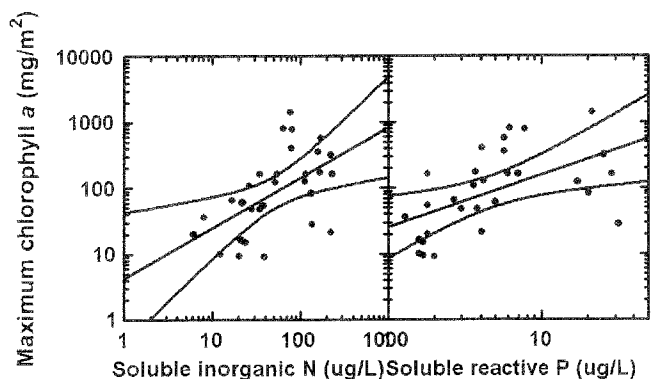


Figure 15.3 Maximum monthly periphyton chlorophyll *a* as a function of mean monthly dissolved nutrient concentrations (reproduced from Biggs 2000b, with permission from Journal of the North American Benthological Society).

Biomass criteria have been developed by Dodds *et al.* (1998) for designating streams in three general enrichment categories: oligotrophic, mesotrophic and eutrophic. Using the statistical model of maximum monthly biomass as a function of nutrient concentration and days of accrual, Biggs (2000b) developed a nomograph from which it is possible to simply read off the likely trophic state of a stream reach for any given mean monthly nutrient concentration and expected accrual period (Fig. 15.4). From the nomograph, it is clear that streams that flood frequently (and thus have shorter accrual periods) can potentially assimilate much higher nutrient levels without exceeding periphyton biomass guidelines than streams with more benign flow regimes.

In lakes, periphyton biomass displays only a weak linkage with water column nutrients compared to phytoplankton (Lowe 1996). In some circumstances there can be an inverse relationship between periphyton biomass and planktonic biomass (i.e., lake trophy). Thus, Hansson (1988) demonstrated experimentally that periphyton could be competitively superior when water column nutrients were low because of access to nutrients released from sediments, whereas phytoplankton out-competed periphyton for light if water column nutrient concentrations increased. This may apply in New Zealand's more nutrient-rich lakes. DeNicola *et al.* (2003) have proposed a hierarchical model of factors that influence periphyton growth in lakes in Ireland (Fig. 15.5)—this model may also be relevant in New Zealand, but remains to be tested.

Hawes and Smith (1994) showed that oligotrophic Lake Taupo supported a very high biomass of periphyton at their study sites, probably as a result of sustained slow growth

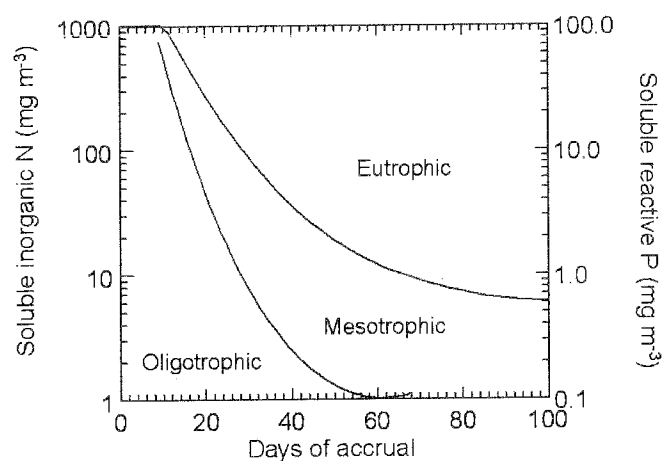


Figure 15.4 Mean monthly soluble nutrient concentrations predicted to result in maximum benthic algal biomass indicating oligotrophic, mesotrophic, and eutrophic conditions in gravel/cobble-bed streams for varying days of accrual. (reproduced from Biggs 2000b).

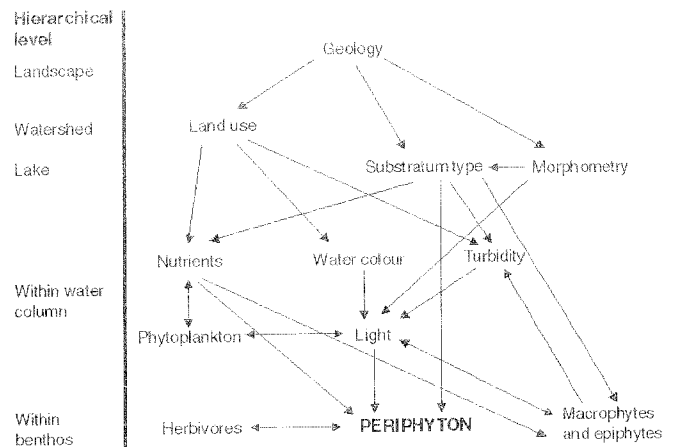


Figure 15.5 Hierarchical arrangement of potential factors influencing periphyton production in Irish lakes: a possible template for application to New Zealand systems (from DeNicola *et al.* 2003).

with few losses. In a separate study they showed that localised enrichment of the lake waters appeared to have a direct effect on both periphyton biomass and community composition, although this was complicated by occasional dramatic losses in biomass as a result of high winds and associated wave disturbance (Hawes and Smith 1993).

Since many of New Zealand's large lakes are oligotrophic, and potentially vulnerable to eutrophication as a result of changes in land use, a broader study of the relationship between lake trophic status, changes in water clarity and periphyton production appears warranted.

Preventing proliferations

Nutrient guidelines have been developed for different durations of low flows to enable various instream values to be protected in New Zealand. For example, where aesthetics and trout fishing are the primary values to be protected, it is recommended that maximum monthly biomass be $<200 \text{ mg chlorophyll } a/\text{m}^2$ of stream bed. To achieve this, it is recommended that mean monthly soluble reactive P be $<1 \text{ mg}/\text{m}^3$ for streams prone to long periods of low flow (e.g., >50 days), but may be as high as $10 \text{ mg}/\text{m}^3$ for more flood-prone streams where accrual time is generally <20 days (Biggs 2000a). Guidelines for protecting biodiversity values are much stricter (maximum monthly chlorophyll $a <50 \text{ mg}/\text{m}^2$) and therefore require much lower nutrient concentrations for this value to be protected.

An important but poorly understood effect of high nutrient concentrations on periphyton biomass is that nutrients not only enhance maximum biomass, but also result in high biomass being maintained for longer. This is thought to be because the greater molecular diffusion to the base of the mats may extend the life of the basal layers before they degrade, causing spontaneous sloughing. For

example, the total duration for which biomass exceeds 200 mg/m^2 chlorophyll *a* increases from 1.5 to 3 months per year for an increase in mean monthly soluble reactive phosphorus from 5 to 15 mg/m^3 , where days of accrual average <50 per year (Biggs 2000b).

Factors controlling periphyton biomass losses

Physical disturbance

In rivers and streams, the time available for periphyton to grow without being severely disturbed by floods is probably the most important factor controlling broad-scale variations in periphyton development in catchments in New Zealand. Indeed, in a study of nine rivers in Canterbury, Biggs and Close (1989) reported that 63% of the variance in mean periphyton biomass was explained by the percentage of time in flood, and in a set of 30 streams around New Zealand, Biggs (2000b) reported that 62% of the variance in maximum monthly biomass was explained by the mean number of days between floods (Fig. 15.6). High biomass (i.e., $>200 \text{ mg/m}^2$ chlorophyll *a*) generally does not occur in streams that have fewer than 30 days between flood disturbances. Floods also affect variation

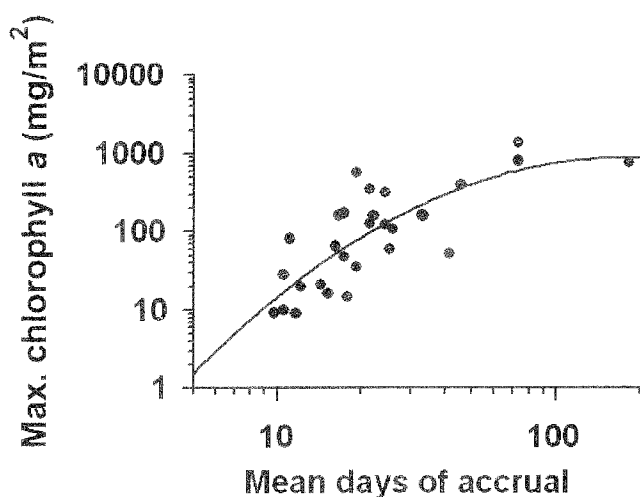


Figure 15.6 Maximum monthly periphyton biomass (as chlorophyll *a* concentrations) at 30 stream sites as a function of mean days of accrual (calculated as $365/[\text{mean number of events per year exceeding } 3 \times \text{median flow}]$) (reproduced from Biggs 2000b).

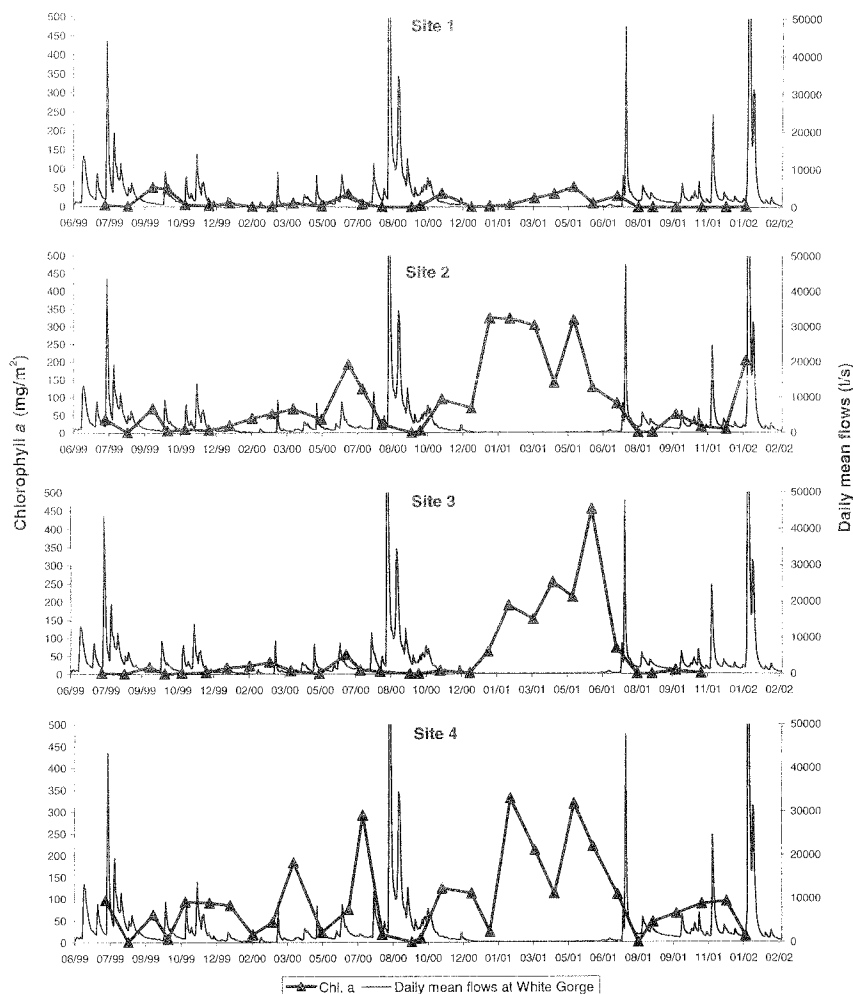


Figure 15.7 Temporal variations in periphyton biomass (as chlorophyll *a* concentrations) at four sites down the Waipara River, North Canterbury, reflecting a downstream gradient of increasing nutrient enrichment. Declines in periphyton biomass reflect floods or freshes as well as natural decay and sloughing (reproduced with permission from Hayward 2003).

through the year, with cycles of growth followed by sloughing being common (particularly in moderately to highly enriched rivers such as the Waipara River, Canterbury; Fig. 15.7).

The amount of biomass lost in a flood depends on the intensity of the event (i.e., flow velocity, and stability of bed sediments) and the resistance of the communities. Intense floods in New Zealand (e.g., with a 0.5–1 year return period) are generally catastrophic for all periphyton community types because they involve so much bed sediment movement. However, the outcomes of small to intermediate-sized floods (commonly called “*freshes*”) that may result in only partial mobilisation of the bed sediments (i.e., “*sub-catastrophic disturbances*”) are much more difficult to predict. For such events, the degree to which the periphyton community can resist being torn from the substrate has a major bearing on the degree of community disturbance. As might be expected, communities com-

posed of low-growing, strongly attached taxa such as some diatoms (*Achnanthisidium*, *Cymbella*, *Cocconeis*, *Synedra*) and larger taxa with strong holdfast structures (such as the green filamentous algae *Stigeoclonium lubricum* and *Ulothrix zonata*) are much less affected by freshes than communities composed of tall-growing, high-biomass, loosely adhering taxa (such as the diatom *Melosira varians*, and the green filamentous alga *Cladophora glomerata*) (e.g., Peterson 1996). Thus, depending on what was growing at a site prior to a fresh, there may be quite different outcomes in terms of disturbance to community form and functioning. Biggs and Thomsen (1995) pointed out that because periphyton growing in New Zealand's oligotrophic streams usually comprises thin films with low biomass, these communities will be less prone to disturbance by freshes than the higher biomass, often filamentous communities that normally dominate eutrophic streams.

A critical problem in determining the importance of floods for growth of periphyton in freshwater systems is identifying which flow events cause a significant loss of biomass (i.e., "reset" the community). Considerably more work is required before this aspect of periphyton growth can be fully incorporated into mechanistic models. It seems that the *relative* magnitude of increase in near-bed velocities is relevant rather than the absolute velocity. The amount of biomass present is also important (Biggs and Close 1989). For example, Biggs and Thomsen (1995) reported, from laboratory flume tests, that < 50% of a low biomass diatom film was removed by velocity increases alone (0.3 to 1.5 m/s), whereas > 80% of a higher biomass filamentous diatom community was removed by the high velocities. Also, once sands are mobilised, they cause additional losses through abrasion (Francoeur and Biggs in press). When larger bed sediments start moving in a flood, the grinding action of the particles removes almost all biomass (e.g., Francoeur *et al.* 1998; Biggs *et al.* 1999).

Different measures have been used to quantify flood disturbances. These include the number of events per year exceeding 3 or 5 times the median flow (or average time between such events), the number of floods in which a specific velocity threshold is exceeded, the events severe enough to move up to 84% of the bed sediments, and the proportion of marked rocks moved from a site (Scarsbrook and Townsend 1993; Death and Winterbourn 1994; Biggs 1995; Clausen and Biggs 1997; Biggs *et al.* 1999; Biggs 2000b). Each approach has advantages and disadvantages. The most thorough is an assessment of the frequency of bed sediment movement at a site, but this requires considerable field work and some modelling to generate the necessary statistics.

The frequency of flood disturbances in streams also has a major bearing on whether nuisance periphyton may occur. A 1992 study (unpublished data, NIWA) reported

data that suggested that proliferations could occur in experimental channels fed by the oligotrophic Waitaki River when there were fewer than 13 floods/year (of $>3 \times$ the existing flow), but proliferations did not occur when there were more than 26 floods/year. Similarly, Clausen and Biggs (1997) found generally high mean monthly biomass of periphyton in streams with fewer than 10 floods/year (of $>3 \times$ median flow; FRE₃) at 25 stream sites distributed across New Zealand (also, see previous section).

Invertebrate grazing

Losses of periphyton by invertebrate grazing are potentially high in New Zealand freshwater systems, but depend on the density and type of invertebrate grazers present and the growth rate of the periphyton (reflecting nutrient and light supply, and temperature). In streams not prone to frequent disturbances, invertebrate communities are often dominated by snails (particularly *Potamopyrgus antipodarum*), with sub-dominant populations of caddisfly larvae and mayfly larvae (e.g., Quinn and Hickey 1990). With increasing frequency of disturbance, snails tend to be eliminated first, then caddis larvae. In highly disturbance-prone streams mayflies usually dominate (e.g., Sagar 1986; Scarsbrook and Townsend 1993). This also appears to create a gradient in grazing intensity, because snails seem to be much more aggressive grazers than caddisflies, which in turn appear to be more aggressive grazers than mayflies (Biggs *et al.* 1998b).

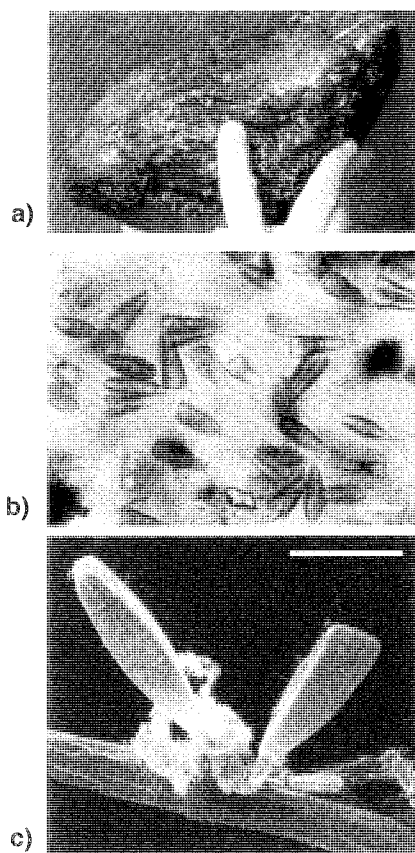
Where densities of grazers are low (e.g., in low-gradient streams where there is a lack of suitable habitat because interstitial spaces have become silted), periphyton growth can proceed largely unconstrained and proliferations may develop. Sometimes, in gravel-bed streams, large amounts of periphyton will accumulate quickly after a major flood, while there are still few invertebrates present. Invertebrates take much longer to colonise and reproduce than periphyton (e.g., Scrimgeour and Winterbourn 1989; Biggs and Stokseth 1996), but after several months may catch up and begin to control the accrual of periphyton biomass, providing that the rate of production is not high. In some cases, invertebrate populations (particularly *Potamopyrgus*) may track local areas of high periphyton productivity. Thus, instead of high periphyton biomass accruing, very high local densities of invertebrates can develop (Biggs and Lowe 1994). Overall, control of periphyton by grazing invertebrates is likely to be most significant in physically stable environments.

Recent studies have discovered moderate to strong "top-down" trophic cascades in many New Zealand streams where trout (which are aggressive predators) reduce the number and activity of grazers, which has then allowed

the accumulation of significantly higher periphyton biomass compared with streams containing only native fish (Flecker and Townsend 1994; Biggs *et al.* 2000; Nyström *et al.* 2003). Because trout displace native fish in many less flood-prone streams (e.g., McIntosh 2000), it has been conjectured that trout streams will generally be “greener” than native fish streams (Biggs *et al.* 2000).

COMPOSITION OF PERIPHYTON

Periphyton communities in New Zealand freshwaters are most commonly dominated by diatoms. In rivers, these can comprise fine dark-brown films of unicells (e.g., *Achnantheidium* spp., *Synedra* spp., *Cocconeis* spp.), through to thick olive-cream mucilaginous mats of stalked/tubed taxa (e.g., *Gomphoneis* and *Cymbella*) (Fig. 15.8) and, occasionally, mats of grey-brown filamentous taxa (*Melosira varians*, *Fragilaria vaucheriae*) (Biggs 1990; authors’ unpublished data and observations) (Fig. 15.9). The mucilaginous mats can be mistaken for “sewage fungus” when fully developed. Periphytic diatoms in New Zealand’s larger oligotrophic lakes tend to be dominated



Photos: (a,c) Barry Biggs, (b) Cathy Kilroy.

Figure 15.8 The diatom genus *Gomphoneis*. (a) Thick mucilaginous diatom growth (mainly *Gomphoneis*) on a rock. (b) Live cells, showing stalks. (c) Scanning electron micrograph of *Gomphoneis* attached as an epiphyte to a filament of the green alga *Cladophora*. Scale bars: 50 μm .

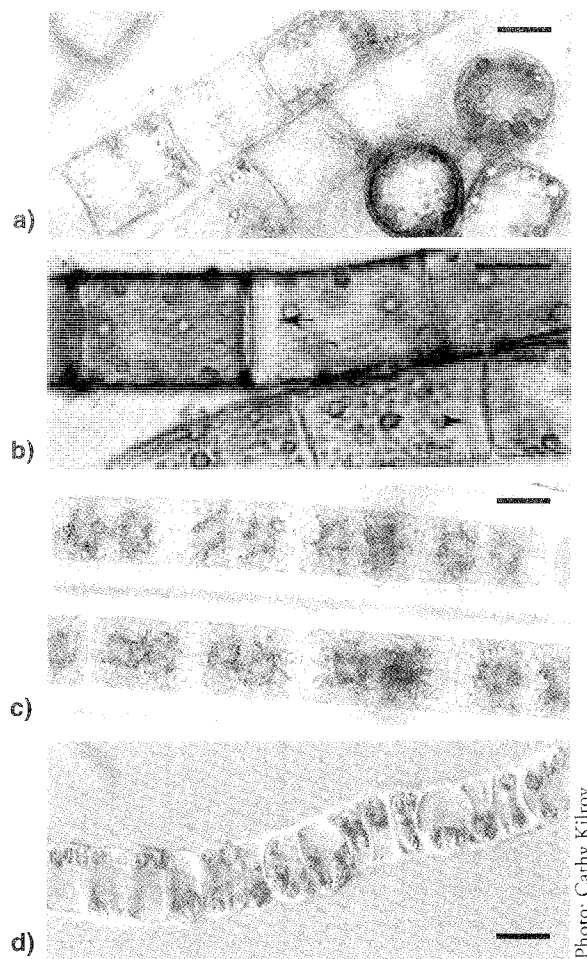
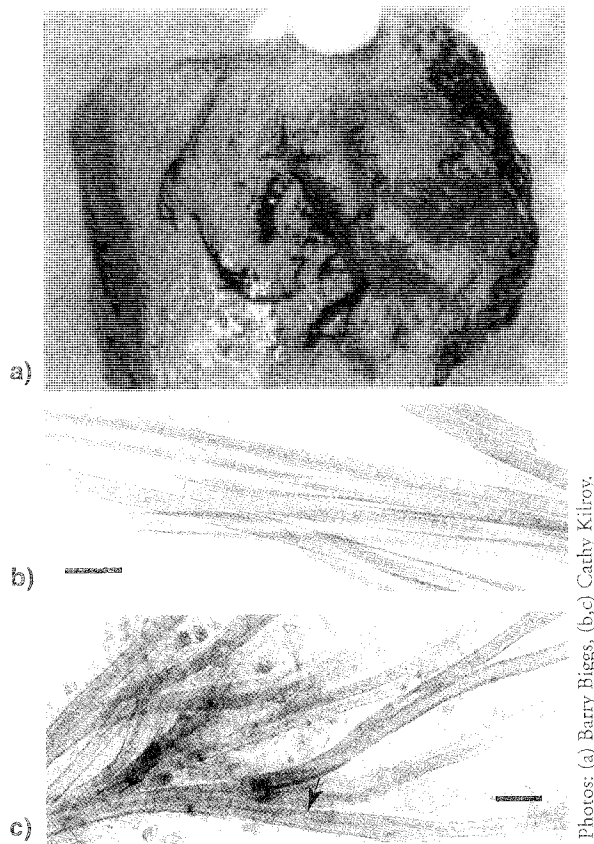


Photo: Cathy Kilroy

Figure 15.9 Common filamentous algae. (a) *Melosira varians*, a filamentous diatom. (b) *Oedogonium* sp. Note the cell scarring (arrow), which is diagnostic in this genus. (c) *Zygnema* sp., a common genus in slow-flowing rivers and in wetlands. (d) *Ulothrix zonata*, common in mountain streams. Scale bars: 20 μm .

by species of *Achnantheidium*, *Fragilaria*, *Staurosira*, *Staurosirella*, *Epithemia* and *Rhopalodia* (Hawes and Smith 1994, authors’ unpublished observations). In lakes and smaller water bodies where the pH is around 7 or less, acidophilous diatom genera such as *Tabellaria*, *Brachysira* and *Frustulia* become more important.

Green algae (usually filamentous) are probably the second most important component of the biomass in freshwaters. In rivers and streams, species of *Spirogyra* are the most common, followed by *Oedogonium* spp. (Biggs and Price 1987; Biggs 1990). Other genera, including *Zygnema* and *Bulbochaete* are more typical of lake and wetland habitats, although *Oedogonium* spp. are also widespread (authors’ unpublished observations) (Fig. 15.9). Filamentous Cyanobacteria are commonly found after prolonged periods of low, stable flow in foothills rivers and on silts in many ponds (Biggs 1990; authors’ unpublished observations). These mats are often made up of *Phormidium* spp., but may also contain taxa such as



Photos: (a) Barry Biggs, (b,c) Cathy Kilroy.

Figure 15.10 Cyanobacteria. (a) Typical black growth of *Phormidium* on a rock. (b) Trichomes of *Phormidium*. (c) Tapering trichomes of *Dichothrix*. Note the heterocysts at the base of each trichome (arrow). Scale bars: 10 µm.

Homeothrix and *Calothrix/Dichothrix* (Fig. 15.10). In stable conditions, thick cyanobacterial mats can develop in the littoral zones of lakes, and in shallow tarns and wetland pools. These generally comprise mucilaginous colonies of Chroococcales and narrow trichomes of Oscillatoriales. A fuller account of the periphyton communities occupying different habitats in New Zealand follows.

Common taxa and regional distributions

A major difficulty in evaluating periphyton in New Zealand is that many of the taxa cannot be identified to species level because, for example, they lack the necessary reproductive structures. This has been particularly problematic for Cyanobacteria and filamentous green algae, although recent work is starting to clarify the possible species assemblages of groups such as the Oedogoniales, Microsporales and Zygnematales (Novis 2003, in press a, b). As a result, often only the genus name is used. The term *taxon* is used to refer to an organism identified to its lowest practical taxonomic level.

Most common periphyton taxa in New Zealand streams also occur elsewhere in the world. The small amount of information available on periphyton in larger lakes suggests

that the same applies to these habitats. However, our less accessible habitats appear to support a much larger proportion of regional species, many of which remain to be identified or described (authors' unpublished observations). Therefore taxonomic guides developed in Europe and North America are generally useful, but have limitations. Checklists have been prepared of freshwater algae found in New Zealand (e.g., Cassie 1984a, b, c) and include mostly cosmopolitan taxa. The small number of "endemics" noted may simply reflect the lack of detailed taxonomic work on algae in New Zealand. Nevertheless, for the common taxa, a developing body of information internationally on the habitat requirements and pollution tolerances of taxa can also be applied in New Zealand to assist with assessing water quality conditions or the effects of water management decisions.

Regional differences in communities

Extensive surveys (e.g., Biggs and Price 1987; Biggs 1990) have indicated that the distributions of river periphyton taxa do not reflect historical geological events such as volcanism or Pleistocene glaciation. Many (if not most) algae are capable of rapid aerial dispersal through spores, and this is likely to have enabled newly created habitat to become colonised by a broad pool of propagules. Indeed, ease of colonisation is implicit in the success of algae in freshwaters, including ephemeral habitats.

Some regional differences have been recognised, but most of these appear to reflect broad-scale controlling factors such as geology and flow variability. For example, proliferations of "blanket weed" (i.e., *Cladophora glomerata* and *Rhizoclonium* sp.) occur most frequently in central North Island streams in an area surrounding, but not in, the Volcanic Plateau. This reflects a combination of high enrichment of streams through seepage from mineral-rich papa-mudstone (Tertiary marine sediments), land development and high summer temperatures. Conversely, communities dominated by the unbranched filamentous green alga *Ulothrix zonata* and the branched filamentous alga *Stigeoclonium* sp. are typically found along the axial ranges of both the North and South Island, reflecting cooler water temperatures and lower levels of enrichment in these areas. An example of a distribution that may reflect regional differences in climate (mainly temperature), is that of the filamentous red alga *Compsopogon coeruleus*. This taxon is most commonly found in enriched streams of the northern North Island (Biggs and Price 1987), where it forms long, ropy, grey-green strands. The regional distributions of some diatom taxa may also reflect climate differences. For example, *Diatoma tenuis* has been recorded mainly in the southern part of the South Island, where it may dominate the periphyton in some rivers (authors' observations).

Periphyton communities of specific habitats in New Zealand

Based on sampling in streams and rivers throughout New Zealand, a set of about 30 algal taxa that account for a large proportion of the periphyton biomass in most streams has been identified (Biggs and Price 1987; Biggs 1990). From this set we can distinguish at least 16 main periphyton community types (Biggs 1995; 2000a,b; Biggs *et al.* 1998a). For convenience, the communities can be separated into groups that are normally associated with un-enriched (oligotrophic), moderately enriched (mesotrophic) and enriched (eutrophic) habitats (Table 15.2). Further periphyton community types are associated with lake and wetland habitats but are not yet well defined. The association of community types with particular habitats tends to be clearer at the extremes of the habitat spectrum (e.g., oligotrophic vs. eutrophic). Also, many of the taxa commonly found in New Zealand freshwater ecosystems appear to have wide ecological tolerances (and are often cosmopolitan). The river and stream periphyton communities described below are not all the possible types, and the lake and wetland communities are at this stage based mainly on the authors' observations rather than published surveys.

Spring streams

The overriding habitat characteristic of spring streams is the high degree of flow stability. In lowland areas these streams generally have fine bed sediments (usually silts, sands and fine gravels), which are often colonised by aquatic macrophytes. Gravel/cobble substrata for epilithic periphyton may be rare except in steeper streams with higher water velocities. Spring streams are often quite enriched, depending on the geology and land use of the catchment, and on whether the seepage waters are derived from gravels in an unconfined aquifer (Biggs and Close 1989). Thus, a broad range of community types can be expected. *Vaucheria* can often be found in cold spring streams, where it forms large mats over silts and sands. Fragile filaments of the diatom *Diatoma heimale* var. *mesodon* may also proliferate in cooler spring streams (Death 1991; authors' observations).

Of all the different stream types, spring streams are most likely to have seasonal variations in biomass rather than biomass changes driven by flow variability. Indeed, in the spring streams of Canterbury, the highest biomass (usually caused by blooms of *Ulothrix zonata*, or *Oedogonium* in more enriched sites, Fig. 15.9) occurs in late summer/autumn and late winter (e.g., Biggs and Close 1989). Between these times, periphyton films usually persist on rocks and intense invertebrate grazing probably prevents the biomass from increasing (Winterbourn and Fegley 1989). The reason for the two seasonal peaks in biomass

of filamentous green algae is unclear. Water temperatures do not change much between winter and summer in spring streams (except in downstream reaches, far from their source) and always appear sufficient for high invertebrate activity (authors' observations).

Lake streams

Lake outlet streams, like spring streams, generally have low variability of flow. This can lead to high densities of benthic invertebrate grazers and high periphyton removal rates (e.g., in mid- to low-altitude lake outlet streams: Harding 1994). However, both bed gradient and trophic state cover a wider range. As a result the periphyton communities dominating such streams are again highly variable.

There have been few investigations of the periphyton communities of lake streams. Biomass in Grasmere Stream (draining Lake Grasmere, Canterbury) showed the most marked seasonal changes of six "stable" sites in the vicinity of Cass, over a 5-year period (Death 1991). The community comprised mainly filamentous cyanobacteria (probably *Lyngbya*), along with common diatom taxa, including *Cymbella* sp., *Achnanthydium minutissimum* and *Gomphonema* sp. Apart from this study, most sampling in lake streams and rivers has been undertaken in outlets regulated for hydroelectric power development and it is unlikely that this information is totally applicable to unregulated lake systems.

Lowland streams

Streams fed by runoff from low-elevation, coastal areas are characterised by low gradients, low water velocities and mainly sand/silt beds. Many of these streams are partially spring-fed and therefore may also have low flow variability. They also tend to be very enriched because of farming activities. Few such streams have natural, intact, riparian and/or catchment vegetation, and we have no knowledge of the composition of natural periphyton communities in such habitats. In areas with agricultural development, lowland streams often lack significant periphyton communities because the waterways are generally choked with macrophytes. Here the algal growth may consist of filaments either entangled on macrophytes or growing on gravels/cobbles in localised steeper reaches. In such streams mesotrophic and eutrophic communities can attain a very high biomass. In particular *Melosira varians*, *Oedogonium* and *Microspora* have been recorded in extensive growths in enriched South Island lowland streams, and *Cladophora glomerata* in North Island streams (Biggs and Price 1987).

Hill streams and rivers

This is by far the most common stream type in New Zealand and has the greatest range in physical conditions.

Table 15.2 Periphyton communities commonly found in New Zealand streams and rivers of different trophic state.

Community identification code and Primary taxon/taxa	Secondary filamentous taxon/taxa	Understorey taxa
Oligotrophic habitats		
O1: <i>Audouinella hermanii</i>	Rare	<i>Cymbella kappii</i> , <i>Synedra ulna</i> , <i>Fragilaria vaucheriae</i>
O2: <i>Lyngbya</i> sp.	Rare	<i>Synedra ulna</i> , <i>Gomphoneis minuta</i> var. <i>cassieae</i> , <i>Navicula avenacea</i>
O3: <i>Schizothrix/Calothrix/Lyngbya</i>	Rare	?
O4: <i>Ulothrix zonata</i>	<i>Microspora</i> sp., <i>Spirogyra</i> spp., <i>Oedogonium</i> spp.	<i>Synedra ulna</i> , <i>Cymbella kappii</i> , <i>Gomphoneis minuta</i> var. <i>cassieae</i> , <i>Gomphonema parvulum</i> , <i>Fragilaria vaucheriae</i>
O5: <i>Nostoc</i> sp.	Rare	?
Mesotrophic habitats		
M1: <i>Cladophora</i> sp.	<i>Oedogonium</i> spp., <i>Melosira varians</i>	<i>Gomphonema parvulum</i> , <i>Cymbella kappii</i> , <i>Synedra ulna</i> , <i>Cocconeis placentula</i> , <i>Navicula rhyncocephala</i>
M2: <i>Fragilaria/Gomphonema tenellum/Synedra ulna/Synedra rumpens/ Cymbella minuta/ Gomphoneis minuta</i> var. <i>cassieae</i>	Rarely <i>Ulothrix zonata</i> , <i>Stigeoclonium lubricum</i>	
M3: <i>Gomphoneis minuta</i> var. <i>cassieae/Cymbella kappii</i>	<i>Ulothrix zonata</i> , <i>Stigeoclonium lubricum</i>	<i>Cymbella kappii</i> , <i>C. minuta</i> , <i>Synedra ulna</i> , <i>Fragilaria vaucheriae</i>
M4: <i>Oedogonium/Microspora/ Zygnema</i>	<i>Spirogyra</i> spp., <i>Melosira varians</i> , <i>Microspora</i> sp.	<i>Synedra ulna</i> , <i>Cocconeis placentula</i> , <i>Navicula rhyncocephala</i>
M5: <i>Phormidium</i> spp.	Rare	<i>Synedra ulna</i> , <i>Cymbella kappii</i> , <i>Gomphoneis minuta</i> var. <i>cassieae</i> , <i>Cocconeis placentula</i> , <i>Gomphonema parvulum</i> , <i>Cymbella minuta</i>
M6: <i>Spirogyra</i> spp.	Rarely <i>Oedogonium</i> spp., <i>Cladophora</i> sp., <i>Phormidium</i> spp., <i>Stigeoclonium lubricum</i>	Rarely <i>Gomphoneis minuta</i> var. <i>cassieae</i> , <i>Synedra ulna</i> , <i>Cymbella kappii</i> , <i>Gomphonema parvulum</i> , <i>Achnantheidium lanceolatum</i> , <i>Cocconeis placentula</i>
M7: <i>Stigeoclonium lubricum</i>	Rare	<i>Gomphonema parvulum</i> , <i>Gomphoneis minuta</i> var. <i>cassieae</i> , <i>Cymbella kappii</i> , <i>Synedra ulna</i>
M8: <i>Vaucheria</i> sp.	Rare	Rarely <i>Navicula</i> spp., <i>Cymbella</i> spp., <i>Synedra</i> spp.
Eutrophic habitats		
E1: <i>Melosira varians</i>	<i>Oedogonium</i> spp.	<i>Synedra ulna</i> , <i>Cocconeis placentula</i> , <i>Navicula cryptocephala</i> , <i>Navicula rhyncocephala</i>
E2: <i>Cladophora glomerata</i>	<i>Rhizoclonium</i> spp.	<i>Epithemia sorex</i> , <i>Cocconeis placentula</i> , <i>Synedra ulna</i> , <i>Cymbella kappii</i> , <i>Gomphoneis minuta</i> var. <i>cassieae</i>
E3: <i>Rhizoclonium</i> sp.	<i>Cladophora glomerata</i> , <i>Melosira varians</i>	<i>Cocconeis placentula</i> , <i>Synedra ulna</i> , <i>Cymbella kappii</i> , <i>Navicula avenacea</i> , <i>Rhoicosphenia curvata</i> , <i>Nitzschia</i> spp.

Most hill-country streams are flood-prone, with typically between 10 and 30 floods per year (e.g., Biggs *et al.* 1999). These streams tend to be steep, with strongly developed reach-scale morphological features (i.e., pool, runs and riffles). The bed sediments are generally gravels and cobbles, and there is a wide range of water velocities. They also encompass the full range of enrichment regimes as a result of differences in land use and catchment geology. During summer low flows, the periphyton community dominating on stable substrates is usually determined by (in decreasing order of importance) shading, enrichment, water velocity and invertebrate grazing. However, since most streams and rivers in New Zealand lack riparian shading except in their headwaters, the latter three factors are usually the most important controllers of periphyton biomass and community composition.

Rapidly colonising diatoms (e.g., of the *Fragilaria* and *Gomphoneis* groups) can dominate in disturbance-prone foothill streams (Biggs 1995; Biggs *et al.* 1998a; authors' unpublished data). However, a range of different communities can develop on stable (armoured) substrates in more stable streams (particularly during low flows), reflecting local conditions of nutrients, light, and velocity (e.g., Biggs 1995; Suren *et al.* 2003). The highest biomass is often in late summer (e.g., Biggs and Close 1989; Biggs *et al.* 1998a; Hayward 2003), when prolonged periods of low flow are most common. In oligotrophic streams draining bush or tussock catchments in the South Island, cyanobacterial communities dominated by *Schizothrix*, *Calothrix*, *Lyngbya* and *Heteroleibleinia* are often abundant as blue-black patches on the stones during summer low flows, particularly in high rainfall areas (Biggs and Smith 2002). The communities appear to be very resistant to grazing by invertebrates. *Tolypothrix* and *Nostoc* can also be common. Cold-water streams in the upper catchments of South Island foothill rivers are often dominated by the green filamentous alga *Ulothrix zonata* (Biggs and Price 1987; authors' observations) (Fig. 15.9). Light purple/pink-coloured mats dominated by *Audouinella* tend to be restricted to stream reaches with very stable boulder or bedrock substrata (Biggs and Price 1987; Biggs and Smith 2002) (Fig. 15.11). With low to moderate enrichment, riffles in foothills streams and rivers often become dominated by thick mucilaginous mats of *Gomphoneis* and *Cymbella* during prolonged low flows. These mats are often overgrown by dark black/brown cyanobacterial filaments cyanobacteria (*Phormidium*) late in the summer, or late in the successional cycle if prolonged low flows occur at other times of the year. Conspicuous large, bright green tufts of the branched filamentous alga *Stigeoclonium lubricum* may also develop among the diatom community, though it is rare for a whole stream reach to be dominated by *Stigeoclonium* (Biggs and Price 1987).

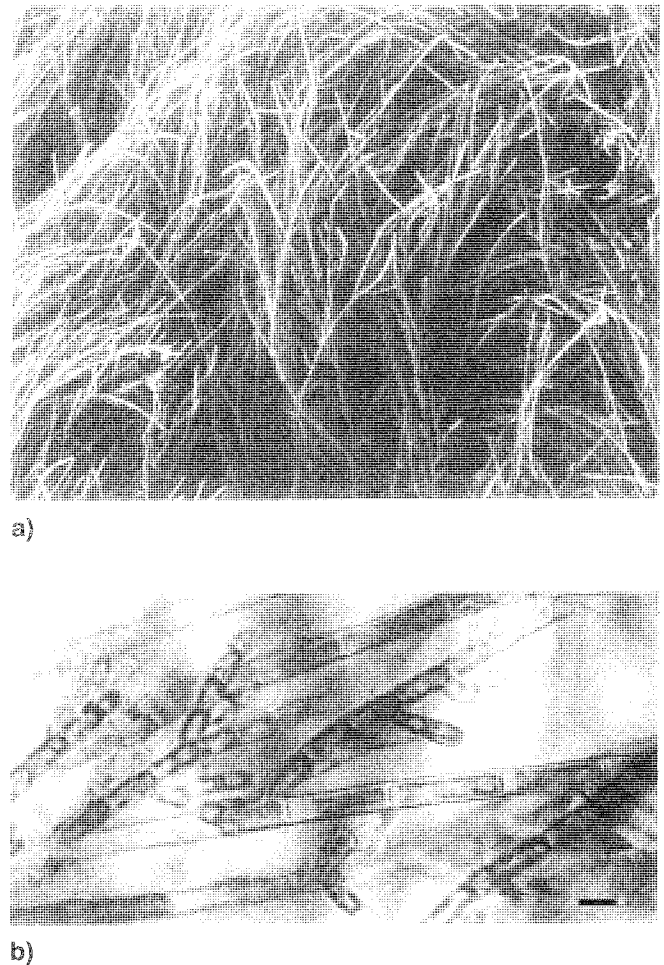


Figure 15.11 *Audouinella bermanii*. (a) Scanning electron micrograph showing three-dimensional form of an *Audouinella* mat. (b) Light micrograph of filaments. Scale bar: 20 μm . Photos: (a) Barry Biggs; (b) Nelson Boustead.

Non-glacial mountain streams and rivers

Non-glacial mountain streams and rivers above the tree-line are usually characterised by frequent floods, steep, high-energy channels with unstable bed sediments, little or no shading and low nutrient concentrations. Stable conditions that enable periphyton communities to mature are rare. The overall species pool in the upper reaches of such rivers is probably limited by the frequency of flood disturbance, and thus even during infrequent periods of low flows the periphyton communities will usually be dominated by rapidly colonising diatoms (e.g., *Gomphoneis*, *Cymbella*, *Fragilaria*). *Ulothrix zonata* is the principal filamentous green alga, because it is resistant to disturbance and can grow well in unenriched waters (Biggs 1990). Zygnematales (e.g., *Spirogyra*) are also common in some areas (e.g., Arthur's Pass, P. Novis, pers. comm.). In periods of steady flows it is common to see widespread bright brown growth of the filamentous diatom *Diatoma*

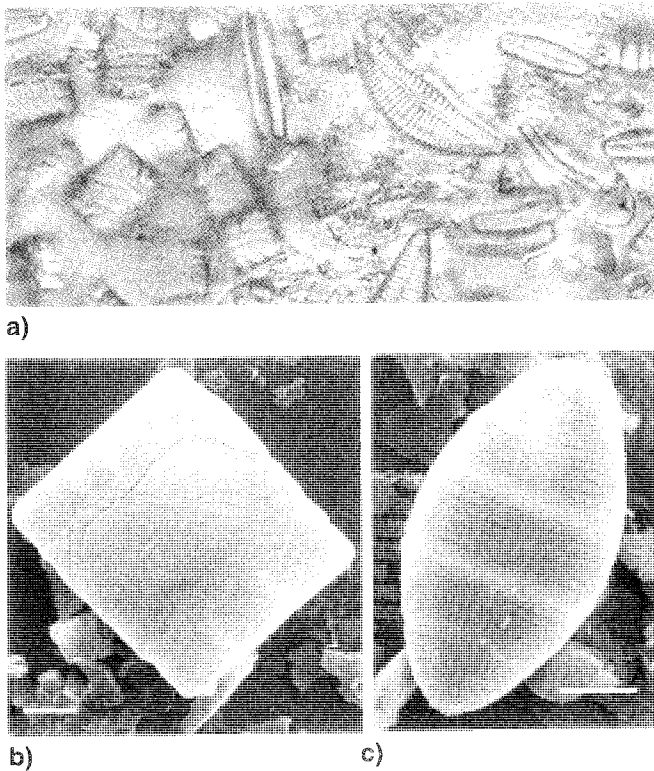


Figure 15.12 *Diatoma heimale* var. *mesodon* commonly forms a bright brown slimy covering on stones in cool-water streams. (a) Light micrograph showing *D. heimale* dominating a community, with *Encyonema* sp. (half-moon shaped) and *Achnantheidium* sp. (b, c) Scanning electron micrographs showing (b) the side (girdle) and (c) top (valve) views of a single cell. Scale bars: 3 μm .

Photos: Cathy Kilroy.

heimale var. *mesodon* in steep rivers that rise at high altitudes (authors' observations) (Fig. 15.12). Another alga characteristic of these rivers is the chrysophyean *Hydrurus foetidus*, which grows in light brown, mucilaginous masses, trailing in the flow (P. Broady, pers. comm.). Grazing mayflies, which can be very common in mountain rivers, may exert considerable top-down grazing control on periphyton communities there (see Chapter 13). In addition, low nutrient levels mean that periphyton growth is usually quite slow. As a result, mean and peak biomass is usually relatively low.

Glacial mountain rivers

These habitats are typified by high levels of suspended sediment, frequent bed sediment movement, low nutrient supplies and high light levels. Flood disturbances are usually more seasonal than in non-glacial mountain and foothills rivers: the lowest flows tend to occur in late summer/early autumn and in winter when most precipitation falls as snow. Because of the high suspended and bedload sediment supply to glacier rivers, benthic

biota are subjected to harsh, abrasive conditions with even small increases in flow. Nevertheless, periphyton biomass has been recorded as building up to noticeable levels on occasions (Biggs and Close 1989; Milner *et al.* 2001). It is important to recognise different valley segments in such rivers, because physical conditions change markedly as one moves downstream. For example, conditions may be extremely harsh in the single-thread, down-cutting phase of these rivers, but where they braid in the less confined outwash plains, many sections of the braids usually have considerably less hydraulic power and very stable interlocking cobbles (armoured) bed sediments (Milner *et al.* 2001). Periphyton is almost always very sparse in the main channels (e.g., Biggs and Close 1989). The only cells present are usually either diatoms washed in from more stable areas, or small diatom species such as *Gomphonema* spp., and *Achnantheidium* and *Achnanthes* spp. that attach themselves to rocky substrates with mucilage stalks or pads. These small tightly adhering species may occupy tiny crevices in the rocks and can regenerate rapidly following floods. Extensive mats of larger diatoms (particularly *Gomphoneis* and *Cymbella*) and of the filamentous green algae *Ulothrix zonata* or *Spirogyra* can develop during periods of stable flow in the braids with stable sediment. Mats dominated by the red alga *Audouinella hermannii* can also occur (authors' unpublished data) (Fig. 15.11).

Regulated rivers

Rivers regulated by mainstream dams have unusual hydrological disturbance regimes that tend to encourage the development of high-biomass communities of diatoms, cyanobacteria and filamentous green algae. Such communities have been recorded below control structures in the Ohau River in South Canterbury (Biggs and Hickey 1994), the Waiau River in Southland (authors' unpublished data), and the Moawhango River in the central North Island (authors' unpublished data). Conditions induced by river regulation include a reduction in bed sediment supply, which results in armouring of the river bed, a reduction in the frequency of small- and medium-sized floods, and enrichment of waters from upwelling groundwater during periods of very low flows.

During low flows, periphyton community composition in such systems may reflect oligotrophic (e.g., *Ulothrix zonata*) to mesotrophic habitats (e.g., *Oedogonium*, *Phormidium*). Thick mucilaginous slimes composed of *Gomphoneis*, *Cymbella*, and *Synedra* populations occur in some South Island regulated rivers during summer low flows (e.g., Biggs and Hickey 1994).

Nutrient supply regimes may also be important in some regulated rivers. Significant groundwater intrusion can occur in the residual channel (probably partly enhanced by the greater hydraulic head from water dammed

upstream). This groundwater is usually enriched and may stimulate periphyton growth rates. Indeed, a common feature of regulated, residual flows throughout the world is proliferations of filamentous periphyton (Lowe 1979; Biggs 1987, 1996).

Conversely, in rare situations, flows are stored for short periods (e.g., overnight) and much higher discharges (approximately double) are released during the day (e.g., in the Clutha River below the Roxburgh hydro impoundment). This creates a variable flow regime that may be quite destructive for downstream periphyton communities, particularly in the wetting/drying zone (Bergey 2000). However, the effects may sometimes be mitigated by increased bed stability. Indeed, if the slope of the bed is low, then the doubling of discharge may only have a minor increase (say < 20 %) in near-bed velocities (e.g., Lower Clutha River).

Wetland streams

Streams fed by wetlands generally have low gradients, with moderate to low water velocities and fine bed-sediments, at least in their upper reaches. These streams can be very low in nutrients and in some places are acidic and stained yellow-brown by tannins. This represents a unique environment for periphyton. Much of the periphyton community is expected to be episammic (associated with sands) and epiphytic with abundant desmids and Zygnemataceae (*Spirogyra*, *Zygnema*, *Mougeotia*) and *Microspora*. However, few wetland-fed streams have been surveyed in New Zealand, so little definitive information exists to confirm these predictions. Collier and Winterbourn (1990) examined diatom communities in two brown-water streams as part of a study of the effects of naturally high acidity on benthic communities and found that they differed from communities in neighbouring clear-water streams. The communities in the brown-water streams were dominated by the filamentous alga *Tribonema*, and diatoms, including *Eunotia* spp. Both these genera are typical of acidic waters. Several new diatom species have been discovered recently in brown-water streams on the West Coast and on Stewart Island (e.g., Sabbe *et al.* 2001).

Lakes

Periphyton communities in lakes have received scant attention in New Zealand compared to those in streams and rivers and to planktonic lake communities. The few published studies focussing on periphyton and epiphyton in large, oligotrophic lakes indicate that diatoms grow to depths of at least 30 m, and that shallower depths are colonised by a mixed community of Cyanobacteria, filamentous chlorophytes, and diatoms. These communities differ markedly in composition in different lakes.

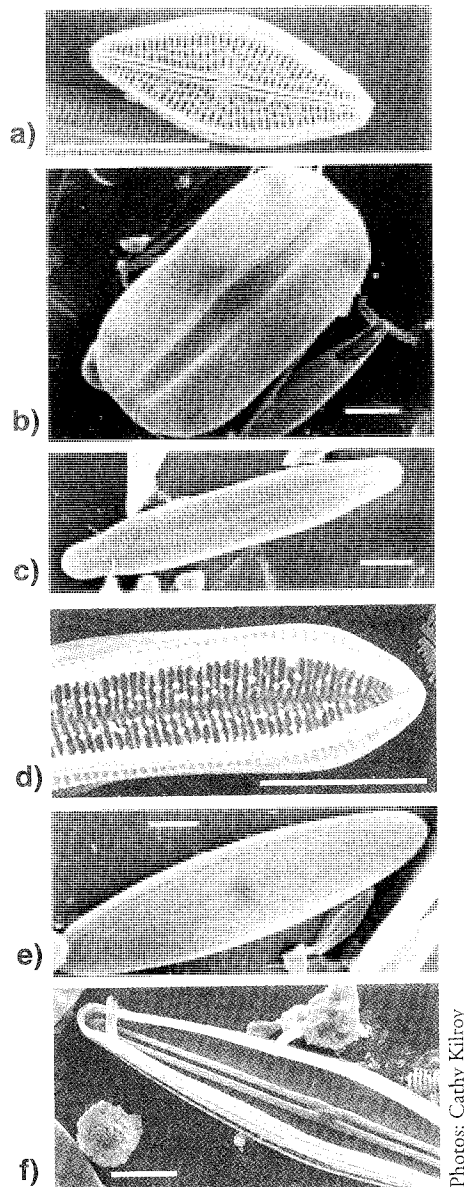
In Lake Coleridge, South Island, the same diatom taxa—*Achnantheidium minutissimum* and *Eunotia pectinalis*—dominated at depth, both as epiphytes on characean algae (Hawes and Schwarz 1996) and on artificial substrates (James *et al.* 2000). Shallower areas were characterised by *Bulbochaete*, in addition to common diatom taxa. Lake Taupo periphyton at depths of 20–30 m comprised *Aulacoseira granulata* (strictly, a planktonic taxon), *Fragilaria* spp., *Epithemia sorex* and *Rhopalodia novaezealandiae*, graduating to Cyanobacteria (*Tolypothrix*, *Aphanocapsa*) along with diatom taxa at mid-depths. In shallower waters the community also included chlorophytes (*Mougeotia* sp., *Spirogyra* sp.) (Hawes and Smith 1993). The taxa differences between lakes Coleridge and Taupo are likely to be related to water chemistry.

A recent survey characterised the diatoms of littoral periphyton from pristine South Island lakes (K. Vanhoutte pers. comm.). Community composition was strongly related to calcium concentration (\approx alkalinity). Lakes at high altitude with sparsely vegetated catchments tended to have very low levels of calcium (< 2.0 mg/l) and benthic diatom communities were dominated by acidophilous taxa including *Brachysira*, *Eunotia*, *Frustulia* and *Actinella*. Lower altitude lakes with higher alkalinities were dominated by *Achnantheidium* spp., and a variety of fragilarioid species.

Wetlands

Wetlands of all types in New Zealand share an almost complete lack of published information on their periphyton communities, apart from a treatment of the desmids (Croasdale and Flint 1986, 1988; Croasdale *et al.* 1994) and some early taxonomic work (Norstedt 1888; Skuja 1976). The latter work (Skuja 1976) was posthumously published and based on cores of benthic material from 19 mires throughout North and South Islands. Over 450 taxa are described, but no environmental information is presented. All these works serve to show the importance and great diversity of desmids (Chlorophyta) as a component of benthic algae in wetlands, especially in acidic, oligotrophic bogs. Skuja (1976) also listed chlorophytes (mainly *Zygnema* spp., *Oedogonium* spp.), Cyanobacteria (e.g., *Cylindrospermum*, *Schizothrix*), and diatoms (mainly acidophilous taxa including species of *Frustulia*, *Eunotia*, *Pinnularia*, *Neidium* and *Stenopterobia*) (Fig. 15.13).

More recently, a survey of diatoms from wetland surface sediments on the West Coast, South Island, showed that certain diatom taxa were good indicators of pristine, oligotrophic areas, particularly *Stenopterobia* spp. Other diatom species were indicative of disturbed, more enriched sites (e.g., *Encyonema* spp.) (P. Gerbeaux, pers. comm., see chapter 28). Benthic diatom communities from 20 sites in



Photos: Cathy Kilroy

Figure 15.13 Scanning electron micrographs of benthic diatoms typically found in acidic conditions.

(a) *Brachysira* sp.; (b) *Eunophora oregonica*; (c) *Pinnularia* sp.; (d) *Stenopterobia* (end of valve); (e) *Neidium* sp.; (f) *Frustulia* sp. Scale bars: 10 μm .

a subalpine wetland complex near Arthur's Pass were very strongly related to a gradient of pH, with conductivity and water colour (due to humic acids) also important. A single, small wetland can include a relatively wide range of water chemistry conditions, most likely as a result of differences in water sources—ombrotrophic (rain-fed) to minerotrophic (groundwater-fed). Thus periphyton taxonomic diversity is potentially high. Overall, although the diatom communities in these wetland sites were typical of a global "peat-bog flora" at the genus level, many species could not be identified from published flora, and probably represent undescribed taxa (Kilroy *et al.*, in press).

A further feature noted in this survey was periphyton in the form of thick cyanobacterial mats in shallow wetland pools. Such mats occur in similar alpine to subalpine wetland areas elsewhere in New Zealand (authors' observations) and resemble those reported from Antarctica, in that they comprise an upper layer of brown-pigmented Cyanobacteria, and a bright green underlayer that contains most of the chlorophyll *a* (Hawes and Schwarz 1999). The brown-pigmented upper layer appears to provide protection for the photosynthetic layer from high light intensities, including UV irradiation. Such mats may be able to form in these environments because grazing invertebrates, which would otherwise constantly disrupt cyanobacterial growth, never attain high densities due to extreme diurnal and seasonal temperature ranges and low pH. Interestingly, these stable mats sometimes harbour dense populations of the diatom *Eunophora*, which is endemic to New Zealand and Tasmania (Fig. 15.13).

Another type of periphytic alga commonly seen in higher altitude wetland areas is *Cylindrocapsa*. This filamentous chlorophyte forms fluffy red masses in pools and damp areas (authors' observations), the red colour being due to pigments in the reproductive structures in this genus.

Tarns

Tarns are small lakes formed in depressions left following glaciation, often with no defined inlet or outlet. Since peat-based wetlands can form in the same type of depressions, sometimes the division between these two types of water body becomes blurred. Most tarns are located at medium to high altitudes, where they are a characteristic feature of the New Zealand mountain landscape. Recent GIS estimates put the number of mapped South Island water bodies of less than 1 ha and lying at > 600 m asl at almost 7000 (H. Hurren pers. comm.). There are likely to be many more pools that are too small to be mapped. As with wetlands, information on the periphyton communities of tarns is sparse. Burns *et al.* (1984) documented the algal flora to genus level in "morainic ponds" near Lakes Tekapo and Ohau. The only other work on periphyton in tarns appears to have been studies of the desmid flora (e.g., Flint and Williamson 1999). In 2001, benthic algae samples were collected from the littoral zone of 50 tarns and pools with area <0.001 ha located mostly in pristine and/or high-altitude areas (K. Vanhoutte, pers. comm.). Diatoms made up a large proportion of the periphyton, and diatom community composition was closely related to a gradient of pH, which was (with a few exceptions) slightly to moderately acidic, with typical acidophilic genera dominating. It was notable that the genus *Eunophora* occurred at almost all the sites sampled, though often in low concentrations.

Unusual and rare periphyton taxa

Knowledge of the taxonomic composition and distribution of periphyton taxa in New Zealand rivers is still far from complete. For example, the branched filamentous red alga *Audouinella bermannii* was reported from New Zealand for the first time in 1987 (Biggs and Price 1987) (Fig. 15.11). At one of the six sites where it was found (Lower Clutha River below the Roxburgh hydropower station), a more extensive survey during very low flows revealed that this species was the dominant taxon covering most of the bedrock (Biggs and Shand 1987). More extensive collecting has since identified *A. bermannii* as one of our most common taxa in cobble-boulder bed streams (Biggs 1990). Similarly, *Coleodesmium* (Cyanobacteria) was recorded for the first time in New Zealand only in 1996 (S. Moore and P. Broady, unpublished data). This genus grows in tiny tufts attached to rocks, usually in cool-water streams.

Recent collecting and taxonomic research on periphyton from still-water habitats in high-altitude areas has identified a number of new diatom genera and species. For example, the genus *Eunophora* was first described from high-altitude tarns and pools mainly in Tasmania, but with rare examples from New Zealand (Vyverman *et al.* 1998) (Fig. 15.13). Since then, at least two species have been found to be widely distributed in pristine habitats in New Zealand and in some cases can dominate the periphyton (authors' observations). Curiously, one of these species was described in 1881 from fossil material at Arthur's Pass as *Amphora berggrenii* (Cleve 1881), but it took 115 years to find live specimens. A new species of the rare genus *Actinella* has been found dominating the periphyton in lowland West Coast streams and alpine streams (Sabbe *et al.* 2001). Two new species of *Fragilariforma* have been described from alpine streams and tarns, where they can form a major component of the periphyton (Kilroy *et al.* 2003).

Further surveys in other remote regions and unusual or extreme habitats are yielding more new or rare populations or communities. For example, new records of desmid species are still being documented from wetland habitats (Flint and Williamson 1999; P. Broady pers. comm.). Extensive growths of the large, branched, filamentous red alga *Batrachospermum* (rarely reported in previous studies) have been noted, in stable West Coast streams (Entwisle and Foard 1997).

Some groups of algae that occur in periphyton are still poorly known. These include a range of small species of green algae, yellow-green algae, and Cyanobacteria.

Polluted systems

Of all the human activities potentially affecting stream periphyton growth, and perhaps initiating proliferations,

point-source nutrient enrichment is probably the easiest to control. Most wastes receive primary and secondary treatment before being discharged to open water-courses. The low-molecular-weight organics that could stimulate sewage fungus growth are generally removed, along with large quantities of inorganic phosphorus and nitrogen. However, depending on the receiving environment, there may be sufficient residual nitrogen and phosphorus in the effluent to enhance peak biomass levels at certain times of the year. Less easy to control, and more prevalent in New Zealand, is the enriching effect of land development.

Effects of intensified land use

The conversion of scrub and forest to pasture or hard urban surfaces increases runoff rates and the magnitude of floods. This is likely to have significant effects on periphyton, though it is difficult to separate these effects from other effects of changes in land use such as increased sediment loads. If the frequency of large floods (i.e., $>7 \times$ the medium flow) increases much beyond about 15 per year, then a low to average periphyton biomass should be expected (Biggs 1995; Clausen and Biggs 1997; Biggs *et al.* 1999).

Changes in land use can result in increased bank destabilisation and increased suspended and bedload sediment (through road construction, etc.). A recent study found that losses of periphyton biomass were much greater with movement of bed sands than without (Francoeur and Biggs in press).

The effects on periphyton of enrichment from agricultural development may become evident with as little as 20% of the catchment in intensive pasture, and maximum effects may be seen after approximately 40% of the land is converted (Biggs 1995). This non-linearity of effect is partly because periphyton communities are highly sensitive to even small increases in nutrients. A common misconception by agricultural scientists has been that the losses of nutrients from pasture lands and top-dressing are usually insignificant. While this may be so in terms of pasture and crop growth requirements, even small losses can represent huge increases in the supply rates to stream periphyton. Where streams flow over alluvial plains, there can be major inputs of enriched groundwater. Elevated nutrient concentrations may originate from upstream farming activities that are far from the stream channel.

In general, there is no simple rule for evaluating or controlling the effects of land development on periphyton growth, as the degree of nutrient loss to streams depends on many different factors, such as soil type, use of irrigation, land relief, the extent of shallow groundwater intrusion into the streams, and riparian vegetation (McDowall *et al.* 2003). In addition, natural enrichment may occur in areas with andesitic volcanic, Tertiary marine mudstone/sandstone, and limestone bedrock (Biggs 1995).

Organic contamination

Organic contamination of New Zealand's freshwaters (for example, from freezing works and abattoirs) has declined over the last 30 years, with the result that there are now few examples of periphyton communities indicative of organically polluted waters. Where organic enrichment does occur, a shift in communities is often observed to those dominated by heterotrophic diatoms (e.g., *Nitzschia* spp.), filamentous green algae such as *Cladophora* spp. and *Stigeoclonium tenue*, and the filamentous Cyanobacterium *Lyngbya* spp. As the concentration of labile organics increases, the periphyton mat usually becomes progressively dominated by heterotrophic filamentous bacteria such as *Sphaerotilus natans* and *Zoogloea* sp. (often termed "sewage fungus") (Biggs 1989). Under high concentrations of organic contamination, the streambed can become smothered with these sewage fungus growths that give the appearance of brownish cotton wool. These growths obliterate habitat for benthic invertebrates and usually smell foul.

The shift in dominance from algae to non-photosynthetic bacteria in the periphyton community can be measured simply as the ratio of mat ash-free dry mass to chlorophyll *a* concentration. This is termed the Autotrophic Index (Weber 1973) and values > 400 can be taken as indicative of organically degraded conditions. Biggs (1989) used this approach, together with artificial substrate sampling, to illustrate the effects on periphyton of increased treatment, and then full diversion, of a freezing-works discharge to the spring-fed South Branch of the Waimakariri River, Canterbury. The heterotrophic index was usually < 100 at the control site upstream of the discharge (where communities were dominated by the diatoms *Gomphoneis minuta* var. *cassieae* and *Cymbella kappii*) and rose to > 2,000 downstream of the discharge (communities dominated by *Sphaerotilus natans* and *Zoogloea* sp.). Following treatment and diversion the Index dropped to 200–400, similar to the values recorded at the upstream control site following the diversion.

Mining

Mining can generate many types of pollutants, depending on the nature and intensity of the activity and on the hydrological links to surface waters. In New Zealand gravel mining and alluvial gold mining in river beds are likely to have the most impact on periphyton communities. The effects of these activities mainly occur through increased siltation of the mat, with implications for higher ecosystem processes.

While many periphyton taxa are well adapted to grow in environments with light to moderate siltation, such as glacier-fed rivers in the South Island (e.g., the stalked diatom *Gomphoneis minuta* var. *cassieae* and the filamentous

green alga *Ulothrix zonata*; Biggs and Price 1987; Graham 1990), high concentrations of suspended solids and associated silt accumulation can influence both the composition, metabolism and palatability to invertebrates of the periphyton mat. For example, in West Coast streams where placer gold was being mined, Davies-Colley *et al.* (1992) reported that the enhanced concentrations of clay particles greatly reduced light penetration into the streams, which in turn reduced benthic primary productivity, periphyton biomass and reduced the organic content of the mat, thus lowering its quality as food for invertebrates. James *et al.* (2000) found that siltation of the periphyton mat on macrophytes in Lake Coleridge, a South Island glacial lake, resulted in an increase in the grazing rate (probably to compensate for lower food quality) and food assimilation by the snail *Potamopyrgus antipodarum*.

There are many old coal mines in Westland with very acidic drainage (pH < 3.0). These streams are often dominated by the filamentous green alga *Klebsormidium* cf. *rivulare*, a species apparently distributed world-wide in this habitat, and a species of *Euglena*. The amount of growth can be considerable, possibly because most invertebrates are excluded by the acid conditions (P. Novis pers. comm.).

Twelve years of monitoring periphyton growth at sites above and below a waste-water discharge from a gold mining operation on the Coromandel Peninsula showed that periphyton biomass and community composition were not usually affected by the discharge. Exceptions occurred in the early years of the programme, when large summer growths of green filamentous algae occurred downstream of the discharge, indicating a nutrient effect, which could potentially have had flow-on effects to invertebrate and fish communities. A further interesting observation was the occurrence on one sampling occasion of deformed specimens of the large, common diatom species *Synedra ulna*, downstream of the discharge (authors' unpublished observations). Deformation of diatoms has been documented as an effect of pollution by heavy metals (McFarland *et al.* 1997). The monitoring continued after mining operations ceased and, overall, the programme showed that the discharges had been managed such that lasting effects on the periphyton community were avoided.

SUMMARY

Most work on periphyton in New Zealand has been undertaken on stream and river communities. Research to date has established the main factors controlling periphyton biomass accrual and loss, with flood disturbance frequency, nutrient supply and invertebrate grazing having the most influence. The effects of human activities on stream periphyton generally relate to flow regime changes, nutrient enhancement and changes in

sediment supply. Taxonomically, the communities comprise largely cosmopolitan species, or closely related taxa, though more unusual populations may be found in more remote or pristine streams. Periphyton communities in lakes and wetlands are less well studied, though the processes controlling their growth and biomass are likely to be similar to those in rivers. High-altitude and pristine small lakes and wetlands appear to support very diverse periphyton communities, with a higher proportion of unusual and possibly endemic taxa. There is scope for considerable further study on periphyton communities in New Zealand. Gaps in knowledge highlighted in this chapter are the composition and role of periphyton in lakes and wetlands and their associated streams, and in unimpacted lowland streams. Another is the detailed taxonomy of some components of the flora (e.g., green filamentous algae, unicellular and colonial green and yellow-green algae, some groups of Cyanobacteria, and diatoms in certain habitats).

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Chapter 14

Aquatic and riparian vegetation of rivers and streams

Paula Reeves, Kevin Collier and Alastair Suren

INTRODUCTION

Aquatic and riparian vegetation influence stream and river ecosystems—the amount and composition of vegetation within a river or stream and their associated riparian corridors can substantially affect water velocities (both flood flow and baseflow), habitats, nutrient dynamics, water clarity and quantity, and ultimately the

abundance and diversity of other aquatic life. Riparian vegetation is particularly important in rural and urban areas, where it may be the only visible remnant of the original land cover (Fig. 14.1).

In the last 15–20 years, research on the functioning of riparian and aquatic vegetation of streams and rivers has been used in management efforts to improve the quality



Figure 14.1 The riparian vegetation along the Oratia and Opanuku streams contribute significantly to the natural landscape present in Henderson, Waitakere City.

Photo: copyright Geosmart.

of water and fluvial habitats. These efforts, together with provisions in the Resource Management Act 1991 to protect the natural character of rivers and their margins and safeguard the life-supporting capacity of waterways, has been matched by a rising commitment by a range of agencies and public interest groups to embark on stream restoration programmes. These programmes are strongly focused on re-establishing riparian vegetation, with the goal of improving water quality and in-stream native biodiversity.

This chapter provides an overview of the physical processes that determine the distribution of aquatic and riparian plant communities, and reviews the recent literature on the functional roles of aquatic and riparian plants, and the key principles and approaches involved in restoring these vital components of stream and river corridors.

Riparian vegetation

The word riparian comes from the Latin “*riparius*” meaning “of or belonging to the banks of a river”. In modern lexicon, it refers to the biotic communities on the shores of streams, lakes and wetlands (Howard-Williams 1991; Naiman and Decamps 1997). These land-water ecotones are characterised by sharp environmental gradients relative to adjacent ecosystems, leading to species zonation and high biodiversity (Howard-Williams 1991). Riparian vegetation can strongly influence ecosystem function through:

- providing shade that helps regulate water temperature and in-stream plant growth,
- filtering rainfall and contaminants from hill-slope runoff,
- assisting in nutrient uptake and cycling from terrestrial ecosystems,
- reducing sediment inputs from bank erosion, and
- delivering inputs of leaf litter, dissolved organic matter and wood that provide food and habitat for aquatic life, influence channel morphology, and attenuate light reaching the streambed.

These functions are illustrated in Figure 14.2.

Before humans colonised New Zealand, most life in small to medium-sized streams in lowland and hill-country settings was under forest shade (Leathwick *et al.* 2003). Litter inputs would have been higher than today in these small streams, providing both food and habitat for invertebrates and fish. The larger rivers, however, would have been open to direct sunlight, and riparian vegetation would have played a relatively minor role in regulating

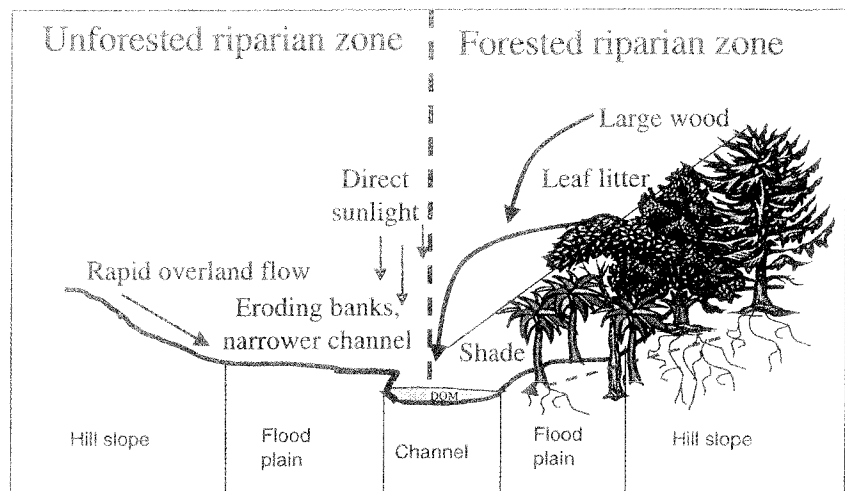


Figure 14.2 Cross-section of a riparian zone illustrating the interactions between the riparian vegetation and the stream.

aquatic processes. Instead, autotrophic processes would likely have driven aquatic production in these larger rivers, although extensive interactions with floodplain vegetation would have played an important role in ecosystem structure and function. Over the last 150 years, with conversion to agriculture, there have been dramatic changes in New Zealand’s vegetation cover, especially on the lowlands, where only 15% of the original forest cover remains. Moreover, many rivers have lost their riparian-floodplain linkages through flood control works that have restricted flood waters to narrow channels. Recent increases in the extent of production forestry and conservation planting may have reversed the direction of this change in some areas, as many of the smaller streams are once again becoming shaded.

Aquatic vegetation

Aquatic vegetation is an all-encompassing term for a vast array of plant species adapted to spending at least part of their life cycle either submerged or partly submerged in water. Several synonyms exist, including macrophytes, hydrophytes and waterplants. The most commonly used term is macrophyte, which refers to aquatic vegetation that can be seen by the naked eye and includes macro-algae (charophytes), mosses and liverworts (bryophytes), ferns and angiosperms (Wetzel 2001). Each of these groups has a very distinctive morphology (Fig. 14.3) and reproductive life cycle. The macro-algae have very large cells—these plants lack conductive tissue, absorb nutrients directly from the water column and are attached to the substrate by small rhizoids that penetrate into soft sediments. Macro-algae are usually restricted to slow-flowing lowland streams and lakes. Bryophytes refer to mosses and liverworts, both of which can be especially common in small, highly turbulent headwater streams where bedrock and boulders are common. They can also be found at great depth (50 m or more) in several clear-water glacial lakes in

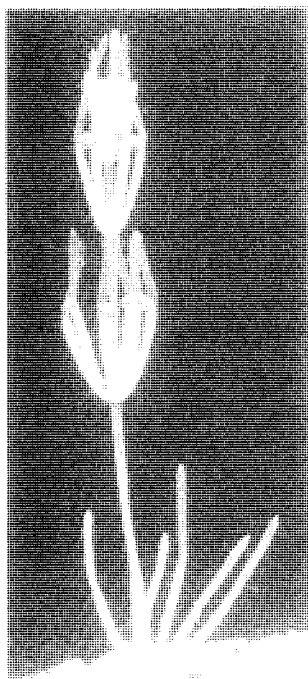


Photo: Rohan Wells

A



Photo: Alastair Suren

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Photo: Paul Champion

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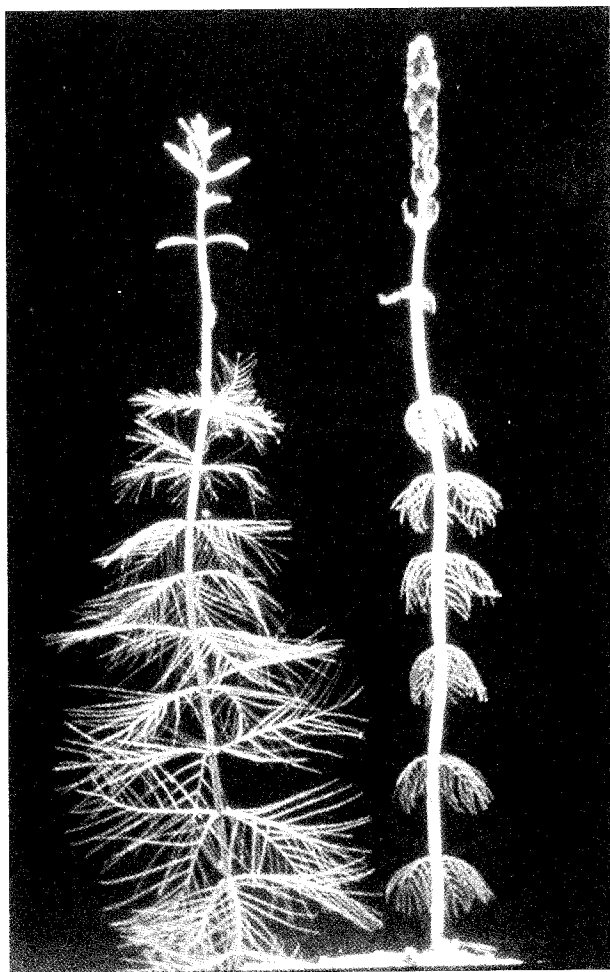


Photo: John Clayton

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Figure 14.3 The distinct morphologies of different aquatic plant groups.

- A. macro-algae (*Chara australis*)
- B. bryophytes (*Anthoceros* spp.)
- C. aquatic ferns (*Azolla rubra*)
- D. angiosperms (*Myriophyllum propinquum* and *M. triphyllum*).

New Zealand (see Chapter 24). These plants have a diverse range of morphologies, ranging from the simple plate-like thalloid plants of some liverworts, to more complex plants with specialised leaves and stems. Bryophytes lack conductive tissue and absorb all their nutrients from the surrounding water column. They are also firmly attached to bedrock or boulder substrates by small rhizoids. Aquatic ferns (e.g., *Azolla*) and angiosperms represent the rooted or free-floating plants that have stems and leaves, and have conducting tissues to transport nutrients from the roots to the shoots. Rooted angiosperms are found mostly growing in soft-bottomed streams or lakes, and like most other aquatic plants, are absent from gravel-bed rivers where the substrate is unstable. This chapter focuses on these macroscopic components of aquatic vegetation. Microscopic aquatic vegetation (e.g., micro-algae) is covered in Chapters 15 and 24.

The aquatic flora of New Zealand's streams and rivers has changed considerably since de-forestation began around 900 A.D. (Wardle 1991). Up until then the flora in streams would have been confined to species that are strongly adapted to shade, such as many bryophytes (Howard-Williams *et al.* 1987), and in the few larger rivers where there were stable substrates and moderate flows the flora would have been dominated by milfoils (*Myriophyllum* spp.), pondweeds (*Potamogeton* spp.) and some charophytes (e.g., *Nitella hookeri*). Deforestation would have modified streams and rivers considerably, increasing light penetration as riparian vegetation was removed, and substantially increasing sediment inputs, creating more silt substrates. These conditions would have facilitated the successful establishment of a number of exotic aquatic plants introduced into New Zealand following European settlement. These conditions, combined with the competitive traits of many exotic species, have led to current conditions, in which most streams and rivers in deforested catchments support often dense growths of exotic species (see Chapter 28 for a discussion of the effects of exotics, their competitive abilities and management options).

VEGETATION DISTRIBUTION

Riparian vegetation

The lateral extent of riparian vegetation can be difficult to delineate because of the mosaic of landforms and environments that these plants occupy. Riparian vegetation is generally considered to grow in edge habitats between low and high water marks, and is influenced by the portion of the terrestrial landscape affected by elevated water tables or flooding, and the ability of the soil to hold water (Naiman and Decamps 1997). Vegetation outside this area can be considered part of the riparian zone if, for example,

it contributes organic matter (e.g., wood from fallen trees) to the floodplain or channel, or provides habitat for species associated with the adjacent aquatic habitat. For example, the river specialist blue duck (*Hymenolaimus malacorhynchos*) is often associated with riparian cover and can nest, sometimes amongst vegetation, up to 30 m away from the river channel (Williams 1991).

The width of the riparian zone and its functional role (e.g., shading) varies with stream channel size, position within the drainage network and the hydrological regime (Naiman and Decamps 1997). For example, small headwater streams may have narrow riparian zones because they are readily shaded by a single row of trees, and flood flows do not extend far up the steep valley sides. In mid-sized streams, the riparian zone is typically represented by a wider band of vegetation whose composition is determined by long-term channel dynamics and the annual flood-flow regime. Large rivers with intact floodplain connections are characterised by extensive, well-developed and physically complex plant communities that may experience flood inundation, channel migration and variations in soil moisture (Naiman and Decamps 1997). In terms of hydrological regime, riparian zones alongside spring-fed or lake-fed streams may be narrower than those of streams fed by run-off because they experience relatively low levels of flood inundation.

The extent and composition of riparian zone plant communities is controlled by physical factors at a range of scales. Variations in large-scale factors such as rainfall, temperature and slope influence the type of riparian plant community that can potentially become established, while smaller scale longitudinal or altitudinal variations can influence the distribution of plants within catchments. Finally, small-scale topographic factors or soil moisture gradients operate on the scale of individual plants. The complex interaction of these factors determines the composition of riparian vegetation occurring at particular sites.

Large-scale factors

The Land Environments of New Zealand (LENZ) classification has been developed using climate, landscape and soil variables as predictors of original forest cover (Leathwick 2001; Leathwick *et al.* 2003). Climatic factors such as temperature, solar radiation, and water supply and demand vary around the country, and this correspondingly affects the types of trees that can grow in riparian areas. For example, species such as towai (*Weinmannia silvicola*), kauri (*Agathis australis*) and taraire (*Beilschmiedia tarairi*) occur only in regions of New Zealand with high solar radiation, whereas southern species such as rimu (*Dacrydium cupressinum*), kamahi (*Weinmannia racemosa*) and beech (*Nothofagus* spp.) occur in regions with low solar radiation. To some extent the spatial patterns identified in

the LENZ classification reflect the location of riparian zones that are often located on recent soils deposited alongside river channels. Many of these recent soils in productive farming areas have been predominantly developed for pasture, and this has significantly affected riparian function throughout New Zealand's lowland and hill-country streams.

Longitudinal factors

Air temperatures and other key factors affecting tree growth also vary with altitude, so riparian vegetation composition changes as streams and rivers flow from the mountains to the sea. The upper altitudinal limit of New Zealand trees occurs where maximum daily summer air temperatures are around 11°C (Leathwick *et al.* 2003). In Fiordland, for example, rimu and miro (*Prumnopitys ferruginea*) are most abundant at low elevations, kamahi is abundant at low to middle elevations, and silver beech (*Nothofagus menziesii*) and southern rata (*Metrosideros umbellata*) are dominant towards the treeline. Above the treeline, subalpine scrub and tussock grasslands dominate. Given the small catchment areas and narrow streams at this altitude, it is likely that scrub and tussocks provide many of the same functions as riparian trees that grow alongside larger streams at low elevation.

Other factors that change longitudinally down catchments, such as valley geomorphic type and biogeochemical processes, can also influence the composition, diversity, productivity and function of riparian vegetation (Harris 1988; Tabacchi *et al.* 1998; Quinn *et al.* 2001). In addition, disturbance and inundation regimes of riparian areas along river margins change downstream. Headwater riparian areas typically experience greater hydraulic stress due to frequent flood flows and high water velocities, but inundation here is less extensive than in lowland areas due to the constrained nature of their valleys. Where floodplain connections are intact and the valley is unconstrained, aquatic and terrestrial ecosystems can interact substantially during floods, providing foraging opportunities for some species such as eels in riparian areas of lowland rivers (Jellyman 1989). Also, in the lower parts of catchments, the penetration of saline waters inland during high tides can affect the types of vegetation that can grow in riparian areas. For example, increased saline penetration up the Heathcote River following flood control works caused riparian willows to die off, leading to increased bank erosion (Roper-Lindsay 1994).

Local factors

At a smaller scale, the slope, micro-topography and aspect of riparian areas can influence plant community composition through its effects on soils, light levels, soil redox potential and microclimate. Soils at the bases of slopes, where hill-country streams often occur, and on alluvial terraces further down the drainage network, are generally deep because of the accumulation of material transported by gravity and water. Although these soils may have high nutrient levels, plant growth can be influenced by poor drainage and localised deoxygenation. Tree species such as kahikatea (*Dacrycarpus dacrydioides*) and pukatea (*Laurelia novae-Zelandiae*) are generally more abundant on flat sites with poor drainage, while a larger number of species are most common on steep sites, including tawa (*Beilschiedia tawa*) and kamahi. Anthropogenic disturbances, such as intensive cattle grazing, also influence riparian vegetation.

Adapting to riparian environments

Active channels and periodically inundated floodplains are harsh environments for the establishment of plants. Naiman and Decamps (1997) recognised several broad strategies used by plants to survive in riparian areas: (1) invaders can produce large numbers of propagules that colonise alluvial substrates, e.g., *Leptospermum scoparium* and *Kunzea ericoides*; (2) endurers can re-sprout after breakage or burial, e.g., many of the introduced willow species and *Leptospermum scoparium*, and (3) resisters can withstand floods, e.g., kahikatea, with its stem buttresses (Fig. 14.4).



Figure 14.4 The buttressed trunks of kahikatea (*Dacrycarpus dacrydioides*).

Photo: Tracey Edwards

Aquatic vegetation

Spatial distribution

Most streams and rivers in New Zealand have sufficient oxygen, nutrients and carbon to meet the requirements for at least some macrophytic growth. Once these fundamental requirements have been met, the factors that principally determine the presence or absence of aquatic plants in streams and rivers are light, substrate stability and flow regime (Sculthorpe 1967; Biggs 1996).

Light availability in water decreases with increasing water depth and may also be reduced by high concentrations of suspended inorganic particles (e.g. glacial silt, sediment from run-off, humic substances), and particulate and dissolved organic suspensoids (e.g., phytoplankton, diatoms, humic substances). Levels of organic suspensoids of algal origin are often positively correlated with nutrient concentrations that contribute to higher rates of phytoplankton productivity (Viner and White 1987). Other external factors that can reduce light reaching the water column are the bank height in relation to the channel width (deeply incised streams are often shaded by their stream banks), bank angle, and the alignment of the reach with respect to the sun (Collier *et al.* 1995a).

The interaction of aquatic plants with flow is complex, with both exerting influences on the other (e.g., Dodds and Biggs 2002). Flow strongly affects the substrate—finer sediments are found in slower-flowing waters and coarser substrates are more common in faster flows. In turn, substrate size and stability are major determinants of the species of aquatic plants present, with certain species favouring specific substrate types (Haslam 1978; Biggs 1996; Biggs *et al.* 2001). Bryophytes, for example, are usually found on stable stones and boulders, while submerged rooted aquatic plants are more common in the fine substrates (clays, silts and sands) that provide a better rooting medium.

Biomass declines as velocity increases (Fig. 14.5; Henriques 1987), and aquatic plants are generally not found where flow velocities are greater than 1 m/s (Chambers *et al.* 1991). Plant morphology and architecture (both below and above ground parts) determine the ability of a species to withstand hydraulic forces. Plants with high drag co-efficients (high branching and surface-to-volume ratios) and low anchoring strengths (e.g., *Apium nodiflorum*, *Ceratophyllum demersum* and *Nasturtium officinale*) are usually found growing only in areas with low velocities, whereas plants with a streamlined form and either deep or tangled roots or strong perennial rhizomes (e.g., *Glyceria fluitans* and *Ranunculus tricophyllus*) can withstand much greater velocities (Haslam 1978).

Like woody riparian plants, aquatic plants can influence stream flows by providing hydraulic resistance that contributes to flooding in lowland streams and rivers, so aquatic plants often must be removed from streams and

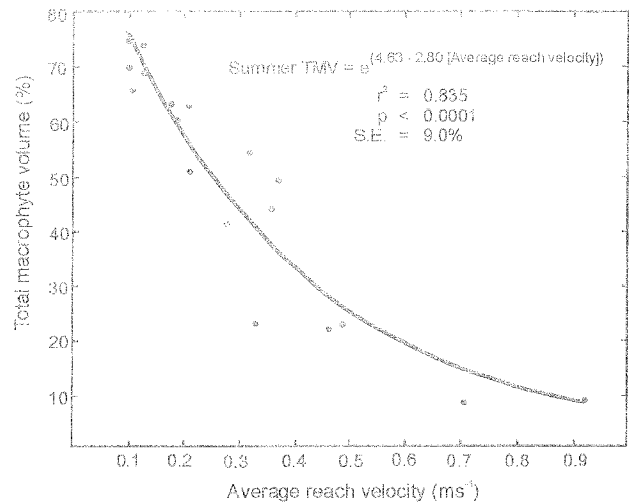


Figure 14.5 Total macrophyte volume (TMV = % cover times average plant height / average reach depth) versus average reach velocity of constant-flow streams on the east coast of the southern half of the South Island, New Zealand. (Adapted from Henriques 1987; re-printed with permission).

channels in flood-prone areas. Velocities can be reduced substantially and the reductions vary laterally, depending on the cross-sectional area occupied by the aquatic plants (Fig. 14.6) (Sand-Jensen and Mebus 1996; Champion and Tanner 2000). Champion and Tanner (2000) found that aquatic plants acted as semi-permeable dams during the summer months in a lowland Waikato stream, retarding flows and increasing stream depth so that it remained relatively constant all year. Even relatively small plants such as aquatic bryophytes can influence near-bed flow regimes. For example Nikora *et al.* (1998) showed how the moss *Fissidens rigidulus* reduced the average longitudinal velocity, Reynolds stress and total turbulence energy in a cobble-bed experimental flume.

Flow stability and the period between floods also determine the distribution of macrophyte communities, with constant flow velocities and long return-periods favouring macrophyte growth (Biggs 1996). In a survey of 15 South Island streams, the abundance, richness and diversity of aquatic plants were significantly correlated with flood frequency, and macrophyte vegetation was absent from streams with more than c. 13 flood disturbances per year in which the floods exceeded 7 times the median flow (Riis and Biggs 2003). However, flood disturbance is not the only variable influencing macrophyte distributions. For instance, Riis and Biggs (2001) adapted the disturbance frequency and resource supply model of Grime (1979) to determine the occurrence of common New Zealand macrophyte species. This habitat matrix model had four primary habitat groups based on the four extreme combinations of high and low frequency of disturbance and high and low resource supply (Table 14.1).

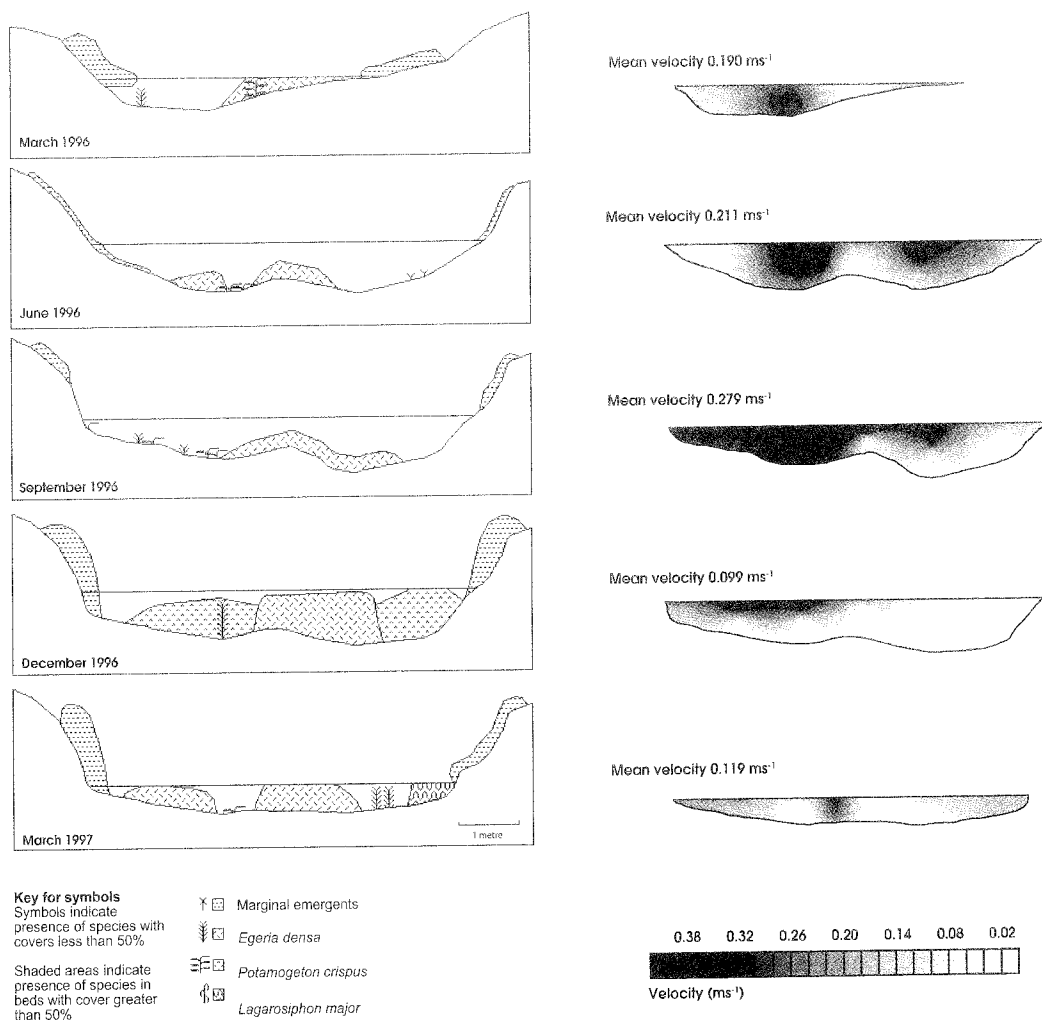


Figure 14.6 Distribution of aquatic plants and stream velocity in Whakapipi Stream, a spring-fed lowland tributary of the Waikato River. (Champion and Tanner 2000; reprinted with permission of Kluwer Academic Publishers).

Table 14.1 Species traits characteristic of the four life history strategies that can be attributed to plants: *C* – competitive; *R* – ruderal; *S* – stress-tolerant; *ST* – stunted (Grime 1979; Kautsky 1988).

Species trait	<i>C</i> -strategy	<i>R</i> -strategy	<i>S</i> -strategy	<i>ST</i> -strategy
Biomass growth	High	Fast	Slow	Slow
Stature	High		Low	Low
Morphology	Highly variable	Flexible stems; High root:shoot ratio	High root:shoot ratio	Low stature
Life-span	Long-lived	Short; fast coloniser	Long-lived	Long-lived

Analysis of species traits associated with 32 common New Zealand aquatic vascular plants showed that seven were allocated to the competitive or ruderal strategy (ruderal plants thrive in disturbed or waste ground), eight to either the ruderal or stress-tolerant strategy, and the remaining nine were allocated as either competitive (2 species), competitive-stress tolerant (3 species) or stress tolerant-stunted (4 species) (Riis and Biggs 2001). All of the ruderal strategists, with a high vegetative reproduction potential, were exotic species, suggesting that they could easily out-compete many native species at sites with relatively high disturbances. This finding was in sharp contrast to the finding that all stress-tolerant strategists were native species, and present only in low-disturbance lakes.

A second habitat template model describing the occurrence of aquatic plants in streams has recently been developed that considers both bed sediment stability and velocity disturbance (Biggs *et al.* 2001). This model (Fig. 14.7) is thought to better explain aquatic plant community structure in streams, as it recognises the fact that some plants are very sensitive to increases in velocity alone, while others are sensitive only to bed sediment movement. A recent study by Riis and Biggs (2003) confirmed this model, and emphasised that substrate movement was one of the major determining factors controlling macrophyte growth in streams.

Temperature fluctuations in aquatic environments are generally much less extreme than fluctuations in air temperature and consequently have a reduced influence on the distribution of aquatic vegetation. The majority of native aquatic plant species occur throughout the country, although the number of species decreases with increasing altitude (Wardle 1991). Many adventives also have widespread geographical distributions, with several highly invasive species, such as *Ceratophyllum demersum* and *Egeria densa*, yet to reach their maximum potential distributions within New Zealand.

Temporal patterns

Macrophyte biomass in streams frequently displays strong seasonal patterns. Biomass has been recorded as eight times greater in summer than winter in a lowland New Zealand stream (Collier *et al.* 1999), although 3-fold increases are more usual (e.g., Duggan *et al.* 2002; Riis *et al.* 2003). Biomass abundance in streams appears to be related to a combination of factors, including the phenological cycles of the dominant species (with the onset of senescence coinciding with declining water temperatures and day lengths shorter than 12 hours), competition between species for resources, and disturbance by floods (Champion and Tanner 2000).

FUNCTIONAL ROLES OF AQUATIC AND RIPARIAN PLANTS

Light, temperature and microclimate

Shade provided by riparian vegetation helps regulate water temperature and in-stream plant growth, particularly along streams where the overhead canopy is closed. As stream channels widen, the influence of riparian vegetation shading decreases. In forested streams, lighting is typically less than 5% of ambient where the channel width is less than 6 m, but riparian vegetation has little shading effect in channels wider than 15–20 m, unless very large canopy trees like kahikatea are present (Davies-Colley and Quinn 1998). Forest structure also influences the spectra of light penetrating the canopy and eventually reaching aquatic habitats, affecting visual cues for animals and the growth form of plants (Endler 1993). Of particular importance with respect to current problems of ozone depletion is the role of riparian vegetation in ameliorating levels of UV-B radiation, which can adversely affect some aquatic biota (e.g., Wüben *et al.* 2001).

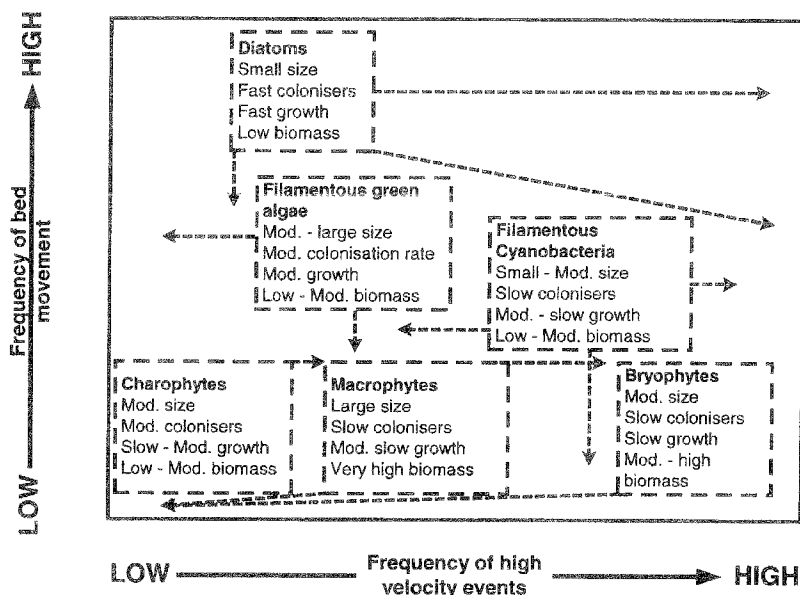


Figure 14.7 Conceptual habitat matrix for stream plant communities based on the disturbance agents of frequency of high velocity events, and frequency of bed movement. The theoretical location of each plant group is shown in the small boxes; arrows show possible extent of conditions that the different plant groups can tolerate (modified from Biggs *et al.* 2001).

Temperature has important influences on aquatic fauna, affecting metabolic rates and survival. Constant water temperatures above around 23–25°C are lethal to sensitive stoneflies and mayflies, but native fish tend to be more tolerant (Quinn *et al.* 1994; Richardson *et al.* 1994). Small streams are particularly susceptible to heating due to their shallow depth, and it is in these streams where riparian shade is likely to be most effective at regulating water temperature. Similarly, air temperatures are influenced by riparian vegetation and can affect the survival of sensitive adult stages of aquatic insects. For example, constant temperatures of 22–23°C over four days caused 50% mortality in adults of a stonefly species (Collier and Smith 2000), although they were more tolerant of diurnally fluctuating temperatures. Other aspects of riparian microclimate may also be important for aquatic species (e.g., high humidity for egg-hatching in some galaxiid fish).

Channel morphology and bank stability

Riparian vegetation influences channel morphology primarily by its effects on the stability of stream banks, sediment entrainment, and the volume of in-stream woody debris, which can influence hydraulics and sediment dynamics (Naiman and Decamps 1997; Baillie and Davies 2002). Conversion of riparian vegetation from forest to pasture has been shown to lead to channel narrowing in small, hill-country streams, due to the entrainment of sediment by bank grasses (Davies-Colley 1997). Conversely, shading of these pasture streams by riparian planting can be expected to lead to a period of increased sediment export as the stream channel readjusts to a morphology typical of shaded areas (Fig. 14.8).

The role of riparian vegetation in stabilising banks depends on the ability of vegetation to (1) reinforce banks through root networks, (2) provide a well-developed turf or a dense root system that protects against surface soil erosion, (3) pump out water from the soil, and provide macropores for drainage, and (4) buttress the toe of the stream bank, protecting it from shear failure (e.g., Abernethy and Rutherford 1999, 2000). These stabilising functions are influenced by a combination of factors, including the height of the stream banks relative to the depth of root penetration, bank angles, the erosive power of the stream under high flows, and whether the banks are protected by other features (e.g., boulders, bedrock or large woody debris). In New Zealand, exotic trees, particularly willows and poplars, have been used extensively to stabilise stream and river banks. Research by Phillips *et al.* (2001) indicates that, of the native species, ribbonwood (*Plagianthus regius*) and tutu (*Coriaria arborea*) have the most promising characteristics for stream bank stabilisation, although tutu is poisonous to stock.

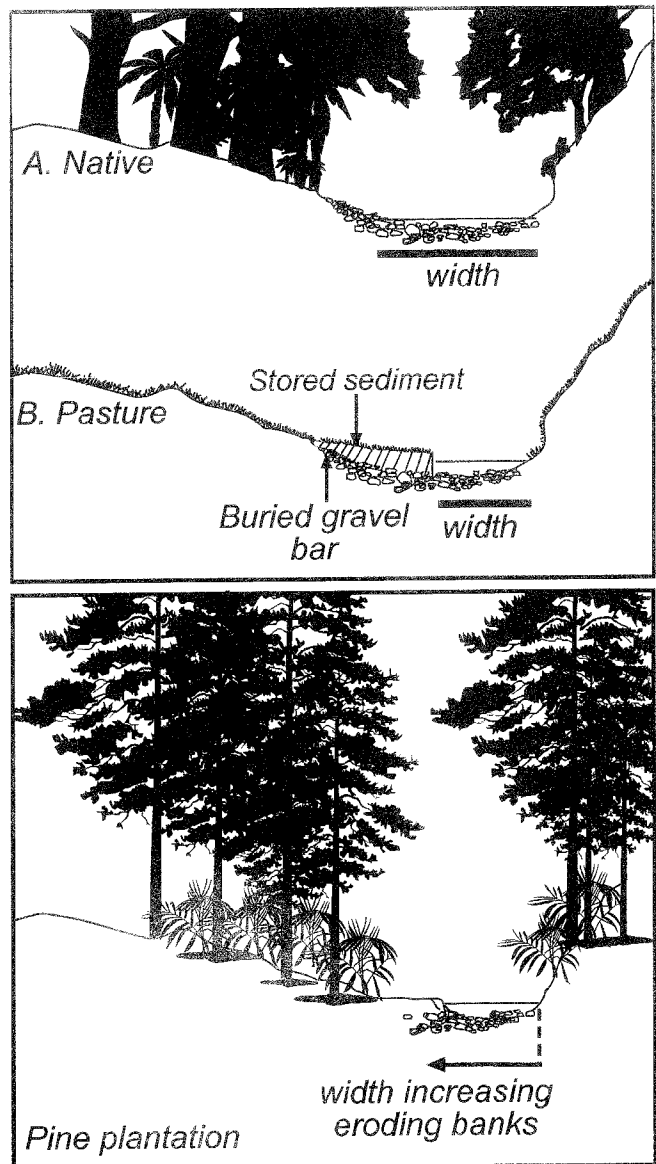


Figure 14.8 Conceptual model of the channel form of small streams at the same cross-section in different vegetation. Upper picture: A: steady-state cross-section in native forest; B: steady-state cross section in pasture; Lower picture: stream-bank recession in a pasture stream planted with pines following canopy closure. Sediment stored in the pasture streams is indicated in B. (Davies-Colley 1997; reprinted with permission).

Aquatic habitat

The role of aquatic vegetation in providing habitat for invertebrates has been extensively studied, for both macrophytes (e.g., Biggs and Malthus 1982; Rooke 1984) and bryophytes (e.g., Percival and Whitehead 1929; Suren 1991). In streams, these plants provide shelter from high currents (Gregg and Rose 1985; Sand-Jensen and Mebus 1996; Nikora *et al.* 1998), and increase the biomass of detritus and periphyton food (Suren 1992), and they can act as sites for egg deposition or pupation by aquatic insects. Invertebrate density has been shown to be much

greater on aquatic macrophytes than on the soft bottoms of lowland Waikato streams (Collier *et al.* 1999), but lower or similar where more heterogenous and stable stony substrate are available (Rooke 1984; Suren and Lake 1988). Invertebrate community composition can also differ between aquatic plants and other substratum types (Rooke 1984; Collier 1995; Collier *et al.* 1998; Collier 2004), but the evidence that invertebrate composition differs among macrophyte species is less clear. Biggs and Malthus (1982) and Marshall (1973) found little difference in community composition between different species, whereas Death (1991) found that different aquatic plant species did support different invertebrate communities.

In contrast to macrophytes, aquatic bryophytes have consistently been shown to support both higher invertebrate densities and communities that differ from those on bare substrata (e.g., Percival and Whitehead 1929; Suren 1991; Nolte 1991; Suren 1993; Linhart *et al.* 2002). The clearer response of invertebrate communities to bryophytes, as opposed to other aquatic plants, may reflect the hydraulically harsher environment typified by the steep, boulder headwater streams that bryophytes are found in. Floods, and associated substratum movement, are common in such streams, so the moderating influence of bryophytes as stable microhabitats may be more pronounced there than in the lowland settings where aquatic plants predominate.

The importance of aquatic vegetation for fish in streams is not as well studied as for invertebrates, however many fish are expected to use larger aquatic plants patches for cover. Studies of the microhabitat preferences of shortfin and longfin eels in four physically contrasting rivers have shown that plant cover was positively related to the abundance of large (> 500 mm) shortfin eels, and moderately large (300–399 mm) longfin eels (Jellyman *et al.* 2003). Such a preference for this habitat may reflect in part the ability of these large animals to hide within aquatic plant beds and prey on smaller fish and invertebrates. For example, Burnett (1952) found that eels living in stable streams with aquatic plants contained a much higher number of trout in their guts than eels living in gravel-bed streams without aquatic plant cover.

Aquatic food webs

There are four potential major energy sources for invertebrates in streams: detritus, epilithon (including both algae and fungal-bacterial “slimes”), aquatic plants, and other invertebrates (Winterbourn 2000; McIntosh 2000). Of these four energy sources, aquatic plants are relatively unimportant as a direct food source to most invertebrates, despite their common occurrence in streams, their often high rates of primary productivity, and the fact that many types of invertebrates dwell amongst them. Although some

stream invertebrates (e.g., nymphs of the aquatic moth *Nymphula*) consume living aquatic plants, most studies have shown that invertebrates generally avoid direct consumption (Mackay and Wiggins 1979; Otto 1983; Suren and Lake 1989), although they may consume the biofilms adhering to plant surfaces. Once aquatic plants senesce and decompose, however, they may be consumed by invertebrates as detritus (Smock and Stoneburner 1980; Suren and Lake 1989). The distinct seasonal growth of aquatic plants, with a winter decline in biomass (Riis *et al.* 2003), may mean that aquatic plants enter the detrital food chain during the winter months, and consequently may represent a carbon supplement for invertebrates during this time, if the decaying material is retained in the stream long enough to be processed.

Although invertebrates generally avoid eating live aquatic plants, some fish, and in particular grass carp (*Ctenopharyngodon idella*) consume considerable quantities of these plants. Recent trials assessing the value of grass carp as a control method for keeping agricultural drains clear of submerged aquatic vegetation have shown that these fish initially thrived and removed almost all the submerged aquatic vegetation from the lower portion of a drain after a four-month period (Wells *et al.* 2003).

Aquatic bryophytes are generally not consumed by invertebrates. For example, Suren and Winterbourn (1991) examined the gut contents of 23 invertebrate taxa that were found among bryophytes in two headwater alpine streams, and found that most had consumed algae and detritus.

Nutrient and contaminant removal

Both riparian and aquatic vegetation play important roles in removing nutrients (e.g., nitrogen and phosphorus) and other contaminants such as sediment, pesticides and pathogens. The key processes responsible for nutrient and contaminant removal in the riparian zone include denitrification, sediment trapping and nutrient uptake by riparian plants. Denitrification involves the microbial conversion of nitrate to nitrogen gas, via nitrite and nitrous oxide. Conditions favourable for denitrification are found in anoxic, waterlogged ground with organic-rich soils. Under these conditions up to 90% of nitrates can be removed (e.g., Cooper 1990), however the efficiency of removal depends on the rates of lateral subsurface flow, with higher rates lowering nitrate removal (Nguyen *et al.* 1999). High flow rates can also lead to scouring of sediments stored in the riparian zone, and once soils become saturated, water can be forced to the surface before running overland, so minimal nutrient removal takes place before the water enters waterways (Elliot and Sorrell 2002).

For sediment trapping and plant uptake of nutrients to be effective in the riparian zone, run-off must infiltrate the soil and move through this zone at slow speed and at

the depth of plant roots (Correll 1996). Optimal conditions would include flat to gently-sloping topography, permeable soils and the presence of a shallow confining layer to prevent groundwater flowing beneath the root zone (Phillips 1989; Smith 1989). Woody vegetation is generally assumed to be more effective in the uptake of nutrients in the riparian zone due to its deeper roots and greater biomass for nutrient storage (Correll 1996). Dense grass cover is the most effective riparian vegetation for trapping sediment (Dillaha *et al.* 1989). Guidelines for New Zealand waterways have been produced for determining the optimal width and effectiveness of riparian grass strips for removing sediment suspended in overland flow (Collier *et al.* 1995b).

In-stream aquatic vegetation has also been found to remove nutrients from streams. Howard-Williams and Pickmere (1999) found that the uptake of nitrate and dissolved reactive phosphorus by in-stream vegetation increased substantially following an increase in the abundance of aquatic plants after stock was fenced out of the riparian zone. However, after riparian shading reduced the amount of in-stream aquatic plants, nutrient uptake decreased to become negligible. In addition Wilcock *et al.* (2002) found that velocity strongly influenced nutrient retention, with greater removal of nitrate, ammonium and dissolved reactive phosphorus in a slow-flowing stream reach than in a swifter-moving reach.

RESTORING AQUATIC AND RIPARIAN VEGETATION

Effects of land use

Loss and degradation of aquatic and riparian vegetation occurs through a range of direct and indirect activities associated with different land uses. Direct land-use activities include the clearing of riparian land for pastoral, horticultural or urban development, forestry, intensive stock grazing, and dredging of streams and rivers to remove vegetation or modify the channel to increase hydraulic conductivity. All these activities decrease habitat heterogeneity and alter hydrology. Indirect activities occur more at the catchment-scale and include increases in run-off and in associated sediment and nutrient inputs. These land-use activities can deliver large amounts of suspended sediment to waterways, reducing light levels. In the riparian zone these activities may change soil type and increase fertility, creating conditions more favourable for invasive plant species.

Setting restoration goals

Restoration involves returning a site to some former (usually pre-human) condition, and includes the reinstatement of ecological processes so that the system becomes

self-regulating and integrated with the surrounding landscape (Middleton 1999). Commonly, "restoration" is referred to as "rehabilitation", as it is rarely feasible to return all biotic elements and ecosystem processes to a former state. Rehabilitation projects in heavily modified landscapes usually have modest goals, as some ecosystem processes that have often been lost (e.g., floodplain inundation), would be too costly or unacceptable to society to reinstate.

Whatever the scale of the restoration project, it is essential to formulate clear and measurable goals at the outset, so that management activities can be monitored, evaluated and amended if progress towards a goal is not occurring. Formulation of realistic goals requires understanding of the functioning and interaction between key factors, and knowledge of what the former state may have been. Reference sites can be useful in providing a template for past or desired conditions, provided they have similar physical and chemical characteristics to the proposed restoration site. Key factors have largely been described in the previous sections of this chapter, and operate at levels of both the catchment (e.g., climate, geology and geomorphology) and reach (e.g., current velocity, channel morphology, bank stability, water quality).



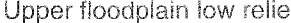







To assist in setting riparian restoration goals, Quinn *et al.* (2001) developed a classification system for identifying the potential functions of riparian areas in the Waikato Region; it recognises that riparian function varies with landform and stream size. The classification recognises 10 different riparian classes, based on riparian and stream morphological characteristics that influence biophysical functions of the riparian area. For each riparian class, an assessment was made of the potential relative importance of different riparian functions (Table 14.2). The table can be used to set restoration goals and prioritise management actions, when combined with the knowledge of the most suitable management actions to attain maximum potential function. For example, replanting riparian vegetation enhances many riparian functions, but if the goal is to reduce in-stream temperatures, then re-vegetation will be most effective along stream sections located in upper floodplains with low relief and V-shaped and U-shaped hill valleys.

Approaches to restoration

Passive restoration

Passive restoration involves ceasing activities that affect the stream corridor, such as fencing-out stock or preventing contaminated run-off from reaching the waterway. It is based on the theory of self-design, where the composition of plant communities is determined by environmental conditions and therefore, if these conditions can be restored, then so will the plant

Table 14.2 Potential relative importance of different riparian functions among riparian classes. The scale ranges from 0 to 5, where 0 means the function is inactive and 5 means it is very highly active. Modified from Quinn *et al.* 2001.

Riparian Class	Riparian Function									
	Stream bank stability	Filtering overland flow	Fish spawning/habitat/cover	Shade for stream temp.	Shade for in-stream plant control	Woody debris input	Plant nutrient uptake	Denitrification nitrogen control	Direct animal waste control	Downstream flood control
Lower floodplain 	3	1	5	1	0	3	2	2	1	5
Entrenched floodplain 	5	2	2	2	4	3	2	1	1	2
Upper floodplain low relief 	5	2	2	5	5	3	4	4	4	2
Upper floodplain high relief 	4	1	2	3	3	2	2	1	4	4
V-shaped entrenched 	5	4	2	3	5	4	3	1	2	2
V-shaped hill valley 	4	4	4	5	4	4	4	4	5	1
U-shaped hill valley 	4	4	4	5	4	4	4	3	5	2
Shallow V-shaped rolling 	4	4	2	3	3	5	4	4	3	2
Vegetated drain 	2	5	1	2	5	1	3	4	5	3
Headwater wetland 	1	5	1	1	0	0	5	5	4	5

communities (Mitsch and Jorgenson 1989). This approach can be very cost-effective, however, the ability of the system to return to some former state will depend on the degree to which it has been disturbed. It relies on the presence of viable seed sources, suitable environmental conditions for breaking dormancy, and a lack of invasive species in the vicinity, conditions that could be very difficult to meet in much of New Zealand's modified landscape. Riparian floodplain vegetation may regenerate naturally from either the seed bank or episodic flooding re-introducing seed or vegetative propagules from native forests further upstream. However the high abundance and competitive traits of many adventive riparian species, particularly willows (*Salix* spp.) and dense pasture grasses (e.g., *Glyceria maxima* and *Holcus lanatus*), frequently prevent the re-establishment of native species in riparian floodplains.

Passive restoration of native aquatic vegetation in running waters is probably a viable option only in small

streams where shade would be sufficient to favour the re-establishment of bryophytes or some of the native aquatics such as *Nitella hookeri* and *Potamogeton ochreatus* that will persist under a closed riparian canopy. In wider streams and in rivers, light levels will be higher, favouring the adventive flora. Seedbanks of native aquatic vegetation from lakes, even from degraded aquatic habitats, have been shown to be ecologically active (de Winton *et al.* 2000), but no comparable studies have been done for streams and rivers in New Zealand.

Active restoration

Without intervention, many native riparian and aquatic plant communities are unlikely to become re-established. Active restoration may involve a range of interventions, including removing weeds and other pests, re-planting vegetation, re-contouring stream channels and banks, de-channelising streams or even "daylighting" piped streams.

One of the most widely practised forms of active stream and river corridor restoration in New Zealand is replanting riparian areas to restore native vegetation. A number of riparian planting guides have been published by government agencies (e.g., Auckland Regional Council 2001; Environment Waikato 2002). Successful establishment of riparian vegetation requires planting species best suited to the microhabitats present in riparian areas, which are delimited by the variation in factors such as soil moisture, soil texture, soil fertility, salinity and flooding frequency.

“How wide should the planted area be?” is a common question posed when replanting riparian vegetation. The answer depends on the objective of the restoration—only a few metres may be needed to shade small streams, but much wider areas may be required to filter overland contaminants (Collier *et al.* 1995b). Creating native riparian plant communities that are sustainable (i.e., are resistant to weed invasion and have sufficient self-recruitment) can be facilitated by reducing edge effects that influence microclimate (sunlight exposure, wind exposure, and air and soil temperature and moisture), which strongly influence the structure and composition of plant communities (Murcia 1995). Two studies in the North Island of New Zealand have reported edge effects persisting at least 40–50 m into the forest from the forest edge (Young and Mitchell 1994; Davies-Colley *et al.* 2000), suggesting that the width of riparian plantings may need to be at least of this order at exposed sites.

Future restoration

Research over the last two decades has significantly advanced our understanding of aquatic and riparian vegetation in streams and rivers in New Zealand, and the environmental factors influencing their abundance and community composition. The rate of uptake of this research by management agencies, and the initiation of riparian restoration and pest control projects, often with community involvement, has been considerable. This work, coupled with improved predictive models and data from long-term studies and forecasts (e.g., Howard-Williams and Pickmere 1999; Collier *et al.* 2001), has provided opportunities to re-evaluate expectations regarding riparian restoration and aquatic plant management, and has highlighted several insights that should prove useful for future restoration efforts.

Firstly, in the absence of native vegetation, exotic plants can provide some of their functions, such as providing shade, inputs of organic matter, and in-stream habitat where benthic substrata are unsuitable. Although many exotic species can cause problems, such as the invasion of channels by willow roots and nuisance macrophytes, it may be desirable to maintain existing stands of non-nuisance exotic vegetation in modified environments. Similarly, it

may be better to limit proliferation of exotic macrophytes while maintaining in-stream function, such as by providing patchy shade with spaced riparian plantings alongside lowland streams (e.g., Collier 2004), rather than to undertake large-scale channel vegetation clearance.

Secondly, the timescales involved in the management or restoration of riparian vegetation may exceed the lifetime of those involved, emphasising the need to manage societal expectations of both the speed and trajectory of change (Collier *et al.* 2001). Recent work has indicated that in some situations, conditions may become worse before they get better, notably where channel widening occurs following the shading of previously open channels (Davies-Colley 1997). The time required for restoration may be affected by how close restored sites are to source areas containing desired species, highlighting the need for broader perspectives when planning restoration works.

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Chapter 13

Stream communities and ecosystem processes

Mike Winterbourn

INTRODUCTION

The current worldwide emphasis on diversity, at levels ranging from the gene to the community and ecosystem, has provided stream ecologists with a strong incentive to explain and understand the less-than-obvious relationships between biotic diversity and ecosystem function. It is well known that loss of biodiversity can impair ecosystem processes, but details of the effects of species loss and the generality of impacts across ecosystem types are poorly understood (Rosenfeld 2002). A major challenge for running-water ecologists, therefore, is to understand how different assemblages of organisms determine ecosystem processes under diverse environmental conditions.

Historically, studies of running-water communities and ecosystems have been undertaken by ecologists with different academic backgrounds and philosophies. Community ecologists tend to be concerned primarily with interactions among populations and species within a community—processes such as competition, predation and disturbance, and the effects of physical and chemical factors on the temporal and spatial structure of communities. In contrast, ecosystem ecologists are more concerned with functional attributes of biological systems, such as energy transformations and the cycling of nutrients and organic matter. Recently, however, these different approaches have begun to be brought together, particularly through multidisciplinary research programmes. Coordinated studies of the effects of introduced trout on populations, communities and ecosystem function in the Taieri River system, Otago, provide an excellent local example (Townsend 2003); the finding that trout can exert strong control over algal and invertebrate community structure and ecosystem function has important implications for river management.

In this chapter the nature of running-water communities and stream ecosystem function in New

Zealand is discussed, with particular emphasis on recent studies.

STREAM COMMUNITIES

Stream communities in New Zealand vary greatly in composition, depending on their geographical locality, physical and chemical conditions, catchment land use, and historical factors. In fact, it is probably fair to say that no two streams or their communities are exactly the same. Nevertheless, some generalizations can be made about the nature of stream communities. The primary producers are principally submerged and semi-aquatic macrophytes, bryophytes (mosses and liverworts), filamentous algae, cyanobacteria (blue-green algae) and microscopic diatoms that are ubiquitous, particularly on stones and other hard surfaces. The primary consumers in New Zealand streams and rivers are almost exclusively invertebrates, particularly insects, molluscs, crustaceans and worms (see Chapter 16). No algal- or detrital-feeding fish occur in our freshwater fauna, although the extinct grayling may have been an herbivorous or omnivorous fish (McDowall 1990). The secondary consumers are principally fish and larval insects such as the dobsonfly *Archichauliodes* and the stonefly *Stenoperla*, but also birds like the blue duck and black stilt. Some fish, such as eels and the introduced trout and perch, can also be tertiary consumers (at least in part) if they feed on other fish. The decomposers in running waters are mainly bacteria and fungi, particularly the filamentous hyphomycetes that colonise and break down dead leaves.

The food webs of short stream reaches can include over 100 species of plants (including microscopic algae), invertebrates and fish, and individual food chains in 10 Taieri River tributaries had on average 2–5 trophic links (range 1–10). Townsend *et al.* (1998) found that, as in streams elsewhere, omnivory was common in the Taieri

system, and that food webs were most complex in stable streams with high levels of algal production.

Community structure at different spatial scales

The structure of running-water communities can be examined in various ways. Most obviously, their species composition can be determined, and both qualitative and quantitative differences in abundance or biomass used to make comparisons among communities. In order to simplify community analyses, certain kinds of taxa are often combined, for example by using coarser taxonomic categories such as orders or families, or by designating animals as members of a limited number of feeding groups. The analysis of food webs, by analysing gut-contents and, to a limited degree, by stable isotope analysis, provides information on links between species within communities. It also draws together aspects of community and ecosystem ecology by focussing on a functional aspect of community structure.

The community present at a particular stream site is a product of many variables operating at different scales (Frissell *et al.* 1986). Macro-scale variables, including climate, altitude, topography, geology, land use and biogeography, are considered to be the main variables determining what may exist at a particular site. Consistent with this contention, studies of stream invertebrates in New Zealand have identified regional differences in stream and river communities (Biggs *et al.* 1990; Harding *et al.* 1997), with regional faunas being most distinct in eco-regions with the least-modified landscapes. The loss of indigenous forest and its replacement by pasture tends to have a homogenising effect on stream faunas, which subsequently tend to comprise more tolerant, habitat-generalists. An example demonstrating both the broad similarities and regional differences of low to medium altitude running-water communities in New Zealand is given in Figure 13.1. Although caddisflies and dipterans exhibit the greatest species richness in all four systems, the diversity of mayflies is clearly much greater in the northern rivers.

Meso-scale variables operate within individual catchments and include channel morphology and riparian vegetation. Good examples of variation in communities at the meso-scale are seen by contrasting streams draining forest and grassland catchments, or forest streams with contrasting light environments, organic matter inputs and levels of

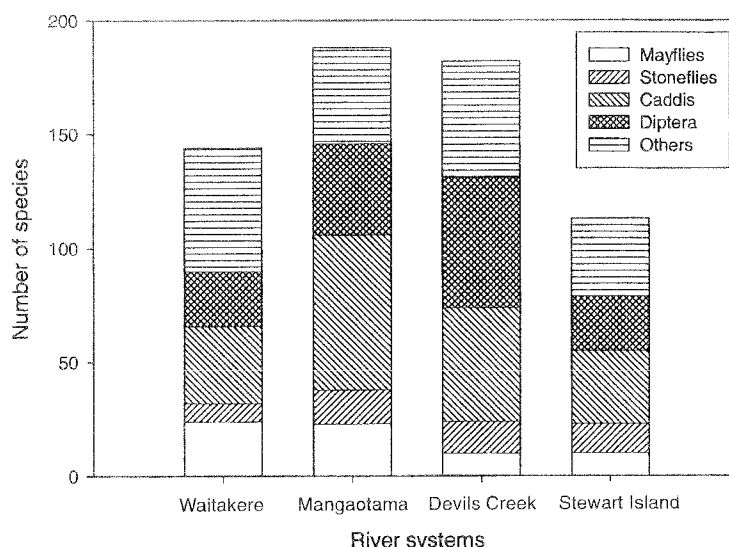


Figure 13.1 Numbers of species belonging to major invertebrate taxa reported from four areas. A—Waitakere River, Auckland; B—Mangaotama Stream, Waikato; C—Devils Creek, Reefton; D—Stewart Island. Data from Towns (1978), Collier *et al.* (2000), Cowie (1983) and Chadderton (1990), respectively.

primary productivity. For example, Death (2003) distinguished three types of invertebrate community in forest streams that differed in size, flow permanence and degree of riparian cover within Te Urewera National Park (Table 13.1). In contrast, the faunas of low-gradient streams are strongly influenced by substratum type, macrophyte community composition and water clarity, as found by Duggan *et al.* (2002). Thus, stony lowland streams in Westland were dominated by mayflies, caddis and elmid beetles, whereas silt and sand-bottomed Waikato streams had more extensive macrophyte beds and a predominance of molluscs (*Potamopyrgus*), amphipods and oligochaetes.

Micro-scale variables influence community structure at local spatial scales (c. 100 m) and include substratum type, current velocity and bed stability. Stream beds composed of more heterogeneous stony substrata often support more diverse communities than do more uniform substrata, as habitat diversity is greater. Invertebrate faunas on stones,

Table 13.1 Characteristics of three kinds of forest streams identified in Te Urewera National Park, North Island. After Death (2003).

Stream type	Width (m)	Flow	Canopy	Dominant fauna
A	<1	Intermittent	Closed	Chironomids
B	1–10	Permanent	Closed	Mayflies, caddisflies
C	12–15	Permanent	Open	Chironomids, stoneflies, mayflies

wood and plants may also differ within the same stream reach, although the difference is often one of relative abundance rather than species composition (Death 2000). The light environment also influences community structure at the micro-scale, as shown by Towns (1981), who found that artificial shading of a section of stream bed resulted in a loss of periphyton and a reduction in numbers of grazing invertebrates. However, the latter were replaced by filter-feeding caddis and blackfly larvae that could attach to the barer substratum.

Functional feeding groups

Functional feeding groups were introduced by Cummins (1973) to simplify invertebrate community data for analysis. They group together invertebrates that feed in different ways, for example by distinguishing filterers of fine particles from shredders (chewers) of leaf detritus. Functional feeding groups used in a number of New Zealand stream studies are shown in Table 13.2, which also gives examples of each.

Although an excellent concept in theory, caution is required in using functional feeding groups in practice, because stream invertebrates may feed in different ways in different habitats, at different times of year (depending on what foods are available) and at different stages in their life cycle. Also, from an ecosystem perspective, trophic designations (e.g., herbivore, detritivore), which are not the same thing as functional feeding groups (Table 13.2), are likely to be more appropriate. In New Zealand the dominant functional feeding group in many stony streams is the collector-browsers. Its members feed on a mixture of fine detritus and algae, and include numerous insects, snails, crustaceans and oligochaetes. Our other common feeding groups are the shredders, mainly chewing insects that feed on dead leaves and wood, filterers that sieve materials from the water column using a net or hairy appendages, and predators, which eat other animals.

In an extensive study of 100 small streams in 10 South Island ecoregions, Harding *et al.* (1997) found that on average collector-browsers made up 83% of the macro-

invertebrates collected. Mayfly nymphs were the dominant members of this group in most forest streams, along with amphipods on Stewart Island and in eastern Southland, whereas snails, chironomids and oligochaetes predominated in pasture streams. Average proportional representation of the other three functional feeding groups were predators (5%), shredders (1%) and filterers (11%), although the latter (mainly *Coloburiscus*, *Austrosimulium* and *Hydropsychidae*) constituted 30% and 24% of the small stream faunas on Banks Peninsula and in north-east Nelson.

A very similar picture of community structure was obtained by Biggs *et al.* (1990) for 88 larger rivers throughout New Zealand, despite their findings being presented in terms of biomass (ash-free dry weight) rather than relative abundance. Thus, on average, collector-browsers constituted 88% of functional feeding group biomass, filterers 8%, predators 3%, and shredders 1%.

Structuring of communities

Multiple interactions of organisms with biotic and abiotic factors result in the distinctive characteristics of communities, including their composition, species richness, abundance, diversity and stability. Competition, predation, symbioses, and the abiotic factors that create environmental patchiness can all have significant effects on their characteristics, although which combination of them is most important varies from one community to another. Furthermore, the structure of communities changes in response to disturbance, the destabilizing activities of the organisms themselves, and as a consequence of ecological succession (Campbell 1993).

Abiotic factors that structure stream and river communities include physical factors relating to flow, water chemistry and the nature of the substratum, whereas biotic factors include food availability and interactions among species, including competition, predation, various mutualisms and disease. The harsh-benign hypothesis of Peckarsky (1984) predicted that abiotic factors would predominate in determining stream community structure

Table 13.2 Functional feeding groups (FFGs) used to categorise stream invertebrates in New Zealand. Trophic roles played by members of each group, and some representative taxa are also given. Note that functional feeding groups are not the same thing as food-based categories, and that some taxa (e.g., *Paranephrops*) can belong to more than one functional feeding group.

FFG	Trophic role	Taxa
Shredders	Detritivore	<i>Austroperla</i> , <i>Zelandoperla</i> , <i>Paranephrops</i>
Filterers	Herbivore, detritivore, carnivore	<i>Coloburiscus</i> , <i>Hydrobiosella</i> , <i>Aoteapsyche</i> , <i>Austrosimulium</i>
Collector-browsers	Herbivore, detritivore	<i>Deleatidium</i> , <i>Pycnocentroides</i> , Orthoclaadiinae
Predators	Carnivore	Hydrobiosidae, <i>Archichauliodes</i> , <i>Paranephrops</i>

in physically harsh environments, whereas competition and predation should become increasingly important in less disturbed (more benign) streams and rivers (Fig. 13.2). Consistent with this hypothesis, New Zealand mountain streams are seen as harsh physical environments in which competition and predation play minor roles in determining invertebrate community structure (Winterbourn 1997), in contrast to spring-fed streams with minimally disturbed flow regimes and high productivity.

Similarly, Biggs (2000) hypothesised that the interaction between disturbance frequency and nutrient supply would result in a habitat gradient along which the composition and biomass of periphyton would respond predictably, as shown in Figure 13.3. Towards the “benign” end of this

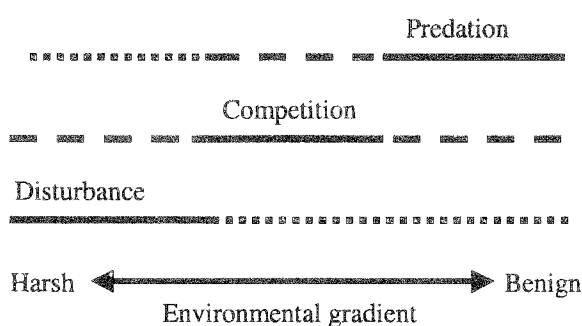


Figure 13.2 The hypothesised importance of disturbance, competition and predation on the structuring of stream communities along a harsh-benign environmental gradient. Solid lines – most important; dashed lines – intermediate importance; dotted lines – least important (adapted from Peckarsky 1984).

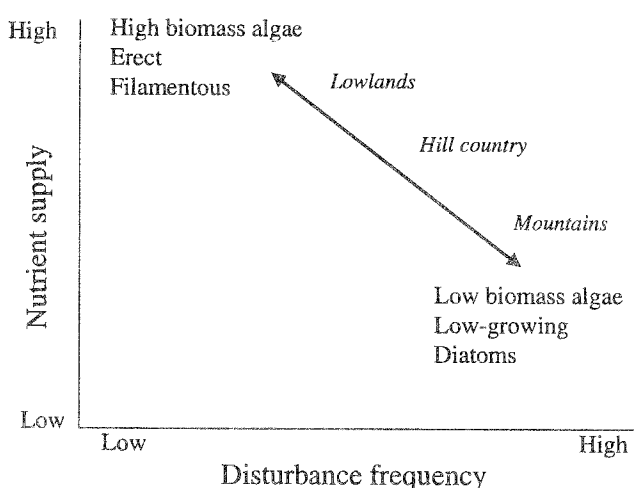


Figure 13.3 The combined effects of disturbance frequency and nutrient supply on algal assemblages in New Zealand streams as envisaged by Biggs (2000). Characteristic geographic locations of the extreme and intermediate communities are also shown.

gradient, grazing invertebrates can also have a significant effect on the architecture and biomass of algae, such that heavily grazed communities will be dominated by low-growing diatoms like *Cocconeis* that are able to resist grazing more successfully than erect and filamentous species.

Of the many biotic and abiotic factors influencing stream community structure, disturbance and predation are receiving considerable attention from ecologists in New Zealand and elsewhere.

Disturbance

Water chemistry and substratum particle size are significant determinants of stream community composition and structure. Recently, however, physical disturbance has received recognition as perhaps the most important abiotic factor influencing benthic stream communities in relatively pristine streams, i.e., those largely unaffected by human-induced disturbances such as pollution or flow regulation. Disturbance incorporates the effects of floods, spates and periods of drying, and includes their effects on the temperature regime, water chemistry, and the movement and redistribution of stream bed substrata. The magnitude, frequency and predictability of disturbances all influence running-water communities through their effects on individual aquatic taxa. Exactly how disturbance should be defined, and therefore recognised and measured, is controversial, however. The definition of White and Pickett (1985) “a disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure, and changes resources, substratum availability or the physical environment” indicates how most stream ecologists use the term, even though they rarely provide an explicit definition in their papers.

Straightforward indicators of physical disturbance used in studies of stream biota include the stream bottom component of the Pfankuch channel stability index (Pfankuch 1975; Table 13.3), and distances moved by painted stones arranged systematically on the stream bed.

Table 13.3 Variables assessed visually in estimating stream bed stability using the bottom component of the Pfankuch index. For details of scoring see Collier (1992).

Rock angularity
Stone surface brightness
Particle packing (consolidation) of substratum
Percentage of stable bed materials
Amount of scouring and deposition in the channel
Amount of attached algae and mosses

In the Cass basin, stable streams have greater invertebrate species richness than unstable ones, and increasing disturbance frequency seemed to act on species richness by reducing the time available for recolonization following disturbances (Death 1995). Ecological theory predicts that intermediate levels of disturbance should promote biodiversity by maintaining habitat complexity, but this prediction was not borne out by the Cass basin study. However, a better fit to the intermediate disturbance hypothesis was obtained by Townsend *et al.* (1997), in a study of invertebrate communities at 54 sites in the Taieri River system. The reasons for these differences are not well understood, but Death (2002) has argued that rather than being directly determined by disturbance intensity/frequency, species richness will depend on periphyton biomass, which in turn is determined by substratum stability and light levels.

In harsh (disturbed) stream environments in New Zealand, invertebrate communities are often dominated numerically by chironomids and mayflies, especially *Deleatidium* species, epilithic algal biomass is low, and retention of detritus is weak. An extreme example is provided by the river outflows from the Fox and Franz Josef glaciers, where in very cold (<2°C), turbid, flood-prone conditions only seven invertebrate species, all chironomids and mayflies, were found, at very low densities (Milner *et al.* 2001). In contrast, more benign (stable) streams usually support a much greater diversity and abundance of invertebrates, and faunal composition is more site-specific (Death 1995).

Effects of predators

In New Zealand the most convincing examples of predators having a dominant influence on stream communities come from studies on the effects of trout on galaxiid fishes in Otago and Canterbury, and their consequential effects on invertebrate and algal communities. In the Taieri basin, brown trout and galaxiids rarely co-exist (Fig. 13.4)—most galaxiid populations are restricted to streams inaccessible to trout (e.g., headwater streams above waterfalls) (Townsend and Crowl 1991). Brown trout have also been implicated in reducing the feeding activity of grazing invertebrates, and by doing so stimulating a trophic cascade, with epilithic algal populations formerly kept in check by grazers increasing their standing crops and biomass (Townsend 2003). In the Waimakariri catchment, where

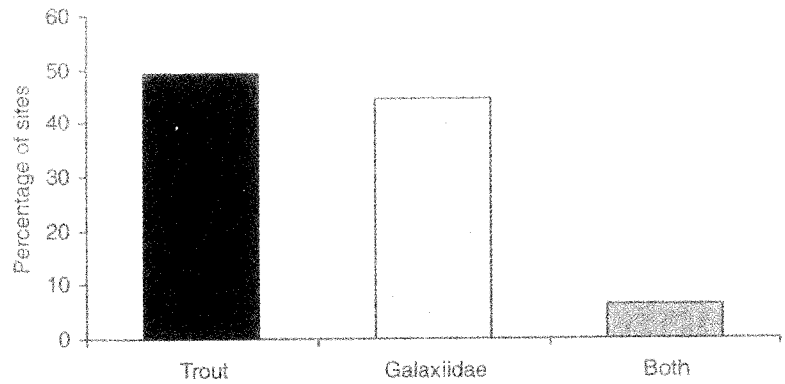


Figure 13.4 The percentage of electro-fished stream sites ($n = 144$) in the Taieri River basin containing introduced brown trout only, native galaxiids only, and both trout and galaxiid fish (after Townsend and Crowl 1991).

the spatial separation of trout and galaxiids appears to be less pronounced, stream invertebrate communities were found to differ in composition and abundance in trout streams and trout-less streams (Nystrom *et al.* 2003). Fewer large predators and shredders occurred in trout streams, and a higher proportion of the non-predatory invertebrates were cased caddisfly larvae and snails. This study indicated that different predators have contrasting effects on stream invertebrate communities, both directly through predation and indirectly by modifying invertebrate behaviour.

In contrast to trout, crayfish (Fig. 13.5) have been described as ecosystem engineers, because they modify habitat by redistributing fine sediment, thereby making patches of substrata available for colonisation by benthic algae and invertebrates. Furthermore, Usio (2000) found that crayfish (*Paranephrops zealandicus*) greatly promoted leaf breakdown rates and reduced the abundance of

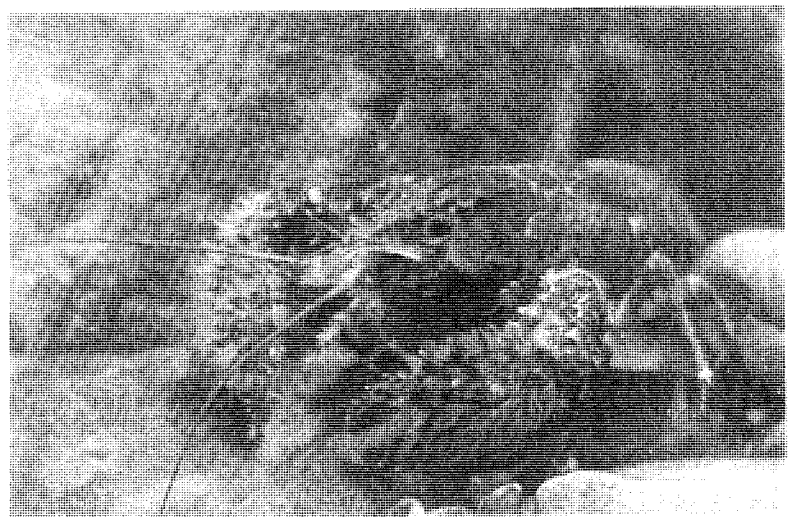


Figure 13.5 The southern crayfish *Paranephrops zealandicus*. The northern crayfish *P. planifrons* has much less hairy chelipeds.

Photo: Angus McIntosh

invertebrates in leaf packs through predation and bioturbation. As crayfish can be common in moderately harsh stream environments, such as the high rainfall region of the West Coast, South Island, their influence on the structure and function of stream communities may have been underestimated in the past.

Colonisation and dispersal

Streams are open systems whose biological communities are continually dependent on colonization events. So, although abiotic factors and species interactions have significant roles in determining community structure, the presence of particular plant, animal and microbial species is determined ultimately by their ability to colonise. The terrestrial adults of most aquatic insects whose larvae inhabit streams are the primary colonisation stage in the life cycle, and their egg-laying behaviour determines where larvae will be found. For example, some blackfly species only lay eggs in forest or open streams, and their larval distributions reflect this well-established behaviour (Timm 1994). The adults of some aquatic insects may also make oriented flights along streams, e.g., upstream or into forest, thereby enhancing the likelihood that eggs and larvae will develop in appropriate places (Winterbourn and Crowe 2001).

Fish provide many examples of colonization movements. The juvenile stages (whitebait) of migratory Galaxiidae swim large distances upstream to find suitable habitat for adult fish, whereas members of land-locked populations (e.g., of koaro) may migrate into tributaries to spawn, and then move into larger streams and lakes. Comparable migratory movements are also found in invertebrates with no terrestrial dispersal phase. Thus, larvae of the shrimp *Paratya curvirostris* develop in estuaries before swimming upstream, while the larvae or juveniles of mussels and pea clams (sphaeriids) disperse passively, attached to fish and flying insects.

Smaller-scale movements that are implicated in the dispersal, redistribution and recolonisation of substrata by invertebrates (and by inference at least some plant and microbial taxa as well) are downstream drift, and movements upstream, laterally and vertically within and on the streambed (Smock 1996). Drift includes catastrophic downstream displacement by floods, but also frequent small-scale movements, which may be accidental or induced by cues such as the presence of a predator, or limited availability of food. The distances drifted and the propensity to drift vary considerably among species, depending on current speed, turbulence and the ability of individuals to reattach. Ledger *et al.* (2002) also found that the mobility of some common New Zealand stream insects was related to their mode of feeding and diet. Thus, leaf-feeders like the stonefly *Spaniocerca zelandica* and the

caddisfly *Zelandopsyche ingens*, which have patchily distributed foods, drifted further and more often than stoneflies, which fed on a wider range of foods.

Although some individuals are likely to be drifting within a reach at any one time, the overall proportion of a population in the drift is almost always very low. However, drift tends to be higher at night than during the day in many species, especially mayflies (Sagar and Glova 1992). Experimental work by McIntosh and Townsend (1994) showed that nymphs of *Nesameletus* from a fishless stream drifted frequently during the night and day, whereas nymphs from a trout stream were strictly nocturnal. They concluded that trout may be providing the selection pressure leading to nocturnal peaks in drift activity by inducing changes in the activity and behaviour of prey organisms, including the time spent grazing on the tops of stones. The nocturnal activity of benthic feeding fish such as some galaxiid species can also induce nocturnal drift in ephemeropterans (McIntosh and Townsend 1994) and may explain why Watson (1971) observed strong nocturnal peaks for mayflies in the Waitakere River, where trout have never occurred.

RUNNING-WATER ECOSYSTEMS

Unlike lakes, which are usually depicted as closed ecological systems, streams are open systems dependent to a large extent on a continuing supply of energy, materials, nutrients, and organisms such as insects from outside the stream itself. These allochthonous inputs are supported by autochthonous inputs provided by in-stream primary producers, whose relative importance can be expected to be greater in larger, more open rivers and where plant nutrients are in good supply. The importance of terrestrial inputs to the functioning of running-water ecosystems and the significant links between aquatic and terrestrial food webs (Fig. 13.6) mean that stream ecology must be viewed in the context of the catchment.

Early studies on energy flow, carbon pathways and organic matter processing in forested headwater streams (e.g., Fisher and Likens 1973; Cummins 1974) established the importance of dissolved organic matter and particulate organic matter derived from outside the stream as both major energy inputs and sources of carbon supporting stream communities (Fig. 13.7). Furthermore, because rivers are one-way flow systems, some proportion of the inputs and materials processed at any one site will be transported downstream before being taken up again. This process of nutrient and material cycling with a downstream component (nutrient spiraling) was incorporated in the river continuum concept (Vannote *et al.* 1980) to indicate that downstream communities and ecosystem functions are influenced by communities upstream. The river

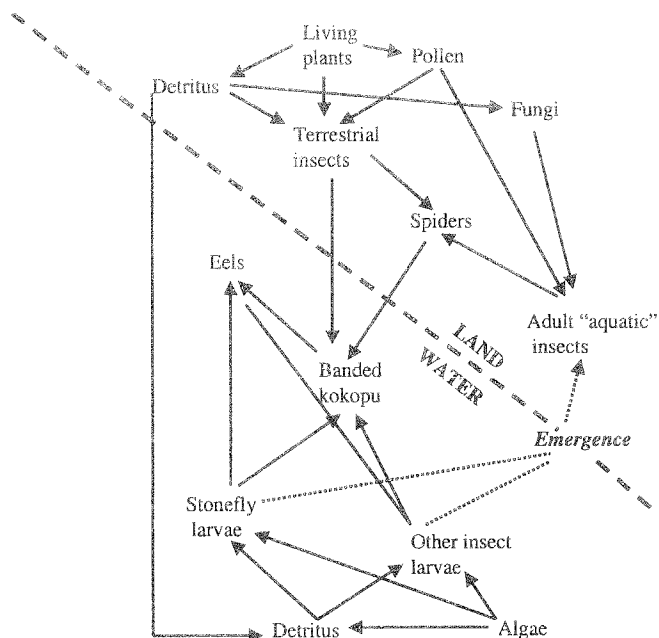


Figure 13.6 Trophic (feeding) links between stream and terrestrial ecosystems at a hypothetical forest stream containing eels (*Anguilla* spp.) and banded kokopu (*Galaxias fasciatus*) as top predators.

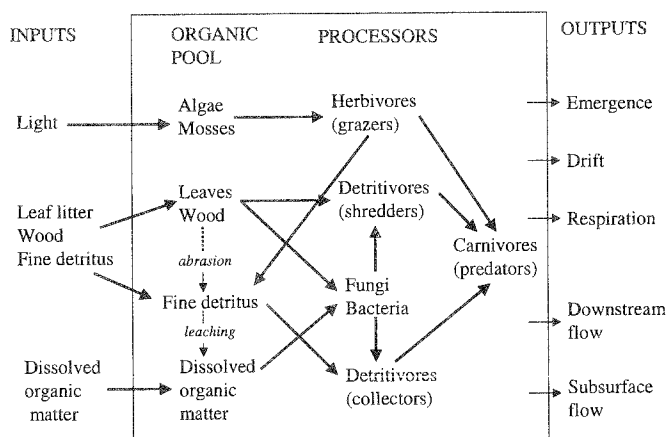


Figure 13.7 A model of energy flow through a forest stream ecosystem showing energy inputs, outputs and major organic-matter processing pathways.

particulate organic matter, some of which enters the aquatic food web. The floodplains of rivers like the Amazon also provide annual breeding and feeding habitat for fishes on a vast scale, and Young and Huryn (1997) reported that brown trout moved onto an inundated floodplain of the Taieri River and fed on invertebrates there. They also found that concentrations of suspended organic particles (seston) and dissolved organic matter were highest in floodplain reaches (Table 13.4) and concluded that the floodplain and submerged macrophyte beds were their primary sources. However, contrary to the predictions of the river continuum concept, no longitudinal patterns in the concentration of organic seston (mostly ultra-fine particles $<30\ \mu\text{m}$) were found along the 310 km length of river studied. Estimates of daily spiraling length for total organic carbon (calculated as downstream carbon transport divided by community respiration) in the Taieri ranged from 10–98 km, and increased downstream, in parallel with increases in channel dimensions and discharge. Elsewhere, estimates of average carbon spiraling length range from <3 km in small headwater streams to 4000 km in the Amazon (Newbold 1992), although not all compounds incorporated in these estimates turn over at the same rates. In particular, the more labile nutrients and fractions of dissolved organic matter utilised by microorganisms have much shorter spiraling lengths. Thus, ammonium uptake lengths calculated from the addition of ^{15}N tracer to nine very different North American streams were all less than 50 metres (Mulholland *et al.* 2002).

Table 13.4 Approximate carbon budget for a 47 km reach of the upper Taieri River based on data in Young and Huryn (1996, 1997). Input from the floodplain is calculated as output total minus all other inputs, i.e., assuming a steady-state situation.

	kg C/day	% input or output
Inputs from upstream		
Particulate organic matter	40	1
Dissolved organic matter	380	9
Reach primary production	2970	70
Floodplain inputs	860	20
Outputs from reach		
Particulate organic matter	320	8
Dissolved organic matter	1550	36
Respiration	2380	56
OUTPUT TOTAL	4250	100

continuum concept also predicted that the importance of primary production would increase in more open middle-sized rivers, and that the nature of the biological community at any point along a river system would reflect the predictable nature of its food supply.

The degree of dependence of downstream communities on materials from upstream is difficult to quantify and is not necessarily as important as processes specific to a given reach or segment. For example, the inundation of floodplains results in a redistribution of dissolved and

Detritus processing

Leaves and wood

When leaves become submerged, various dissolved substances are leached from them, depending upon their state of dryness, and some are taken up by microorganisms elsewhere in the stream. The leaves themselves are colonised by bacteria (including Gram-negative rods and filamentous actinomycetes) and fungi (most notably hyphomycetes), the principal decomposers in running waters (Fig. 13.8). Subsequently, rates of leaf breakdown are influenced by a variety of factors, including water temperature, water chemistry, physical abrasion, the numbers and kinds of colonizing microorganisms and invertebrates, and both the physical and chemical nature of the leaves themselves.

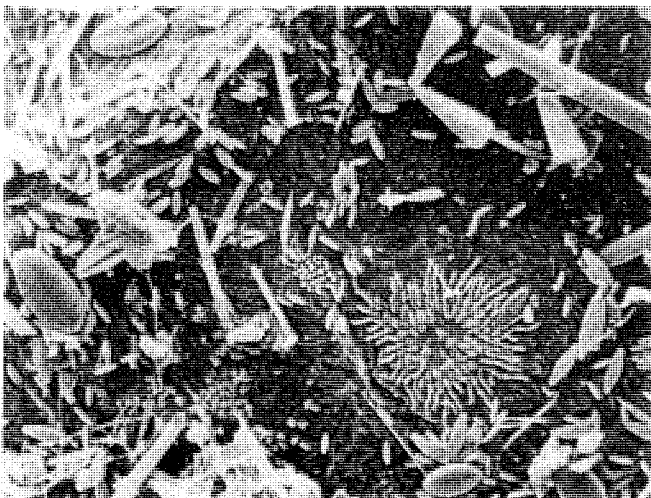


Figure 13.8 Diatoms, bacteria, filamentous organisms and amorphous detritus colonising the surface of a decomposing leaf taken from a stream.

Photo: Jan McKenzie

Quinn *et al.* (2000) concluded that leaf toughness was a key determinant of breakdown rates for several New Zealand native and introduced tree species and had a strong effect on the pattern of microbial activity. Thus, large numbers of microbes begin to decompose soft leaves such as those of mahoe (*Melicytus ramifloris*) rapidly, whereas microbial build-up is very slow on the tough leaves of rewarewa (*Knightsia excelsa*), mountain beech (*Nothofagus solandri* var. *cliffortioides*) and kauri (*Agathis australis*). The effect of site-specific factors on leaf breakdown is indicated in Table 13.5, which shows that the time for 50% weight loss by kamahi (*Weinmannia racemosa*) leaves ranged from 163–480 days in four West Coast streams.

Wood is also widespread in forested streams and rivers, although in New Zealand streams its standing stock often appears to be lower than reported elsewhere, possibly because New Zealand streams are flood-prone, especially

Table 13.5 Time for 50% weight loss of kamahi leaves in four West Coast streams differing in pH and shredder abundance. After Collier and Winterbourn (1987).

Stream	Mean pH	T ₅₀ (days)	Shredders
A	7.5	163	Common
B	7.3	198	Some
C	4.9	321	Occasional
D	4.7	480	Absent

in the mountains. Wood is important in defining channel morphology, forming the structural basis of debris dams and providing habitat for invertebrates and fish. In contrast to the decay of dead leaves, wood decay is very slow, and initially, at least, is largely a surface phenomenon. As on leaves, the main microbial colonists of wood are filamentous fungi and bacteria, some of which have high cellulase activity and can be embedded in layers of mucilage 2–3 mm thick (Tank and Winterbourn 1995).

The shredder pathway

As well as being decomposers of leaves and wood, microflora make leaves more attractive foods for invertebrates. Cummins (1974) compared the leaf microflora with peanut butter on crackers, and predicted that following such “conditioning” a high proportion of the autumn-shed leaves in deciduous forest streams would be eaten by large-particle detritivores (or shredders). Smaller particles egested by the shredders, which can digest only a small proportion of the plant material they ingest, would then be eaten in turn by smaller detritivores, collectively termed collectors (Fig. 13.7). This route was predicted to be a major pathway of energy flow within forest stream ecosystems, and placed emphasis on the pivotal role of shredders in their functioning.

In New Zealand, shredders are rare or absent from many forest streams, especially in the mountains and high rainfall areas (Winterbourn *et al.* 1981; Thomson and Townsend 2000). However, large populations of leaf- and wood-shredding caddisflies and stoneflies have been reported in small native forest streams on Banks Peninsula (*Oeconesus maori* and *Pycnocentria forcipata*; Linklater and Winterbourn 1993), and in exotic forest streams near Hanmer Springs (*Austroperla cyrene*; Harding and Winterbourn 1995) and in the central North Island (*Pycnocentria funerea*; Collier and Smith 2003). Where shredder feeding is minimal, the production of fine particles by physical and microbial breakdown in streams and the riparian zone must be of considerable importance, since invertebrates that feed on fine particles are widely distributed and abundant in forested and many other streams throughout the country.

Epilithon and other biofilms

The surfaces of stones are major sites of carbon transfer in streams, since a great diversity of invertebrates feed there on complex biofilms (organic layers) composed of algae, microorganisms and detrital particles, often embedded in a polysaccharide matrix (Fig. 13.9). They also feed on the microflora colonising the surfaces of submerged and semi-submerged branches and logs; here Anderson (1982) found insects belonging to at least 35 families in seven orders, as well as snails, isopods and worms.

The composition of epilithic biofilms varies greatly among streams, with the relative importance of heterotrophic and autotrophic elements being influenced in large part by the available light. Adsorbed or precipitated organic carbon was a major component of epilithon in two small, acid brown-water streams studied by Collier and Winterbourn (1990), who speculated that humic substances, low nutrient availability and frequent physical abrasion of stone surfaces may keep periphytic biomass low. Although it is usually assumed that algae are the most important components of epilithon eaten and digested by grazing invertebrates, Rounick and Winterbourn (1983) found that heterotrophic organic layers grown in the dark in a forest stream were capable of supporting growth of *Deleatidium* larvae. Similarly, ^{14}C -glucose labeled biofilm on sticks incubated within the sediments of West Coast forest streams was incorporated into body tissue by the amphipod *Paraleptamphopus* (Tank and Winterbourn 1995). On reflection, these results are not unexpected, since large numbers of grazing invertebrates live and presumably feed within the streambed and hyporheic zone (see Chapter 32), where living algae are rarely found and

cannot photosynthesise. Despite the limited significance of a shredder pathway in many New Zealand streams, it is apparent that detrital carbon is incorporated into stream food webs by other effective routes.

Primary producers and primary production

Headwater streams

Small forest streams are characterised by low algal biomass, and primary production may be limited by low light intensity. However, many aquatic and semi-aquatic mosses and liverworts can thrive in low-light environments and also under high light intensities, where they may produce protective pigments. Thick coverings of bryophytes occur commonly on stable substrata such as boulders, and bedrock under waterfalls. Of 103 tributary streams surveyed by Suren (1993) in Arthurs Pass National Park, half of those above and below the tree line contained bryophytes, and their occurrence was strongly correlated with stream bed stability measured with the bottom component of the Pfankuch index. The maximum biomass recorded was over 650 g ash-free dry weight/m², with the greatest individual species biomass provided by the moss *Fissidens rigidulus*. As well as being primary producers in their own right, bryophytes provide colonization surfaces for periphyton species, especially diatoms, and fine detritus is trapped amongst their stems. The periphyton and detritus provide food for an extensive fauna of mainly very small invertebrates that live in bryophyte mats, although few species feed on the bryophytes themselves.

Nutrient limitation

In larger streams and rivers, nutrient availability rather than light may limit primary production, although self-shading can reduce rates of production within thick algal mats. Limitation of benthic algal production by the availability of nitrogen, phosphorus, or both has been demonstrated in New Zealand streams using nutrient-diffusing substrata (Winterbourn 1990; Biggs and Lowe 1994). Nitrogen in particular is present in low concentrations in many New Zealand freshwaters and has led to a number of studies of nitrogen cycling (Fig. 13.10) at the land-water interface (Howard-Williams 1991). These, and studies elsewhere, have shown that streams are responsible for much of the transformation and transport of nutrients within catchments, with phosphate, ammonia and nitrate all capable of being taken up rapidly and in large amounts by algae, macrophytes and bacteria.

River metabolism

Primary production and respiration by stream or river communities can be estimated from measurements of the oxygen produced and used in closed or flow-through chambers, or by monitoring changes in oxygen concen-

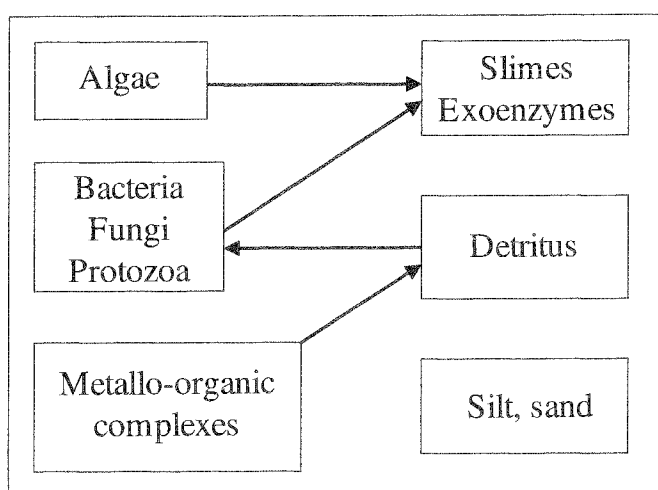


Figure 13.9 A compartment model of stone-surface biofilm (epilithon) showing pathways of carbon flow. The relative sizes and importance of the compartments depends on many factors such as water chemistry, flow rate, nutrient availability, and degree of shading.

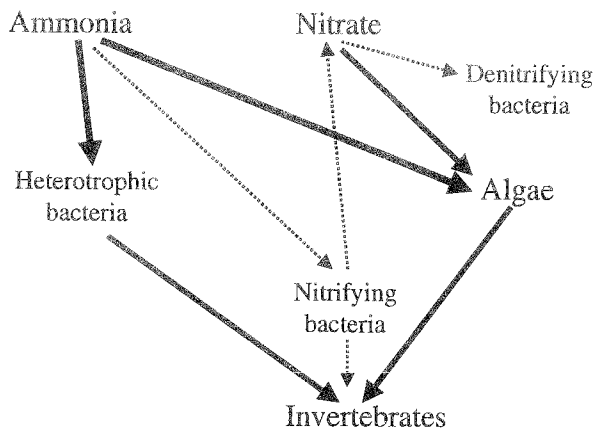


Figure 13.10 Major (solid lines) and minor (dotted lines) nitrogen cycling pathways involving microbes, algae and invertebrates in streams (after Hall and Tank 2003).

tration at one or more river sites over 24-hour periods. Quinn *et al.* (1997) used sealed recirculating chambers to measure primary production in small Waikato streams and found that rates in pasture streams ($\sim 5 \text{ mg O}_2/\text{m}^2/\text{d}$) were about 2.5–11 times higher than those in native and exotic forest. An average production/respiration ratio of 2.8 in the pasture streams indicated they were autotrophic systems, whereas the native forest streams had an average production/respiration ratio of 0.6, indicating they were more heterotrophic. In contrast to the Waikato research, an open-channel method was used by Young and Huryn (1996) to measure river metabolism at 14 stations along the Taieri River. Gross primary productivity estimates for the Taieri, which is bordered by tussock grassland, pasture and willows, ranged from <0.3 – $9.6 \text{ g O}_2/\text{m}^2/\text{d}$, and production/respiration ratios indicated autotrophy throughout the 310 km of river studied. Patterns of primary production observed appeared to be controlled largely by water depth and turbidity rather than by position along the river continuum, and comparisons made between years suggested that hydrological variability is likely to have a very strong influence on longitudinal patterns.

Secondary production

Secondary production is the accumulation of biomass by consumers. It is affected by a variety of environmental factors, food resources and biotic interactions, and its accurate calculation requires detailed field and laboratory measurements. Consequently, it is not surprising that most production studies have been of individual species rather than whole communities. Two aspects of particular interest to stream ecologists have been the relative importance of terrestrial and aquatic sources of energy in stream ecosystems, and the basis of fish production. Stable

isotopes have been used successfully in recent studies to examine the first of these, i.e., the basis of secondary production.

Stable isotope studies

Secondary production in rivers and streams depends ultimately on the availability of food derived from autotrophic (in-stream) primary production and allochthonous (terrestrial) organic matter. The relative importance of the two sources to consumers can be determined using stable carbon isotope analysis, since the ratio $^{13}\text{C}:^{12}\text{C}$ in consumer tissue reflects that of the foods assimilated, with little change (Rounick *et al.* 1982). To be effective, the energy sources of interest must have distinct, non-overlapping isotope ratios, but this is not always the case, and therefore is a limitation of the method.

In general, stable isotope studies have confirmed the importance of terrestrial organic matter as the principal source of carbon converted into invertebrate tissue in forest streams. However, despite typically low levels of in-stream primary production, in-stream algae may support a high proportion of the production of some invertebrate species (e.g., snails), and play a more significant role in essentially heterotrophic stream ecosystems than expected (Thorpe and Delong 2002). Stable isotope studies have also confirmed the greater dependence of more open (e.g., grassland) streams and rivers on in-stream carbon sources, particularly algae, although the production of some invertebrates may still be supported primarily by detritus of terrestrial origin. Thus, in open-canopy streams within the Waimakariri catchment, stable isotope ratios indicated that *Deleatidium* depended mainly on algal carbon, whereas the production of *Olinga* was supported by carbon of terrestrial origin (Nystrom *et al.* 2003).

The ratio of the stable isotopes of nitrogen ($^{15}\text{N}:^{14}\text{N}$) can also be used to indicate the approximate trophic level occupied by individual species, since an increase in this ratio of about 3 ppt occurs at each transfer in a food chain (Fig. 13.11). It needs to be appreciated, however, that the interpretation of stable isotope values of organisms (both C and N) is not always straightforward and they are best used in conjunction with other criteria, such as gut-content analyses and knowledge of feeding methods. The value of a combined approach is shown by a recent study of the New Zealand freshwater crayfish *Paranephrops planifrons*, whose stable isotope ratios indicated strong dependence on animal prey (Parkyn *et al.* 2001). However, although some invertebrates had been ingested, its gut contents were predominantly leaf detritus. To account for this, the authors concluded that detritus was primarily a source of energy used for maintenance (respiration), whereas invertebrates supported tissue growth. In this example, an examination of either gut-contents or stable isotopes alone

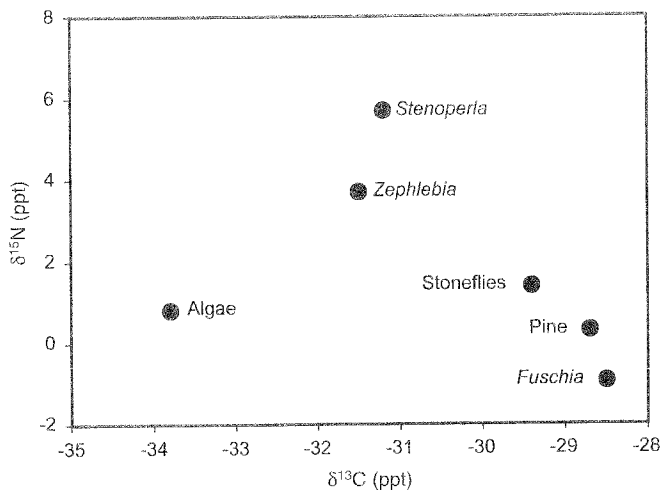


Figure 13.11 Stable carbon and nitrogen isotope ratios for leaves, algae and three types of insect (a mayfly *Zephlebia*, small stoneflies *Zelandobius* and *Spaniocerca*, and the large predatory stonefly *Stenoperla*) in a small stream near Reefton. Note the increases in $\delta^{15}\text{N}$ with trophic level, and the relative positions of the symbols for primary consumers, which suggest more algae are assimilated and converted to tissue by *Zephlebia* than by stoneflies (M.J. Winterbourn, unpublished data)

would have resulted in different and incorrect inferences being drawn.

Lastly, stable isotopes have been valuable in revealing unsuspected trophic links between streams and the riparian zone. For example, Collier *et al.* (2002) found that the isotope signatures of spiders alongside two North Island streams indicated they fed on aquatic prey or on the terrestrial adults of insects such as stoneflies that have stream-dwelling larvae (Fig. 13.6). An analogous situation, whose importance has been substantiated by stable isotope studies in North America but not yet in New Zealand, is the incorporation of marine carbon and nitrogen into stream food webs through the decomposition of salmon carcasses following spawning in headwater streams (Wipfli *et al.* 1998). It is also probable that marine-derived nutrients are introduced to terrestrial and aquatic ecosystems from seabird colonies, such as those of burrowing petrels, a biogeochemical pathway that may have been much more significant in pre-human times in New Zealand (Holdaway 1989).

The basis of fish production

The importance of fisheries for both food and recreation has resulted in many studies of fish production, one of the most famous being that of Allen (1951) on a brown trout population in the Horokiwi Stream near Wellington. Later, Hopkins (1971) calculated production of fish in two small streams in the lower North Island,

determined their annual food consumption, and estimated the annual production of invertebrate communities from quantitative benthic samples (Hopkins 1976). His production estimates for six sites on the Hinau and Horokiwi streams ranged from 10–85 g dry weight/m², with an average production/biomass ratio of about 7. As his estimate of annual food consumption by fish in the main stem of the Hinau Stream was about 67 g dry weight m², and the calculated production of invertebrates was 82 g/m², this suggests that almost all biomass produced by the benthos was consumed. In his earlier study, Allen (1951) calculated that in a year trout consumed 40–150 times the mean biomass of invertebrates present, a deficit that he could not explain.

Reasons for the so-called Allen paradox were re-examined by Huryn (1996), in a comprehensive production study in an Otago stream containing brown trout but no other fish. His results indicated that trout consumed most (>80%) benthic invertebrate production and that consumption of terrestrial invertebrates, hyporheic fauna, and juvenile trout was necessary to balance the budget (Table 13.6). However, he found that primary consumers did not eat all the algal biomass produced, the surplus (about 79% of total primary production) being the result of “top-down” control of herbivores by trout. In contrast to the trout stream, little top-down control of invertebrates by fish predation was found in a second Otago stream inhabited by galaxiid fishes (*Galaxias eldoni*), and a much large proportion (about 75%) of the algal biomass produced was consumed by invertebrates (Huryn 1998; Table 13.6).

Table 13.6 Food demand by fish and invertebrates and net production available to them in two Otago streams inhabited by brown trout and Eldon’s galaxias, respectively. Confidence limits (95%) around the means shown range from 4–65%. Data from Huryn (1996, 1998).

N.A. = not applicable; N.M. = not measured.

	Trout stream g/m ² /yr	Galaxias stream g/m ² /yr
Primary production	310	50
Primary consumer demand	66 (21%)	37 (74%)
Trout demand	11.3	N.A.
Galaxias demand	N.A.	1.2
Available food		
Benthic invertebrates	10.3	5.3
Terrestrial invertebrates	0.6	N.M.
Juvenile trout	0.3	N.A.
Total available food	11.2	5.3

Although production estimates for whole stream communities require many assumptions and approximations, the results of New Zealand studies indicate that introduced trout can have different and stronger effects on stream ecosystems than at least some native fish species. In particular, the cascading effect of trout on algal biomass through predation on invertebrates and modification of their prey behaviour is impressive and provides an excellent example of how a merging of the approaches of community and ecosystem ecology can provide new and cogent insights in ecology.

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Chapter 12

Stream sediment load and organic matter

Murray Hicks, John Quinn and Noel Trustrum

INTRODUCTION

The particulate load of a stream or river exerts fundamental control on channel form and behaviour, affects the appearance, geochemistry and shore morphology of receiving waters, influences the type and diversity of the river ecosystems, and imposes a host of engineering and management problems when supplied in excess or deficit.

The particulate load includes material of organic and inorganic composition and ranges in size from clays less than 1 micron to boulders or tree trunks. Inorganic material, often referred to as sediment or more loosely as “silt”, is usually dominant, comprising mineral grains and rock fragments eroded from hillsides and soil layers. The organic material is generally plant debris that has been washed or has fallen directly into the stream. It includes fine organic material derived from topsoil erosion, leaf litter, wood fragments and whole trees. The finer fragments are termed particulate organic carbon, while branches and tree trunks are termed large woody debris.

As explained in Chapter 8, the sediment supplied to a river is one of the primary controls on the channel form. For example, an abundant supply of sand and gravel, and a vigorous transport regime that is the result of steep gradients and frequent high flows, have produced the wide braided, gravel-bed rivers of Canterbury. In contrast, an excess of finer sediment can cause a river to form a narrow, high-banked meandering channel (e.g., lower Waipaoa River, East Cape).

Large woody debris can substantially affect the morphology of small streams. Logjams may dam the flow and trap sediment, imposing a step-pool structure on the stream profile.

The concentration of fine sediment suspended by a river determines its appearance, clarity, and the suitability of the river as a source of potable water. Turbid, sediment-

laden water limits light penetration for photosynthesis in aquatic ecosystems; it can inhibit feeding by fish and birds, and it can hinder the migration of some fish species (Rowe *et al.* 2000; Davies-Colley and Smith 2001). Too much fine, muddy sediment degrades the habitat quality of a riverbed if the spaces between gravel particles become clogged or if the gravel is buried.

The yield of sediment delivered over time is important for many reasons. For example, when dams are placed across rivers, sediment accumulates in the reservoir, while the river channel downstream has its sediment supply cut off. The rate of sediment entrapment determines the life of the reservoir. The reduced downstream supply can result in scour of the riverbed (called degradation) and a host of related problems, such as bank erosion and the failure of bridge piers. Similarly, degradation can result if sand and gravel aggregate is extracted from the riverbed at rates that exceed the rate of supply from upstream.

Coastal erosion can be initiated or accelerated when supplies of river sand and gravel to the coast are reduced, while massive supplies of fine sediment to the ocean during large floods can result in a mud drape over the floor of the continental shelf, smothering benthic habitats (Foster and Carter 1997).

The flux, or yield, of particulate organic carbon from the land to the ocean via rivers, estimated to be in the range of 90–240 Mt/year globally (Lyons *et al.* 2002), is a significant component in the global budget of carbon. It is a factor that needs to be considered in international initiatives, such as the Kyoto Protocol, that aim to manage global climate by managing atmospheric carbon-dioxide levels (Preston *et al.* 2003).

This chapter describes the characteristics, sources, and fluxes of the sediment load and organic matter carried by New Zealand’s rivers and streams. It reviews approaches for estimating loads and fluxes, and discusses how these may be affected by land use.

SEDIMENT LOAD

Definitions and approaches

Sediment load is often described as suspended load (if in suspension) or bedload (if moving in contact with the stream bed), reflecting the mechanism of transport (Einstein *et al.* 1940). The suspended load is dispersed through the flow by turbulence. It generally consists of finer-grained sediment of clay, silt, and fine sand grades, because this material is more easily held in suspension. Sand and gravel grains that are too heavy to be suspended may still be rolled or driven in a hopping motion along the channel bed. This bedload transport often results in the creation of bedforms such as ripples and dunes (commonly with sand grades) or lobes and sheets (with gravel).

The velocity of river flow required to entrain a sediment grain increases with the density of the mineral grain and generally with its diameter (Fig. 12.1). However, cohesive bonds between silt or clay grains mean that these grades require a higher entrainment velocity than sand, although once dispersed they can remain in suspension at much lower velocities before being deposited again. The threshold velocity for suspending a grain also increases with grain diameter. Consequently, as the velocity increases past the entrainment threshold, sediment finer than about 0.1 mm is entrained directly into suspension, while coarser sediment moves at least initially as bedload (Fig. 12.1).

Most sediment transport occurs during freshes and floods, when flow velocities are high and sediment is supplied to the channel from runoff. The sediment forming the bed of a river, as observed at normal, low-flow conditions, may be transported during floods as bedload or in suspension, depending on its size and the competency of the flow to entrain and suspend it. The gravel load

usually only moves as bedload, and is left stranded on the bed when a flood recedes. The distance that gravel particles move downstream during a flood is relatively short, typically of the order of one to several channel widths. In contrast, the finer fractions of the suspended load are, by and large, flushed right through the channel network during a flood, and so they usually appear only in trace concentrations in the bed-material. This portion of the load is termed the washload, and is derived largely from the erosion of hillslopes.

The capacity of a river to transport washload is very high, since relatively little energy is expended on keeping fine grains in suspension. Indeed, when the concentration of mud exceeds several hundred thousand mg/l, the fluid properties change from those of a water flow to a hyperconcentrated flow (Costa 1988). Such high concentrations have not been sampled in New Zealand rivers (the maximum recorded concentration from a New Zealand River is ~77,000 mg/l), and in New Zealand rivers the washload concentration is largely dependent on supply, not on capacity. The suspended sand load and bedload are more likely to be transported at capacity, in which case they may be estimated using a formula. However, in the case of gravel-bed or bedrock channels, the supply of entrainable sand and gravel may also be limited, either by scarcity or because the bed may be protected by an armour layer of coarser grains. Armour forms naturally, as finer grains on the bed surface are preferentially winnowed by the current and not replaced from upstream (Parker and Sutherland 1990).

Measuring the suspended load

For the above reasons, the suspended-sediment load in New Zealand rivers is, by and large, dependent upon supply and must be sampled to be quantified. The concentration of suspended sediment varies with depth and distance across a river section, depending on the local turbulence intensity. Thus a complete measurement—or gauging—of the suspended load passing a river section requires the use of special samplers and techniques to properly capture these variations in concentration (Hicks and Griffiths 1992; Hicks and Fenwick 1994).

Commonly, a device called a depth-integrating sampler is used. This typically consists of a streamlined brass “bomb” with an internal glass sample bottle (Fig. 12.2a). The sampler is designed to sample isokinetically; that is, the flow velocity through the intake nozzle matches the ambient stream velocity. The sampler is lowered to the streambed and recovered from it at a steady rate, and the sample collected has a sediment concentration equal to the mean velocity-weighted concentration in the vertical. Samples are collected from several verticals to determine variations in concentration across the channel.

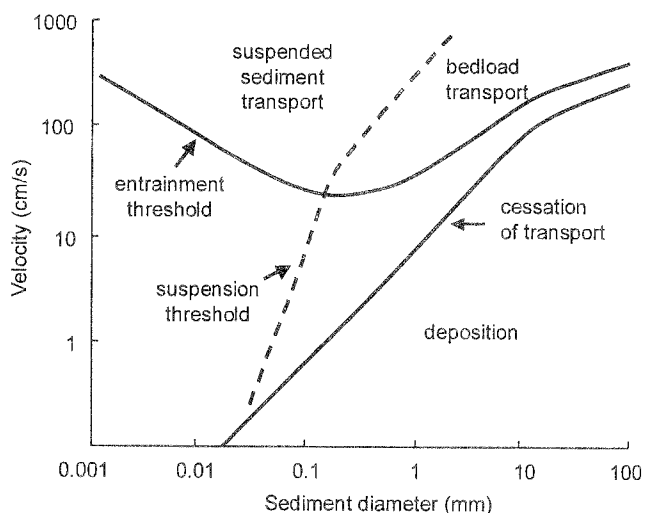


Figure 12.1 Threshold velocities for sediment entrainment, deposition, and suspension (after Hjulström 1935; Richards 1982).

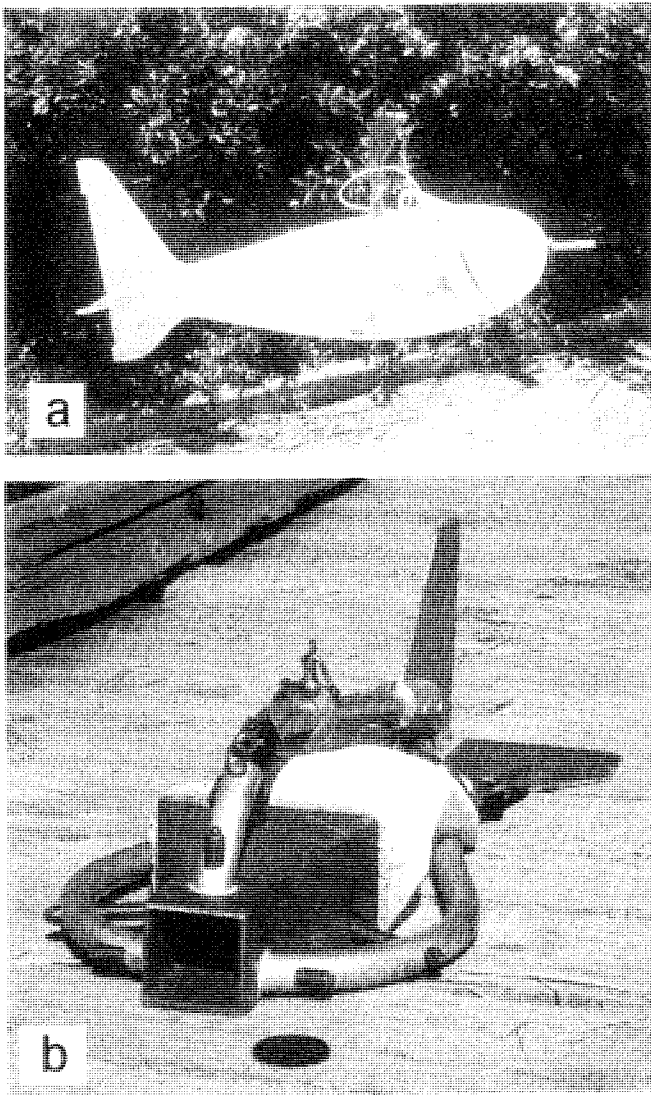


Figure 12.2 a) P-61 suspended sediment sampler. b) Helley-Smith pressure-difference type bedload sampler (from Hicks and Gomez 2003). Photos: Basil Gomez

Continuous records

The effort required for a single suspended-sediment gauging is often impractical to sustain, particularly at remote sites where water discharge and sediment concentration change rapidly. Thus if a continuous record of suspended sediment is required, it is usually sampled at only one point in the cross-section, either with an automatic pumping sampler or using an optical sensor to measure water turbidity (Gippel 1995; Hicks and Gomez 2003). The point suspended-sediment concentration or turbidity may be related to the cross-section mean concentration by undertaking calibration gauging with depth-integrating samplers. A common approach is to use turbidity sensors and auto-samplers together—the main purpose of the auto-sampler is to collect enough samples to calibrate the turbidity sensor.

Auto-samplers can be programmed (either with on-board processors or more commonly when coupled to a

data-logger) for a variety of sampling schedules, including fixed time intervals, fixed changes in water level, fixed flow volume, or using a “stratified” approach that changes the schedule according to a set of rules. A disadvantage of auto-samplers is that they typically have only 24 to 28 sample bottles. Multiple samples can be collected into a single bottle, however, and when flow-proportional sampling is carried out, the concentration of the composite sample represents the discharge-weighted average concentration for the bottle-filling period. Flow-proportional sampling involves collecting a sample whenever a fixed volume of water has passed by.

Backscattering-type (or nephelometric) turbidity sensors have proven popular for monitoring suspended sediment, since their signal-to-noise ratio increases as sediment concentration increases. The turbidity generated by a given concentration of sediment increases as grain size gets finer, thus the signal of these sensors is also sensitive to the size grade of the suspended load. For this reason, the relationship between suspended-sediment concentration and turbidity varies from river to river, and even within runoff events at a given site. While this introduces scatter into the calibration relationship, in practice this is less important than obtaining a continuous time series record. Different types of turbidity sensor, however, can give different nephelometric turbidity unit (NTU) readings for a given sediment suspension, even if they are all calibrated from the same standard solution. For this reason, each sensor should be field-calibrated to the actual sediment in the river. As a rule of thumb, however, suspended-sediment concentrations (in mg/l) in New Zealand rivers are often found to be about twice the turbidity (in NTU, Fig. 12.3).

A nuisance often encountered with turbidity sensors is biological fouling. This can be controlled with varying degrees of success by hand-cleaning, mechanical wipers, algae-repelling polymer coats, and jets that squirt water onto the lens.

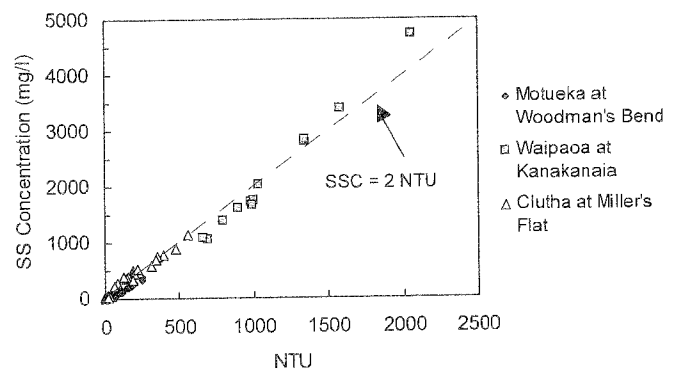


Figure 12.3 Relation between nephelometric turbidity (NTU) and suspended sediment concentration (SSC) at several New Zealand rivers.

Suspended sediment ratings

A continuous sediment load record is not necessary if the aim of sampling is to estimate the long-term average sediment yield. In that case, a relationship between sediment load (i.e., mass per unit time) or concentration (i.e., mass of sediment per unit volume of water) and water discharge—termed a sediment rating—is established from field measurements. This rating is then combined with a discharge record to estimate the long-term average yield. For faster calculations, the discharge record may be compressed into a flow-duration table.

The rating relation should represent the conditional mean sediment concentration (as a function of water discharge) over the time period of interest. Thus sufficient samples need to be collected to ensure that the variability in sediment load within given discharge intervals has been sampled in an adequate and unbiased manner (Hicks and Gomez 2003). This variability stems largely from sediment supply factors. Particularly in smaller streams, there is a tendency for concentrations to be higher on the rising limb of flood hydrographs due to an initial flush of readily available sediment. The sediment supply may also vary seasonally or over longer periods, for example due to changes in land use in the catchment. Concentrations may gradually wane in the years following a particularly severe storm as a slug of sediment is flushed through the catchment and as hillslope erosion scars heal.

There are various techniques for fitting functions to rating curves. A common approach is to use linear least-squares regression on logarithmically-transformed data. This generates a simple power-law function of the form $C = aQ^b$ (C is concentration, Q is water discharge, and a and b are coefficients determined by the regression). By using log-transformed data, however, curve-fitting techniques tend to model the conditional geometric mean concentration rather than the conditional arithmetic mean, and so the resulting sediment yield often tends to be underestimated. Various approaches have been suggested to correct for this (Duan 1983; Ferguson 1986; Cohn *et al.* 1989). Also, some care is required with the use of linear regression, because while the overall fit of the curve may appear good, the calculated sediment yield is often highly sensitive to the fit over the high discharge band. Curve-fitting methods such as non-linear regression or locally-weighted scatterplot smoothing (LOWESS) may be superior in cases where the rating is curved.

For example, for the Waipaoa River at Kanakanaia Bridge (Fig. 12.4), estimates for suspended yield range from 24.5 Mt/year when the rating is modelled by least-squares (of log-transformed data) to 10.7 Mt/year when a log-bias-adjusted LOWESS curve is fitted. The main difference arises where the two curves diverge at the high discharge range. Since the LOWESS curve follows the data, it provides a more accurate result.

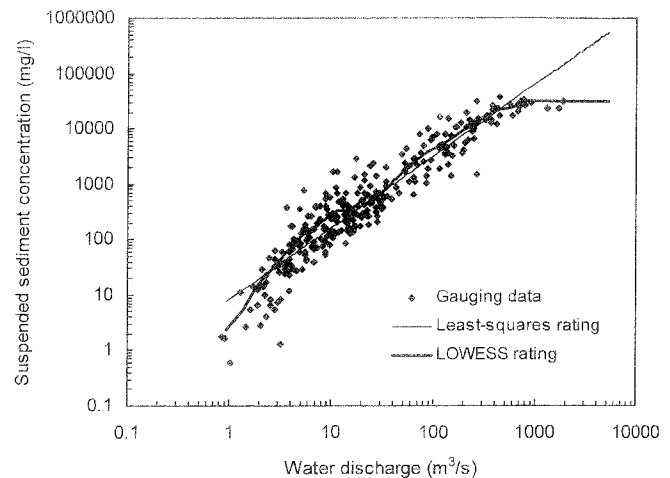


Figure 12.4 Suspended sediment rating relation for Waipaoa River at Kanakanaia Bridge. Straight line fitted by linear least-squares to log-transformed data. Curved line fitted using log-bias adjusted LOWESS (Hicks *et al.* 2000).

Determining bedload

Bedload may be estimated using a variety of approaches, including sampling, *in situ* traps, formulae, tracers, surveying deposition, or simply assuming the bedload to be a proportion of the suspended load (which is usually more easily determined). All of these approaches have their limitations, however, and it is common practice to estimate bedload using at least two approaches in order to increase confidence in the result, or to at least set bounds to the uncertainty (Hicks and Gomez 2003).

Much has been learnt about the variability of bedload transport in natural channels from bedload traps. These structures are sunk or built into the channel bed and have a cavity that captures the bedload (Fig. 12.5; Hayward and Sutherland 1974). In the more sophisticated versions, the accumulating weight is measured with a sensor (e.g., Reid *et al.* 1980). Unlike bedload sampling, which involves collecting discrete samples of bedload across the channel for limited intervals of time, traps permit a continuous record. Continuous records from gravel-bed rivers have shown that, within the span of a few hours, the bedload

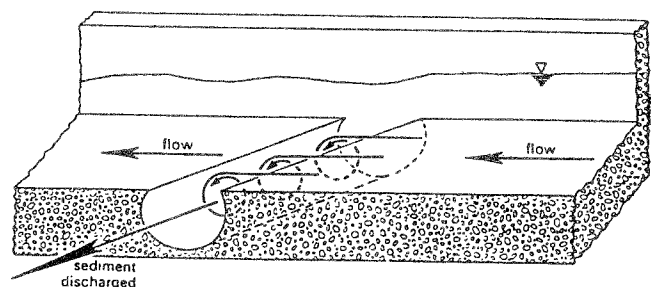


Figure 12.5 Vortex-tube bedload trap installed at Torlesse Stream (after Hayward and Sutherland 1974).

transport rate can vary from zero to ten times the average rate. Such fluctuations relate to the passage of gravel “slugs” or simply to the intermittent availability of entrainable sediment. Unfortunately, bedload traps are expensive to construct and operate and so are limited to research devices on relatively small streams.

There are two main types of bedload sampler in use: pressure-difference and basket samplers. The former have an expanding orifice upstream of the sample bag. The expansion is designed to create a pressure drop and so allow the inlet velocity to match the ambient velocity. This helps optimise the sampling efficiency (the ratio of trapped bedload to ambient bedload). The most commonly used sampler of this type is the “Helley-Smith” (Fig. 12.2b; Helley and Smith 1971), and it is suited best for sand and finer-grade gravel. Basket samplers are simply mesh boxes; they usually have a lower sampling efficiency but have been used on cobble-bed streams.

Because of the great variability that typifies bedload transport, a large number of bedload samples are required for adequate measurements. Recent research suggests that 20 to 40 samples are required for one measurement (Gomez and Troutman 1997). For this reason, sampling is usually undertaken only to develop a bedload rating curve (i.e., bedload vs. water discharge) or to verify estimates calculated by formulae.

Tracer techniques involve tagging grains with fluorescent dye, paint, magnets, or radio-transmitters (e.g., Laronne and Duncan 1992). The distance traveled by the tracer particles and their depth of burial are used to calculate the time-averaged bedload transport rate, usually for a particular flood.

Morphological approaches involve surveying bedload accumulation over some depositional area, such as a river delta or alluvial fan (e.g., Mosley 1978). Traditionally, these deposits have been measured with ground-based cross-section surveys (e.g., Griffiths 1979a). However, modern remote-sensing technologies such as digital photogrammetry and airborne laser scanning may be used to survey deposition volumes (Lane *et al.* 2003; Hicks *et al.* 2003). Morphological methods also deliver a time-averaged bedload transport estimate.

Bedload formulae relate bedload transport rate to either water discharge (e.g., Shulits 1934), shear stress (e.g., Meyer-Peter and Muller 1948), or stream power (e.g., Bagnold 1980). To a greater or lesser extent they all rely on some degree of empirical calibration from flume or field measurements. Even the best formula can do a poor job at predicting the instantaneous bedload transport rate, owing to the large variability of bedload transport rates shown in nature, even at constant discharge or shear stress (Gomez and Church 1989). Formula predictions tend to become more reliable when the result is averaged over a period of time (Carson and Griffiths 1989). Predicting bedload

transport rates in gravel-bed rivers, with bed-material grading from sand to cobbles, is a complex exercise, since the different size fractions do not always move at the same time. In such cases, the transport rate must be calculated for each size fraction (e.g., Parker 1990) or at least for the sand and gravel fractions separately (e.g., Wilcock and Kenworthy 2002), and the size distribution of the bed surface layer needs to be adjusted as the relative mobilities of the different size fractions change.

Bedload transport in rivers can be profoundly influenced by basin-scale “slugs” or waves of bed material, stemming perhaps from landslides, large storms or changes in land use (Nicholas *et al.* 1995), so bedload formulae have been incorporated in numerical models that route the sediment through basins. In the process, the models adjust local streambed levels, bed composition, and bedload transport rates according to the supply of bed-material from upstream (Pizzuto 2003).

The contribution of bedload to the total load, and so the ratio of bedload to suspended load, tends to decrease downstream as the bedload abrades and river gradients decrease. Hicks and Griffiths (1992) note that in New Zealand mountain streams the bedload can be about 90% of the total load, while further downstream in larger rivers the bedload is less than 25% of the total load. For example, on the braided Waimakariri River near Christchurch, the bedload (reliably established from historical cross-section surveys by Griffiths 1979a) forms approximately 13% of the total load. Griffiths and Glasby (1985) estimated that the bedload delivered to the coast was 3–10% of the total load.

Characteristics of the suspended load of New Zealand rivers

Data sets and analyses

Routine sediment sampling programmes in New Zealand rivers have focused on the suspended load. Sampling began in the 1950s in response to soil conservation problems, and was undertaken by hydrological survey teams of the Ministry of Works. Through the 1960s and 1970s, sampling was extended into catchments of interest for hydroelectric power development and into representative basins. In the 1980s particularly, smaller basins were investigated in order to establish the effects of land use and land-use conversion on sediment yields. Suspended-sediment sampling was resumed or begun at a selection of sites within NIWA's National Hydrologic Reference Network in the early 1990s, but has since stalled through lack of funding. Sediment sampling in recent years has been undertaken on a relatively *ad hoc* basis by regional councils and for specific research projects. NIWA maintain a national database of suspended-sediment gaugings in their Water Resources Archive (Pearson 1998).

12.6 Freshwaters of New Zealand

At the regional to national level, the data compiled up to the late 1970s were analysed by Thompson and Adams (1979), Adams (1979, 1980), and Griffiths (1979b, 1981, 1982) to estimate suspended-sediment yields. Griffiths' results were used by Griffiths and Glasby (1985) to estimate yields to the coast. More recently, Hicks *et al.* (1996) and Hicks and Shankar (2003) have analysed the

national database for data collected into the late 1990s. All of these investigators have used the sediment rating approach to estimate yields.

Suspended load characteristics of major New Zealand rivers (as determined by Hicks and Shankar 2003) are summarised in Table 12.1.

Table 12.1 Suspended-sediment concentration and yield information for New Zealand rivers. The most effective flow is the flow rate that transports more suspended sediment than any other flow.

Region	River	Catchment area (km ²)	Suspended sediment yield (t/km ² /a)	Temporal mean concentration (mg/l)	Maximum gauged concentration (mg/l)	Mean flow (m ³ /s)	Mean annual flood (m ³ /s)	Percent yield carried by flows less than mean flow (%)	Percent yield carried by flows less than mean annual flood (%)	Most effective flow (m ³ /s)
North Island northeast coast	Waitangi	302	208	24	471	8	223	0.8	66	124
	Waihou	1233	55	42	200	42	148	33.2	95	99
	Kaituna	948	55	35	289	39	131	27.5	97	61
	Tarawera	681	144	89	710	31	54	32.1	93	50
	Rangitaiki	2893	29	23	5119	71	255	16.6	74	589
	Whakatane	1557	380	88	2937	57	846	4.2	81	239
	Waioueka	661	988	70	7099	32	667	0.4	78	355
North Island east coast	Motu	1393	2531	245	14600	91	1619	1.8	87	484
	Waiapu	1378	20520	1219	49291	86	1750	0.6	76	608
	Waipaoa	1580	6753	1687	36814	35	1320	3.0	86	358
	Wairoa	1811	509	76	2381	71	1502	2.2	79	993
	Mohaka	2370	566	138	3648	81	805	2.6	62	262
	Ngaruroro	1994	661	100	7247	43	913	0.6	66	915
	Tukituki	2385	433	86	4178	45	1398	1.5	69	753
North Island west coast	Ruamahanga	2340	228	52	2210	81	882	2.4	83	929
	Otaki	305	1029	31	4030	32	952	0.5	83	807
	Manawatu	3900	820	342	7182	103	1446	6.5	86	447
	Rangitikei	2684	360	96	10542	63	687	0.8	79	395
	Whanganui	6643	687	224	5838	214	2251	4.3	88	799
	Waitara	705	1263	154	3300	34	605	1.9	80	179
	Mokau	1046	158	54	1440	37	272	4.9	87	125
Central North Island Nelson	Waipa	2826	65	36	1014	87	437	10.6	92	250
	Waikato	12421	21	19	118	370	868	29.0	95	647
	Tongariro	772	177	34	1501	41	415	2.6	87	127
	Aorere	562	174	4	407	73	2007	0.6	90	1207
	Motueka	1750	180	38	710	60	1086	3.7	85	288
	Wairoa	456	136	8	2810	16	752	0.4	71	309
	Perolus	376	336	11	3604	20	862	0.3	57	1072
South Island east coast	Waiiau	2008	1184	214	4688	97	1039	2.7	86	279
	Waimakariri	3210	989	256	7574	123	1506	5.4	82	429
	Rakaia	2582	1680	173	5577	204	2521	2.0	93	680
	Rangitata	1461	1079	147	4832	101	1049	2.6	82	604
	Shotover	1088	1208	392	11336	41	452	5.0	94	130
	Clutha	20582	28	21	337	565	1793	11.1	88	1246
	Mararoa	1191	130	26	1850	34	516	1.4	71	156
South Island west coast	Spey	99	111	4	1314	14	228	1.4	93	179
	Haast	928	4500	133	6234	191	3713	2.0	90	1149
	Hokitika	322	6313	119	31103	99	1744	1.4	96	1130
	Cropp	13	32119	894	77393	5	189	7.8	98	40
	Grey	3830	501	49	1226	357	3716	2.3	94	1234
	Buller	6350	782	81	967	429	4845	2.0	74	2326
	Inangahua	1000	366	22	568	75	1533	1.1	81	617
Mokihinui	1160	188	17	277	89	2035	2.7	97	461	

Concentrations

Peak suspended-sediment concentrations during floods and freshes typically range from a few hundred to a few thousand mg/l in rivers draining basins with moderate relief and firm bedrock. In the large rivers of Canterbury and Otago draining the eastern Southern Alps, concentrations are higher, but rarely exceed about 12,000 mg/l. On the South Island west coast, concentrations above about 6,000 mg/l are less common, because of the frequent flushing of suspended material by rainfall. However, very high concentrations, of about 77,000 mg/l, have been sampled in the Cropp Basin in the headwaters of the Hokitika River. In the North Island, concentrations of 30,000–60,000 mg/l are not uncommon in the East Cape region, where the combination of erodible rocks (e.g., mudstones and sheared argillites) and forest removal has led to severe soil erosion.

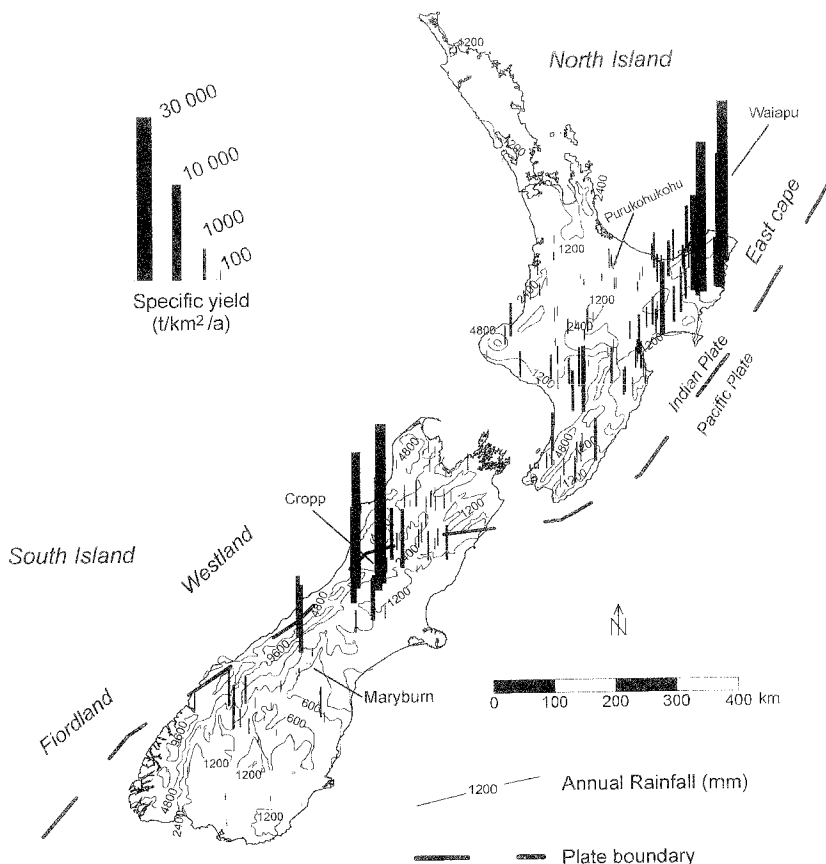


Figure 12.6 Gauged suspended sediment yields in New Zealand rivers (from Hicks *et al.* 1996).

Sediment yield and its controls

Specific suspended-sediment yields (i.e., annual sediment yield divided by catchment area) vary enormously around New Zealand (Fig. 12.6), largely in response to variations in rainfall and geology (Hicks *et al.* 1996). When sediment yields from basins in relatively uniform lithology are compared (Fig. 12.7), it is clear that both rainfall and rock type can each induce a factor of 1000 difference in yield.

In the South Island, yields range from 1.7 t/km²/year at the low-rainfall intermontane Maryburn Basin to 32,119 t/km²/year in the Cropp Basin in the high rainfall, fissile schist zone of the Southern Alps. The highest yields generally coincide with the peak of the orographic rainfall distribution along the western flanks of the Southern Alps. Despite heavy rainfall, yields from Fiordland are low (28–209 t/km²/year; Pickrill 1993), reflecting the presence of hard, erosion-resistant rocks.

In the North Island, yields range from 13 t/km²/year at the forested Purukohukohu Basin on the central volcanic plateau to 20,520 t/km²/year at Waiapu Basin in the East Cape region. In the North Island, however, where generally rainfall variations are small relative to those in the South Island, the effect of rainfall is not obvious and much of the variation appears to relate to geological factors—basement rock lithology, rate of tectonic activity, or both. Indeed, in both the North and South Islands, the highest yields tend to cluster along the boundary between the Pacific and Indian tectonic plates (Fig. 12.6). In the South Island, high

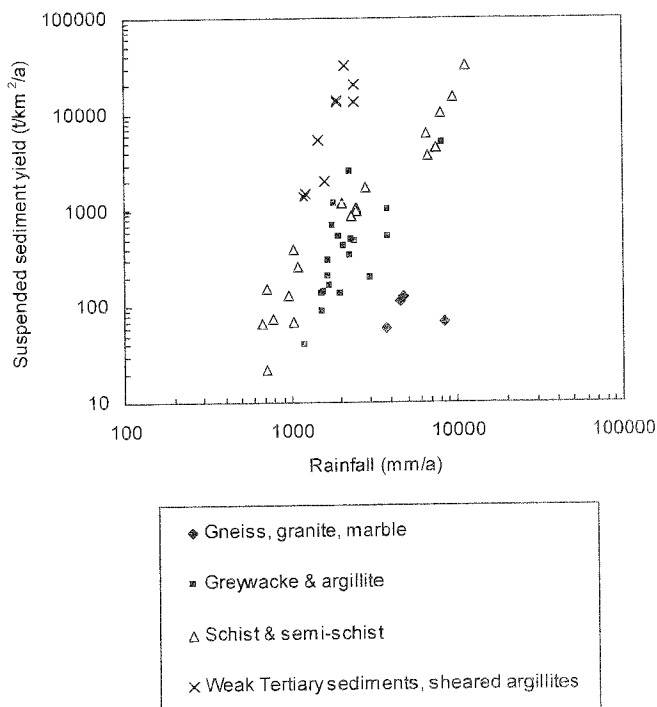


Figure 12.7 Suspended sediment yield vs. catchment mean rainfall and dominant rock-type.

yields occur in the Southern Alps on the up-thrust eastern side of the Alpine Fault (a dominant feature marking the Indian-Pacific plate boundary), while yields are much lower on the western side of the fault.

Land use, generally represented by vegetation cover, exerts further control on sediment yield. Generally, a forested basin yields something like 2–5 times less sediment than an equivalent basin under pasture (Hicks 1990; Fahey *et al.* 2003). A change in land use typically results in an initial increase in sediment yield associated with disturbance of the land surface and accelerated erosion and runoff. The yield then decays with time back towards a relatively stable level, as the vegetation associated with the new land use establishes and matures. Studies in the East Cape region (e.g., Page and Trustrum 1997) indicate that the widespread deforestation that followed European settlement may have increased sediment yields in the region by up to a factor of ten. Detailed studies of small native-forested catchments during logging operations (e.g., O'Loughlin and Pearce 1976) have shown that sediment yields may increase up to 100-fold following clear-felling. Fahey *et al.* (2003) measured sediment yields from an exotic forest in Hawkes Bay before and during harvesting. They found that the specific yield from the mature forest during the pre-harvesting phase (7.6 t/km² over two years) was one third that from a nearby pasture catchment (21.2 t/km²), but the yield from the forest during the harvesting phase (84.4 t/km² over 17 months) was twice that from the pasture catchment (43.8 t/km²). Hicks and Harmsworth (1989) estimated that the several-year period of harvesting activity of an exotic forest near Glenberrie, Northland, provided approximately 70% of the total sediment yield over the 32-year growth-logging cycle. Generally, sediment yields fall to lower levels within a few years of forest clearance, as vegetation re-establishes, but in the East Cape case high yields have continued for a century and relate in part to the formation of large gully complexes following forest removal. Urbanisation can induce similar temporary increases in sediment yield (Williamson 1993).

Land tilling or disturbance for agriculture may or may not increase stream sediment loads. While tilled fields can erode severely during rainstorms, the proportion of the mobilised sediment that is delivered into watercourses (as against being deposited on-field at slope breaks or in vegetated riparian zones) can vary substantially. For example, Basher *et al.* (1997) showed, from a sediment budgeting study, that while market gardening fields near Pukekohe eroded at a rate of 5680 t/km²/year, less than 1% of this entered the drainage system.

Particle size

The particle-size distribution of the suspended load also varies widely. It depends mainly on the catchment rock

type but also varies with flow conditions. Averaged size distributions from rivers draining a variety of lithologies (Fig. 12.8) show that the suspended load tends to be dominated by clay and silt grades (i.e., washload) in basins with glaciers, loess deposits, bedrock composed of mud-silt grade sediments, or well-weathered bedrock. Sand begins to form a significant proportion of the suspended load in rivers draining relatively un-weathered schist and greywacke, while it dominates the suspended load from basins formed in volcanoclastic material and coarsely-crystalline rocks such as granite and gneiss.

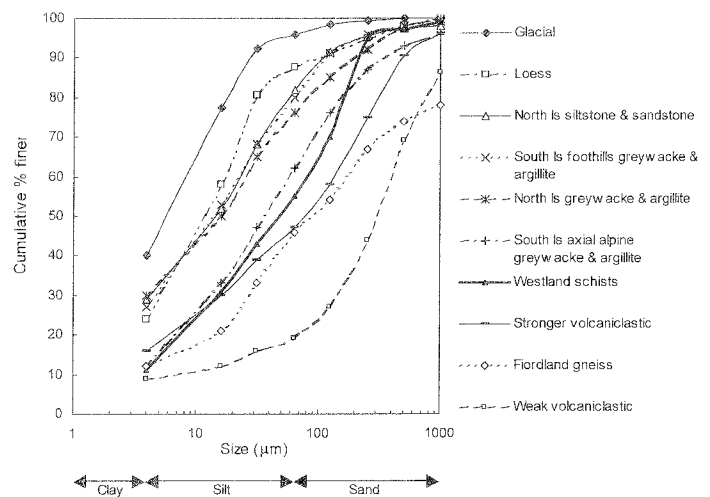


Figure 12.8 Averaged particle-size distributions of suspended load from rivers draining various geological terrains.

Fluxes to the coast

Hicks and Shankar (2003) assessed suspended-sediment yields to the New Zealand coast (Fig. 12.9). Their yield estimates for approximately half of the country were based on sediment rating curves and flow records established at river gauging sites. They used an empirical model to predict yields from ungauged basins. The model relates specific suspended-sediment yield to mean annual rainfall and an erosion-terrain classification based on slope, rock type, soils, main erosion processes, and expert knowledge. The model was calibrated with river suspended-sediment gaugings from over 200 sites and with data on sedimentation rates in Fiordland from Pickrill (1993).

River suspended-sediment delivery to the New Zealand coast totals 209 Mt/year. This is approximately evenly split between the North and South Islands (91 Mt/year and 118 Mt/year, respectively). The highest yields from the North Island are to East Cape (Fig. 12.9). These reflect a combination of moderately high rainfall, erodible rocks such as mudstone and sheared argillites, tectonic uplift close to a convergent plate boundary, and almost complete removal of the native forest cover. The region accounts for

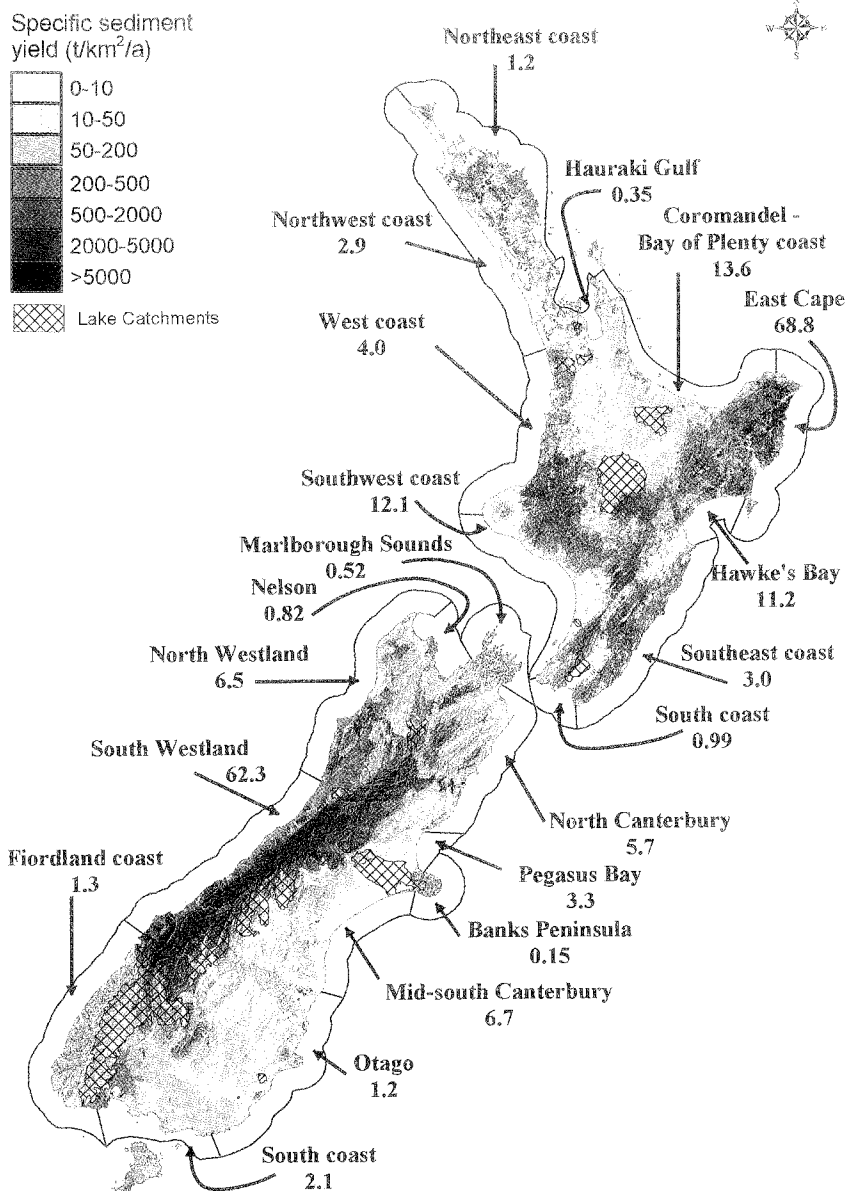


Figure 12.9 Suspended sediment yields to New Zealand coast (Mt/a). Grey scale indicates sediment sources in $t/km^2/a$; hatched areas are lake catchments that intercept sediment delivery to the coast (after Hicks and Shankar 2003).

58% of the North island yield, while the Waipau River alone delivers 35 $Mt/year$. The relatively low-relief areas of the northeastern North Island generate comparatively little sediment (e.g., only 0.35 $Mt/year$ is delivered to the Hauraki Gulf).

In the South Island, the highest suspended-sediment yields are from the western flanks of the Southern Alps in South Westland. There, a combination of steep slopes, heavy rainfall, high uplift rates along the eastern side of the Alpine Fault, and erodible schist lithology generate a regional yield (62 $Mt/year$) amounting to 68% of the South Island yield. In contrast, and despite heavy rainfall, the sediment yield in Fiordland is very low (1.3 $Mt/year$).

This reflects the hard plutonic-gneissic terrain, almost complete forest cover, and the scouring of regolith by Pleistocene glaciations.

The large natural lakes of the South Island trap 15 $Mt/year$ of suspended sediment, compared to less than 1 $Mt/year$ for North Island lakes. Hydro-lakes in the South Island's Clutha and Waitaki catchments intercept 2.4 $Mt/year$, while dams on the Waikato and Rangitaiki Rivers trap some 0.4 $Mt/year$. In total, natural lakes entrap 15% of the potential sediment delivery to the New Zealand coast, while hydro-dams have reduced the natural suspended sediment yield to the coast by 1.3%.

Globally, New Zealand is one of the major producers of sediment, accounting for about 1% of the input to the world's oceans from less than 0.2% of the land area.

A previous estimate of suspended sediment yield to the New Zealand coast was made by Griffiths and Glasby (1985), using the empirical relations of Griffiths (1981, 1982) to estimate yields from ungauged areas. Their yield to the North Island coast was much the same as that determined by Hicks and Shankar (2003), but their South Island yield was three times larger. The largest difference occurs along the Fiordland coast, and is likely due to the paucity of data available to Griffiths and Glasby at the time.

Magnitude-frequency characteristics of suspended-sediment yields

The frequency distributions of sediment yields tell which flows do the most work in transporting sediment and also provide clues about sediment availability. Analysis of the proportion of the total suspended load transported within specific discharge bands from over 200 sites around New Zealand (examples given in Table 12.1) shows that, on average, the most effective flow (i.e., the flow transporting the most sediment) equates to 13 times the mean flow or alternatively to 0.57 times the mean annual flood. Also, on average, 76% of the suspended sediment yield is carried by flows between the mean flow and the

mean annual flood. In other words, flows that occur during relatively common freshes and floods are more important in the long term than are the rare large floods. This is a feature of basins with temperate climates worldwide (e.g., Wolman and Miller 1960).

The magnitude-frequency characteristics of sediment yields during discrete runoff events can convey information on sediment supply within a basin and also the characteristic erosion processes. For example, in the Waipaoa Basin, Hicks *et al.* (2000) found that, for tributaries dominated by gullying, the magnitude-frequency distribution for event yields was flatter than that of tributaries in which the characteristic erosion process was landsliding (Fig. 12.10). In the gullied basins, sediment was readily washed into stream channels during even small rainstorms, so these events were relatively important. In contrast, substantial rainfall was required before landslide erosion was activated, so in the landslide-dominated basins the less frequent larger events transported more sediment.

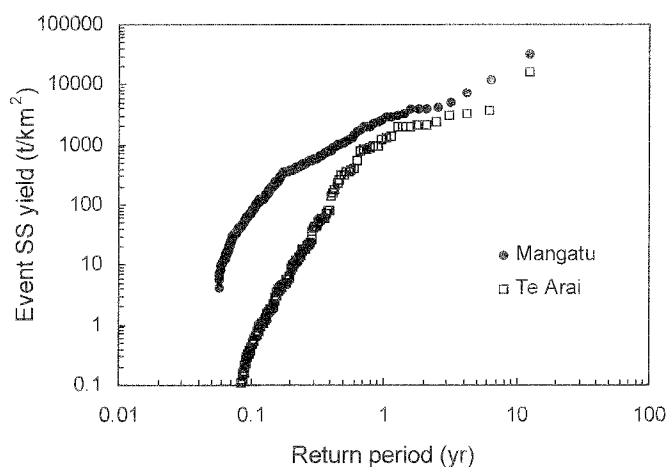


Figure 12.10 Magnitude-frequency relationships for event suspended-sediment yield for Te Arai River (landslip-dominated) and Mangatu River (gully-dominated), East Cape.

Hicks (1994) found that in an urban basin, event yields increased relatively little as return period increased, due to sediment exhaustion. In contrast, yields increased substantially with return period in a nearby basin undergoing urban development, due to easy erosion of bare-earth subdivisions.

ORGANIC MATTER

Particulate organic matter, dissolved organic carbon and large woody debris

Particulate organic matter in streams ranges from fragments of eroded soil and leaf litter to large wood debris from riparian forests. Organic matter is normally measured

as dry mass, organic (ash-free) dry mass, or carbon (c. 50% of organic dry mass). The terms organic matter and organic carbon are used interchangeably. It is often defined operationally by size—e.g., dissolved, fine, coarse (>1 mm), small wood, large wood (>10 cm in diameter), or by source—as allochthonous (synthesised elsewhere) or autochthonous (synthesised locally).

Catchment-derived particulate organic carbon (i.e., soil carbon, plant debris) is termed allochthonous. Particulate organic carbon produced in-stream by bacteria, periphytic algae, and rooted plants on the streambed is termed autochthonous. Stream carbon is cycled between particulate organic carbon, dissolved organic carbon, dissolved inorganic carbon (= $\text{HCO}_3^- + \text{CO}_2$), and CO_2 in the atmosphere (Fig. 12.11). Streamflow interacts with this cycling to produce “carbon spiraling” as material moves downstream.

The relative importance of catchment-derived vs. in-stream sources of organic matter generally decreases with distance down forest streams as the flow depth increases, reducing the trapping of leaf litter (Broekhuizen and Quinn 1998), and the channel widens, increasing sunlight for photosynthesis (Davies-Colley and Quinn 1998) and reducing litter input (Winterbourn *et al.* 1984). In lakes, autochthonous photosynthesis by attached algae and rooted plants on the bed and margins and/or algae in the water column (phytoplankton) are usually the key sources of organic matter for food webs (James *et al.* 2000). This autochthonous lake production may constitute a substantial component of the particulate matter suspended in lake-outflow rivers and often results in an abundance of filter-feeding invertebrates in these rivers (Harding 1992).

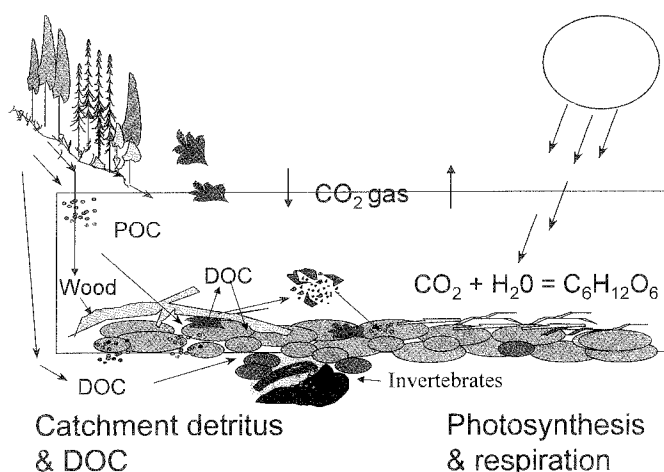


Figure 12.11 Summary of main factors influencing in-stream carbon dynamics. POC is particulate organic carbon and DOC is dissolved organic carbon.

Wood in New Zealand streams

A small but growing body of research on the abundance and role of wood in New Zealand streams suggests that wood can be relatively abundant and functionally important. In-channel wood volumes measured in pristine native forest streams less than 6 m wide range from 6 m³/ha to 469 m³/ha (Mosley 1981; Evans *et al.* 1993; Quinn *et al.* 1997; Baillie and Davies 2002; Mark Meleason pers. comm.). This range is similar to that found in northern Colorado streams (93–254 m³/ha in Richmond and Fausch 1995) and streams in the Smoky Mountains of Tennessee (70–300 m³/ha in Harmon *et al.* 1986). In-channel volumes measured in pine plantation streams prior to harvest or in abandoned pine plantations range from 0 m³/ha to 327 m³/ha (Evans *et al.* 1993; Quinn *et al.* 1997; Baillie *et al.* 1999; Baillie and Davies 2002).

Large woody debris in both pine and native streams has been found to influence channel morphology and pool formation (Mosley 1981; Quinn *et al.* 1997; Baillie and Davies 2002) and provide habitat for aquatic invertebrates (Tank and Winterbourn 1996; Collier *et al.* 2003; Collier and Smith 2003). Wood is particularly important as habitat for periphyton and invertebrates in soft-sediment streams in lowland and pumice catchments, where it provides stable surfaces for colonisation and retention of finer organic matter (Collier *et al.* 2003).

Effects of land use on organic matter dynamics and export

Changing land use from forest to pasture alters stream organic matter dynamics, including the rate and type of allochthonous inputs (via changes in riparian vegetation, wetlands, and soil erosion) and autochthonous inputs (via changes in shade and nutrient concentrations). Change in land use can also reduce in-stream retention of organic matter via its effects on channel width (narrower in pasture), stream bed particle size (often smaller in pasture) and in-stream wood (less in pasture).

Extensive studies at Whatawhata, in the western hills of the Waikato Basin, showed that pasture streams received less leaf litter input than forest streams. Vertical (litter fall) and lateral (wash/blow in) particulate organic matter inputs to Whatawhata streams increased as the percentage canopy cover increased (Fig. 12.12). Pine and native forest streams had similar seasonal patterns of litter input (greatest in summer, least in winter), whereas pasture streams dominated by deciduous trees had greatest input in autumn (Scarsbrook *et al.* 2001). On the other hand, measurements (Quinn *et al.* 1997) and computer simulations run for several years' conditions (Quinn and Broekhuizen 2003) indicate that higher light and nutrient levels in pasture result in c. 10-fold higher instream production of attached algal (periphyton) biomass than in

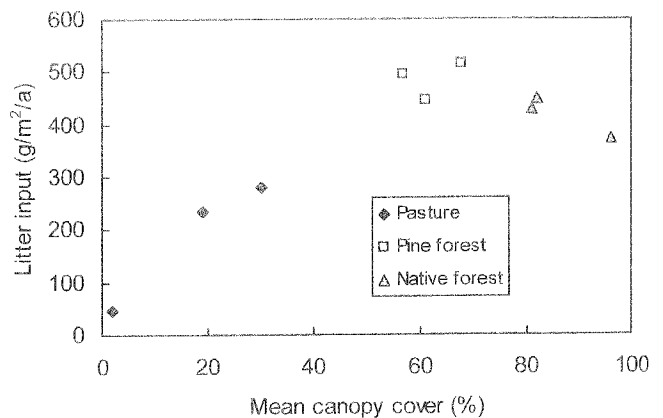


Figure 12.12 Relationship between canopy cover and total input of coarse particulate organic matter to Waikato headwater streams in contrasting landuses (adapted from Scarsbrook *et al.* 2001).

native forest. Pasture streams also receive greater inputs of erosion carbon (bound to sediment particles). Particulate organic carbon export was estimated to be 4- to 10-fold higher in pasture (41–51 kg/ha/year) than in forest streams for a dry year (Quinn and Broekhuizen 2003). This was based on measurements of suspended sediment and total organic nitrogen exports (Quinn and Stroud 2002) and stream-specific relationships between particulate organic carbon, suspended sediment and total organic nitrogen. Dissolved organic carbon export was also 25% higher from pasture (25 kg/ha/year) than from native forest (Quinn and Stroud 2002).

The change in land use from native forest to pasture at Whatawhata has reduced the amount of wood in stream channels by c. 80% by volume (means 12 m³/ha c.f. 50 m³/ha in native forest—Quinn *et al.* 1997). Wood was more abundant in streams in 15-year old pine plantations (planted to the stream edge, mean 200 m³/ha) than in streams in native forest. These values indicate that wood represents a significant carbon store in Whatawhata streams, with 200 g C/m² in pasture, 1450 g C/m² in native forest, and 5,000 g C/m² in pine forest streams. These figures assume a specific gravity of 0.58 for native wood (tawa; Hinds and Reid 1957) and 0.4 for pine and pasture wood (Harmon *et al.* 1986), and assume that carbon is 50% of wood dry mass. Studies of old growth forests in Alaska indicate overall wood input and loss (dissolved organic carbon leaching, fragmentation, respiration and export) rates of 1–3% (Murphy and Koski 1989).

Some of this wood will be exported as leached dissolved organic carbon and fine particulate organic carbon fragments generated from invertebrate grazing and abrasion, and thus will be measured in catchment monitoring of these components. However, an unknown fraction will also be exported as logs and branches during extreme storms, and this has not been measured. Logs are expected to be a

small item in the pasture catchment carbon export budget, due to minimal inputs, but they may play a more significant role in forest streams.

Soil organic carbon fluxes: within the landscape and out to the coast

The flux of sediment through the drainage network is a vector for the redistribution of a whole range of nutrients and contaminants. Not least among these is soil organic carbon (which is a substantial component of the particulate organic carbon). As illustrated in Figure 12.13, carbon generated in soils (termed labile C) or recycled from erosion of rocks (recalcitrant C) can follow a number of pathways and rest in a range of storage sites before being transported into the ocean via the river network. There are also reverse fluxes, such as where landslide scars re-vegetate. Carbon may also be cycled directly to the atmosphere as CO₂.

Soil erosion is a key process in providing organic carbon into streams. In New Zealand, the extensive deforestation and establishment of pastoral agriculture between 1860 and 1920 initiated marked increases in soil erosion over many areas of the country (Page and Trustrum 1999). Trustrum (1999) used a sediment budget of the Waipaoa River basin to demonstrate that the high rates of anthropogenic erosion were also likely to lead to significant erosion-related carbon transfers. That study also showed that because the largest proportion of soil carbon is usually stored in the upper soil horizons, there is an inverse relationship between the depth of erosion and the proportional loss of carbon from hillslopes.

Similarly, using the high-resolution record of storm-induced erosion from the period of pastoral settlement that is preserved in sediments on the bed of Lake Tutira, Page *et al.* (in press) found that although landsliding generated

about 75% of the sediment entering stream runoff, it delivered only about half the soil carbon yield. The other half was associated with sheetwash, which acts on surface soils that have higher carbon contents than the material removed by landsliding. This research also showed that, due to sequestration of eroded material on slopes and in storage sites such as lakebeds and floodplains, net losses of carbon down the stream network were only about one-quarter of the total carbon eroded by landslide and sheet erosion.

A recent multi-disciplinary study (Trustrum *et al.* 2002) used a combination of empirical and process-based approaches to quantify the flux of soil organic carbon to the ocean from the New Zealand landmass. This combined knowledge of sediment yields to the ocean, the spatial distribution of soil organic carbon in the landscape, and the predominance of various erosion processes. The approach was to multiply the suspended-sediment erosion rates (illustrated in Figure 12.9) with a soil organic carbon loading and an erosion-depth factor. Soil carbon loading was mapped across New Zealand using data compiled within the national Soil Carbon Monitoring System (Scott *et al.* 2002), interpolating between measurement sites on the basis of rainfall, slope and vegetation type. With this, soil organic carbon contents were mapped for three depths of soil: 0–0.1 m, 0.1–0.30 m and 0.3–1.0 m. As discussed above, the amount of soil organic carbon mobilised depends on the depth of erosion, which is determined by the erosion process. The erosion depth factor was mapped using the New Zealand Land Resource Inventory (Fletcher 1988) to identify erosion type and severity and using expert knowledge to infer depth of erosion for each process on the basis of its mapped severity. The resulting source map of erosion-derived particulate organic carbon is shown in Figure 12.14.

This exercise produced a carbon yield to the ocean of ~4 Mt/year, which is comparable to estimates of potential CO₂ sequestration in plantation forestry (Tate *et al.* 2000). The highest yields of carbon are from the west coast of the South Island and the east coast of the North Island (Fig. 12.14). On the South Island West Coast, the high yields are considered to be natural: being a forested area, the soils there have a high carbon content and the area has naturally high rates of erosion. On the east coast of the North Island, however, where changes in land use over the last century are considered to have exacerbated already high rates of erosion, the high carbon fluxes have a substantial anthropogenic component.

Lyons *et al.* (2002) note that New Zealand, along with other high-standing islands of the

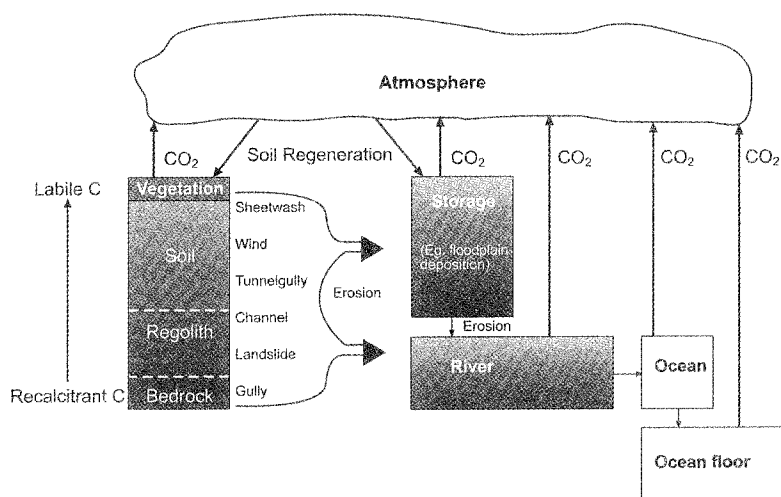


Figure 12.13 Carbon flux pathways between the soil and ocean floor.

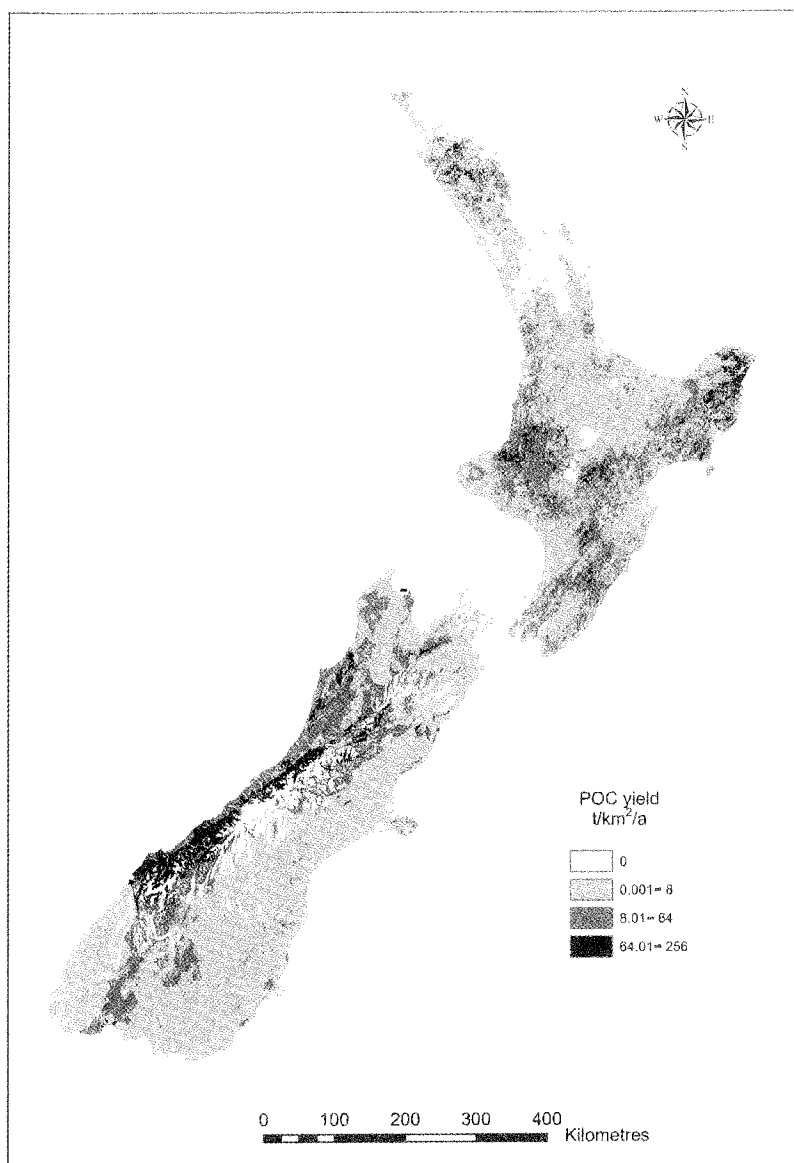


Figure 12.14 Specific yields of particulate organic carbon (as supplied from soil erosion) into the New Zealand river network (from Trustrum *et al.* 2002).

southwest Pacific (e.g., Indonesia, Taiwan, Papua New Guinea), may contribute as much as 35% of the global supply of particulate organic carbon to the oceans—from an area ~ 3% that of the continents.

SUMMARY

The sediment load of a river is a primary control of channel form and, when supplied in excess or deficit, can lead to a variety of problems relating to erosion or deposition. It may be classified as suspended load or bedload according to how it is transported by the flow. The suspended load is typically sampled with a depth-integrating sampler, and, since it is a primary determinant of stream turbidity, it may often be monitored using turbidity as a proxy. The rating relationship between

suspended-sediment concentration and water discharge may be used with a flow record to estimate the long-term average yield of suspended sediment. The bedload may be sampled, trapped, surveyed, or measured with tracer grains, but most often it is either calculated by formula or assumed to be equal to a proportion of the suspended load.

Measurements of the suspended load in New Zealand's rivers have shown great spatial variability in sediment concentration and specific yield. By and large, this variability relates to rainfall, geology and, to a lesser extent, land cover and land use. Suspended-sediment particle size relates mainly to catchment rock type. Changes in land use and forest harvesting result in phases of increased sediment yield that may last from several years to many decades. The flux of suspended sediment carried by rivers to the New Zealand coast totals some 209 Mt/year. Much of this is derived from the East Cape and South Westland regions, and most is transported at flow rates between the mean flow and the mean annual flood flow.

Organic material in rivers includes fine catchment-sourced particles, such as fragments of eroded soil and leaf litter, fragments generated by in-stream biologic activity, and large woody debris. The large woody debris can be a major control on the morphology of small streams and is an important provider of in-stream habitat. Land use, and changes in land use, can exert substantial control on the dynamics of in-stream organic matter, for example by affecting the supply of large woody debris, leaf

litter and carbon from soil erosion, and by affecting the growth of algae through the supply of nutrients and light.

The supply of organic carbon to streams by soil erosion is affected by the type and depth of erosion process, since the soil carbon is concentrated in the topsoil layers. The flux of particulate carbon carried to the New Zealand coast by rivers is approximately 4 Mt/year. The greatest supplies stem from the forested South Island west coast and the North Island east coast, where historical changes in land use have accelerated naturally high soil erosion rates.

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Chapter 11

Water quality and chemistry in running waters

Rob Davies-Colley and Bob Wilcock

INTRODUCTION

The composition and chemical characteristics of river water strongly influence its suitability for aquatic life and its use by humans. In this chapter we discuss research since the last nation-wide reviews of water quality in New Zealand, namely a chapter on water quality by Hoare and Rowe (1992) in "Waters of New Zealand" and the Ministry for the Environment (1997) "State of the New Zealand Environment" report. We consider first the composition of running waters and their chemical "behaviour", and then discuss water quality of New Zealand rivers and streams with reference to current guidelines.

COMPOSITION OF RIVER WATERS

An extremely wide variety of substances may occur dissolved or suspended in river and stream waters: these materials may be categorised as dissolved gases, non-volatile solutes, and particulates (suspended solids) (Table 11.1).

Gases

Running waters, precisely because they are running and thus turbulent, tend to equilibrate with the atmospheric gases nitrogen and oxygen, and a wide variety of atmospheric trace gases, most of which need not concern

Table 11.1 Major and minor constituents of running waters

Category	Constituent	Importance
Dissolved gases	Oxygen (O ₂) Carbon dioxide (CO ₂)	Respiration of aquatic life Source of inorganic carbon assimilable by plants
Non-volatile solutes	Major inorganic constituents (e.g., Na ⁺) Dissolved inorganic phosphorus Dissolved inorganic nitrogen Dissolved organic matter (DOM)	Important in (salinised) arid lands, but of little concern in NZ. Sometimes useful as a water tracer. Important plant nutrient, often limiting growth in waters Important plant nutrient, often limiting growth in waters Degradation lowers oxygen concentration, contains nutrients (C, N & P), absorbs light (shifts water colour), photochemical 'catalyst'
Particulates	Mineral solids Organic solids Microbiological contaminants	Major component of sediment load, scatter light and transport sorbed pollutants (esp. clay minerals) Component of the sediment load, pool of the nutrients C, N, P that may be mineralised by bacterial action, absorb and scatter light. Pathogenic micro-organisms render water unsafe for consumption, contact recreation, and shellfish harvesting.

us here. Dissolved nitrogen gas is almost irrelevant to water quality, but dissolved oxygen (DO) is extremely important. Oxygen is only slightly soluble in water (its “saturation” solubility is around 10 g/m^3 , depending on temperature), but it is essential for aerobic forms of river life. This “slight solubility” presents a problem for aquatic life, because oxygen-consuming respiration by aquatic life, ranging from bacteria to large animals and plants, tends to deplete the dissolved oxygen in water. Consequently, if there is any delay in re-aeration processes that tend to re-establish equilibrium at “saturation” concentration, oxygen can fall to dangerously low levels (the Resource Management Act 1991 standard is a minimum of 80% saturation.) Conversely, production of oxygen by aquatic plant photosynthesis can increase oxygen well above saturation level in stream waters that are comparatively poorly-aerated. Models capable of representing the competing processes of reaeration, respiration, and photosynthesis have been developed for predicting oxygen dynamics in streams and rivers, and inverted forms of these models may be used for calculating rates of re-aeration, respiration, and photosynthesis from diurnal records of dissolved oxygen (Wilcock *et al.* 1998).

Dissolved carbon dioxide is much more soluble in water than oxygen is, and inorganic carbon is typically present at concentrations comparable to oxygen, even though its atmospheric abundance is 600-fold lower. Inorganic carbon in waters provides an essential nutrient (carbon) in a form that can be used by aquatic plants to synthesise organic carbon during photosynthesis. The “carbonate system” and its relationship to CO_2 and pH in rivers is discussed in the section below on chemical processes.

Non-volatile solutes

The main non-volatile solutes in running waters, in common with other freshwaters, are the two anions Cl^- and SO_4^{2-} , dissolved silica (uncharged SiO_2), and the four cations Na^+ , K^+ , Ca^{2+} , and Mg^{2+} (Hoare and Rowe 1992) (the third, and most abundant anion, bicarbonate HCO_3^- is volatile). Although these constituents dominate the total dissolved solids content of river waters, they are seldom of much concern in New Zealand from a water quality viewpoint. However, they are of interest to geochemists in terms of riverine flux to the oceans (e.g., Meybeck 1987), and may also be useful for tracing water masses, particularly river plumes in lakes. The major ion composition of some New Zealand rivers, notably the Waikato, is influenced by geothermal inflows.

Of greater interest than the major ionic constituents are the inorganic forms of the nutrient elements phosphorus (P) and nitrogen (N): phosphates, nitrate-N, nitrite-N, and ammoniacal-N. When over-abundant, they stimulate excessive plant growth and are important pollutants of waters. A wide range of inorganic constituents present at

trace concentrations in river water may sometimes be of interest—usually because of their toxicity (Hoare and Rowe 1992). In New Zealand, geothermal discharges may cause high levels of the toxic elements arsenic, boron, lithium and mercury, and levels of various toxic metals such as zinc may be elevated by mining operations and may be high in urban storm water (Smith 1986).

Dissolved organic material (DOM) in river waters includes a vast range of natural and anthropogenic compounds that are seldom individually characterised. Instead various “lumping” measures such as dissolved organic carbon (DOC) are assayed. Dissolved organic material is a major pool of the nutrient elements carbon, nitrogen and phosphorus. The oxygen that may be consumed in a river by heterotrophic bacteria engaged in metabolising organic matter is of particular interest in organic wastewater discharges and is indicated by the 5-day biochemical oxygen demand test (BOD_5) (Hoare and Rowe 1992). The BOD_5 test is usually applied to unfiltered water samples, although it may be measured on a membrane-filtered sample as an index of easily degradable dissolved organic material. Typically, most of the dissolved organic material in river waters is very refractory in the biochemical oxygen demand BOD test, as it comprises macromolecules similar to soil humus, to which it is related (Thurman 1985). The macromolecular dissolved organic matter in river waters, in common with that in lake waters (Chapter 21), is light-absorbing (“coloured dissolved organic matter”, CDOM) and is of interest for its influence on optical properties (light penetration and colour) and photochemical behaviour.

Particulates

Most of the remaining important constituents of water are not dissolved, but are present as suspended solid particles (Table 11.1). Inorganic (mineral) particles can be distinguished from organic particulates. Mineral particles are comparatively dense (e.g., quartz is 2.65 times as dense as water) and so sink rapidly, even in turbulent river waters, unless they are very fine-grained (typically, mineral particles suspended in rivers are $< 63 \mu\text{m}$ diameter, Hicks and Griffiths 1992). Finely-divided silicates and oxides, and clay minerals (layer silicates) derived from weathering dominate mineral sediments, and may be classified into clay ($< 2 \mu\text{m}$) and silt ($2\text{--}63 \mu\text{m}$) size ranges. Suspended mineral particles are important because they scatter light intensely and reduce the visual clarity of running waters, causing them to appear cloudy or turbid (Davies-Colley and Smith 2001). Fine suspended particles may also act as carriers of other pollutants, notably sorbed phosphorus and various trace organic and metal biocidal materials.

Particulate organic matter (POM, “organic detritus”) in running waters is of great importance because, along with dissolved organic material, it contains large amounts

of the nutrient elements, especially carbon, nitrogen and phosphorus, in a fixed form. Degradation (mineralisation) of this organic matter mobilises nutrient elements, initiating a process sometimes referred to as nutrient “spiralling” (refer to section on water chemistry below). Organic particles in running waters are at least partly composed of heterotrophic bacteria and other microorganisms engaged in degrading or mineralizing the organic matter (and exerting a biochemical oxygen demand—BOD) in doing so. Much of the particulate matter in running waters is neither purely inorganic nor organic, but a mixture of both, aggregated into flocs bound together by surface forces and “sticky” organic polymers of bacterial origin (Droppo 2001). Microbiological pathogens may be regarded as a particular type of “organic particulate”, of special interest for public health (Table 11.1), and include the parasites *Cryptosporidium* and *Giardia* (occurring as resistant cysts), virus particles such as hepatitis-A virus, and various bacteria, notably *Campylobacter* spp.

PHYSICAL VARIABLES

Besides the material composition of running waters, we are also concerned with their physical properties as they affect conditions for aquatic life and water uses. The most important physical properties are temperature and optical properties.

The temperature of running waters controls the rates of chemical and biochemical reactions and the position of various equilibria, such as the saturation solubility of gases, notably oxygen. Water temperature and its variation has a direct influence on the suitability of a habitat for aquatic organisms (the Resource Management Act standard is a maximum of 3°C change). Temperature increases in rivers due to waste heat discharges from geothermal or thermal power stations or industries can sometimes be environmentally damaging (Hoare and Rowe 1992). Temperatures typically fluctuate diurnally around a (seasonal) mean that expresses the balance of heat loss by back radiation (long-wave or thermal infra-red), evaporation and convection; and heat gain by solar radiation (short-wave, visible plus near infra-red) (Rutherford *et al.* 1997). The seasonal mean water temperature of New Zealand rivers is highest in summer and lowest in winter (Mosley 1982). Diurnal temperature change is driven by solar radiation, usually with a faster rise to the mid-afternoon maximum temperature than fall to the minimum near dawn (Fig 11.1).

Large rivers have considerable thermal inertia, consequently their diurnal thermal range is small. However, in small, unshaded streams in New Zealand, temperature in the mid-afternoon in summer often reaches the high 20s (Fig. 11.1), which can stress aquatic life (Quinn *et al.* 1994; Cox and Rutherford 2000a,b).

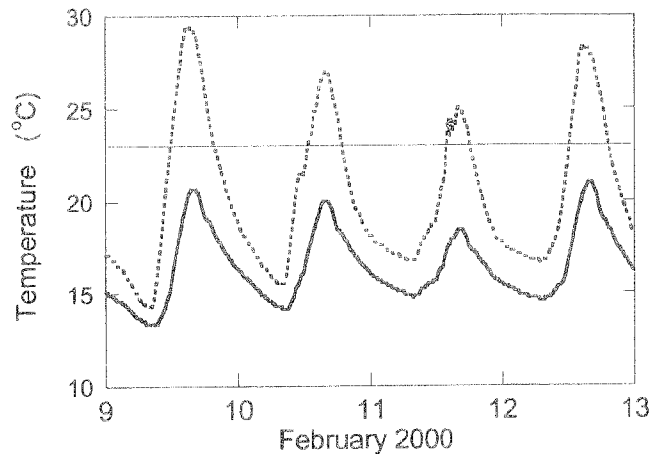


Figure 11.1 Diurnal temperature change in an unshaded pasture stream (dotted line) contrasted with that in a nearby shaded native forest stream (continuous line), Te Arai Stream, south of Gisborne (authors’ unpublished data). The line at 23°C indicates a value above which stoneflies are likely to be excluded.

Optical properties control the behaviour of light in running waters, in common with waters generally. Chapter 21 discusses the optics of lake waters in more detail. Less attention has generally been given to the optics of river water, possibly because aquatic plant growth in shallow running waters is rarely limited by the availability of light. Light penetration has seldom been measured in New Zealand rivers, but Davies-Colley *et al.* (1993) give an indirect calculation suggesting that our rivers are usually “optically shallow” (i.e., their beds are usually shallower than the euphotic depth, which is defined in Chapter 21 as the depth at which light is reduced to 1% of incident light). Shading of comparatively small streams by riparian vegetation is often the major light-attenuating factor limiting aquatic plant growth (e.g., Davies-Colley and Quinn 1998).

Visual water clarity is important for aquatic life and for human use (discussed below in the section on guidelines). Black disc visibility, a measure of visual water clarity based on horizontal observation of a submerged black disc, yields an estimate of the beam attenuation coefficient, which is an inherent optical property that depends only on water composition and not the ambient light field (Davies-Colley 1988). The horizontal sighting range of the black disc is similar to visual ranges in water that are of practical importance for aquatic animals as well as man. Black disc visibility is routinely monitored in the National Rivers Water Quality Monitoring Network (NRWQN), Smith and Masdaam 1994). Also measured routinely in the NRWQN are two laboratory optical measurements: nephelometric turbidity, an index of light scattering by suspended particles that correlates inversely with black disc

visibility (Davies-Colley and Smith 2001), and the absorption coefficient of filtered water samples at 440 nm (a_{440} , Davies-Colley *et al.* 1993), an index of coloured dissolved organic matter (CDOM) or “yellow substance”.

CHEMICAL PROCESSES

Water chemistry is concerned with the description of chemical reactions and processes affecting the distribution and circulation of chemical species in natural water (Stumm and Morgan 1996). Chemical processes can be subdivided into *equilibrium processes* affecting the distribution and relative proportions of chemical substances, and *kinetic processes* affecting the rates at which chemical changes occur.

Equilibrium processes are independent of time and are characterised by constants that fix the proportions of concentrations at specified conditions (notably temperature). For example, carbon dioxide dissolves in water to form carbonic acid H_2CO_3 , which dissociates to produce bicarbonate (HCO_3^-), carbonate (CO_3^{2-}) and hydrogen ions (H^+); concentrations of all these species are related by equilibrium constants (Butler 1991). Tables of equilibrium constants (Table 11.2) enable calculation of equilibrium concentrations. For example, Figure 11.2 shows the distribution of dissolved inorganic carbon (DIC) species in water. Dissolved organic carbon is important in aquatic ecosystems as the source of carbon for aquatic plants and because it buffers water against pH changes. Changes in pH affect solubility and speciation of compounds, thereby affecting their bioavailability and toxicity. Running waters in New Zealand are mostly dominated by bicarbonate and have a total dissolved organic carbon concentration of $\sim 10^{-3}$ mole/l (Close and Davies-Colley 1990a). The pH range of 7–8 typical of New Zealand rivers reflects pH buffering by bicarbonate, which is converted to carbonate by addition of a base and to carbonic acid on addition of an acid (Fig. 11.2).

During periods of active plant photosynthesis, dissolved CO_2 is taken up by plants and converted to organic matter (Equations 1 and 2 below) producing O_2 and consuming acid (H^+), causing the pH and dissolved oxygen to rise during photosynthesis (Fig. 11.3). The reverse happens at night when respiration is dominant. The consumption of CO_2 in photosynthesis cannot be compensated for by conversion of bicarbonate to CO_2 , because alkalinity (dominated by HCO_3^-) is a conservative quantity (Stumm and Morgan 1996). Therefore, uptake of dissolved CO_2 means that H^+ must fall (pH rises) to maintain equilibrium (refer K_1 in Table 11.2).

Physical processes such as the solubility of dissolved atmospheric gases (e.g., O_2 and CO_2) and the evaporation of water at different temperatures are also described by equilibrium constants. Kinetic processes are time-

Table 11.2 Equilibrium constants for dissolved inorganic carbon at 25°C

Equilibrium expression	Equilibrium constant
$[CO_2(aq)]/[H_2CO_3]$	$K = 650$
$[H^+][HCO_3^-]/[H_2CO_3^*]$	$K_1 = 10^{-6.35}$
$[H^+][CO_3^{2-}]/[HCO_3^-]$	$K_2 = 10^{-10.33}$
$[H^+][OH^-]$	$K_w = 10^{-14.0}$
$[H_2CO_3^*] = [CO_2(aq)] + [H_2CO_3]$	

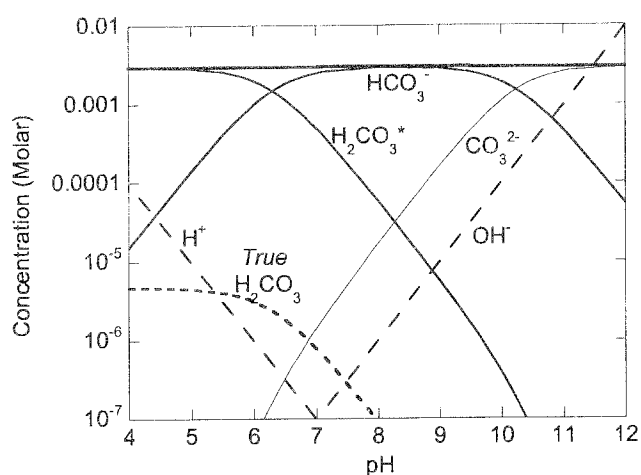


Figure 11.2 Distribution of inorganic carbon species in water at 3×10^{-3} M dissolved inorganic carbon (DIC = 36 g/m³). DIC is the sum of concentrations of aqueous carbon dioxide, $H_2CO_3^*$, HCO_3^- and CO_3^{2-} , and calculations were made using the equilibrium expressions in Table 11.2. (modified after Stumm and Morgan 1996).

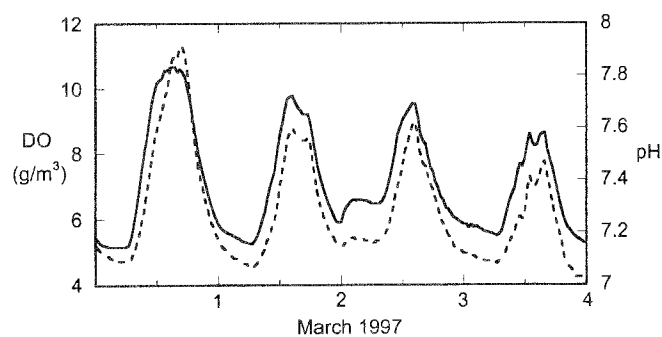


Figure 11.3 Covariation of dissolved oxygen DO (continuous line) and pH (dotted line) in a eutrophic lowland stream (Whakapipi, near Pukekohe) during summer.

dependent and characterise the rates at which chemical and physical transformations occur. The rate at which chemical reactions occur (chemical kinetics), molecular movement in response to concentration gradients (diffusion), oxygen exchange across the air-water interface (reaeration), and mixing and dispersion of contaminants within rivers (Rutherford 1994) are all examples of kinetic processes. Many processes in streams are characterised as being first-order with respect to one or other of the variables. For example, the uptake of ammoniacal-N (NH_4) in streams is modelled as a first-order process: $d\ln[\text{NH}_4]/dt = k_C[\text{NH}_4]$. Values of k_C for a wide range of rivers and streams are 1–5/d (Wilcock *et al.* 2002).

Dissolved and particulate substances

Natural waters contain mineral and biologically derived species that may be charged, such as Na^+ , or uncharged (e.g., SiO_2 , NH_3). For simplicity, water constituents are categorised as either dissolved or particulate (Table 11.1). In many instances, an operational definition of “dissolved” is used to describe material passing through a filter, such as a membrane filter, of a certain pore size (e.g., 45 μm), and thus may include colloidal particles that are not truly in solution (Fig. 11.4). For this reason, filter-passing constituents are sometimes better referred to as “filterable”. An example of this is dissolved reactive phosphorus (DRP), also referred to as soluble reactive phosphorus (SRP), but which is more accurately designated as filterable reactive phosphorus (FRP). The solubility of a substance limits the concentration that can be in solution, although supersaturation can occur, for example when a saturated solution of salt in water is slowly cooled so that the salt concentration exceeds its saturation value at the lower temperature. Supersaturated solutions are generally unstable and tend to revert readily to equilibrium conditions.

Particulate substances can settle out (precipitate) or be trapped on the surfaces of plants, whereas dissolved substances are more mobile and not so easily trapped. The form of chemical constituents in water affects their biological activity. Dissolved species are more readily taken up by fish, aquatic insects and other water-dwelling organisms (e.g., across gill membranes), whereas particulate chemicals are mostly incorporated via ingestion by benthic fauna (Rand *et al.* 1995). Dissolved chemicals in sediment pore waters are considered to be in equilibrium with their particulate forms (ANZECC 2000).

Plants have been thought to uptake nitrogen and phosphorus primarily from sediment via roots, but recent findings show leaf uptake to be important in nutrient-rich streams (Chambers *et al.* 1989; Madsen and Cedergreen 2002). Foliar uptake is the primary pathway for uptake of carbon, calcium, magnesium, sodium, potassium and sulphate by plants (Barko *et al.* 1991).

Carbon, nitrogen and phosphorus in aquatic ecosystems

Carbon (C), nitrogen (N) and phosphorus (P) are three essential macronutrients for stream life. Other macronutrients are sulphur, potassium, magnesium and calcium, but they are usually plentiful in streams (as major ionic species) (Odum 1971). Trace elements, or micronutrients, are elements that are necessary for living systems in minute quantities, often as components of enzymes. Eyster (1964) listed ten micronutrients that are essential for plants—Fe, Mn, Cu, Zn, B, Si, Mo, Cl, V and Co. Most of these are essential for animals too, and a few others, such as iodine, are essential for vertebrates.

The carbon cycle describes the circulation of carbon through ecosystems. Carbon dioxide is incorporated into organic compounds in green plants during photosynthesis. These compounds are eventually oxidised during respiration by plants, or by herbivores, carnivores, and saprophytes, releasing CO_2 back to the atmosphere (Allaby 1983). Photosynthesis and respiration by plants are the key processes transferring carbon between the atmosphere and biosphere. They can be characterised by the following chemical equations for algal photosynthesis that show the relative importance of C, N and P forms, according to whether N is in the form of nitrate (NO_3^-) or ammonium (NH_4^+) (Stumm and Morgan 1996). In each case the forward reaction (left to right) describes photosynthesis, while the reverse reaction (right to left) describes respiration, and the quantity in braces { } represents biomass (actually, the average composition of algae). It is important to note that for both equations the C:N and N:P *mole* ratios are 106:16 (or about 7:1) and 16:1. For

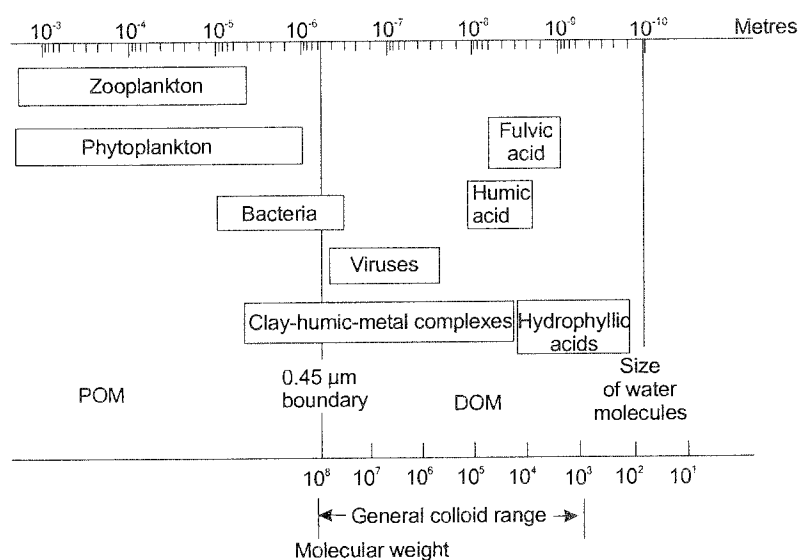
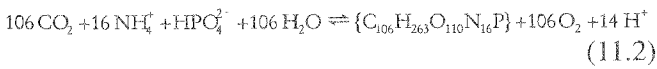
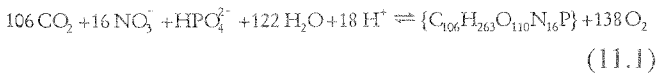


Figure 11.4 Size spectrum of particulate and dissolved matter in river waters (modified after Thurman 1985).

limnologists using *masses* instead of molar values, the C:N and N:P ratios are approximately 6:1 and 7:1, respectively.



Carbon is supplied to rivers from external and internal sources. External (allochthonous) inputs are mainly terrestrial litter, such as leaves, tree stems, branches and flowers that fall or are washed into streams. Internal (autochthonous) inputs are generated within stream channels, photosynthetically by algae, bryophytes, and large aquatic plants (macrophytes).

The nitrogen cycle (Fig. 11.5) describes the conversion of atmospheric nitrogen into organic compounds (N-fixation) by certain bacteria and blue-green algae (cyanobacteria), and the other (mainly biological) pathways by which N circulates through the ecosystem in organic and inorganic forms. Nitrogen enters streams mainly as dissolved inorganic N (represented by NH_4 for ammoniacal-N, and NO_3 for nitrate plus nitrite) and organic N. NH_4 removal is due to uptake by primary producers, bacteria, and fungi, plus direct nitrification. Indirect nitrification is the conversion of NH_4 mineralised from organic matter to NO_3 . NO_3 is removed from the water primarily via assimilation by biota and denitrification on the channel bottom. Regeneration is the release of NH_4 and NO_3 from the stream bottom back to the water column and is the net result of several interacting processes, including mineralisation, indirect nitrification, denitrification, and reuptake by organisms. NO_3 and NH_4

remaining in the water are exported downstream. Denitrification is the process occurring under anaerobic conditions in which nitrate nitrogen is returned in gaseous form to the atmosphere as N_2 and N_2O (Peterson *et al.* 2001).

Phosphorus is an essential constituent of DNA and RNA and is involved in energy transfer processes in living cells (ADP-ATP processes) and it is a component of fats of cell membranes (Allaby 1983). Phosphorus moves slowly from deposits on land and in sediments to living organisms, and even more slowly back into soil and sediment. The phosphorus cycle is given in various texts on limnology and environmental science and is not reproduced here.

Nutrient spiralling

Nutrient spiralling describes the downstream transport process whereby nutrients cycle between biotic and abiotic compartments of the stream ecosystem (Webster and Patten 1979; Newbold *et al.* 1981); it is a useful concept for considering eutrophication of running waters. While some upstream movement such as eddies, fish migration, and flight of adult insects occur, net fluxes in stream ecosystems are in the downstream direction (Stream Solute Workshop 1990). Newbold *et al.* (1981) defined spiralling length as the average distance an atom travels during one complete cycle from the dissolved state in the water column, to a streambed compartment, and eventually back into the water column. Thus, the more rapidly nutrients are taken up and released, the shorter the spiralling length. Spiralling length comprises two parts: uptake length is the distance travelled in dissolved form, and turnover length is the sum of distances travelled in particulate form (Stream Solute Workshop 1990). Uptake length, S_w , is related to

the first-order uptake rate coefficient, k_c (calculated from the slope of the plot of \log_e of the nutrient concentration versus time), stream velocity (u) and depth (h) and to the mass transfer coefficient (v_f) as follows:

$$S_w = \frac{u}{k_c} = \frac{uh}{v_f} \quad (11.3)$$

The mass transfer coefficient (v_f) is the velocity at which a nutrient molecule moves from the water column to the stream bottom due to biological or sorption processes (Peterson *et al.* 2001). Stream uptake lengths vary according to the nutrient concerned and its specific pathways for retention and release, climatic variables and seasonality, and features of individual streams (e.g., bed type, slope, light climate, plant community) (Table 11.3).

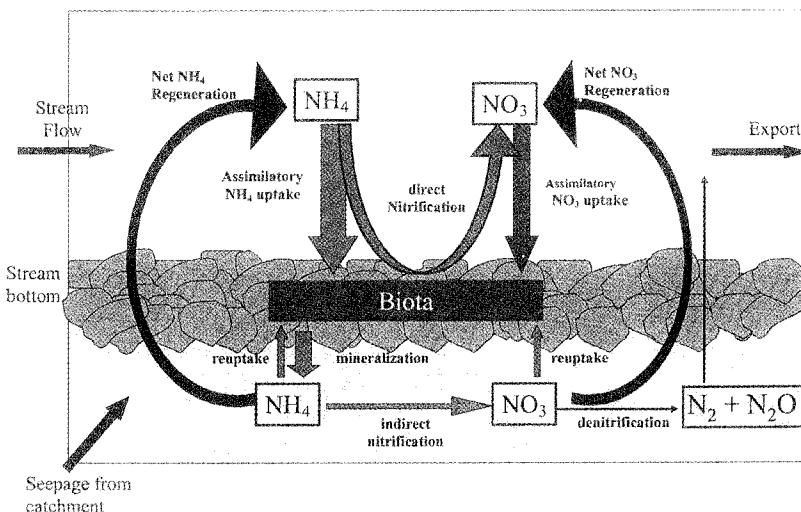


Figure 11.5 The nitrogen cycle in streams showing inputs and outputs of inorganic nitrogen forms (NH_4 and NO_3) and some key processes (from Peterson *et al.* 2001).

Table 11.3 Uptake (spiralling) lengths (S_w) for New Zealand streams, and some streams elsewhere.

Stream	S_w (m)			Source
	NH ₄ -N	NO ₃ -N	DRP	
Whangamata Stream, N.Z. grazed, not grazed		1100, 220		Quinn <i>et al.</i> 1993
Purukohukohu Stream, N.Z. open, shaded		21, 400		Quinn <i>et al.</i> 1993
Toenepi Stream, N.Z. open, shaded	410, 80	450, 70	70, 30	Wilcock <i>et al.</i> 2002
Bear Brook, U.S.A.	60-151		44-54	Hall <i>et al.</i> 2002
Paradise Brook, U.S.A.	105		29	Hall <i>et al.</i> 2002
Hubbard Brook U.S.A.	214-271		85	Hall <i>et al.</i> 2002
Ter River, Spain sandy-cobbled	85-402		96-127 48-65	Marti <i>et al.</i> 1994
rocky	31-104			

WATER QUALITY GUIDELINES

Water quality is more than merely the composition of water and may be defined, briefly, as its suitability for desired use—including use as a habitat for aquatic life (ANZECC 2000). Thus a river may be regarded as of low water quality if its composition limits its use, for example, for stock watering or swimming. A large variety of constituents may affect the water quality of rivers in New Zealand, but three particularly important categories of contaminant (Table 11.4) are: (1) *suspended sediment*, which may settle on the bed with associated damage and which, together with coloured dissolved organic matter, affects the optical character of waters; (2) the *nutrient* elements phosphorus and nitrogen, which may promote aquatic plant growth to nuisance levels (some forms of nitrogen are also toxic); and (3) *faecal microbiological contaminants* that represent a health risk (indicated by faecal indicator bacteria) to people using water for recreation and to livestock (Ministry for the Environment 1997). Also particularly important to overall water quality are the physical and chemical state variables: *water temperature*, *pH* and *dissolved oxygen*, which sometimes vary appreciably in running waters owing to diurnal fluctuation of insolation and the metabolism of aquatic plants (e.g., Wilcock and Nagels 2001).

In order to define the quality of a water it is necessary to compare its composition with water quality guidelines, which may be defined as numerical values below (or above) which a designated water use is supported and maintained (ANZECC 2000). Guidelines are not standards—they have no legal force. While this is sometimes viewed as disadvantageous, guidelines have an inherent flexibility, in that updating of guidelines as science advances is much simpler than re-legislating standards.

Table 11.4 lists guidelines in use in New Zealand for a number of attributes and contaminants of freshwaters. The

table does not include very large number of chemicals (particularly organics and toxic metals) that may, on occasion, contaminate water, with consequences to aquatic life. An extensive compilation of guidelines is given in ANZECC (2000: Table 3.4.1) for assessment of toxic risk from some such chemical contaminants.

Guidelines for major categories of contaminants

Sediment and optical effects

Several workers have suggested that fine-grained sediment transported in suspension (measured in mass concentration units: g/m^3) is *the* most important contaminant of waters, not merely in terms of mass load (g/s), but in terms of damage caused to aquatic habitat and human use (e.g., Clark *et al.* 1985). The effects of sediment while suspended differ from those when the sediment is deposited.

The most damaging consequence of excessive fine-grained suspended sediment is probably its effect on light transmission (optics) of waters (Davies-Colley and Smith 2001). Certainly the most visually obvious effect of suspended sediment is the cloudiness (turbidity) caused by intense light scattering. Direct damage to respiratory structures of aquatic animals occurs only at high suspended-sediment concentrations, much higher than the guideline for optical effects (Wood and Armitage 1997). Light attenuation by suspended sediment (and coloured dissolved organic matter) in river waters has two main optical effects—reduced light penetration and reduced visual clarity. These two aspects of clarity are distinct: doubling suspended-sediment concentration usually halves visual clarity, but causes much less than a halving of light penetration (Kirk 1994). Table 11.4 gives a guideline for light penetration, which is influenced by coloured dissolved organic matter (yellow substance) as well as suspended sediment in river waters (Davies-Colley *et al.* 1993).

Table 11.4 Water quality guidelines in use in New Zealand

Variable	Rationale (water use protected)	Guideline (notes)	References
Optical variables and suspended sediment			
Suspended sediment	Aesthetics and contact recreation	< 4 g/m ³	Smith <i>et al.</i> (1993)
Turbidity	Aesthetics and contact recreation	< 2 NTU	Smith <i>et al.</i> (1993)
Visual clarity (black disc visibility)	Aesthetics, aquatic life	< 20-50% change (depending on water class, site conditions)	MfE (1994)
Visual clarity (black disc visibility)	Contact recreation	> 1.6 m	MfE (1994)
Light penetration (euphotic depth)	Light for photosynthesis	< 10% change in euphotic depth, < 20% change in lighting at bed	MfE (1994)
Colour attributes (e.g., hue)	Aesthetics	< 5 or 10 Munsell units change (depending on water class)	MfE (1994)
Nutrients (and toxicity of some forms of nitrogen)			
Dissolved inorganic nitrogen (nitrate + ammoniacal-N)	Nuisance periphyton growth	< 0.04-0.10 g/m ³ (depends on accrual time)	* MfE (1992)
Dissolved reactive phosphorus	Nuisance periphyton growth	< 0.015-0.030 g/m ³	MfE (1992)
Nitrate-N	Potable water supply (toxicity)	< 11.3 g/m ³	Ministry of Health (2000)
Ammonia (NH ₃)-N	Toxicity to aquatic life	< 0.021 g/m ³	# ANZECC (2000)
5-day Biochemical oxygen demand, BOD ₅	Nuisance bacterial slime growth	< 5 g/m ³	MfE (1992)
Bacterial indicators of faecal pollution			
Faecal coliforms	Contact recreation	< 150/100 mL (bathing season median)	ANZECC (2000)
	Stock water supply	< 100/100 mL	ANZECC (2000)
	Shellfish gathering waters	<14/100 mL (median) <43/100 mL (90 percentile)	MfE (2003)
<i>Escherichia coli</i>	Contact recreation	< 126/100 mL (bathing season median)	Department of Health (1992)
		< 235-576/100 mL (single sample, depending on extent of contact)	Department of Health (1992)
	Beach grading (A-D) based on 95 percentiles	MfE (2003)	
	<i>Single sample interpretation:</i> Acceptable (Green) Mode <260 <i>E. coli</i> /100 mL Alert (Amber) Mode 260-550 <i>E. coli</i> /100 mL Action (Red) Mode > 550 <i>E. coli</i> /100 mL	MfE (2003)	
Enterococci	Contact recreation	< 33/100 mL (bathing season median)	Department of Health (1992)
		< 61-151/100 mL (single sample, depending on extent of contact)	Department of Health (1992)
		<35/100 mL (bathing season median)	ANZECC (2000)

* See also the more sophisticated "sliding scale" guidelines given by Biggs (2000b) based on the work of Biggs (2000a)

Using the ANZECC (2000) trigger values protocol based on the work of Hickey and Vickers (1994).

Visual clarity strongly affects contact recreation safety and the visual amenity of rivers (Davies-Colley *et al.* 1993). For primary contact recreation, black disc visibility should exceed 1.6 m, and for protection of visual amenity and sighted aquatic life a maximum (natural) visibility change of 20–50% is recommended (Ministry for the Environment 1994). These guidelines (Table 11.4) have been endorsed by the Australian and New Zealand Environment and Conservation Council (ANZECC 2000). Visual clarity also affects the habitat of sighted aquatic life, notably fish and aquatic birds. To protect fish such as trout from the effects of suspended sediment, Newcombe and MacDonald (1991) proposed a “severity-of-effect” model combining suspended-sediment concentration and exposure time to this pollutant. More recently, Newcombe (2003) developed a visual clarity “severity-of-effect” model, which predicts, for example, a threshold of effect at 1 or 2 days exposure to a black disc visibility of 0.55 m, and 1–2 weeks for a visibility of 1.09 m, and “significant impairment” of sensitive fish exposed > 2 weeks to 0.55 m visibility.

Several people, including Ryan (1991) in New Zealand, have reviewed the range of biological effects of deposited fine sediment. Sedimentation of fines is expected in standing bodies of water such as lakes, but can also occur within flowing streams and rivers, despite their turbulence (Wood and Armitage 1997). For example, Graham (1990) showed that fine sediment is captured within periphyton growing on the beds of streams. Fine particles can also settle in quiet areas such as pools at relatively low flows, and within the interstices of the bed sediment. Deposited fine material can partially seal surface bed sediment (a process known as “colmation”), thereby isolating the interstitial water from the overlying river water and degrading the quality of interstitial habitat for invertebrate animals. Fines deposition also particularly degrades fish spawning gravels. Currently, it is not possible to enumerate simple guidelines to protect rivers from sedimentation.

Table 11.4 also gives guidelines for colour attributes of water, which are particularly affected in rivers by light-absorbing particulate organic material (organic detritus) and coloured dissolved organic matter.

Nutrients

Nutrient promotion of biological growths in waters is a natural process, but proliferation of growths to nuisance levels (“undesirable biological growths” in the words of the Resource Management Act 1991) may occur when nutrient concentrations are high. Dissolved inorganic forms of phosphorus and nitrogen, which are immediately available to aquatic plants, are of most concern in rivers. Table 11.4 lists guidelines for dissolved inorganic nitrogen (DIN—the sum of concentrations of nitrate-N, nitrite-N and ammoniacal-N), and also for dissolved reactive phos-

phorus (DRP), because these immediately bioavailable forms of N and P promote nuisance benthic algae (periphyton) growths in unshaded running waters. Many factors besides nitrogen and phosphorus availability affect the growth of benthic algae, including light, substrate stability, grazing by invertebrate animals, and flood frequency (Biggs 2000a). A “sliding scale” of nutrient guidelines that accounts for different periods of undisturbed periphyton growth in rivers has been proposed by Biggs (2000b).

The growth of larger aquatic plants (macrophytes) in rivers is also promoted by nutrients, and can have undesirable consequences for habitats. Dense stands of macrophytes cause oxygen and pH fluctuation (Wilcock and Nagels 2001), induce sedimentation (Sand-Jensen 1998), impede water currents, and may encroach on the channel-forming marginal wetlands (Champion and Tanner 2000). However, macrophytes can take up nutrients from sediment through their roots, as well as from water, so control of nutrient concentrations in the water appears less relevant to macrophytes than to periphyton. Nuisance macrophytes are more readily controlled by light limitation than nutrient limitation (e.g., Sand-Jensen 1998).

Although periphyton and other aquatic plants are net producers of oxygen on average, their respiration at night often causes levels of dissolved oxygen to drop too low for aquatic animals. The daily fluctuation of dissolved oxygen in eutrophic streams is accompanied by a daily fluctuation of pH caused by inorganic carbon uptake by aquatic plants during photosynthesis. pH typically reaches a peak in the mid-afternoon, which promotes toxicity of ammoniacal-N ($\text{NH}_4^+ + \text{NH}_3$). The un-ionised ammonia (NH_3) form is very toxic, with a guideline value (Table 11.4) of 0.021 g/m^3 (ANZECC 2000), based on the work of Hickey and Vickers (1994), who demonstrated high sensitivity of some New Zealand stream invertebrates to NH_3 . The ratio of toxic free ammonia to ammonium ion depends on the pH, with approximately equal ammonia and ammonium concentrations at a pH of 9.2. Thus, high ammoniacal-N in receiving streams may not be toxic if the pH < 8, at which value the concentration of NH_3 is less than 1/10 of ammoniacal-N. However, some eutrophic, plant-infested agricultural streams in New Zealand have ammoniacal-N concentrations of $1\text{--}2 \text{ g/m}^3$, and pHs > 8.5 are common in mid-afternoon (free ammonia > 0.4 g/m^3) when incipient toxic effects can be expected.

Nitrate in drinking water is toxic to humans at high concentrations, and the guideline given in Table 11.4 is seldom exceeded in rivers in this country (Smith and Masdaam 1994).

Unightly growths of heterotrophic bacterial slimes were once fairly common in New Zealand rivers receiving

organic waste discharges (Cooper 1983). Ministry for the Environment (1992) recommended a five-day biochemical oxygen demand (BOD₅) of filtered samples < 2 g/m³ in order to prevent such growths. It is a testimony to the extensive clean-up of point-source organic pollution of rivers in New Zealand over the last two decades that this guideline is now mainly of historical interest.

Faecal contamination

In order to protect the public from the risk of contracting a communicable disease from waters contaminated by faeces of humans or warm-blooded animals, guidelines have been promulgated for faecal indicator bacteria such as faecal coliforms or their main constituent species, *Escherichia coli* (Table 11.4). Faecal indicator bacteria indicate the risk of infection from ingesting water, particularly during contact recreation (Ministry for the Environment 2003). Faecal contamination by livestock may represent a different disease risk than faecal contamination by human wastes, but currently guidelines do not make any distinction. This is in part just practicality—there is no simple way of telling whether *E. coli* (the most commonly used freshwater bacterial indicator) in a water sample has come from animal or human faeces, but it also recognises that numerous disease agents (notably *Campylobacter*, *Cryptosporidium*) may be present in animal faeces.

The most recent microbiological quality guidelines (Ministry for the Environment 2003) have moved beyond the use solely of guideline values for faecal indicator bacteria towards the grading of swimming sites based on 95 percentile values for historical *E. coli* data (grades ranging from A: *E. coli* 95 percentile < 130/100 mL, to D: 95 percentile > 550/100 mL), supported by “sanitary inspection” of faecal contamination sources. Single samples of water taken during bathing season monitoring are interpreted using the guidelines in Table 11.4: *E. coli* concentrations < 260/100 mL (Green mode) indicate acceptable water for contact recreation, whereas a single sample > 550/100 mL is ‘Action’ (Red) mode (whereupon efforts are to be made to warn the public, for example by erecting warning signs). These guidelines are based on the estimate that about 5% of campylobacteriosis cases in New Zealand (currently about 300 cases/100,000 people per year) may be from water-borne exposure.

A guideline given in Table 11.4 for livestock drinking water in terms of faecal coliforms (ANZECC 2000) is 10 times lower than a previous guideline for this use, and would be difficult to meet in streams directly polluted by domestic animals. Ministry for the Environment (2003) maintain the Department of Health (1992) guidelines for shellfish-gathering waters—based on assay of faecal coliforms (Table 11.4).

Trigger values

An approach to guidelines promulgated by ANZECC (2000) is to define “trigger values” below which there is a low risk of any adverse biological effects. Ideally trigger values would be developed from actual studies of ecological effects, however, in the absence of such information, the 80 (or 20) percentile for “slightly to moderately disturbed” ecosystems is suggested as a guide. Table 11.5 lists 80 percentiles (or 20 percentiles) for lowland rivers in the National River Water Quality Network (NRWQN) (ANZECC 2000). It is immediately evident that these do not necessarily indicate anything useful about the limits of tolerance of aquatic ecosystems. In particular, the trigger values for ammonical-N, dissolved oxygen, and pH are not likely to be helpful, being well within guideline values for adverse biological effects. However, the percentiles in Table 11.5 may provide useful points of reference for some purposes.

Table 11.5 “Trigger” values for lowland rivers in New Zealand (ANZECC 2000)

Variable	Value
Total nitrogen	614 mg/m ³
Nitrate	444 mg/m ³
Ammonical-N (<i>not</i> NH ₃)	21 mg/m ³
Total phosphorus	33 mg/m ³
Dissolved reactive phosphorus	10 mg/m ³
Dissolved oxygen	98%, 105%
pH	7.2, 7.8
Visual clarity	0.56 m
Turbidity	5.58 NTU
Temperature (February)*	21.5 °C

* Based on 80 percentile values for NRWQN rivers (Smith and Masdaam 1994).

Water quality indexing

Simple guideline numbers have a major deficiency in that they give no information about changing suitability of water for intended use as water quality variables approach guidelines from above or below. The “sliding scale” (multiple guideline) approaches of Biggs *et al.* (2000b) for nutrient limitation of periphyton and the Ministry for the Environment (2003) microbiological guidelines appear to be, in part, a response to this deficiency. When a guideline is established, the tendency is to think that suitability for use changes steeply (changing abruptly from suitable to unsuitable) as the guideline is passed (Fig. 11.6). However, in reality, suitability-for-use curves often slope quite gently.

Ideally, continuous curves giving suitability-for-use of water as a function of particular water quality variables would be available to managers. Such curves are the basis

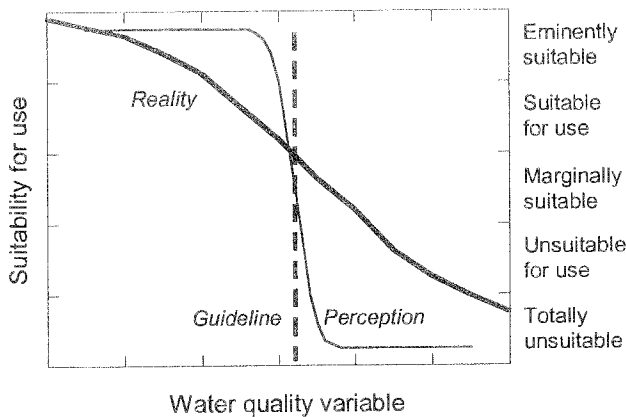


Figure 11.6 Schematic of a water quality guideline value compared with suitability-for-use curves. A perception may be that suitability-for-use changes abruptly near the guideline value (dashed vertical line) as shown by the thin curve. However, more typically, the suitability-for-use changes slowly, with no clear threshold (thick curve).

of water quality indexing, which is an attempt to reduce a complex water quality dataset to a single number denoting overall quality or suitability for use (Ott 1978). The suitability-for-use scores for individual water quality variables are aggregated to provide an overall index value for the intended use (usually by summing individual scores with or without weighting, although more complex approaches have been tried). In New Zealand, Smith (1990) developed a water quality index with two important advantages over other international efforts: (1) the suitability-for-use curves were developed by a panel of experts drawn from a cross-section of the water industry via an (anonymous) Delphi dispute resolution procedure, so reducing sensitivity to “outlier” opinions of one or a few experts, and (2) the minimum suitability-for-use score is taken as the index score (rather than aggregating the suitability-for-use scores of individual variables). Figure 11.7 shows suitability-for-use curves for visual clarity in a contact recreation index for New Zealand, based on Smith’s earlier work (Nagels *et al.* 2001).

SOURCES OF POLLUTION

Aquatic pollution is conventionally defined as emanating from two kinds of sources. Point sources are clearly identifiable, have specific locations, and are typically pipes and drains discharging wastes from industry and municipal waste disposal networks. Non-point (or diffuse) pollution has been defined as: “Pollution arising from land-use activities (urban and rural) that are dispersed across a catchment or sub-catchment...” (Novotny 2003). Non-point source pollution occurs when water flows over or

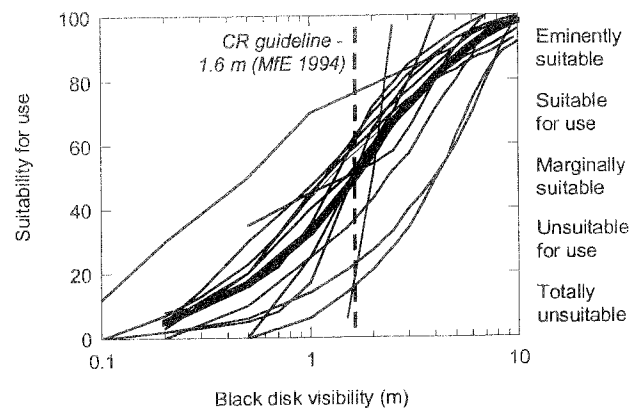


Figure 11.7 Suitability-for-use curves for visual clarity in a water quality index for contact recreation developed by Nagels *et al.* (2001). The opinions of 16 panellists are shown as thin curves and the thick curve is the mean (“consensus”) curve used for indexing. The Ministry for the Environment (1994) contact recreation guideline is shown as a vertical dashed line.

through the land, picks up pollutants, and deposits them in surface or ground waters. Pollutants often tend to enter waterways predominantly from either a diffuse or a point source (Table 11.6), due to the nature of their sources and the main pathways for transport to waterways.

Pollution loads to streams differ according to the dominant land use, for example, dairying versus vegetable growing. Specific yields (sometimes called catchment export coefficients) are the annual load per unit area of land and have the units, kg/ha/year. To characterise each land use properly, yields should be calculated from data sets averaged over long periods, so that they are not unduly biased by unusually wet or dry years that affect runoff. Examples of specific yields for suspended solids, total nitrogen (TN) and total phosphorus (TP) are given in Table 11.7.

Nationally, point source yields of nutrients to lakes and rivers are considerably lower than yields from non-point sources. The total phosphorus load from non-point sources is roughly 50 tonnes/day, which is considerably greater than the 6.1 tonnes/day from point sources entering inland waterways. The total nitrogen load from non-point sources is about 400 tonnes/day, compared with the point-source estimate of 29 tonnes/day (Elliott and Sorrell 2002).

GENERAL FEATURES OF RIVER WATER QUALITY IN NEW ZEALAND

Overview

By comparison with World Average Fresh Water (WAFW, Meybeck 1987), many New Zealand rivers are dilute, soft, and low in alkalinity, and may be characterised

Table 11.6 Point source and non-point sources of pollution, and key contaminants.

Pollution source	Pollutant type	Contaminant
<i>Point source</i>		
Surface and subsurface drains	Farm wastes, irrigation water, dairy pond effluent	N, P, SS, faecal microbes
Sewer	Community sewage effluent	BOD, SS, faecal microbes, N & P
Industrial drain	Processing wastes	Heavy metals, BOD, toxic organics
Storm water outfall	Urban storm water	Oil and grease, faecal microbes, N, P, BOD
<i>Non point (diffuse) source</i>		
Surface runoff from agriculture	Particulate pollutants*	TP, TN, SS, faecal microbes
Subsurface runoff	Dissolved pollutants**	DIN, DRP
Riparian grazing by livestock (including livestock in channels)	Animal wastes, sediment, reduced streambank stability	Faecal microbes, SS, N, P
Atmospheric deposition	Urban, industrial emissions	Acid rain, polycyclic aromatic hydrocarbons
Spray drift	Farm operations	Pesticides, fertilizer

* Surface drains often collect drainage from subsurface drains and hence collect dissolved and particulate pollutants.

** Subsurface drains can convey particulates if there are soil macropores (e.g., soil cracks).

Table 11.7 Specific yields (kg/ha/year) for different land uses in New Zealand.

Land use	SS	TN	TP	Reference
Intensive dairy	142	35	1.16	Wilcock <i>et al.</i> 1999
Average grazed pasture	600-2000	4-14	0.3-1.7	Wilcock 1986
Urban development	200-2000	2.5-11	0.4-1.6	Williamson 1993
Exotic forest – disturbed	300-2000	0.06-0.8	0.4-8	Wilcock 1986
Exotic forest – undisturbed	500	0.07-0.2	0.15	Wilcock 1986
Native forest	27-300	2-7	0.04-0.68	Wilcock 1986

as “sodium-calcium bicarbonate waters”, whereas WAFW is “calcium bicarbonate water”. Close and Davies-Colley (1990a) reported that the composition of 97 New Zealand rivers sampled at baseflow in the “100 Rivers” project ranged widely, particularly in nutrient content, although the quality was generally good by comparison with guidelines (Table 11.4). Visual clarity of rivers at baseflow varied by 40-fold (from 0.25 to 11 m, median 2.04 m), and g_{440} by more than 100-fold (from about 0.02–3/m) and accounted for one quarter of the variation in colour (hue), which ranged from “orange” through to “blue” (Davies-Colley and Close 1990).

Smith and Masdaam (1994) studied the first two years of data collected in the National River Water Quality Network (77 river sites). They confirmed Close and Davies-Colley’s (1990a) finding that New Zealand rivers are generally of good quality in terms of the guidelines (Table 11.4). The most important exceptions were some rivers with comparatively high nutrient concentrations associated with intensive agriculture, and some turbid rivers draining rapidly eroding mudstones. Smith *et al.* (1996) reported that temperature, dissolved oxygen, pH,

conductivity and visual clarity tended to decrease with flow in New Zealand rivers, whereas other variables, including turbidity, yellow substance and nutrients, tended to increase with flow. Smith *et al.* (1997) confirmed that an inverse relationship between visual clarity and flow was typical of New Zealand rivers, and that yellow substance (indicated by g_{440}) typically increased with flow. These flow trends explain the familiar tendency for New Zealand rivers to be visually clear and green to blue-green coloured at baseflow, and turbid and yellow-coloured (“muddy”) in flood flow.

Scarsbrook *et al.* (2003) reported that patterns in 13 years of water quality data in the National River Water Quality Network were related to climatic fluctuations associated with the El Niño-Southern Oscillation (ENSO) Index. The influence of climate on water quality was apparently mediated via insolation and temperature, as well as by flow (reflecting rainfall). Clearly, detection of water quality trends in response to human influences such as an intensification of land use is complicated by the need to account for climatic influences.

Obtaining a national picture of the state of faecal

pollution of New Zealand rivers is difficult because indicator bacteria are not measured in the National River Water Quality Network. Another difficulty is that indicator bacteria concentrations vary widely at an individual site, partly with the state of flow, with concentrations in flood flow typically being more than 100-fold greater than at base flow (Nagels *et al.* 2002). However, Smith *et al.* (1993) inferred considerable faecal pollution of streams and rivers draining grazed farmland in New Zealand, based on patterns of indicator bacteria in various regional monitoring programmes (e.g., in the Wellington Region, rivers draining farmland had median values that were 20 times higher than forested rivers). Thus, a general trend to increasing faecal pollution with increasing intensity of farming may be expected. However, high faecal bacterial numbers may sometimes be encountered in streams draining forested land (e.g., Davies-Colley and Nagels 2002), and have been attributed to feral mammals or stray livestock accessing the stream channel, together with reduced sunlight-induced die-off of indicator bacteria under tree shade.

Current issues

Until perhaps the early 1990s, most attention in New Zealand was given to the effects of point sources of pollution on water quality, although several authors (notably McColl and Hughes 1982; Wilcock 1986) had earlier drawn attention to the effects of diffuse sources on water quality. There are still important problems related to point sources such as dairy farm wastewater and sewage from small towns (both still often treated by traditional two-pond waste stabilization systems using designs developed in the 1970s). However, the main source of degraded water quality in New Zealand is increasingly shifting to diffuse sources related to land use. The MAF Policy Report on effects of agriculture on freshwaters in New Zealand (Smith *et al.* 1993) highlighted the importance of diffuse pollution by livestock agriculture. Pollution by point sources is now well on the way to being cleaned up in this country and the more difficult task of controlling diffuse pollution (Ministry for the Environment 1997) from arable farming and urban stormwater, as well as farmland used for livestock grazing, must now be addressed this millennium.

A case study illustrating the change in focus from point to diffuse sources is provided by the Waikato River, which is now in much better condition than it was in the 1950s, when it was grossly polluted with organic wastes from industry and sewage from towns (Vant 2001). However, the river has recently suffered cyanobacterial blooms due to eutrophication (causing taste and odour problems for water supply), and faecal indicator bacteria concentrations indicate that the river is still not suitable for swimming in reaches near Hamilton and downstream. Vant (2001)

warned that the Waikato's quality may worsen with increasing intensification of farming unless measures are instituted to control nutrients and faecal pollution. The "clean streams" accord between the dairy company Fonterra, regional councils and central Government recognises the problem and sets a timetable for action (e.g., 50% of stream crossings by dairy herds are to stop by 2007 and 90% by 2012).

Intensification of farming elsewhere in New Zealand has also led to water quality degradation. For example, Hamill and McBride (2003) recently reported a trend of increasing dissolved reactive phosphorus in rivers in Southland, which they attributed to the increasing dairy herds in that region.

REGIONAL WATER QUALITY PATTERNS

Close and Davies-Colley (1990b) reported clustering of river sites into eleven groups, based on major ions, phosphorus, organics and nitrogen, calcium, and nitrate. Their cluster map shows considerably more complexity in the North Island than in the South Island, and indicates that water quality is generally better in the South Island. They found that regional patterns in river water quality reflect the influences of environmental factors such as specific water yield, and the proportion of catchment area in soft rock and intensively grazed pasture.

A similar clustering analysis was carried out by Masdaam and Smith (1994), who found that water quality varied with mean water temperature, catchment elevation and soil properties, and degree of pasture development. Water quality was more variable in the North Island, consistent with the greater overall environmental variability of this island (also seen in soils, geological, and climate parameter maps, for example). Sites with comparatively low water quality included the Waipaoa River (Gisborne District), characterised by very low visual clarity and very high conductivity—both features reflecting rapidly eroding soft rocks in the catchment, and the Tarawera River (Bay of Plenty) and lower Manawatu River, subject to organic pollution with high nutrients, low visual clarity and high conductivity. Three lowland rivers in Northland (Hoteo, Rangitopuni and Wairua) had low visual clarity, high coloured dissolved organic matter, high nutrients, and fairly high conductivity, and a number of rivers had some degree of industrial pollution or intensive farming pressure, resulting in elevated nutrients, fairly high coloured dissolved organic matter and conductivity, and low-moderate visual clarity (Waipa, lower Waihou and lower Waikato River, in the Waikato Region; Waingongoro River, Taranaki; and Mataura River, Southland—the last being the only South Island site in the National River Water Quality Network with appreciably degraded water quality at that time).

Broad patterns identified by Masdaam and Smith (1994) and Close and Davies-Colley (1990b) can be summarised as follows.

- Few New Zealand rivers suffer major industrial pollution to the extent that water quality is appreciably degraded. The lower Tarawera River, which receives pulp mill effluent from the Kawerau paper mills, is a notable exception, but to a lesser extent the Waikato, lower Manawatu and Maitai rivers all suffer some industrial pollution.
- Lowland rivers in the northern North Island tend to cluster together, being strongly coloured by coloured dissolved organic matter and rather turbid, reflecting strongly leached and weathered clay soils.
- Rivers draining erodible mudstone catchments (notably the Waiapu and Waipaoa in the Gisborne District and the Turakina in Taranaki) are very turbid and have high conductivity—both features related to high sediment yields from erodible mudstones.
- Rivers draining intensively farmed land tend to have comparatively high nutrient concentrations, particularly nitrate. These rivers would also be expected to have moderate to high levels of faecal contamination.
- Upland rivers in pristine or lightly grazed land have generally good water quality.

Neither river site-clustering study captured all the water quality patterns in New Zealand rivers, particularly those of small streams, which may be locally polluted by industrial discharges, sewage discharges, urban runoff, or intensive livestock or arable farming. Furthermore, because bacterial indicators are not currently included in the National River Water Quality Network, faecal pollution by animal and human wastes could not be assessed.

Rivers on the West Coast, South Island provide an interesting regional pattern. Lowland rivers and streams in the region tend to have high coloured dissolved organic matter concentrations (Moore 1989; Moore and Jackson 1989), as indicated by yellow substance absorption coefficients (e.g., Davies-Colley *et al.* 1992; Davies-Colley and Nagels 2002), with associated dark yellow to orange-coloured waters. The high yellow substance may be attributed to the strongly leached soils in the region, which have low levels of iron and aluminium hydrous oxides that would otherwise complex these organic acids. In sharp contrast, large rivers with alpine headwaters in the same region have low yellow substance (e.g., 0.15/m in the Wanganui River; Davies-Colley and Nagels 2002) and tend to be blue-green in hue and somewhat turbid, with high concentrations of glacial flour.

FUTURE TRENDS

Recent changes in agriculture have seen a decline in the national sheep flock and a dramatic increase in cattle numbers (see Chapter 34), with an overall intensification

of land use. Numbers of other livestock, notably deer, are also rising rapidly. Increased stocking rates and fertilizer use are likely to increase diffuse pollution of our freshwater resources, unless there is widespread adoption of sound management practices, such as the fencing of riparian areas. However, such practices are less effective at controlling dissolved pollutants, most notably nitrate, which is likely to be of increasing concern in areas like the Lake Taupo catchment (see Chapter 34).

The effects of climate change may lead to more extreme weather patterns, with increasingly drier eastern areas (Hawkes Bay, Wairarapa, Marlborough, Canterbury, Otago) and more rainfall in western areas. The effects of global warming will be to increase the potential productivity of some areas and lead to a further intensification of land use. Increased demand for irrigation water to sustain increasingly intensive land use, despite projected drier conditions in parts of New Zealand, will place increasing stress on freshwater resources, and widespread adoption of sustainable land and water management practices will be needed to maintain river water quality.

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Chapter 10

Floods and low flows

Charles Pearson and Roddy Henderson

INTRODUCTION

This chapter focuses on methods for analysing extremes of river flows, that is, low flows and flood peaks. These extremes have serious consequences for the New Zealand economy, from shortages of water (including ground water) for irrigation, hydropower generation, and survival of instream biota, to the devastation and destruction caused by river floods. Low flows often occur over broad areas as part of a drought, through a lack of rainfall over a sustained period. Flood flows tend to occur at a smaller spatial scale, in response to one or two periods of intense storm rainfall over a number of catchments. A meteorological distinction between the two flow extremes is that low flows are related to climate—the prevailing climate over a matter of months, whereas floods are related to weather—one storm of up to 5 days duration, or a sequence of storms over a matter of weeks.

The principal questions asked about floods and low flows are “How severe, frequent, and prolonged are they?” and “When will the next one happen?” The first question is asked in the design of structures such as stopbanks or of an irrigation scheme, or in the preparation of water management plans. The second is asked when preparations must be made to mitigate the immediate effects of a flood or drought, for example by evacuating stock from flood plains or scheduling thermal electricity generation to conserve hydroelectricity generating capacity.

This chapter focuses on the processes, characteristics, frequency and occurrence of New Zealand low and flood flows. A section on frequency analysis, applicable to both low and flood flows, covers statistical terms, data checking procedures, key assumptions (such as time series “stationarity”), and “at-site” and “regional” methods. In separate sections for floods and low flows, the nature of the frequencies of their occurrence are covered, including the effects of climate shifts and changes. For floods,

information on extreme storm rainfalls is provided, and for low flows, their context in terms of drought severity is explained.

Costs of floods and low flows

Insurance industry payments for flood damages caused by 42 events between 1976 and 2003 averaged \$17 million per year (Source: Insurance Council of New Zealand Inc.). (All costs in this chapter have been inflation adjusted to March 2000.) One of the most costly events for insurers was the Southland and Invercargill floods of 1984 (\$100 million). In February 2004, floods in the Manawatu, Wanganui and Taranaki region have been estimated to cost over \$100 million in insurance payouts, and cost a great deal more to the communities devastated. In the decade 1976-1985, seventeen lives were lost through floods and total costs were over \$1,300 million. In 1988, a single storm, Cyclone Bola, alone caused an estimated \$130 million of damage, particularly in the East Cape region (\$52 million in insurance payments). In 1999 and 2000 floods had a devastating impact on several regions of New Zealand: after the April 2000 floods in Tauranga, insurers paid out \$1.9 million; in Queenstown in December 1999 they paid nearly \$50 million; in South Canterbury July 1999, \$600,000; in Whangarei and Rotorua, May 1999, \$2.1 million; in Dargaville, April 1999, \$1.72 million, and in Northland and Pukekohe, January 1999, over \$5 million.

At the opposite extreme, droughts can be equally expensive—the Canterbury drought of 1987-88 cost the community an estimated \$500 million. Similarly, the cost of the 1997-98 Otago El Niño drought on farm gate returns was \$500 million. That is about half a percent of gross domestic product, although the total effect on the economy can be multiplied by anything between three and twelve. Climate is now recognised as an important

contributor to the New Zealand economy, and in particular to the recession of 1998 (Buckle *et al.* 2002). High-profile hydro-electricity crises occurred during droughts in 1992, 2001 and 2003, when storages in key hydro-lakes (principally Pukaki, Tekapo and Taupo—70% of New Zealand's hydro-electric storage) failed to meet demand. For the latter two events, the crises were more related to an increase in demand for electricity than to extreme shortages in the lake inflows. Events as severe as that of 1992 occurred in the 1930s and 1970s.

The effects of low flows and floods are not limited to economic costs; they also have major impacts on the well-being of New Zealand society and on the environment. Everyday fluctuations in flow are important to recreation, river and lake ecosystems and water quality, and to the estuaries, and coastal and oceanic waters that the rivers flow into. Extreme flows have major effects on the biota of rivers, e.g., Jowett (2001) showed that numbers of fish are reduced during low flows in the Waipara River, while Hayes (1995) showed the “cost” of floods for populations of trout in the Kakanui River.

Flood and low flow data

An important basis for understanding extreme flows is reliable records of flow. Routine monitoring of New Zealand rivers began between 1900 and 1930 for hydro-electric power generation. Major regional rivers came under monitoring programmes in the 1930s for designing flood protection works. During the International Hydrological Decade (1965-74) representative basins were selected throughout New Zealand and comprehensive national and regional monitoring networks were established.

The National Hydrometric Network (Pearson 1998) is a key source of New Zealand stream flow data, along with the complementary monitoring networks operated by regional and district councils. Data on variables such as air temperature, barometric pressure, wind direction, rainfall, lake and river water levels, river flows (c. 300 stations) and sediment loads, and river water quality are routinely collected and stored, and disseminated by the National Hydrometric Network, a part of the “Climate and Water Resources Archive”.

To ensure national consistency and provide assurance that the data can be confidently used for scientific research and planning, stringent quality control procedures are applied. The quality assurance of the National Hydrometric Network leads international procedures (Mosley and McKerchar 1989; Hudson *et al.* 1999) in being accredited to International Standards Organisation levels. Overall, New Zealand has had 2,100 water level recording stations opened, representing over 33,000 station years of water level and flow data (Walter 2000;

data available from NIWA and regional and district councils). Around 1,000 stations are currently operating.

Also available (from NIWA) for analyses of low flows is a database of summer “spot” low flow measurements, which has data for over 5,000 New Zealand river locations, with over 36,000 individual measurements (excluding data from the water-level recording stations). For major rivers, which are usually near human settlements, local councils have records of historical flood water levels, which can be used to place more recent floods into a long-term perspective. Other data of importance in understanding extremes of flows are provided by national databases and information on land resources, land-use, geology, and topography (digital elevation data).

New Zealand floods and low flows

Variations in weather and climate are reflected in fluctuating river flows. The peaks in a natural river's flow (Fig. 10.1) are responses to rainfall and snowmelt, and less frequently to dam breaks (e.g., washing out of a natural dam caused by a landslide). Flows rise sharply in response to rainfall, and decrease more slowly as the soils and rocks in the catchment gradually release the water that was stored during the storm. The flows of many rivers in New Zealand (e.g., Waikato, Clutha) are modified by large lakes and hydro-power dams, which reduce the size of peak flows and maintain low flows during rainless periods.

For practical purposes, a flood is defined as a sharp rise in a river's flow, above its mean flow. The magnitude of a flood depends principally on the amount and duration of rainfall, and on how wet the catchment was beforehand. The extent to which flows decline depends on the period of time to the next rainfall, and on the amount of water stored in the catchment in the form of groundwater, soil moisture, snow, and water in lakes.

Low flows are normally considered to be an extended “recession” of a river's flow—a decay over time of the flow hydrograph to well below its mean flow. The magnitude of low flows depends upon the severity and duration of rainfall deficit across a catchment and the speed of release of water from aquifers and soil moisture.

Broad patterns of average annual maximum flood peaks and average annual minimum low flows across New Zealand are shown in Figure 10.2. High flood flow rates occur in catchments draining mountainous regions, particularly the Southern Alps, with lower flood flows on average in eastern regions sheltered by these mountains. Low flows exhibit a similar pattern, with the notable exception of the higher low flow rates in the pumice regions of the central North Island, which release water slowly. New Zealand has a maritime climate, and lies within a zone with predominantly moist westerly winds, and floods can occur at any time of year. Storms come

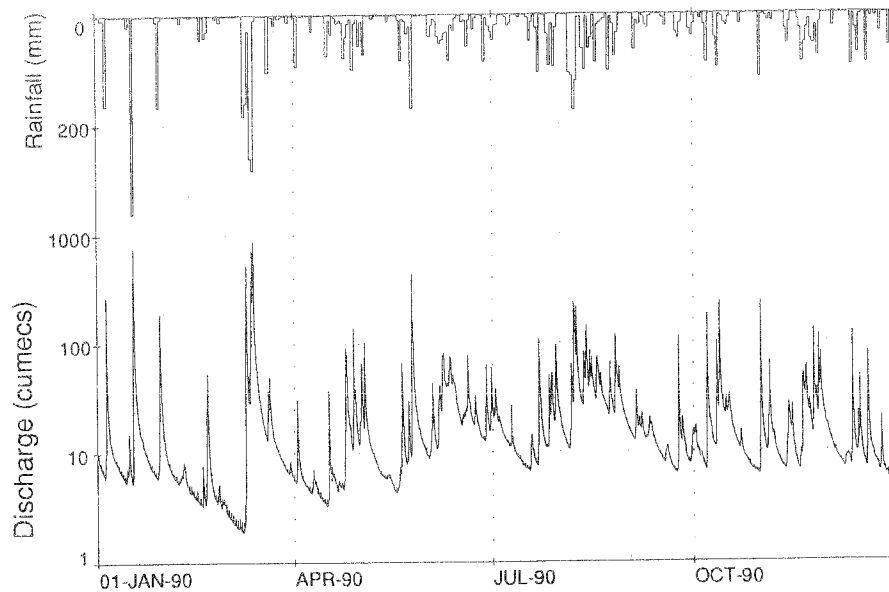


Figure 10.1 The 1990 flow hydrograph for the Hutt River at Birchville (site number 29818, catchment area 427 km², Walter 2000). Daily rainfall totals from Phillips automatic raingauge (located inside Hutt Catchment) are shown for the same period.

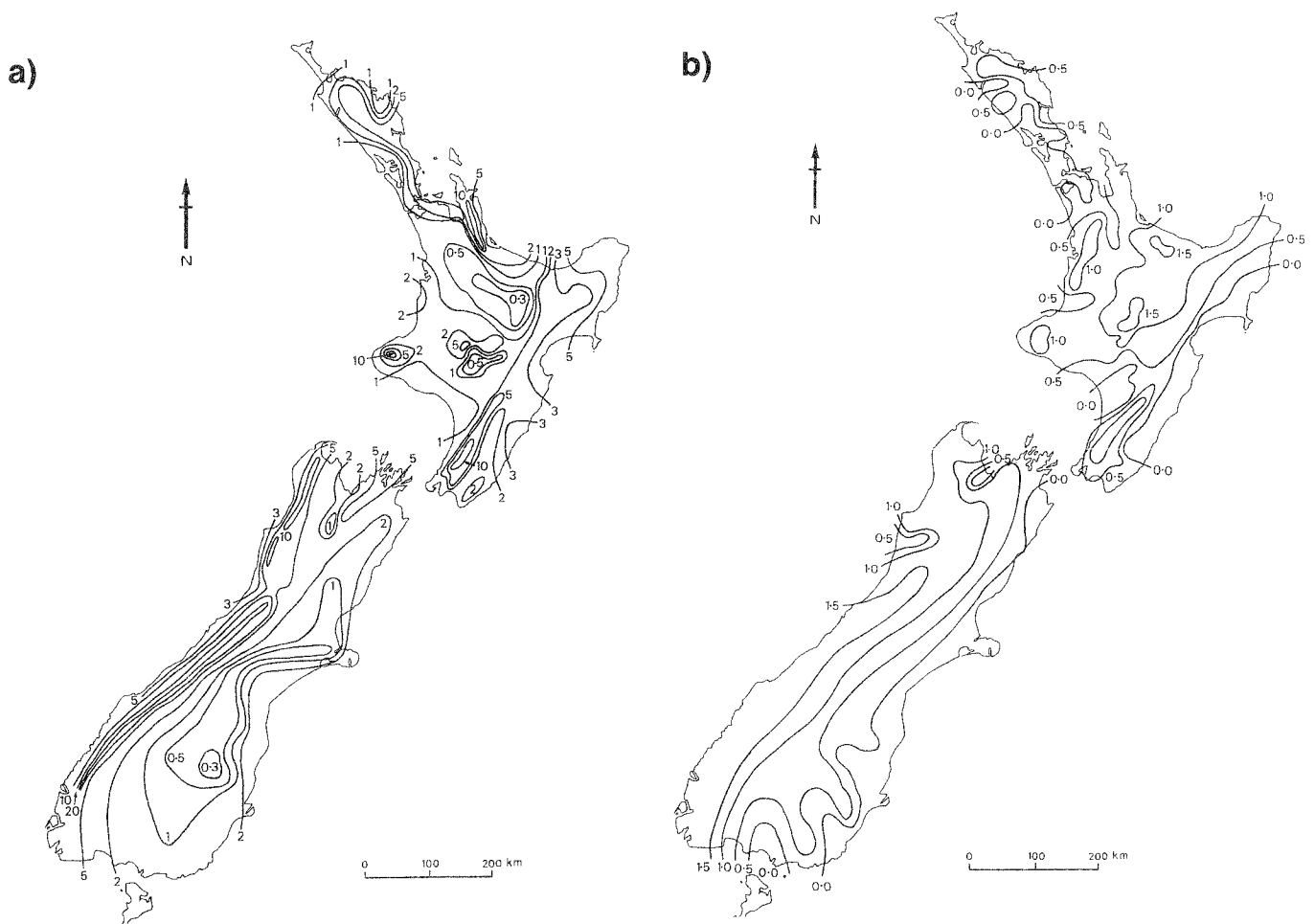


Figure 10.2 Patterns of average New Zealand floods and low flows: (a) mean annual floods per catchment area raised to the power of 0.8 ($\text{m}^3 \text{s}^{-1} \text{km}^{-1.6}$ units, from McKerchar and Pearson, 1989), and (b) logarithms (base 10) of mean annual seven-day low flows per catchment area ($\log_{10} \text{l s}^{-1} \text{km}^{-2}$ units, from Pearson 1995).

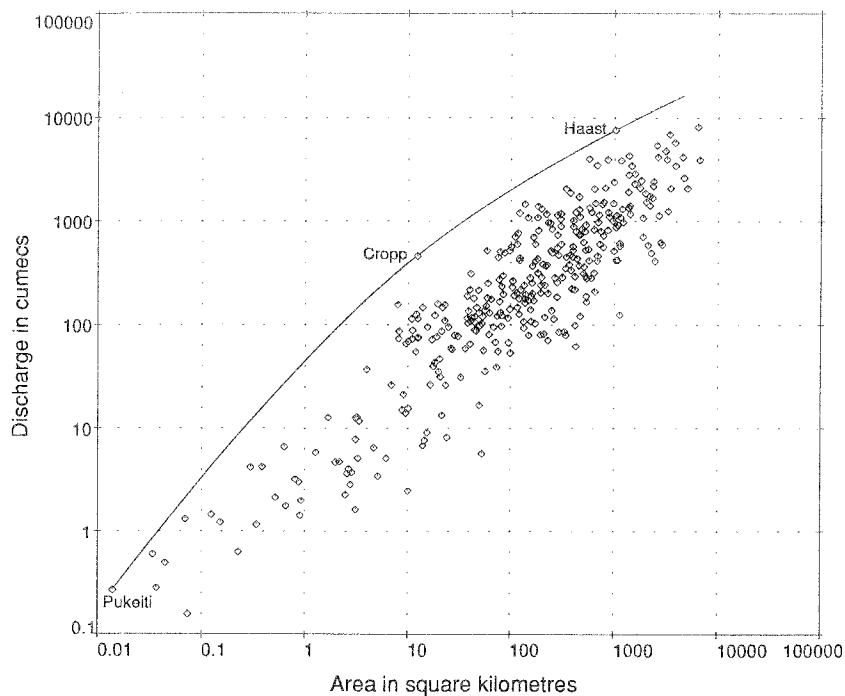


Figure 10.3 Maximum recorded flood peaks from 343 New Zealand catchments (taken from data in McKerchar and Pearson 1989).

from the west, from the south (cold Antarctic fronts), and from the north (tropical cyclones). Low flows tend to occur mainly in the summer period. Mosley and Pearson (1997) provided a review of the major processes leading to floods and low flows in New Zealand rivers. The most extreme flood flows are observed in catchments draining the Southern Alps (such as the maximum recorded flood peaks of the Cropp and Haast rivers shown in Figure 10.3). The envelope curve of New Zealand flood peak extremes in Figure 10.3 is not far below that of world records (Costa 1987).

FREQUENCY DEFINITIONS

To gain some understanding of the likelihood of occurrence of floods and low flows, we need to introduce statistical and probability definitions.

For a river location (one spot on the river network), the probability distribution $F(x)$ of its flood peaks or low flows is a basic characteristic. The distribution function $F(x)$ is the probability of the variable being less than a particular flow value, x . Frequency analysis aims to identify and estimate these probabilities at river locations with flow records (“at-site” frequency analysis). These distributions can be extrapolated to events of low probability, and the information can be used for river locations that lack flow records (“regional” frequency analysis). For effective “regional” frequency analysis—the use of information from monitored catchments to estimate flows at unmonitored river locations—it is important to understand the physical

processes that influence the upper and lower tails of the probability distribution for floods and low flows. Knowledge of these factors can help in estimating the frequency of flow extremes. These factors include climate (rainfall and storm characteristics), catchment physiography (topography, slope, soil properties, geology, vegetation, land-use), catchment scale (drainage area, hillslope versus river channel processes, Blöschl 1996), and river hydraulics (channel networks, lakes, river gorges, Wolff and Burges 1994).

Commonly analysed data in New Zealand include the annual minimum low flows (usually 7-day mean flows), annual maximum flood peaks, and “peaks over a threshold” (also known as “partial duration series”). For low flows, the one-year periods are usually taken from mid-winter to mid-winter, to avoid splitting the summer low flow period. For floods, the calendar year is mostly used to define the

annual partition, to avoid splitting the winter period. In New Zealand large floods can occur at any time of the year, but in many catchments are less likely in summer. For peaks over a threshold, the partitioning is not done by time (year) but by flow threshold value. All values greater than a specified threshold are selected for analysis. Both annual maxima and partial duration series are used mainly for analysing flood flows, but can also be used to analyse low flows when looking at maxima of streamflow deficits.

Observations on flows over time are subject to measurement and systematic errors because of the equipment and procedures used. The frequency analyst needs to be aware of the likely extent of these errors (i.e., the quality of the data) prior to analysis; the analysis also introduces its own sampling and model errors. The “accuracy” of data and of estimation methods relates to the extent of systematic error in the data and bias in the method, respectively. Similarly, the “precision” of data and statistical estimates relates to the measurement errors in the data and sample size for the analysis, respectively.

Before frequency analysis is carried out, the underlying time series must be checked for errors. A time series plot will show up data spikes, gaps and unexpected datum shifts. Data from nearby stations should be plotted wherever possible to verify or exclude data. For example, if a period of missing record were compared with data for that period from a nearby station, it might show that the missing record would have contained an annual minimum or maximum, or an event above or below a key threshold. If an accurate estimate cannot be made, then the statistic

that would have been extracted for that piece of record should be excluded from the rest of the analysis.

While water storage or flow data from rivers, lakes, reservoirs and groundwater reserves may be an accurate and precise measurement of the storage or flow fluctuations, the storages or flows may be subject to human interference. Particularly during summer periods and during growing seasons when droughts occur, substantial quantities of water may be abstracted for water supply and irrigation. However natural records are needed for statistical analyses of low flows and droughts. Data affected by abstractions therefore need to be “naturalised” using time series estimates of the abstractions prior to analysis.

Important definitions and concepts for understanding frequency methods are described below (more details can be found in, for example, Pearson 1992; Stedinger *et al.* 1993; Pearson and Davies 1997).

To analyse a sample, we usually assume “independence” and check that each observation is not influenced by earlier observations. If we select events that are too close together (such as weekly flow minima) then we may find that they are correlated in some way (i.e., not independent), which will violate the assumptions underlying most statistical analysis.

Distribution functions $F(x)$

For random variables, which can take a value between a minimum and maximum, we can define a probability density function (such as the normal distribution’s bell-shaped curve) and cumulative distribution functions to apportion probability to the random variable. Distribution functions are defined using mathematical formulae with a number of unknown “parameters” (usually two or more). Parameters, once estimated for a sample, completely specify the mathematical relationship between flow (x) and probability distribution (F). For distributions with more parameters, the function shape has more flexibility. Mathematical formulae for distributions mentioned in this chapter and their parameters are listed in Table 10.1.

Parameter estimation

There are numerous methods of fitting statistical distributions to sample data, such as graphical methods, methods of moments, maximum likelihood, least squares, maximum entropy, and probability-weighted moments (e.g., Hosking 1990). All methods have in common the aim of quantifying parameters of statistical distributions using the sample data.

Extreme value distributions

The Extreme Value Type 1 distribution (EV1 or Gumbel distribution) is the simplest of the extreme value family of distributions. It is not symmetrical, like the more

familiar normal distribution. The EV1 distribution is positively skewed, with most values in the lower range, and fewer in the tail of higher values. It has two parameters, one a measure of distribution “location” and the other a measure of “scale”. Other statistical distributions used for frequency analysis have longer and/or thicker tails, reflecting differences in the flow regimes of the rivers.

In contrast to other statistical distributions used for extremes, the EV1 distribution has a theoretical justification for flood peaks, although strict application of extreme-value theory requires that the number of events in each time period is large (so that the maximum for each year is the maximum of say 10 or more events). The EV1 distribution belongs to a family of extreme value distributions, the Generalised Extreme Value (GEV) distribution (Hosking *et al.* 1985). The GEV has three parameters (location, scale and “shape”), which describe its mathematical form. The shape parameter (k) determines whether the distribution is of Type 1, 2, or 3 (EV1, EV2, or EV3). When k has a value of zero the distribution is Type EV1 (that is, it becomes a two-parameter distribution). When $k < 0$ the distribution is termed EV2, and when $k > 0$ the distribution is termed EV3. On “Gumbel” paper an EV2 distribution curves upwards, and an EV3 distribution curves downwards. Statistical tests can be used to test whether the EV1 distribution is a better fit than the EV2 or EV3 alternatives (e.g., Hosking *et al.* 1985). Figures 10.4 and 10.5 show extreme value distributions fitted to New Zealand low flows and flood flows, illustrating the difference in fitted distributions $F(x)$, and different scales often used in hydrology for the x axis: Gumbel reduced variates, normal variates and logarithmic scales.

Traditionally, the exponential distribution (exponential decay related to the likelihood of larger events) has been used to model exceedances above a threshold level in the partial duration series (Shane and Lynn 1964; Todorovic and Zelenbasic 1970). More recent research has focused on the generalised Pareto distribution for partial duration series (e.g., Hosking and Wallis 1987; Rosbjerg *et al.* 1992), which contains the exponential distribution as a special case. The assumptions of, respectively, generalised Pareto-distributed exceedance magnitudes and a Poisson-distributed (exponential distribution of time periods between events) number of threshold exceedances in the partial duration series imply that the annual maxima follow the generalised extreme value (GEV) distribution with the same shape parameter (k) as in the generalised Pareto distribution. The same relationship applies between the exponential and EV1 distributions.

Stationarity

Stationarity is a key assumption of frequency analysis—that the data exhibit no trends or cycles, and that they are

Table 10.1 Some cumulative probability distribution functions $F(x)$ used for floods and low flows (x). In some cases the formula for x in terms of F is given, for estimation of frequency flows for a given F (e.g., for an AEP=0.01, $F=0.99$, and this value of F can be inserted into an $x(F)$ equation to give a 100-year return period flood estimate; similarly $F=0.2$ can be used to give a 5-year return period low flow estimate).

Distribution	$F(x)$	Parameters	Remarks
Normal	$F(x) = \Psi [(x - \mu) / \sigma]$	Two: mean μ , standard deviation σ	Used mainly for averaged data. Ψ is the Standard Normal cumulative distribution function
Exponential	$F(x) = 1 - \exp[-(x - u) / \alpha]$ $x(F) = u - \alpha \log(1 - F)$	Two: location u , scale α	Used for partial duration series
Extreme Value Type I (EV1, or Gumbel)	$F(x) = \exp[-\exp(-(x-u) / \alpha)]$ $x(F) = u - \alpha \log(-\log F)$	Two: location u , scale α	Used for floods mainly
Weibull	$F(x) = 1 - \exp[-\{(x - \epsilon) / \beta\}^\delta]$ $x(F) = \epsilon + \beta \{-\log(1 - F)\}^{1/\delta}$	Three: lower bound ϵ , scale β , shape δ	Used for low flows
Generalised Extreme Value (GEV)	$F(x) = \exp[-\{1 - k(x - u) / \alpha\}^{1/k}]$ $x(F) = u + \alpha \{1 - (-\log F)^k\} / k$	Three: location u , scale α , shape k	EV2: $k < 0$; EV3: $k > 0$ Reverts to the EV1 distribution for $k = 0$
Generalised Pareto (GPA)	$F(x) = 1 - \{1 - k(x - u) / \alpha\}^{1/k}$ $x(F) = u + \alpha \{1 - (1 - F)^k\} / k$	Three: location u , scale α , shape k	Used for partial duration series. Reverts to the Exponential distribution for $k = 0$
Log-normal	$F(x) = \Psi [\{\log(x - \epsilon) - \mu\} / \sigma]$	Three: lower bound ϵ , and mean μ , standard deviation σ of logged values	Used for both floods and low flows
Two-Component Extreme Value (TCEV)	$F(x) = \exp[-\lambda_1 \exp(-x/\theta_1) - \lambda_2 \exp(-x/\theta_2)]$	Four: rate of occurrence λ_1, λ_2 , with means θ_1, θ_2 , for the basic and outlying series	Equivalent to two EV1 distributions

drawn randomly (independently) from the same statistical distribution that has applied in the past and will continue to apply in the future. With our improved understanding of climate change, land-use change and climate variability patterns such as El Niño-Southern Oscillation (ENSO) and the Interdecadal Pacific Oscillation (IPO), we are more aware of variations over time (“non-stationarity”) in the frequency and magnitude of droughts, and the non-stationarity of probability distributions for low flows and floods.

Plotting positions

Plotting positions are formulae for plotting data against probabilities. Plotting positions are important for visually checking the behaviour of the frequency distribution at the tails. The formula used most often to calculate plotting positions for floods (e.g., Fig.10.4) and low flows is that of Gringorten (1963; see e.g., Pearson 1992).

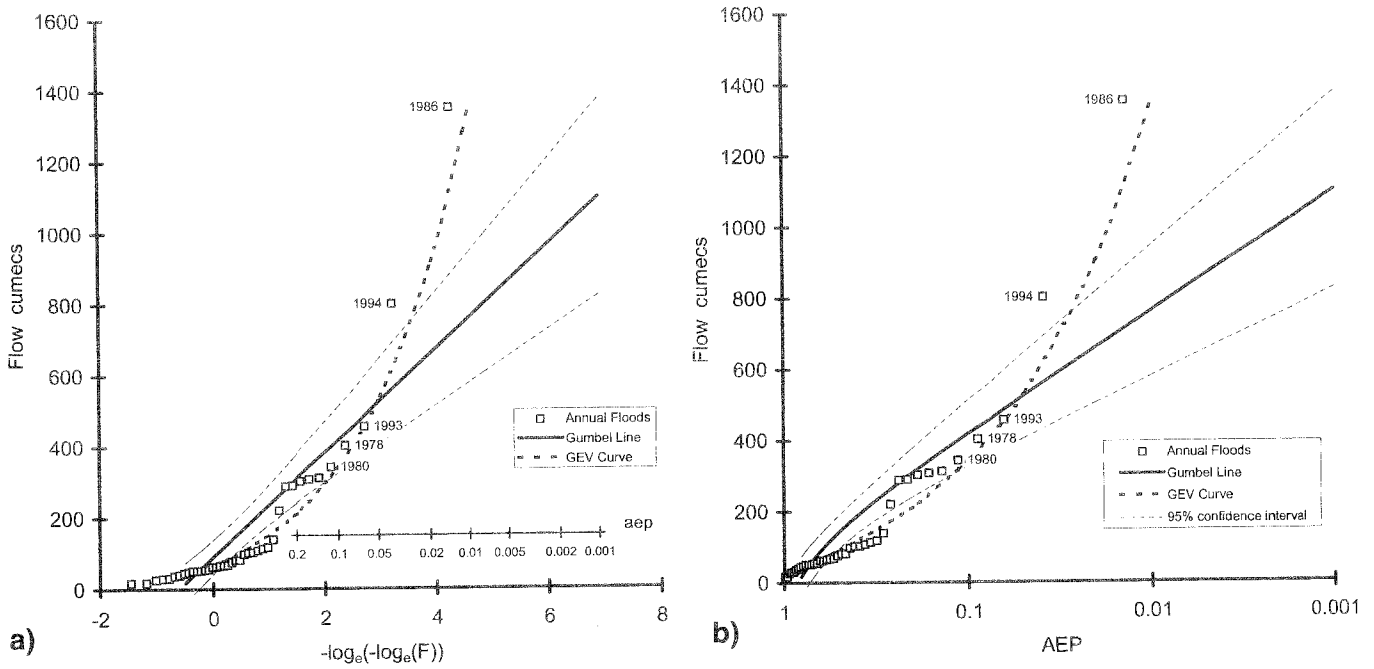


Figure 10.4 Flood distributions $F(x)$ fitted to Hakataramea at Above Main Highway Bridge (site 71103, catchment area 899 km², Walter 2000) annual maximum flood peaks (1964-2003) using (a) Gumbel reduced variate and (b) logarithmic horizontal axes.

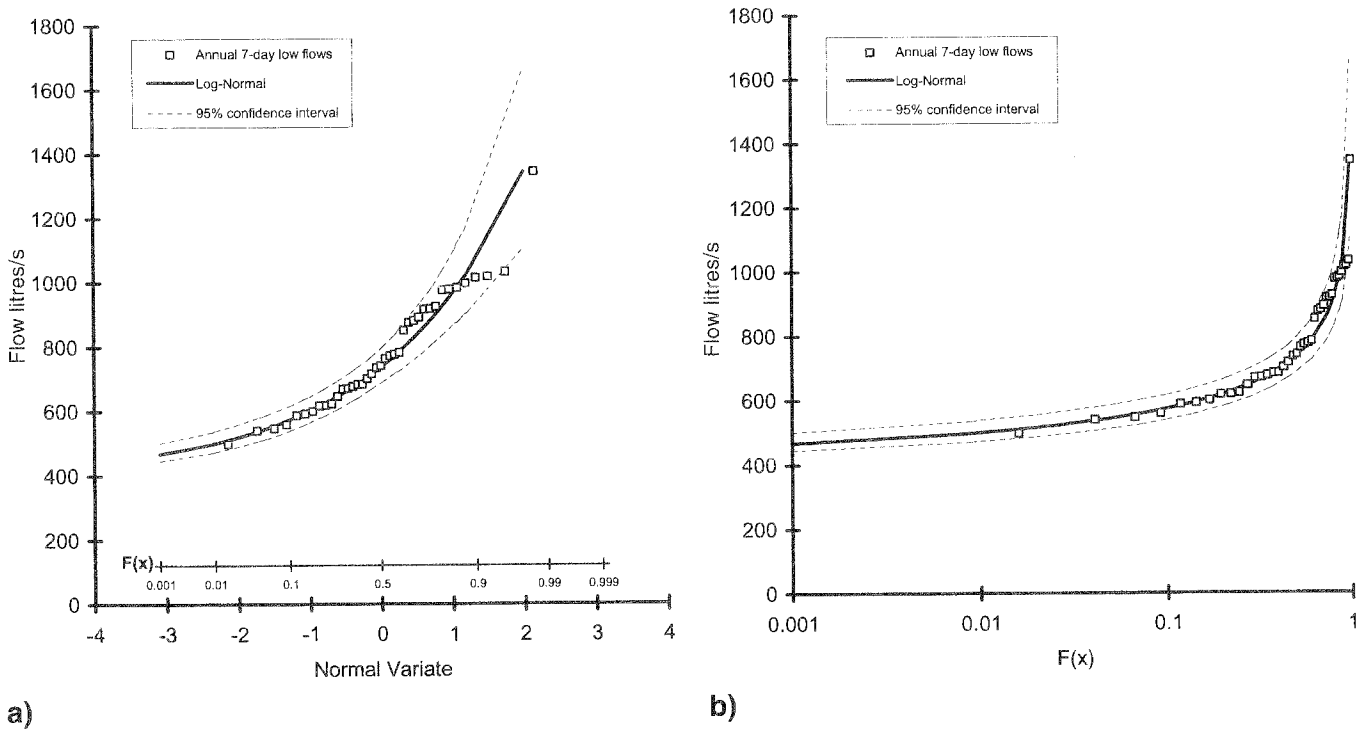


Figure 10.5 Low flow distributions $F(x)$ fitted to Selwyn at Whitecliffs (site 68001, catchment area 164 km², Walter 2000) annual minimum 7-day low flows (1965-2003) using (a) normal reduced variate and (b) logarithmic horizontal axes.

Annual exceedance probabilities, percentiles, quantiles, and return periods

Floods may be described by reference to their peak discharge or water level, the time to peak, and the volume of storm runoff. A commonly used statistical index is the return period of a flood, which is related to the probability that a given discharge is equalled or exceeded in a stated time interval. For example, a flood peak discharge with a 1% probability of being equalled or exceeded in any one year (a 1% annual exceedance probability, AEP) is often described as the flood with a $T = 100$ -year return period (i.e., $T = 1 / \text{AEP}$). A 1% AEP event is defined by the peak value x where $F(x) = 0.99$; this x value is defined as the 0.99 quantile or 99% percentile. The return period T of flow minima is taken as the reciprocal of the probability of non-exceedance, $T = 1/F = 1/(1-\text{AEP})$. For example, the 20-year return period low flow x is the 0.05 quantile or 5% percentile, where $F(x) = 0.05$.

Risk

The binomial risk formula used to estimate the probability r ("risk") of a T -year return period low flow or flood peak occurring at least once in the next L years is $r = 1 - (1 - 1/T)^L$. Because of the different definitions of return period for low flows and floods in relation to $F(x)$, the risk for low flows is $r = 1 - (1 - F)^L$, and $r = 1 - F^L = 1 - (1 - \text{AEP})^L$ for flood flows.

For example, the probabilities that a 100-year return period low or flood flow will occur at least once in the next 20, 50, 100, and 200 years are 18%, 40%, 63%, and 87% respectively. So, a 100-year return period flood or low flow will not necessarily occur once every 100 years. The probability of a flow of this severity or greater occurring once in a 100-year period is 63%. On the other hand, there is a probability of nearly 2% that two or more 100-year return period extreme flows will occur in a given 20-year period. There is a significant probability that rare events may recur in a short time interval, and conversely, not recur for long periods of time. For example, several large floods on the Tongariro River have occurred in the 1960s and 1990s, with few floods in the 1970s.

Low flow distributions

The form of the probability distribution of low flows is more restricted than that of floods. For low flows, the focus is on extreme minima, and as flows cannot be less than zero, the lower bound of $F(x)$ has to be considered. As with annual maximum floods, annual series of flow minima are usually positively skewed. The distributions recommended for low flows include the Weibull (Nathan and McMahon 1990), normal and the log-normal distributions (the latter two distributions are shown in Figure 10.5). The EVI and GEV distributions can be used

for low flow series as well, although they were developed theoretically for maxima. The Weibull distribution is an extreme value distribution developed explicitly for minima. It has a fixed lower bound, and is equivalent to the EV3 distribution for $-x$. For low flows the lower bound can be zero (for ephemeral streams) or a positive value. The normal distribution, with its symmetry and infinite lower bound, is less likely to apply to low flows, which are, like floods, most often found to be positively skewed. The log transformation (log-normal distribution) is more likely to be applicable to low flows (and floods) than the normal distribution.

L-moments

L-moments are linear moments of statistical distributions (Hosking 1990). An advantage they have over conventional statistical moments is that they avoid raising data to powers of 2, 3 and 4 as required for variance, skewness and kurtosis respectively. This gives better parameter estimates when the data contain outlying values, as is the case when dealing with extremes. Another advantage is that L-moment ratio analogues of conventional moment ratios such as coefficients of variation, skewness and kurtosis are more reliable for discerning homogeneous regions and identifying likely parent statistical distributions.

L-moment ratios (L-CV, L-skewness, L-kurtosis) are analogous but superior to conventional moment ratios (coefficients of variation, skewness and kurtosis; the coefficient of variation is the standard deviation divided by the mean). L-moment ratio diagrams are used to compare sample values from a number of sites with their population values for statistical distributions, and so for determining the nature of flow frequencies for regions and countries (e.g., Vogel and Wilson 1996). Decisions on regional groupings and likely distributions for frequency analysis can be made using these diagrams. Hosking and Wallis (1993) have developed statistical tests for these purposes.

At-site analysis

For sampling locations with more than ten observations, a statistical analysis of these data can be carried out using a two-parameter distribution to obtain estimates of return periods. A rule of thumb is not to extrapolate to return periods too far beyond twice the sample size (if dealing with annual minima or maxima).

Regional analysis

Hydrological time series available for a river location are usually too short (or, all too often, non-existent) to make reliable predictions of extreme events. Regional information can be used to reduce the uncertainty by

introducing more data from monitoring stations in a region likely to have extreme hydrologic behaviour that is similar to that of the site being considered.

Regional analyses of recorded floods and low flows can be used to predict flows for locations for which no flow measurements are available. In such a regional study, the flow series at all sites in the region are analysed, and the flows having some specified return period are estimated. These values, commonly in the form of specific flows (discharge per unit area), can then be applied to catchments for which no measurements are available.

For low flows, storage reservoirs are designed to meet the demand for flow for an acceptable fraction of the time at an acceptable cost. Reservoir capacity can be made equal to the water deficiency volume exceeded with an acceptable probability of occurrence or return period T . If the storage provided is less than the actual water-deficiency volume in a particular year, then failure occurs. Hence, the annual maximum series of drought volumes can be analysed in the same way as for floods. Such analyses assume constant demand for the water. However, if there is a danger that the water-deficiency volume may exceed storage, then demand can be reduced, for example, by increased reliance on thermal rather than hydroelectric electricity generation.

Flow may be measured in as many streams in an area as possible, during periods of low stable flow. If this is done over a range of low flows, with enough measurements (e.g., ten or more), equations may then be developed for the relationships between the flows in streams with continuous records of flow and those lacking records. These can then be used to predict flows in an unmonitored catchment (e.g., Henderson *et al.* 2003). If no low flow measurements are available for the site of interest, a second method involves development of regression equations (e.g., Vogel and Kroll 1992) for low flow statistics from monitored catchments in the region, using as independent variables catchment attributes such as catchment area, mean rainfall, hydrogeology, and slope. These equations can be used to estimate low flow frequencies at the unmonitored site.

NEW ZEALAND FLOOD FREQUENCIES

New Zealand has a temperate, maritime climate. Its rugged relief, especially in the Southern Alps of the South Island, leads to orographically enhanced storm rainfalls and extreme floods. This section describes the development of an improved understanding of flood frequency behaviour in New Zealand using regional methods. By 1990, geographic flood regions (Beable and McKerchar 1982), flood estimation contour maps and the Extreme Value Type I (EV1) distribution (McKerchar and Pearson 1989) had been prescribed as part of regional flood estimation schemes (e.g., Fig. 10.2). Since 1990 other advances have

been made in regional flood estimation for New Zealand, based upon the method of L-moments. These include the use of catchment characteristics rather than regions to group small catchments, the identification of EV2 tendencies in flood peak and storm rainfall annual maxima, the use of peaks-over-a-threshold sampling for regional analyses, and the use of the Two-Component Extreme Value distribution for regional studies.

Two studies of regional flood frequencies of New Zealand were carried out in the 1980s (Beable and McKerchar 1982; McKerchar and Pearson 1989, 1990). For each of these national flood studies, careful quality checks were carried out (checking stage-discharge rating curves and gaps in records) to obtain annual maximum flood series for analysis.

Beable and McKerchar (1982) followed the regional "index flood" approach of the British flood studies report (National Environment Research Council 1975). More than ten regions were identified for estimation of mean annual flood (the index flood) and dimensionless flood frequency quantiles (growth curves). Mean annual flood was estimated using multiplicative regression equations with catchment characteristics, such as basin area (A), rainfall and slope as independent variables. Distributions used for the flood frequency growth curves were either the EV1 or EV2 distributions. Beable and McKerchar's work substantially improved upon earlier methods, which were based mainly upon storm rainfall statistics. Mosley (1981) used a clustering approach of specific mean annual flood and coefficient of variation of annual maxima to confirm the flood regions identified by Beable and McKerchar.

Almost a decade later, McKerchar and Pearson (1989, 1990) updated this work using an index flood method based upon contour maps (Fig. 10.2a). Mean annual flood (divided by $A^{0.8}$, 343 catchments) and dimensionless 100-year return period flood (x_{100} , 275 catchments) were mapped for New Zealand, and used as the basis of a flood estimation procedure. The EV1 distribution was shown to be satisfactory against the Generalised Extreme Value (GEV) distribution alternative (EV2, EV3) for most annual maximum flood peak series, although series from small catchments (area less than 100 km²) and from catchments of eastern regions did not match this distribution as well. For larger catchments (area greater than 100 km²), mean annual flood and x_{100} had prediction standard errors of 17% and 30%, respectively.

Regional L-moments methods developed by Hosking and Wallis (1993) and Hosking (1990) allowed groups of catchments to be tested for homogeneity with respect to flood frequency, to identify the most appropriate distributions for these groups. L-moment ratio plots of L-kurtosis versus L-skewness assist greatly in specifying upper tail behaviour of flood series distributions, $F(x)$ (Vogel and

Fenessey 1993).

Follow-up studies (Pearson 1991a, 1991b) to the national studies used L-moment statistics to show that for some catchments, the EV2 distribution was more appropriate than the EV1 distribution for the McKerchar and Pearson flood data set. Numerous regional studies since have used these tests to identify tendencies toward EV2 distributions in many New Zealand flood series (including those of the North Island), particularly for eastern regions where flooding is less frequent on an annual basis. In eastern regions (e.g., coastal Canterbury of the central South Island), annual maximum series had predominantly low flood peaks, punctuated by a few relatively large values. In general, the EV1 distribution was found to be satisfactory for the flood frequencies of western New Zealand, but elsewhere the EV2 was more appropriate.

Pearson (1991b) examined the flood frequency of over 100 small catchments (less than 100 km²) of McKerchar and Pearson's data set, by identifying sub-groupings based on catchment characteristics. The characteristics used included catchment geology and soil parameters. Optimal groupings were found using storm rainfall and catchment slope parameters with L-moment ratios in a grouping scheme developed by Wiltshire (1985). The groups were not contiguous regions. Different regional flood frequency growth curves were found for six groups, partitioned into low or high catchment slopes and low, medium or high catchment storm rainfall. The tendencies toward an EV2 distribution were greatest for catchments with low storm rainfalls and high catchment slopes. For all but the high rainfall groups, the McKerchar and Pearson EV1 100-year flood estimates were shown to be underestimates.

The value of using peaks-over-a-threshold series compared with annual maximum flood series was investigated for east and west South Island regions (Madsen *et al.* 1997). Selecting flood data using peaks over a threshold allowed a two-fold or greater increase in flood series sample size (n). Rather than identifying east and west regions, an optimal partitioning was achieved using the Pearson (1991b) scheme with annual rainfall—west catchments had rainfall over 1300 mm, whereas east catchments were below this threshold. In general, the exponential distribution was found to be satisfactory for the flood frequency of the peaks over a threshold series of the western catchments, and the Generalised Pareto distribution was more appropriate for the east. These results were equivalent to the EV1 and EV2 findings for annual maxima of the west and east respectively. More importantly, the use of larger flood sample sizes for peaks over a threshold series (versus annual maxima series) led to greater precision in the L-moment ratios, and hence greater ability to identify homogeneous groupings and

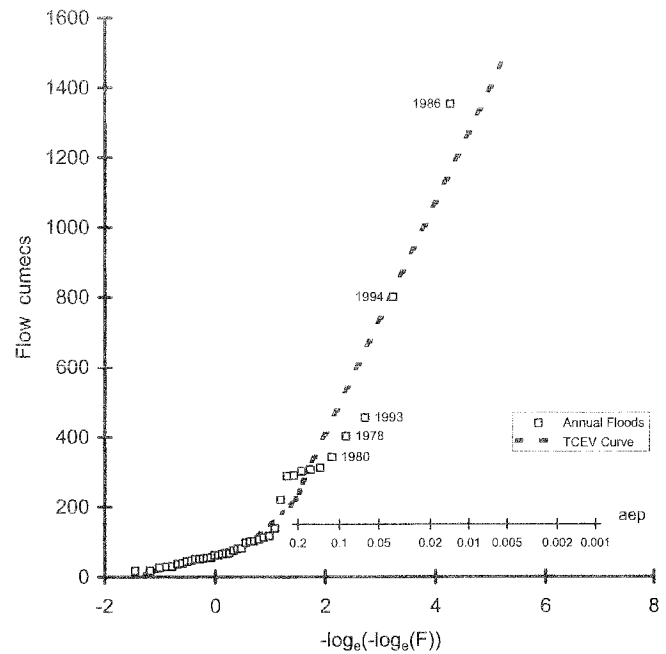


Figure 10.6 Two-Component Extreme Value distribution $F(x)$ fitted to Hakataramea at Above Main Highway Bridge (site 71103, catchment area 899 km², Walter 2000) annual maximum flood peaks (1964-2003).

corresponding distributions.

The EV2 tendencies of flood series from Canterbury rivers (central, east South Island) were investigated further by Connell and Pearson (2001). Rather than the extreme curvature of some EV2 distributions (e.g., Fig. 10.4), the more conservative approach of the Two-Component Extreme Value distribution (Rossi *et al.* 1984; Beran *et al.* 1986) was used. The Two-Component Extreme Value distribution comprises a frequent basic series and an infrequent “outlying series”, and is equivalent to two EV1 distributions—one line less steep on Gumbel axes for routine annual floods, and a steeper line for large, outlying floods (Fig. 10.6).

Analysis of annual maximum flood series from Canterbury showed that the Two-Component Extreme Value distribution fits many of these series, and that it is a better distribution to use for regional flood studies than the EV2 distribution. Distinct regions were identified for Canterbury using L-moment ratios. South Canterbury east coast rivers have a marked two-component tendency while North Canterbury east coast rivers have a lesser but significant two-component tendency. The difference in strength of the Two-Component Extreme Value tendencies between the north and south east coast rivers was due to orographic effects and to the alignment of the upper catchment boundaries. South Canterbury rivers receive the most severe storm rainfall from warmer north-easterly

storms, whereas North Canterbury rivers receive most rainfall from cooler south-easterly storms. For Main Divide rivers, the third Canterbury region, the predominant storm direction is from the west. This was strong enough to be the only EV1 process evident in the flood series and hence this distribution continues to be the preferred option for these (and West Coast) rivers. As flood series from some West Coast rivers tend towards an EV2 distribution, some weak Two-Component Extreme Value signals were detected in some series from this region.

The additional data, results of more recent studies and better methods available since the national flood studies of the 1980s, provide a basis for revising national procedures for estimating New Zealand flood frequencies. Other methods under investigation include using improved physically-based rainfall-runoff models now available with continuous rainfall simulations and data; more studies on the importance of scaling and on the multi-scaling of flood peaks; development of a peaks-over-a-threshold equivalent to the Two-Component Extreme Value distribution; and GIS methods to delineate parameters of $F(x)$ with river channel networks (as illustrated in the Swiss Hydrological Atlas, Weingartner 1999).

Storm rainfalls

Estimates of the frequencies of storm rainfalls are used with rainfall-runoff models to estimate the frequency of flood peaks and volumes in rivers. The tendencies toward EV2 distributions in many New Zealand annual maxima flood series are associated with both atmospheric and land factors. With storm rainfalls, Tomlinson (1980) concluded

that the EV1 distribution was satisfactory to estimate frequencies of annual maximum storm rainfalls in New Zealand. Pearson and Henderson (1998) checked the applicability of the EV1 distribution for annual maximum storm rainfalls for durations of 1, 6 and 24 hours. The EV2 distribution was found to be more applicable for most of New Zealand, in that the shape parameter k of the GEV distribution was predominantly negative. Figure 10.7 shows the storm rainfall distribution $F(x)$ for 24-hour storm rainfalls for the east Southland region, indicating the extent to which the EV1 distribution underestimates storms at higher return periods. The GEV distribution has since been used in a national review of storm rainfalls (Thompson 2002).

NEW ZEALAND LOW FLOW FREQUENCIES

Low flows and droughts

Low flows most often occur in summer (except in alpine catchments such as the Hooker and Rakaia), and extreme low flows are usually related to droughts. Droughts can be defined and classified in numerous ways, such as by the community they affect (e.g., agricultural, horticultural, water supply, instream ecosystem droughts) or by a component or process of the hydrological cycle (e.g., rainfall, soil moisture, groundwater, or streamflow). A drought is essentially a shortage of water that critically affects one or more water uses, and therefore users. The primary cause of drought is lack of rain, followed by depletion of soil moisture, wilting vegetation, low streamflows and lowered ground water tables. The usual cause of drought is a lack of rain-producing weather systems, for example when a stable anticyclone prevents moist airflows from reaching land. Droughts manifest themselves slowly over time (compared with the sudden onset of storms). During a drought, a variable such as rainfall, soil moisture or streamflow can be monitored at a particular location continuously through time. The resulting time series can be analysed to define the onset and breaking of the drought at this location, and its overall duration. Both the severity at a particular location and the spatial extent of a drought usually grow with time. Mapping of variables plays an important role in drought analysis—from point measurements of rainfall or soil moisture, the spatial development of droughts can be mapped through time.

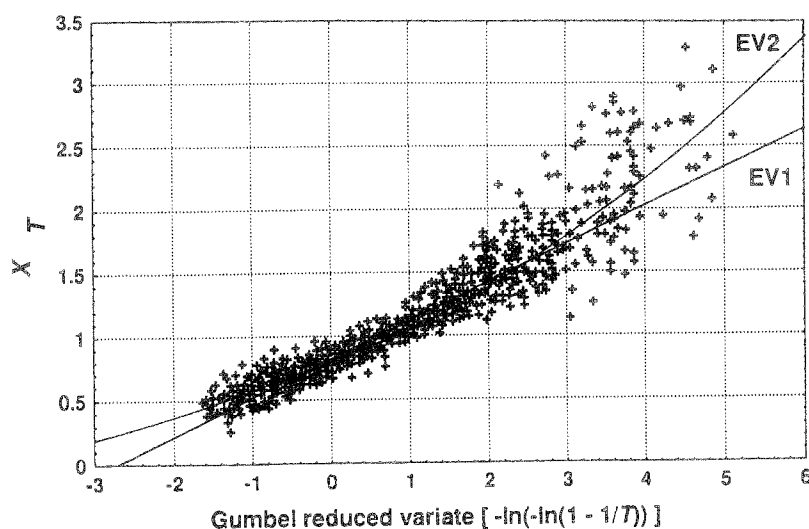


Figure 10.7 Storm rainfall distributions $F(x)$ fitted to 83 series of eastern Southland annual maximum 24-hour duration storm rainfalls (each divided by their mean annual 24-hour maxima), with fitted EV2 (curve) and EV1 (line) distributions.

Hydrological analysts, in conjunction with irrigators, stream ecologists, resource managers

or other end-users of the statistical information, need to select an appropriate variable to describe water shortage for a particular situation, and define the extreme occurrence or threshold of this variable that defines drought. The variable is usually extracted from time series of measured hydrological data, but could also be a variable that defines the effect of the drought, such as cost to a community or stream ecosystem.

Frequency analysis can be applied to a variety of defined meteorological, climatological and hydrological variables to establish probability statements about droughts. The underlying time series data need careful consideration before analysis. Continuous climate and hydrological time series data, and relevant land and catchment characteristics, are usually available from national and local hydrological, climatological and terrestrial databases. Long records of rainfall and river flows typically extend back to the 19th century for many such databases in the world. Historical and anecdotal data extending back beyond the start of continuous recording can also be used, as well as surrogate data such as time series of tree rings and lake varves.

Useful data types for analysing droughts are single (univariate) time series of variables such as rain totals, barometric pressure, air temperature, wind speed, solar radiation, evapotranspiration, snow volume and elevation, soil moisture, groundwater, lake levels and streamflow. Variables may be combined in some form such as the Palmer Drought Severity Index, which uses monthly mean precipitation and air temperature as part of a monthly time series water balance model (see e.g., Guttman 1991; Guttman *et al.* 1992; Hu and Willson 2000), or estimates of soil moisture deficit using daily rainfall and air temperature data.

Some variables can be measured spatially (using techniques such as aerial photography, remote sensing, radar) or mapped to provide time series maps of the spatial extent of the variable. These can also be analysed statistically to measure the severity, duration and frequency of a drought. Maps of mean monthly flows across New Zealand give water users and resource managers a clear picture of the

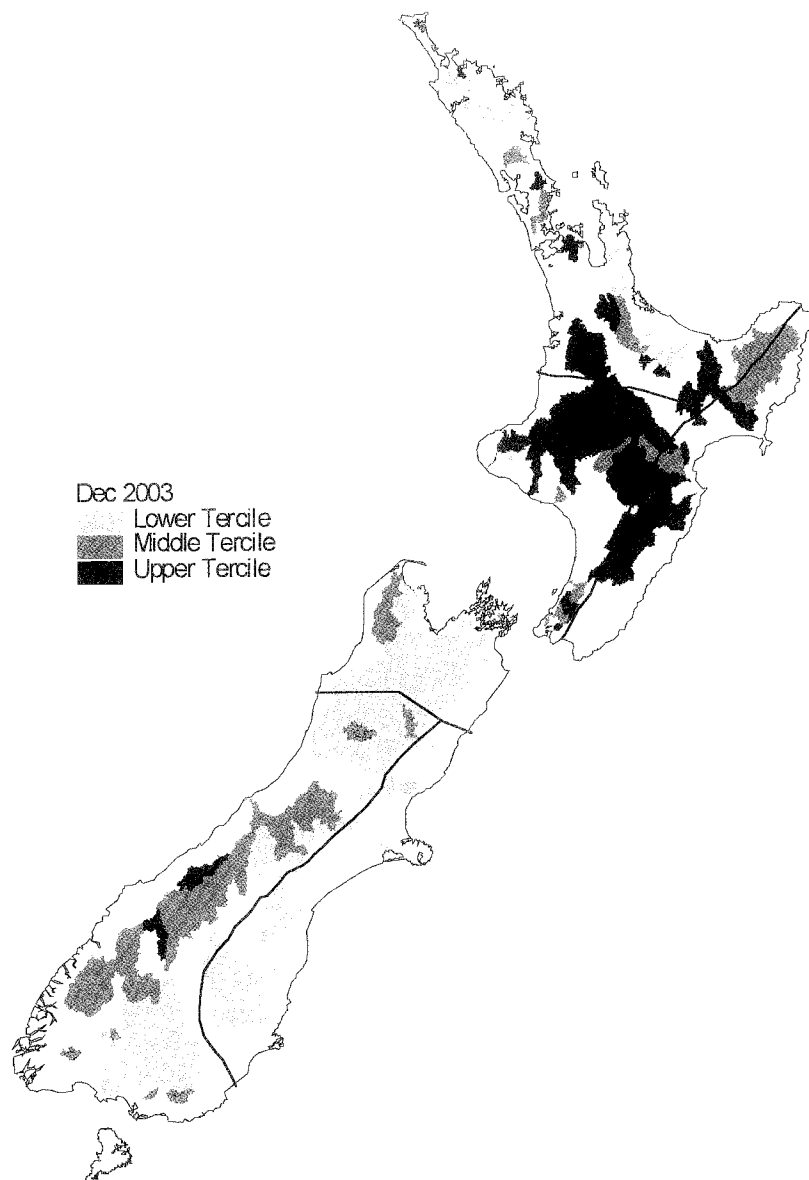


Figure 10.8 December 2003 monthly mean flows for monitored New Zealand catchments (shaded regions), expressed as “terciles” (lower, middle, and upper distribution thirds) of historical December flows for each river catchment.

development of droughts across regions and catchments and also show the end of droughts (e.g., Fig. 10.8).

Low flow frequencies

In low flow hydrology, hydrologists are principally interested in whether or not a river or stream can supply a given demand for water. Water may be needed for domestic or industrial use, for irrigation of farmland, for hydroelectric power generation, for recreational river use, or to maintain wildlife habitat. If demand cannot always be met, water must be stored or the demand must be managed.

As with analysis of flood flows, analysis of low flows gives estimates of the low flow with a probability of

occurrence of $1/T$. If the value of a given low flow is large in comparison with the demand flow, then the river can meet the demand satisfactorily. Particular care needs to be taken in analysing low flows to ensure the flows have not been affected by upstream abstractions. If they have been, they need to be adjusted by comparing them with flow records from unaffected upstream sections or streams nearby, with rainfall records, or with records of known abstractions.

Low flow statistics used in frequency analyses are usually annual minimum n -day mean flows, where n is often taken to be 7 days, i.e., the week in each year with the lowest average flow.

Difficulties arise in the frequency analysis of low flows when the data include zero values, because a stream has dried up. Nathan and McMahon (1990) addressed the zero flow problem by first analysing low flow frequency for the non-zero flows, and then modifying this result by considering the probability of zero flows.

Pearson (1995) used annual minimum low flow series (1-day, 7-day and 30-day mean flows) from 500 catchments to investigate regional patterns and frequency distributions of low flows within New Zealand (Fig. 10.2b). Catchment characteristics (rainfall, soil porosity, vegetation, slope, elevation, hydrogeology) were used to help explain regional variations. Probabilities of zero flows in annual minimum series were estimated using logistic regression on river catchment area and mean precipitation. Even though the regression result was too weak for satisfactory prediction, it showed that in drier regions streams with smaller catchment areas were more likely to run dry. L-moment ratios of non-zero low flow series were used to test the homogeneity of regional and non-geographic groupings of river catchments, and to identify candidate statistical distributions for each group. Regional groupings based on Hutchinson's (1990) low flow regions were of varying homogeneity. A homogeneous Bay of Plenty-Rotorua sub-region was identified from the heterogeneous North Island central volcanic region. The heterogeneous nature of most of the groupings analysed reflects the wide range of frequency distributions required to describe adequately New Zealand annual low flow series. The prediction equations of Pearson (1995) are positively biased in very dry catchments. A new approach under investigation at this time is to use a physically based conceptual model that separates the climatic causes of low flow from the catchment characteristics that modulate the release of water. The climatic effects are generally smoothly varying across the landscape, whereas the catchment characteristics vary sharply along discontinuities in geology and topography. Early results are promising, but the ultimate application relies on being able to estimate the recession behaviour of ungauged catchments from the available physical characteristics.

There is considerable practical interest in improving our understanding of climate and streamflow variability and in its prediction, as community and environmental benefits will accrue if flood and drought conditions can be predicted. Our growing understanding of the climate patterns associated with the El Niño-Southern Oscillation (ENSO) phenomenon (Chapter 2) provides a potentially powerful tool for medium- to long-term prediction of droughts and low flows. Several recent dry periods are ENSO-related. The El Niño-Southern Oscillation is an irregular inter-annual fluctuation in global climate and the circulation of the tropical Pacific Ocean and atmosphere—warmer than normal sea temperatures in the eastern equatorial Pacific Ocean define an El Niño event. The Southern Alps drought of 1992 was associated with a low Southern Oscillation Index (an indicator of an El Niño event). There were few weather systems bringing moist westerly winds to the west coast of the South Island (of national importance for inflows to hydroelectric lakes). The Canterbury drought of 1987-1989, on the other hand, was associated with a high Southern Oscillation Index.

McKerchar *et al.* (1996) showed that for rivers draining into major Southern Alps lakes there is an inverse correlation between the summer (December/January/February) inflows and the precursor spring (September/October/November) Southern Oscillation Index (SOI). On average, summer inflows are lower and less variable when the spring SOI is positive (indicating the La Niña state), as compared with summer inflows when the spring SOI is negative, indicating the El Niño state. The result is of interest for long-term prediction and for scheduling the operation of lake storages for hydroelectric power generation. The reason for the observed dependency is that summer rainfalls in the wetter parts of the Southern Alps also show some dependency on SOI, in that inflows and rainfalls are reduced in years when the SOI is positive—associated with predominantly mild north and east air flows across the country. Conversely, summer snowmelt, which contributes on average 70 percent of the summer inflow to the Waitaki lakes, tends to be greater in years when the SOI is positive, which is consistent with the generally warmer temperatures when SOI is positive.

McKerchar and Henderson (2003) showed that the shift in the Interdecadal Pacific Oscillation in 1978 to a predominance of El Niños over La Niñas, brought an increase in low flows (and mean and flood flows) over the south west zone of the country. This is explained by the predominance of south and west wind flows across New Zealand during El Niño events. This discovery implies that, in fitting probability distributions to flow series, the basic assumption of stationarity may not be valid for some regions of the country. Modifications of conventional

frequency analysis methods are under development to account for climate shifts, and, in the longer term, climate change.

Clausen and Pearson (1995) carried out a regional frequency analysis of annual maximum streamflow deficits to investigate the variability of droughts, using three geographical regions in New Zealand with different climate and physical characteristics—Bay of Plenty, West Coast of the South Island and Canterbury. Using a truncation-level approach (Dracup *et al.* 1980), the critical parameter becomes the flow truncation level, whereas in low flow analysis it is the fixed time duration (e.g., the commonly used 7-day duration). Examples of applied truncation levels are the mean (Bonacci 1993), the median (Griffiths 1990), and lower percentage exceedances, e.g., 90% or 95% exceedance flows found from flow duration curves (Zelenhasic and Salvai 1987; Chang and Stenson 1990). The Clausen and Pearson study showed that alternative definitions of drought river flows can be useful in estimating return periods for flow deficits, and that the regions studied had quite different frequency distributions of streamflow volume deficits.

Auckland 1993-94 drought and low flows

In the North Island a prolonged drought in 1993-94 caused a major water shortage in the Auckland metropolis and increased water demand in the drought-prone northern and eastern regions of the island. The drought was associated with a long lasting El Niño event (the longest since the mid-1910s), where persistent anticyclonic conditions over eastern Australia brought westerly and southwesterly winds and dry conditions over the North Island.

Clausen and Pearson (1997), using streamflow and rainfall data, investigated the frequencies of the 1993-94 drought using different indicators of drought. Frequencies of streamflow deficit and duration were estimated and compared with the frequency of the annual minimum 7-day flow. Also, the frequencies of the total rainfall deficit over three-month seasonal periods, half-annual and annual periods for the Auckland metropolitan area were estimated.

The results revealed the importance of distinguishing between different definitions of droughts, in that droughts can be extreme in one way, but not in another. The study showed that the accumulated deficit in flow for streams was particularly extreme, which led to the water supply crisis. Paradoxically the 7-day minimum for 1993-94 was not exceptionally low when compared with other years. The drought in Auckland was caused by low rainfall during all of 1993-94, an event occurring only once every 50 years on average. However, shorter duration rainfall totals within that year were not particularly extreme. The conclusion was that the 1993-94 drought was extreme because of its duration rather than its magnitude. It is clear that short-

term, fixed-duration measures of drought (e.g., 7 days) cannot always indicate the extremity of a streamflow drought, and other flow measures such as deficits and monthly and seasonal means need to be considered.

CONCLUDING COMMENTS

Extremes in New Zealand's river and stream flows are caused by natural processes, and even though we have developed an understanding of the nature and frequencies of low flows and flood flows, we cannot prevent their occurrence. River floods can be contained within stopbanks and alleviated by floodways and flood detention dams, to a level of risk that a community is prepared to pay for. Similarly, farmers can design their management practices to cater for certain levels of risk that water will not be available when it is needed. Hydrologists provide the underlying data and interpretation of those data in the form of frequency analysis so that decision makers can decide on acceptable levels of risk for communities, businesses and the natural environment.

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Chapter 9

Catchments, streamflow and the use of models

Richard Ibbitt, Alistair McKerchar and Ross Woods

INTRODUCTION

Imagine that you are a raindrop that has just reached the ground. What will your fate be? If you are an “introverted” raindrop you will quietly seep into the ground, and slowly make your way through the soil to the water table, unless some thirsty plant “drinks” you and evaporates your molecules back into the atmosphere. If you escape beyond the reach of plant roots you will eventually join other introverted drops at the water table. Here you are likely to change direction from moving in a predominantly vertical direction to moving with the crowd in a more or less horizontal direction. Eventually you will seep out of the ground, either onto the surface or straight into a stream channel, to join other introverted raindrops and also the “extrovert” raindrops that shunned your quiet seepage path for the more exciting overland route. Extrovert raindrops quickly crowd together in small ponds and as their numbers increase the pond overflows to form a transient streamlet that rushes headlong down hill to join other streamlets that eventually join to form a permanent stream and become part of the streamflow making for the sea.

The streamflow that you add to flows along a channel that you and your predecessors have formed on the ground surface. Your channel will join other channels to create larger channels, and the whole collection of connected channels forms a branched network. At any point in a channel network it is possible to “look” upstream and identify the entire area from which all the water in the channel at the observation point drained. This area is termed the “catchment” draining to the point on the channel of the “observer”. Sometimes a catchment is termed a river basin, a drainage basin, or, in American and Australian usage, a watershed. Whatever term is used, the fascinating property of a catchment is its ability to gather together, under the influence of gravity, all the runoff from precipitation that is widely spread across the landscape, and concentrate it in the narrow threads of river channels.

The purpose of this chapter is to outline the methods commonly used to describe and quantify the pathways followed by raindrops as they pass through the catchment and channel network on their way to the sea. Computers and mathematical tools have enabled the development of quantitative descriptions—or models—of catchment hydrology, and a selection is reviewed by Singh and Woolhiser (2002). This chapter focuses on the underlying principles of model building and application, so that readers are equipped to appraise models that they may encounter. It highlights a particular catchment model, TOPNET, which is widely used in New Zealand and is representative of a number of modern “distributed” catchment models that are used in other countries.

Catchments and drainage networks

Catchments come in many different shapes and can change shape over thousands of years through the process of river capture. The result of millennia of rainfall on the landscape is a complex mosaic of catchments, each drained by its own network of stream channels. The distinguishing features of drainage networks depend mainly on the tectonic activity of a region, the rock types to be found in the catchment, the steepness of the land inside a catchment boundary, and the long-term precipitation on the catchment. Major tectonic features such as the Southern Alps or Mt. Taranaki can cause catchments to drain in either parallel directions (Fig. 9.1a), or in radial directions (Fig. 9.1b). Within a catchment, the rock type (hard versus soft) and the soil type derived from it (shallow versus deep, sandy versus clayey etc.) can determine the form of the network. Figure 9.2 shows some New Zealand stream networks where the differing shapes of the networks can be attributed to geological factors.

Various ways to quantitatively describe stream networks and catchment shapes have been developed (Mosley and

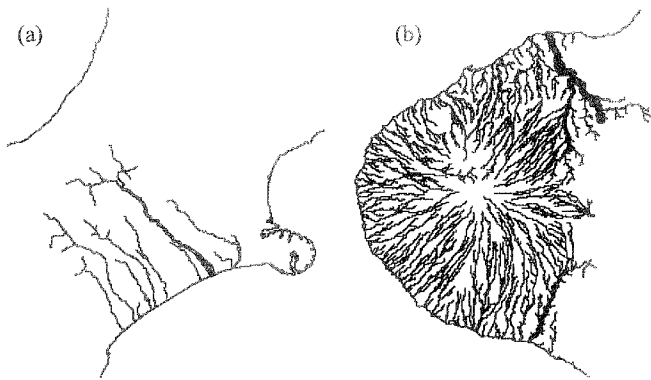


Figure 9.1 Tectonic influence on stream networks (a) parallel channels along the Southern Alps; (b) radial channels around Mt. Taranaki.

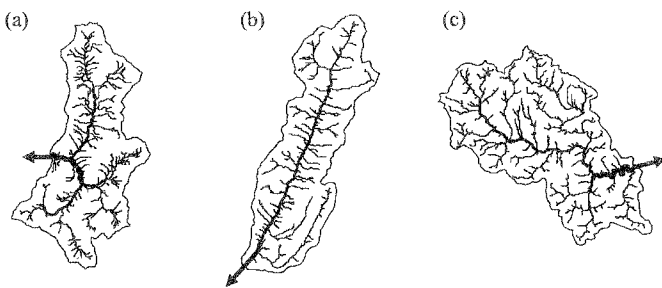


Figure 9.2 Different patterns of stream network (a) Cleddau at Milford Sound; (b) Jollie at Mt Cook; (c) Wharekopae inland from Gisborne.

McKerchar 1993). “Stream order” is one such description. In the Strahler system for example, “fingertip” tributaries are assigned order one. A channel below the junction of two first-order streams is a second-order stream. Two second-order streams combined form a third-order stream and so on. Another useful descriptor is the “width function”. The width function (Kirkby 1976) has important connections to the way that a catchment responds to rainfall. It counts the number of stream channels at each distance from the catchment outlet (also termed the “mouth”). Figure 9.3 shows the width functions corresponding to the stream networks shown in Figure 9.2. As the network is traced upstream from the outlet, the width function generally increases at first, indicating the greater number of channels present in the “middle” of the catchment; the width function decreases as the most distant headwater streams are reached. The importance of the width function can be appreciated by considering what would happen if a coloured block of wood were to be thrown into each stream channel, throughout the network. The colour of each block corresponds to its starting distance along the stream network from its outlet, e.g., red for near the outlet, shading through the colours of the rainbow to purple at the points farthest from the outlet. Assuming for the moment that the water in the stream flows at a constant velocity, then an observer at the catchment outlet would see the following happen. First, only a few red blocks would pass the observer. As time passes, blocks with shades of yellow

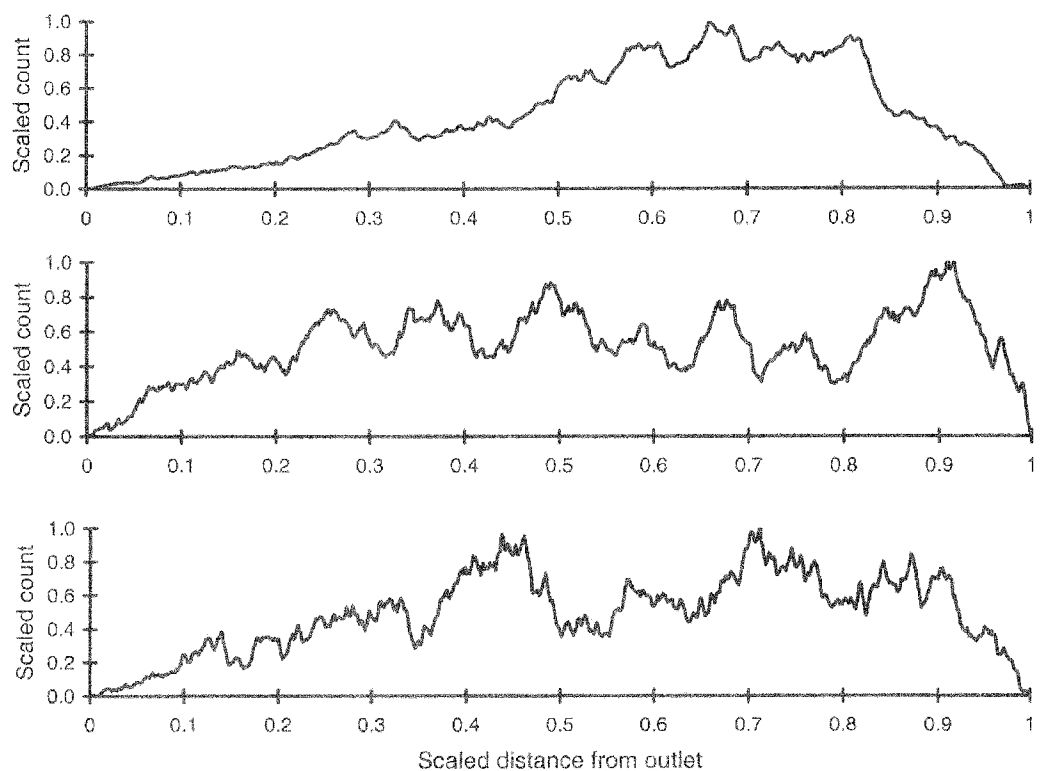


Figure 9.3 Width functions for the networks shown in Figure 9.2. Cleddau (top), Jollie (middle) and Wharekopae (bottom). The axes have been scaled to aid comparison.

and then green, blue, etc. and finally purple arrive. Given that there are usually more stream channels in the “middle” of a catchment, e.g., the Cleddau catchment shown in Figure 9.2a, one might reasonably expect to see more green blocks than either red or purple. The colour of a block indicates how far it has travelled, while the number of blocks of a particular colour indicates how many stream channels are at a particular distance from the outlet. If each block is associated with a fixed amount of water, then the arrival of the blocks indicates the increase and decrease in the amount of water flow coming from different parts of the stream network. A graph of water flow against time is called a hydrograph and is a very important concept in hydrology. Changes in river flow have great significance for people who live near rivers (e.g., when will a flood come?) or who use river water (e.g., will there be enough drinking water?). Figure 9.4 shows a streamflow hydrograph for each of the networks shown in Figure 9.2. This demonstrates some predictability between the shape of a channel network and the hydrograph.

While the width function is useful for understanding how water travels through a catchment, its use for calculating the arrival of the water at the catchment outlet assumes that the water in the stream network travels at a

constant velocity and that the rainfall immediately enters a stream on reaching the ground. This is seldom true—some of the rain never gets to the stream at all but is lost by evapotranspiration or seeps to deep groundwater. In addition, where parts of the network are steeper, water tends to travel faster, although sometimes the increased roughness of the channel bed compensates for the accelerating effect of gravity. Channels may become rougher as the water level in the channel rises because the water has to swirl around vegetation protruding into the space that the river wants to occupy. Also, water flowing in channels travels at different speeds at different depths. Despite these limitations, the importance of the width function is that it enables an approximate calculation of the times at which rain falling on a catchment will arrive at the outlet, without the need to measure the actual flow in the river, which is an expensive and time-consuming process.

UNIT HYDROGRAPHS

First developed by Sherman (1932) in the pre-computer era, the unit hydrograph is an easy-to-use method to calculate the rate at which streamflow arrives at a catchment outlet during a storm. The unit hydrograph, or unit graph, is a standard response of a catchment to a standard amount of rainfall; it assumes that catchments always behave in the same way when subject to the same rainfall. It is a very useful concept, as well as a practical engineering tool. If 1 mm of rain falls on an impermeable catchment in one hour, then the unit hydrograph for that catchment estimates how much water will reach the catchment outlet within that hour, within the next hour, and so on. The unit hydrograph is applied to each hour of rain independently, and the results are added together. The four “staircase” graphs in the centre of Figure 9.5 show a unit hydrograph being used with four different hourly rainfalls. Sherman’s approach had the advantage that by working directly with measured streamflows he was able to automatically take into account variations in the velocity of streamflow throughout a stream network, as well as the time taken for rainfall to find its way into the nearest channel. However, the unit hydrograph involves some complications. The flows observed are caused by temporally varying rainfall, and since no two storms have the same temporal pattern, a method is required to account for the effect of rainfall variation if the main unit hydrograph assumption is not to be violated. Even with

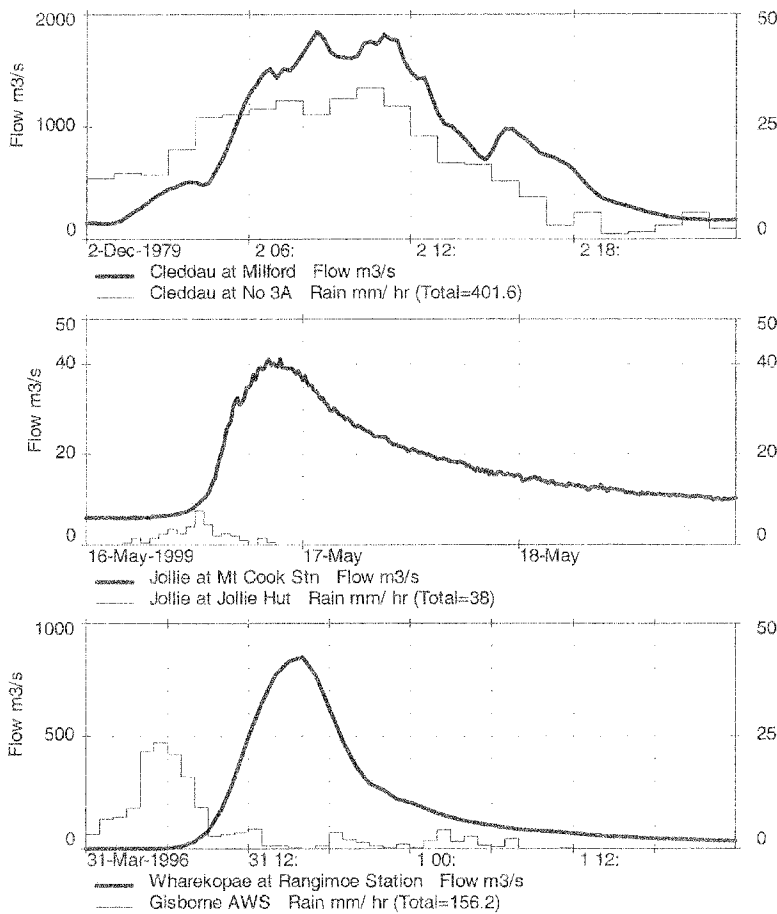


Figure 9.4 Typical flow hydrographs for the catchments shown in Figure 9.2

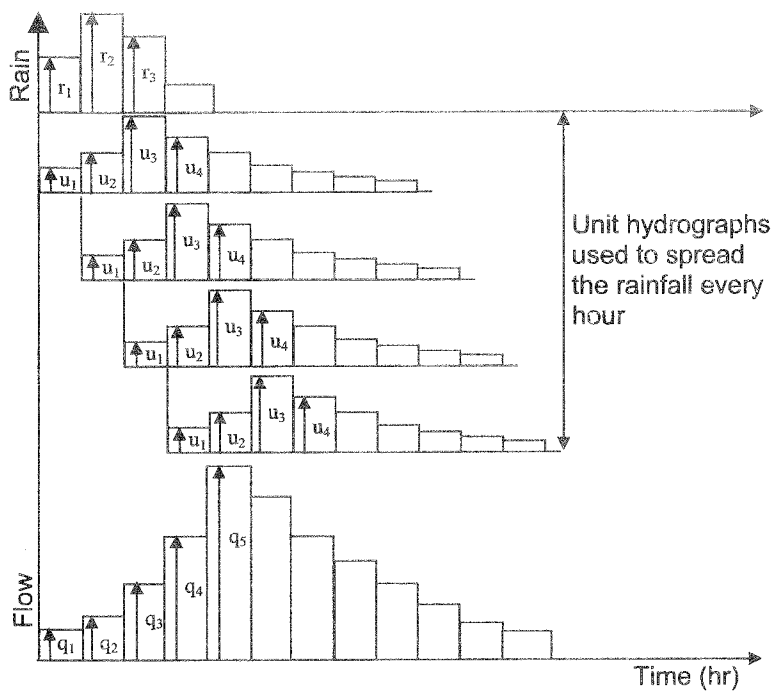


Figure 9.5 Hydrograph convolution – r_n = the rainfall in the n^{th} hour, u_m = the m^{th} unit hydrograph value, and q_n = the flow in the n^{th} hour. $q_1 = r_1 u_1$, $q_2 = r_2 u_1 + r_1 u_2$, $q_3 = r_3 u_1 + r_2 u_2 + r_1 u_3$, $q_4 = r_4 u_1 + r_3 u_2 + r_2 u_3 + r_1 u_4$, etc.

similar storms, different amounts of runoff to the stream network can occur because a catchment can be in different states of wetness/dryness before each rainfall event.

It is first necessary to determine how much of the flow past a catchment outlet belongs to the current rainfall event, and how much belongs to previous events. In all but the smallest streams, water is flowing before a flood even begins. This pre-event flow is called “base flow” and is caused by drainage of water from past storms that has soaked into the ground to become part of groundwater storage. This water is gradually released over long time periods compared to the average length of a storm, and is important because it maintains the ecological health of a river between rainfall events. Since base flow cannot be attributed to a single rainfall event, ways must be found to remove its effect from measured hydrographs so as to derive the storm runoff attributable to a particular rainfall event. More art than science was used in what has become known as “base flow separation”, until Hewlett and Hibbert (1967) produced an empirical but objective method.

Despite more than 40 years of work by many hydrologists, there is still no definitive method for base flow separation that is physically defensible in anything but general terms. This is because it is extremely difficult to make catchment-wide observations of how groundwater storage is behaving. Fortunately, for the purposes of the unit hydrograph, base flow in New Zealand is often small enough compared to the flow at the peak of a flood that

errors in its derivation are unimportant. (An exception to this is rivers at the outlets of lakes, where the storage in the lake can mean that the base flow is a relatively large proportion of the flood flows.) However, this does not mean that estimation of base flow is unimportant, for as already indicated, base flow is particularly important for the sustainability of river ecosystems.

Having separated the base flow from the total hydrograph to leave the quick flow hydrograph, the next problem is to separate out the effect of variation between storm rainfalls. This problem has two components. First the “correct” rainfall to use in deriving the flood hydrograph at the catchment outlet has to be found. Much of the rain that reaches the ground soaks in, and to a hydrologist concerned with floods, the important quantity is that fraction of the rainfall that does not soak in but quickly makes its way to the nearest section of the stream network. Derivation of the “effective” rainfall, i.e., that which does not soak in, was helped by the advent of early hypotheses of water infiltration into the

ground and the fact that the amount of water passing a catchment outlet during a flood, must, by definition, match the amount of effective rainfall. When rain starts falling onto dry ground, the initial drops tend to seep into the ground to fill available spaces in the ground. The water used for filling the spaces is referred to as the “initial” loss. Once the initial loss has occurred the rate of entry of water into the ground surface reduces, because now each drop that enters the ground must effectively wait for one to leave. The ongoing loss of rainfall into the ground is referred to as the “continuing” loss. Both the initial loss and the continuing loss must be estimated and subtracted from the measured rainfall to derive the effective rainfall to be used in estimating the correct amount of surface runoff that becomes streamflow. The second task is to eliminate the effect of variations between storm rainfalls. A mathematical procedure called “convolution” is used to do this. Convolution is the process by which the effective rainfall is spread through time to simulate the different travel times of flow from different distances along the channel network. Figure 9.5 shows the process of convoluting a unit hydrograph with a series of hourly rainfalls. Convolution takes a particular hour of rainfall and divides it up into those parts that will arrive at the outlet in the next hour, the hour after that, and so on. The process then advances one hour and repeats the process. The final step is to add together the contributions from all the hourly rainfalls that arrive at the outlet during the same

hour. Convolution is used every time a flood hydrograph is constructed from a temporally varying rainfall using the unit hydrograph method.

MODERN CATCHMENT MODELS

The development of the unit hydrograph was a breakthrough for calculating the development and progress of flood flows at a fixed point in a catchment. Today, estimates of flow are required at multiple points inside a catchment, and over the full range of flow conditions possible within a catchment. No longer are floods the only reason for calculating river flow, although they are still an important reason. With the growth in population, industrial development and environmental awareness, increasing pressure has been placed on river resources to not only provide water for use by society and absorb its waste products, but also to preserve the natural appearance and ecology of the stream network. The picture is further complicated by the ability of society to have an impact on the amount of water reaching the stream network through changes in land use and even changes to the climate. And last but not least, the increasing pressure on water resources requires that they be managed efficiently, which means that new projects must be accurately planned and designed and fine-tuned for efficient everyday operation. So yesterday's techniques need to be supplemented by ones that are spatially-distributed, multi-purpose, flexible and capable of being used for both planning and operational purposes.

To achieve all the above aims has required a greater understanding of the mechanisms by which rainfall is transformed into stream flow. This in turn has required better mathematical descriptions of the transformation processes, and better physical understanding of catchment processes. With few exceptions, the elegant simplicity of the unit hydrograph approach has been replaced by more complex mathematical procedures that have to be applied to every sub-catchment contributing flow to the stream network. Computers have freed hydrologists to use their imaginations as to how best to include all the different facets of water movement observed in a catchment. The result of combining the power of the computer with human imagination is a set of computer instructions that express the hydrologists' ideas about water movement in a catchment. The set of computer instructions is referred to as a "model", sometimes preceded by a word or phrase such as "catchment", "hydrological", or "rainfall-to-runoff".

MODEL COMPONENTS

Like any model, those discussed here are made up of components. These come in two basic categories:

1. the "physical" equations used to transform rainfall into

streamflow and that are usually associated with components of the hydrological cycle, e.g., the initial interception of rainfall by vegetation.

2. the spatial components, i.e. the sub-catchment models and channel models, that combine to represent the spatial framework of a catchment.

The members of each group of components under item 2 are basically similar to one other, but are adjusted for such things as different sub-catchment areas. The challenge in their use lies in connecting them together in the correct sequence. This is a relatively simple and mechanical process and techniques for doing it follow standard GIS processing techniques. The components under item 1 provide more of a challenge. This is because there is, as yet, no standard theory for hydrology, and so every hydrologist has their own view as to exactly what happens, and what is important, to a model's objectives. Increasingly, component models are being improved as field observations lead to better mathematical descriptions.

One of the simplifications that can be introduced into catchment models is based on the time step used to step through calculations in a model. If hydrological answers are needed for a time step of a year, then an elaborate description of short duration processes, such as the routing of a flood down a channel network, can be ignored. Conversely, if a model is to be used for "real" time forecasting, then accurate descriptions of rapidly changing processes will be needed and small time steps should be used to ensure that the precision of the calculations is maintained. The issue of what time step to use is referred to as the temporal scaling problem of models. While a complicated model run for many small time steps should give a result that is similar to that from the same model run for fewer longer time steps, this can seldom be guaranteed. The difference between the results of this type of experiment can be taken as a measure of model error and can be used to assess the reliability of a model simulation.

The temporal scaling problem arises because, with a few exceptions, the components of a catchment model are based on incomplete descriptions and inexact measurements, and so the component model's mathematical description lacks the generality to span different time frames. Until unified and generally accepted component descriptions are available, the best advice that can be given to model users is to use the smallest time step that is economically practical, and assess the probable consequence of temporal scaling problems by making some model runs at different time steps.

No mention has been made of the principles available to help hydrologists design model components. Driven by incomplete descriptions of the processes that transform rainfall into flow, hydrologists have tended to adopt the

KISS principle (keep it simple stupid!). Where a component description can be based on a full range of physical principles, it should be. Generally, however, the only principle that can be applied is that of conservation of mass. Termed “achieving a water balance” by hydrologists, this principle basically states that all the water entering a model component must be accounted for exactly by outflows and changes of storage within the component. Any model or model component that fails this principle is seriously flawed and should not be used.

After the water balance principle, there are few others that can be generally applied—e.g., conservation of momentum, force balance, conservation of energy—simply because measurements to justify components based on these principles cannot yet be made, and/or there are no ways to adjust those measurements that can be made, often in a laboratory, to apply at a different spatial scale. The way hydrologists have tried to get around this problem is to use the power of the computer to sub-divide the spatial description of a catchment into small pieces to which components based on small-scale measurements can be more appropriately applied. However, there still remains a gulf between the smallest scale at which models can practically operate, e.g., 50 m, and that at which laboratory and many field measurements are made, e.g., 0.05 m. Bridging this gulf requires innovative methods, e.g., Franchini *et al.* (1996), Saulnier *et al.* (1997), Ibbitt and Woods (2002), or exploitation of raw computing power, e.g., Ewen *et al.* (2000) and Ewen *et al.* (2002). While the former type of solution is often particular to one model, e.g., Topmodel (Beven and Kirkby 1979), the latter are currently restricted by computing power and will ultimately be limited by the supply of appropriate input data, i.e., even if models are run at a spatial scale of 0.05 m, providing digital elevation data at this scale is unlikely to be possible for many decades.

GOOD MODELLING PRACTICE

In summary, good modelling practice requires models that:

- 1) use all the available measured information;
- 2) can be applied to a wide range of hydrological conditions;
- 3) are flexible in their application; and
- 4) whose fit to measured data (whether the fitting is done using calibration or by some other means, e.g., assignment of values from a look-up table) has been fully assessed using data not used in any fitting process, and qualitatively, using criteria of reasonableness.

The modelling flexibility that is now possible has led to a number of models that offer similar output options. The main differences are in the amount of input required to make the models run, and the physical rigour of compu-

tations inside the model. Since no catchment model is perfect, and many of them are far from perfect, it is vital that the results of their use be perceived as having been produced under circumstances most likely to lead to the correct conclusions. There are two basic parts to building faith in model results.

Part one consists of being able to check that the operation of a model is consistent with independent measurements and with qualitative perceptions about the physical reasonableness of component operation. For example, if the peak of a flood is simulated by a rapid rise in groundwater with no increase in surface runoff, most hydrologists would question the validity of the simulation, since their experience would tell them that, except in unusual circumstances, flood flows are derived from water running off the surface of the ground, not percolating along small subterranean paths.

Part two compares a model's general performance with that of other models. This is necessarily more subjective than the part one tests. It considers such things as how much information about the catchment a model is able to use, and a model's capability to simulate the full range of hydrological phenomena likely to be encountered in a particular situation. For example, in areas where snow is an important contributor to catchment runoff, a model without a snow component is less likely to give correct results than one with such a component. Besides having components for the more important hydrological processes operating in a catchment, a model should be able to simulate a wide range of catchment sizes and time steps, for then its results can be compared to other models. While it is difficult to be categorical about the part two tests a model should be subject to, it is a good idea to keep an open mind when assessing the results from a model study and to keep asking the questions: “how might the results have been different if another model had been used, and why?”

Much of the uncertainty arising from the use of a catchment model comes from assigning values to the numbers that make a general model specific to a particular catchment. In modelling parlance, the numbers that make a general model specific to a particular catchment are called parameters. They cover such things as the maximum amount of water that can be stored by the soil and the vegetation, or the rate at which water can seep into a saturated soil. These parameters are often guesses, but most modellers try to relate their abstractions to physically identifiable entities. Partly because of differences between the definition of a physical process in a catchment and how it is represented in a model, and partly because of the effects of the spatial and temporal scales used, it is frequently necessary to estimate the numerical values of parameters by indirect trial and error methods. For example, while the saturated hydraulic conductivity, K_s ,

of a soil is a perfectly respectable parameter with an exact physical meaning, (it is the velocity with which water enters a saturated soil under a unit pressure gradient) and is measurable at spatial scales of 0.05–1.0 m, the correct value to use at the spatial scale of a one kilometre is problematic. This occurs because K_s can vary greatly over short distances, due, for example, to large cracks that can develop in some soils under dry conditions. K_s values at a sub-catchment scale can be further removed from laboratory measurements because of the impacts of plant roots and their seasonal growth and decay.

So how are these problems overcome and what does their solution mean for model results? The traditional way to overcome the potential lack of representativeness of measured parameter values, and to estimate those for which measured values may not even be available, is to guess a set of values and use them in the model to see what sort of flows the corresponding rainfall sequence is transformed into. These flows are then compared to measured flows and new values for the parameters are assigned. While this process can be done randomly, there is a wealth of literature on how it can be done systematically and objectively (Wilde and Beightler 1967; Kuczera 1983a,b). The process of calibrating a model is illustrated in Figure 9.6. With the flexibilities offered by today's models, it is almost always possible to explain at least 70 to 80% of the natural variation in the data used for the calibration. This result, if quoted as the sole indicator of model quality is almost meaningless. A better indicator of model performance is the amount of the natural variation explained for a sequence of data not used in the calibration process. The process of using new data to check on model performance is called split-sample

validation. This sort of test should always be done in any model study, and while it is regarded in some quarters (e.g., Klemes 1983) as a relatively weak test of model performance, it is usually relatively easy to carry out since it uses the same types of data as were used to calibrate the model. Split-sample testing can be enhanced if the data used for validation contain a wider range of values than those used for the calibration, as this gives an indication of a model's ability to deal with circumstances outside its calibration range.

It is not unusual for a model calibrated to just a single flow record to behave less well in a validation test. This is because the model has many "degrees of freedom" and it is possible for an error in the estimate of one parameter to be compensated for by an error in another parameter. With new data, the compensation may be less satisfactory. To reduce problems caused by large degrees of freedom, models should be fitted to more than one sequence of data. At its weakest, the model might be fitted to flow records at both the catchment outlet and to some other point on the stream network, for which flows are measured. The success of this approach depends to some extent on the relative sizes of the catchments upstream of both measurement points. If the areas are very different, it is likely that the flows from the larger catchment will dominate the result and only a small improvement in the validation results will occur. The more measured flow records that can be put into this process the better will be the validation results.

An alternative way to use flow data from ancillary sites, particularly if these are of poor quality (i.e., containing periods of missing data or known errors at high flows), is to generate validation results from the model for these sites and then visually compare the model results with the

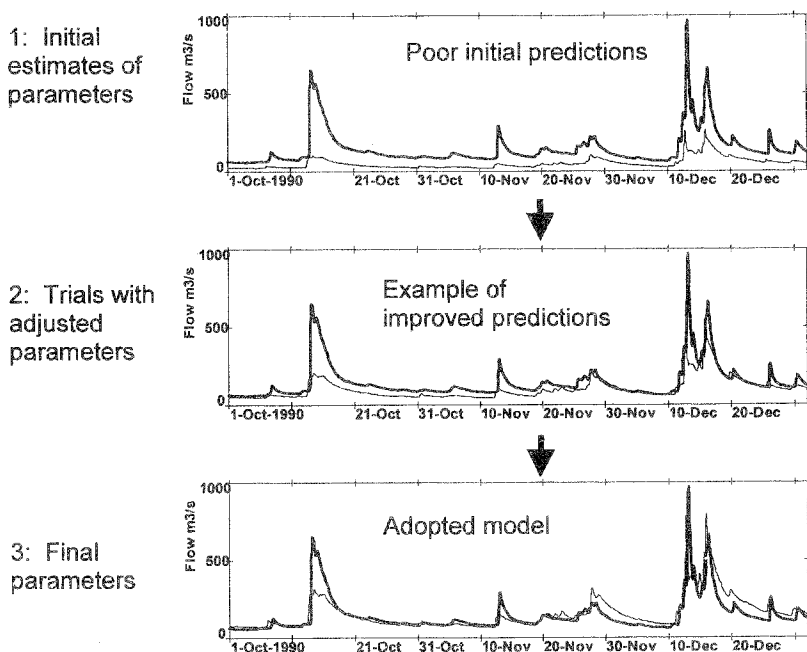


Figure 9.6 Example of calibrating a model to three months of flow data for the Waimakariri River near Christchurch. The heavy line is the recorded flow; the light line is the model prediction.

available measurements. One would hope to see a temporal pattern similar to that measured, with modelled flows showing little bias, i.e., consistent under- or over-estimation.

A stronger test of model reliability is to reproduce a measured temporal sequence of some internal state or flux variable in the catchment. For example, in some catchments soil moisture and/or near-surface water table position are routinely measured. Comparison of these measurements with the equivalent variable in the model can give considerable insight into the behaviour of a model.

All the tests so far have described comparisons of model output with measurements made in the same catchment. Such tests have the potential to hide model bias caused by factors that affect a whole catchment, for example a systematic over-estimate of catchment rainfall. The strongest form of test is therefore to apply a model calibrated for one catchment to data from a similar catchment. Of course suitable steps have to be taken to adjust for the obvious changes in stream network shape and catchment areas and to use appropriate input data. This type of testing has to be approached with caution, since it is very difficult to find two catchments whose hydrological differences can be quantitatively assessed as “small”. This is because the hydrological properties of the soils in a catchment and its underlying geology are seldom completely known with any certainty. However, where such a test can be applied, it can lead to model improvements, since it highlights potential parts of the model that are insufficiently general.

When the quantitative checks on model performance have been exhausted, it often pays to assess the internal states in a model. Application of this test to the Mahurangi catchment north of Auckland has shown that although the model generated credible flows and internal soil moisture states in a small sub-catchment, there were sub-catchments for which the water table was frequently well above the ground surface, and soil moisture values were often at a maximum in summer, when they would have been expected to be approaching their minima. The problem in this case was traced to a spatial scale used and led to a technique aimed at eliminating this type of difficulty (Ibbitt and Woods 2004).

The problem with assigning numerical values to parameters has been discussed in terms of their indirect estimation from flow data. An alternative approach to minimising the effort needed to calibrate a model is to provide look-up tables, developed from a nation-wide analysis, for the more important parameters in the models. Through the GIS tools used to build the structure of each model, the areas of various soil and vegetation types in each sub-catchment are first identified. Look-up tables can then be used to assign a numerical value to each hydrological property associated with a particular soil or veg-

etation type. Finally the various estimates can be combined into a weighted sum, using the amount of area of each type as the weights. While this approach has been found to give satisfactory results, it frequently benefits from the use of calibration techniques to refine the values assigned from the look-up tables. Effectively, the look-up tables provide initial values that are approximately correct. Where look-up tables are unavailable, measured values for parameters can be used in one of two ways. The measurements can be used to assign initial values, or final calibrated values can be compared with the measurements to see what level of agreement has been achieved.

EXAMPLE OF A MODERN MODEL

The TOPNET modelling system used by NIWA is typical of many spatially-distributed, physically-based modelling systems being developed around the world. Like most spatially-distributed modelling systems, it relies heavily on GIS-based techniques to process the map data that provide information on ground elevations, channel locations, and the spatial distribution of soil and vegetation types.

Construction of a TOPNET model begins with digital elevation data for the region being modelled. In New Zealand the size of the cells (pixels) that make up the national digital elevation model (DEM) is 30×30 m. Where possible, DEM data at this resolution is used to build TOPNET models. For large catchments, such as the Clutha, use of the DEM data at a pixel resolution of 30×30 m leads to very large and unwieldy computer files and so the DEM grid has been re-gridded to a 150-m resolution. The elevation data in the DEM has been modified by “burning in” the blue line channel network coverage from the 1:50,000 NZ Map Series (Terralink 1997) to ensure that in the next stage of the DEM processing, the correct stream network is used to delineate sub-catchments.

From the coordinates of a catchment outlet, flow paths are followed upstream until no further upslope points can be found. These “end” points are on the catchment boundary or the boundary of some internal sub-catchment. All such points are stored to help with the process of sub-catchment definition. During the upslope processing, various pieces of useful information are extracted. These include the upslope area that flows through each DEM pixel and the local slope between two pixels. Depending on the instructions given to the GIS processing procedure, sub-catchments can be defined at various levels of average size. Typically, sub-catchments in the range $1\text{--}5$ km² are aimed for, but this can be varied. Once the sub-catchments have been defined, the DEM data for each sub-catchment are used to compute the distribution of a quantity called the “topographic index”.

The topographic index is based on the local slope between two DEM pixels and the area potentially contributing surface flow to those pixels. The index shows how prone a pixel is to saturation. Pixels with large upslope drainage areas and flat gradients get high index values, signifying they are likely to be wet areas, while pixels on catchment boundaries have small contributing areas, and often steep slopes, and have small index values, signifying likely dry areas. During the processing of the DEM, the slopes for the channel segments are extracted, along with the areas potentially contributing flow to that segment. This area is used as a substitute variable for flow in the downstream hydraulic relationships (discussed in Chapter 8), to derive approximate channel dimensions for routing flow down the stream network.

The TOPNET system models a catchment as a collection of sub-catchments, linked by a branched river network (Fig. 9.7). Flow is routed through the river network using kinematic waves and the shock-fitting technique of Goring (1994).

Once processing of the digital elevation model is complete, the sub-catchment areas are overlain on soil-type and land cover maps, and the amounts of different soil types and vegetation types in each sub-catchment are extracted for use with look-up tables that define reasonable values for the rainfall interception and soil moisture properties for each vegetation and soil type. An area-weighted sum of these properties is then formed for each sub-catchment.

The GIS processing also examines information provided on the locations of rain gauges across the catchment and derives weights that define how much of each gauge's catch is likely to fall on each sub-catchment. If required, and where the rain gauge network is sparse, an additional weighting function can be used to adjust rain gauge totals to be consistent with the long-term pattern of rainfall over

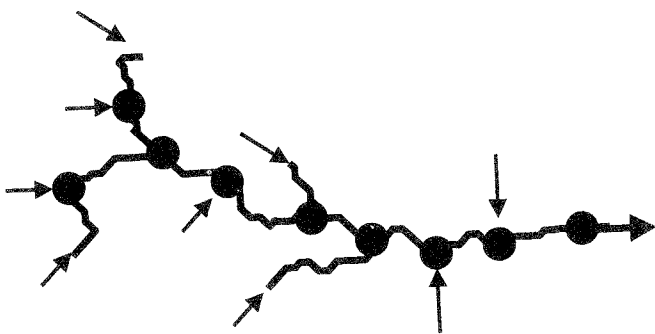


Figure 9.7 Detail of the river network routing structure. Arrows indicate inflows to the river network from modelled sub-catchments, and filled circles indicate junctions in the river network.

a sub-catchment. This is particularly useful where measured data with fine time resolution are available only for the lower parts of a catchment, and yet it is known from long-term storage rain gauges that the rainfall is higher in the mountains.

The output from the GIS processing is a model specification file with information on the spatial characteristics of the catchment, including the correct interconnections between sub-catchments and the stream network, and between channels segments within the stream network, and a sub-catchment summary file that provides information on the mean elevation of each sub-catchment for use in the snow-melt component of TOPNET models. When creating the model specification file, the GIS process cannot assign data-derived values for all the model parameters. For such parameters, sensible values are assigned, in the expectation that the user will refine these later using a calibration procedure such as that of Kuczera (1983a, b).

The information put into the model specification file is used to make the TOPNET model code specific to a sub-catchment. Each sub-catchment is modelled (Fig. 9.8) using an adaptation of Topmodel (Beven and Kirkby 1979). The TOPNET model code itself provides component models for all the main processes identified as part of the hydrological cycle:

- rainfall interception by vegetation and its subsequent evaporation from plant surfaces;
- calculation of snow accumulation or melt, if required;
- infiltration into unsaturated ground;
- infiltration excess runoff where rainfall rates exceed the ground's ability to absorb it;
- overland flow routing to the nearest stream channel;
- infiltration to, and drainage and plant transpiration from, unsaturated soil moisture;
- seepage to groundwater and calculation of the level of the water table;

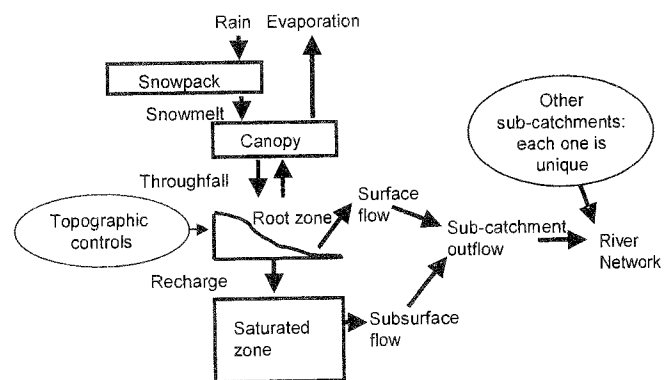


Figure 9.8 Topnet model structure, showing the snowpack, canopy, root zone, saturated zone and river network components of the model. Arrows indicate flows of water from one model component to another.

- estimation of the fraction of the catchment that is saturated, and hence the amount of the rainfall that cannot infiltrate and so becomes surface runoff;
- calculation of the drainage from groundwater into the stream channels; and
- complete routing of all flows into the stream network and then to the catchment outlet.

Topmodel assumes that available soil water storage can vary within a sub-catchment because of topographic effects—valley bottoms and flat places are wetter than ridges. It uses the topographic index to measure the propensity for soil wetness at each location in a sub-catchment. The actual amount of soil water storage depends on the level of storage in the saturated zone (which varies with time) as well as with the topographic index. Each sub-catchment model assumes that vegetation and soil characteristics are uniform within the sub-catchment.

By modelling the whole hydrological cycle, the whole hydrograph is simulated using measured rainfall rather than some “effective” rainfall, and problems such as differing degrees of wetness before a storm and base-flow separation are automatically taken into account. Furthermore, TOPNET models are designed for continuous simulation of catchment water balance and river flow, and can provide flow predictions for many locations in a catchment. They are well suited to operational flood forecasting (Ibbitt *et al.* 2000), but can also be used to simulate the potential effects of changes in vegetation and climate, *i.e.*, the system is multi-purpose.

The model uses continuous sequences of rainfall and temperature data (*e.g.*, at hourly intervals) from one or more locations. TOPNET models have been used with rain input from weather radar, measured data from rain gauges, and rainfall forecasts from atmospheric models. The output from TOPNET models, while voluminous, allows the moisture fluxes and levels within any subcatchment to be examined and to be compared between subcatchments. Comparisons can be made with any available measurements.

The hydrographs in Figure 9.9 show modelling results for the 2000 km² Motueka catchment near Nelson. Information was assembled from the following sources:

- the Climate Database (Penney 2001)
- the Land Cover Database (Terralink 1997)
- the River Environment Classification (Snelder and Biggs 2002)
- the Land Resources Inventory and National Soils Database (NWASCO 1979)

From these sources, several hundred sub-catchments were defined and their physical properties estimated, and then daily time series of rain and temperature were used to run the TOPNET model for more than 15 years. Three of the model parameters were calibrated to improve the

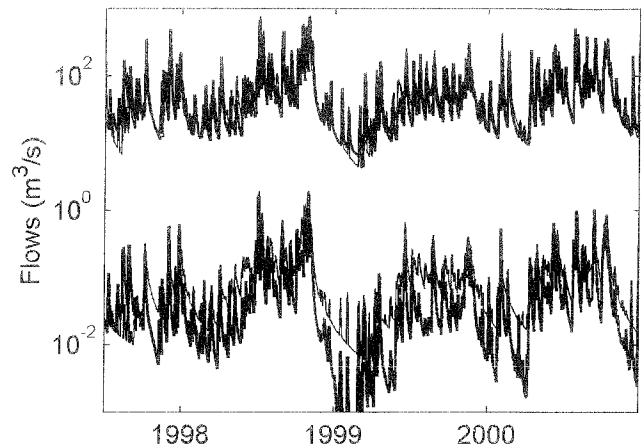


Figure 9.9 Measured and simulated river flows for two contrasting locations in the Motueka River catchment. The upper lines show flows for Motueka at Woodstock (1750 km²), and the lower lines for Hunters Gully at Weir (5 km²) (thick line is measurement, thin line is model). Note logarithmic vertical scale.

match to flow data for one year of data from all available river flow recording sites in the Tasman region. The simulations in Figure 9.9 are validation tests for years that were not used for the model calibration. For the region as a whole, the simulations are very useful for catchments of 100 km² and larger, and are also useful indicators of annual and seasonal water balance at smaller scales. More research is underway to understand the complex low flow behaviour in smaller catchments, which, as the hydrograph for Hunters Gully shows, is at present not always well reproduced by the TOPNET model.

USE OF MODELS

For sustainable and efficient use of New Zealand’s water resources, tools are needed that allow accurate and reliable estimates to be made of the consequences of society’s use of the available resource. Catchment modelling is one such tool. Initially, models were for a specific purpose, simply because the use of multi-purpose models was computationally impractical. To a large extent modern computers have removed the computational limitations, exposing weaknesses arising from differing human perceptions of the hydrological cycle, and an inability to measure the important properties of hydrological processes. It is therefore important that the results of a model study are demonstrated to be reliable. A number of objective ways to do this are available but they have yet to be fully appreciated by those seeking to use the results of model studies. An important “soft” measure of modelling credibility is to be able to show the use of “best” practice, *i.e.*, the use of a widely accepted model that uses all the available information and that produces outputs that show

that appropriate steps have been followed in the model's application. To meet these requirements the users of catchment models still need insight into both the hydrological cycle and a particular model's representation of it.

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Chapter 8

Rivers and the riverscape

Paul Mosley

INTRODUCTION

Rivers and streams occupy only a very small proportion of New Zealand's land area, but they are significant and fascinating elements of the landscape. They present many scientific questions, they are important natural resources that demand well-informed and wise management, and they provide the habitat for a biologically diverse component of New Zealand's flora and fauna.

Qualitative approaches to describing and explaining rivers—principal tasks of the science of fluvial geomorphology—are well developed, and are based on historical geology and geomorphology (Cotton 1958; Schumm 1972, 1977; Gage 1980). In a geologically young and active landscape like that of New Zealand, such approaches make an essential contribution to our understanding of the form and behaviour of rivers, and to our ability to manage them wisely. A strongly quantitative approach to fluvial geomorphology also has been established. This is based on the morphometric work of American scientists such as Strahler (1952), the hydraulic geometry developed by the United States Geological Survey (Leopold *et al.* 1964), and regime theory (an empirical rather than theoretical methodology of open-channel design) popularised by British engineers (Blench 1957). Measurements provide the ability to develop and objectively test quantitative relationships between the form and behaviour of rivers and the factors that supposedly control them. Very importantly for the resource manager, they make it easier to predict changes in river form as a result of changes in the wider environment, including the effects of human activity.

Study and management of rivers also draw on the sciences of river mechanics and hydraulics (Henderson 1966; Raudkivi 1998). Increasingly, as computers become more powerful, two- and three-dimensional hydraulic modelling is being applied to river channels, to provide an

understanding of river form and behaviour that is based on physical principles, rather than the statistical relationships that are the product of hydraulic geometry and other empirical approaches.

Finally, we should recognise the role of the landscape analysis and landscape architecture approaches to the analysis and management of rivers. Many aspects of a river and its surroundings—the river corridor or “riverscape”—that are mentioned in New Zealand's resource management legislation are visual, and are perceived in a more or less subjective way by onlookers. Appraisal of such aspects, in terms of concepts such as legibility (how obviously the landscape expresses the processes that formed it), is a matter of experience and judgement. There are ways, however, of quantifying people's perception of rivers (Mosley 1989). Nowadays, landscape architects play a considerable role in decisions that relate to proposals that could alter the character and appearance of rivers.

This chapter considers rivers from these several perspectives, but with an emphasis on the types of information that are required to manage rivers in the context of the Resource Management Act. It reviews the ways in which we might characterise or describe the characteristics of a river, the relationship between the nature of a river and the factors that control it, and the ways in which we might classify rivers as a means of organising our knowledge and understanding.

RIVER CHARACTERISATION

Scientific study and wise management of rivers both require that we are able to measure and describe their relevant characteristics, as a step towards understanding river form and process and the effects thereon of human activity. In the past, many different approaches to river characterisation have been developed (Mosley 1987). In

New Zealand, the Resource Management Act has required resource managers to develop new ways of considering rivers, and therefore of characterising them.

River characterisation: hydraulics of flow

This section provides a very brief introduction to the ways in which a river or stream channel can be described, in terms of how water flows along it.

Natural channels usually exhibit *gradually varied flow*. In such flow, the streamlines (the paths taken by packets of water travelling downstream) are not parallel, because the shape and dimensions of the channel cross-section change downstream (Fig. 8.1). Accordingly, the depth and velocity of the water change along the reach, and divergence or convergence of the streamlines produces centrifugal forces, so the pressure distribution in the water column is not hydrostatic (i.e., it is not solely dependent on depth below the surface). In small streams and torrents, *rapidly varied flow* is common, where streamline curvature is extreme, for example around boulders or down cascades and rapids. The other class of flow—*uniform flow*, in which the bed, water surface, all streamlines, and the energy gradeline are all parallel—is rarely found in nature, except over short distances.

Natural rivers and streams normally have *unsteady flow*, in which discharge (Q), depth (d), cross-sectional area (A) and flow velocity (v) vary with time at a given cross-section. *Steady flow* may be found in an artificial canal into which a constant discharge is introduced; in steady flow, the continuity equation of flow applies:

$$Q = A_1 v_1 = A_2 v_2 = \dots = A_n v_n \quad 8.1$$

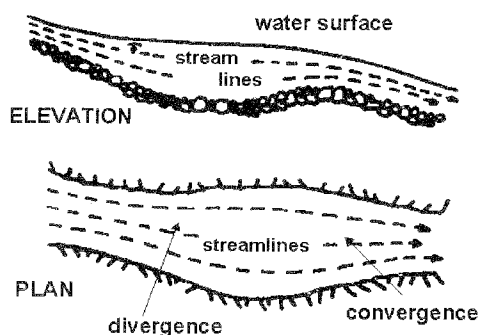


Figure 8.1 Diagram of non-uniform flow in a stream channel.

that is, discharge is constant, and is the same at all cross-sections 1, 2...n along a channel. With unsteady flow, the passage of a flood wave would mean that the discharge at different cross-sections along the channel is not precisely the same. Discharge normally changes sufficiently slowly that, for many practical purposes, the continuity equation can be taken to apply.

Natural rivers and streams invariably have *turbulent flow*, in which inertial forces predominate over viscous forces, and water velocity (speed and direction) at any point is continually changing as random eddies pass. In contrast to turbulent flow, is *laminar flow*, in which streamlines are smooth and linear, demonstrated, for example, when dye is injected with a syringe into a sheet of water flowing smoothly over plate glass. The Reynolds number Re is used to distinguish between turbulent and laminar flow:

$$Re = \rho v d / \mu = v d / \nu \quad 8.2$$

in which $\rho v d$ is the inertial force of the flowing water, ρ is the density of water, μ is dynamic viscosity¹, and ν is kinematic viscosity². Turbulent flow is characterised by $Re > 2000$; in laminar flow, normally $Re < 500$. Flow turbulence is important—it is responsible for entraining sediment from the bed, maintaining sediment in suspension, and creating continual fluctuations in water velocity that are significant to fish and other aquatic biota.

Normally, flow in natural channels is *sub-critical or tranquil*, except in steep, bouldery channels with falls and cascades, where the flow can be *super-critical*. The dimensionless number that is used to identify whether flow is sub-critical or super-critical is the Froude number Fr , which relates the inertia of a unit mass of streamflow to the celerity of a shallow wave:

$$Fr = v / (gd)^{0.5} \quad 8.3$$

in which g is the acceleration of gravity. When $Fr < 1$ (flow is sub-critical), ripples on the water surface can travel upstream; when $Fr > 1$, ripples are swept downstream. Critical flow, when $Fr = 1$, occurs in natural channels in restricted locations, such as at and downstream of steep drops (Fig. 8.2). In Figure 8.2, water drops over a steep fall caused by a group of boulders. Upstream, water surface slope, depth and velocity are such that velocity is less than $(gd)^{0.5}$; depth is said to be less than the critical depth. Downstream of the boulders there is a steep drop, such that velocity exceeds $(gd)^{0.5}$, and the depth is greater than the critical depth. Where the water passes over the

¹ Dynamic viscosity is a measure of the force per unit area (stress) required to maintain a unit difference of velocity (strain rate) between two parallel layers separated by a unit distance.

² Kinematic viscosity is a measure of the interference between adjacent layers of fluid.

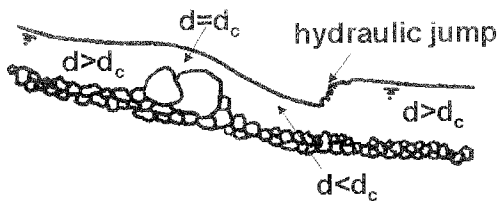


Figure 8.2 Diagram of sub-critical and super-critical flow in a stream channel.

boulders, $v = (gd)^{0.5}$, and depth is critical. Further downstream, where the stream bed and water surface slope become gentler again, sub-critical flow is re-established, and a “hydraulic jump”—a stationary zone of extreme turbulence—forms. Hydraulic phenomena such as hydraulic jumps and the effects of changing channel slope and cross-section are analysed in detail in texts such as Henderson (1966), using the energy equation and the momentum equation.

At any point n along a channel, the total energy or head H_n of the water is the sum of potential energy due to the elevation z_n of the bed above base level, water depth d_n and kinetic energy $v_n^2/2g$, that is:

$$H_n = z_n + d_n + v_n^2/2g \quad 8.4$$

If we consider two cross-sections in a channel reach with gradually varied flow and constant discharge, the total head at the downstream section d equals the total head at the upstream section u , less the head “loss” H_L (i.e., the conversion of the water’s potential energy into heat) in the reach between them (Fig. 8.3):

$$z_d + d_d + v_d^2/2g = z_u + d_u + v_u^2/2g - H_L \quad 8.5$$

The stream bed gradient, the water surface slope, and the energy gradient are not necessarily the same along a channel reach (Fig. 8.3): the stream bed and even the water surface can rise in the downstream direction, but the energy gradient always falls downstream.

Energy in moving water is continually dissipated, as the water overcomes resistance to flow, or friction. Three well-known formulae have been developed to characterise the relationship between resistance to flow and the velocity of the water, in terms of three roughness coefficients: Chezy C , Darcy-Weisbach f , and Manning n :

$$v = C(RS)^{0.5} \quad 8.6$$

$$v = (8gRS/f)^{0.5} \quad 8.7$$

$$v = R^{0.67}S^{0.5}/n \quad 8.8$$

In these equations, S is the energy gradient, and hydraulic radius $R = Q/A = \text{discharge}/\text{cross-section area}$, which is closely approximated by mean depth d in natural

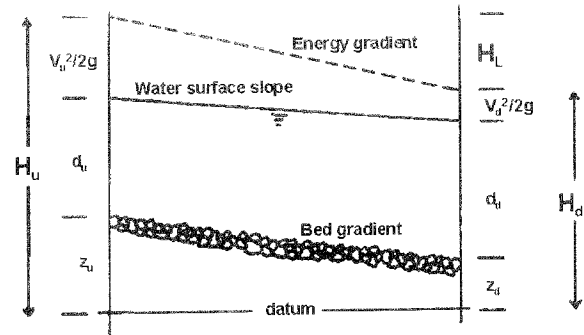


Figure 8.3 Diagram showing gradually varied flow in a stream channel.

channels. Equations 8.6 and 8.7 are soundly based on theory, but the empirically-derived Manning equation (Eq. 8.8) is the most widely used, and values of n have been tabulated for many different types of channel. Typical values for different types of streams and rivers are shown in Table 8.1. Hicks and Mason (1991) have presented the physical and hydraulic characteristics, including Manning n and Chezy C , together with photographs, for rivers throughout New Zealand.

In simple terms, the Manning equation shows that water velocity increases as depth and slope increase, and decreases as resistance to flow increases. Resistance to flow is caused by a number of factors, including channel boundary roughness, bedforms such as dunes or riffles, variations in cross-section shape and dimensions, obstructions such as trees and boulders, in-channel vegetation, and channel sinuosity. For regular, straight channels where the other components of flow resistance are unimportant, the grain size of sediment along the channel boundary is the main determinant of the Manning coefficient:

$$n = 0.015 D_{50}^{1/6} \quad 8.9$$

in which D_{50} is the median diameter of the bed surface material, in millimetres. Hicks and Mason’s data show that in many rivers the roughness coefficients are correlated with discharge: as discharge increases, so do cross-sectional area and depth, and the frictional effect of the channel boundary declines.

Table 8.1 Typical values of Manning n

Type of channel	Manning n
Regular, weed-free river	0.020-0.030
Winding stream, with pools and riffles	0.033-0.040
Very winding and/or weedy stream	0.075-0.15
Bouldery stream	0.050-0.30

River characterisation: the geomorphological perspective

A river can be characterised in terms of a wide range of variables or descriptors, the choice of which depends on the particular purpose at hand. From the perspective of the processes that control river form and function, one might characterise a river in terms of its morphological attributes at several scales (Table 8.2):

- the catchment (also known as the drainage basin or watershed): the hillslopes and drainage channels that contribute runoff to a given point;
- the channel network: the system of streams and rivers that drain from a catchment to a particular point;
- the channel reach: a section of river that is more or less homogeneous, although it may display considerable within-reach variability associated with such repeating features as riffles and pools (Fig. 8.4);
- sub-reach features: riffles, pools and runs;
- the channel cross-section: a “slice” across the river at one place (Fig. 8.5); and
- the river bed at a single point on a cross-section. This scale of measurement is considered more fully in Chapter 43.

Many channel characteristics vary with discharge, and it is necessary to reference measurements to a standard hydrological condition when making comparisons between rivers or between reaches. Bankfull discharge, the flow that just fills the channel to the top of the

banks, commonly has been used by geomorphologists (Fig. 8.5). In many rivers in temperate climates this is approximately equal to the mean annual flood, with a mean recurrence interval of once in 2.33 years, although in New Zealand recurrence intervals of the bankfull discharge vary from less than one year to more than ten years (Mosley 1981). The bankfull channel may be defined in the field by delimiting the intersection of the channel's steep banks with the surface of the floodplain or terrace. Where a distinct floodplain is not present, an equivalent may be identified by the boundary between colonising or short-lived vegetation species in the channel, and long-lived species that have established above the “bankfull” level. The concept of bankfull is not always easy to apply in New Zealand, where rivers often are incising actively into terraces or have bedrock banks.

The channel reach

The characteristics listed for the channel reach in Table 8.2 that relate to river form in plan view (meander wavelength etc.) are best measured from aerial photographs, although for large rivers (> 250 m wide), the 1:50,000 topographic maps are adequate for some purposes. Many of the characteristics, such as bed slope, water surface slope and channel width, vary from place to place along a reach, often in a repetitive way in association with regularly repeating features such as riffles and pools. These must be evaluated using field survey techniques.

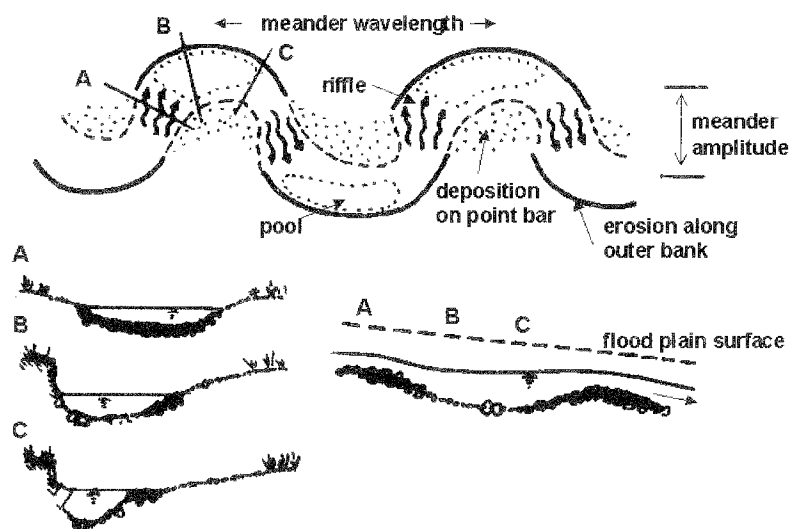


Figure 8.4 Idealised plan geometry, cross-sections and longitudinal section of a meandering river.

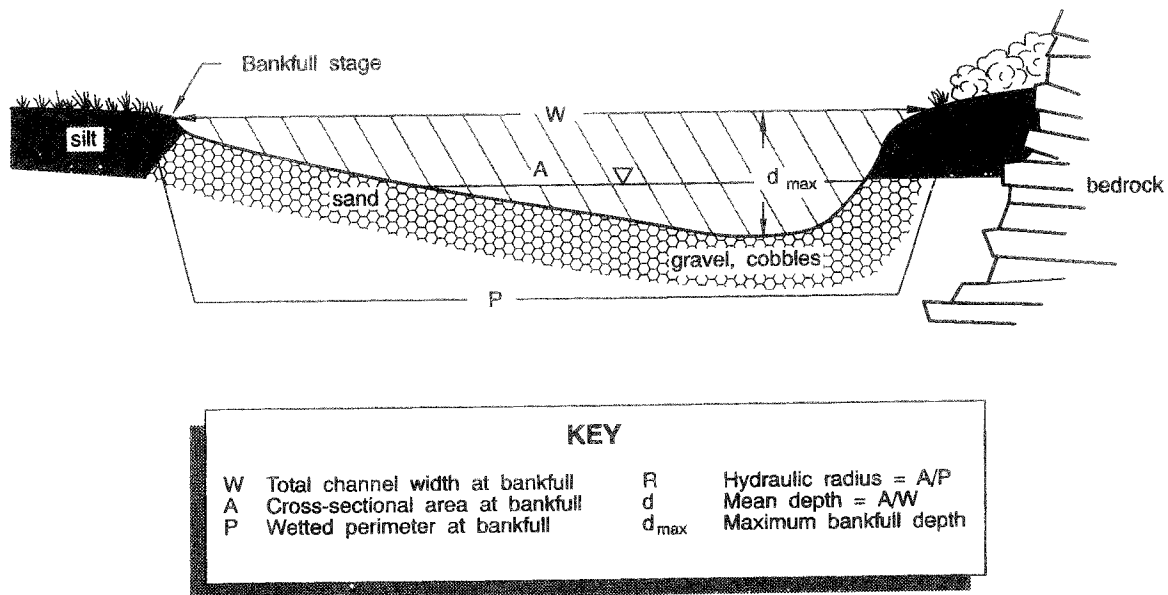


Figure 8.5 Diagram showing channel cross-section definitions.

Table 8.2 Morphometric characteristics of rivers and catchments.

	Cross-section	Reach	Network	Catchment
Length or scale measures	Elevation Width Wetted perimeter Mean depth Hydraulic radius Maximum depth Elevation	Meander wavelength Bend amplitude Radius of curvature Bedform wavelength Bedform amplitude Width wetted perimeter Mean depth Hydraulic radius Maximum depth	Total channel length Drainage density Network diameter Mainstream length	Perimeter Length Outlet elevation Maximum elevation Relief
Area or extent	Channel capacity (cross-sectional area)	Channel capacity (cross-sectional area) Area of riverbed	Stream order, magnitude Number of n th order streams	Area Area in lakes, ponds, wetlands Area under forest cover
Shape	Form ratio (width/mean depth) Depth ratio (maximum depth/mean depth) Roughness (Manning n etc.) Sediment size	Sinuosity Braiding index	Bifurcation ratio	Elongation
Angle or gradient	Energy gradient Water surface slope Traverse water surface slope	Energy gradient Water surface slope		Relief ratio

There are strong statistical relationships between the characteristics of a river reach and the water and sediment that it carries. Thus, for example, meander wavelength WL commonly increases as the 0.5 power of mean annual flood Q_{maf} ; for a sample of South Island rivers, Mosley (1981) found the relationship:

$$WL = 46 Q_{maf}^{0.51} \quad 8.10$$

Channel sinuosity (the ratio of distance between two points along the channel to the straight-line distance) is related to the nature of the sediment load, bank material, energy expenditure in the river, and geologic environment. Such relationships are very useful in reconnaissance studies, for example, to make inferences about channel stability or sediment load from aerial photographs (see below, Fig. 8.14).

Sub-reach features: riffles, pools and runs

One of the most common means of characterising a channel, at a scale intermediate between cross-section and reach, is to describe it in terms of the simple habitat classification of riffle, pool and run (Fig. 8.6):

- Riffle: an area of the riverbed with higher than average water surface slope, Froude number and velocity, and lower than average depth. A riffle is likely to have broken water with breaking waves, and coarse gravel and cobbles on the bed.
- Pool: an area of the riverbed with lower than average (possibly near-zero) water surface slope, Froude number and velocity, and greater than average depth. A pool is likely to have relatively smooth water, with eddies that carry the flow back upstream at the side; the bed sediment often is composed of sand and silt in low-velocity areas, but in the deepest parts there may be a “lag” of cobbles.
- Run: an area of intermediate depth, velocity, and water surface slope, often in a relatively straight section of channel in which flow is moderately uniform. Runs are not always easily distinguished from riffles and pools.

Jowett (1993) found that the velocity/depth ratio best discriminates among riffles, pools, and runs, although he noted that slope is the ultimate determinant of habitat type. Within these broad groupings, there are different types of feature. Often, a riffle takes the form of a bar that stretches diagonally across the channel (“skew shoal”), while in a winding river it is likely to be at the inflection point between bends. Often, pools are located downstream of skew shoal-type riffles, where the flow is directed into the riverbank and scours deeply along the base of the bank, while in a meandering river they are found at the outer bank of bends. In braided and semi-braided rivers, riffles

and pools often are associated with channel confluences—a pool at the confluence, with riffles immediately upstream. In rivers that have bedrock outcrops at intervals, pools are commonly sited at the outcrops, with riffles up and downstream.

Other features that can be recognised at this sub-reach scale include (Fig. 8.6):

- Cascades – steep sections of channel, usually with outcropping bedrock and boulders, with much turbulent water and super-critical flow (Montgomery and Buffington 1997).
- Step-pools – regularly repeating features, akin to riffles, in headwater streams. They are composed of cobbles and boulders that extend right out of the water, creating turbulence, drops, areas of super-critical flow, and intervening pools of deep, relatively slow-flowing water (Montgomery and Buffington 1997).
- Braid bars and channels – a complex variety of bedforms in braided rivers that would require a separate chapter to describe fully (e.g., Ferguson 1993).
- “Microform bed clusters” – aggregations of cobbles and boulders that are identifiably organised structures, and that tend to be arranged in a repeating, although seemingly random, way along the river channel (Biggs *et al.* 1997).

The occurrence of these features is strongly related to channel slope, and to position along the river profile. This is considered later, in the river classification section.

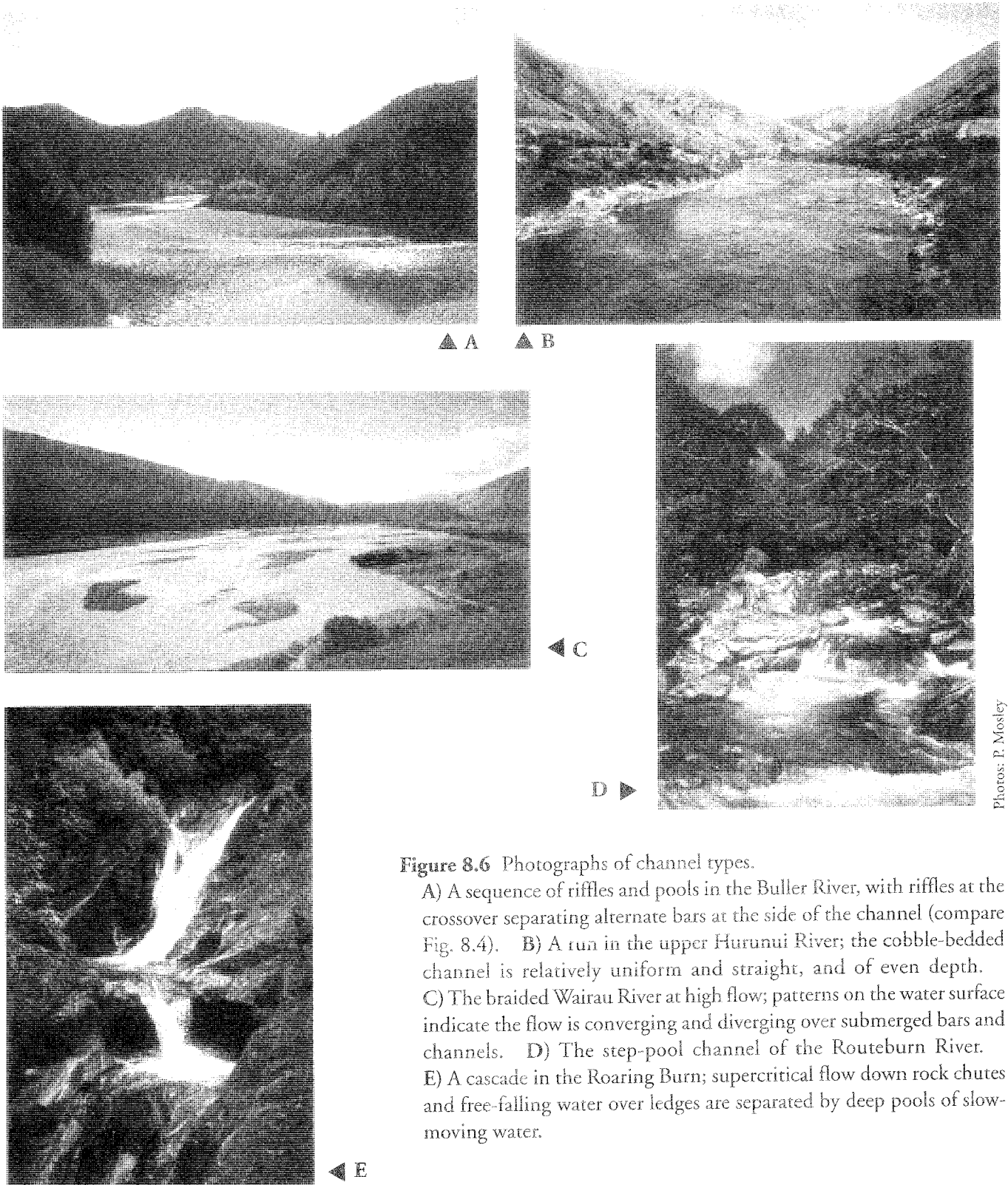
The cross-section

The measurements used to characterise channel cross-sections are based on standard surveying or levelling techniques (Fig. 8.5). Many scientists have used the observations of depth and velocity made across a channel during a stream gauging to define representative cross-sections. Caution is needed when using such data, however, because stream gauging sites almost always are selected at stable sections with flow as uniform as possible, which are likely to be rather unrepresentative of the river as a whole.

Cross-section characteristics strongly reflect the amount of water conveyed by the river, and the type of material—sediment, *in situ* bedrock, and vegetation—composing the channel boundary. The relationships are considered in more detail in the following section on at-a-station hydraulic geometry.

River characterisation for resource management

From a resource management perspective, for example when assessing the environmental effects of a proposal to abstract water from a river for irrigation, one would need



Photos: P. Mosley

Figure 8.6 Photographs of channel types.

A) A sequence of riffles and pools in the Buller River, with riffles at the crossover separating alternate bars at the side of the channel (compare Fig. 8.4). B) A run in the upper Hurunui River; the cobble-bedded channel is relatively uniform and straight, and of even depth. C) The braided Wairau River at high flow; patterns on the water surface indicate the flow is converging and diverging over submerged bars and channels. D) The step-pool channel of the Routeburn River. E) A cascade in the Roaring Burn; supercritical flow down rock chutes and free-falling water over ledges are separated by deep pools of slow-moving water.

to characterise a river in terms of a much wider range of attributes. These could include its overall landscape setting, the floodplain setting, channel character, water character, and scenic and recreational value (Table 8.3). In practice, the choice of characteristics and measurements often will depend on matters that must be considered in meeting the requirements of the Resource Management Act (1991).

Characteristics specified in resource management law

The matters defined in Part II (Purpose and Principles) of the Resource Management Act that may be relevant for managing a river include:

- life supporting capacity of ... water (section 5(2)b),
- natural character of ... rivers and their margins (section 6a),

- outstanding natural features and landscapes (section 6b),
- areas of significant indigenous vegetation and significant habitats of indigenous fauna (section 6c),
- public access to and along ... rivers (section 6d),
- the relationship of Maori and their culture and traditions with ... water ... and other taonga (section 6e),
- amenity values (section 7c),
- intrinsic values of ecosystems (section 7d),
- heritage values (section 7e),
- quality of the environment (section 7f),
- finite characteristics of natural and physical resources (section 7g), and
- habitat of trout and salmon (section 7h).

Where the environmental effects of a proposed resource consent must be assessed, the Fourth Schedule of the Act lists a number of additional matters that must be considered. Part IX (Water Conservation Orders) of the Act extends even further the characteristics of a river that one might consider. The purpose of a water conservation order is to recognise and sustain amenity and intrinsic values of a water body. Section 199 of the Act lists the characteristics that might be protected as those related to habitat for terrestrial or aquatic organisms; fishery; wild, scenic or other natural characteristics; scientific and ecological values; and recreational, historical, spiritual or cultural purposes.

Almost all of the characteristics referred to by the Act and listed in Table 8.3 need to be converted into an attribute that can be measured, or at least described, as far as possible, in an objective way. Mosley (1982) provided some specific recommendations, but appraisal of many of the characteristics requires judgement by experienced landscape architects and other specialists.

Natural character

A particularly important concept introduced, but not defined, in the Resource Management Act is that of “natural character”. Decisions by the Environment Court and other legal authorities have provided some guidance about what is included in natural character, although most of this refers to the coastal environment. Taken together, this case law—which can be extended to the river environment—suggests that natural character includes:

- location, setting and context within the broader environment;
- topography, landforms, and land-forming processes, including specific landform features such as a gorge;
- vegetative ground cover, biota, ecosystems, and ecological processes;
- combination of natural elements, natural patterns, and natural processes;
- resilience to disturbance, potential naturalness, and likely future recovery; and

- presence or absence of built structures and artificial patterns, and their relative dominance with respect to natural elements.

Regional councils have further extended the dimensions of natural character, in their various regional plans for resource management. In summary, Mosley (1999) concluded that “the natural character of a locality might be defined as deriving from those observable elements, attributes, and patterns in the environment that were produced by naturally occurring biological and physical processes, and those processes themselves. It excludes any that were produced by human agency. The elements, attributes, patterns, and processes that are viewed as contributors to natural character, and the importance attached to each, may depend on the observer, and may vary from place to place.”

CONTROLS ON RIVER MORPHOLOGY

Schumm (1969, p. 256) pointed out that “the dimensions, shape, gradient and pattern of stable alluvial rivers should be controlled by the quantity of water and quantity and type of sediment moved through their channels”, and he summarised the relevant relationships in a series of conceptual equations.

In New Zealand, a study of the hydraulic geometry of South Island rivers indicated that 53% of the variability of river morphology could be “explained” in statistical terms by variables characterising river flow, sediment character, bank erodibility, and flow variability (Mosley 1981). For some morphological variables, the level of explanation was much higher. For example, nearly 90% of the variability in cross-sectional area XS could be “explained” in terms of mean annual flood Q_{maf} , mean diameter of bed sediment D_{mean} , and percentage of silt and clay in the banks $SILT$ (an indicator of the resistance of the banks to lateral erosion), according to the relationship:

$$XS = 0.41 Q_{maf}^{0.86} D_{mean}^{-0.30} SILT^{-0.16} \quad 8.11$$

Nevertheless, Mosley’s statistical analysis indicated that 47% of morphological variability was *not* explained by flow and sediment characteristics. Schumm and Lichty (1965) showed that river morphology is influenced not only by water and sediment discharges, but also by:

- geology (lithology, structure, historical geology);
- paleoclimate and paleohydrology (the present-day influence of earlier climates and hydrologic regimes, particularly during the Pleistocene);
- topographic relief, base level changes, and mass above base level;
- valley dimensions and the degree of constriction of the channel by the valley sides;

Table 8.3 Characteristics of a river relevant to resource management requirements.**NATURAL CHARACTER AND SOCIO-ECONOMIC MODIFICATION****A. LANDSCAPE SETTING – LANDSCAPE BEYOND THE RIVER FLOODPLAIN**

Valley type and dimensions; other significant landforms

Lithology

Vegetation, land use, erosion types

Presence, number and character of terrace levels

Degree of constriction of channel by valley sides

B. RIVER MARGINS AND FLOODPLAIN SETTING

Presence, extent and dimensions of floodplain

Floodplain surficial material and soil type

Floodplain vegetation and land use

River margin vegetation and land use (riparian habitat)

C. CHANNEL CHARACTER

Channel pattern and presence of islands and bar forms

Channel dimensions – width, sinuosity, meander wavelength, slope, degree of braiding

Obstructions in channel – degree and type (including organic debris)

Bank height and type/degree of erosion

Bed material (including rock outcrops)

Bank material (including rock outcrops)

Instream habitat and cover for biota

Degree of human modification, including: bed disturbance and resulting water turbidity; structures; introduction or removal of vegetation; deposition of substances; reclamation of the bed.

D. WATER CHARACTER

Hydrologic regime – flow variations

Colour, turbidity/clarity, odour, taste

Flow velocity, depth, turbulence

Temperature Water chemistry (selected determinands such as nitrates, DO, including pollutants)

Presence of floating solids, solids on bed, floating liquids

Bacteriological water quality

Fauna present (fish, invertebrates, etc.)

Flora present (macrophytes, periphyton, etc.)

E. SOCIO-ECONOMIC CONDITIONS

Human population: numbers, distribution, settlements, waste disposal to water Industries: type, products, output, waste disposal to water

Water resource uses

AMENITY VALUE

F. VISUAL QUALITY (the following can be ranked on a scale 1 to 5, for the valley segment or river reach)

Unity: the degree to which the visual elements coherently fit together as a whole

Identity: the extent to which a particular landscape has its own identifiable character

Diversity: the degree to which variation in form, texture and colour provides contrast

Mystery: the extent to which patterns in the landscape are partially hidden and “mysterious”

Uniqueness: the degree to which the landscape is unlike any other in the region

Sensitivity: presence of areas that are susceptible to undesirable modification

Visibility: the extent to which a landscape can be seen

Naturalness: the extent to which the landscape has been modified by people

G. RECREATIONAL POTENTIAL

Litter and other human waste

Presence of beaches, islands and campsites suitable for recreation

Presence of facilities such as picnic grounds, toilets, children's play areas, interpretive/visitor centres, etc

Accessibility of banks for vehicles and walkers

Ease of access to water for trailer mounted craft, and individual people

Points of interest, including archeological, historical and cultural features, nature walks, etc.

Grade of river on International River Classification Scale, and number of rapids

Floatability and obstructions to navigation for different craft

Flow fluctuation and permanence of flow

- climate and hydrologic regime in the source area; and
- vegetation cover along the river banks.

The form and behaviour of many New Zealand rivers clearly are affected by rapid tectonic uplift, outcropping bedrock, geologic structure, and the presence of Pleistocene and Holocene fluvio-glacial terraces, into which the rivers are cutting (Mosley and Schumm 2001). The influence of vegetation cover on channel morphology has become well-recognised in recent years and is discussed more fully below.

Dominant discharge and extreme events

Streamflow varies with time, and discharges during flood peaks may be two or three orders of magnitude (100 to 1,000 times) greater than during low flows (Chapter 10). As discharge increases, so too do depth, velocity, and shear stress τ :

$$\tau = \gamma RS, \quad 8.12$$

in which γ is the unit weight of water. Eventually, the shear stress exerted by the flowing water exceeds the threshold at which sediment particles on the stream bed can remain stable, and sediment transport is initiated (Chapter 12). This does not happen simultaneously at all locations on the stream bed, but as discharge increases, the area of the bed on which sediment is moving also continues to increase. As discharge increases, the rate of sediment transport at a given point increases also. This implies that changes to channel shape, which are accomplished by the movement of sediment from one place to another, become increasingly rapid as discharge increases. Two-dimensional computer models of rivers, which show the distribution across the riverbed of local slope, depth, velocity and sediment size, are becoming increasingly useful in studying the way in which sediment is entrained, transported and deposited in a channel, and therefore how the channel may evolve (e.g., Duncan and Carter 1997).

Extreme floods, with recurrence intervals of once in 50 or 100 years or so, can cause catastrophic changes to river channels. This happened over a large area of southwestern South Island during a severe storm in 1994, when floods eroded streambanks and terraces, transported vast quantities of sediment, and transformed streams and rivers from narrow, stable channels to wide, raw expanses of mobile gravel and cobbles (Fig. 8.7). Such events can have an impact on the channel that is visible for decades. Part of the reason for this is that, during an extreme event, land-forming processes in addition to sediment transport by flowing water become important.

Wolman and Miller (1960) pointed out that catastrophic events, which may cause dramatic changes in the landscape, happen too infrequently and for too short a



Figure 8.7 Valley bottom of Temple Stream, near Lake Ohau, devastated by a major flood in 1995. Photo: P. Mosley

period to accomplish a significant proportion of the total work expended in reshaping the landscape. On the other hand, low flows, which occur for much of the time, do not have the energy to erode banks, transport bed material, or reshape the channel. Instead, they suggested that moderate events, occurring every few months or years, accomplish most geomorphic work, in total. This notion is summarised in the concept of dominant discharge, the discharge that is most effective in transporting sediment and reshaping the channel (Fig. 8.8). Dominant discharge is commonly taken to be equivalent to the bankfull discharge or mean annual flood. The concept of dominant discharge simplifies the reality that the form of a channel at any one time reflects the integrated effects of all the events—some small and some large—that have reworked the river bed in the preceding months and years, since the last “catastrophe” completely reshaped the river.

A graphic way of considering the relationships between channel shape, flow and sediment is to compare the hydraulic geometry of different types of river. Hydraulic geometry relationships describe the way in which water at a range of discharges accommodates itself to a river bed that has been created largely during earlier flood events. Hydraulic geometry can be considered at one location, as discharge varies with time (at-a-station hydraulic geometry), and at different locations along the course of river, as discharge varies with drainage area (downstream hydraulic geometry).

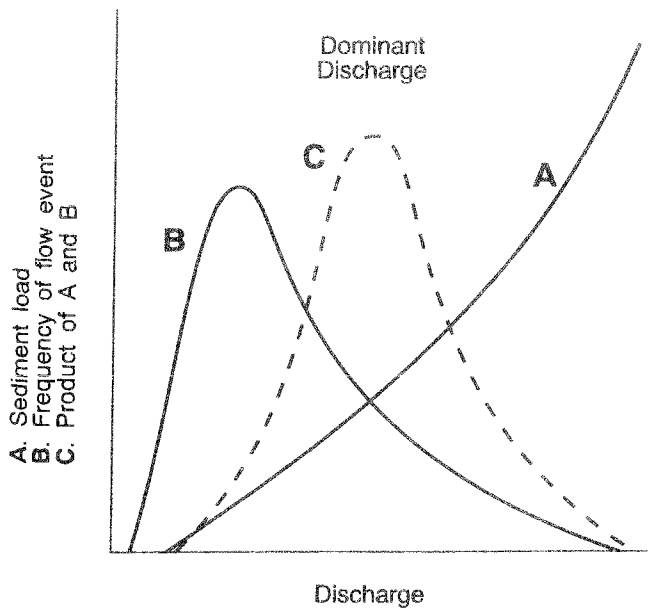


Figure 8.8 Definition of dominant discharge in terms of sediment transported by a range of flow events (after Wolman and Miller 1960).

At-a-station hydraulic geometry

The at-a-station hydraulic geometry of three rivers, the Buller, Ohau and Waipaoa Rivers, shows the influence of different bed and bank conditions on channel shape (Fig. 8.9). The sites on the Buller and Waipaoa are at gauging stations and therefore are in particularly uniform channel sections; the sites in the Ohau River just above its entry to Lake Benmore were randomly selected, and are representative of this section of the river.

The Waipaoa River near Gisborne has a broadly parabolic bed composed of sand and fine gravel, and almost vertical silt banks that are rather resistant to erosion. Water surface width increases as a power function of discharge up to flows of about $30 \text{ m}^3/\text{s}$, at which point the water level reaches the foot of the banks. Thereafter, water surface width increases more slowly, although still as a power function of discharge. The plot for mean depth as a function of discharge also has two segments that reflect the shape of the channel, but mean velocity increases as a simple power function of discharge across the full range of flows.

The Buller River near Murchison has a rock-bound channel with an almost flat gravel bed. It shows a similar set of relationships to those of the Waipaoa, but with its rock banks it is significantly narrower for the same discharge than the Waipaoa, with silt banks. For example, at flows of $100 \text{ m}^3/\text{s}$, the Buller is 47 m wide, and the Waipaoa is 87 m wide. Their depths are similar at this flow, about 1.9 m, but the mean velocity of the Buller is higher—1.6 m/s, in comparison with 1.1 m/s in the Waipaoa—and increases more rapidly as discharge increases.

The Ohau River (now dewatered by the Upper Waitaki power scheme) had a broad, gravel-bedded braided channel with non-cohesive gravel banks and multiple channels. The average width of flowing water, summed across all channels at several cross-sections, was much greater than in the Waipaoa or Buller, about 250 m at a discharge of $100 \text{ m}^3/\text{s}$. This reflects the lack of cohesion of the Ohau's banks, and the ease with which branch channels could migrate across the river bed and undercut the banks.

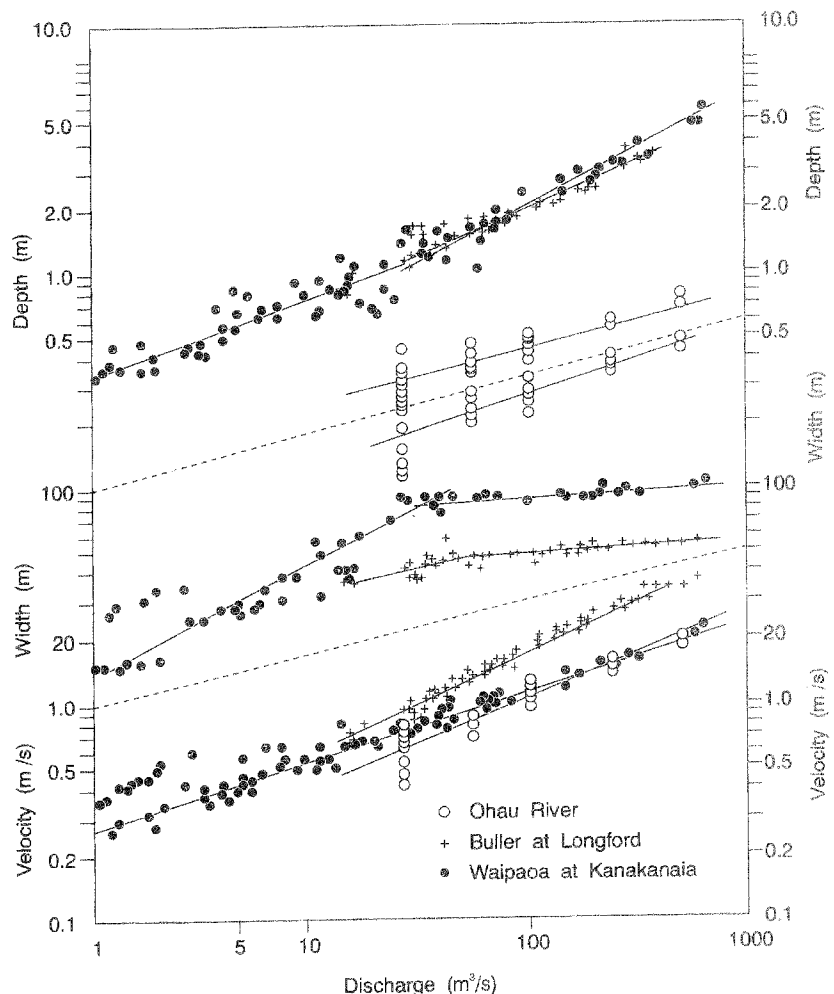


Figure 8.9 At-a-station hydraulic geometry of the Buller, Ohau and Waipaoa Rivers.

Average depth was substantially less, about 0.45 m at 100 m³/s, reflecting the broad, dish shape of the branch channels and the presence of wide sheets of migrating gravel. However, the average flow velocity in the braided Ohau was similar to that in the single-channel of the Waipaoa across a wide range of flows, probably because the greater channel slope compensated for the coarser bed material and shallower depth (Eqs. 8.6-8.8).

Downstream hydraulic geometry

Downstream hydraulic geometry relations are best developed using data measured along the course of a single river system, but scientists often have combined data from different systems, to draw on readily available observations at flow gauging sites. In either case, hydraulic geometry relations must be developed for a single characteristic discharge, such as the mean annual flood or the mean annual discharge.

Griffiths (1980) demonstrated good agreement between exponents for combined data from six New Zealand rivers and the range of values that have been found in other studies. There were no obvious differences between the hydraulic geometry relationships for the six rivers, although sample sizes for each river were far too small to permit any statistically valid comparison (Fig. 8.10). More recent, high quality data sets for New Zealand rivers have shown somewhat different results, however. Jowett (1998) obtained data for an additional 73 sites on rivers and streams throughout New Zealand, while Ibbitt (1997) reported the results of more intensive data collection in

the Ashley and Taieri River catchments (Table 8.4). In all cases, the exponents for the width relationship are close to 0.5, the “traditionally accepted” value. However, the exponents in the power function relationships between discharge and depth were substantially less, and between discharge and velocity were substantially more, than the “traditionally accepted” values found in many other studies. This implies that, for the rivers considered by Jowett and Ibbitt, downstream increases in flow tend to be accommodated to a greater extent by an increase in flow velocity, and to a lesser extent by an increase in depth, than is commonly found. Jowett suggested that differences in stream gradient may provide the explanation, since a steeper river should be shallower and swifter than an equivalent low-gradient river, and the sites included in his dataset tended to be on smaller and steeper stream and rivers, while Ibbitt’s data were from headwater streams.

Influence of vegetation on channel morphology

A basic assumption that underlies much river control work in New Zealand is that the form and behaviour of river channels is influenced by riparian vegetation (Collier *et al.* 1995, p. 15-45). It is believed that the roots of trees growing along the riverbank reinforce the floodplain soils and inhibit erosion. Even if the trees are undercut, they tend to fall into the river and subsequently protect the bank from high velocity flows. In addition, thickets of trees on the floodplain reduce the velocity of overbank flows and encourage deposition of fine sediment, which reduces the risk that overbank flow will enter neighbouring farmlands.

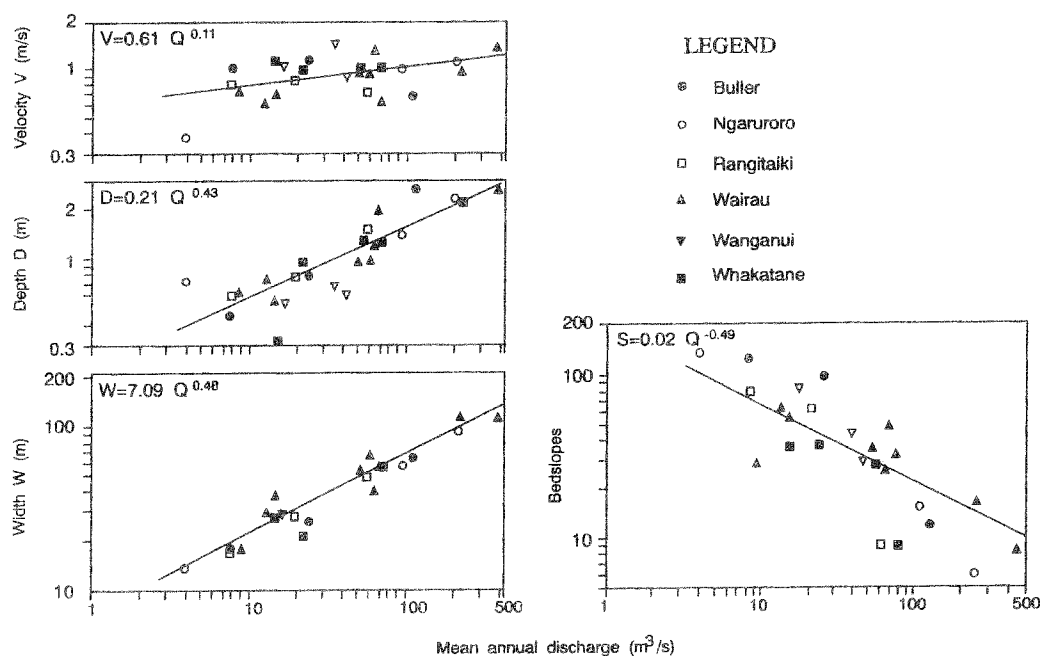


Figure 8.10 Downstream hydraulic geometry of six New Zealand gravel-bedded rivers (Griffiths 1980).

Table 8.4 Values of exponents in the power function relationships for downstream hydraulic geometry.

Source	b in $W \propto Q^b$	f in $d \propto Q^f$	m in $v \propto Q^m$
Six NZ rivers (Griffiths, 1980)	0.48	0.43	0.11
73 NZ rivers (Jowett, 1998)	0.49	0.24	0.24
Ashley River (Ibbitt, 1997)	0.44	0.24	0.32
Taieri River (Ibbitt, 1997)	0.52	0.25	0.24
Hutt River (Ibbitt, 1997)	0.52	0.14	0.34
“traditionally accepted” values (Griffiths, 1980)	0.50	0.40	0.10

Thousands of kilometres of stream and riverbank have been protected from erosion by planted willows (Fig. 8.11). There has been surprisingly little research in New Zealand to assess the relationship between riparian vegetation, streambank stability and channel morphology, although there is a growing international literature on the subject (Millar 2000). Various case studies suggest that vegetation removal tends to permit bank erosion and channel widening, whereas vegetation re-establishment leads to narrower and more stable channels. For example, the Tamaki River, which flows from the eastern Ruahine Range, responded rapidly to forest clearance along its banks in the early years of the 20th century. In the first 2.5 km from its exit from the range, it widened from less than 10 m in the early 1920s, only a few years after the forest was cleared from its floodplain, to an average of 54 m in 1942 (Mosley 1978). From 1942 to 1976, channel width increased by only another 6 m. The widening during the earlier period, which occurred particularly during large storms, was probably a response to declining bank resistance as tree roots decayed.

On the other hand, many rivers have become narrower

in recent years, apparently in response to natural colonisation of their riverbeds by willows and shrubby weeds. For example, the Waipara River, North Canterbury, has narrowed progressively and dramatically since 1950 (the date of the earliest available aerial photographs), with no obvious environmental changes other than encroachment of woody vegetation onto the riverbed that might have caused the trend (Fig. 8.12; Mosley 2003). In some respect, this process appears to be returning many Canterbury rivers to a stable, anastomosing form that they might have had before the natural forest cover was removed by colonising Maori and Europeans (Blakely and Mosley 1987), albeit with exotic rather than native species. Effectively, such rivers are being shifted along the continuum of channel types depicted in Figure 8.14, as channel and bank stability, and sediment load are modified.

A particularly interesting case is the Waitaki River, presently braided, but becoming progressively less so. Hicks *et al.* (2002) have considered the Waitaki in terms of Millar’s (2000) model to discriminate between braided and meandering rivers:

$$S^* = 0.0002 D_{\text{median}}^{0.61} \phi^{1.75} Q_{\text{bankfull}}^{-0.25} \quad 8.13$$

in which S^* is the threshold channel slope between braided and meandering rivers, Q_{bankfull} is bankfull discharge, D_{median} is the median diameter of bed material, and ϕ is the bank sediment friction angle in degrees. ϕ is an index of bank stability and resistance to erosion, and is partly a function of bank vegetation (Millar 2000). Hicks *et al.* used observed values for the Waitaki of 23 mm for D_{median} , 1,000 m³/s for Q_{bankfull} , and 0.0033 for channel slope S , and values for f obtained from the literature. They concluded that the Waitaki should be braided only under unvegetated conditions, and that development of a partial tree/shrub cover would increase the bank sediment friction angle to the point that the river would cross the threshold from braided to meandering. There has been a progressive increase in vegetation cover in the Waitaki River bed, from



Figure 8.11 Bank protection by planted willows and poplars along the North Branch, Ashburton River.

Photo: P. Mosley

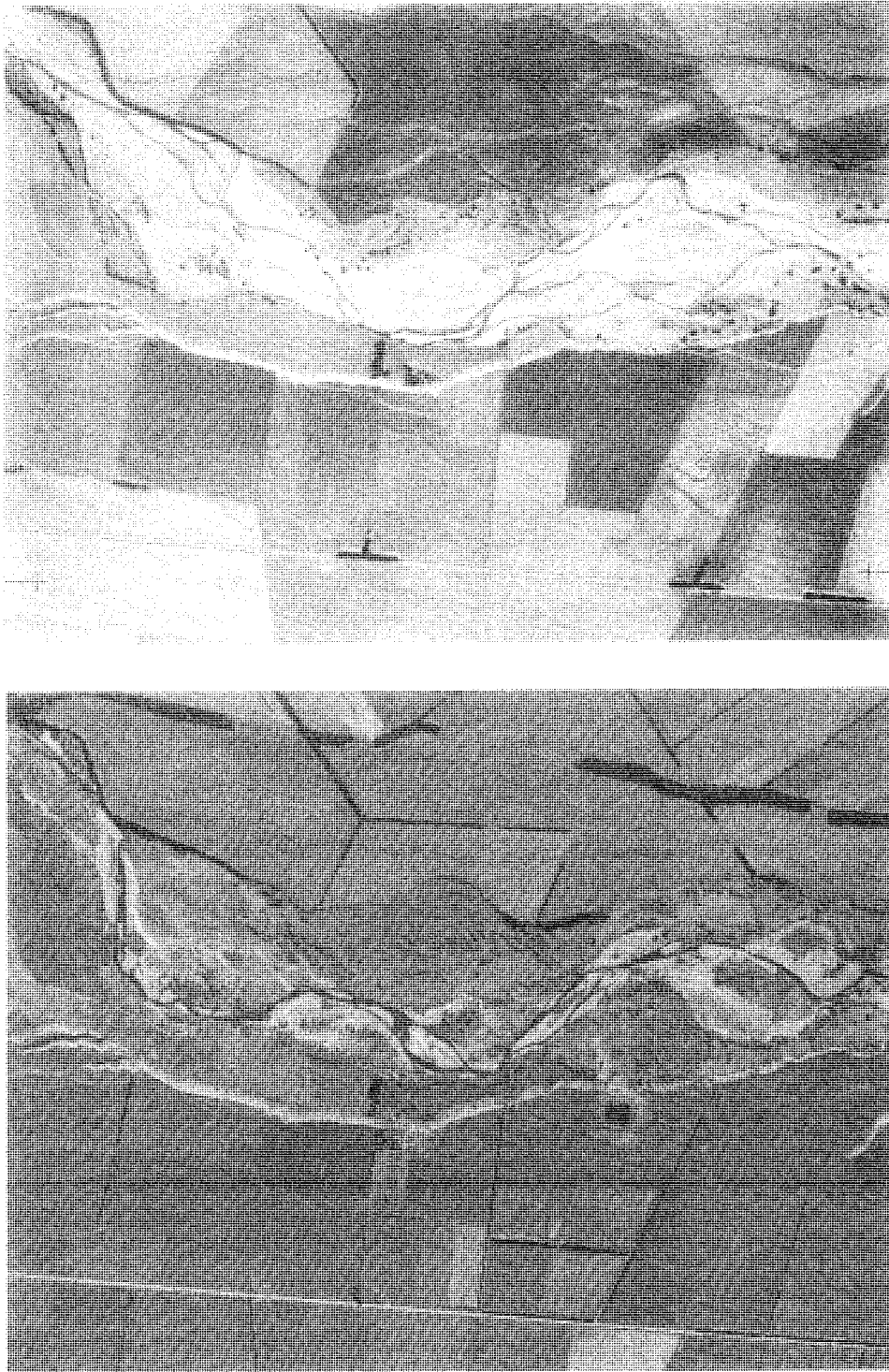


Figure 8.12 Waipara River just upstream of State Highway 1 in 1950 and 1995. Flow is from left to right (west to east), and the areas shown are 2.9 km across.

Photos: P. Mosley

a largely bare riverbed in 1936, to a 51% vegetation cover by 2001. At the same time, there has been a progressive reduction in channel width by 10-15 m/year. Hicks *et al.* conclude that the Waitaki is becoming narrower, more stable, and less inclined to braid, a tendency that has been encouraged by vegetation, and also by the reduction of flood flows, associated with hydro-electricity generation.

CLASSIFICATIONS OF RIVERS

Classifications of rivers can be helpful in clarifying the relationships between channel morphology and the factors that control it. Many different classifications have been developed, with variables to discriminate between classes chosen to fit the intended purpose of the classification. Nevins (1965), a river control engineer, classified New

Zealand rivers on the basis of “the fundamental relationship between the quantity of detritus (sediment) carried by rivers and its dominant grain size, and the quantity of water and the river slope”. He recognised four “phases” from source to sea:

1. The mountain or torrent phase. This phase is characterised by steep slopes, falls and gorges, and an abundant supply of coarse, angular sediment. Such streams and rivers are widespread in the mountains.
2. The shingle phase. Where the river leaves the mountains, it flows in one or more shallow channels that wind across a wide gravel bed. The course tends to be straight, with a steep slope; the river may be aggrading, and probably reworking earlier deposits, as it erodes its bank and migrates laterally.
3. The silt phase. In some rivers, there is an abrupt transition to a channel with sand bed and silt banks, and the river flows in a deep, narrow meandering channel across a flood plain.
4. The tidal phase. This class of river is rather restricted in distribution in New Zealand, with only the lower few kilometres of most rivers being included. The tidal phase of most rivers is very similar to the silt phase, with a deep, narrow, sinuous channel composed of silt banks and sand bed.

This simple classification is in terms of channel gradient, sediment load, degree of confinement by valley sides, and position along the longitudinal profile, from source to sea. In addition, Nevins identified regions in which catchment geology differed and could be expected to provide different types of sediment load.

Recently, the River Environment Classification (REC) has been developed in New Zealand to assist in managing aquatic ecosystems (Snelder and Biggs 2002). The REC classifies channel segments in terms of climate, source of flow, geology, land cover, stream order, and valley slope, which are all variables that are obtainable from GIS databases. It does not use channel morphology as part of the classification, because morphologic data that are useful for discriminating channel classes are not readily obtainable from GIS databases. In the process of developing the REC, however, a classification procedure for channels was prepared, based on a mixture of “causal” and morphologic variables:

1. Channel gradient (causal: geologic environment)

2. Degree of entrenchment (causal: geologic environment)
3. Bed material size and nature of bank materials (causal: geologic environment)
4. Degree of channelisation (causal: human influence)
5. Degree of tidal effect (causal: boundary condition)
6. Degree of braiding (morphological)
7. Sinuosity and presence/absence of rhythmic bedforms (morphological)

The resulting channel types may be arranged along the river profile (Fig. 8.13), in the same way as the “phases” of Nevins (1965), but need not be. Channelisation may be imposed on any of the types of channel identified in Figure 8.13.

An alternative approach is to recognise that rivers tend to be distributed in a continuum of channel types, which grade into each other. This is because most of the variables that control channel morphology, such as sediment load or valley slope, are continuous variables, without distinct breaks that might cause a change from one channel type to another. The channel types are not necessarily associated with their position along the channel course, as is implicit in Nevin’s classification. The model of channel types proposed by Mollard (1973) includes types that are recognised as distinct classes by other scientists, but shows how they grade into each other, in response to the variables that control morphology (Fig. 8.14). Mollard’s model was developed for alluvial rivers in northern Canada, but it is readily usable in New Zealand. It is particularly interesting, because it suggests that the many efforts to discriminate between meandering and braided rivers are based on an over-simplified assumption that these classes of river are distinctly different. Detailed studies of New Zealand rivers

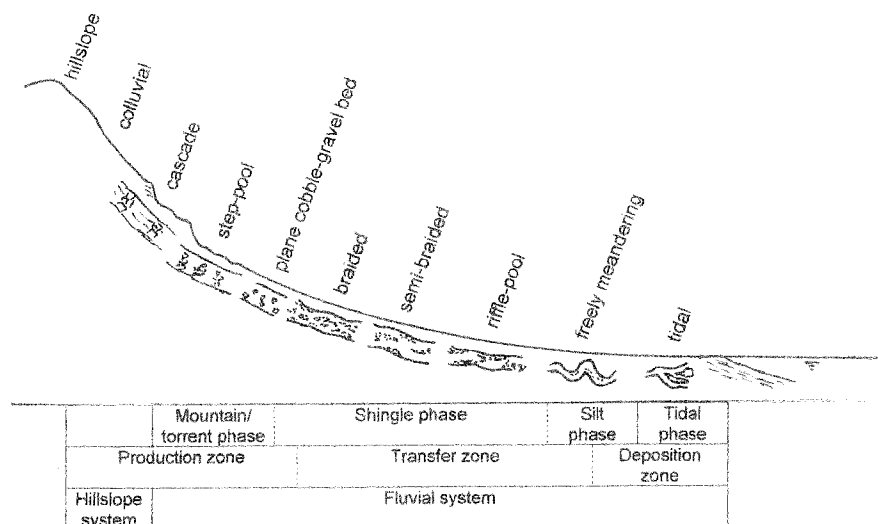


Figure 8.13 Idealised model of river types, arranged according to the longitudinal profile (Mosley and Schumm 2001).

such as the Rangitata, which changes progressively from a meandering course to a fully-braided course as it flows across the Canterbury Plains, have shed considerable light on what Carson (1984) termed the “meandering-braided river transition”. Carson developed his own descriptive scheme, in which the principal variables were bank erodibility and relative bed material supply rate. Meandering rivers tend to have low bank erodibility and low rates of supply of bed-material; the converse is the case with braided rivers, but there is a gradation between them, as indicated also by Figure 8.14.

THE RIVERSCAPE

The appearance of the landscape along the course of a river—the riverscape—is an important element of the amenity value of a river, which is defined by the Resource Management Act as “the natural or physical qualities or characteristics of an area that contribute to people’s appreciation of its pleasantness, aesthetic coherence, and cultural and recreational attributes”. In New Zealand resource management practice, the riverscape commonly is dealt with in qualitative terms, with landscape architects providing their personal appraisals of landscape quality and, where relevant, the likely impacts of proposed developments.

The types of analysis that might be carried out are exemplified by expert evidence submitted to hearings of applications for water conservation orders and resource consents. Normally, such analyses are carefully referenced to relevant provisions of resource management law and planning documents (regional policy statements, regional and district plans, etc.), because they must be defensible in court. For example, in an appraisal of the scenic qualities of the Rangitata River, Di Lucas (2001, unpublished evidence to a Special Tribunal on behalf of Department of Conservation) grounded her analysis on a discussion of the provisions and definitions of four acts of Parliament and interpretive case law, the New Zealand Coastal Policy Statement, as well as regional, district and local planning documents. She described the river, section by section, in terms of:

- the scale and openness of the landscape,
- the geological processes evident,
- the aesthetic quality of such scenic elements as braided channels,
- intrinsic values such as the presence of birdlife,
- visibility of human activity,
- the presence of specific natural features such as the river mouth and lagoon,
- availability of viewpoints,
- technical aspects of the riverscape such as legibility and aesthetic harmony, and even

- the extent to which the river appears in literature and art.

This “expert paradigm” approach to landscape assessment by landscape professionals is not the only one available. The “psychophysical paradigm”, which assumes that observable or measurable properties of the landscape provide a stimulus to which individuals respond, has emerged internationally as perhaps the dominant direction for both research and practice. This approach characterises the landscape using objective and quantitative measures; this not only avoids the subjectivity of the “expert paradigm”, but enables quantitative and predictive relationships to be developed with, for example, streamflow or data from surveys of public preferences. Mosley (1989) used this approach to investigate the characteristics of a river scene that people respond to in deciding their scenic preferences. He calculated the “scenic attractiveness” of 190 river scenes, using surveys of 18 groups of people with different interests and backgrounds, and analysed their relationship with 63 variables describing the scenes. The seven variables that together were best able to explain preferences statistically (coefficient of determination $R^2 = 0.73$) were:

- the percentage of the view composed of indigenous forest,
- the angle of the line of sight to the most prominent point in the view,
- the elevation of the most prominent point in the scene above the viewpoint,
- the percentage of the view composed of alpine land cover,
- the percentage of the view composed of water surface,
- the confinement of the river by overhanging vegetation, hillsides, etc., and
- the colour class of the river (on an 8-point scale from colourless to muddy brown and grey).

The preferences included only two that relate to the river itself, and two that relate to its margins. One variable alone, the percentage of the view composed of indigenous forest F , explained 41% of the variation in scenic attractiveness SC , according to the equation:

$$SC = 4.6 + 3.56 F \quad 8.14$$

Mosley’s study provides a basis for predicting the likely attractiveness of a particular riverscape to members of the public, either using the best-fit equation or comparison with rivers in his set of 190. It also provides a means of judging whether a proposed change to the riverscape might modify its amenity value.

An important source of information about New Zealand’s riverscapes is the national survey of river scenery and recreational value carried out by Egarr and Egarr

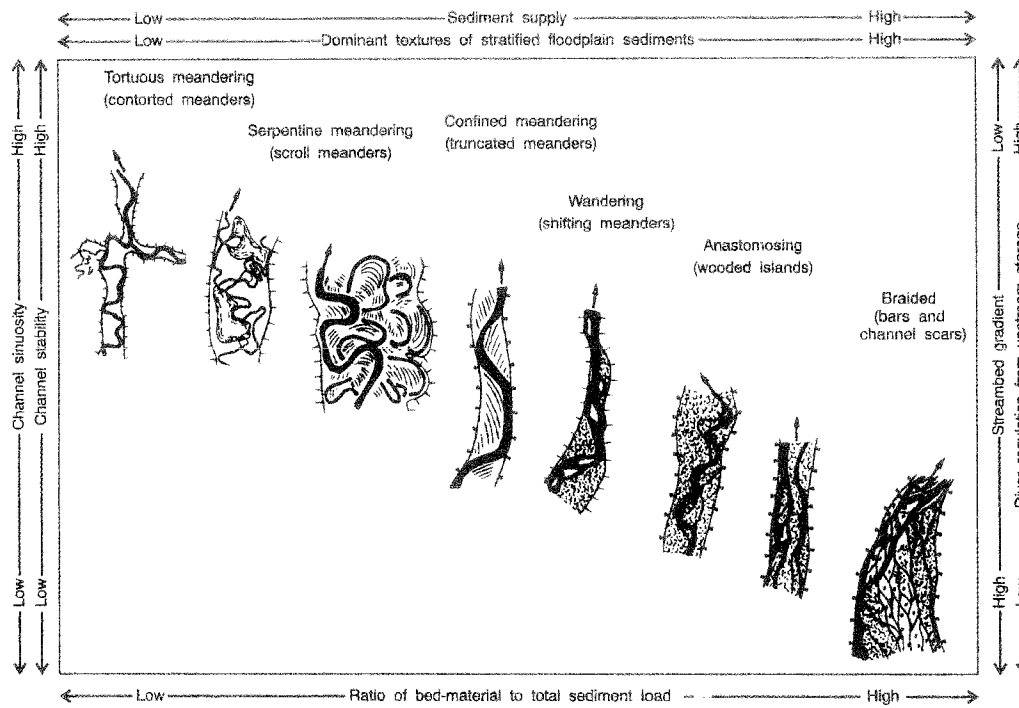


Figure 8.14 The continuum of river types, and controlling variables (after Mollard 1973).

(1981). This survey was carried out, in consultation with local river users, by visiting, traversing and completing checklists for each river. Among other things, each section of river was ranked in terms of scenic values (vista, grandeur, solitude, naturalness, colour/texture, water quality, river spectacle) to produce an overall scenic ranking, on a six-point scale from "dull" to "exceptional". While perceptions of river scenery and requirements for recreation have changed since Egarr and Egarr's survey was completed, it provides a useful starting point for considering riverscapes today.

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Chapter 7

Flow regimes

Maurice Duncan and Ross Woods

INTRODUCTION

In New Zealand you can see a wide range of types of rivers. There are boulder-filled mountain torrents, issuing from glaciers in mountains only a few kilometres from the coast; wide, braided, gravel-bedded channels; meandering silt watercourses; tree-lined urban waterways. Some that rise in the high mountains may change dramatically along their course, before discharging via a lagoon or estuary to the sea. Rivers arising in the foothills or from lowland springs tend to be more uniform.

But every river has its own unique character. What makes each one different? The answer lies in the combination of physical and climatic features that influence what we call the “flow regime” or “hydrologic regime” of a river.

WHAT IS FLOW REGIME?

The flow regime (or hydrologic regime) of a river is the unique way that its flow changes from day to day, season to season and from one year to another. Regime defines the character of a river, how liable it is to flood or to experience long periods of low flow, what it looks like; what lives in it; whether it is potentially useful. For particular management purposes, various aspects of flow regime may be significant, but in general we require information about extreme high flows, extreme low flows, average flows (equivalent to the total volume of water discharged by the river), flow variability, and the frequency or spacing of significant events, such as “flushing flows” (Chapter 43).

Differences in flow regime are best illustrated by looking at graphs of flow from different rivers. A hydrograph is a graph of the change in either a river’s water level (often called stage) or its flow (discharge) over time. Two main components of river flow can be identified from a hydrograph: baseflow and flood flows (often termed

quickflow) (Fig. 7.1). The baseflow of a river is derived from seepage of ground water into the channel or from lake outflows; it may be large or small, but it tends to change slowly. Flood flows occur on top of the baseflow. They are produced from precipitation directly into the channel, from overland flow down surfaces sloping into the channel, from water that infiltrates into the soil and moves quickly to the stream channel (interflow) and from runoff from wet areas near stream channels (Chapters 4, 9). Flow regimes differ in the magnitude and frequency of high and low flows due to differences in total precipitation, as well as in their flow variability, the magnitude of high and low flows relative to base flow, and their flashiness (Snelder and Biggs 2002).

Hydrographs of floods commonly show the rise of floodwaters (termed the “rising limb”) and their recession

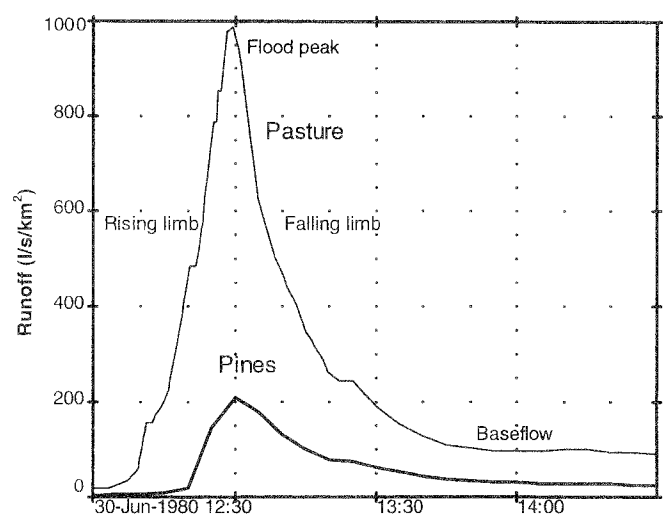


Figure 7.1 The key components of a hydrograph. The difference in regime caused by change in land use is shown by hydrographs from a pasture catchment (thin line) and a pine catchment (thick line) of similar size (approximately 7 hectares) for the same rainstorm.

(“falling limb”) (Fig. 7.1). The slopes of the rising and falling limbs tell us about the nature of the rainfall that caused the flood and also about the catchment itself (Chapter 9). For example, during “flash floods” caused by intense rain falling onto an already saturated catchment, streams rise rapidly because a greater proportion of the water runs directly into the stream network rather than soaking into the ground.

In hydrograph analysis, baseflow and floodflow are separated by drawing a line from the start of the rising limb of a flood to a point on the falling limb. Because the baseflow is usually higher after a flood, the line has an upward slope that is usually selected based on experience—a figure of $0.004 \text{ ml/s/km}^2/\text{s}$ (Hewlett and Hibbert 1967) was used to separate baseflows and flood flows in Figure 7.2.

The particular combination of baseflows and flood flows for a river is a crucial aspect of its flow regime. Rivers may have a stable regime with a limited variation in flow, or a regime with very variable flows. The Buller River at the outlet of Lake Rotoiti (Fig. 7.2A) shows small, regular, slowly rising and falling floods throughout the year, on top of a large sustained baseflow. The opposite extreme is illustrated by the Whareama River (Fig. 7.2D) which shows a clear seasonal pattern of virtually zero flow in summer but a sustained baseflow in winter, with frequent, short, flashy floods.

Floods may happen regularly, e.g., virtually weekly on the South Island’s West Coast, or only occasionally. In some rivers, floods are seasonal—often in winter and spring in east coast streams.

WHY IS THE FLOW REGIME IMPORTANT?

The river as a habitat

The flow regime of a river, in combination with other factors such as temperature and water quality, influences the plants and animals that can live in it. As an example, consider the conditions favourable for brown trout. They like cool, clear, bouldery rivers that have stable flow regimes with few floods and high baseflows (Chapters 17, 43). There are several reasons for this. When the riverbed is nearly always covered in water the food chain can maintain full production. Algal slimes can grow on the

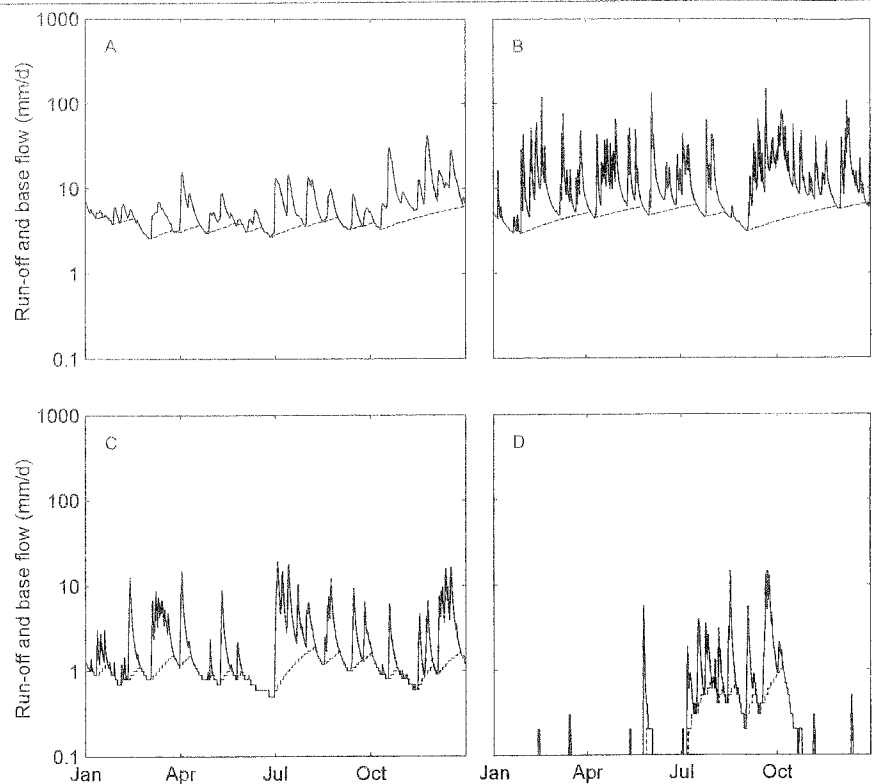


Figure 7.2 Examples of annual runoff and base flow hydrographs of rivers with low- and high- variability flow regimes. Note the logarithmic scale. The dotted line separates base-flows from flood flows using the method of Hewlett and Hibbert (1967).

Graph	River	Year	Catchment Area (km ²)	FRE3
A.	Buller at Lake Rotoiti (Nelson)	1984	195	4.6
B.	Ahaura River (Westland)	1981	790	17.5
C.	Wairoa River (Nelson)	1984	464	19.9
D.	Whareama River (Wairarapa)	1984	398	0.7

gravels and boulders on the riverbed, and aquatic insects, the main food of trout, in turn can feed on the slimes. If there are frequent floods, the slimes (Biggs 1990) and insects (Jowett and Richardson 1989; Quinn and Hickey 1990b; Sagar 1986; Scrimgeour *et al.* 1988) get washed away or ground off as the riverbed moves in the flood, so there is less food for both insects and trout. Streams with high base flows almost always have water that is deep enough for trout to hide and rest. Another reason relates to spawning. In a stable flow regime, it is less likely that floods will wash away the redds (areas where the eggs are laid), and there will usually be enough water to carry oxygen through the gravels to the eggs. The clear water preferred by trout has little sediment to clog up the redds. Boulders provide both white water to hide, and still areas to rest. Thus, trout are adapted to survive in a stable regime. However, other creatures may prefer conditions associated with other types of flow regime (Biggs 1990; Quinn and Hickey 1990a,b; Sagar 1986; Scrimgeour *et al.* 1988).

Human use of rivers

A river's flow regime also affects the way in which people can use it. For example, monthly flows in rivers like the Rakaia and Ahuriri Rivers (Fig. 7.3), which drain from the Southern Alps, are highest in spring and summer. This is also the time of highest demand for irrigation waters. Therefore, water can be taken directly from the river on a "run-of-river" basis, and there is no need for costly storage reservoirs. On the other hand, demand for hydro-electric power peaks in mid-winter, and control structures have been built to augment and manage the storage capacity of lakes such as Tekapo, Pukaki and Hawea, which have river inflow patterns with a winter minimum, similar to those of the Rakaia and Ahuriri (Fig. 7.3).

Sometimes a river's flow regime is very suitable for one use but poor for another. For example, Nelson rivers have a monthly flow regime similar to that of the Hakataramea River in South Canterbury, (Fig. 7.3) in which summer flows are low, with slow clear water. These conditions are ideal for swimming, and suit holidaymakers visiting Nelson. On the other hand this same flow regime restricts the amount of water available to irrigate Nelson's important horticultural crops.

INDICES OF FLOW REGIME

What methods are used to describe and compare various aspects of (i.e. to "index") the flow regime of a river? Flow regimes can be discussed in terms of the variation of the flows, for example the frequency of floods above a given threshold, the sequence of mean monthly flows through the year, or the mean annual 7-day low flow. Each of these tells us something different about the flow regime of a river. Indices of flow often are expressed in terms of specific discharge (flow per unit area, in $l/s/km^2$), to help in comparing catchments of different sizes.

Maps and information on mean flows, sediment discharges, river temperatures, low flows and floods of New Zealand rivers can be found in Duncan (1987) and in Figures 7.3 and 7.4. Other studies concentrate on single aspects of the flow regime, such as low flows (Hutchinson 1990) or floods (McKerchar and Pearson 1989).

The River Environment Classification (Snelder and Biggs 2002) used 13 different variables to help characterise the intra-annual variation in flow conditions relevant to five ecologically significant aspects of flow regime suggested by Richter *et al.* (1996), Poff *et al.* (1997), and Poff and Ward (1989): magnitude of the flow variation, frequency of flows above a threshold, duration of high and low flows, timing of flows and rate of change of flows. One useful and ecologically significant measure of the frequency of high flows is the average number of floods per year exceeding three times the median flow (FRE3) (Clausen and Biggs 1997, 1998).

Variation of flows

The value of FRE3 unlike other measures of flow variability such as the coefficient of variation (CV) of flow (the standard deviation of the instantaneous flows divided by the mean flow), has been shown to be ecologically relevant (Clausen and Biggs 1997, 1998). It provides a simple index of the flow variability that in part determines the ability of algae, macro-invertebrates and other aquatic biota to become established. From measurements of periphyton biomass in 26 New Zealand rivers, Clausen and Biggs (1998) showed that as FRE3 increased, the amount of biomass decreased. FRE3 is an index of the frequency of disturbance by a combination of shear stress and bed abrasion, so that the more frequently floods exceeded FRE3, the more frequently periphyton was removed from the bed, and the shorter the time available for biomass to accumulate. The same study examined the relationship between FRE3 and benthic invertebrate numbers for 63 sites, and found a slightly curvilinear relationship with high densities for intermediate values of FRE3 (10–20 freshes per year).

FRE3 must be considered as an indicator of flood events that cause ecological disturbance, rather than as a threshold. Successively higher flows cause increasingly high shear stresses over increasingly large areas of the riverbed, so that sediment movement and sloughing of algal mats become increasingly frequent and widespread. Although a threshold of motion for sediment transport can be defined for a given sediment type (Chapter 12), it is more difficult to identify a "threshold of ecological disturbance"—indeed, algal mats can slough off even at low flows, if the mats become large enough. A given "FRE3 flood" (one that peaks at or above the FRE3 for a particular river) is not necessarily more ecologically significant than a smaller fresh; factors such as the preceding sequence of flows must also be considered.

A low FRE3 value indicates a stable flow regime. Rivers with few floods (FRE3 <5) tend to be controlled by large lakes or are mainly spring-fed, such as the Buller River at Lake Rotoiti (Fig. 7.2A). Their flow is mainly baseflow, and there are very few, small floods. Such rivers are typically rich in nutrients and they normally support a large amount of stream life. Rivers with much more variable flow (FRE >10) tend to drain high rainfall areas: they have a high baseflow, but also have frequent, large floods that disturb the riverbed. These conditions do not allow aquatic plant and animal communities to develop fully (Biggs *et al.* 1990). West Coast rivers such as the Ahaura (Fig. 7.2B) fall into this category. Some rivers have long periods of low flow, low base flows, large infrequent floods of short duration, and very low FRE3 values, e.g., the Whareama River (FRE3 <1)(Fig. 7.2D). Periphyton growth can rise to nuisance levels, and midges, snails and worms dominate the invertebrate fauna.

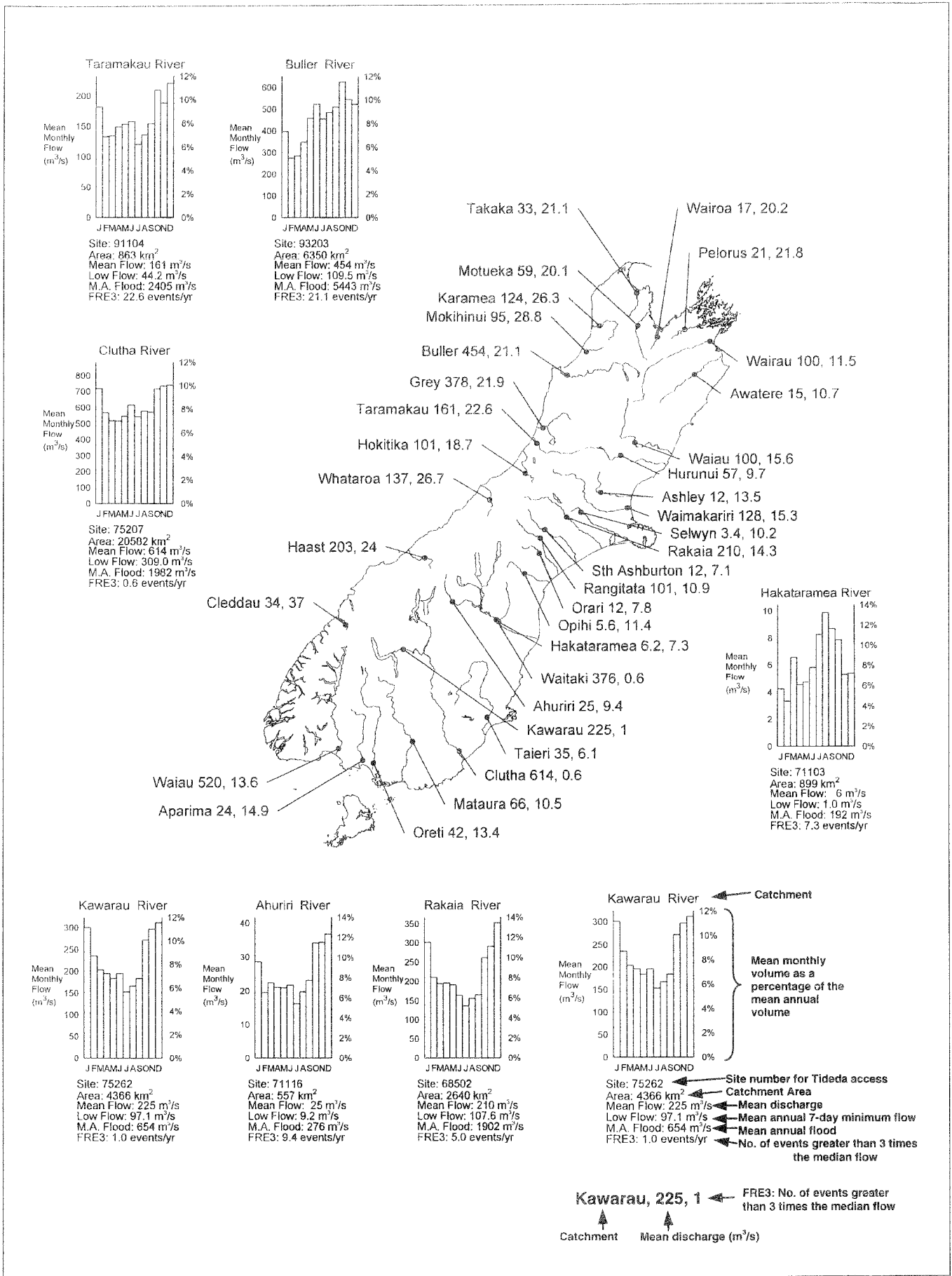


Figure 7.3 Indices of flow regime for representative catchments, South Island (from National Institute of Water and Atmospheric Research and Regional Council archives).

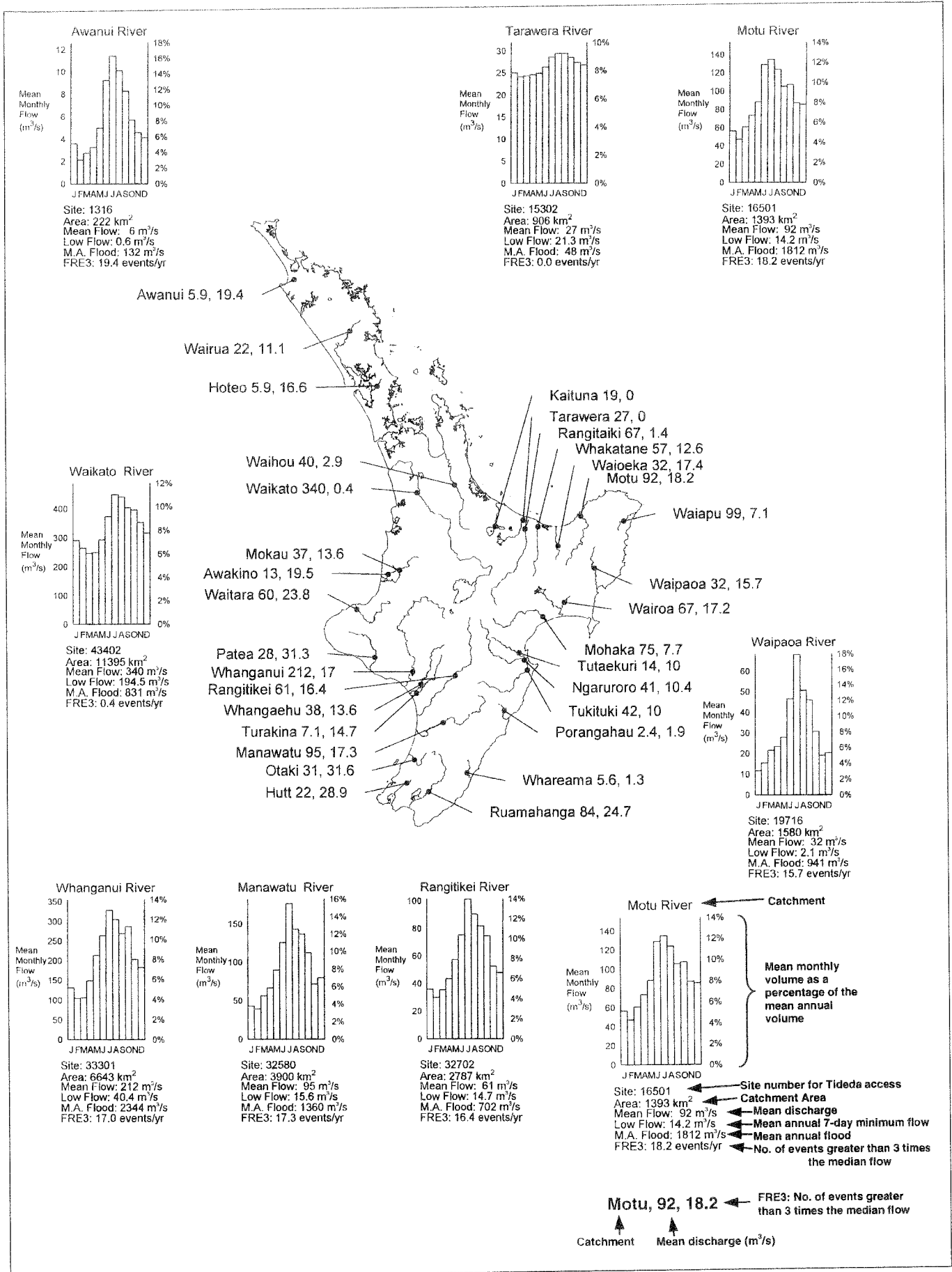


Figure 7.4 Indices of flow regime for representative catchments, North Island (from National Institute of Water and Atmospheric Research and Regional Council archives).

Figures 7.3 and 7.4 show the mean flows and FRE3 of 67 of the country's larger or more economically important rivers. Some large South Island West Coast rivers and large east coast rivers such as the Clarence River in Marlborough are not included because their flows have not been reliably measured. The mean flows shown are the natural river flows, i.e., they are the flows that would be expected if there were no man-made diversion of flow from one catchment to another.

Monthly flow histograms

Month-to-month variations in river flow (Figs. 7.3 and 7.4) primarily reflect the seasonal distribution of rainfall and snowfall in New Zealand. The winter rainfall peaks in the north are reflected in the flows of the Awanui, Motu, Whanganui and Manawatu Rivers, while the more even distribution of rainfall in central New Zealand is illustrated by the flow of the Buller River.

The monthly flows of the Rakaia, Kawarau and Ahuriri Rivers are characteristic of alpine snow-fed rivers, where winter precipitation is held in the snow pack and released in the spring and summer thaw. However, the high spring and summer flows are also a response to rainfall from the northwest winds that prevail then. The flow pattern of the Taramakau River, which is typical of the many short, steep and large rivers draining the Southern Alps to the west, shows the same traits but is less influenced by snow melt.

In the Volcanic Plateau of the central North Island, rainfall percolates through the fractured pumice into the groundwater system and is released evenly by spring-fed streams, as in the Tarawera River (Fig. 7.4). Many of the rivers with headwaters in the central North Island show some influences of their pumice cover. A comparison of the Waipaoa River, which drains Tertiary sedimentary rocks, with the Rangitikei or Whanganui River shows how much the flows are moderated in the latter rivers.

Lake storage and the even release of water for hydroelectric power generation are responsible for the uniform monthly flow of the Clutha at Clyde (Fig. 7.3). Its monthly flows vary much less than those of its tributary the Kawarau, even though Kawarau monthly flows are somewhat moderated by uncontrolled storage in Lake Wakatipu. The Buller River also shows the smoothing influence of Lakes Rotoiti and Rotorua on its monthly flow fluctuations.

The Hakataramea River is typical of foothills-fed east coast rivers, with high flows in late winter and spring, and low flows in summer and autumn. These reflect the generally low east coast rainfall, and dry summers when the soil dries out (Fig. 7.3). Only when soil moisture is fully replenished by a combination of winter rainfall and low rates of evapotranspiration, is there sufficient rainfall to increase flow substantially.

Specific discharge

Specific discharge (also known as specific yield) is the flow per unit of catchment area, usually expressed in litres per second per square kilometre ($l/s/km^2$). It allows the flows from catchments of different sizes to be directly compared. It can also be converted to depth of runoff in millimetres, and is therefore easily compared with rainfall. Further insights into a river's regime may be made by examining its specific discharge for various parts of its flow, such as low flow, mean flow or flood flow.

The mean specific discharge strongly reflects catchment rainfall and evapotranspiration (ET). Figures 7.3 and 7.4 show data for the major rivers of New Zealand, and flow rates for 95 smaller river sites can be found in Close and Davies-Colley (1990). The range for the North Island rivers shown in Figure 7.4 is 8 to $101 l/s/km^2$ (290 mm and 3190 mm) for the Porangahau and Otaki Rivers respectively. However, most of the catchments yield about $34 l/s/km^2$ (1070 mm) reflecting the relatively even distribution of rainfall over the North Island. The range for South Island sites is much wider, with the Whataroa River yielding a high $310 l/s/km^2$ (9840 mm) and the Hakataramea River only $6.7 l/s/km^2$ (210 mm). The Whataroa catchment runoff of nearly 10 metres is by no means uncommon, as the Hokitika at Colliers Creek (catchment area $352 km^2$) yields 8700 mm of runoff. To this, estimated evapotranspiration of 600–700 mm per year must be added (Finkelstein 1961), indicating an annual rainfall of about 9500 mm over the whole catchment.

Specific mean annual flood flows (that is, the average of the annual peak flows, per unit area) reflect storm rainfall intensities, which normally increase with annual rainfall. However, the highest rates for the North Island (Fig. 7.4) are for rivers towards the north, which are subject to storms originating from tropical cyclones. These include rivers such as the Awanui, Motu and Waipaoa, which have specific mean annual floods of 630, 1140 and $690 l/s/km^2$ respectively. Many of the other North Island rivers have specific mean annual floods of about $300 l/s/km^2$. The Waikato and Tarawera Rivers have very low specific mean annual floods of 60–70 $l/s/km^2$, because lake and ground water storage in the pumice of the central volcanic plateau has a strong damping effect on their flood regimes. The annual maximum floods and related statistics for 343 rivers nationwide can be found in McKerchar and Pearson (1989).

Low flows are determined by the recency of rainfall, catchment groundwater storage and its rate of outflow (a function of the underlying rocks and lakes), and catchment area. The lowest flow per unit area during a period of 7 consecutive days that could be expected to occur on average every 2 years is called the specific 2-year return

period 7-day low flow. It is a particularly important index for management of instream flows because it represents the extreme low flows that are likely to limit the life-supporting capacity of a waterway. Such flows vary from about 500 l/s/km² in the Taramakau River to as little as 1 l/s/km² in the Hakataramea River; they are primarily a function of annual rainfall and geology. Catchments with small low flows also tend to have long periods with low flows. Hutchinson (1990) lists low flow magnitude and frequency from 428 sites nationwide.

SHAPING FLOW REGIMES

Flow regimes principally reflect climate (precipitation and evapotranspiration), geology, vegetation cover, and human activity such as flow diversion for hydroelectricity generation or irrigation.

Climatic influences

Rainfall and evapotranspiration distribution

The major climatic factors influencing flow regimes are how often and how hard it rains, and how rapidly moisture is returned directly to the atmosphere by evapotranspiration. Examination of the annual pattern of rainfall and evapotranspiration goes a long way towards explaining why a particular river has a particular regime.

New Zealand's rainfall pattern results from its long narrow shape, steep topography and isolated island position. The country's mountain backbone lies directly across the path of the eastward-moving anticyclones and low-pressure troughs that are characteristics of the "Roaring Forties". The passage of these weather systems results in a high and regular rainfall over much of the country, although some places get much more rain, more often, than others (Chapter 2). Mean annual rainfall varies from as little as 300 mm in a small area of Central Otago to over 10,000 mm in a long narrow strip to the west of the crest of the Southern Alps (Griffiths and McSaveney 1983; Henderson and Thompson 1999). However, over most of the country it is between 600 and 1500 mm. Some areas with average rainfall under 600 mm are found in the South Island to the east of the main ranges. North Island mountains are lower, and annual rainfall is more uniform. Much of the island receives about 1500 mm, and the dry areas (central and southern Hawke Bay, Wairarapa and Manawatu) about 700 mm (See Chapter 4).

Evapotranspiration varies less from place to place; annual rates are on the order of 500–1000 mm. Evapotranspiration is therefore relatively small in comparison with precipitation in the Southern Alps, but large in comparison with precipitation in the drier east of the country. In summer, in particular, potential evapotranspiration can exceed precipitation for several months.

Reflecting these differences in the water balance—the difference between precipitation and evapotranspiration—rivers draining westwards from the Southern Alps have annual runoffs of the order of 5000 mm, whereas those draining the Wairarapa have annual runoffs of the order of 300 mm.

The greatest seasonal contrast in rainfall occurs in Northland, East Cape and the Wairarapa, where winter rain is almost double that of summer. The resultant effect on stream flows is evident from the patterns of monthly flow of the Awanui, Motu and Waipaoa River (Fig. 7.4). This predominance of winter rainfall diminishes southwards (Chapter 2), although it is still discernible over the northern part of the South Island and its effect can be seen in the flow of the Buller River (Figs. 7.2A and 7.3). Further south, winter is the season with lowest precipitation, and inland areas receive most rainfall in summer, from convective showers. The effect of low winter precipitation can be seen in the Taramakau River (Fig. 7.3), but the higher summer rainfall of the coastal catchment is more commonly due to northwest rainfall than convective showers. The highest variations in seasonal rainfall from year to year are in areas to the east of the mountain ranges. Here very dry conditions may develop in late summer and autumn, particularly in Hawke's Bay, Canterbury and North Otago. The Hakataramea River monthly flows (Fig. 7.3) and the Whareama River hydrograph (Fig. 7.2D) illustrate the effect of these high seasonal variations in rainfall, and the high rates of evapotranspiration during the summer months.

Usually it rains hardest where it rains the most (Tomlinson 1980; Whitehouse 1985). The highest 24-hour rainfall on record is 758 mm, which fell at Prices Flat in the Hokitika catchment, in the high rainfall zone of the western Southern Alps (Henderson and Thompson 1999). A storage rain gauge at Alex Knob on the south bank of the Waiho River, Fox Glacier, recorded 1800 mm in 3 days in March 1982. If rainfall at Alex Knob has a similar intensity pattern to that at neighbouring recording rain gauges, and we think it does, then about 1350 mm would have fallen in 24 hours (Thompson and McKerchar 1992). Such high and intense rainfall produces frequent flashy floods imposed upon a sustained baseflow, as is evident in the hydrograph of the Ahaura River (Fig. 7.2B).

The Gisborne and Auckland regions, which have considerably lower annual rainfalls than the Southern Alps, can also receive heavy daily falls of up to 140 mm. In contrast, the plains of Otago and Southland rarely receive daily falls greater than 100 mm and 80 mm respectively (Thompson 1987, 2002).

El Niño, ENSO, IPO

The eastward passage of anticyclones and low-pressure troughs across New Zealand, the weather pattern

responsible for the sequence of rainstorms and dry periods normally experienced, is influenced by the state of the El Niño-Southern Oscillation (ENSO) (Chapter 2). During an El Niño phase, sea surface temperatures around New Zealand are lower and the westerly winds are stronger, resulting in greater rainfall in the south and west of New Zealand and lower rainfall in the northeast. La Niña conditions tend to give warmer sea surface temperatures around New Zealand and lead to fewer westerly winds. This leads to less rainfall in the south and west and more in the northeast (Mullan 1995).

The Interdecadal Pacific Oscillation (IPO) is a measure of the phase of a decadal-scale oscillation of temperatures within the Pacific Ocean. Shifts in the IPO alter the frequency of occurrence and intensity of El Niño and La Niña phases of ENSO. This oscillation shifted phase in the mid-1940s and again in 1977/1978 (Salinger *et al.* 2001) and may have shifted phase again in 1999 (McKerchar and Henderson 2003). McKerchar and Henderson (2003) investigated whether the IPO affected high and low flows in New Zealand. They found less severe floods occurred between 1978 and 1999 in the Bay of Plenty region and more severe floods occurred in the south and west of the South Island. No consistent changes in flood frequency were found elsewhere in New Zealand. They also found, for the same period, that low flows have generally increased in the South Island, particularly in the south, but not in the North Island. They note that, contrary to previous assumptions, some hydrology statistics are not stationary, and instead change at timescales of 20–40 years. The changes they found in some parts of New Zealand are relatively large and need to be taken into account when planning flood structures such as spillways and stop-banks, or abstractions for water supply or irrigation.

Geological influences

Some types of rock transmit water much more readily than others—i.e., their transmissivity, defined as the rate at which water moves through the ground for a given water table gradient, is higher. Similarly, some types of rock store groundwater in greater volumes than others, because their porosity is greater. Hence, the type of rock, or the lithology, in a catchment controls the way in which rainfall passes through the catchment to the river. For example, Tertiary mudstones, shales and siltstones have low transmissivity and little storage, and tend to produce flow regimes that have flashy floods, steep recessions, and low base flows. Rocks of this type occur in the Whareama River catchment, in the Wairarapa (Fig. 7.2D). Catchments with high infiltration, transmissivity and water storage tend to have small floods with slowly receding flow, and high, persistent baseflows. Examples are the Maryburn River in

the McKenzie Basin, which has deep permeable gravels at the surface, or the Rangitaiki River (Fig. 7.4), which drains an area with a deep pumice cover.

In his study of summer low flows in Northland, Waugh (1970) found that fissured basaltic lava absorbed rainfall and released it slowly, thus sustaining low flows. Areas with other rock types such as Cretaceous shale and sandstone were less absorbent, and their streams had lower low flows. A study of water resources of the Nelson area (Scarf 1972) showed that the rivers draining from the marbles of the Mt Arthur Range had substantial low flows, some issuing from caves (e.g., Riwaka River) and springs (e.g., Pupu Springs near Takaka). This was in contrast to the very low flows of streams draining areas covered by the impervious Moutere outwash gravels, where streams commonly dry up in summer. Although rainfall distribution plays a part, catchment geology has a major influence on Nelson flow regimes.

Lake storage has an effect on flow regimes that is similar to that of rocks with high storage characteristics. For example, the Buller River at the outlet of Lake Rotoiti (Figure 7.2A) shows flow peaks that are much more subdued than those of the Ahaura River, because of the damping effect of the lake.

Human influences

Hydroelectric power

In many New Zealand rivers the natural flow regime has been altered, particularly by hydroelectric power projects or changes in land use (Chapters 33, 37). Hydroelectric development has substantially affected the Waiau (Southland), Whanganui, Waikato, Clutha and Waitaki River systems. The mean flow of the Waiau River has been reduced from 561 to 157 m³/s by the 404 m³/s of flow that has been diverted to Doubtful Sound via the Manapouri Power Station. The flow regime of the Waiau River at Tuatapere consequently has been affected by a reduction in the full range of flows (Fig. 7.5).

Some of the headwater streams of the Whanganui River have been diverted into the top of the Waikato system—much of their low and median flows are now redirected, leaving only small residual flows and flood flows. However, the normal regime of the Whanganui is partially restored as undiverted tributaries add to its flow. Hydrographs of the remaining flow and simulation of the natural flow of the Whanganui River at Te Maire (Fig. 7.6) illustrate that the low flow part of the flow regime is most affected by the diversion, but to protect important features of the instream environment, between December and May, diversions must stop when flow is less than the mean annual low flow. The loss of 18 m³/s from the Whanganui River system is the Waikato River's gain. The Waikato also gains 14 m³/s from the Rangitikei River via the Moawhango

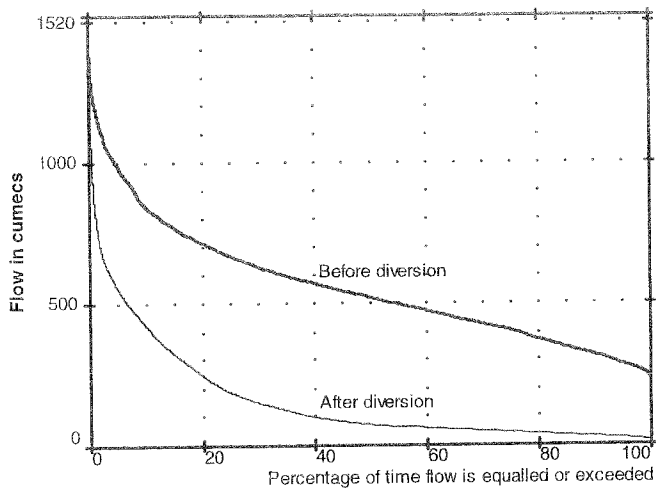


Figure 7.5 Flow duration curves of the flow in the Waiau River at Tuatapere before and after diversion via Lake Manapouri to Deep Cove.

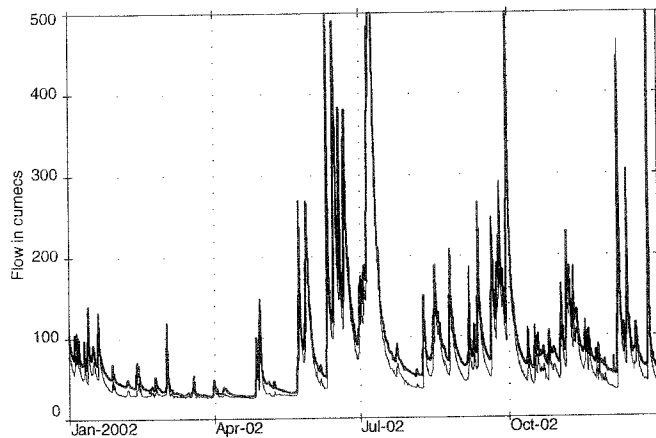


Figure 7.6 Flow recorded in the Whanganui River at Te Maire (fine line) and flow that would have occurred without the diversion (thick line). To protect the in-stream environment, diversions are reduced between December and May, and cease altogether when the flow is less than the mean annual low flow (e.g., as illustrated in March and April).

Tunnel. It is further modified by controlled outflows from Lake Taupo and eight hydroelectric power stations further downstream. The net effect has been to reduce flood flows and increase low flows in the Waikato River.

At the Roxburgh hydroelectric power station, the release of extra water from the lake to meet peak electricity demands produces a daily flood wave on the Clutha River (Fig. 7.7). It has been suggested (Otago Catchment Board 1986) that this, combined with the tidal and wave pattern at the coast, has resulted in periodic shifts of the river mouth, leading to regular flooding in the Lower Clutha delta. The monthly Clutha flows (Fig. 7.3) mask the daily

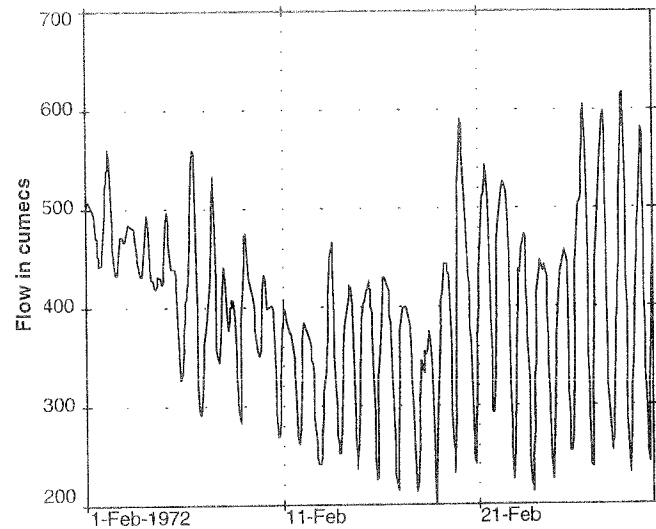


Figure 7.7 Daily flood waves in the Clutha River at Balclutha, caused by the response of Roxburgh hydroelectric power station to varying demand for power.

fluctuations. The monthly regime is even because of the moderating effects of the large lakes Wakatipu and Wanaka and the manipulation of water storage in Lake Hawea.

Hydroelectric storage dams and diversion canals in the Waitaki Catchment have made dramatic changes to the flow regimes of its large rivers. The Ohau River previously had a mean flow of $80 \text{ m}^3/\text{s}$, but now has either no flow or occasional flood flows. However, agreement has been reached on releasing a residual flow of $10 \text{ m}^3/\text{s}$ in exchange for being able to operate Lake Ohau over a larger range of lake levels. The Pukaki River now has no flow and there are only occasional flood flows in the Tekapo River. The Tekapo River further downstream is now much clearer than before and conditions for trout have been enhanced (Chapter 43, Teirney *et al.* 1982). Flood flows have been reduced and low flows increased in the lower Waitaki River. However, the Roxburgh and Waitaki dams have prevented chinook salmon from returning to their previous spawning grounds, and the salmon runs are reported to have been substantially reduced (Teirney 1980). More biological effects of changes in flow regime downstream of New Zealand hydroelectric dams are described in Chapter 43.

Irrigation

Irrigation is the largest consumptive use of surface water in New Zealand (Lincoln Environmental 2000). Irrigation occurs mainly in the drier east of the country. Irrigation abstraction changes flow regimes by reducing flows from spring through to autumn. If irrigation rates exceed the water-holding capacity of the soil, then some irrigation water may flow into streams as “bywash”, increasing the flows in the receiving stream during the irrigation season,

and possibly also carrying nutrients and contaminants such as faecal coliform bacteria to the stream. Resource consents usually limit abstractions to leave a residual flow, variously set as the mean annual low flow or the 5-year low flow, or some other ecologically relevant flow. Often this leaves the stream with a relatively constant flow throughout most of the summer, but some resource consents specify a sharing rule to maintain a degree of flow variability within the range affected by abstraction. For example, the Rangitata diversion race takes up to 32 m³/s from the Rangitata River, under a 1:1 sharing rule, for irrigation from September to April and for hydro-electricity generation during the balance of the year. The abstraction does not alter flow variability much, but low flows are reduced to only half the natural flow for significant periods of time (Fig. 7.8).

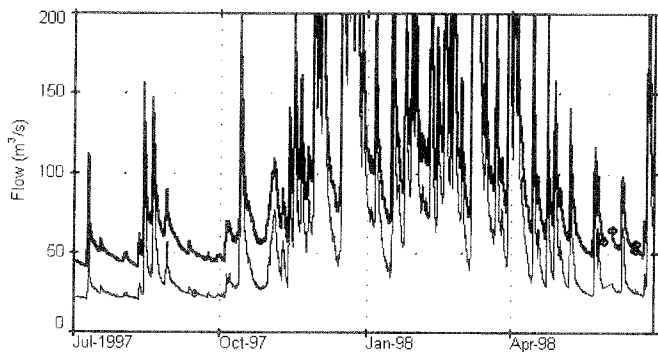


Figure 7.8 Flow recorded in the Rangitata River in a typical year (thick line). The fine line shows the residual flow in the river after abstraction of water for irrigation in summer and hydropower in winter.

Changes in land use

Both Maori and European settlers in New Zealand have influenced river flow regimes by making large-scale modifications to the vegetation. The moa hunters effectively converted large areas of bush to tussock country by burning. Europeans in turn have converted tussock and bush country to pastoral farms, and bush and scrub country to pine plantations. Chapter 33 details the effects of some of these changes.

When land is cleared, runoff from the land increases markedly, thus increasing floods and low flows. When pine plantations replace pasture, flood peaks may decrease by 80% and annual yields and low flows can halve (Fig. 7.1); the opposite happens when pasture replaces pines. These changes occur primarily because of differences in interception of rainfall and evapotranspiration by different types of vegetation. Interception is the rain which falls on vegetation and which evaporates before reaching the ground. Thus it is not available for transpiration by the

plants or for runoff. Interception in tall vegetation like trees or scrub can account for 20–40% of rainfall.

ESTIMATING FLOW REGIMES OF UNGAUGED CATCHMENTS

Streamflows have been measured for only a limited number of rivers and streams in New Zealand. It is often necessary, however, to estimate the magnitude of floods and low flows for catchments that do not have a streamflow record. To do this, hydrologists have tried to define regions in which the basins are sufficiently alike to apply the measured relationships between rainfall and runoff from gauged basins to ungauged basins within that region. The high variability of geology, topography, and especially rainfall in New Zealand has made this a difficult task.

Regionalisation

Toebe and Palmer (1969) divided New Zealand into 90 regions based on geology and climate, and proposed that representative basins monitoring rainfall and runoff be established in each region. Fifty-three regional basins (Duncan 1987) were instrumented and, together with those rivers instrumented for flood warning and power or irrigation development, served as the basis for flow estimates.

In the North Island, where regional geology and soils vary more than in the South Island, cluster analysis suggests that useful regions cannot be easily identified (Mosley 1981).

Beable and McKerchar (1982) proposed regions for the estimation of flood size and frequencies based on regional equations. They defined 7 and 6 regions, respectively, in the North Island and 6 and 3 regions in the South Island. While this was a useful exercise, difficulties arose at regional boundaries, where flood estimates could vary widely depending on which regional equation was adopted.

A more recent study, using a larger data set and longer records, demonstrated that contour maps of mean annual floods and 100-year floods could be drawn for the whole country (McKerchar and Pearson 1989). This shows that flow regimes vary relatively smoothly across New Zealand, rather than abruptly changing at well-defined regional boundaries.

Equations for estimating the low flow of ungauged catchments, based on 11 regions nationwide, were proposed by Hutchinson (1990). Many regional equations were quite similar, with the differences justified by providing more precise estimates. Paradoxically, the Southern Alps region and North Island central volcanic plateau, regions of quite different geology and rainfall regime, had similar equations for the estimation of low

flows. The regular Southern Alps rainfall and the porous volcanic plateau bedrock both have the effect of sustaining low flows.

Pearson (1995) used annual minimum low-flow series from nearly 500 catchments nationwide to draw contour maps of specific mean annual 7-day low flow. He also used catchment characteristics (e.g., area, annual rainfall, vegetation, elevation, % bare land, slope and hydrogeology index) from a subset of sites with longer records to predict mean annual 7-day low flows. These two methods were adequate to predict regional variations in low flows, but the catchment characteristics method was biased for catchments with low specific discharges.

Flow variability was the basis for the classification of 130 river sites by Jowett and Duncan (1990). They did not attempt to map hydrologic regions but they did identify six groups. Rivers with the lowest flow variability were associated with the large South Island montane lakes because the lakes attenuate the flows, and with the volcanic plateau of the North Island, where precipitation is absorbed by the porous pumice lithologies and emerges evenly in springs. The next group was also in the central portions of the North and South Islands where regular, but not constant precipitation, resulted in a relatively constant base flow. The group with the greatest flow variability was on the east coast of both islands, where rainfall is irregular and low in relation to evapotranspiration. Here summer flows are very low, and winter flows are quite high, as the underlying geology has low transmissivity and little storage, so once the soil is saturated a large proportion of the precipitation runs off. An intermediate group included rivers around Mt Taranaki, the Tararua Ranges, and in the Nelson region.

Because of their links with biological communities, rock type (soft and hard sediments, igneous rocks, volcanic ash), flow variability, and water quality (mainly conductivity) were the basis of defining regional groups of rivers to form riverine "ecoregions" in a study by Biggs *et al.* (1990). Five principal riverine ecoregions were distinguished. Particularly distinctive were the hydrological, geological and water quality conditions of the central North Island volcanic plateau and the eastern, Hawke Bay-Poverty Bay region of the North Island. Other regions were the Tararua Ranges, the remainder of the North Island comprising Taranaki, Waikato and Northland, and the South Island. Defining such regions could have considerable

benefits for establishing river management goals, especially where unmanageable factors such as catchment geology may cause naturally poor water quality compared with other regions. Many of these ideas have been incorporated into the River Environment Classification discussed below.

In summarising regional hydrological regimes, Mosley (1981) stated that in the South Island, climatic regime, as modified by topography, appears to be the major influence. Much of the South Island is underlain by relatively impermeable rocks, and has steep topography. They are less important as sources of a variation in flow regime than climate, which is spatially highly variable.

The North Island is more complex, with variations in flow regime influenced by climate (e.g., the Northland sites), lithology and soils (e.g., pumice area sites), and topography (e.g., sites draining the Tararua Range and Mt Taranaki).

River Environment Classification

The River Environment Classification (REC) uses a six-step hierarchical approach to describe the main causes of variation in river environments. At the top of the hierarchy is climate, and then source of flow (SOF). Sources of flow are determined by rules applied to the catchment upstream of each river reach—e.g., glacial mountain sources of flow applies to catchments with >2% of catchment area with permanent snow. Catchments with Glacial Mountain sources of flow have low flows in winter and high flows in spring and summer as the snow melts (Table 7.1). Other

Table 7.1 Hydrological variables for different sources of flow.

CVW = cold very wet (mean annual temperature <12°C, precipitation >2200 mm/a); CW = cold wet (precipitation 1200 to 2200 mm/a); CD = cold dry (precipitation <1200 mm/a); WVV = warm very wet (mean annual temperature >12°C); WW = warm wet; WD = warm dry; MaxF = mean annual 7-day high flow/median; Tmin, Tmax = month with the lowest and highest flows where January = 1.

Source of flow	FRE3	CV	MaxF	Tmin	Tmax
CVW, Glacial Mountain	15	1.0	35	6.4	11.8
CVW, Mountain	22	1.3	49	4.8	10.7
CVW, Hill	27	1.7	87	3.08	8.6
CVW, Lowland	27	1.9	115	1.8	7.8
CVW, Lake	2	0.6	8	2.7	10.7
CW, Mountain	13	1.1	48	3.3	9.9
CW, Hill	15	1.5	90	2.4	8.3
CW, Lowland	18	1.9	163	1.9	7.3
CW, Lake	2	0.5	9	2.4	9.6
CD, Hill	11	2.4	366	2.7	8.3
CD, Lowland	14	2.1	165	2.4	6.8
WVV, Lowland	19	1.8	151	2	7.0
WW, Lowland	18	2.3	230	1.2	7.4
WD, Lowland	19	3.2	471	1.9	7.4

sources of flow are Mountain, Hill, Low elevation and Lake—each one has a characteristic flow regime with different seasonal timing and amplitude of flow regimes, e.g., lake sources of flows have low FRE3, coefficient of variation (CV) of flow and MaxF (Table 7.1) (Snelder and Biggs 2002). Flow regimes at 335 flow sites nation-wide were described by 13 flow variables. Data were analysed to determine the mean values for 14 climate/source of flow classes. Statistical tests were used to see how well the sites in each class clustered and to see if there was clear separation between the classes. The River Environment Classification was compared with a climate classification, Hutchinson's (1990) classification and New Zealand water management regions. The River Environment Classification was stronger than any of the others and was able to predict the general pattern of a flow regime. However, it was not strong enough to be used to reliably predict the flow regime characteristics of a specific site (Snelder and Biggs 2002).

Hydrological models

The flow regimes of unmodified rivers can be estimated with hydrological models, using information on climate, vegetation, soils and topography. If human influences are present (e.g., irrigation, hydropower development), then the models will also need to include these factors. The choice of modelling approach depends on the amount of data available, and the resources available. For example, summary information on annual and seasonal climate (rainfall, temperature, potential evaporation), vegetation type and soil properties, can be used to estimate annual and seasonal flows. These estimates use very simple models of water balance for the plant canopy, soil water, and shallow groundwater (e.g., Woods 2003).

If more detailed information on catchment and climate properties is available, then a detailed catchment simulation model such as Topnet (Chapter 9) can be used to produce modelled daily or even hourly flow hydrographs for ungauged catchments. Figure 7.9 shows the application of this technique to the Waikohu River, a small (26 km²) tributary of the Waipaoa River. The model estimates are broadly similar to the measured values, which were not used in the development of the model. Models can always be adjusted to improve the fit to observed flow data: Figure 7.9 shows what level of accuracy might be achieved by this method if no measured data are available.

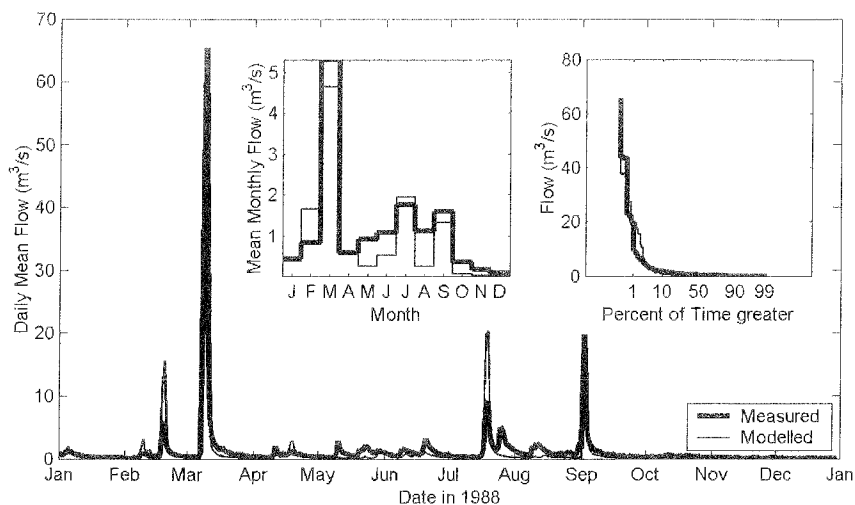


Figure 7.9 Measured and modelled daily flows for the Waikohu River using the TOPNET model. Insets show measured and modelled monthly flows and flow duration curves.

SUMMARY

The flow regimes of New Zealand rivers are determined primarily by the country's abundant rainfall, steep topography and long narrow shape, which generally give rise to short, swift gravel- or cobble-bedded streams. Local weather conditions, geology, lakes and hydro-electricity schemes further modify flow regimes.

The mean annual flood, the number of floods above a threshold (FRE3), the mean flow, and the mean annual 7-day low flow are good indices that together describe a flow regime. Values of FRE3 vary from 0 for spring-fed streams to over 20 for some West Coast streams. Specific mean flows, reflecting annual rainfall, vary from about 280 l/s/km² for South Island West Coast rivers to less than 7 l/s/km² for small east coast rivers. Specific mean annual flood flows, which are influenced by rainfall intensity, catchment storage (lakes, lithology) and catchment area, vary from over 5000 l/s/km² to less than 60 l/s/km². Seven-day minimum flows with return periods of 2 years, determined primarily by rainfall regime and catchment storage, range from over 50 l/s/km² to less than 1 l/s/km².

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Chapter 6

Glaciers – perennial snow and ice

Trevor Chinn

INTRODUCTION

New Zealand has a large amount of fresh water stored in glaciers as perennial snow and ice. There are some 3,140 glaciers, covering an area of 1,160 km², with an estimated volume of 53.3 km³. They are distributed from Mt Ruapehu, at 39°15' S in the North Island, to southern Fiordland at 45°57' S (Fig. 6.1). A glacier is basically a mass of snow and ice that has accumulated above the permanent snowline, where snow gain exceeds melt. The ice forms from the compressed, residual snow packs left over at the end of many summers, and creeps downhill to

be melted away at lower, warmer altitudes. For the New Zealand Glacier Inventory, a practical working definition was required to eliminate temporary seasonal snow that persists for more than one year, and to include those debris-covered ice bodies commonly referred to as rock glaciers. The Inventory counted:

“Those ice bodies of 1 ha or greater in area which have remained in existence during the most negative balance years (warmest years) over the past two decades”.

This definition was used in the early 1980s to make an inventory of every glacier in New Zealand (Chinn 2001).

The Inventory, which followed international standards (IAHS-AISH 1980) included about 20 common attributes measured for each glacier, the most important being dimensions, altitudes, areas and estimated volumes. Glaciers respond to climatic changes, and as climate change monitoring is one of the foremost reasons for compiling the Inventory, it was necessary to establish a baseline time against which to compare changes to the glaciers. A 1978 set of oblique aerial photographs gave the best coverage of the glaciers, and the inventory has been mapped for autumn 1978. One exception is the North Island glaciers on Mount Ruapehu, where no suitable photographs were available until a glacier flight was made in 1988.

SUMMARY OF NEW ZEALAND GLACIER DATA

The distribution of the 3,140 individual glaciers of New Zealand is shown on the map of Figure 6.1. They are plotted as single points at the centroid of each individual glacier and *not* as glacier outlines, as the majority of the small glaciers would not be visible at this scale. In the North Island, 18 glaciers were recognised, all on Mt. Ruapehu. In the northern South Island, where 35 glaciers have been identified, peaks between Nelson Lakes

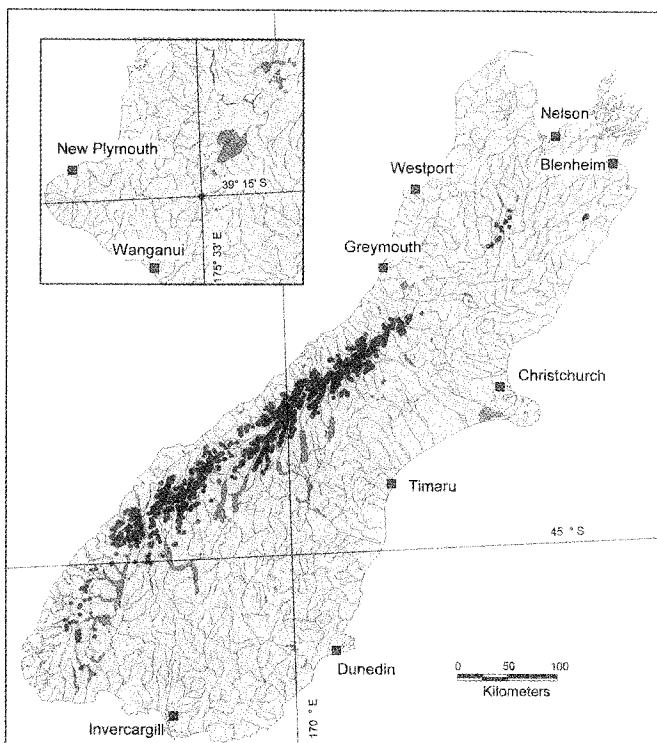


Figure 6.1 Distribution of the glaciers of New Zealand, plotted as a filled circle for each individual glacier centroid (not as glacier outlines). Mt Ruapehu inset.

and Lewis Pass barely intercept the snowline, while the 9 glaciers of the Inland Kaikoura Range are in an arid zone and contain some of the finest examples of active rock glaciers in New Zealand. Southward, between Lewis Pass and the headwaters of the Taramakau and Waimakariri Rivers, none of the mountains intercept the permanent snowline, so there are no glaciers. The area between Arthur's Pass and Hollyford Valley contains the bulk of glacier ice (Fig. 6.2). South of Hollyford Valley, in the lower subdued topography of Fiordland, the mountain summits again only barely reach the regional snowline, so there are only small scattered glaciers. The southernmost glacier lies on Caroline Peak above Lake Hauroko.

Density of glaciation

The visually obvious changes in the amount of ice cover or degree of glacierisation of the various river basins cannot be expressed as a percentage of basin area because this value depends on where one arbitrarily determines the position of the bottom of the catchment. However, distance along the Main Divide is independent of catchment areas, and



Figure 6.2 Northeast view across the Alps of south Westland showing the multitude of glaciers throughout the central Southern Alps. Photo: T. Chinn

Figure 6.3 shows the intensity of glacierisation as ha per km southward along the Divide, together with a topographic profile. The greatest areas of ice cover are on the highest ranges in the South Canterbury, Mt. Cook and Mt. Aspiring areas.

Glacier areas and volumes

The glaciers show an enormous variation in area, from 1 ha to nearly 10,000 ha for the outstandingly large Tasman Glacier, which is over twice the area of the next largest glacier, the Murchison. The nation-wide distribution of glaciers in size interval classes was examined by counting the number of glaciers in intervals of area. Because of the large range in glacier areas, a linear scale is inappropriate, and area is best plotted exponentially (Fig. 6.4). The modal size for the glaciers of New Zealand is in the 4 to 8 ha interval, although by far the greatest area is covered by the few larger glaciers.

Glacier volumes can be estimated by using a crude relationship with area (Fig. 6.5). The modal volume for the glaciers of New Zealand is in the 0.16 to 0.32 Mm³ interval, but the greatest volume by far is held by the few larger glaciers, with nearly 50% of all of New Zealand's ice contained in the ten largest glaciers.

Glacier mean elevations

On small glaciers the mean elevation of the glacier approximates the regional snowline and, along the Southern Alps, snowline elevations descend southward with increasing latitude, from 2500 m on Mt. Ruapehu to 1600 m in Fiordland. Across the Alps snowline elevations are inversely related to total precipitation—snowlines are lower in the west, at about 1600 m, and rise markedly to over 2200 m on far eastern glaciers (Chinn and Whitehouse 1980). Figure 6.6 shows glacier mean elevations plotted on a background of the topographic profile, with Mt. Ruapehu arbitrarily positioned at -100 km. Values are the averages for all glaciers in each river basin, separated into those east and those west of the Main Divide. These data include the noise introduced by a 300 to 320 m elevation difference between north- and south-facing glaciers. A simple curve fitted to all of the mean elevation data shows that the regional snowline over New Zealand descends with increasing latitude, at a slope of 1.09 m/km.

GLACIERS AND CLIMATE

Glacier fluctuations are amongst the clearest signals of climate change because glaciers are highly sensitive, large-scale climate instruments, which ideally should be picked up and weighed once a year. However they do not measure simple temperature or snowfall changes. Air temperature and precipitation, the two factors most commonly

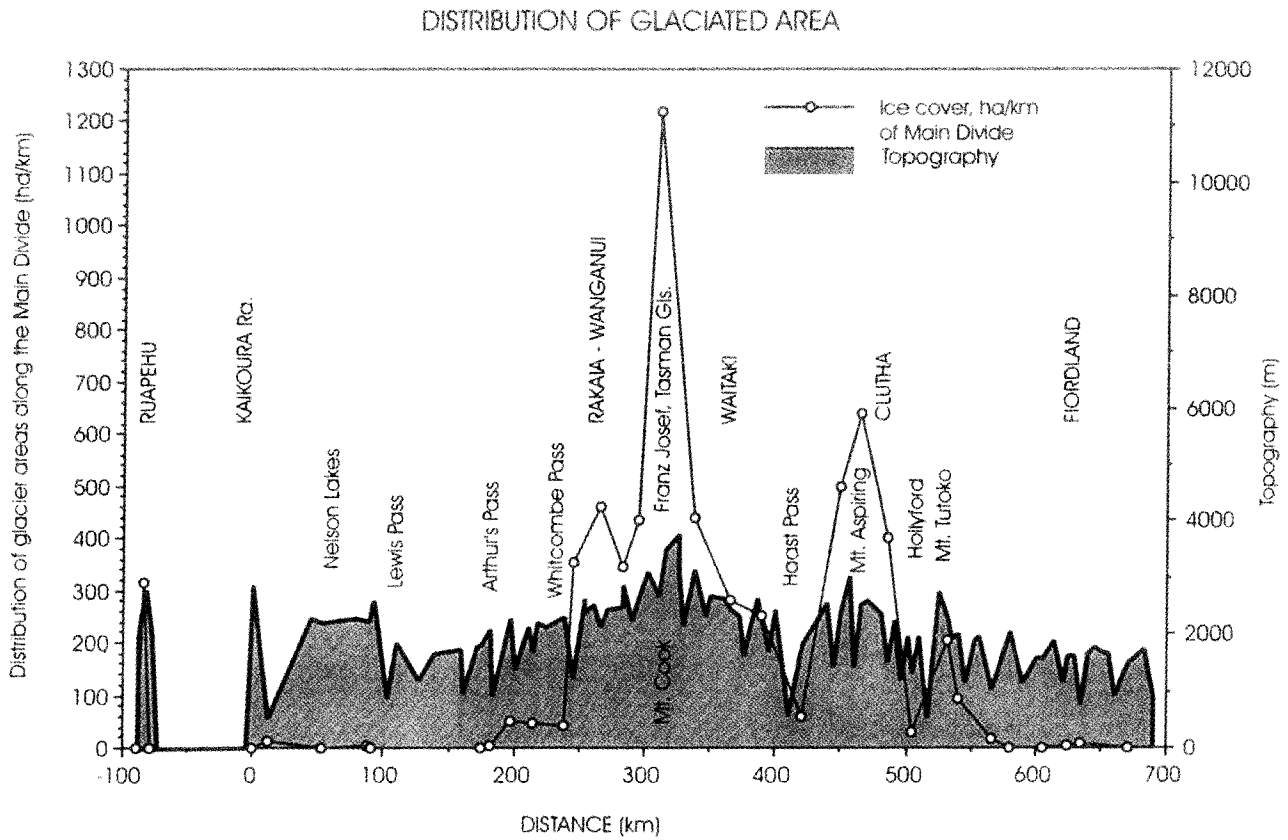


Figure 6.3 Intensity of glacierisation along the length of the Main Divide expressed as ha per km of distance along the Main Divide. Associated maximum and minimum topographic heights are also plotted in profile. Position of Mt. Ruapehu not to scale.

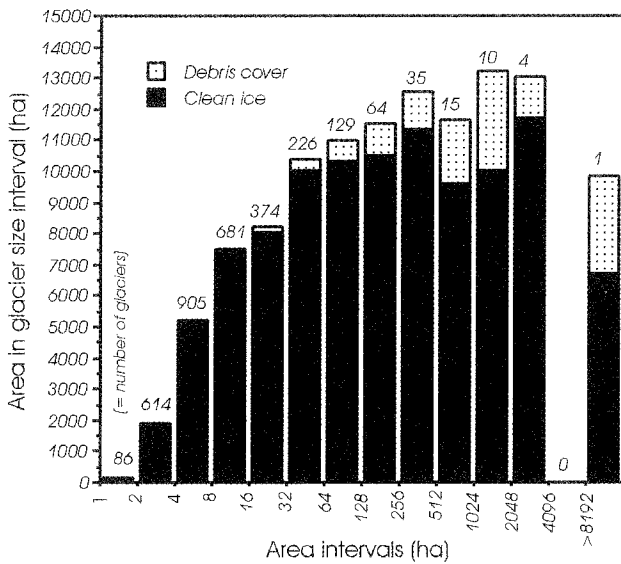


Figure 6.4 Distribution of glacier areas for all New Zealand glaciers. Count of all glaciers in exponential size intervals of 2^n ha plotted as the sum of both clean ice and debris-covered ice within each area interval.

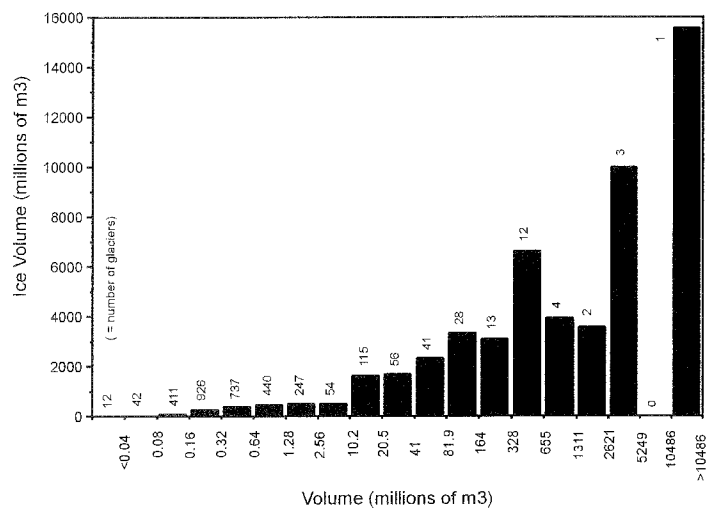


Figure 6.5 Distribution of glacier volumes of all New Zealand glaciers by volume intervals, using an exponential scale of 2^n million m³. The number of glaciers in each interval is indicated.

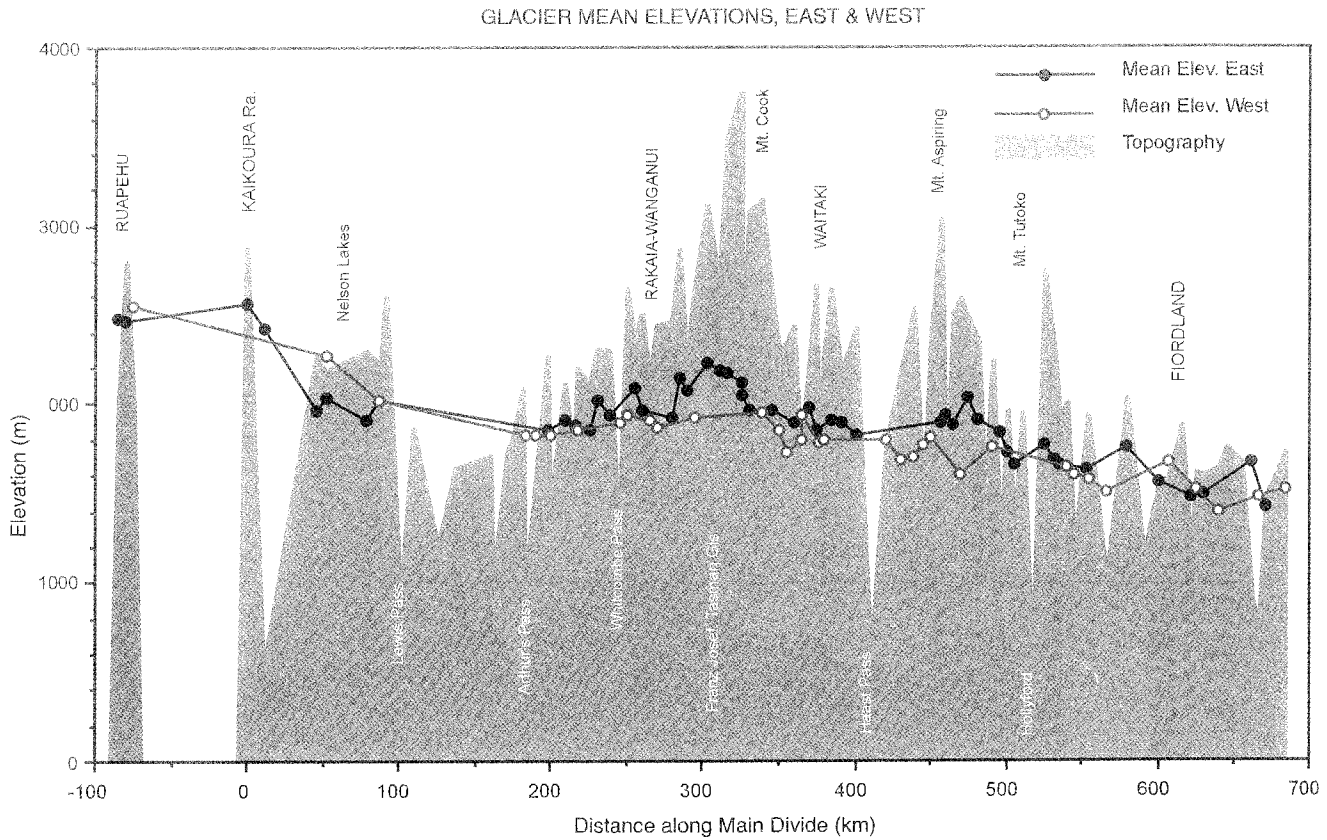


Figure 6.6 Mean glacier elevations for each basin, plotted separately for eastern and western glaciers, over a topographic profile along the Main Divide. Position of Mt. Ruapehu not to scale.

correlated with glacier fluctuations, are only two parts of the complex linkage between climate and glacier fluctuations. This system involves mass balance; snowline altitude; sunshine; wind; humidity and albedo (reflectivity) of the glacier surface, heat exchange at the surface; heat distribution inside the glacier; glacier shape and the mathematics of ice flow. Glacier mass balance is the net gain or loss of snow and ice over the entire glacier surface (accumulation minus ablation) over a given year. This value is the most informative measurement of glacier change. Mass balance values provide a direct, undelayed measurement of the climate of the past year, without the signal being distorted and delayed with the passage of ice through the glacier system.

The equilibrium line or glacier snowline

The snowline at the end of summer separates the accumulation area of the upper glacier, where there is a net gain to the glacier of the winter snow, from the area of loss or ablation on the lower glacier (Fig. 6.7). This line is called the equilibrium line, as it lies where the amount of summer melt equals the amount of winter snow. Its position will move up and down the glacier each year according to the change in mass balance of the glacier. In positive years the “snowline” will be at a lower altitude

than in negative years. The equilibrium line altitude (ELA) is also one of the fundamental glacier variables. On larger glaciers, when they are in equilibrium with the climate, the position of the equilibrium line separates the accumulation area (A_c) from the ablation area (A_b) roughly in a ratio of 2:1 (Fig. 6.8). This ratio, called the accumulation area ratio (AAR), changes drastically with climate changes.

Response time

The response time of a glacier is the time lag from a change in mass balance to the first response at the glacier terminus. Let us assume that the average climate over a number of years suddenly takes a step shift to a different average. This change in climate is registered immediately at the glacier surface by a change in the annual mass-balance. The mass gain takes a few years to be transmitted down through the glacier system as a wave, to produce a change to the terminus position. This lag is the terminus response time, which for most valley glaciers is in the order of 10–50 years. It is analogous to the beginning of the rise in a river after a storm. The glacier then continues to change in size until it gains an equilibrium with the new climate. This volume response time, the period from the initial mass balance change until the glacier has fully adjusted to a new equilibrium size, occurs many years later

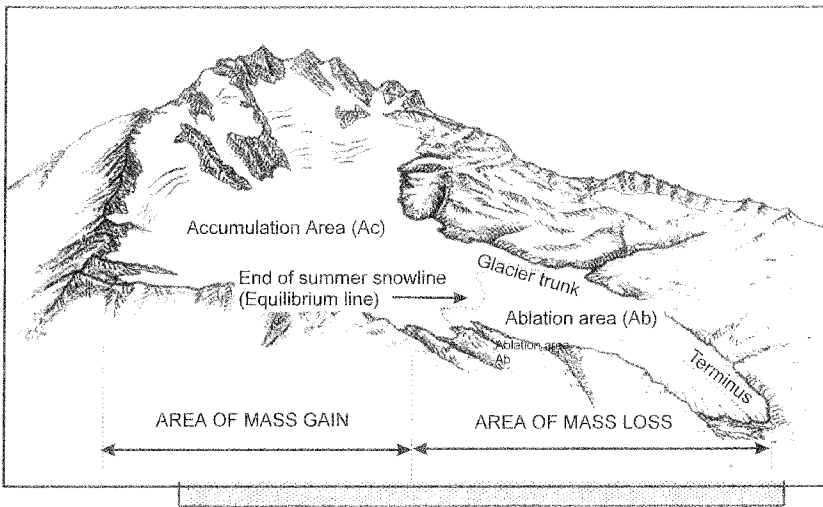


Figure 6.7 Position of the various zones and the equilibrium line on a glacier.

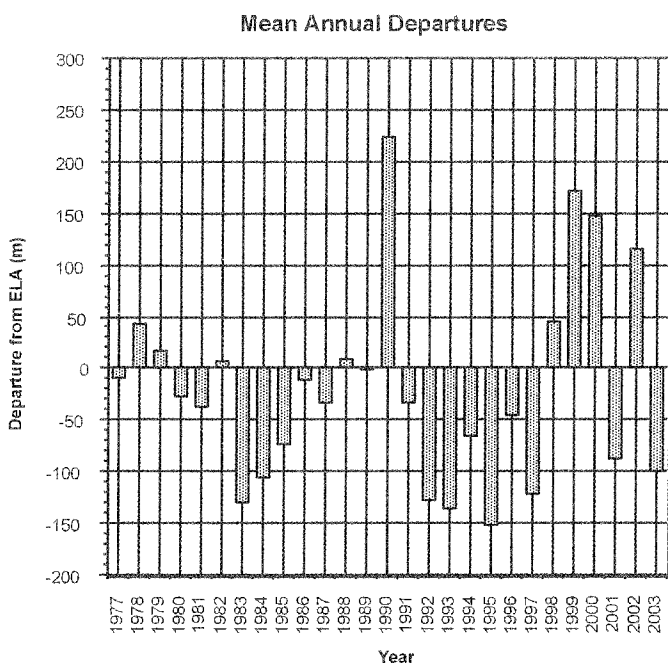


Figure 6.8 Mean annual snowline departures from the average equilibrium position (ELA) of the snowlines for the index glaciers. Positive (high) values indicate less snow, while negative glacier balance and negative (low) values indicate “snowy” years of mass gains to the glaciers.

ice squirts down the valley, like stomping on a tube of toothpaste. Such glaciers, like the Franz Josef and Fox, take only 5 to 8 years for the terminus to respond. Conversely, in large, low-gradient glaciers it may take over a century for the terminus to react to a climate change. The input of additional snow into glaciers like the Tasman and Murchison has an effect similar to that of pouring a jug of water into a bath—there is a minor rise in ice levels, but no shift in the position of the terminus.

HOW ARE GLACIERS MEASURED?

Measuring the glacier’s frontal position is the simplest way of determining changes in glacier length but it is also amongst the least useful. This is because, in the number of years taken to flow down to the terminus, the climate signal is seriously delayed and distorted, destroying the relationship to any particular year. The best way to measure a glacier is to measure the annual mass balance change each year. This requires a field team to make forays onto the glacier throughout both the winter and summer to measure mass changes at a set of stakes—expensive and arduous work. Simpler, cheaper alternatives, such as determining the differences between topographic maps have been tried, but the accuracy of these methods can be unacceptably low.

New Zealand glacier mass balance measurements

Only one significant series of full mass balance measurements has been made in New Zealand. These studies were made at the Ivory Glacier, a small cirque glacier in the Waitaha catchment, Westland, from 1969 to 1975 as part of an IHD Programme of representative basin studies (Anderton and Chinn 1978). Annual mass balance was consistently negative during the study period. The

than the first terminus response and, continuing the river analogy, would be the time from the storm to the peak of the flood (if the storm were to continue with uniform intensity).

Depending on the shape and gradient of the glacier, a given boost of snow to the névés will result in different responses. Short, steep, alpine glaciers have a fast response time of some 10 to 20 years for additional ice to make the terminus advance. The shortest response times are found where the névé is funnelled into a very steep trunk and the

runoff pattern was dominated by rain. The annual mean precipitation was 9630 mm for the period 1971-75. Snow represented about 25% of annual precipitation. Melt contributed 21% of runoff, including 9% contributed by a loss of glacier volume (Anderton and Chinn 1978; Hay and Fitzharris 1988). A glacier balance measurement programme has also been carried out on the Tasman Glacier from 1965 to 1975 (Anderton 1975). This programme measured the mass changes along a longitudinal profile, so the data is for specific points only and not area-averaged to give volumes. These studies were terminated in mid 1970s, mainly because of the cost of maintaining the programmes.

Glacier monitoring using the “snowline” or ELA

Over the past two decades New Zealand glaciers have been monitored by recording the equilibrium line altitude (ELA as defined by Meier 1962) in lieu of mass balance measurements. The series of oblique aerial photographs made to compile the New Zealand Glacier Inventory also picked up the end-of-summer snowline positions, along with many other features such as snout positions, glacial lakes, etc. This record of snowlines was seen to be a useful and cheap way of monitoring the glaciers and annual photographic surveys have been continued at a selected set of “index glaciers” (Chinn 1995). This ongoing programme uses oblique aerial photographs of the end-of-summer snowline positions, or ELA as an alternative, economical method of monitoring glaciers. The surveys currently cover some 48 selected index glaciers distributed along east-west transects throughout the Southern Alps. The data collected have shown that the trend of glacier recession over the last 100 years has reversed, with the glaciers showing inferred positive balances in most years since 1978 (Chinn and Salinger 1999)

The series of ELA values shows that the Southern Alps behaves as a single climatic unit, with a high degree of correlation of ELAs, coupled with little change in the ELA trend surface slopes (Chinn and Salinger 1999; Lamont *et al.* 1999). Significant differences between behaviours of termini are the result of differing response times, not of climates. Atmospheric circulation patterns exert a strong control on the glacier balances, with a dominant SW circulation anomaly in positive balance years and a strengthened NE anomaly in negative years (Fitzharris *et al.* 1997; Clare *et al.* 2002). These circulation anomalies and glacier balance

changes are in harmony with oscillating hemispherical atmospheric systems like the IPO (Interdecadal Pacific Oscillation) and El Niño events (Fitzharris *et al.* 2002). Associated with these events are strong teleconnections between the glacier fluctuations and other climatic events around the Southern Hemisphere (Tyson *et al.* 1997; Clare *et al.* 2002), including New Zealand sea surface temperatures.

A CENTURY OF GLACIAL RECESSION

The last significant world-wide glacial expansion, known as the “Little Ice Age”, was a cool period from the 15th century to the mid 19th century. New Zealand glaciers reached their maximum Little Ice Age extents variously between the 1700s and late 1800s, leaving big grey, unvegetated moraines closest to the present ice. Along with glaciers worldwide, those of New Zealand have been steadily shrinking since the end of the Little Ice Age maximum, mainly in response to climate warming, as there has not been a corresponding change in precipitation. Over this period the Southern Alps has lost an average 26% of its ice-covered area, with the greatest loss, of over 30%, suffered by smaller alpine glaciers (Chinn 1996).

Length changes have been more variable. Cirque and alpine glaciers have lost nearly half of their Little Ice Age lengths, whereas the valley glaciers have lost only one-quarter of their original lengths (Fig. 6.9). The average length losses are roughly proportional to original glacier length, with the greatest losses in the largest glaciers. Average retreat rates range from 7.8 m per year for cirque

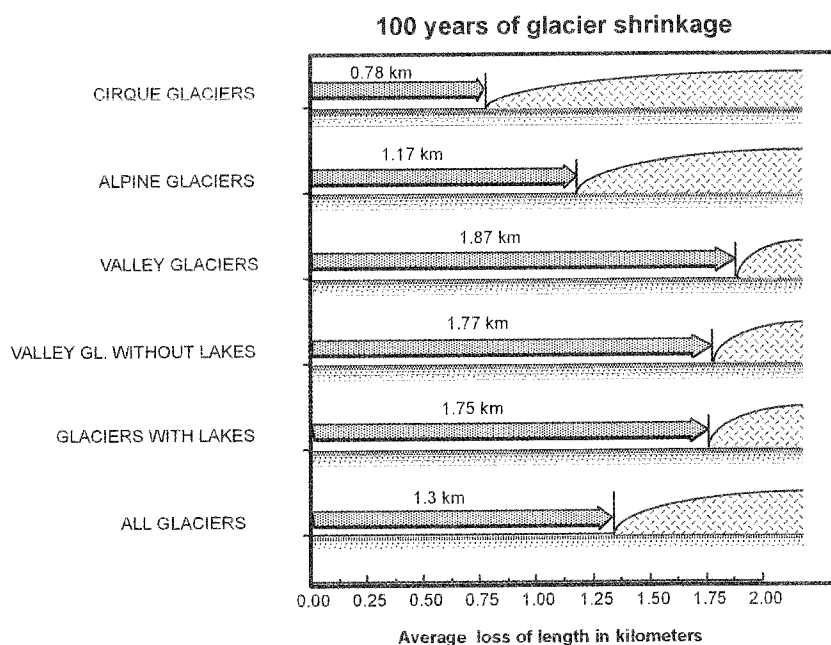


Figure 6.9 Length changes for the last 100 years, averaged for each of the various categories of glaciers.

glaciers to 17.7 m per year for valley glaciers, with a mean for all glaciers of 13.3 m per year. This includes a number of the larger glaciers that have not changed at all, but so far have undergone only surface lowering. It is mainly these large glaciers with century-long response times, which have yet to come to equilibrium with the present climate, that are now disintegrating into lakes. An insulating blanket of surface moraine that inhibits melt is also an important factor in prolonging their retreat. Over the same period, there has been an average loss of glaciated area of some 26%.

What does recession tell us about the climate?

Glacier recession is a response to a general rise of the ELA or snowline. This rise can be approximated by the rise in mid-elevation of the glacier determined from the increase in altitude of the retreating termini. This “snowline” rise, calculated as simple arithmetic means, shows that the changes in mean elevations range from 84 m for the cirque glaciers to 137 m for the average of alpine glaciers (Chinn 1996). If we assume that temperature alone was responsible for the recession of the glaciers, then by using the rise in snowline of only the cirque glaciers (as they have smaller shape errors) and an average atmospheric lapse rate of 0.7°C per 100 m, the mean upward snowline shift of c. 84 m for the cirque is equivalent to a warming of c. 0.6°C. The meteorological records suggest that measured temperatures over the past century show a warming of c. 1.0°C, with the bulk of the rise occurring since the 1950s.

The gains of the 1980s and 1990s

Commencing about 1977, there was a reversal of the glacier recession trend, with a gain to the glaciers which, in most years, continued through until 1998 (Chinn 1999). These gains have accumulated to send advances down to the fronts of most of the fast-response glaciers on both sides of the Alps. The different types of glaciers have shown differing responses to the balance gain. Advances of the small cirque glaciers have been too small to be detectable. On the steep responsive mountain glaciers (Fig. 6.10) the dominant resurgence took place in 1983, followed by a decline in the number of glaciers advancing. A second advance pulse is evident from 1989 to 1995. Response times range from 5 years to more than 20 years, as a few glaciers have yet to begin the general readvance. Valley glaciers have shown a characteristic slow, dampened response of low-gradient glaciers (Fig. 6.10). The 1983 initial pulse evident in mountain glaciers has been dampened for valley glaciers into a more general resurgence culminating in 1986, some 2 to 4 years later than that for the mountain glaciers. Again there appears to have been a second pulse of advances peaking in 1994. Waves of

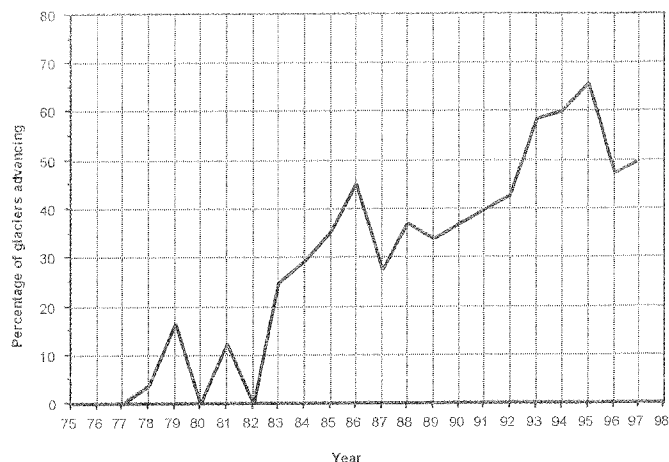


Figure 6.10 Percentage of a sample of 64 glaciers advancing during the past two decades.

thickening did pass into the trunks of the largest valley glaciers, but these pulses were dampened out before they affected the termini.

Water resource changes associated with this period of glacier gains have been crudely estimated by using the annual snowline altitudes (ELAs) of the index glaciers (Chinn *et al.* 2002). The method requires that the area-altitude curve and the gradient of the mass gains and losses is known for each index glacier to calculate the annual mass change. Volume changes from the index glaciers are then extrapolated throughout the entire New Zealand Southern Alps, based on glacier area correlations. Annual ice volume changes for a trial period 1993 to 2001 have been calculated (Table 6.1). The results of this study indicate that there has been a net volume gain of 6.7 Mm³ or 12.6% from 1993 to 2001.

Table 6.1 Estimated annual and total changes in ice volume for the glaciers of the New Zealand Southern Alps from 1993 to 2001.

Glacier Year	Volume change (km ³) Southern Alps	Cumulative volume (km ³) Southern Alps
1992/1993	0.00	53.28
1993/1994	1.70	54.96
1994/1995	2.93	57.86
1995/1996	-1.00	56.84
1996/1997	2.47	59.29
1997/1998	0.08	59.35
1998/1999	-1.44	57.89
1999/2000	-1.04	56.83
2000/2001	2.44	59.25

Glacier fronts that ignore the climate

Glacier recession by icebergs calving into a lake is a much faster process than ordinary glacier ablation (i.e. melt and evaporation). In fact it is so much faster than shrinkage by melt that, once a lake begins to form at the glacier terminus, the glacier is doomed to suffer a massive recession. The calving process will normally overtake any effect of the climate and recent lake expansion has steadfastly continued through the recent decade of glacier expansion. A thickening pulse travelling down the glacier trunk may be observed on many of them, but this has served only to retard or temporarily halt the expansion of the proglacial lakes.

WATER RESOURCES OF THE GLACIERS

Glacier ice constitutes a significant water resource held in New Zealand mountains. Fluctuations in the volumes of glaciers have a direct impact on downstream hydrology and on New Zealand's essential hydro-electric industry. The main impacts will be from changes in water storage held as ice. Recent scenarios of climate warming suggest that glaciers may have a measurable effect on river discharges. Scenarios of 1.5°C and 3°C warming were investigated (Chinn 1990) and it was reported that these warming trends would reduce the volume of perennial snow and ice by 25% and 50% respectively. By applying these ice losses over a 50-year period to the main hydro-electric generating rivers of the Waitaki and Clutha, and to the larger Canterbury rivers, the additional contributions to daily mean discharge from diminishing ice storage (ignoring the 0.9 conversion for the density of ice) were calculated (Table 6.2). For the majority of the rivers, the increase in flow would be insignificant, but for the Clutha it would be a significant 0.5 to 1 m³/s increase, and for the Waitaki there would be an important 8 m³/s increase in flow.

Comparative estimates of the contribution to the Waitaki River system from depletion of ice storage are available for the Tasman Glacier. The Tasman Glacier catchment contains 65% of the ice area of the Waitaki catchment. From surveys on the lower Tasman Glacier, Skinner (1964) calculated that discharge from down-wasting alone over the 72 year period from 1890 to 1962 contributed a discharge of 1.43 m³/s to the Tasman River. Since the development of the pro-glacial Tasman Lake, the rate of ice loss has increased dramatically and in a recent study, Purdie and Fitzharris (1999) found that recession of the Tasman Glacier now discharges 4.3 m³/s into the system, of which 0.7 m³/s is supplied by direct calving of ice into the lake. Hydrology downstream of glaciated areas is also very dependent on glacier behaviour which, in addition to water discharge, affects potential hazards from

Table 6.2 Discharge to main eastern rivers from ice storage over a 50-year period for two climate-warming scenarios (Chinn 1990). Scenario 1 is for a 1.5° C temperature rise, leading to a 120 m rise in snowline and a 25% loss of present ice volume. Scenario 2 is for a 3° C temperature rise, a 230 m rise in snowline and a 50% loss of present ice volume.

River	Scenario 1 25% ice reduction 50 yr meltwater runoff (m ³ /s)	Scenario 2 50% ice reduction 50 yr meltwater runoff (m ³ /s)
Waimakariri	0.009	0.018
Rakaia		
Cameron	0.011	0.022
Rakaia	0.216	0.433
Mathias	0.008	0.016
Wilberforce	0.010	0.019
Avoca	0.001	0.002
Entire Rakaia	0.246	0.493
Rangitata		
Havelock	0.026	0.051
Clyde	0.047	0.094
Lawrence	0.008	0.015
Entire Rangitata	0.080	0.161
Waitaki		
Ahuriri	0.009	0.019
Ohau	0.0004	0.0008
Huxley	0.009	0.018
Hopkins	0.057	0.115
Dobson	0.016	0.033
Ben Ohau	0.007	0.015
Hooker-Mueller	0.421	0.842
Tasman	2.477	4.953
Murchison	0.609	1.217
Jollie	0.004	0.009
Cass	0.013	0.025
Godley	0.388	0.777
Macaulay	0.009	0.018
Entire Waitaki	4.021	8.042
Clutha		
Humboldts	0.020	0.040
Dart	0.278	0.556
Rees	0.053	0.106
Shotover	0.006	0.012
West Matukituki	0.071	0.143
East Matukituki	0.041	0.081
Wanaka	0.0007	0.001
Wilkin	0.029	0.057
Makarora	0.011	0.021
Hunter	0.031	0.062
Entire Clutha	0.540	1.079

glacier floods and sediment discharge, with its implications for river aggradation and erosion.

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Chapter 5

Seasonal snow and water

Ian Owens and Blair Fitzharris

Of the total global precipitation, only about 6% falls as snow, so at first glance snow does not seem an important part of hydrology. However, storage of water as snow exerts a powerful influence on runoff at many time scales. Seasonal snow accumulates and then melts within a single hydrological year. In so doing it tends to reduce river flows in winter, but enhances them in spring and early summer.

Southern Alps, but is generally less than 1000 mm in the eastern mountains (Chinn 1969; Fitzharris 1979). By the end of summer, the snow line retreats to elevations of between 1500 m and 2200 m, which marks the perennial snow margin. Earlier reviews of seasonal snow in New Zealand can be found in Fitzharris *et al.* (1992) and Fitzharris *et al.* (1999).

SNOW IN NEW ZEALAND

New Zealand lies between latitudes 34° and 47°S in the southwest Pacific. Surrounded by ocean, its main axial ranges trend across the prevailing westerlies, and reach elevations above 2000 m in many places. Most seasonal snow is concentrated in the mountains of the Southern Alps, but it also accumulates on the volcanic cones of the North Island and on its main axial ranges. The combination of high relief and moist on-shore winds ensures that up to 35% (53,000 km²) of the South Island is snow-covered throughout the winter (Fig. 5.1). The winter elevation of the snow line is highly variable, but averages 1000 m in the south and 1400 m in the north. High summer solar radiation levels at New Zealand's mid-latitude location ensures strong melting and a supply of snow water to rivers.

Seasonal snow accumulation exceeds 4000 mm water equivalent near the Main Divide of the

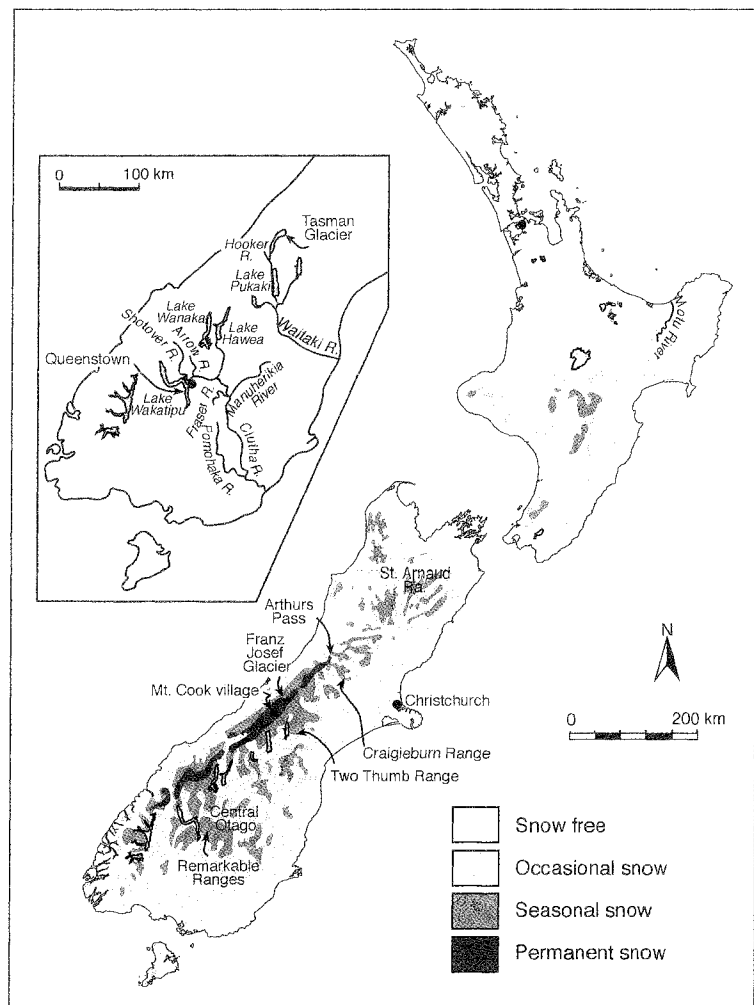


Figure 5.1 Snow- and ice-covered areas of New Zealand and names of locations referred to in the text (modified from Technical Subcommittee on Snow, New Zealand Committee for the I.H.D. 1969).

MEASUREMENT OF SNOW

Snow depth and water equivalent

Many different techniques are used to measure the amount of snow (Fig. 5.2). The most straightforward is to measure snow depth at a point. Accumulated snow depth is assessed with reference to a graduated reference stake, while snow boards allow a measure of new snow fall amounts, as they are usually cleared of snow daily. Snow depth can also be remotely measured, for example with an ultrasonic sensor. However, the enormous spatial variability of snow cover, its transient nature, and great depth in the western mountains, makes comprehensive measurement over a large catchment rather difficult (Chinn 1969).

For hydrological purposes, measurement of snow water equivalent, rather than snow depth, is more desirable and can be obtained in various ways. If a sample of known volume is weighed, the density can be determined (Fig. 5.2a), and the water equivalent given as the product of depth and density. A standard procedure in snow hydrology is to measure water equivalent by sampling the entire thickness of the snow pack with a coring device, such as the Federal Sampler (Fig. 5.2d). Often a snow course is set up, where a series of such samples are made in an area of similar slope, elevation and exposure.

In some cases it is necessary to obtain detailed information about individual layers within the snow pack. In these areas, a snow pit is dug (Fig. 5.3), if possible down to ground level. The observer identifies layers, and measures their thickness, temperature and density. A full snow pit analysis will often involve other parameters such as snow type, hardness and wetness.

Snow deposition

Snow deposition is obtained from recording precipitation gauges, though a turbulence-damping device such as an Alter shield (Fig. 5.2e) is required in windy environments typical of alpine catchments so as to ensure an accurate catch. A measure of both accumulation and melt is obtained with a snow pillow (Fig. 5.2f) in which the pressure of the snow pack on a reservoir of non-freezing liquid is recorded.

Area of snow

The areal distribution of snow cover is often synthesised from a large number of point observations, as Chinn (1981) and Hughes (1974) have done for storms affecting low elevations in the eastern South Island. Greater detail of snow distribution is sometimes available when snowstorms affect urban areas, and press reports may also allow the recurrence interval of snow falls of different depths to be estimated (Fig. 5.4).

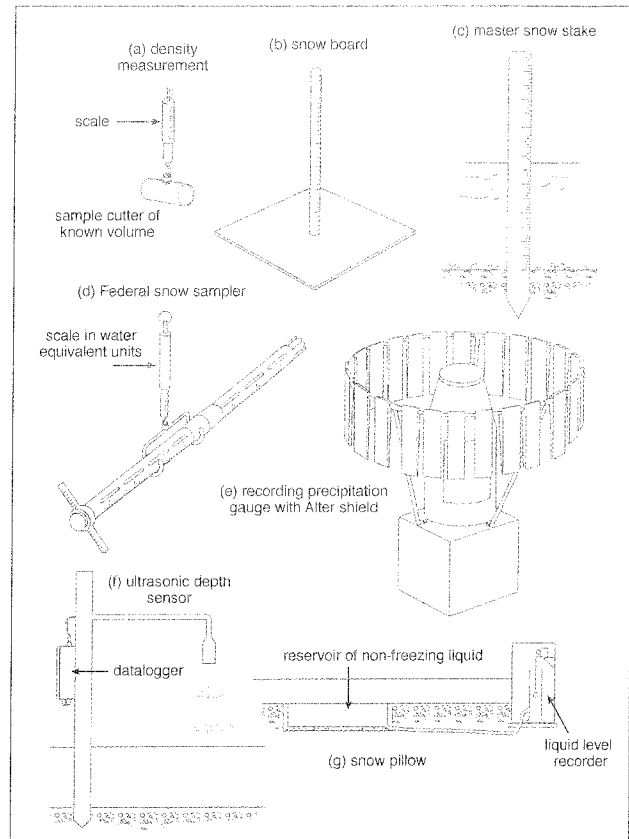


Figure 5.2 Instruments for measuring snow accumulation and melt.

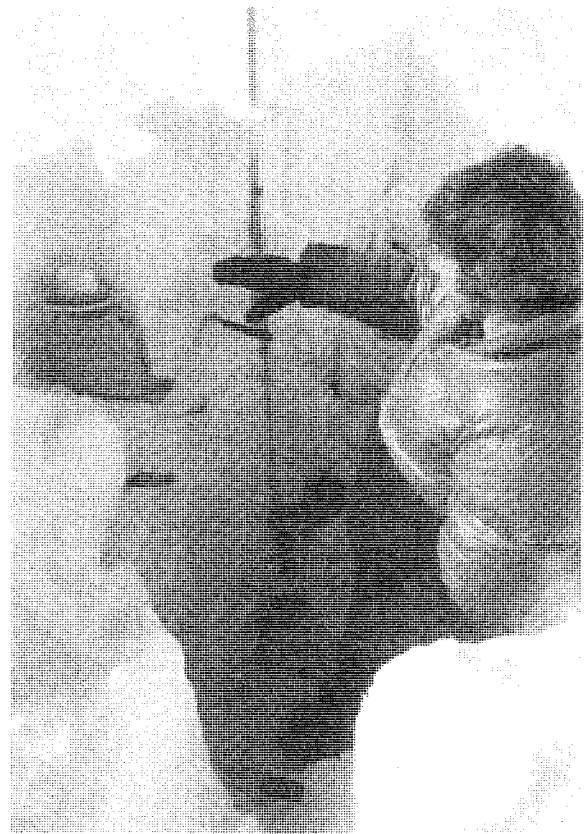


Figure 5.3 Making observations of snow structure and characteristics within a snow pit.

Photo: Ian Owens

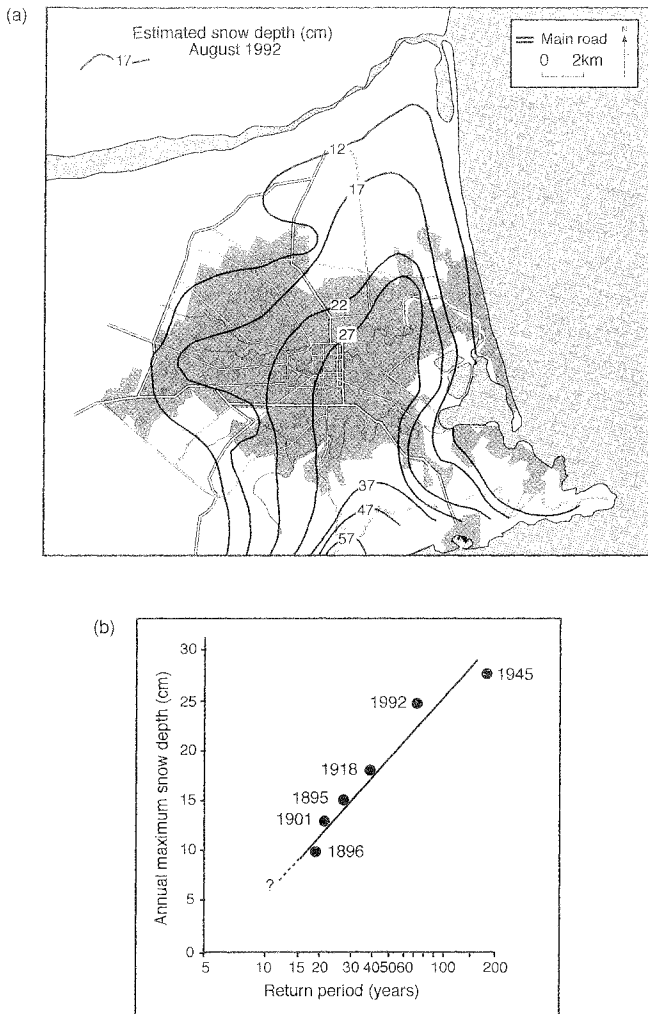


Figure 5.4 Snow falls in Christchurch. (a) Distribution of snow depths for August 1992 storm (b) Recurrence intervals of snow depth in snow storms for Christchurch (modified from Christchurch Engineering Lifelines Group 1997).

The extent of snow can be determined remotely by aerial photography, or more commonly by using radiation sensors on satellites. An example showing the distribution of seasonal snow over the South Island after a mid-winter southerly blast is given in Figure 5.5. Thomas *et al.* (1978) used Landsat imagery for snow investigations on the St Arnaud Range. Although these images have good resolution (about 15 m), they found them to be of limited use for hydrological applications, because the satellites orbit over the country at 18-day intervals and cloud cover frequently obscures the ground surface. Weather satellites have a much lower resolution (about 1 km), but imagery is available at least daily. The potential for their use was examined by Hickman (1972) and Fitzharris and McGann (1989). They showed that there is reasonable agreement between snow line estimates from satellite imagery and



Figure 5.5 MODIS satellite image of South Island showing snow cover during the coldest part of the winter (NASA image for 15 June 2002).

from weather stations. Fitzharris and McAlevey (1999) presented three case studies that explore the effectiveness of using weather satellites for determining snow-covered area and snow lines. They concluded that, by itself, satellite monitoring is not sufficient to provide a complete snow climatology. It can, however, give valuable and low-cost monthly checks as to changes in snow cover during the year. They illustrate how, in combination with computer snow simulation models, the area and volume of seasonal snow cover can be obtained.

Snow melt

Snowmelt can be assessed by repeated measurements of snow depth—the water equivalent is then obtained by multiplying the depth by a standard or known snow density. Alternatively, a lysimeter, placed within or at the base of the snow, catches melt water and measures it through repeated weighing or with a tipping-bucket mechanism at an outlet (Neale and Fitzharris 1997).

MODELING THE SNOW COVER

If direct measurements of snow depth and water equivalent are not available, snow accumulation is sometimes estimated from standard climate observations. A general index of snow pack accumulation may be derived using daily or monthly deviations from long-term average

rainfall and temperatures measured at surrounding lowland climate stations (Fitzharris 1987). Perhaps more effectively, computer simulations can be made of processes of snow accumulation and melt. In this way Moore and Owens (1984) modelled snow water equivalent for the Alan's Basin snow course at 1750 m elevation in the Craigieburn Range, using daily precipitation and temperature data from climate stations located in the nearby mountains. The model requires the optimization of parameters for the rain/snow boundary, a precipitation correction factor to allow for increases with elevation and a degree-day factor for snow melt.

More elaborate definitions of the rain/snow threshold and more detailed specification of temperature variation with height can improve such models. For example, Barringer (1989) simulated snowline elevation on the north-facing slope of the Remarkables Range. He used data from the meteorological station at Queenstown airport and three automatic recording stations at 950 m, 1295 m and 1615 m elevations. Fitzharris and Garr (1995) developed a similar model called *SnowSim*, but which estimates seasonal snow in the main hydroelectric catchments of the South Island. It uses daily climate data from surrounding long-term climate stations. The model is calibrated against the long-term water balance and the output agrees with available historical data.

These simulation models are lumped-parameter models. Apart from variations with elevation, they give no indication of the spatial variability of water stored as snow. McAlevey (1998) extended *SnowSim* to meet this need. His model provides spatial estimates of snow for every 1 km² pixel for the entire seasonal snow zone of New Zealand. Data is interpolated from 41 climate stations using neural networks, an inverse distance-weighted algorithm and lapse rates for temperature and precipitation. Satellite-derived snow-covered area agrees well with that determined from the model (Fitzharris and McAlevey 1999).

PHYSICAL PROPERTIES OF SNOW

New snow types

Most snow originates in the atmosphere at altitudes of 2,000 to 12,000 m, where cloud temperatures are normally below freezing and ice and supercooled water coexist. Ice crystals grow large enough to fall, either by crystal growth (at the expense of water droplets), or from riming (when droplets fall onto crystals and freeze). The exact form of an ice crystal depends on the temperature and humidity conditions at which it grows. The few reported observations for New Zealand mountains, for example by O'Loughlin (1969), Prowse (1981), and Weir and Owens (1981), show that crystals such as heavily rimed needles

are very common. Typically, they are associated with humid conditions near the freezing level.

Cold content of snow

After snow has been deposited, its role in subsequent stages of the hydrological cycle is dependent on the cold content, or heat deficit of the snow pack, which is a function of snow temperature, density and depth. Heat deficits develop in early winter and are not lost until the snow pack becomes isothermal at 0°C in spring. It is then "ripe" for melt. The New Zealand snow pack, however, is relatively warm and may become isothermal at any time. For example, Prowse (1981) calculates that the maximum heat deficit of the Craigieburn Range snow pack could easily be overcome by one day's energy gain, even in winter.

Snow metamorphism

Before the snow pack becomes ripe, it undergoes metamorphism, which affects its hydrological behaviour (Colbeck *et al.* 1990). If the snow temperature remains lower than 0°C, the snow is said to be dry and metamorphism is controlled by the temperature gradient. Under small gradients, "equilibrium growth" produces rounded, well-bonded snow grains. However, when the temperature change with depth is large, even for small vertical distances, faceted crystals, and eventually hollow cup-shaped crystals called depth hoar, are formed. This process, called "kinetic growth", is common in cold inland locations with shallow snow packs, whereas equilibrium growth is more common in maritime areas with deep, warmer snow packs. Metamorphism of wet snow occurs very rapidly and produces clusters of grains, which eventually lose their identity to form large snow grains known to skiers as spring corn. Refreezing of layers following melting may form very strong and thick ice crusts.

In dry, cold snow, equilibrium metamorphism dominates. Several studies show that depth hoar is surprisingly common in New Zealand, despite our maritime climate. It is observed in colder winters, especially on ranges east of the Main Divide, where snow packs are shallower and cold clear conditions may persist for long periods (McNulty and Fitzharris 1980; Weir and Owens 1981; Prowse and Owens 1984; McGregor 1990). However, most of our snow can be regarded as wet, being at or close to 0°C for much of the time, so that equilibrium metamorphism dominates. Weather processes often create crusts at the snow surface, some of which are subsequently buried.

Apart from indirect effects through avalanche activity, the hydrological significance of snow metamorphism is through the potential of the snow pack to retard delivery of heavy rainfall or large snow melt quantities to river systems. For instance, Moore (1984) showed that liquid

water in snow at Temple Basin near Arthur's Pass is trapped and diverted horizontally two to three metres along ice crusts. Nevertheless, measured percolation rates are moderately high because ice layers decay rapidly during rain-on-snow events (Moore and Prowse 1988).

Snow density

Snow density is the most useful single property for classifying different types of snow and ice phenomena. New snow densities depend on conditions of snow formation and fall. The bulk density of the seasonal snow pack is controlled by processes of metamorphism. In the longer term, firnification leads to eventual formation of glacier ice. Typical densities are given in Table 5.1. At low elevations, new snow densities of about 300 kg/m³ are common (Chinn 1981). At high elevations, most new snow is less dense, but still classified as "damp". Where windy conditions occur, dense slabs are produced from new snow falls (Prowse and Owens 1984).

Table 5.1 Snow types and their density

Type	Density (kg m ⁻³)
Wild snow	10 – 30
New snow immediately after falling in calm air	50 – 70
Damp new snow	100 – 200
Settled snow	200 – 300
Wind-packed snow - soft slab	100 – 290
Wind-packed snow - hard slab	290 – 450
Firn	400 – 800
Glacier ice	917
Water	1000

Source: after Seligman(1936), Paterson (1969), Male and Gray (1981)

PROCESSES OF SNOW ACCUMULATION

The maritime location of New Zealand means that air masses approaching the Southern Alps have freezing levels that are well above sea level. As a consequence, snow lines from storms usually intercept the mountains at some elevation above their base. At the same time, proximity to the cold air of Antarctica ensures that strong southerly flows bring snow to the mountain ranges, even in summer.

With the passage of successive frontal systems, freezing levels frequently rise and fall over many hundreds of metres within a few days, so that snow conditions change rapidly. Precipitation may fall as rain and substantial melt may take place at elevations up to 2500 m or more in mid-winter, yet snow can fall to below 500 m in summer. Thus snow accumulation is not always a steady process, and snow fall or melt occur in every season. The typical evolution of a snow pack is illustrated in Figure 5.6, which shows detailed measurements over a snow season made at two pit sites on the Round Hill Ski Field, Two Thumb Range, Tekapo. Snow falls are related to storm precipitation, as measured a few kilometers away at Mount John, and to the elevation of the freezing level. Each snow fall makes a step increment to the snow-pack depth. Individual snow layers thin exponentially with compaction and metamorphism. Towards October, when the freezing level rises, few additions are made to the snow pack, as precipitation mainly falls as rain.

Processes of deposition and melt create a pattern of increase of snow accumulation with elevation above the snow line that usually resembles a wedge (Fig. 5.7). The shape and slope of the snow wedge are seldom consistent from winter to winter, and are dependent on the frequency of weather types. As snow crystals fall they are deposited on mountains. Those that pass below the freezing level begin to melt, finally turning into rain. Some snow in the layer immediately below the freezing level does not melt

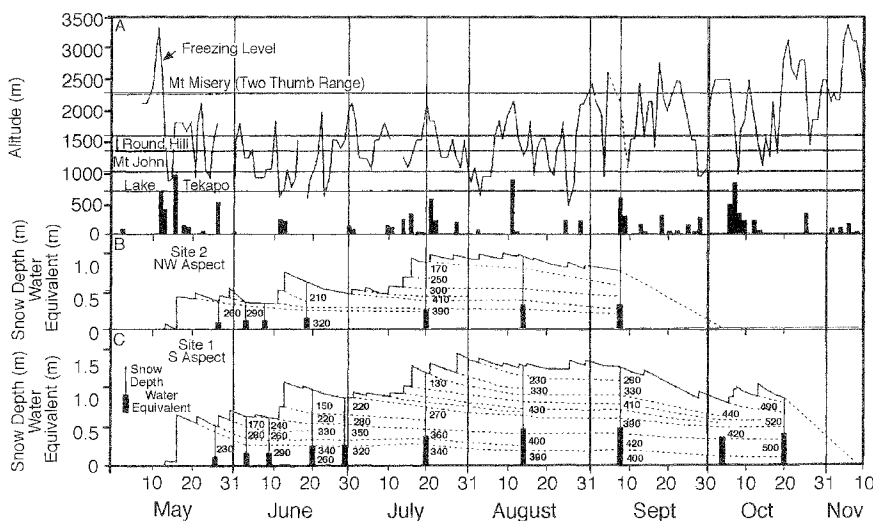


Figure 5.6 Detailed growth and decline of a snow pack for a typical eastern mountain range: Round Hill, Tekapo, snow course. (Fitzharris *et al.* 1992)

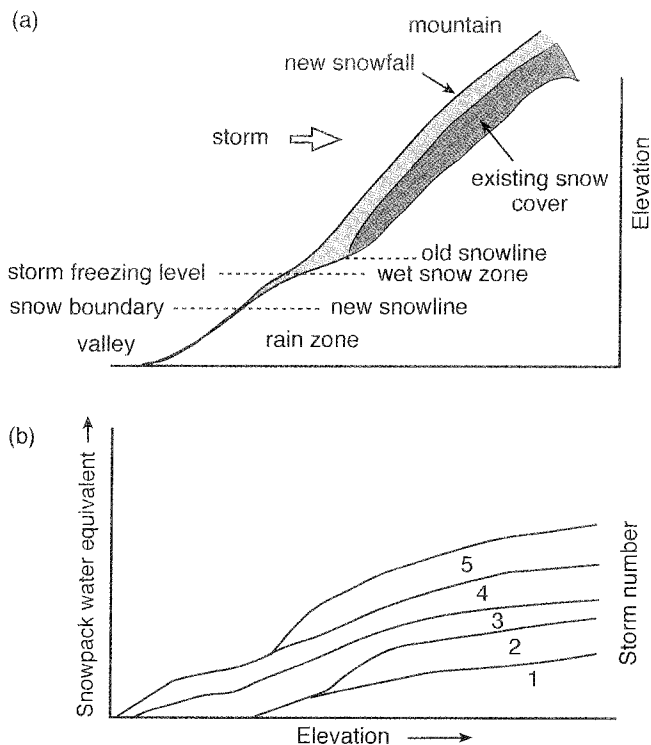


Figure 5.7 Development of the seasonal snow wedge.

completely before it reaches the ground, and so forms a wet snow zone. The net result is that after a typical storm there is no snow in mountain valleys, but there exists a well-defined fresh snow line part way up the mountain slope. New snow depths increase steadily with elevation in the shape of a snow wedge (Fig. 5.7a).

Along the Southern Alps a typical frontal system creates a compound snow wedge—a higher elevation one is produced by snow in the pre-frontal airstream, which tends to have a higher freezing level; superimposed on this is a second wedge with a lower elevation snow line formed by colder air behind the front. Each storm in the normal sequence of westerly circulation patterns creates its own snow wedges of different shape. The snow line moves up or down the mountain slope, depending on the orographic precipitation gradient and freezing level. Successive wedges then accumulate to give an increase in size of the snow pack with elevation (Fig. 5.7b).

Periods between storms may cause melt at lower elevations, a process that acts to steepen the overall wedge. In spring, as freezing levels rise, more precipitation falls as rain rather than snow, and increased temperatures promote more melt. Consequently, the snow wedge is eroded and finally disappears in spring or summer, with the timing of its demise dependent on elevation. For high mountains, the snow wedge never completely melts, such as in the example given by Fitzharris (1978) for the Tasman Glacier, and snow accumulates to become incorporated as névé.

There is little or no tall vegetation in the main snow

areas of New Zealand, and winds are persistently strong. These factors create considerable movement and losses through sublimation or through snow being blown to lower elevations where it melts. Plumes of snow are trapped in the lee of obstructions such as tors and tall tussocks, and large drifts form downwind of terrain breaks in slope and in gullies. The result is a highly variable spatial pattern of snow deposition (Twaddle 1995).

In a study of snow accumulation on the wind-swept mountains of central Otago, Harrison (1986) found that aspect is important in controlling snow depth and water equivalent. Snow tends to accumulate in specific locations. Exposed southeast to west slopes record a water equivalent of less than 200 mm, and sheltered northeast slopes over 450 mm. Snow drifts form a useful water resource, storing approximately 4 million cubic metres in a 120 sq km catchment and contributing up to 9% of spring runoff. Snow patches continue to supply melt water for downstream irrigation well into late spring and summer. For cooler years, some can even persist into the autumn.

PROCESSES OF SNOW MELT

The melting of snow and ice is controlled by energy transfers to the surface. Melt energy (Q_m) is available from several sources (Fig. 5.8), and melt can begin as soon as the snow pack or upper layers of the glacier become isothermal. In spring and summer, Q_m is large, even at high elevations. Radiation energy (Q^*), which varies mainly with solar (short-wave) radiation, depends on the time of year, time of day and albedo (reflectivity) of the surface. New snow may have an albedo as high as 95%, but for melting snow and surfaces, albedoes are usually closer to 50%. Upward long-wave radiation normally exceeds incoming long-wave radiation, unless warm clouds or high-angle snow-free terrain are present.

Sensible heat (Q_H) and latent heat (Q_E) are transferred by turbulent motions above the surface. Their respective magnitudes depend, first, on the temperature and humidity gradients above the surface, and second, on the eddy conductivities for heat and moisture. Because melting snow or ice maintains a surface temperature of 0°C , sensible heat gain occurs when the overlying air has a positive temperature. Latent heat gain occurs when vapour pressures in the air exceed 6.11 hPa, the saturation value at 0°C , and is the release of energy as condensation occurs on the melting surface. The eddy conductivities depend mainly on wind speed, but they are also influenced in a complex way by atmospheric stability and surface roughness. As shown by Moore (1983), good estimates of the turbulent energy transfers to melting snow and ice can be obtained from measurements of wind speed, temperature and humidity at a single height above a

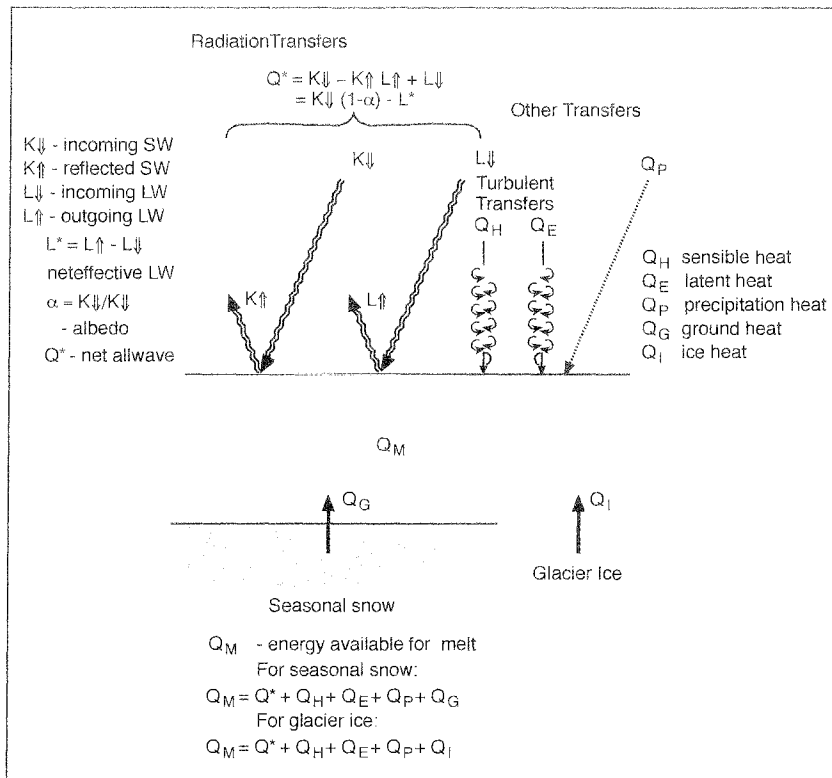


Figure 5.8 Sources of energy for snow melt.

melting surface. The energy transfer due to rain may be estimated from rainfall rates and temperature. The energy transfer by conduction from the ground or deeper layers of a glacier may be estimated from temperature profiles.

Worldwide studies of energy transfer for melting snow show that radiation energy, Q^* , is usually the main source for melt, especially in forested locations and at high elevations. In New Zealand, however, the turbulent transfers of sensible and latent heat, Q_H and Q_E , dominate because of our temperate, humid and windy conditions and nearness to the ocean. Precipitation heat transfer (Q_P) also makes significant, though not dominant, contributions (Table 5.2).

Lysimeter and energy balance measurements at 1750 m elevation near the Main Divide (Mueller Hut near Mount Cook village) show that radiation energy is the predominant energy source for melt of seasonal snow, but it is confined to daytime hours. During fine periods, there is a broad seasonal variation in energy supply, but nor'west storms at any time of year can produce very high turbulent and precipitation energy transfers. For example, Marcus *et al.* (1985) found that during one storm, in which 300 mm of rain fell in 10 hours on the lower Franz Josef Glacier, there was 155 mm of ice melt, even though it was mid-winter. East of the Main Divide, sensible heat is often dominant, especially during nor'westers—a frequent foehn wind in the lee of the Southern Alps (Fitzharris *et al.* 1980; Moore 1983; Neale and Fitzharris 1997). The highest elevation experiments come from the Tasman Glacier névé.

Here at an elevation of 2440 m, sensible heat (Q_H) is the dominant source of energy for melt, with up to 78 mm/day recorded (Cutler 2002).

For many applications in hydrology, use of the energy balance approach to estimate snow melt is not very practical. Measurement of components of the energy balance requires sophisticated instrumentation in severe field environments. Estimation is possible, but uncertainty remains over key parameters, particularly exchange coefficients for sensible and latent heat (Q_H and Q_E). Accordingly, daily snow melt is more usually estimated using the degree-day model:

$$M = f \cdot T_{\text{mean}}$$

where

$$\begin{aligned} M &= \text{snow melt in mm/day water equivalent} \\ T_{\text{mean}} &= \text{mean temperature for the day (}^\circ\text{C)} \\ f &= \text{a melt factor (mm/}^\circ\text{C/day)} \end{aligned}$$

This simple empirical model is found to work surprising well for catchments, provided an appropriate value of f is chosen. The overseas literature considers that f is between 2–8 mm/ $^\circ\text{C/day}$ for melting snow, depending on albedo, the age of the snow and whether it has rained (Male and Gray 1981). Moore and Owens (1984) found melt factors ranging from 4–8 mm/ $^\circ\text{C/day}$ in the Craigieburn Range. Neale (1996) reports an f of 2.3 mm/ $^\circ\text{C/day}$ at Mueller Hut at 1750 m elevation, near Mount Cook village, but some data suggests that during nor'west storms it could rise to as high as 11.5 mm/ $^\circ\text{C/day}$. Cutler (2002) repeated

Table 5.2 Energy sources for melting snow

Location	Months	Q_M (MJ m ⁻² d ⁻¹)	Q^* (%)	Q_H (%)	Q_E (%)	Q_P (%)	Source
Central Otago	Oct 1978	18.1	23	58	19	<1	Fitzharris <i>et al.</i> (1980)
Craigieburn (1550 m)	Nov 1976	8.2	42	43	14	<1	Prowse (1981)
	Oct 1977	15.1	33	64	3	<1	
	Oct 1979	13.0	17	61	21	<1	
	Oct 1980	9.2	27	60	12	1	
	Average	10.3	30	57	13	<1	
Temple Basin	Oct-Nov 1982	10.6	16	57	25	2	Moore (1984)
Craigieburn Ra	Oct-Nov 1982	13.2	18	78	4	0	Moore (1984)
Craigieburn (1750 m)	Oct-Nov 1984	11.8	60	16	14	10	Pearce and Owens (1985)
Mueller Hut	Oct-Nov 1995	3.5	63	27	4	3	Neale and Fitzharris (1997)
Tasman névé	Jan-Feb 2001	5.8	33	52	15	<1	Cutler (2002)

his experiments, but at a higher elevation of 2440 m, on the Tasman névé and found a mean f of 3.4 mm/°C/day, rising to 9.1 mm/°C/day during nor'west storms.

ROLE OF SYNOPTIC CLIMATOLOGY IN CONTROLLING SNOW AND RUNOFF

Water inflows into the storage lakes can alter much from year to year because of weather fluctuations. Research into atmospheric circulation in the vicinity of New Zealand for very high and very low inflows to hydro lakes shows that very low winter inflows are caused by a higher-than-usual incidence of westerly to southwesterly airflow (Fitzharris and Garr 1996). Such anomalies diminish runoff because in winter a higher proportion of precipitation falls as snow. For inflows to be above average in winter, higher-than-normal frequencies of northerly to northwesterly conditions are required. These bring warm rain, rather than snow, and help to melt the snow pack. Low inflows in spring and summer are produced by anomalous easterly circulations. These typically induce little rainfall near the Main Divide and are often associated with anticyclones.

The El Niño/La Niña phenomenon, as measured by the Southern Oscillation Index, appears to exert an influence on the snow hydrology. A large-scale atmospheric teleconnection exists between El Niño conditions and higher frequencies of southwesterly airstreams onto New Zealand. As a working hypothesis, early winter is colder than usual, a larger-than-normal snow pack results and hydro lake inflows are consequently reduced. By contrast, during La Niña, there are anomalous airflows over New

Zealand from the northeast. Winter tends to be warmer and drier in the Southern Alps, and the snow pack is smaller. Summer inflows tend to be higher when the Southern Oscillation Index is negative in spring, and vice versa when it is positive (McKerchar *et al.* 1998).

There is a divergent circulation signal between years with high and low snow storage (de Lautour 1999). For high snow years, the winter anticyclone over Australia is larger, more intense, and it extends further south. The westerlies over New Zealand are stronger and extend further north. There is also anomalous circulation from the southwest. For low snow years, the winter anticyclone over Australia is weaker and smaller. The westerlies are weaker, and are located further south. There is anomalous anticyclonic circulation from the northeast.

INFLUENCE OF SNOW ON FLOW OF NEW ZEALAND RIVERS

Seasonal patterns of runoff vary, depending on the relative proportions of permanent or seasonal snow. Contrasting types of distribution are illustrated in Figure 5.9. For the glaciated Hooker catchment, maximum flows occur in January, and this month alone accounts for almost 20% of the annual total. There follows a steady decrease in runoff to the winter months, when most precipitation falls as snow and there is little energy available for melt. Thus July provides only 2% of the annual flow. River flows rise again in spring and closely follow the seasonal insolation cycle, so that over half of the annual runoff occurs in the three months December to February.

By contrast, the Manuherikia catchment has no

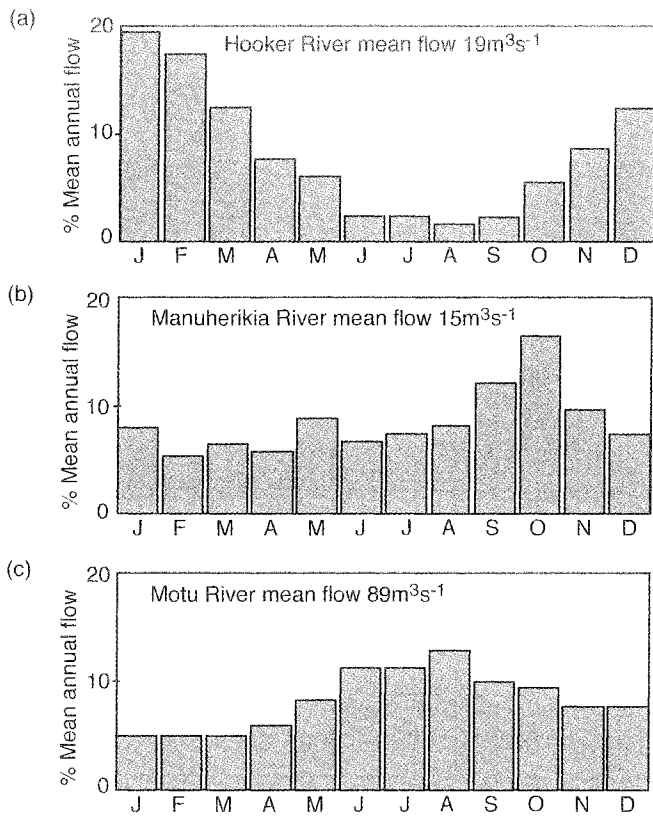


Figure 5.9 Mean monthly runoff for contrasting catchments showing the dominant role of (a) perennial ice, (b) seasonal snow, and (c) no snow or ice.

permanent ice, but is substantially covered by seasonal snow over winter. Runoff peaks in October as temperatures rise in the spring and snow melt begins in earnest. As the area covered by snow shrinks towards summer, the flow diminishes rapidly to reach a minimum in February, and is sustained only by rare rainfalls, melt from remnant snow patches and base flow from upland bogs and groundwater.

Catchments with no snow or ice, such as the Motu, tend to produce a more even flow distribution throughout the year, with any variation being produced by seasonal changes in evaporation and rainfall. Flow tends to be higher in winter months.

More detail on the influence of seasonal snow on river flows is provided by examining the runoff patterns of tributaries of the Clutha (Jowett and Thompson 1977). Here the average altitude of the catchment is important. All rivers have low flows in July, the month with the lowest temperatures. Thence the energy available for snow melt steadily increases towards the summer solstice. The lowest catchments, such as Manuherikia, have their highest rates of snow melt and runoff in September, intermediate elevation catchments such as the Arrow and Fraser a month later, while runoff peaks in the highest catchments of Wanaka, Wakitipu, Hawea and Shotover in November.

The Fraser River shows the extreme case for a high

catchment with pronounced winter snow accumulation and spring melt. Diurnal variations in river flow are clearly defined as spikes superimposed on an upward bulge in spring base flow (Fig. 5.10). The size of the spike for any given day depends on the overall synoptic weather pattern and the energy available for snow melt. On anticyclonic days, there is much solar radiation available for melt and the diurnal runoff spikes are pronounced. They grow even larger with the advent of nor'westers ahead of a cold front, when sensible flux contributes to melt, and all but disappear as cloudy, cold southerlies follow the frontal passage.

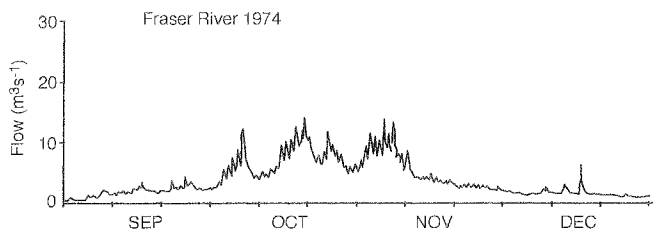


Figure 5.10 Typical pattern of spring flow for the Fraser River, Central Otago.

HOW IMPORTANT IS SEASONAL SNOW MELT TO RUNOFF?

Next to rainfall, the amount of water stored as snow and ice is a key factor controlling the hydrology of many mountain catchments, including those upstream of the South Island's main hydroelectricity storage lakes. Thus knowledge of these aspects is important for water management decisions. From Table 5.3 it can be seen that the average accumulated storage in these hydro catchments amounts to 15% of the annual runoff. This is predominately due to storage as seasonal snow, and represents water that is guaranteed to appear as river flow over the period October-February. It is almost the same size as controlled

Table 5.3 Estimated long-term water balance for the main South Island hydroelectric catchments.

Parameter	Water units (mm)	Energy units (GWh)
Precipitation	2 820	–
Evaporation	550	–
Runoff	2 270	15 550
Seasonal snow storage	330	2 260
Controlled lake storage	394	2 700

Source: modified from Fitzharris (1987)

lake storage. In this context, mean seasonal snow storage is crucial in determining the seasonal timing and amount of spring and summer inflow into the hydro lakes. For the hydro catchments, spring (Oct, Nov, Dec) runoff over the period 1931-84 varied by ± 500 mm about a mean of 800 mm. Fluctuations in snow storage account for about 40% of this variation, compared with fluctuations in spring precipitation which account for 60% (Fitzharris 1987). In some smaller mountain catchments, seasonal snow melt makes larger relative contributions. For example, in the Fraser catchment of Central Otago, where runoff is used for local hydro-electricity generation, for irrigation of horticultural land, and for frost fighting, melt from snow storage provides 33% of the annual river flow on average (Fitzharris and Grimmond 1982). In many eastern mountains of the South Island, slow melt of long-lasting snow patches (Pearce and Owens 1985) can provide valuable late spring and early summer water at times of low stream flows.

Unfortunately, there are only three snow courses in New Zealand with sufficiently long records to estimate annual variability in seasonal snow storage. At Alan's Basin, at an elevation of 1750 m in the Craigieburn Range, peak snow accumulation varied between 230 mm and 1030 mm over a 12-year period, with a standard deviation of 240 mm and coefficient of variation of 44%. A 19-year record of snow pack at elevations above 1300 m in the Two Thumb Range showed a coefficient of variation of 52% (Chinn 1981). Harrison (1986) estimated snow accumulation in the Fraser basin of Central Otago over 17 winters. The driest winter gave 40% lower water equivalent and 55% less runoff than the winter with the heaviest snow cover. All these snow courses are in relatively low-precipitation areas, but demonstrate that seasonal snow storage, and hence its contribution to runoff, has a wide inter-annual range.

Fitzharris and Garr (1995) simulated daily values of seasonal snow storage for catchments of the South Island's main hydro lakes back to 1930. Seasonal snow tends to build from about May to a maximum in October. Melt is often completed by February, but water stored as seasonal snow can occasionally carry over into the following year. There is a large year-to-year variability of snow. Over 62 years, the maximum seasonal snow storage varied from less than 200 mm (1500 GWh) to over 650 mm (5000 GWh). The time series of this variability is given in Figure 5.11 and indicates considerable inter-annual variability. There is no marked trend in seasonal snow storage since 1930, although there may be larger amounts in recent years.

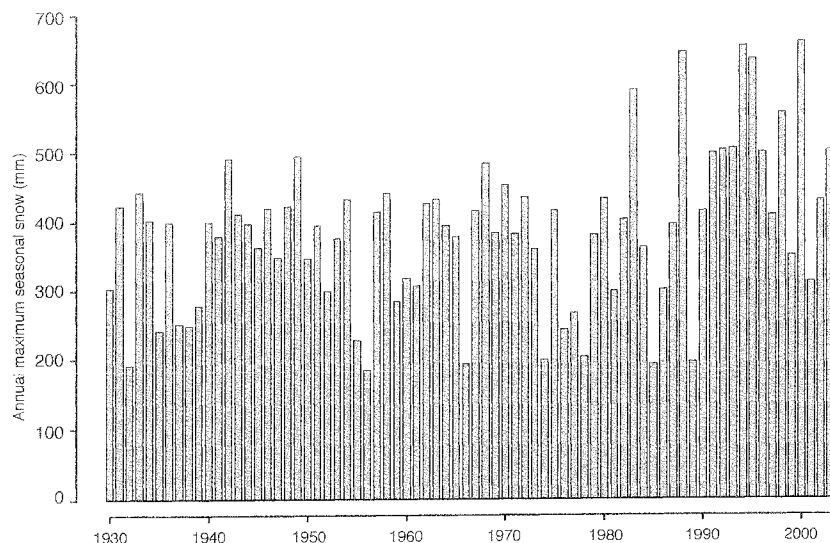


Figure 5.11 Estimated amounts of water stored as seasonal snow in the main South Island hydro catchments 1930-2003.

These results confirm that the amount of water stored as snow is markedly different from one winter to the next, and as a consequence spring and summer snow-melt runoff is never consistent from one year to the next.

SNOW MELT FLOODS

Under some conditions, rapid snow and ice melt make important contributions to floods. The role of snow melt during a major flood in the Clutha River was assessed for two subcatchments by Fitzharris *et al.* (1980). In the Fraser, where 80% of the catchment is above the normal snow line, melt provided 40% of flood runoff. Melt rates of nearly 3 mm/hr preceded rainfall. These saturated and primed the catchment, so that very quick rises in streamflow occurred when 100 mm of rain fell. In the Pomahaka, only 10% of the catchment is above the snow line, so that snow melt provided only 10 mm of the estimated 66 mm storm flow. Thus the relative contribution of snow melt to floods depends upon snow-line elevation and snow-covered area, as determined by the hypsometric curve of a catchment.

Melting of transient snow also contributes to small floods because when the snow line is lower than normal, considerable proportions of mountain catchments are involved. For instance, Moore and Prowse (1988) described a July storm that caused road washouts in the Craigieburn area. Precipitation of 123 mm in one day was augmented by over 25% by melting of a shallow snow pack at relatively low elevations. In drier mountain areas of the eastern South Island, floods can arise from snow melt alone. For example, the largest flood in one year from Camp Stream (Craigieburn Range) occurred on a day with no precipitation, but with foehn nor'west conditions producing strong melt (Moore and Prowse 1988).

CAN SNOW SURVIVE FUTURE CLIMATE CHANGE?

Future changes in water resources, electricity supply and electricity demand are likely if global warming occurs. This is because snow and glaciers are very sensitive to changes in temperature. The sensitivity of snow storage to changes in precipitation and temperature is shown as a response surface for South Island hydro catchments in Figure 5.12. Snow storage declines as temperature increases and precipitation decreases, but the curved isolines indicate that the relationship is not linear. For example, with temperatures colder than present, snow storage becomes more sensitive to changes in precipitation. Climate warmings envisaged by two plausible scenarios produce much reduced snow storage: 170 mm for warmer and wetter Scenario 1 (compared with 330 mm at present); and about 50 mm for warmer and drier Scenario 2 (Fig. 5.12). Seasonal snow storage is so diminished in Scenario 2 that the seasonal distribution of runoff will be greatly affected.

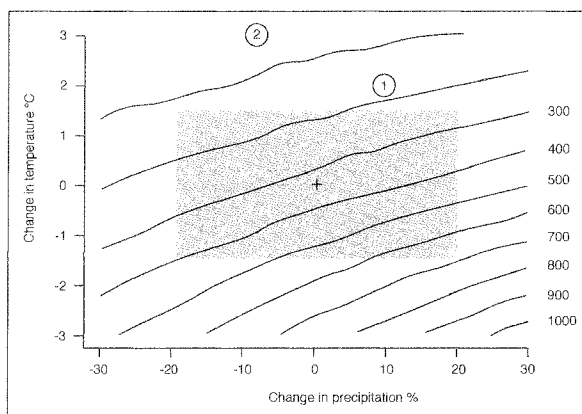


Figure 5.12 Impact of climate change on water stored as seasonal snow in the main South Island hydro catchments. Diagram is a response surface, which shows the sensitivity of snow with respect to changes in temperature and precipitation. Snow is given as mm water equivalent. Shaded area shows range of inter-annual variability for present climate.

Assuming increases in temperature of 3°C and precipitation of 15%, Fitzharris (1989) suggested a rise in snow line of 300–400 m, a decrease in snow accumulation below 2300 m and a reduction in winter snow-covered area for South Island hydro catchments from 45% to 28%. These changes will markedly alter the flow regime of many South Island rivers. Modelling studies by Fitzharris and Garr (1996) suggest an increase in inflow to hydro storage lakes of 40% in winter, but a decrease of 13% in summer. Annual runoff would increase by 14%. All these changes favour increased hydro generation and reduce the demand for water storage. Such forecasts are preliminary because

current year-to-year variability in snow storage is larger than the simulated changes in the mean, and because regional estimates of future climate are at present rather crude.

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Chapter 4

Soil water, runoff and streamflow generation

Tim Davie

INTRODUCTION

How much rainfall ends up in a stream is the most fundamental question a hydrologist can be asked, and many years of research have gone into investigating runoff mechanisms and pathways. One of the greatest motivations for the research has been to predict how much streamflow will result from a given amount of rainfall. Variability in soil characteristics and runoff processes make streamflow generation a highly complicated process that does not allow for simple mechanistic solutions. To understand runoff and streamflow generation it is important to understand the soil properties that influence runoff, and in particular to understand the role of soil water. This chapter describes the main properties of soil water and how they influence runoff mechanisms and streamflow generation.

SOIL WATER

Soil is a mixture of mineral particles, organic matter, air and water. The ratio of the amount of water to the other three constituents is the soil water content or soil moisture level. There are many different ways of expressing soil water content: it can be a percentage of saturation, a percentage of volume, a percentage of mass, a depth of water held in the soil, or even a volume of water held in the soil. The most widely used measure of soil water content is volumetric soil water content, θ , the fraction of soil volume consisting of water (Equation 4.1). The easiest method of obtaining volumetric soil water content is to take a soil sample of known volume and weigh it before and after drying it in an oven.

$$\theta = \frac{V_w}{V_t} \approx \frac{M_w - M_d}{V_t} \quad (4.1)$$

where θ is the volumetric water content (cm^3/cm^3), V_w is

the volume of water (cm^3), V_t is the total volume of soil (cm^3), M_w is the mass of water (g) and M_d is the mass of dry soil (g). The density of water is close to 1 g/cm^3 , so the weight of water can be assumed to equal the volume of water—hence the \approx symbol in Equation 4.1.

Gravimetric soil water content is similar to volumetric water content except that it refers to the ratio of water mass to soil mass, i.e., there is no need to know the volume of soil. Gravimetric and volumetric soil water content can be related to each other by the soil bulk density—the density of soil *in situ*:

$$G = \frac{M_w - M_d}{M_d} = \frac{\theta}{\rho_b / \rho_w} \quad (4.2)$$

where G is the gravimetric water content (g/g), ρ_b is the bulk density of soil (g/cm^3 ; Equation 4.3), and ρ_w is the density of water (g/cm^3). As the density of water is close to 1 g/cm^3 it can be ignored in Equation 4.2.

Soil bulk density (ρ_b) is the ratio of the mass of dry soil to the total volume of the soil (Equation 4.3). Soil bulk density gives an indication of soil compaction, with a cultivated topsoil having a value of around 1 g/cm^3 and a compacted subsoil being as high as 1.6 g/cm^3 (McLaren and Cameron 1996).

$$\rho_b = \frac{M_d}{V_t} \quad (4.3)$$

Porosity (ε ; cm^3/cm^3) is another important soil water property. It is the fraction of pore space in the total volume of soil:

$$\varepsilon = \frac{V_p}{V_t} = 1 - \frac{\rho_b}{\rho_p} \quad (4.4)$$

where V_p is the volume of pores (cm^3) and ρ_p is the density of soil particles (g/cm^3).

In theory, water can fill all of the pores in a soil; therefore porosity is the maximum potential volumetric water content. In practice the volumetric soil moisture seldom reaches the porosity value and, if it does, gravity acts on the water to force drainage through the profile, so that moisture levels quickly drop back below porosity. Field capacity is the stable point of saturation after rapid drainage.

As well as the total porosity of a soil, it is important to know about the pore-size distribution, as this influences the wetting and drying of a soil. Pores can be classified according to size or function (McLaren and Cameron 1996). Macropores can be defined as pores greater than 30 μm (microns) in diameter, but they can also be defined by a drainage characteristic—the amount of pressure required to remove water from the pore. A well-structured soil consists of stable aggregates with a wide range of pore sizes within and between the aggregates, and macropores may make up at least 10% of the soil volume. This structure provides numerous interconnected pathways for the flow of water, with flow at a wide range of velocities. In less well-structured soils, biological activity (e.g., roots and worms) can produce macropores providing flowpaths for water that are largely separated from the main soil matrix (Clothier *et al.* 1998). These are essentially two different types of macropores—those large pores within the soil matrix and those that are essentially separated from the matrix.

Soil water acts as a store within the water balance equation (see Chapter 1). Water enters the surface through infiltration and leaves either through evaporation (including transpiration by a plant) or by percolating through to the saturated zone (groundwater). Plants rely on water stored in the soil to transpire. Although there is a relationship between the amount of transpiration from a plant and the soil moisture status, it is non-linear. Two factors influence this non-linear relationship:

1. As the soil moisture level decreases, the remaining water is held in smaller and smaller pores within the soil matrix. In order to utilise the water from the small size pores the plant has to “suck” very hard.
2. Plants have the ability to shut down their stomata, and therefore transpire less, in response to water stress. Some plants, including many pasture grasses, are not very good at shutting stomata and therefore transpire fully until they cannot obtain any more water, at which point they start to die. The corresponding soil moisture level is referred to as wilting point. Other plants (e.g., the snow tussock *Chinochloa rigida* and many other New Zealand native plants) are very effective in shutting down their stomata under water stress and can survive drought conditions effectively.

Plants exert an osmotic pressure from within their roots to pull the water from the soil. Water is held in the matrix at a pressure less than atmospheric pressure: essentially a negative pressure potential. The size of negative pressure potential is called the soil suction. The best way of imagining soil suction is to think of yourself as a plant! As the soil dries out you have to exert a large and larger force to draw the water from smaller and smaller size pores. The strength of force you (the plant) exert is the soil suction.

The relationship between soil moisture and soil suction is non-linear (Fig. 4.1), so that there comes a point when plants can no longer withdraw water from the soil (i.e., they cannot exert enough suction to get water from the very small pores). This is the wilting point. By convention the wilting point of a soil is normally assigned a soil suction value of 15 times atmospheric pressure, although in reality it will depend on soil structure and plant physiology. The shape of the suction moisture curve (Fig. 4.1) is an important characteristic of a soil and is important for understanding how the soil drains or responds to rainfall. The shape is governed by two factors: the soil’s physical properties (e.g., mineralogy, pore size distribution) and whether the soil is wetting or drying (hysteresis). Soil physical properties vary with soil type and the suction moisture curve differs between soils (Fig. 4.1).

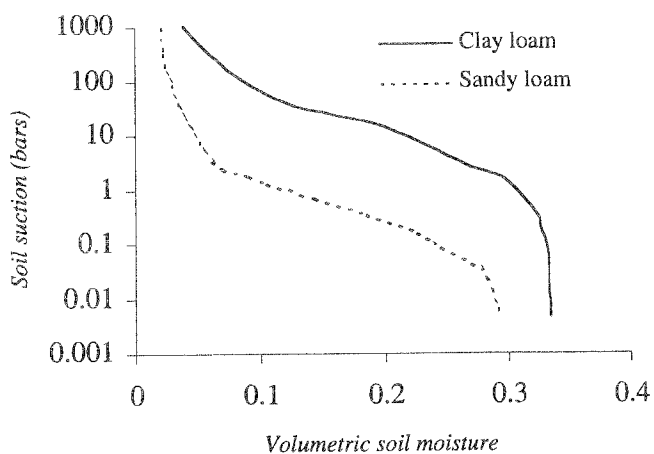


Figure 4.1 Suction moisture curve for two soils, showing the effect of soil texture (redrawn from McLaren and Cameron 1996).

Hysteresis is a characteristic of soil—the equilibrium soil moisture content for a given soil suction is less under wetting conditions than under drying conditions (McLaren and Cameron 1996). This is demonstrated in Figure 4.2.

The amount of water stored in the soil between wilting point and field capacity is the available water, i.e., it is available to plants for transpiration. Wilting point and field capacity are not absolute limits on the soil water content

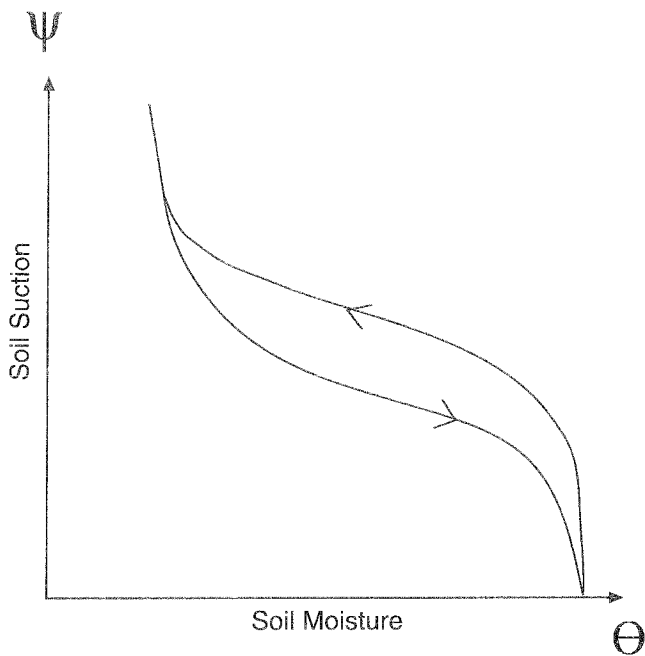


Figure 4.2 Idealized suction-moisture curve for soil. The arrows on the diagram indicate hysteresis: a difference in relationship between a soil that is wetting up (moving from left to right on x-axis) and drying (right to left). In reality there are more than two curves, even for the same soil, so that a direct relationship between soil moisture and soil suction is difficult to derive.

(i.e., a soil can be drier than the wilting point and wetter than the field capacity) but they do represent the range of limits normally observed in the field.

Infiltration, evaporation and percolation from a soil are controlled by the amount of water already present in the soil; the soil moisture level is thus an extremely important characteristic to measure. Unfortunately, it is also highly variable. Spatial variation in soil moisture is a result of spatial variation in the soil physical properties that control moisture levels (porosity, pore-size distribution, soil mineralogy) and topography. As a relatively fixed variable, topography can be accounted for, but the variation in soil physical properties is much harder to assess. Techniques are available to measure soil moisture changes over time extremely accurately (e.g., manual sampling, neutron probes, time domain reflectometry techniques), but they are always applied at a particular point. The scaling up of point measurements to areal estimates is a thorny problem for hydrology that Murray *et al.* (1975) have attempted to solve for soil moisture. Sensors that measure parameters that can

be linked to soil moisture over an area have been mounted on several satellites. Analysis of these data can assess soil moisture under optimum conditions (flat ground with little vegetation cover) at the hectare scale (Davie *et al.* 2001), but they detect only near-surface soil moisture and conditions are rarely optimal. The linkage between near-surface and root-zone soil moisture levels is not easy to establish. An intensive study of soil moisture distribution at Mahurangi (Northland) showed a poor correlation between soil moisture in the 0–6 cm zone and the 0–30 cm zone (Wilson *et al.* 2003). A future challenge is to measure spatial soil variations at a scale relevant to hydrological investigations.

Soil water drainage

A fully wet soil will drain in a non-linear fashion (Fig. 4.3). If gravity were the only force at work within a soil, then all water would drain out and the shape of the curve in Figure 4.3 would be linear. The fact that the soil drainage rate decreases with time is due to retarding forces that act against gravity—these are referred to as capillary forces. Capillary forces are due to the adsorptive forces (e.g., electrostatic attraction) that attract the liquid water to soil mineral faces and the surface tension of the liquid water. The initial high drainage rate occurs as gravity drains water from the large pores within a soil. Capillary forces within these large pores are small compared to gravity. After this “easy drainage” has occurred the water starts to drain from smaller size pores in which the capillary forces are much higher, until eventually the capillary forces become greater than gravity and the drainage curve reaches the near flat point evident in Figure 4.3. This flat point (reached at about day 5 in Figure 4.3) is the field capacity. The length of time taken for the soil moisture to drain from saturation to field capacity is dependent on the pore-size distribution and soil structure.

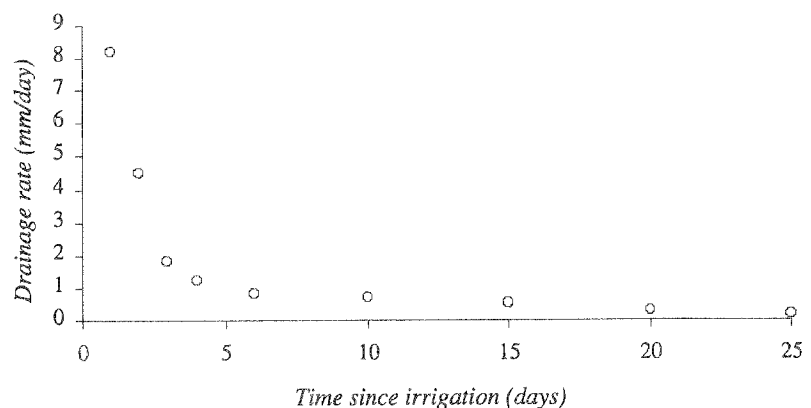


Figure 4.3 Drainage from an irrigated Templeton soil (adapted from Reid *et al.* 1984)

RUNOFF PROCESSES

Runoff is a generic term describing the movement of water from where it falls as precipitation to where it reaches a stream or river channel. Figure 4.4 shows the main runoff processes observable at a hillslope scale. It is common mistake to refer to runoff as only surface runoff—runoff also includes water moving below the ground surface in either the saturated or unsaturated zone.

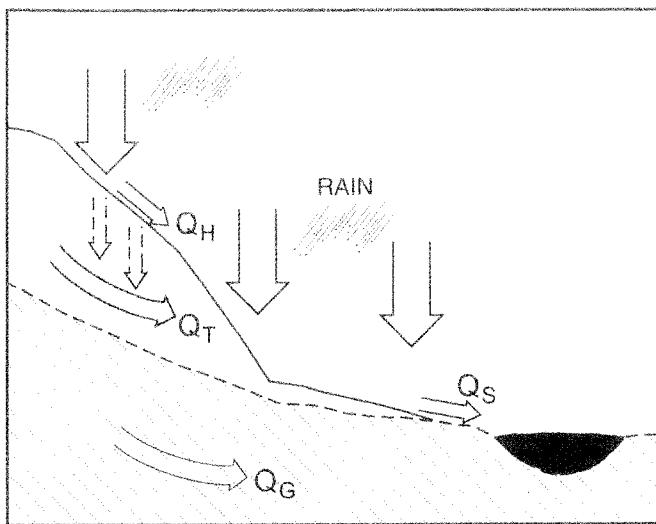


Figure 4.4 Runoff processes. The subscripts for Q refer to: Hortonian overland flow (H), saturation overland flow (S), throughflow (T) and groundwater flow (G).

Overland flow

Water that flows over the ground surface is referred to as overland flow. Two overland flow mechanisms generate surface runoff: infiltration excess and saturation overland flow.

Infiltration excess overland flow

The first mathematical representation of overland flow was by Horton, an engineer who worked mainly in the American mid-west. Horton (1933) proposed that overland flow occurs when the rainfall rate exceeds the infiltration rate of the soil surface and that this water flows over the surface as a thin film of water until it reaches a channel of some form. This simple formulation allows for the amount of overland flow to be calculated with knowledge of the surface infiltration rate and the rainfall intensity, both of which are reasonably simple to measure. In recognition of the pioneering work of Horton in formulating a mathematical understanding of overland flow, in-

filtration excess overland flow is often referred to as Hortonian overland flow.

Infiltration rate is a time-dependent property of the soil, although, as with many soil properties, measurements show a huge spatial variability. The generalized shape of the infiltration curve can be seen in Figure 4.5. The initial high infiltration rate slows down as the large, empty soil pores fill with water. This part of the infiltration curve is due to soil sorptivity—the capability of soil to absorb water (McLaren and Cameron 1996). Once the infiltration curve reaches a steady state, the infiltration rate is due to the ability of the soil to transmit water through it. This is sometimes referred to as the infiltration capacity, although this terminology is confusing, as it is not a capacity *volume* but the *rate of flow* when the soil is filled to capacity. Infiltration capacity varies with soil texture. Rawls *et al.* (1993) report infiltration capacities of 5 mm/hour for a clay soil and 25 mm/hour for a sandy loam.

The spatial variability of infiltration is due to the high spatial variability in soil structure. Webb (1989) traced the wetting front in a 32-m trench dug into a Templeton soil (Canterbury) after rainfall. The average wetting front was at around 20 cm depth, however in three places it had exceeded 60 cm in depth and there was a high degree of variation. The uneven wetting front was attributed to micro-topographic variations at the soil surface and channelling of water by pasture plants; there was no evidence of macropore flow (Webb 1989).

Simultaneous measurements of soil infiltration rate and rainfall intensities frequently show that infiltration rates are higher than rainfall rates. Selby (1970) reported steady-state infiltration rates of between 60 and 600 mm/hour on short grazed pasture on yellow-brown pumice soils in the central North Island. The values were higher for ungrazed grass and under kanuka (Selby 1970). Basher and Ross (2001) reported infiltration rates of 400 mm/hour in

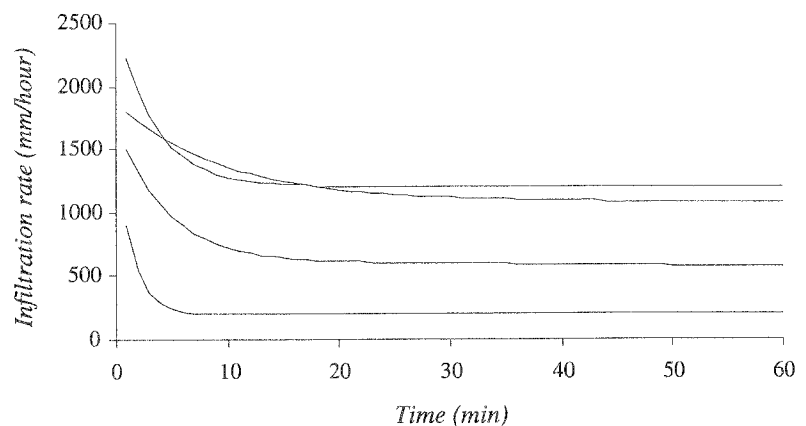


Figure 4.5 Examples of measured infiltration rate curves, adapted from Selby (1970), showing variability in surface infiltration rates for a yellow-brown, pumice-derived soil in the central North Island.

onion beds near Pukekohe; these rates increased during the growing season to as high as 900 mm/hour.

Maximum rainfall intensities for New Zealand are in the region of 20–60 mm/hour, with the highest recorded being 107 mm/hour at Whenuapai in 1966 (NZ Meteorological Service 1979). A comparison of rainfall intensities and soil infiltration rates suggests that infiltration excess overland flow seldom occurs. This is in line with the observation that during rainfall it is uncommon to see a thin layer of water flowing over the surface as overland flow—most rainfall does infiltrate. Horton was not wrong, as there are times when the rainfall rate exceeds the infiltration rate, particularly in the types of environment in which he worked. Soil compaction may lead to low infiltration rates, e.g., Basher and Ross (2001) show a decline in infiltration to as low as 0.5 mm/hour in compacted wheel tracks. Infiltration may be low during extremely intense rainfall during storms, and where human modification to the earth surface leads to low infiltration rates (e.g., urbanisation and road paving). It is also important to note that rainfall intensities quoted above are for an hour's duration; much higher intensities are often recorded for shorter time intervals and these may lead to a short burst of infiltration excess overland flow.

Another cause of infiltration excess overland flow is that soils may exhibit hydrophobicity or water repellancy. This has often been observed in New Zealand soils (e.g., van't Woudt 1959; Wallis *et al.* 1993) but has not received a great degree of study. Clothier *et al.* (2000) showed how a yellow-brown earth/loam of the Ramiha series (Manawatu) changed from an initial steady-state infiltration rate of 2 mm/hour to 14 mm/hour as the soil's water repellancy broke down. Hydrophobicity has been linked to several factors, including the presence of mycorrhizal fungi and swelling clays such as allophane.

Saturation overland flow

Saturation overland flow was hypothesised by Hewlett and Hibbert (1967) and first described by Dunne and Black (1970). The mechanism of saturation overland flow is that all rainfall infiltrates to reach a groundwater table, and the flux of water reaching the saturated zone causes the groundwater table to rise until it reaches the surface. Overland flow occurs at this stage, through a combination of rainfall falling onto the already saturated soil and water returning from the saturated soil matrix. In many places in a catchment the groundwater table is too deep for it to reach the surface with the addition of rainfall, so saturation overland flow will occur only in limited areas. Particular areas that are likely sites for saturation overland flows are places where the groundwater table is frequently close to the surface—near stream margins, topographic convergence zones (e.g. valley heads), at sharp breaks of slope,

and where perched water tables develop. These areas are described as variable source areas. In their original paper, Hewlett and Hibbert (1967) referred to variable source areas acting like extended channel networks that allowed rapid delivery of runoff to the stream. McColl *et al.* (1985) mapped variations in the extent of saturated areas over time at a small experimental catchment near Wellington and related this to streamflow characteristics. They found the surface-saturated areas extended during large storms, and that much of the water reaching the stream during the storm was derived from these areas.

Subsurface flow

After water has infiltrated the soil surface it continues to move, either through the soil matrix, or along preferential flow paths; this is referred to as lateral or preferential flow. Flow along both of these paths moves water through the soil mantle and can be referred to together as throughflow. The rate of soil water movement through a saturated soil is described by Darcy's law:

$$Q = -K_{sat} A \frac{dh}{dx} \quad (4.5)$$

in which Q is the amount of water in a unit time flowing through a cross-sectional area A , K_{sat} is the saturated hydraulic conductivity, and dh/dx is the hydraulic gradient (change in hydraulic head with distance). Saturated hydraulic conductivity is a measure of the ability of the soil to transmit water, which is controlled by the pore-size distribution within the soil and the connectivity of pores.

In soil physics an important distinction is made between K_{sat} , the saturated hydraulic conductivity and K^{40} which is the hydraulic conductivity at a suction of 40 mm of water. K^{40} is essentially the hydraulic conductivity through the main soil matrix, excluding pores greater than approximately 0.75 mm in diameter (Watt and Burgham 1992).

The movement of water through a soil mantle water is not just vertical. With a sloping water table (e.g., Fig. 4.4) the hydraulic gradient is controlled by the slope of the water table, and water moves down slope. The movement of water through a saturated soil matrix is not rapid. Kelliher and Scotter (1992) reported a K_{sat} value of 13 mm/hour for subsoil in a fine sandy loam, which they term as "common for New Zealand soils".

Ward (1984) used the analogy of a thatched roof to describe throughflow. When straw is placed on a sloping roof, water moves very efficiently to the bottom of the roof, without visible overland flow. This is due to preferential flow along the straws. Measurements of hydraulic conductivities in soils normally show a decrease with depth, which leads to a preferential flow direction down-slope rather than vertically downward. This suggests that the analogy of a thatched roof is valid, however preferential

flow is still a relatively slow process if the water is moving through the soil matrix.

The slow movement of water down a hillslope may maintain a wetter area at the bottom of the slope, the so-called saturated wedge (this can be seen at the base of the slope in Figure 4.4 where the water table is close to the surface). It is caused by the decreasing hydraulic gradient as the slope angle decreases at the base of a slope. Here the rate of flow slows down and hence fills a greater volume of soil within a time period. The saturated wedge provides an area in which saturation overland flow may occur, i.e., a potential variable source area.

It is also possible for a saturated wedge to develop as a result of the infiltration of overland flow at the base of a slope, creating a wetter area. This has been observed during summer storms at Whatawhata (near Hamilton), where runoff plots on steep slopes have overland flow, but less overland flow is recorded on more gently sloping plots, and none in the ephemeral stream channel (Fig. 4.6).

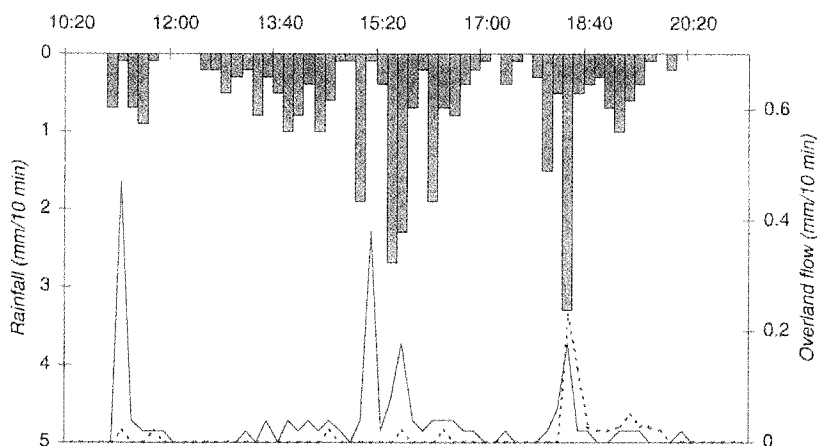


Figure 4.6 Overland flow from surface runoff plots at Whatawhata during a summer storm (December 2002). The solid line is from a steep runoff plot (34°); the dashed line is for a more gently sloping plot (12°). During this storm no flow was recorded in the ephemeral stream channel downslope. The overland flow observed on the hillslopes has probably re-infiltrated by the time it reaches the stream channel.

When a soil is unsaturated, a $K(\theta)$ term replaces K_{sat} (Equation 4.5). The unsaturated hydraulic conductivity ($K(\theta)$ in Equation 4.6) is a function of pore size, connectivity and soil moisture status.

$$Q = -K(\theta)A \frac{dh}{dx} \quad (4.6)$$

The relationship between $K(\theta)$ and soil moisture is strongly non-linear, i.e., a small increase in soil moisture may lead to an increase in flow that is several orders of magnitude higher. However the maximum flow rate is still the same as for saturated flow (i.e., K_{sat}) and is relatively slow.

More rapid movement of water through soils may occur through connected macropores such as wormholes and cracks, which allow rapid infiltration of water into the soil. It is unlikely that macropores form a continuous network that allows rapid movement of water all the way down a hillslope, but they may deliver water rapidly to an impermeable boundary along which water may move as lateral flow. Observations of lateral flow in New Zealand soils go as far back as van't Woudt (1954) for ash-derived soils near Taupo, and the combination of macropores and lateral flow has been put forward as a possible explanation for hillslope flowpaths in the Maimai catchment, Westland (McGlynn *et al.* 2002).

STREAMFLOW GENERATION

This chapter began by stating the most fundamental question for a hydrologist to answer—how much rainfall ends up as streamflow? In Figure 4.7 a time series of rainfall and streamflow shows that the relationship between the two variables is seasonally variable and complex at many timescales.

The complexity in streamflow response can be attributed to two main factors. First, spatial and temporal variability in soil properties that control the rainfall-runoff relationship lead to a large variability in runoff leading into the stream. Second, streamflow can be thought of as the final term in the water balance equation, so that streamflow variability is a sum in the variability of all other processes (rainfall, evaporation, storage).

A hydrograph is a time series of river or stream flows (Figs. 4.7 and 4.8). The water that flows down the river during peaks in the hydrograph is referred to as stormflow. Baseflow is the stable or slowly declining flow between peaks, which conventionally is considered to extend through and beneath peaks (Fig. 4.8). The method used to distinguish between stormflow and baseflow is called hydrograph separation—the most common technique for separation involves drawing a somewhat arbitrary line that separates stormflow peaks from the baseflow (Fig. 4.8 and Chapter 6). It is also possible to separate hydrographs by analysing isotopic or chemical tracers to determine the proportion of “old” and “new” water in the stream. This requires continuous measurement of an isotope (e.g., ^{18}O or ^2H) or chemical tracer (e.g., Cl, Si). In these techniques the assumption is made that the isotopic or chemical signature of the “new” water is similar to rainfall and that of “old” water is closer

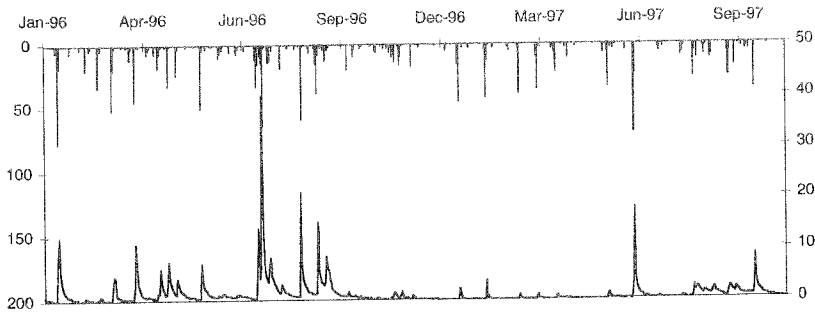


Figure 4.7 Streamflow and rainfall record for the Waiwhero catchment (near Motueka). The rainfall and streamflow relationship is complex and shows marked seasonal variation.

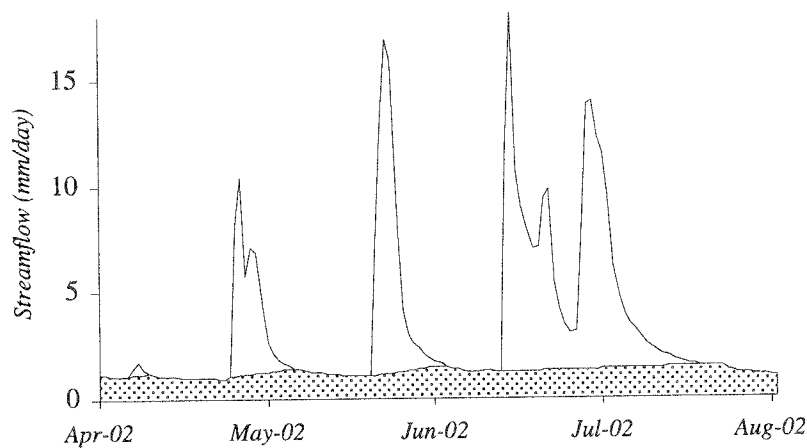


Figure 4.8 Hydrograph record for the Maitai River, near Nelson (data provided by Tasman District Council). Baseflow is the shaded portion, with stormflow occurring above the shaded area.

to groundwater, or at least very different from the rainwater (Kendall and McDonnell 1998).

Baseflow generation

Baseflow is the water that continues to flow down the river irrespective of rainfall. It maintains river levels during summer months, and normally is generated from groundwater and hillslope throughflow. Rivers that have a high baseflow index (i.e., a large percentage of their total flow occurs as baseflow) have a large groundwater store associated with them, e.g., limestone geology or other stores such as lakes and glaciers (Chapter 6).

Isotopic dating of baseflow usually shows that the water is old, supporting the idea that it is derived from groundwater or hillslope throughflow. For example CFC and ^{18}O dating shows that baseflow in the Mangaotama stream, Waikato (the perennial stream below the runoff plots; see Figure 4.6) has an average residence time of 7 years (M. Stewart, pers. comm.).

The drainage from a saturated soil (Fig. 4.3) does not stop once field capacity is reached; it continues but at a much slower rate. The same process occurs with baseflow—the water draining from a hillslope or bedrock decreases with time (unless it is refilled from above) so that there is a typical baseflow recession curve (see Figs. 4.7 and 4.8).

Stormflow generation

There is much more controversy over how stormflow is generated than there is for baseflow generation. The Maimai catchment in Westland has been the centre of world leading research in the subject of stormflow generation. McGlynn *et al.* (2002) provided an excellent overview of hillslope runoff research at Maimai, some of which is covered here.

The first views on how stormflow is generated were that stormflow is caused by infiltration excess (or Hortonian) overland flow across the surface entering the stream channel network. However rainfall rates do not often exceed infiltration rates (certainly not at mid-latitudes) and overland flow seldom is observed at the hillslope scale, so this cannot be a complete explanation. The concept of variable source areas and saturation overland flow (Hewlett and Hibbert 1967) helped explain the lack of observable overland flow across the whole

hillslope, but calculations of water volumes suggest that variable source areas would have to be larger than those commonly observed to produce enough water for stormflow peaks. Pearce and McKerchar (1979) analysed stormflow from a variety of small catchments throughout New Zealand and found that saturation overland flow accounted for all stormflow during small events but not during larger events.

Two other field observations make it difficult to explain stormflow purely in terms of overland flow:

1. Subsurface flow has been observed as a stormflow-generating mechanism (e.g., van't Woudt 1954), although theoretical analysis suggested that it should be too slow to generate much beyond the recession limb of the storm hydrograph.
2. Tracer studies indicate that a considerable amount of stormflow water is “old”, i.e., it has resided in the soil or groundwater for longer than the duration of the storm (e.g., Martinec *et al.* 1974).

The Maimai catchment study (near Reefton in

Westland) was established in 1974 for research into the effects of logging native forest and replanting the area with different non-indigenous species. The installation of hydrological measuring equipment and the fact that rainfall events are frequent made it an ideal place for studying stormflow generation mechanisms in detail. Annual rainfall is approximately 2600 mm, with an average of 156 rain days a year, and stormflow makes up 65% of the total streamflow (Rowe *et al.* 1994; Pearce *et al.* 1986).

Mosley (1979, 1982) investigated the role of macropores as conduits for rapid movement of rainfall to the stream. Observations of macropore flow rates using cut soil faces and dye tracers suggested that rainfall could travel down the short, steep hillslopes at Maimai in less than 3 hours (*i.e.*, within the time frame of a storm). However, subsequent chemical and isotopic analysis of streamflow, rainfall and water exiting the cut soil pit faces showed that the majority of measured streamflow was “old” water, suggesting that rapid extensive macropore flow was not the main mechanism for stormflow generation (Pearce *et al.* 1986).

McDonnell (1990) investigated this further, in particular looking at groundwater ridging—the groundwater ridging hypothesis was put forward by Sklash and Farvolden (1979) as a possible mechanism for large amounts of old water being part of stormflow. The idea relies on the strongly non-linear relationship between soil water content and soil water suction. At the capillary fringe (the area of soil immediately above the water table) the addition of a small amount of water by infiltration can lead to a large and rapid change in soil water suction, which can rapidly become a positive pressure (*i.e.*, instead of water being sucked into the soil, it is expelled). At the stream margin the capillary fringe is close to the surface, and the addition of a small amount of water from rainfall may lead to a rapid change in soil suction and the release of large volumes of groundwater into the stormflow. Although this could be observed at Maimai, the amount of water held near the stream prior to an event was not large enough to account for all of the old water, which suggested another mechanism might be working (McDonnell 1990).

The concept of piston, or translatory, flow in hillslopes has been around for a long time (*e.g.* Horton and Hawkins 1965), and had been observed in laboratory experiments with soil columns (*e.g.*, Germann and Beven 1981). According to the piston flow concept, as water enters the top of a soil column (*e.g.*, through infiltration) it displaces old water at the bottom of the column as stormflow. The analogy is drawn with a piston where downward pressure from the piston at the top of the piston chamber leads to a release of air at the bottom of the chamber. Although this is a simple analogy there are considerable differences in a

field situation. For example, the soil on a hillslope is not confined on either side or above by impermeable boundaries, as in a piston chamber. This would suggest that the addition of water at the top of the hillslope or soil column would be accommodated elsewhere on the slope, and there would be very little piston displacement. The accommodation would occur through the water being absorbed into dryer soil above or forced onto the surface.

McGlynn *et al.* (2002) presented a summary conceptual model of runoff mechanisms on Maimai hillslopes that combines many of the features described above (Fig. 4.9). In this model there is rapid infiltration of water through macropores to reach the bedrock. Here, a form of piston flow occurs, as the saturated zone at the base of the soil mantle is confined by the soil matrix above it. At the bedrock interface there may be a network of macropores, or the conditions may be equivalent to those in a confined aquifer, in that the soil matrix above has a much lower hydraulic conductivity. There is also a mixing of the new water with old water sitting in bedrock hollows, creating a rapid movement of old water into the stream during storms.

One important point to emphasise about piston or translatory flow contributions to stormflow is that the water entering the stream is not the same as the rain falling on the hillslope. The mechanism is a pressure wave that can be modelled as a kinematic wave. This has important implications for modelling the movement of pollutants and nutrients, as pollutant and/or nutrient-rich water entering the soil profile will take a long time to reach the stream, even though rainfall generated a stormflow response in the channel system.

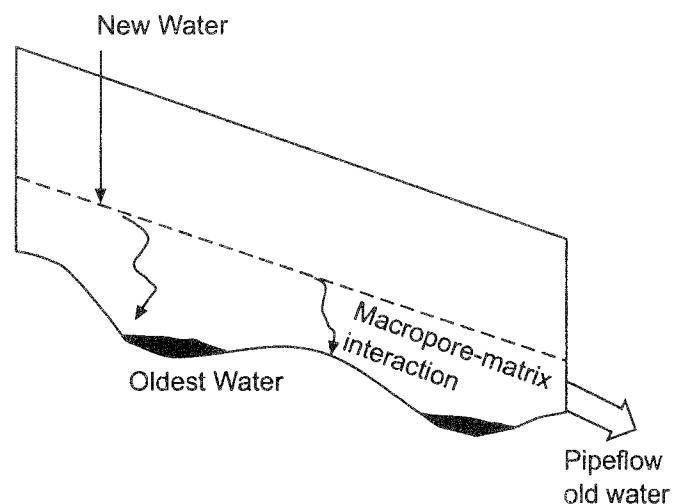


Figure 4.9 Summary hypothesis for hillslope stormflow mechanisms at Maimai (adapted from McGlynn *et al.* 2002).

How relevant are the Maimai stormflow generation studies?

The studies at Maimai have been extremely important in influencing hydrological thinking around the world. However, the conditions at Maimai are far from generally applicable elsewhere. The main study catchment has short, steep slopes (approximately 300 m, with angles of around 35°) and is in an area of high and frequent rainfall. The soils are extremely porous (infiltration rates in excess of 1600 mm/hour have been measured) and remain within 10% of saturation for most of the year (Mosley 1979). These conditions are not common elsewhere in New Zealand and it would be difficult to generalise the concepts beyond Maimai. In their review of the stormflow response of different small catchments around New Zealand, Pearce and McKerchar (1979) presented graphs that showed Maimai had quite different quickflow versus catchment area relationships to other catchments of similar size and land use. However, even though Maimai may not be representative of the rest of New Zealand, an important concept to come from the research is that under conditions ideal for stormflow generation the mechanisms are still extremely complex and spatially variable. This is true wherever in the world the study is taking place.

SUMMARY

There is no single mechanism that describes how rainfall reaches a stream. One of the most important controls on runoff and streamflow generation is the soil moisture status of the soil. This influences the infiltration rate and the hydraulic conductivity through the soil matrix. At present, no measurement techniques are suitable for assessing soil moisture at a spatial scale appropriate for catchment hydrology; we have to infer spatial distribution from a series of point measurements.

Baseflow, the steady flow of a river between storm peaks, is derived from subsurface flows, either from deep groundwater or water moving through the soil mantle. The relative importance of overland flow and subsurface flow in stormflow generation varies from site to site and between storms. New Zealand has hosted several important studies on runoff mechanisms, the best known of which is in the Maimai catchment in Westland. These studies have highlighted the complex nature of runoff generation and the importance of long-term study sites to the understanding of hydrological processes.

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Chapter 3

Evaporation and transpiration

David Scotter and Francis Kelliher

INTRODUCTION

Evaporation is the change of liquid water into vapour, and it includes transpiration by vegetation as well as the evaporation of free water from wet leaves, moist soil and lakes. Evaporation from vegetation also is called evapotranspiration. This chapter introduces the principles governing evaporation and illustrates them with New Zealand examples. Evaporation models are presented which both incorporate established theory and provide some essential tools for hydrology.

To allow comparison with rainfall, the evaporation rate is usually expressed as the volume of water lost from a unit of land area over a unit of time, which can be expressed as depth per unit time, commonly given in mm/day. Evaporated water returns as precipitation, so globally the two are in balance, averaging about 1,000 mm/yr (Sturman *et al.* 2001). However terrestrial evaporation averages only about 500 mm/yr (Kelliher *et al.* 2001), with a third of the land area producing less than the average and being classified as semi-arid. Imbalances between rainfall and evaporation drive the hydrological cycle, and are responsible for the distribution of arid and humid regions and seasons. More extreme imbalances cause floods and droughts. To understand freshwater, where it comes from and where it goes, we need to understand evaporation. It is a large component in calculations of the soil water balance, which are used to analyse groundwater recharge and stream flow from catchments, to model pasture production, and to schedule irrigation.

A lot of fresh water is used for irrigation, which increases evaporation. In 1937, New Zealand's first major irrigation scheme began operations near Ikawai, northwest of Oamaru; here water drawn from the Waitaki River was applied to 1,863 hectares of pasture. In 2000, 82% of New Zealand's irrigated land area was in Canterbury and Otago (416,260 hectares of their total area of 7,638,733 hectares:

Christina Robb, unpublished Ministry for the Environment report, 2000). In these two regions, 85% of the 340 m³/s of allocated water (80% of New Zealand's total) was used for irrigation, with 72% of the land area used for pastoral agriculture.

RULES AND TOOLS

Energy and water supply limitations on evaporation

Two ingredients are needed for evaporation, water and energy: the rate of evaporation tends to be limited by the rate of supply of one of these. With New Zealand's generally high and reliable rainfall, the soil is often moist, so energy is often the limiting input. To understand evaporation we need to know the source of that energy. Evaporation needs a lot of energy—to evaporate a kg or litre of water takes about 2.5 MJ, so 2.5 MJ/m² is required per mm of evaporation. By comparison, a 1 kW heater takes 40 minutes to give out 2.5 MJ. Evaporation involves an input of latent or hidden heat, in contrast to the sensible heat that we can sense because it warms the air.

The energy for evaporation comes from the sun, much of it from direct solar radiation—sunlight—falling on the evaporating surface. We can see the earth's surface during the day because it reflects some sunlight. A lake reflects about 7%, a pine forest about 17%, and pasture about 24% of the incident short-wave solar radiation. There is also an exchange of invisible long-wave radiation between the earth's surface and the atmosphere. The difference between the incoming and outgoing radiation is called the net radiation (R_n), which is the energy absorbed by the earth's surface. R_n is usually strongly positive during the day, but weakly negative at night. Over a typical 24-hour period in summer in New Zealand the incoming solar radiation or irradiance usually is 15–30 MJ/m², and for

vegetation R_n is 60% – 70% of the solar radiation. Both the irradiance and the net outgoing long-wave radiation are reduced by cloudiness.

Ignoring the small but vital amount used for photosynthesis, the absorbed net radiation does three things—it causes evaporation, it warms the air above the evaporating surface, and it warms the vegetation and soil, or in the case of a lake it warms the water. During the night when R_n is negative it does the opposite, cooling the air, the vegetation and soil, and the lake water, sometimes causing negative evaporation (dew fall). This energy budget can be expressed by:

$$R_n = \rho_w \lambda E + H + G \quad 3.1$$

where R_n is the net radiation (W/m^2), ρ_w is the density of water ($1,000 \text{ kg}/\text{m}^3$), λ is the latent heat of vaporisation ($2.45 \text{ MJ}/\text{kg}$ at 20°C), E is the evaporation rate (m/s), H is the sensible heat flux into the air (W/m^2), and G is the heat going into the vegetation and soil or the lake water (W/m^2).

Measured values for R_n , $\rho_w \lambda E$, H , and G for a pasture near Palmerston North on a sunny November day are

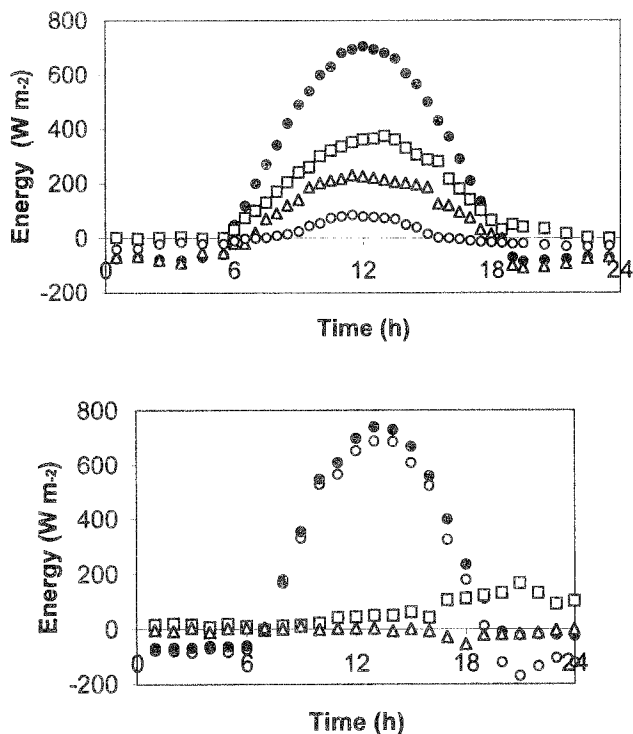


Figure 3.1 Net radiation R_n (*), energy used for evaporation $\rho_w \lambda E$ (○), energy going to heat the atmosphere H (△), and energy going to heat the soil or lake G (□), on a sunny day (a) for well-watered pasture near Palmerston North in November (from Plant Physiology Division, DSIR), and (b) for Lake Tekapo in February (after Oliphant 2000).

shown in Figure 3.1(a), and for the surface of Lake Tekapo on a sunny February day in Figure 3.1(b). For the pasture, the incoming solar radiation, estimated from the measured sunshine hours, was $25 \text{ MJ}/\text{m}^2$. Summing the area under the curves over the 24-hour period shows that the net radiation was $17 \text{ MJ}/\text{m}^2$ or 66% of the solar radiation, and G was just $0.3 \text{ MJ}/\text{m}^2$. Over periods of 24 hours, G in equation (3.1) often is negligible for pasture or forest. In Figure 3.1(a) roughly 70% of the net radiation was used for evaporation ($\rho_w \lambda E$), with the evaporation rate peaking in phase with the net radiation at around noon. The remaining 30% of the absorbed radiation warmed the atmosphere (H). If the soil had been drier and the pasture under stress, less energy would have been used for evaporation and more for heating the atmosphere.

The net radiation values for Lake Tekapo in Figure 3.1(b) are very similar to the R_n values for the pasture in Figure 3.1(a), but the values for the other three variables are very different. On that particular day, 78% of the net radiation went into warming the lake (G), due to the lake water's transparency to sunshine, and its large thermal mass. The energy used for evaporation ($\rho_w \lambda E$) was only 25% of R_n , and the evaporation rate was independent of R_n , peaking at 9 pm when the wind speed was greatest. Further, averaged over this particular 24 hours, H was negative, as the colder lake gained heat from the warmer air above.

If we can find ways of estimating how net radiation is partitioned between evaporation, heating the air, and heating the vegetation and soil or the lake water, Equation 3.1 can lead to a way of estimating evaporation from radiation data, as we shall see below. The partitioning is not always simple, however, as the quite different values for H and G in Figures 3.1(a) and 3.1(b) show. Also, extra energy for evaporation can be carried inland by the wind from the relatively warm sea surrounding New Zealand, a process called advection. This is illustrated by the fact that high evaporation rates occur from a forest during rainfall, and these rates can occur during daytime or at night.

Evaporation, humidity and air flow: the bulk aerodynamic transfer equation

A commonly used equation that describes evaporation is the bulk aerodynamic transfer equation. For a wet evaporating surface it can be written as (McNaughton *et al.* 1979):

$$E = (e_s^* - e_a) C_a / (\gamma \lambda r_a) \quad 3.2$$

where e_s^* is the saturated vapour pressure at the wet surface (kPa), e_a is the vapour pressure at some height in the air above, C_a is the specific heat capacity of air ($993 \text{ J}/\text{kg}/^\circ\text{C}$), γ is the psychrometric constant ($0.067 \text{ kPa}/^\circ\text{C}$ at altitudes less than 200 m), and r_a is the aerodynamic resistance (s/m).

The vapour pressure (e_a) is a common measure of how much water vapour air contains, and the saturated vapour pressure (e_s) is the vapour pressure when air holds as much water vapour as possible at that temperature. Any excess will have condensed into water droplets as fog or dew. The amount of water vapour that air can hold depends strongly on its temperature, as Figure 3.2 shows. If saturated air at 10°C is heated to 20°C while e_a stays the same, e_s almost doubles from 1.23 kPa to 2.34 kPa, so the relative humidity (e_a/e_s) almost halves, going from 100% to 53%.

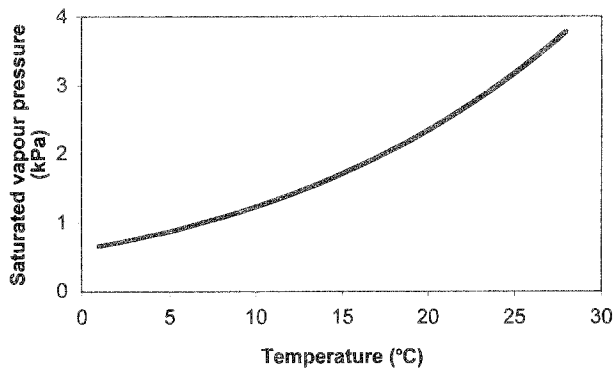


Figure 3.2 The relationship between saturated vapour pressure (e_s) and air temperature.

As evaporation involves the movement of water vapour away from the evaporating surface into the drier air above, Equation 3.2 states that the evaporation rate depends on the difference between the vapour pressure in the air at the saturated surface and in the air above, and on how easily the water vapour can be transported away from the evaporating surface. The windier and more turbulent the air, the less resistance there is to vapour movement, and the smaller is the parameter called aerodynamic resistance (r_a). Also, the taller the vegetation, the more it projects up into the more turbulent air, and the lower is the aerodynamic resistance. Thus if we know the difference between the vapour pressure at the evaporating surface and in the air above, and if we can estimate the aerodynamic resistance from the wind and the crop geometry, we can estimate the evaporation rate using Equation 3.2. The problem is that we usually do not know the temperature at the evaporating surface, and therefore the saturated vapour pressure at that surface. In the next section we see how this problem can be circumvented.

Evaporation and weather stations: The Penman-Monteith and Priestley-Taylor models

Evaporation from vegetation is difficult to measure, and is not measured at weather stations. The daily data usually available from weather stations are rainfall, the maximum and minimum air temperatures and the vapour pressure at a height of about 1.5 m, the solar radiation or number of sunshine hours, and the wind run. Some stations do

measure the evaporation from a large water-filled pan, but as we shall see, the evaporation from vegetated surfaces and lakes is different to that from a raised pan. A lot of effort has gone into finding ways to estimate evaporation from the data collected at weather stations. In 1948 Penman developed a way of doing this for wet surfaces. He argued that the transport of both heat and water vapour was caused by air turbulence, so the same aerodynamic resistance can be used to describe both. With this insight he combined the energy balance Equation 3.1 and the bulk aerodynamic transfer Equation 3.2 to give an approximate evaporation equation that needed only commonly available weather data. It did not need the temperature at the actual evaporating surface. In 1965 Monteith included a (bulk) surface or canopy resistance in the Penman equation, enabling it to be used for vegetation under a wide range of conditions. This resistance roughly accounts for the ease with which water vapour escapes from leaves through the small holes (stomata) on their surface. The Penman-Monteith equation can be written (Allen *et al.* 1998):

$$E = \frac{s(R_n - G) + \rho_a c_a (e_s - e_a) / r_a}{\rho_w \lambda [s + \gamma(1 + r_s / r_a)]} \quad 3.3$$

where s is the slope of the curve relating saturated vapour pressure to temperature at the prevailing air temperature (kPa/°C), ρ_a is the density of air (kg/m³), c_a is the specific heat of air (993 J/kg/°C), and r_s is the canopy or surface resistance (s/m). To use the Penman-Monteith equation, the minimum data requirements are daily sunshine hours or solar radiation, air temperature and humidity, wind speed, and information about the vegetation and its water status.

An alternative, simpler equation was proposed by Priestley and Taylor in 1972 for oceans and lakes, and for large areas of vegetation adequately supplied with water:

$$E = \frac{\alpha s (R_n - G)}{\rho_w \lambda (s + \gamma)} \quad 3.4$$

where α is a dimensionless parameter. For $\alpha = 1$, which applies for an extensive wet surface supplied with constant available energy ($R_n - G$), a dynamic equilibrium is reached with the saturated atmosphere, so that the air warms just enough to be able to hold the extra water vapour produced by evaporation. This is often called the equilibrium evaporation rate, and provides a lower limit for the evaporation rate from an extensive moist surface. Considering daytime conditions, Priestley and Taylor argued that an upper limit for α comes from the observation that usually $H > 0$, so from Equation 3.1, $\rho_w \lambda E < (R_n - G)$. Thus in Equation 3.4, $\alpha < (s + \gamma) / s$, that is less than 1.5 at 20°C, or 2.1 at 5°C. They then examined a wide range of published evaporation data for extensive

moist surfaces, and found that $\alpha \approx 1.26$. The reason that $\alpha > 1$ is because extra energy to that coming directly from the sun is supplied by advection, energy brought from elsewhere by the wind. As stated earlier, a common source for such extra energy in New Zealand is the relatively warm surrounding sea. The Priestly-Taylor equation can be used when only sunshine hour data (to estimate R_n) and air temperature data (to find s) are available, if G is either known or is negligible.

The cooler the air, the lower is s . At 5°C , s is 77% of its value at 15°C . When evaporation is described by Equation 3.4, s/γ is the ratio of $\rho_w \lambda E$ to H . Therefore, for a given energy input, the colder the temperature, the lower the evaporation rate, and the higher the sensible heat loss. This is one reason for the low evaporation rate from snow and ice. Another reason is the large fraction of solar radiation that is reflected rather than absorbed by such surfaces.

There is an apparent contradiction between Equations 3.2 and 3.4—Equation 3.4 implies that the evaporation rate from a moist surface can be found from just the air temperature and the net radiation less G , while Equation 3.2 that implies that the evaporation rate can be found from just the wind speed, the surface roughness, the saturated vapour pressure at the surface, and the vapour pressure in the air above. The reason that both equations can simultaneously be valid is that the incoming radiation (less G) tends to determine the vapour pressure at the moist surface and in the air above.

The above is a very simplified and abbreviated treatment of a complex topic. Raupach (1999) provides a more detailed review, and references to the key papers.

MEASURING EVAPORATION

There is currently no easy way to routinely measure evaporation. Here we briefly describe the three main approaches used in research. To separate the total evaporation into its components of interception (evaporation from wet foliage), transpiration, and ground surface evaporation, the three approaches are sometimes used in combination (e.g., Kelliher *et al.* 1992).

The first approach uses micrometeorology. It can involve measuring the net radiation and soil heat flux, and the air temperature and vapour pressure at two heights above the vegetation. The ratio of sensible to latent heat, $H/\rho_w \lambda E$ (called the Bowen ratio), can then be estimated and rearrangement of Equation 3.1 gives E . The data in Figure 3.1(a) were obtained this way. A more direct approach is the eddy covariance technique. Fast-response wind and humidity sensors (commonly 20 Hz or 20 measurements per second) are deployed above the evaporating surface to measure the high-frequency turbulent fluctuations in vertical wind speed and humidity.

E then is simply taken as the covariance of the wind and humidity fluctuations over a period, commonly 30 minutes. It is this turbulence that carries water vapour away from the evaporating surface. The data in Figure 3.1(b) were obtained in this way.

The second approach uses lysimeters. A lysimeter consists of soil confined in a container, inset into the ground so that the soil surfaces inside and outside the lysimeter are at the same level. Rainfall is measured independently. Lysimeters range from about 100 mm deep (to measure evaporation from bare soil or the forest floor) to several metres deep. They are usually planted with a crop or pasture, but sometimes contain shrubs or trees, and drainage water from the base is collected, ideally under suction. Sometimes the lysimeter is weighed to give the change in soil water storage over periods as short as an hour. Evaporation is found as the change in water storage plus rainfall, less drainage. Over periods of several days when drainage is negligible, water content measurements in soil not contained in a lysimeter, obtained using a neutron probe or time domain reflectometer, may be used in a similar way find the evaporation rate. The data in Figure 3.5 were obtained this way.

Lysimeters are usually impractical for larger trees. For them a third approach is often adopted, which measures transpiration by monitoring xylem sap flow. This is usually done by injecting heat into the sap stream at some point and measuring its transport.

As mentioned above, evaporation is measured at some weather stations by using Class A evaporation pans. An estimate of the evaporation rate from well-watered pasture can be obtained by multiplying the pan evaporation rate by a pan coefficient. A problem is that this pan coefficient can vary from 0.45 to 1.0, depending on the wind speed, air humidity, and whether the grass surrounding the pan is well-watered or water-stressed. Allen *et al.* (1998) provide a table of pan coefficient values for various conditions.

PASTURE EVAPORATION

Evaporation from well-watered pasture

To estimate evaporation from well-watered pasture, E_o , we may use Equation 3.3. Following Allen *et al.* (1998), we assume that:

$$r_a = 108/u_2 \quad 3.5$$

where u_2 is the average wind speed (m/s) at a height of 2 m. Given a typical wind speed of about 2 m/s, this implies a value for r_a of about 50 s/m. We also assume that $r_s = 70$ s/m (Allen *et al.* 1998). Figure 3.3 shows daily E_o values calculated this way for Palmerston North in 2000. Typically in New Zealand E_o values fall between 2 and 6 mm/day in summer, and 0.5 to 1.5 mm/day in winter.

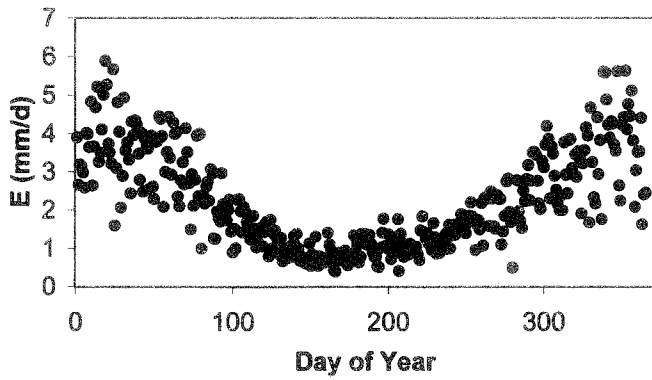


Figure 3.3 Daily values for 2000 of the evaporation from well-watered pasture near Palmerston North, estimated using Equations 3.2 and 3.4.

Average monthly E_o values vary around New Zealand and through the year (Table 3.1). The highest value in Table 3.1 is 156 mm for Blenheim in January, and the lowest is 10 mm for Alexandra in July. The high January value for Blenheim is due to a combination of lots of sunshine, a high average temperature, and a low vapour pressure. The low July value for Alexandra occurs because of a lack of wind there in winter. For any particular month the coefficient of variation for the year-to-year variation in evaporation for well-watered pasture is about 8%, a fifth of the value for rainfall. Thus for periods of 10 days or more, E_o estimates from Table 3.1 can be used with daily rainfall data to estimate when irrigation is needed (Scotter and Heng 2003).

When the E_o values in Table 3.1 are compared with the average monthly rainfall, the main features of the water balance at a site become apparent. Figure 3.4 provides such a comparison for Napier. There is usually excess water in winter but a large deficit in summer, and the management of land on the Heretaunga Plain needs to take both these features into account.

During late spring, summer and early autumn, solar radiation values are relatively high, and for well-watered pasture the first term in the numerator of Equation 3.3 is much greater than the second term. However, during winter there is a lot less solar radiation and the second term dominates. This suggests that during winter most of the energy for evaporation is provided by advection from the surrounding ocean. In winter the average lowland air temperature in New Zealand ranges from 5°C in the south to 12°C in the north (New Zealand Meteorological Service 1983), while the mean surface temperature in the sea surrounding New Zealand is about 4°C higher, ranging from 10°C to 15°C (Uddstrom and Oien 1999). This large difference between average sea and land temperatures disappears in summer.

The Priestley-Taylor approach of Equation 3.4 with $\alpha = 1.26$ gives somewhat different values to the Penman-Monteith values in Table 3.1. For example, for Palmerston North the Priestley-Taylor equation estimates 16% more evaporation over summer, but 24% less over winter.

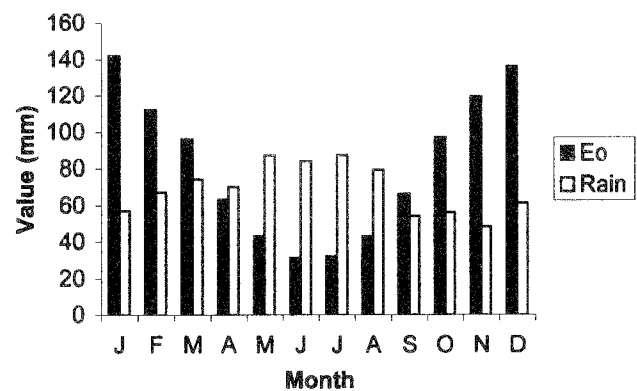


Figure 3.4 Average monthly rainfall and evaporation from well-watered pasture at Napier.

Table 3.1 Average monthly and yearly evaporation rates from well-watered pasture, estimated using Equation (3.3) (after Scotter and Heng 2003).

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
Kaitaia	133	109	97	67	49	36	39	47	63	86	101	125	951
Rotorua	133	109	93	62	41	29	32	43	60	87	107	125	918
Tauranga	141	116	101	69	46	35	40	47	66	92	113	132	999
Napier	142	112	96	63	43	31	32	43	66	97	119	136	978
Palmerston North	123	103	86	55	35	24	25	37	53	77	97	116	831
Blenheim	158	130	108	73	47	35	36	48	73	103	125	146	1080
Hokitika	101	84	70	45	32	27	28	38	48	67	86	101	727
Lincoln	133	107	85	53	34	25	25	36	57	90	109	128	881
Alexandra	131	104	81	43	21	11	10	23	50	84	110	129	795
Invercargill	104	85	69	45	33	22	23	35	52	75	90	107	741

When pasture is irrigated, the plants become wet, and transpiration is suppressed. Immediately downwind, the air's saturation deficit and the evaporation rate will be reduced. As for rainfall interception, some of the applied water will evaporate without reaching the ground. However a compensating reduction in pasture transpiration reduces the significance of this interception. McNaughton (1981) developed a physically based model and predicted that the net interception loss during sprinkler irrigation usually was much less than 10% of the applied water. This is much lower than the percentage interception losses from forest during rainfall discussed below.

Evaporation from water-stressed pasture

As soil dries out, the remaining water is held more strongly, and moves to roots more slowly. If the roots cannot extract water from the soil at the E_0 rate, evaporation becomes limited by water rather than energy. To avoid desiccation, the pasture plants close the stomata on their leaves, so r_s increases, growth virtually stops, and the evaporation rate drops. Pasture plants are well adapted to dry conditions, and can usually extract enough water from the moister subsoil, sometimes from depths greater than 1.5 m, to keep some foliage alive and green. At the end of a dry summer, the soil profile under pasture is likely to be over 200 mm drier than it is at field capacity. The whole profile does not have to re-wet for the evaporation rate to return to the E_0 of well-watered pasture value. Even a few millimetres of rain will usually suffice, as that will re-wet the leaves and the surface soil where roots are usually most active.

For estimating drought days and irrigation requirements (e.g., New Zealand Meteorological Service 1986) or for modelling pasture growth (e.g., Moir *et al.* 2000), the simplistic approach of assuming a readily available (growth-producing) soil water-holding capacity of about 75 mm is often used. The evaporation rate is then assumed to be equal to E_0 until the 75 mm of water stored in the root zone at field capacity is used up, after which evaporation is assumed to be zero until rain or irrigation is added.

For estimating groundwater recharge or stream flow from a pasture catchment, this simple approach is inadequate. As already mentioned, even though growth may be negligible after about 75 mm of water has been extracted, pasture can extract another 100 mm or more of "survival water" from the root zone of most soils, but at a lower extraction rate. Before groundwater recharge can occur, there has to be enough rainfall in excess of evaporation to replace the whole soil water deficit, thus bringing the root zone back to field capacity. To estimate groundwater recharge (e.g., Thorpe and Scott 1999),

evaporation often is assumed to equal E_0 until about 75 mm of water has been removed. It then is assumed to become controlled by the soil, and to drop linearly with decreasing water storage in the root zone, approaching zero when a total of 150 to 250 mm of water has been extracted from the root zone. Figure 3.5 shows one such measured relationship between the water-limited evaporation rate and the soil water deficit. In this particular soil, during summer when the average evaporation rate is about 4 mm/day, pasture evaporation changes from being energy-limited to water-limited at a soil water deficit of about 120 mm. Variations in soil type and rooting depth affect the available water storage capacity. Also, the presence of a shallow water table sometimes can provide extra water via capillary rise into the root zone, greatly enhancing the amount of water available (e.g., Scotter 1989).

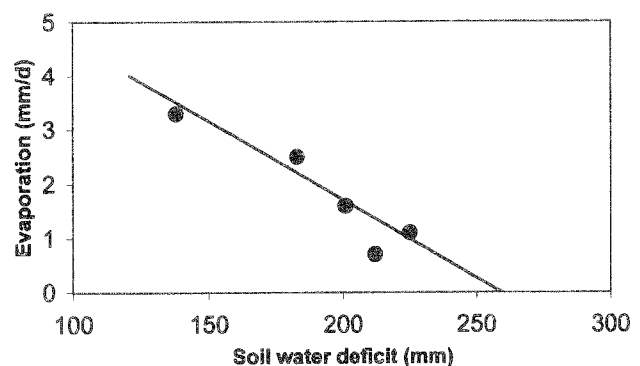


Figure 3.5 The relationship between pasture evaporation rate and the soil water deficit relative to field capacity for Tokomaru silt loam (after Scotter *et al.* 1979).

Approaches that assume a unique relationship between E and the soil water deficit do not deal well with re-wetting of the rootzone. Once rain wets the topsoil the evaporation rate jumps back to E_0 , regardless of how dry the subsoil is. Scotter *et al.* (1979) and Woodward *et al.* (2001) describe one way of getting around this problem.

FOREST EVAPORATION

There are a number of ways in which forest and pasture evaporation differ. Firstly, forest projects up into the turbulent air a lot more than pasture, so its aerodynamic resistance is a lot lower. A typical r_a value for forest 20 m high is 5 s/m, an order of magnitude smaller than for pasture. In Equation 3.3, the second term in the numerator is thus much larger than the first term, so that the evaporation rate depends more on the wind and vapour pressure deficit than it does on the net radiation.

When the leaves are dry, the effects of a low aerodynamic resistance are offset by a significant canopy resistance (r_s), and evaporation rates can be less than the rates for well-

watered pasture given in Table 3.1. However when intercepted rainfall wets the foliage, r_s is effectively zero, and high evaporation rates can occur. Such interception by forest has been found to be the fate of from 12% to 50% of rainfall (Kelliher and Jackson 2001).

A second difference is that, where soil depth allows it, trees are deeper rooting than pasture, and so have access to a greater store of water to tide them over dry spells. For example, *Pinus radiata* growing on a pumice soil has been found to extract 900 mm of water from the top 2.7 m of a pumice soil (Will and Stone 1967).

There is also a lot more feedback between the parameters in Equation 3.3 for forests than there is for pasture. Tall forest generates turbulence that couples the conditions at its evaporating surface to the saturation deficit in the air above. Further, a tree's "plumbing system" can only supply water up to a certain rate, so stomatal control has evolved to prevent desiccation. The stomata of forest trees are thus sensitive to the air saturation deficit. For long-lived trees, it is no doubt better to reduce carbon uptake and photosynthesis by stomatal closure at times of excess demand, rather than to risk death by desiccation. The leaf area, and this connection between canopy resistance and air saturation deficit, largely determine the forest evaporation rate when the leaves are dry.

While it is more difficult to use meteorological data to model evaporation from forest than from pasture, it is possible (e.g., Whitehead and Kelliher 1991). Interception and forest floor evaporation need specifically to be taken into account, as does the effect of humidity and sunlight on canopy resistance. Such models can be used to predict the effect of clearing or thinning forest on the water yield from a catchment.

COMPARING PASTURE AND FOREST EVAPORATION

How the soil-plant-atmosphere system determines pasture and forest evaporation in different environments is neatly illustrated in the scenarios described by Kelliher and Jackson (2001) (Table 3.2). They considered the

annual water balances of closed-canopy pine forest and pasture located near Christchurch, Rotorua and Reefton, which have low, medium and high rainfall.

At Lincoln near Christchurch, well-watered pasture evaporates about 880 mm/year (Table 3.1). However, in the year considered by Kelliher and Jackson only 600 mm of rain fell, and without irrigation that was the maximum amount of evaporation that could occur, leaving an unsatisfied deficit of 280 mm, and negligible runoff or drainage from the soil. Pine forest at the same location also evaporated 600 mm/year, 200 mm/year as interception. In both cases the vegetation increased its surface or canopy resistance to restrict evaporation and avoid desiccation. The deeper rooting system of the forest was of limited help due to a lack of available water in the subsoil.

In contrast, near Rotorua the annual rainfall was 1,400 mm, 500 mm higher than the 900 mm/year evaporation rate of well-watered pasture (Table 3.1). However, the relatively shallow pasture root zone had limited ability to store rain until it is needed, so there was still a 100 mm unsatisfied deficit. This brought the actual evaporation down to 800 mm, leaving a runoff surplus of 600 mm/year to appear as groundwater recharge or stream flow. With 350 mm of interception, the pine forest near Rotorua was estimated to evaporate 1000 mm/year. The greater soil water buffer accessible to the deep-rooting trees satisfied their needs during dry spells, and the pine forest did not experience a deficit. But a consequence of the extra 200 mm of evaporation by the forest relative to pasture was 200 mm less runoff during the year.

The annual rainfall near Reefton was 2,400 mm and quite uniformly spread throughout the year, so neither the pasture nor the forest experienced any water stress. Runoff from pasture of 1,650 mm was calculated when the evaporation of 750 mm/year was subtracted from the rainfall. The Reefton pine forest lost 600 mm/year as interception, giving a total evaporation rate of 1,050 mm/year. This left 1,350 mm/year as runoff, 300 mm less than for pasture.

Table 3.2 Components of the water balance for pasture and full-canopy pine forest at three locations. All numbers are mm/year (after Kelliher and Jackson 2001).

	Christchurch		Rotorua		Reefton	
	Pasture	Forest	Pasture	Forest	Pasture	Forest
Rainfall	600	600	1400	1400	2400	2400
Evaporation	600	600	800	1000	750	1050
Deficit	280	220	100	0	0	0
Runoff	0	0	600	400	1650	1350

These scenarios demonstrate that it is not just the evaporative demand that determines evaporation rate, even in a humid country like New Zealand. The rainfall, the soil, and the vegetation also play important roles.

EVAPORATION FROM OTHER SURFACES

Lakes

In 1992 Spiegel and Viner stated that there were no published studies of lake evaporation in New Zealand. With the notable exception of the short-term data collected by Oliphant (2000), some of which are graphed in Figure 3.1(b), this is still seems to be the case. This is surprising, given the large number of lakes in the country, and their importance for hydroelectric generation. However, the principles outlined above apply also to evaporation from lakes, as detailed studies of lakes at similar latitudes in the northern hemisphere have shown (Schertzer 1997).

Let us first consider how Equation 3.3 applies to a lake. The surface resistance (r_s) is zero, increasing the denominator. If the water surface is smooth, the aerodynamic resistance (r_a) would be higher than for pasture, reducing the size of the second term in the numerator, but as most of the net radiation is absorbed by the lake, the first term, ($R_n - G$), is even smaller. Thus, as for forest, but for quite different reasons, the second term again dominates, and the evaporation rate depends mainly on the wind speed (through its effect on water roughness and on r_a), and on the vapour pressure deficit ($e_s - e_a$).

For lakes with an average depth of less than 10 m, seasonal evaporation is highest in summer and lowest in winter, as for pasture (Schertzer 1997). However, for deeper lakes the damping and phase lag induced by the large thermal mass of the lake water becomes more important. For the Great Lakes of North America, evaporation is highest in winter and lowest in summer (Schertzer 1997), and on a day-to-day basis tends to be highly episodic. In one study the highest daily evaporation, 13 mm, occurred in autumn after the passage of a cold front.

We cannot use Equation 3.3 to make monthly estimates of lake evaporation, as we did for well-watered pasture, because long-term data are not available for the vapour pressure deficit in the air above any New Zealand lakes. The vapour pressure deficit cannot be assumed to be the same as that measured above the land nearby, due to the quite different energy balances over land and water. Until further work proves otherwise, we suggest that Equation 3.4 may provide rough estimates of the seasonal evaporation rate from a lake, with the value for α probably between the equilibrium value of 1.0 and the Priestley-Taylor value of 1.26.

The application of Equation 3.4 to provide rough estimates of lake evaporation can be illustrated by

considering lakes Tekapo and Taupo. Climate data collected on land near both lakes are given in New Zealand Meteorological Service (1983). Water temperature data for Lake Tekapo are given by Irwin and Pickrill (1982) and for Lake Taupo by Green *et al.* (1987). Both lakes are warm monomictic—they show summer stratification (due to a stable density gradient during warming) but circulate fully in winter (due to an unstable density gradient during cooling), and stay above 4°C (the temperature at which water is most dense). The heat flux density into or out of a lake can be found as

$$G = Z\rho_w c_w \Delta T / \Delta t \quad 3.6$$

where Z is the lake depth (m), c_w is the specific heat capacity of water (4,190 J/kg/°C), and ΔT is the change in lake temperature averaged over its depth (°C) during time period Δt (s).

Lake Tekapo is at latitude 44°S and at 700 m altitude. It is about 95 km² in area and 100 m deep. Consideration of the inflows and outflows gives a mean residence time for water in the lake of 2.4 years. The lake is warmest in February, reaching 15°C in the top 20 m, but only 8°C below 60 m. The lake is coolest in September when it is about 6°C at all depths (Irwin and Pickrill 1982). Monthly calculations with Equations 3.4 and 3.6 give an estimate of 780 mm for the equilibrium annual lake evaporation, and 1.26 times this value, or 990 mm, for the Priestley-Taylor rate. These values are similar to those for well-watered pasture in Table 3.1, but the seasonal distribution is quite different, with the highest evaporation rate in autumn, when the cooling lake provides energy for evaporation, and the lowest in summer, when most of the solar radiation goes into warming the lake, as shown in Figure 3.1(b). The estimated seasonal equilibrium evaporation rates are 90, 50, 340 and 300 mm for spring, summer, autumn and winter respectively. The winter evaporation rate is five times the summer rate, which is exactly the opposite to the evaporation from well-watered pasture (Table 3.1).

Lake Taupo is at latitude 39°S and at 360 m altitude. It is 620 km² in area and also 100 m deep. The lake is warmest in March, reaching 21°C in the top 20 m, but is only 11°C at 70 m. The lake is coolest in August when it is about 10°C at all depths (Green *et al.* 1987). The estimated equilibrium annual evaporation rate is 950 mm, and the Priestley and Taylor value is 1,190 mm, with seasonal equilibrium evaporation rates of 150, 150, 390 and 260 mm for spring, summer, autumn and winter respectively. Due to the warmer winter air temperature at Lake Taupo compared to Lake Tekapo, the difference between summer and winter evaporation rates is smaller. However it seems that there is still about twice as much evaporation from Lake Taupo in winter as there is in summer.

Tussock, bare soil and peat bog

Tussock may look like tall grass, but it sometimes behaves more like forest than pasture. Due to its upright growth and often-windy habitat, it can have a low aerodynamic resistance, sometimes leading to high interception losses, which were measured as 21% of rainfall in one study (Campbell and Murray 1990). However when tussock grassland was converted to pine forest, Fahey and Watson (1991) found a 20% reduction in water yield, which they attributed to smaller interception losses from tussock than forest. Also, Hunt *et al.* (2002) found that degraded tussock grassland invaded by *Hieracium* evaporated at the equilibrium rate only when well watered, and evaporation dropped to a fraction of that rate as the shallow soil dried out.

The evaporation rate from bare soil when the surface is moist occurs at about the rates given for well-watered pasture in Table 3.1. However, after moist soil has lost between 2 mm and 12 mm of water (Allen *et al.* 1998), the soil surface becomes air-dry (indicated by a lighter colour) and the evaporation rate starts to fall. This decline is because the rate of water movement through the soil to the surface, rather than the energy supply, now limits evaporation. About another 6 mm to 30 mm can be lost from bare soil during this phase of declining rate of evaporation.

Peat bogs provide an example of vegetation protecting their environment by maintaining the moist conditions favourable for peat formation. Thompson *et al.* (1999) found that the evaporation rate from a raised peat bog dominated by rushes averaged only 55% of the equilibrium rate, but equalled or exceeded it when the foliage was wet. They commented that, despite the moist peat surface, "a combination of plant physiological and canopy structural factors ... combined to prevent high rates of evaporation".

The clothesline and oasis effects

There are two conditions in which the evaporation rate can be much higher than might otherwise be expected. The first is due to the "clothesline effect". This comes into play where a particular shrub or tree, or a line of trees such as a shelterbelt, is much higher than the surrounding vegetation, reducing the aerodynamic resistance. By extending up into the turbulent air, such trees behave like washing hung out to dry, particularly when the leaves are wet and conditions are windy. Because advection provides a large amount of energy from the surrounding environment, the evaporation rate can be unusually high in such situations.

The "oasis effect" comes into play where there is an irrigated area in the middle of parched, non-irrigated land. It also occurs where plants are surrounded by surfaces with minimal evaporation, such as bitumen, concrete, or iron roofs. Most of the solar radiation absorbed by the

surrounding dry surfaces goes into heating the air, which increases the vapour pressure deficit ($e_s - e_a$). When this drier air passes over plants that are not water stressed, it increases their evaporation rate, as can be seen from Equations 3.2 and 3.3.

It is the varying degree to which a Class A raised evaporation pan is subject to the clothesline and oasis effects that causes the variability in pan coefficients.

CONCLUSIONS

Evaporation is the consequence of a complex set of interactions in the soil-plant-atmosphere system. It can be highly variable, but in the humid environment characteristic of most of New Zealand for much of the time, the evaporation rate is anchored to the equilibrium evaporation rate. At such times Equation 3.4 approximately describes evaporation, with α often falling between 1.0 and about 1.26, due to advection providing extra energy from the atmosphere and surrounding ocean, in addition to that provided by insolation.

Climate change has occurred historically, and it is generally accepted that global warming is occurring, with minimum air temperatures increasing by about 0.2°C per decade and maximum temperatures by 0.1°C per decade (Roderick and Farquhar 2002). Decreases in pan evaporation have been observed in the northern hemisphere over the last fifty years (Peterson *et al.* 1995), and there has been some controversy about the cause. Roderick and Farquhar (2002) argue that the incoming solar radiation has been decreasing by about 3% per decade, causing a drop of the same magnitude in the evaporation from moist surfaces. They suggest that the drop in solar radiation is due to increased cloud cover and/or aerosol concentration in the atmosphere. It is not known if similar drops have occurred in New Zealand, and if so, whether similar decreases will occur in future.

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Chapter 2

Atmospheric circulation and precipitation

Jim Salinger, Warren Gray, Brett Mullan and David Wratt

INTRODUCTION

The source of New Zealand's freshwaters is precipitation—especially rain and snow. This chapter outlines factors that determine how this precipitation varies in time and space. Present spatial patterns of precipitation are documented, variations and trends observed over the past century are described, and the likely future influence of increasing global concentrations of greenhouse gases is examined.

New Zealand's location at 34 to 47°S in the Southwest Pacific means that it lies largely within the prevailing westerlies of the mid-latitude Southern Hemisphere. The travelling anticyclones, depressions, and fronts within this flow predominantly govern the progression of weather. However, weather systems that originate from within the tropics can also have an influence (Sturman and Tapper 1996).

These circulation patterns are ultimately driven by the extra solar radiation arriving at the equator compared to the poles, the differential surface heating caused by this radiation difference, the atmospheric and oceanic motions which arise to redistribute this heat, and the influence of the earth's rotation on these motions (Barry and Chorley 1998). Another important aspect is the overturning of the atmosphere - called convection- which occurs when warm, unstable air from near the earth's surface rises forming cloud and, in more intense cases, rainfall, hail, thunder and lightning. The weather is also influenced by topography. Uplift and resultant cooling of air as it impinges on mountain ranges leads to condensation, clouds and precipitation on the upwind slopes. This depletes the air of moisture, and together with downward motion and resultant warming on the lee side of the mountains leads to dry conditions downwind. New

Zealand's precipitation patterns are influenced by all of these factors, and their interactions. For example convection can occur preferentially along the "fronts" between warm and cold air masses, or it can be triggered as potentially unstable air is pushed upward over mountains.

New Zealand precipitation varies with fluctuations in the prevailing westerlies and in the strength of the subtropical high-pressure belt. The country can be divided up into eight regions that show internally coherent variability in monthly precipitation patterns. Temporal variations in precipitation in the individual regions can be related, at least in part, to seasonal variations in the strength of the westerlies, to interannual variations associated with the El Niño-Southern Oscillation (ENSO) and to decadal variations associated with the Interdecadal Pacific Oscillation (IPO)¹. Since the late 1970s, there has been more frequent airflow from the west and southwest onto New Zealand, and these circulation changes have produced significant trends in precipitation. The north of the North Island has become drier, whereas the north, west, and south of the South Island have all become wetter, compared to earlier decades. Much of this change is due to natural variability in circulation. It is likely that the phase of the IPO changed in 1998 so there may be some reversal in this behaviour over the next two to three decades. However, climate model simulations driven by scenarios of increasing greenhouse gas concentrations also predict a strengthening of the west-east precipitation gradient. Hence this pattern of wetter in the southwest and drier in the northeast of the country, overlaid with year-to-year and decade-to-decade natural variability, forms a likely long-term prognosis over the duration of the 21st Century.

¹ The ENSO and the IPO are described later in this chapter.

WEATHER SYSTEMS

The Weather that affects New Zealand

Westerlies

The mid-latitude weather systems that dominate the weather, particularly in the south of the country, are influenced strongly by the jetstream. This high altitude flow of strong westerly winds results from the tropical air ascending and moving southward, away from the equator. This tropical air rotates around the north-south axis through the centre of the earth. Hence, its speed is greater than that further south, as it lies further out from the axis. As the tropical air moves south it gains a component of motion towards the east, contributing to the formation of the jetstream at about 30°S. The westerlies that flow around the roaring forties are a response of the atmosphere as it balances out the effects of the mid-latitude highs and lows that form south of the jetstream (Fig. 2.1). (See Brenstrum 2002, for a fuller but non-technical description of weather systems.)

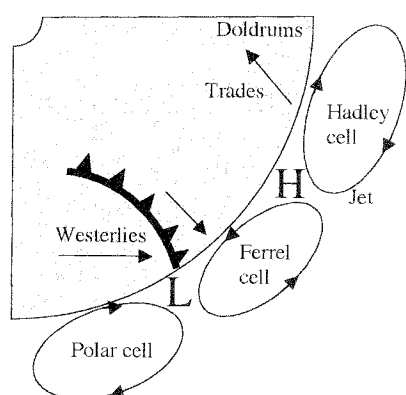


Figure 2.1 A schematic of the average circulation of the winds around the earth.

Many of our weather systems form on the boundary between this warm, moist tropical air and the colder air from the regions around the poles. The colder air will tend to move under the warmer air, the warm air sliding up and over the cold air. These motions induce eastward and westward components that result in rotation and the formation of depressions. Secondary circulations can further concentrate the air mass contrasts into narrow regions—these are marked on weather maps as fronts. These weather systems are known as mid-latitude depressions or lows, and form much of the interest on weather maps.

North Tasman Lows

Depressions forming in the North Tasman sea are an example. The development of these systems is bolstered by the energy available from the warm sea temperatures

near the Queensland coast, and the lifting caused by jetstreams aloft. As warm, moist air is lifted it forms cloud. This releases more heat from the condensation process—known as latent heat. This heating enhances the lifting, leading to an enhancement of the low's circulations. The “weather bomb” that crossed New Zealand on the 20 June 2002 was an example of just such a system (Fig. 2.2). It was called a “bomb” as its central pressure decreased by over 24 hPa in 24 hours. This system led to extensive flooding and wind damage over Northland and the Coromandel.

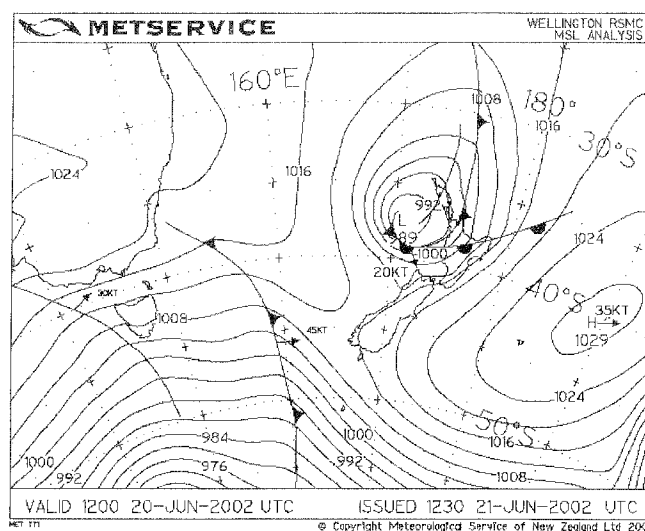


Figure 2.2 The mean sea level pressure analysis of the situation at midnight on 20 June 2002.

Courtesy of Metservice (NZ) Ltd

Ex-Tropical systems

Tropical cyclones form by a different process. These systems draw heat and moisture from the sea. As this air is drawn into the storm, these energy sources intensify the storm, drawing in more air, and hence more heat and moisture—which further enhances the storm. This process is only effective when there is sufficient heat and moisture available, and that is generally only when the sea surface is above 26°C. Tropical cyclones are often nearly circular when viewed in satellite imagery, showing the rapidly rotating nature, with winds up to 250 km/h. The centre of the tropical cyclone is the eye, a small area of calm winds. Tropical cyclones are often very small in scale, with the radius of strong winds and intense rain only extending out 100 km from the centre (c.f. mid-latitude lows of 1000 km across). When these systems move south, they encounter colder seas and strong winds aloft, which destroys the structure of the storm. However, the nature of these storms can change with an injection of cold air into the system, transforming them from a warm-cored tropical cyclone to a cold-cored, broader-scale mid-latitude

depression. While tropical cyclones are small scale systems, the ex-tropical storms that pass by New Zealand can be much larger in scale. Storms like that resulting from cyclones Bola and Giselle are examples of tropical cyclones that transformed into intense mid-latitude systems, causing damage from strong winds and intense rainfall. While most of the weather systems affecting New Zealand can occur at any time of the year, the warm sea temperatures needed for tropical cyclones only occur between December and April (Fig. 2.3).

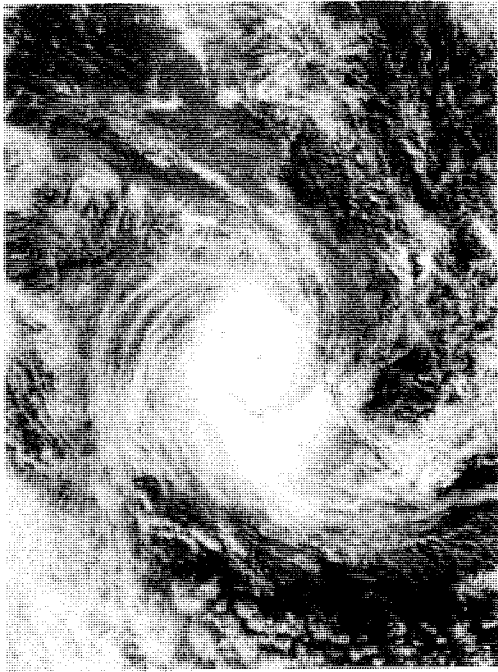


Figure 2.3 A satellite image of the Tropical Storm Erica. This storm passed over New Caledonia on 14 March 2003, injuring 15 and killing 1 person. NASA/GFSC(2003)

PRECIPITATION PROCESSES

There are several steps in the process by which upward motion leads to condensation and rain. The maximum amount of moisture air can hold increases with increasing temperature. As a parcel cools, as it would when it is lifted and expands, so the water holding capacity decreases. When air is lifted and cooled to the point where it can no longer hold all its moisture then condensation can start to form. Generally water will condense out on the small particulates in the air, known as cloud condensation nuclei. Drops continue to grow as the air is lifted and more moisture becomes available. The larger the droplets, the more condensation that forms on them. Thus the biggest drops grow the quickest. As the drops get large, they start to fall through the cloud of smaller drops. This leads to collisions, and the big and small drops can coalesce. It takes around 1 million cloud droplets to form 1 raindrop.

This process of forming rain by condensation and coalescence is quite slow, and can be quickened by the injection of ice. Condensation takes place faster on to an ice surface than on to liquid water. When an ice crystal is injected into a cloud of liquid water drops, it will quickly take the moisture from the liquid drops. This enables rain to develop faster when ice is involved. For example, in a cumulus cloud it probably takes about 20 minutes for droplets to grow to raindrop size by coalescence in clean maritime air (Banta 1990) but half this time or less when ice crystals are present (Barry 1992). A strong westerly flow air may travel several kilometres in the time raindrops or snowflakes take to form and fall to the ground, a factor which is important in modelling the precipitation distribution across the Southern Alps. Most of the rainfall in New Zealand forms from a process involving ice.

Upward Motion

The nature of the upward motion often characterises the precipitation that results. For example, the gentle widespread lifting that results from fronts leads to broad scale precipitation bands; the small-scale intense updrafts of thunderstorms lead to very heavy precipitation over a small area. Squall lines are lines of organised convection that generally have scales of tens of kilometres along the line, but less than 10 km across.

Orographic precipitation

One source of upward motion results from the lifting of the air as wind flows over hills. This leads to condensation, often seen as a cap-cloud on the hill top. However, there are a number of ways of turning this condensation into precipitation. Rain can form by the growth of cloud droplets within this cloud, but often this would take too long, and the parcel of lifted air would have traversed the hills in that time, particularly for the smaller hills like those in Coromandel. Another way of capturing the moisture in the cap cloud is for precipitation from another system, e.g., a frontal precipitation band, to fall through the cap cloud, washing out the cloud drops by collision. This process is called seeder/feeder enhancement, as the seeding rain falls through the feeder cloud (Gray and Austin 1993). A third way of hills enhancing precipitation is through the triggering of extra showers and convection over the hills. The instability of an airmass may be released by the triggering resulting from the uplift. This would lead to extra showers over the hills.

These processes lead to the upwind side of hills being a preferred location for precipitation, and the loss of moisture leads to the lee side being drier. This is the dominant factor in the pattern of precipitation seen in Figure 2.4, the map of the average annual precipitation over New Zealand. The predominant westerlies drive the

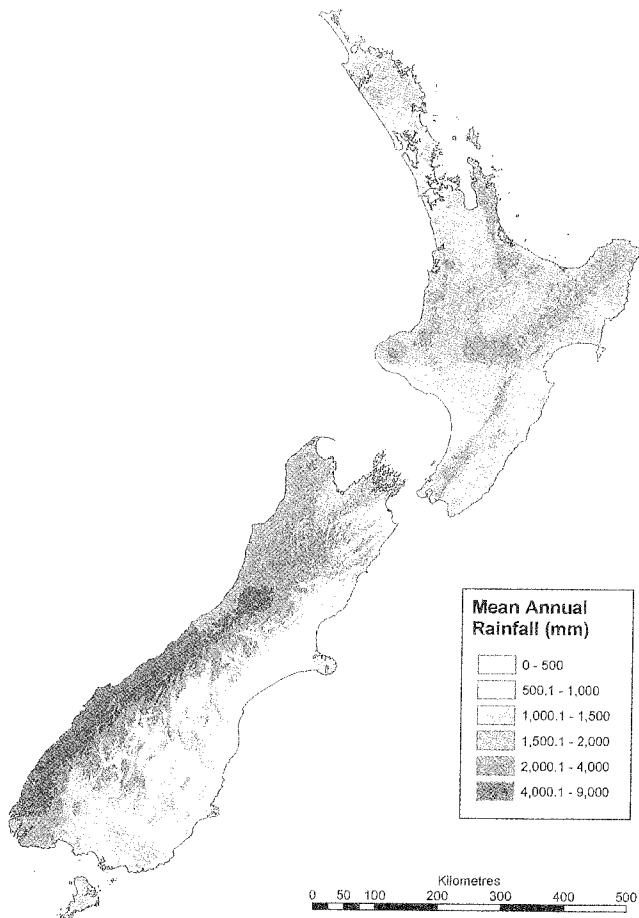


Figure 2.4 Annual mean precipitation, 1961-1990 normals (mm), courtesy of NIWA.

wind up and over the Southern Alps. This leads to annual precipitation totals in excess of 12 m on the region just upwind of the crest. In contrast, areas over Canterbury and Otago see as little as 700 mm of rain each year. The regular passage of fronts in the westerlies provides the source of moisture, uplift and seeding that forms the precipitation that fills the hydro-lakes. As these lakes lie on the eastern side of the divide, it is generally only in stronger winds that the rain, formed over the alpine crest, blows onto the lee side to fill the lake catchments. This spillover is fortunately a frequent occurrence, making the lakes a reliable source of hydropower.² More details on factors influencing precipitation in the Southern Alps and spillover into the South Island hydro catchments are detailed in the SALPEX research project (see Wratt *et al.* 1996, 2000).

Fronts

Fronts are regions where much of the hazardous weather occurs. This is where the heaviest precipitation and strongest winds occur. The character of precipitation associated with fronts depends on the nature of the front. Warm fronts, formed as warm air moves over the top of cold air, are generally characterised as having precipitation that starts as light drizzle and becomes more intense—a result of the cloud become thicker and lower. Cold fronts can have several sequences of precipitation with them. Often they start with lighter rainfall, but can finish with a heavy band of rain. This can take the form of embedded convection, where broad scale rain bands co-exist with more intense and smaller scale convection.

A southerly change is a cold front with an altogether different structure. The southerly change is often heralded with a burst of heavy precipitation that is followed by broader scale rain. These southerly “busters” are one of the more dramatic precipitation systems that affect the east coasts.

Convection and Thunderstorms

Much has already been said of convection. This intense overturning of the atmosphere is responsible for much of our hazardous precipitation, particularly in the urban context where the hard impervious surfaces lead to rapid runoff with little of the retention that might be expected from a rural catchment.

Thunderstorms are the grandest example of convection. These systems take air from the surface and lift it through the depth of the troposphere, leading to columns of rising air 10-15 km high. These strong updrafts are able to hold aloft large water drops and hail. Updrafts as fast as 10 m/s would enable hail to be suspended in the air while they accrete moisture, and then eventually fall, often by moving out of the updraft region. The intensity of these system can lead to rain rates in excess of 100 mm/h. Convection also describes the smaller showers that form, for example, with a cold outbreak. The cold air moving up from the south flows over a warmer surface. Heated from below, the air becomes unstable, and overturns and this is manifested as an area of showers (Fig. 2.5).

Surface heating, as occurs in the middle of summer, can lead to the atmosphere becoming unstable. The formation of the afternoon showers often seen in summer occurs by this process. Triggering of these showers is often achieved by the lifting of the air in a sea breeze. The long coastlines

² This is not always the case. There have been several actual (or potential) New Zealand energy supply crises since the 1950s, associated with unusually long periods of weakened westerlies and more anticyclones, with reduced West Coast precipitation.

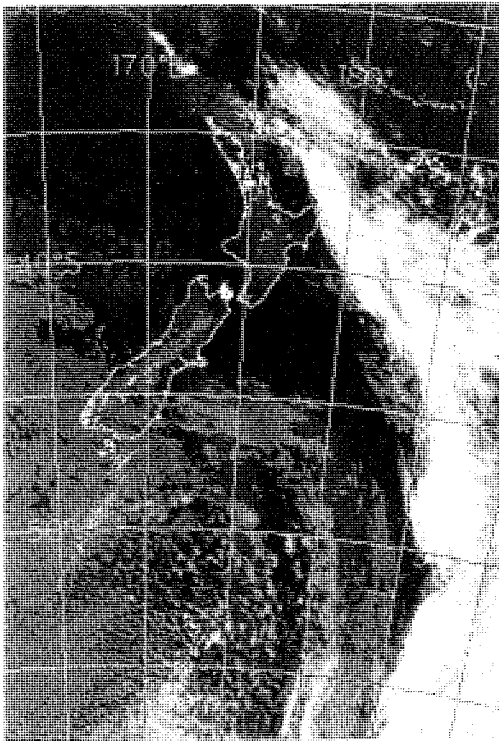


Figure 2.5 A satellite image showing frontal cloud to the East of the North Island and a cold outbreak over the ocean to the south of the South Island. Copyright NIWA

encourage the formation of sea breezes, and the upward moving air over the land can trigger the formation of convective showers and thunderstorms.

SPATIAL PATTERNS OF PRECIPITATION

On the monthly and longer timescale, precipitation over New Zealand often displays strong spatial patterns because of interactions between the circulation and southwest/northeast alpine ranges of the country. This section describes how New Zealand can be divided into eight regions, such that within each region there is a coherent precipitation response to the interaction between circulation and orography. The eight regions are associated with specific precipitation-producing circulation types. Precipitation trends in these regions can also be linked to changes in regional atmospheric circulation.

Site-specific precipitation trends for the period 1920-1990 have been documented by Salinger *et al.* (1992a and 1992b), using records from 21 stations, with rigorously quality-controlled measurements. Although these studies have identified overall New Zealand trends, and station trends, it is preferable to describe trends for regions that show spatially coherent climatic variation. This is because New Zealand is a long narrow country spanning from 34 to 47°S, with high axial ranges rising to 3764 m. The mountains in the South Island extend above 1000 m over

a distance of 750 km. They present a significant barrier to the mean airflow, which is from the west-southwest (Tomlinson 1976). The interaction between atmospheric circulation patterns and the orography is an important mechanism for determining precipitation distribution (Watts 1947, and previous section). It is thus important to define coherent regions of New Zealand that have a similar response to the interaction between circulation and orography, so as to derive district variations in climate.

A first attempt to numerically define such regions was undertaken by Salinger (1980). This study demonstrated that the resulting regions are strongly localized because of the interaction between circulation and New Zealand's rugged orography. Eight coherent regions of New Zealand were defined by correlation of precipitation at 250 stations with time series of indices of atmospheric circulation (Trenberth 1976), and by rotated principal component analysis for 100 stations for the period 1951-75. Salinger and Mullan (1999) used longer time series of precipitation for 88 stations to confirm whether the eight precipitation regions were robust. They validated the division of New Zealand on the basis of climate variation, as well as documenting the regional precipitation variability and trends over the period 1930-94, and their results are discussed in the following sections.

Spatial characterisation

Complete, or near complete, precipitation records from the NIWA Climate Database for the 1930-1994 period were used by Salinger and Mullan (1999). Observed monthly precipitation totals over this 780-month period were extracted from the Database, and carefully checked for errors and site changes. A reasonably complete national coverage was achieved using 88 precipitation stations over the 65-year period.

Principal component analysis is a statistical technique that has been applied very successfully to analysing the variability of multiple climate time series (see, e.g., Craddock and Flood 1969; Pittock 1980). The principal components can be rotated around the origin until each is maximally collinear with a distinct cluster of vectors. This rotation derives components that delineate separate groups of highly intercorrelated stations for analysis of climate variation.

A rotated principal component analysis (Richman 1986) was carried out on all data to identify characteristic patterns of interannual precipitation variation. A set of *unrotated* principal components (empirical orthogonal functions) was calculated first. A varimax rotation, that retains orthogonality between components, was then performed. The number of components rotated was decided by the usual criteria (Kaiser 1960; Craddock and Flood 1969), where the unused higher unrotated com-

ponents accounted for a very small proportion of the relative variance.

To interpret the rotated principal component (RPC) patterns, each pattern has been related to circulation variations by correlation analysis. Correlations were calculated between the RPC scores on each month and various circulation indices. Trenberth (1976) defined a number of circulation indices (Table 2.1) for the New Zealand region (Z1, Z2, Z4, M1 and M2) that used pressure differences between station pairs to identify airflow anomalies. Z1 measures the anomalous mean-sea-level pressure difference between Auckland and Christchurch. It indicates the strength of westerly circulation anomaly over central and northern New Zealand. A zero anomaly implies normal flow conditions. Z2 measures the anomaly in mean-sea-level pressure difference between Christchurch and Campbell Island. It indicates the strength of westerly circulation anomaly over southern New Zealand and to the south. Z4 measures the anomaly in mean-sea-level pressure difference between Raoul Island and Chatham Island. M1 measures the anomaly in mean-sea-level pressure difference between Hobart and the Chatham Islands. It indicates the strength of southerly circulation in the New Zealand area. Finally M2 is the anomaly in mean-sea-level pressure difference between Hokitika and the Chatham Islands, and is indicative of the north/south flow just to the east of New Zealand. Additionally, precipitation anomalies for high score months have been examined and case studies were taken of months which demonstrated a precipitation anomaly pattern which bore a strong similarity to a RPC. Salinger (1980) identified the monthly circulation anomalies and key synoptic events that cause the high precipitation anomaly pattern. Months with low precipitation anomalies and component scores have also been examined to identify atmospheric circulation that results in low precipitation.

Salinger and Mullan (1999) also used some newly defined circulation indices (Table 1). M3 (= M1 - M2) measures the anomaly in mean-sea-level pressure difference between Hobart and Hokitika, and is indicative of the north/south flow to the west of New Zealand and MZ2 is the anomaly in mean-sea-level pressure difference between Gisborne and Invercargill. It thus measures anomalous northwesterly and southeasterly airflow across New Zealand. Lastly, MZ3 measures the anomaly in mean-sea-level pressure difference between New Plymouth and the Chatham Islands. It indicates the strength of southwesterly and northeasterly circulation over New Zealand. The Southern Oscillation Index, SOI, is a measure of the anomalous atmospheric pressure gradient across the Pacific-Indian Ocean region, and is the normalized pressure difference between Tahiti and Darwin.

Precipitation Patterns

The precipitation patterns are shown in Figure 2.6, reproduced from Salinger and Mullan (1999). The number of rotated components retained to describe precipitation patterns was eight, according to the various criteria discussed earlier and by Salinger (1980). The eight precipitation RPCs explained 73% of the monthly inter-annual variance over the period 1930-1994. Table 2.2 records the correlations between the circulation indices of Table 2.1 and the time series of the RPC patterns over this 780-month period. Note that the correlations of the SOI with the eight RPC time series are all close to zero. The SOI is not a good predictor of airflow across New Zealand for an individual month, and should be used for data aggregated into seasons or longer. Conversely, the seasonal SOI does display some useful *lag* relationships, unlike the Z and M circulation indices which have no future predictive value and should only be used diagnostically.

Table 2.1 Definition of circulation indices, and corresponding airflow anomalies over New Zealand. The indices are calculated from monthly mean sea level pressure differences (using deviations from long-term monthly means), over the base period 1951–1980. Indices that do not start at or prior to 1930 are: Z2 (1941), Z4 (1940), MZ2 (1948) and MZ3 (1944).

Index	Definition	Wind Anomaly (+/- index)
Z1	Auckland – Christchurch	west/east (central New Zealand)
Z2	Christchurch – Campbell Is.	west/east (south of New Zealand)
Z4	Raoul Is – Chatham Is.	west/east (east of North Island)
M1	Hobart – Chatham Is.	south/north
M2	Hokitika – Chatham Is.	south/north (east of New Zealand)
M3	Hobart – Hokitika	south/north (west of New Zealand)
MZ2	Gisborne – Invercargill	westnorthwest/eastsoutheast
MZ3	New Plymouth – Chatham Is.	southwest/northeast
SOI	Tahiti – Darwin	northeast/southwest

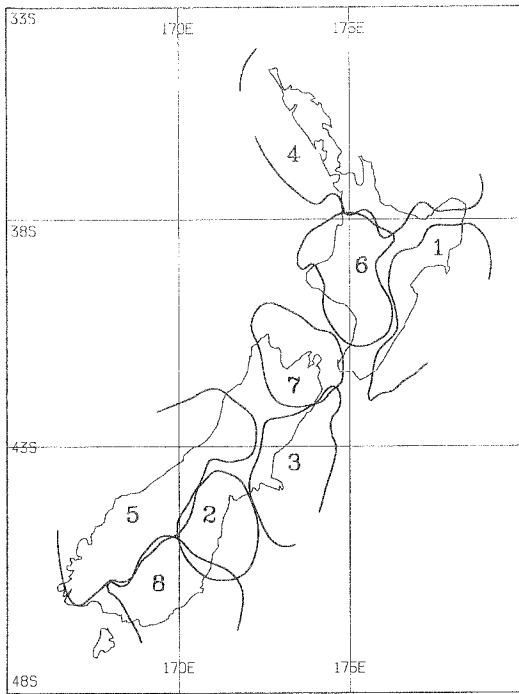


Figure 2.6 Core regions of precipitation rotated principal component patterns (RPCs) 1 to 8 of monthly pre-cipitation for the period 1930-1994, using 88 stations, representing spatially coherent patterns of variation over New Zealand. The variance explained by each pattern is given in the text.

East of the North Island

Precipitation in the east of the North Island is represented by RPC1 (Fig. 2.6) and accounts for about 12% of the variance. Correlations were highest with the zonal flow indices across New Zealand indicating much above average precipitation in east to southeast flow situations, and below average precipitation in west to southwest flow. Months with high precipitation anomalies in the east of the North Island have cyclonic disturbances

that traverse the north of the country and associated anti-cyclones that move eastward south of the country, resulting in a flow of moist, east to southeast air over the area. A good example was afforded in February 1970. While the remainder of the country remained dry, precipitation anomalies were double average in Gisborne and Hawke's Bay. East and southeast winds were enhanced during this period. The high totals were due to a depression that lingered to the northeast of the country for several days. Months with low precipitation were much less consistent, but there was a tendency towards strong west to southwest flow across New Zealand with anticyclones over northern New Zealand and frequent cyclonic disturbances passing to the south of New Zealand.

South Canterbury

This pattern (RPC2), representing precipitation in South Canterbury, accounted for 11% of the variance. Correlations were moderate only with Z2, indicating above average precipitation anomalies with easterly flow onto southern New Zealand and to the south, and below average precipitation when westerly flow is stronger to the south of New Zealand. In the former case (high precipitation), this flow is commonly associated with anticyclones to the south of New Zealand (Salinger 1980), exposing the eastern South Island areas to moist winds from the east. Dry months revealed no dominance of any consistent synoptic pattern.

North Canterbury/Kaikoura

A local maximum from central Canterbury to the northeast of the South Island and southern extremity of the North Island is shown in precipitation RPC3, which explains 10% of the variance. Significant negative correlations occurred with Z2 and MZ2, and positive correlations with M3, denoting strong southeasterly flow anomalies over southern New Zealand are the cause of above average precipitation, and northwesterly flow

Table 2.2 Correlation (*100) of RPC scores with circulation indices. Neglecting autocorrelation effects, which are small for these data, cross-correlations of 0.09 for precipitation are significant at the 99% level. The precipitation RPCs are over the period 1930-94.

RPC No.	Z1	Z2	Z4	M1	M2	M3	MZ2	MZ3	SOI
1	-54	-40	-23	-1	-27	18	-54	-40	7
2	-12	-46	2	3	-30	26	-17	-25	0
3	-24	-47	10	25	-10	40	-44	-14	0
4	-26	-37	-16	-18	-49	11	-9	-50	14
5	65	20	39	-6	-9	-2	70	20	-1
6	23	-26	39	14	-16	30	15	-1	6
7	2	-40	3	-13	-45	14	4	-33	3
8	33	-15	40	32	10	35	15	21	-8

anomalies the cause of below average precipitation. High precipitation rates were recorded in the key area in July 1963. A depression moved from the north Tasman Sea and remained quasi-stationary west of the North Island for three days whilst two anticyclones moved eastwards in latitudes south of New Zealand. These caused moist air from the east to flow onto the east of the South Island, resulting in monthly precipitation totals in this region of three or four times normal. Again, no consistent pattern emerged for the dry months.

Northland/Auckland/Bay of Plenty

Precipitation in northern New Zealand is represented by RPC4. The highest correlations were with MZ3 and M2 (negative) denoting above average precipitation anomalies with north to northeast flow. The converse, south to southwest flow produces below average precipitation. This pattern explained 10% of the variance. Synoptic patterns that promote high precipitation here are where depressions become slow moving just to the north or northwest of the North Island, giving warm, moist advection from the northeast. These months have low pressure anomalies northwest of the North Island giving stronger than normal winds with northerly or northeasterly components onto northern areas. Such features occurred in October 1955, with double the normal precipitation over Northland, and negative anomalies in areas other than northern New Zealand. In this month during the period of high precipitation a quasi-stationary anticyclone remained centred to the east of the North Island with two depressions occurring to the north and northwest of Northland. No consistent pattern was detected for dry months.

West Coast of the South Island

Precipitation on the west coast and alpine region of the South Island is represented by RPC5, which accounted for 9% of the variance (Fig. 2.6). Very significant positive correlations occurred with Z1, indicating that much above average precipitation months are a result of enhanced west to northwest airflow. The opposite easterly flow situations cause below average precipitation months. All months with high precipitation anomalies are characterized by the frequent passage of wave depressions moving east-southeast and passing near the south of the South Island. These months have a very strong Z1 pressure anomaly: high pressure to the north and low pressure to the south of the country. In these months the westerly circulation is very vigorous. May 1958 provides a very clear case study. Eleven depressions passed just to the south of New Zealand while six anticyclones moved across the north Tasman Sea to the north of the country, causing high zonal circulation pressure anomalies. Months of low precipitation on the

South Island west coast are dominated by above average sea-level pressure near southern New Zealand or to the south of the country and below average air pressure to the north, resulting in a prevalence of airflow from the east or southeast.

Western North Island

RPC6 has its highest station loadings in the west of the North Island, from Waikato to the Manuwatu. Strongest correlations were with Z4, which represents zonal flow to the east of the North Island. Thus high precipitation months occur with westerly flow across the North Island, and low precipitation months with easterly flow. Surface pressure anomaly maps confirm this flow with higher than usual pressure north of New Zealand and lower pressures over the South Island. Dry months were also consistent where the flow of air from the southeast is much enhanced over the North Island. Case studies identified positive pressure anomalies across the South Island and negative anomalies to the north and northeast of the North Island. The data suggest that both dry and wet months are equally important in this anomaly pattern that accounted for 8% of the variance.

Nelson/Marlborough

Precipitation in the Nelson and Marlborough is represented by RPC7. This pattern explained about 8% of the variance. Significant correlations occurred with circulation index M2 (negative), indicating that above average precipitation months are associated with more north to northwest flow, and below average precipitation with southerly flow months. High precipitation anomalies occur when depressions from the Tasman Sea move southeast cross the South Island. These disturbances are preceded by stronger than usual north to northwest winds onto the north of the South Island. Dry months gave quite disparate synoptic anomalies.

Southland/Coastal Otago

The final pattern identified, RPC8, has its highest station loadings in the south of the South Island, and accounts for 6% of the variance. Correlations were not strong with any circulation index, but the positive correlations with Z1, Z4, M1 and M3 indicate higher precipitation months are likely in west or southwest flow. This precipitation pattern relates to two synoptic situations. The first occurs when quasi-stationary fronts lie over the area in a strong westerly circulation, and the second occurs when depressions east of the south Island track north eastwards along the coast promoting strong south to southwesterly winds over southern New Zealand. Dry months also show some consistency in this region. During December 1955 under half of normal precipitation was recorded in the area. The associated pressure anomaly

pattern is one of high positive departures to the southeast of the South Island, with a prevalence of anticyclonic airflow with trajectories from the northeast.

The eight regions described in this section are fairly robust, although slight variations are possible. In particular, Mullan (1998) used data from 74 precipitation stations over the period 1949-1991, and arrived at six regions. These were very similar to RPCs 1-8 of Salinger and Mullan (1999), except that the lowest ranking pattern (RPC8) was dropped, and RPCs 2 and 3 (Fig. 2.6) were combined into one (which became the leading RPC with the greatest explained variance). This conveniently produced three localised patterns for the north, west and east of each Island of New Zealand, and became the basis for the six forecast regions used in the seasonal outlooks issued by NIWA (<http://www.niwa.co.nz/ncc/>).

TEMPORAL VARIABILITY IN PRECIPITATION

New Zealand climate varies naturally with fluctuations in the prevailing westerlies and in the strength of the subtropical high-pressure belt. The westerlies contain a succession of migratory depressions and anticyclones (Watts 1947; Garnier 1958; Maunder 1971). Generally, the depressions lie south of the anticyclones. The westerly belt migrates north in winter and south in summer, so that over New Zealand there is a double maximum (in the transition seasons) in the strength of the westerlies. However, except for the far south of the country, the spring maximum in westerly circulation is markedly the strongest.

There is a subtropical ridge in the longitudes of New Zealand that results from the continual eastward migration of anticyclones, and also coincides with the latitude of strongest sinking motion in the Hadley Cell (Figure 2.1). These anticyclonic cells reach their southernmost position in February and most northerly position in October. In winter, when the subtropical ridge is near 35°S, there is also a second maximum in anticyclone frequency at 44°S. This latter frequency peak represents “blocking” anticyclones that are often located east of the South Island, at the same time that wave depression activity reaches its highest frequency over the North Island.

This climatological pattern of atmospheric circulation can be quite different from one year to the next. Many of the circulation fluctuations that affect New Zealand are short-lived or random. However, other changes are associated with large-scale patterns over the Southern Hemisphere or Pacific Ocean, which show persistence and therefore some predictability.

There are two key natural cycles that operate over timescales of years (El Niño-Southern Oscillation, ENSO) and decades (Interdecadal Pacific Oscillation, IPO). Both these natural phenomena are confined largely to the Pacific

Ocean, but there is evidence that sea temperature conditions elsewhere, such as in the Indian Ocean, can also affect New Zealand climate at some times of year (Mullan 1998). On the intra-seasonal timescale, the Antarctic Oscillation, also known as the High Latitude Mode as first described by Kidson (1988), causes hemispheric fluctuations in the strength of the westerlies (eg, Z1, Z2), and influences precipitation over southern New Zealand. Research to date has not found useful predictability of the Antarctic Oscillation (Kidson and Watterson 1999).

This section describes New Zealand precipitation trends and variations during the 20th century that occurred as a result of regional circulation changes, focussing on the contrasts across the eight regions previously discussed. The variations in regional circulation are closely tied to the natural cycles of ENSO and IPO. Precipitation projections to the late 21st century, based on climate model simulations of global warming, are also outlined. The expectation is for global warming to strengthen the prevailing westerly circulation across New Zealand (Mullan *et al.* 2001), and enhance the existing west-east rainfall contrast.

Day to day variability in precipitation, its forecasting, and the subject of extremes are discussed. Information on seasonal variation of precipitation in the eight “precipitation regions” described above is discussed, followed by that of the variability associated with ENSO and the IPO. Measured long-term trends and variations in precipitation and circulation during the 20th century are documented.

Day-to-Day Precipitation Variability, Forecasting and Extremes

Day-to-day variations of precipitation across New Zealand result from the progressions of anticyclones, depressions and fronts across the country, and their development or weakening. Weather forecasters predict the precipitation over the next few days using various global or regional scale synoptic analysis and guidance products from models run overseas, together with satellite imagery, statistical guidance based on the model products, and their own understanding of the interaction of weather systems with New Zealand’s complex orography. (e.g. Brenstrum 2002). A developing tool in the forecasters’ armoury is the mesoscale weather prediction model (e.g. Bormann and Marks 1999). This is driven by predictions from larger-scale global models but runs at a much higher spatial resolution, thereby simulating (for example) local orographic influences on precipitation. Development work is currently underway to improve mesoscale modelling systems for New Zealand by assimilating satellite measurements and local observations into them. The output from such predictive mesoscale meteorological

models can in turn be used to drive a catchment runoff and flow model, to provide objective forecasts of river flows and floods (Ibbitt *et al.* 2001).

The processes leading to extreme heavy precipitation over periods from minutes out to a few days can be different across the country, and for different precipitation accumulation periods. The highest precipitation over short periods of time, (minutes out to an hour or two) are generally caused by strong convection. Remnant tropical cyclones travelling across New Zealand can cause very heavy 1–2 day local precipitation in affected places. Intense slow-moving mid-latitude depressions can also cause heavy precipitation over periods ranging from hours to days.

Statistical analysis of measured precipitation time series is generally used to estimate precipitation intensities expected with various return periods, for infrastructure design and hazard assessment purposes. The reader should consult Thompson (2002) for details. Estimates of such intensities for any location in New Zealand can be obtained using the HIRDS (High Intensity Rainfall Design System) software package, which is based on analyses of NIWA, MetService and Local Government precipitation measurements (Thompson 2002).

Seasonal Changes in Precipitation

The seasonal variations in circulation produce an annual cycle of precipitation variation. Figures 2.7 and 2.8 display average monthly precipitation (as a percentage of the annual total) at 8 locations selected near the centre of each of the precipitation regions. The seasonal patterns shown for each are representative for the region as a whole. These are ordered largely from north to south.

Precipitation at Gisborne (R1) has two maxima (Fig. 2.7): one linked to maximum wave depression activity during

the winter months, and the second in early autumn at a time of year when easterly storms, sometimes originating from the subtropics, can occur. Northern New Zealand (R4), represented by Kaitaia, shows the most marked annual cycle of the eight stations selected. It receives a summer precipitation minimum coincident with the southernmost position of the subtropical ridge, and a winter maximum when wave depression activity is highest. The main features shown in the seasonal variation for the western North Island, represented by Wanganui (R6) is a broad precipitation maximum in early winter and again in spring months, with a minimum in summer. The maxima are linked to a combination of more wave depression activity in winter, and stronger westerlies in spring, and the minimum to the migration southwards of the subtropical ridge. Lower late summer precipitation and higher winter seasonal precipitation are also the features for the north of the South Island (R7, Nelson), again a result of winter wave depression activity and summer anticyclonic activity.

The seasonal variation for region R2 (Fig. 2.8), represented by Timaru, displays an autumn precipitation maximum, and a winter precipitation minimum. The latter occurs when the frequency of blocking anticyclone activity is highest. Kaikoura seasonal precipitation (R3) displays similarities to Gisborne with the early autumn maximum and late spring/early summer minimum. However there is only a slight increase in winter months. Milford Sound, representative of the west coast of the South Island (R5), displays a more marked annual cycle. There is a winter precipitation minimum when the frequency of blocking anticyclone activity is highest, with precipitation maxima in January, May and October. Region R8 (represented by Invercargill) is very similar to region R5, except that the

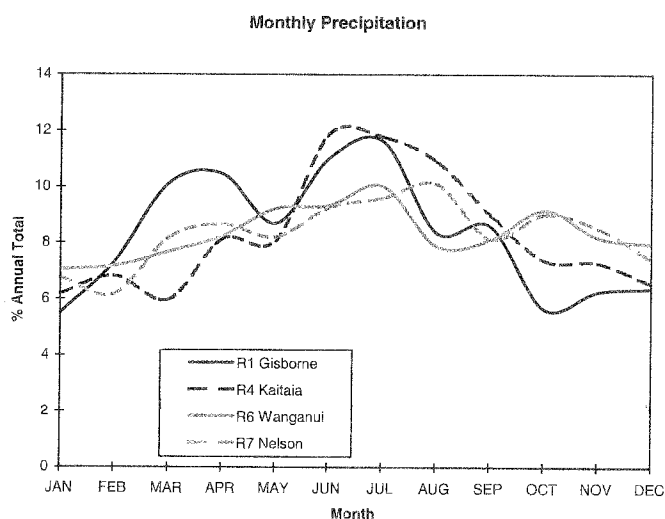


Figure 2.7 Average monthly precipitation as a percentage of the annual total for the 1971–2000 period for stations representing RPC regions 1,4,6 and 7.

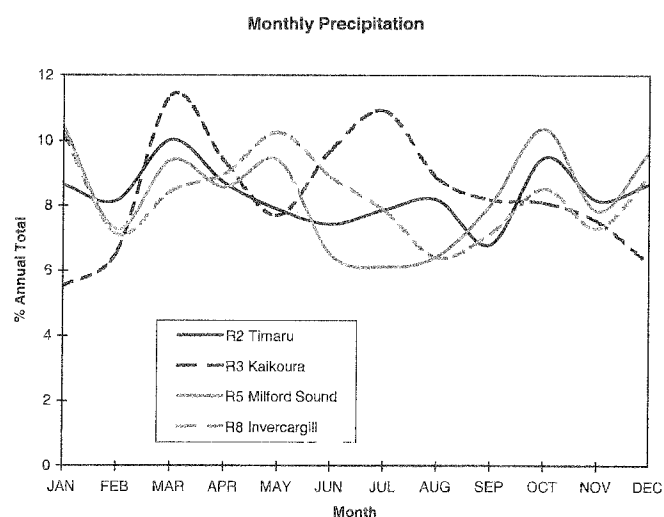


Figure 2.8 Average monthly precipitation as a percentage of the annual total for the 1971–2000 period for stations representing RPC regions 2,3,5 and 8.

winter precipitation minimum occurs later in winter. This minimum relates to the time of year blocking anticyclone frequency is highest over southern New Zealand.

Interannual Changes in Precipitation—the El Niño-Southern Oscillation

The Southern Oscillation, or more generally El Niño-Southern Oscillation (ENSO), is a tropical Pacific-wide oscillation that affects pressure, winds, sea-surface temperature (SST) and precipitation. ENSO is the primary global mode of natural climate variability in the 2-7 year time band defined by sea surface temperature (SST) anomalies in the eastern tropical Pacific. The most common indicator of the intensity and state of ENSO events is the Southern Oscillation Index (SOI, Fig. 2.9), a measure of the anomalous atmospheric pressure gradient across the Pacific-Indian Ocean region. Persistence of the SOI below about -1 coincides with El Niño events, and periods above $+1$ with La Niña events. Another common ENSO index, known as NINO3.4, is the SST anomaly in the region 5°N - 5°S , 170 - 120°W . NINO3.4 and the SOI are very highly (negatively) correlated.

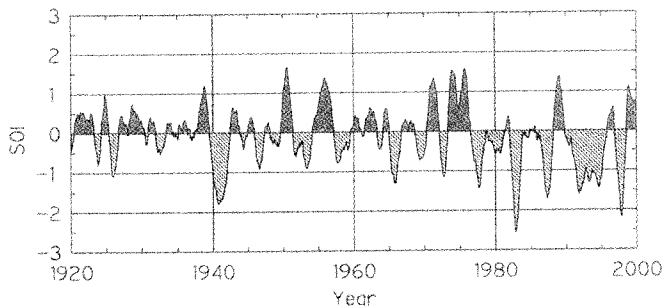
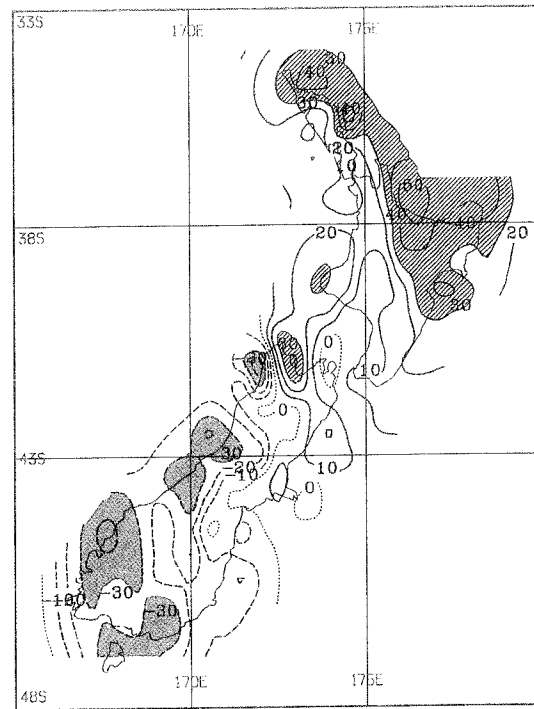


Figure 2.9 Times series of 12-month running averages of the Southern Oscillation Index, 1920-2000.

As El Niño develops, the easterly trade winds weaken and warmer waters in the central and eastern Pacific develop, causing a systematic eastward shift of convection out into the Pacific. Higher than normal air pressures develop over northern Australia and Indonesia, leading to drier conditions or drought. At the same time, lower than normal air pressures develop in the central and eastern Pacific, with excessive rains in these areas and along the west coast of South America. In the El Niño phase, New Zealand experiences stronger than normal southwesterly airflow, which generally results in drier conditions in the northeast of the country, and wetter conditions in the southwest (Gordon 1986; Mullan 1995). Approximately reverse patterns occur during the La Niña phase of the phenomenon, enhancing northeast circulation over New Zealand, with wetter conditions in the north and east, and



to southwest over New Zealand. The El Niño and La Niña extremes of the Southern Oscillation are most evident in the opposite annual precipitation anomalies for RPC4 - Northland/Auckland/Bay of Plenty, compared with the West Coast of the South Island (RPC5, Fig. 2.6).

Decade to Decade Changes—The Interdecadal Pacific Oscillation

The Interdecadal Pacific Oscillation (IPO), also known as the Pacific Decadal Oscillation (PDO, Mantua *et al.*, 1997) is an “ENSO-like” feature of the climate system that operates on time scales of several decades, and has been shown to be associated with decadal climate variability over parts of the Pacific Basin (Folland *et al.* 2002). Three phases of the IPO have identified during the 20th century: a positive phase (1922-1944), a negative phase (1947-1977) and the most recent positive phase (1978-1998). There is now evidence that the recent positive phase has ended (Fig. 2.11).

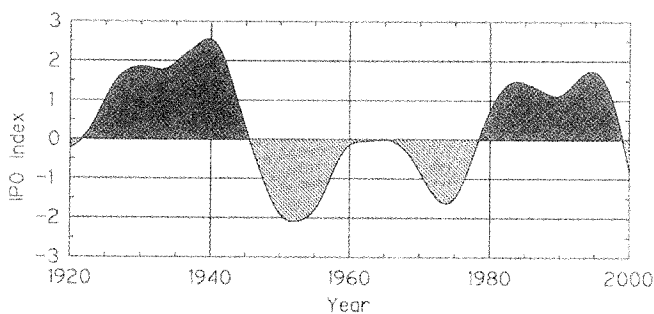


Figure 2.11 Smoothed time series of an index of the Interdecadal Pacific Oscillation, 1920-2000.

Changes in the background state of the tropical Pacific ocean and atmosphere have been documented across the 1977 IPO phase shift boundary (Graham, 1994), and theoretical and modelling studies have shown how these “climate shifts” can alter the development of ENSO events in the same way as has been observed (Wang and Soon-Il 2002). That is, post-1977 El Niños have been stronger and more frequent, and the developing warm SST anomaly spreads eastward from near the Dateline rather than spreading westward from near South America. The IPO also appears to modulate the impacts of interannual ENSO climate variability over Australia (Power *et al.* 1999), New Zealand and the southwest Pacific (Salinger *et al.* 2001), and elsewhere around the Pacific basin (Gershunov and Barnett 1998).

Around New Zealand, in the positive phase westerly winds are stronger with more frequent El Niño events. The negative phase brings weaker westerly winds with more of a balance between El Niño and La Niña events. The shifts in New Zealand regional precipitation, described above

for three periods within the 1930-1994 record with change-points around 1950 and 1975, match the IPO changes in circulation well. After 1950, precipitation increases occurred in northern New Zealand, parts of the east of the North Island as well as in Nelson and Marlborough, whilst regional precipitation decreased in the West Coast of the South Island, southern New Zealand, and parts of the west of the North Island. At the same time, New Zealand climate warmed substantially (as noted by Salinger 1979), consistent with the negative phase IPO and more airflow from the east and northeast. After 1975, there were precipitation increases in the west and south of the South Island, and decreases in the north and east of the North Island, with stronger prevailing west-southwest circulation. Figure 2.12 shows the precipitation change between the negative IPO phase (1946-1977) and the following positive phase (1978-1998). The pattern is similar to that discussed earlier between the periods 1950-1975 and 1976-1994.

Observed 20th Century Variations in Precipitation

New Zealand precipitation trends and variations during the 20th century have occurred as a result of regional circulation changes. Because various parts of the country respond differently to these circulation patterns, the focus is on contrasts across the previously defined eight precipitation regions. The variations in regional circulation are closely linked to the natural cycles of ENSO and IPO.

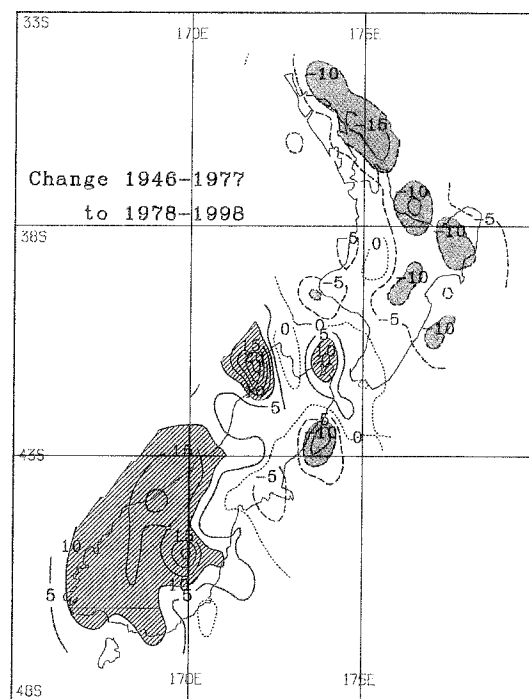


Figure 2.12 Change in annual precipitation, as % of 1961-1990 climatology, from 1946-1977 to 1978-1998. Changes exceeding $\pm 10\%$ are shaded.

The RPC scores on each month (from the Salinger and Mullan 1999 analysis) were averaged over the January to December period to give annual scores. The time series of these annual scores of each precipitation region are shown in the Figure 2.13 for the 1930-1994 time period. A ten-year Gaussian filter was used to smooth the annual values to produce time series to depict longer timescale trends and variations. The main trends in each of the series are considered in the time periods of 1930-1950, 1951-1975 and 1976-1994, with period values shown in Table 2.3. These time periods were chosen after careful examination of the RPC scores, including the corresponding temperature analysis (not discussed here) carried out by Salinger and Mullan (1999), and consideration of time series of the circulation indices in Table 2.1. However, these periods also correspond approximately with alternating phases of the Interdecadal Pacific Oscillation.

East of the North Island

Precipitation anomalies in the east of the North Island were slightly lower in the first period, and slightly higher in the second period, compared with the 1930-1994 average. In particular, winter and spring were drier in the first period, while winter was wetter in the second (1951-1975) period. The last period was the driest of the three: summers were significantly drier and springs wetter.

South Canterbury

Of the eight regions, this shows the least trend in precipitation between the three periods. Autumns were drier and summers wetter in the first period, compared with the 1930-1994 average. Wetter autumns occurred in the second period, and winters were significantly wetter during the last (1976-1994) period.

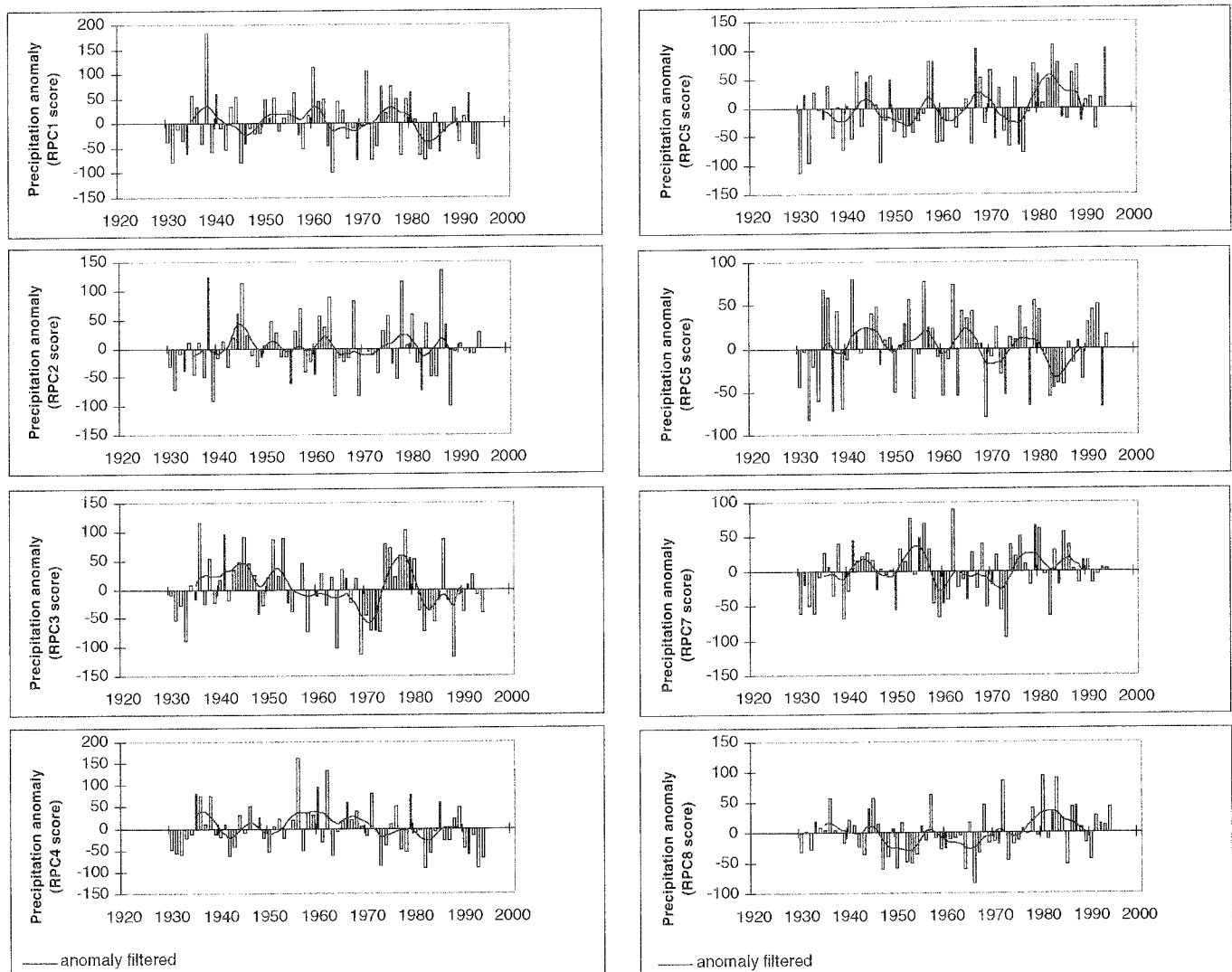


Figure 2.13 Annual loadings (histogram bars) of 8 RPC patterns of Fig. 2.6 for period 1930-1994. Smoothed variations shown as solid line.

Table 2.3 Average annual and season precipitation rotated principal component pattern (RPC) loadings for the time periods 1930–50, 1951–75 and 1976–94.

RPC	1	2	3	4	5	6	7	8
1930-50								
Annual	-3.8	-3.2	10.6	-3.1	-13.5	-1.8	-9.8	-2.3
Summer	11.8	21.7	33.6	2.7	12.0	14.5	12.6	14.8
Autumn	4.8	-26.2	-13.5	-12.1	-26.0	-22.2	-31.1	-13.4
Winter	-16.1	-12.0	-1.8	-1.0	-20.9	3.2	-12.4	-8.1
Spring	-16.4	2.8	16.0	-3.7	-18.2	-6.6	-13.2	0.8
1951-75								
Annual	7.6	1.7	-6.6	16.4	-5.4	3.7	-0.3	-11.9
Summer	7.6	-3.2	-13.4	4.8	-34.7	-0.8	-14.9	-36.8
Autumn	-5.1	24.9	19.6	40.8	15.8	14.8	29.3	14.3
Winter	20.9	-11.3	-5.2	10.1	-14.6	0.9	-6.5	-20.8
Spring	2.0	-1.5	-23.9	10.1	14.9	2.3	-3.7	-5.0
1976-94								
Annual	-5.8	1.3	-3.0	-18.1	22.1	-2.8	11.3	18.2
Summer	-21.6	-19.1	-22.0	-8.9	38.0	-13.8	8.0	36.2
Autumn	1.7	-3.5	-10.6	-40.1	7.8	5.1	-4.0	-4.0
Winter	-9.9	28.2	8.7	-12.5	42.0	-4.9	22.0	36.3
Spring	15.6	-1.3	13.6	-9.5	0.6	4.2	19.4	5.6

North Canterbury/Kaikoura

Trends between the three periods are more distinct for the North Canterbury/Kaikoura area. The first period was the wettest, and the second the driest for this region. The 1930-1950 period was the wettest mainly as a result of significantly wetter summers. The 1951-1975 drier period was a consequence of drier spring and summer seasons. The last period was slightly drier than the longer period average. This was because of drier summers and autumns.

Northland/Auckland/Bay of Plenty

More significant annual trends are seen in RPC4, representing precipitation in northern districts. Although in the first period precipitation is close to the longer period average, the second period is significantly wetter, mainly as a result of wetter autumns, although winter and spring precipitation was above average too. The 1976-1994 period was particularly dry, in comparison with the other periods. The drying trends are largely in autumn.

West Coast of the South Island

This region shows a steady trend of increasing precipitation over the three time periods. The 1930-1950 period was the driest in the west of the South Island, and the 1976-1994 period was the wettest. The early drier period was a result of drier autumns and winters. In the next period, summers were significantly drier, winters slightly drier and autumns wetter than the longer-term average. In the last period both winters and summers were significantly wetter.

Western North Island

The west of the North Island shows very little trend in the annual time series of the RPC scores. There are few seasonal anomalies also. Autumns are drier in the early (1930-1950) period, and wetter in the second period. Summers are drier in the last period.

Nelson/Marlborough

As with the west of the South Island, the area represented by RPC7 shows a similar trend of increasing precipitation. Annual scores have the first period as the driest, with the second close to average. Autumns are drier in the early period, and wetter in the second period, along with drier summers. The third period (1976-1994) is the wettest annually with wetter winter and spring seasons.

Southland/Coastal Otago

This region shows quite distinctive annual differences between the three periods. The first period is close to average, and the second drier than average. There are no strong seasonal anomalies in the first period, but in the second period winters are drier and summers significantly drier than the long-term average. The third period is much wetter than average, a consequence of wetter summers and winters than average.

Circulation Changes

Changes have occurred in the circulation over the New Zealand region. The period from 1930-1950 was one of more south-to-southwest flow over the New Zealand

region, particularly in the 1940s. Precipitation was higher in North Canterbury, particularly in summer, and drier conditions in the north and west of the South Island. This was a consequence of more southerly quarter airflow.

In the 1951-1975 period increased airflow from the east and northeast occurred, compared with the earlier period. The main trends in precipitation were towards wetter conditions in the north of the North Island, particularly in autumn, yet drier conditions in the south east of the South Island, especially in summer. More northeasterly flow accounts for all these trends (Tables 2.2 and 2.3).

The latest period from 1976-1994 is notable for more frequent circulation occurring from the west to southwest over New Zealand. These circulation changes produced significant annual precipitation trends. The north of the North Island has become drier, the only region to do so on an annual basis, mainly because of drier autumns. Conversely the north, west, south and southeast of the South Island have all become wetter, caused largely by increased precipitation in the summer and winter seasons. Winters have become wetter in the north of the South Island, summers wetter in the southeast of the South Island, and both seasons wetter in the west and south of the South Island. Seasonal trends are apparent in other regions: e.g., summers have become drier in the east of the North Island.

CLIMATE CHANGE AND PRECIPITATION

The growth in greenhouse gases in the atmosphere because of anthropogenic activities is expected to be the most important factor forcing climate to change during the 21st century. Within the atmosphere there are naturally occurring greenhouse gases, which trap some of the outgoing infrared radiation emitted by the earth and the atmosphere. The principal greenhouse gas is water vapour, but also carbon dioxide (CO₂), ozone (O₃), methane (CH₄) and nitrous oxides (N₂O), together with clouds, keeps the Earth's surface and troposphere 33°C warmer than it would otherwise be. This is the *natural greenhouse effect*. Changes in the concentrations of these greenhouse gases will change the efficiency with which the Earth cools to space. The atmosphere absorbs more of the outgoing terrestrial radiation from the surface when concentrations of greenhouse gases increase. This is emitted at higher altitudes and lower temperatures and results in a positive radiative forcing, which tends to warm the lower atmosphere and Earth's surface. This is the *enhanced greenhouse effect* -an enhancement of an effect that has operated in the Earth's atmosphere for billions of years due to naturally occurring greenhouse gases (Houghton *et al.* 2001). Over the last 20 million years, the natural concentration of carbon dioxide has ranged from about

190 parts per million (ppm) to 280 ppm. When CO₂ concentrations were low, so too were temperatures, and when CO₂ concentrations were high, it was warmer. By 1999, CO₂ concentrations had reached 367 ppm (IPCC 2001). By 2100, the atmospheric concentration of CO₂ is expected to be at least twice, and possibly almost four times, the pre-industrial level (of 280 ppm). Other greenhouse gases besides carbon dioxide are also increasing.

In order to make projections of future climate, models incorporate past and future concentrations of greenhouse gases and aerosols. The IPCC has modelled climate using seven main scenarios of greenhouse gas and other human-related emissions, based on the IPCC Special Report on Emissions Scenarios (SRES) (Nakicenovic and Swart 2000). From these, a full range of 42 SRES scenarios of atmospheric concentrations have been produced.

The latest scenarios and climate model results indicate that the westerly circulation is likely to strengthen over New Zealand as the globe warms through the 21st century. Changes in annual and seasonal precipitation for Regional Council areas (Table 2.4) have been recently prepared (Wratt *et al.* 2003). These results are based on the New Zealand downscaling study of Mullan *et al.* (2001), but the projected precipitation changes have been broadened to cover the full range of emission scenarios from the IPCC Third Assessment. Figure 2.14 shows the range of precipitation changes from 1990 to 2070-2099 (the so-called "2080s") from the MfE (2003) study. The two panels of Figure 2.14 should be used together to identify the total range at any location. The "Lowest" panel maps the most negative change (or least positive if no model/emission combination gives decreased precipitation), and the "Highest" panel maps the largest positive change (or least negative). In the southwest of New Zealand, the projected long-term trend in annual precipitation varies from no change (or a slight decrease) to a large increase. In the east of the North Island, and northeast of the South Island, the projected long-term trend varies from a large decrease to no change (or only a slight increase) in the annual mean precipitation. These extremes in precipitation changes could occur together; that is, a model/scenario combination with a marked increase in westerly airflow would increase precipitation in the southwest at the same time as decreasing it in the northeast of the country.

These projected changes should not be considered in isolation from the natural variability in precipitation covered earlier in this chapter. These natural variations will continue to impact New Zealand through the coming century, and will be superimposed on the human-induced climate trend. This combination of underlying mean climate (with appropriate global warming adjustments) plus natural variations will produce the extremes which future New Zealand society will have to adapt to.

Table 2.4 Changes for each Regional Council area in seasonal and annual precipitation (in %), for 1990 to 2070–2099, scaled to the full IPCC range of global warming.

	Summer	Autumn	Winter	Spring	Annual
Northland	-3 to +17	-17 to +16	-21 to +13	-46 to +9	-17 to +8
Auckland	-5 to +22	-14 to +15	-15 to +31	-47 to +14	-16 to +19
Waikato	-10 to +37	-21 to +29	-16 to +63	-47 to +24	-19 to +32
Bay of Plenty	-9 to +31	-18 to +15	-10 to +32	-41 to +4	-15 to +10
Taranaki	-5 to +39	-9 to +26	-18 to +50	-26 to +28	0 to +29
Manawatu-Wanganui	-12 to +43	-10 to +43	-26 to +67	-25 to +31	-3 to +39
Hawkes Bay	-31 to +26	-54 to +11	-43 to +25	-59 to -1	-37 to +5
Gisborne	-22 to +19	-56 to +13	-38 to +24	-72 to -4	-35 to +4
Wellington	-20 to +48	-46 to +11	-31 to +62	-40 to +21	-25 to +26
Nelson	-21 to +40	-13 to +9	-2 to +39	-24 to +13	-7 to +12
Marlborough	-23 to +46	-22 to +12	-32 to +37	-37 to +14	-24 to +15
West Coast	-3 to +55	-4 to +46	-16 to +113	-23 to +40	+1 to +55
Canterbury	-28 to +51	-44 to +36	-70 to +93	-47 to +34	-41 to +42
Otago	-11 to +46	-25 to +46	-22 to +129	-16 to +45	-9 to +57
Southland	+2 to +48	-17 to +51	-21 to +105	-17 to +39	0 to +50

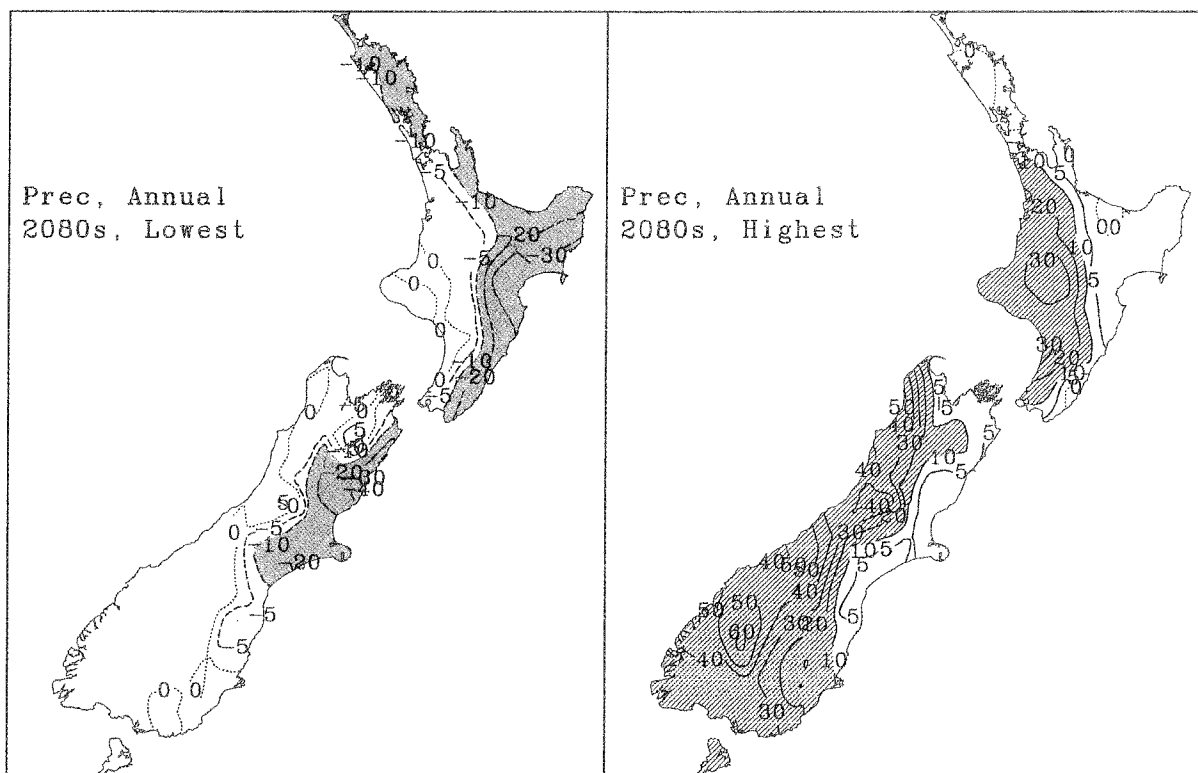


Figure 2.14 Annual mean precipitation changes (in %) relative to 1990, for 2070–2099 (“2080s”), scaled to full IPCC range of global warming. Contours every 10%, plus contours at -5% and +5%. Shading for changes more extreme than -10% and +10%.

Figure 2.14 shows just the annual mean changes in precipitation. Changes in variability of precipitation are also very important in considering future water resources. It would be desirable to know about potential changes in rain-bearing storms, and also changes in extreme precipitation at the daily (or even shorter) timescale. This is an area of active research, but presently there is only limited information available. Mid-latitude storms (also known as extra-tropical cyclones) derive their energy from two sources: the large-scale meridional temperature gradient and the condensation of moisture. In the Southern Hemisphere, both these energy sources are projected to increase under global warming. It would therefore be reasonable to infer that “more storminess” was likely. However, it is currently uncertain whether this would mean more intense storms, or a higher frequency of passing cold fronts, or some combination of these. Moreover, a general increase in the Southern Hemisphere does not equate with an increase locally in the small sector of the hemisphere that New Zealand occupies. The regional changes would be sensitive to changes in prevailing wind strength and direction, which vary considerably between models.

As far as extreme precipitation goes, a warmer atmosphere can hold more moisture (about 8% more for every 1°C increase in temperature), so the potential for heavier extreme precipitation in a warmer world certainly exists. The IPCC in its Third Assessment declared that more intense precipitation events are “very likely over many areas”. An early study for the Australasian region (Whetton *et al.* 1996) suggested there could be a reduction in the return period of heavy precipitation events. By 2030, “no change through to a halving of the return period of heavy precipitation events” was expected, and by 2070 “no change through to a fourfold reduction in the return period”. Subsequent analyses that give guidance for the New Zealand region (Semenov and Bengtsson 2002; MfE 2003) support this earlier finding. There are many aspects that still need to be clarified. For example, are increases in heavy precipitation (and associated flooding) just as likely in the northeast of New Zealand, where mean precipitation is likely to decrease, as in the southwest where wetter mean conditions are expected? If increases in heavy precipitation were to occur at the same location as decreases in the mean, this would imply longer and more extreme dry periods between the heavy precipitation events.

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Chapter 1

Advances in freshwater sciences and management

Ross Woods and Clive Howard-Williams

INTRODUCTION

New Zealand has some of the world's highest quality fresh water, ranking in the top ten for both its abundance and its cleanliness (United Nations 2003). Although, with a population of just 4 million people, the human pressures on our freshwater resources are relatively light by international standards, New Zealand has a wide range of water-related hazards and problems, including floods, landslides, droughts, pollution, aquatic pests, algal blooms and a diminishing natural biodiversity. A key to sustaining our freshwater environments, maintaining our "clean, green" image, and reducing risks from some of these environmental hazards is to continue to improve our knowledge of freshwater resources and hazards, and the effects on them of human activities. A basic understanding of how freshwater systems function is fundamental to good management. New Zealand's freshwater research is continually adding to that storehouse of understanding and knowledge, which is applied through water resources management and hazard management.

This chapter provides an overview of the current state of freshwater science and management, with particular emphasis on developments over the last decade. When *Waters of New Zealand* (Mosley 1992) was published, New Zealand science and resource management were just beginning to operate within a completely new system of organisational and legislative structures. After a decade of living in this new world, we take this opportunity to analyse some of the changes, looking first at science and then at resource management, and concluding with a glimpse into the future.

SCIENCE

The development of understanding in any area of science generally follows a pathway from definitions to

theory to validation or application. A typical development might follow these steps.

- Define the discipline: e.g., freshwater science includes studying the hydrology and ecology of lakes, rivers, catchments, aquifers and wetlands.
- Classify the objects being studied, such as freshwater environments: e.g., classifying lakes as oligotrophic, mesotrophic or eutrophic.
- Develop techniques to quantify the objects being studied: e.g., water-level recording, nutrient concentration sampling, biotic indices.
- Develop conceptual ideas of cause-and-effect: e.g., the inputs, storages, transformations and outputs of a lake nutrient balance.
- Cycle repeatedly through the elements of the scientific method: observe, make a hypothesis, test the hypothesis.
- Add to our conceptual ideas those hypotheses that best explain our observations. This is the development of theory.
- Use the combined knowledge in a predictive capacity: e.g., provide models of systems that will allow them to be managed.
- On very rare occasions, adopt a new paradigm that overturns our collection of accepted concepts.

In this chapter we review progress in each of these stages of freshwater science, with a focus on the most recent developments, as they relate to New Zealand.

Definitions

The breadth and complexity of work within freshwater science is increasing rapidly. While limnology and hydrology are discrete areas of science in their own right (Hutchinson 1957; Chow 1964; Wetzel 1975; Maidment 1993; Dingman 1994), the two disciplines are intimately interlinked. For example, there has been a better appreciation of this with the development over the last

decade of “eco-hydraulics” (Jowett 2000; Nikora *et al.* 2004; Chapter 15). Advances in the area of “hyporheic ecology” have strengthened our understanding of this linkage. The hyporheic zone, the subsurface region beneath and beside a stream where sediments are saturated, is a distinctive environment (e.g., Stanford and Ward 1988; Boulton *et al.* 1997). The special hydrological conditions in the hyporheic zone provide an environment where surface and subsurface waters are exchanged and important biogeochemical transformations change the nutrient status of freshwater, and so a new ecological niche is recognised and named (see Chapter 32). For example, unconfined aquifers in Canterbury have been shown to support a diverse and apparently complex ecosystem of biofilms on the sub-surface gravels, supporting a variety of small animals (e.g., Fig. 1.1) in a complex food web (Chapter 29). It is believed that these organisms maintain groundwater quality by grazing on living and non-living biofilms, which trap organic contaminants entering these valuable aquifers. The ecology of groundwater ecosystems is an exciting emerging research area.

We now view groundwater and surface freshwater resources as a single linked unit. In New Zealand this sits well with the Maori concept of hauora, in which there is a wholeness or unifying structure to all ecosystems. There is also recognition, with increased understanding and quantification, of the connections between freshwaters and the marine coastal zone. The physical structure of the water column of coastal embayments, such as the Marlborough Sounds, is critically dependent on freshwater inflows. It is this structure, in terms of stratification of the water column, that provides the habitat for plankton and therefore contributes to the productivity of the Marlborough Sounds, the location of almost 80% of New Zealand’s mussel farming industry (Ross *et al.* 1998). These

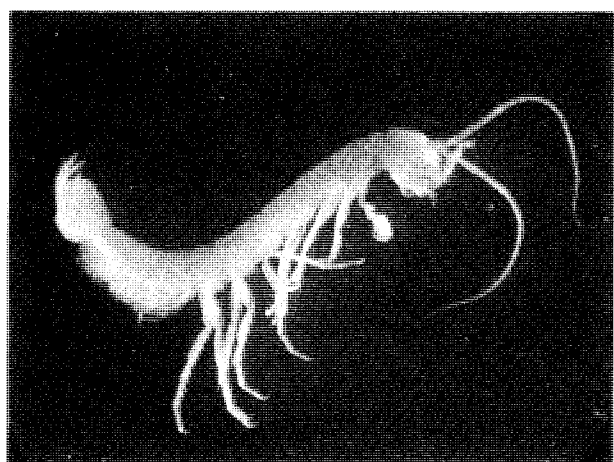


Figure 1.1 *Phreatoicicus typicus*, a Canterbury groundwater isopod, grows to about 20 mm long and can occur in very high densities at sites of organic enrichment.

Photo: G. Fenwick

same freshwater inflows are also the source of faecal coliform bacteria, whose presence determines when, or indeed if, mussels can be harvested.

Freshwater science is therefore a part of our overall study of the global environment that links climate through the land to the ocean.

Classification of freshwater environments

Classification is a necessary step in the organisation of knowledge, providing a framework for describing patterns and structures. It also allows us to put new information in context and assess its significance. Lake and wetland classification systems have been with us for several decades (Hutchinson and Löffler 1956; Cowardin *et al.* 1979). These classifications have an international basis, but frequently are overlain with subdivisions based on local conditions. New classification schemes for both river and land environments have recently been developed in New Zealand, in response to both the availability of suitable data in electronic form, and the need for this information for resource management. With the apparently vast array of river types globally, it is perhaps not surprising that there is no agreed international classification system for rivers. As a result, local classifications are developing in many countries, in response to local needs for a scientific basis to water resource management.

The “100 Rivers” study (Biggs *et al.* 1990), with its nationwide “snapshot” sampling of water quality, periphyton, invertebrates and fish, provided the overview needed to organise knowledge of river ecosystems in New Zealand. Using the lessons learned from that wide-ranging study, especially the contrasts in community composition between adjacent sampling sites on rivers with distinctive hydrological regimes, a classification of rivers emerged. The River Environment Classification, or REC (Snelder and Biggs 2002; Snelder *et al.* 2004) is a tool for ecosystem-based resource management, providing a context for inventories of river resources, and a framework for effects assessment, policy development, development of monitoring programmes and interpretation of monitoring data, and state-of-the-environment reporting. The River Environment Classification has been used to classify all the rivers of New Zealand at a 1:50,000 mapping scale. It characterises river environments at six levels, with names reflecting factors that influence the river environment—Climate, Source of Flow, Geology, Land Cover, Network Position and Valley Landform (Fig. 1.2). The classification system is now well developed, and it has been used in several instances as a method for managing water resources (Snelder and Biggs 2002)

This river classification system joins established classifications for lakes (according to depth, stratification cycle and nutrient enrichment status), wetlands (Ward and

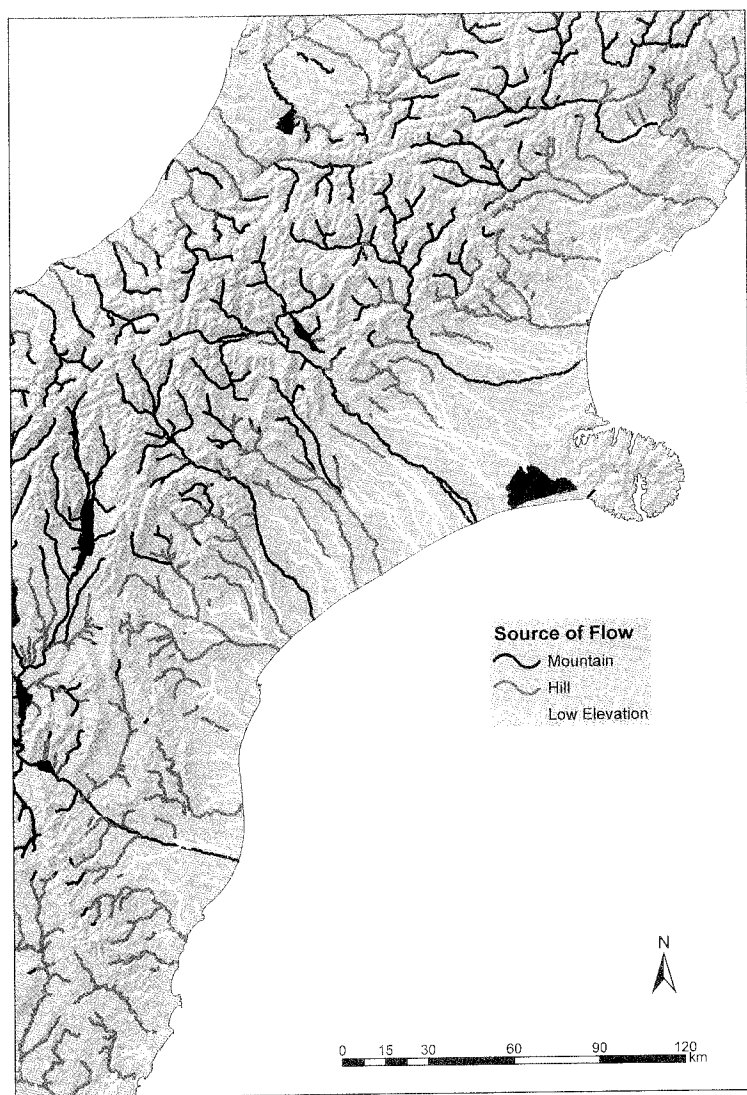


Figure 1.2 River network of New Zealand showing rivers partitioned into three classes based on source of flow.

Lambie 1999) and, more recently, riparian zones (Quinn *et al.* 2001) and estuaries (Hume *et al.* 2003) now being used for management in New Zealand. It will be interesting to see how this river classification system influences those being developed in other countries, and whether a unifying global classification of rivers will eventually emerge.

Quantitative assessment and measurement

Quantitative assessments in freshwater research frequently centre on the volume and flow of water, the concentrations of its constituents (e.g., ions, nutrients and sediments), and the diversity and abundance of aquatic plants and animals. Together, these parameters quantify the structure and dynamics of aquatic systems. Measurement has been strongly affected by advances in technology—for example, developments in electronics have completely revolutionised some aspects of the storage,

transmission and analysis of data. The most common development is an enhancement of existing methods for measuring a given environmental parameter.

- Equipment for measuring water velocity has advanced from propeller meters to acoustic Doppler current profilers and velocimeters (e.g., Goring and Nikora 2002), with a consequent enormous increase in spatial and temporal detail. Acoustic Doppler current profilers have revolutionised lake and coastal studies, allowing near instantaneous measurements of currents in a vertical structured water column (Fig. 1.3).
- Weather radar has been enhanced to measure rainfall with resolutions as fine as a few seconds and a few hundred metres (Austin 2001). This opened new opportunities for understanding the role of rainfall variability in hydrology (e.g., Fig. 1.4) and engineering design (Seed *et al.* 1999).
- Stand-alone water quality analysers have been developed, such as automatic nitrate analysers that can provide automated laboratory analyses at remote sampling sites.
- Automated soil moisture sensors and soil moisture mapping techniques (e.g., Wilson *et al.* 2003) allow point measurements to be interpreted in terms of variations in soil properties, land use, and topography (Fig. 1.5).

Continuous water quality monitoring sensors, either mounted singly or in off-the-shelf instrumentation packages such as “multi probes”, have become reliable, increasingly complex and cheaper. The “Biofish” (Gall 2002) for example, is an instrument that is towed behind a boat—it undulates through the water column to specified depths, continuously recording water temperature, salinity, light transmittance and chlorophyll fluorescence, providing a real-time picture of the structure of the water column (Fig. 1.6).

Remotely operated measurement systems, using advances in the field of robotics, are revolutionising limnological measurements. One such development is in autonomous underwater vehicles. *Tantan*, a Japanese underwater vehicle (Fig. 1.7), can stay underwater for several days, travelling on pre-programmed pathways at varying depths and locations, sampling water, analysing its quality (including using high-resolution optical microscopy) and storing the data (Kumagai *et al.* 2001). Data are downloaded continuously through surface and satellite transmission systems. Such instrumentation has a future in New Zealand, particularly for lakes of the central North Island, which are undergoing rapid change and are

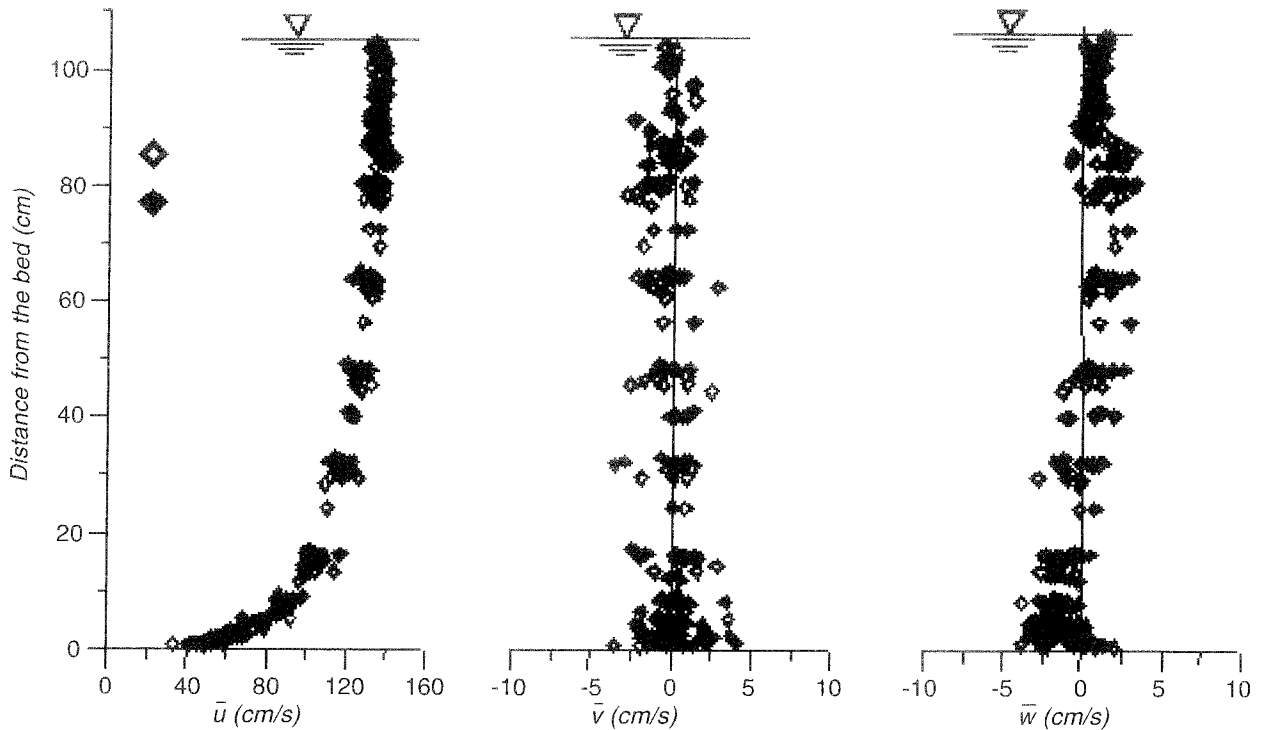


Figure 1.3 Vertical distribution of local mean longitudinal \bar{u} , transverse \bar{v} , and vertical \bar{w} velocities. The data are from acoustic Doppler velocimeter (ADV) measurements in the Balmoral Irrigation Canal, Canterbury (Nikora and Goring 2000).

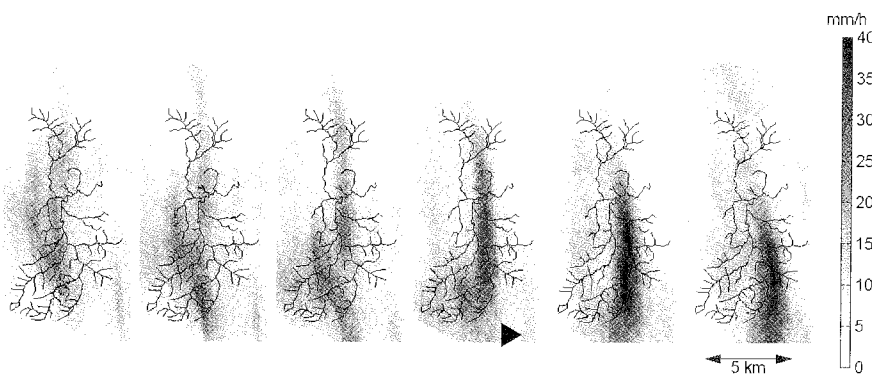


Figure 1.4: Six maps of rainfall rate measured on a 150 m grid (one map every two minutes), over a 120 km² region near Warkworth on 8 August 1998, during a MARVEX campaign (e.g., Woods *et al.* 2001). Note the movement and intensification of rainfall structures of the order of a few hundred metres wide, whose lifetimes are of the order tens of minutes.

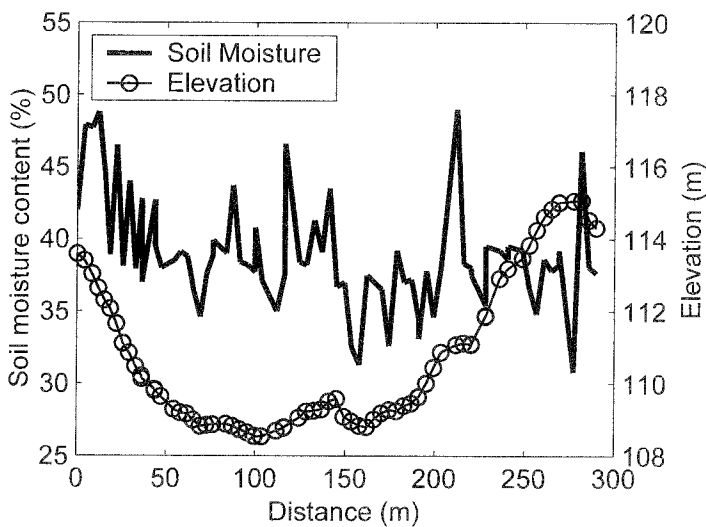


Figure 1.5 Soil moisture content in the top 30 cm of the soil, and elevation, every 4 m along a 300 m transect near Warkworth, in March 1998 (see Woods *et al.* 2001; Wilson *et al.* 2003).

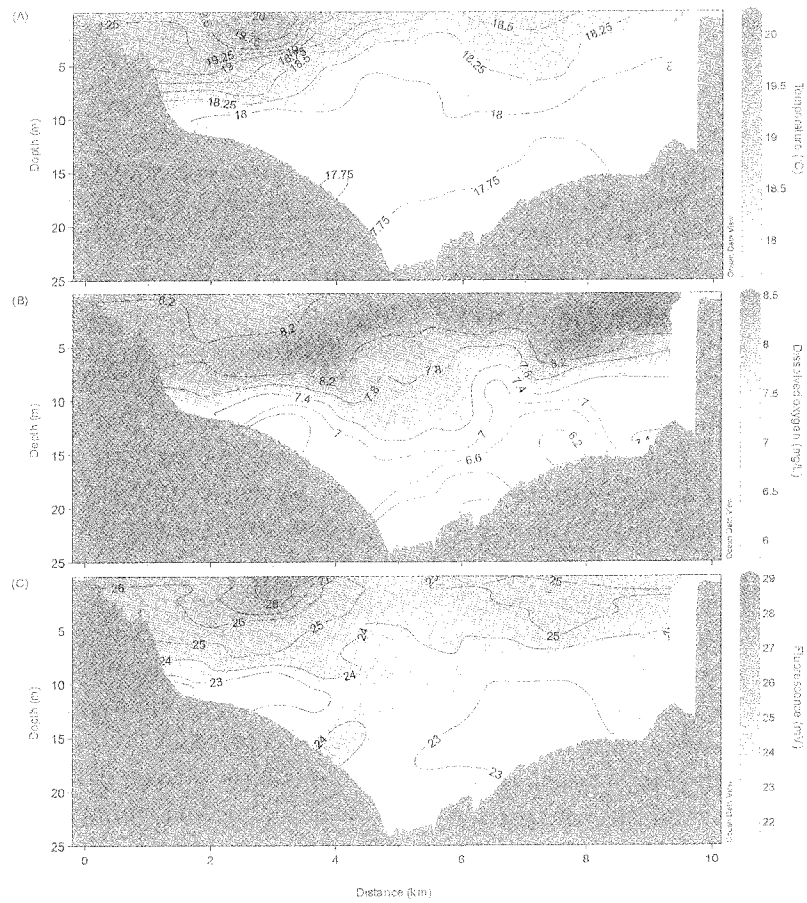


Figure 1.6 Biofish profiles of (A) temperature, (B) dissolved oxygen and (C) fluorescence in Lake Rotorua, March 10, 2004. Transect from Ngongotaha (0 km) to Ohau Channel (10.1 km). Profile ends at 9.2 km. Data from David Burger, The University of Waikato.

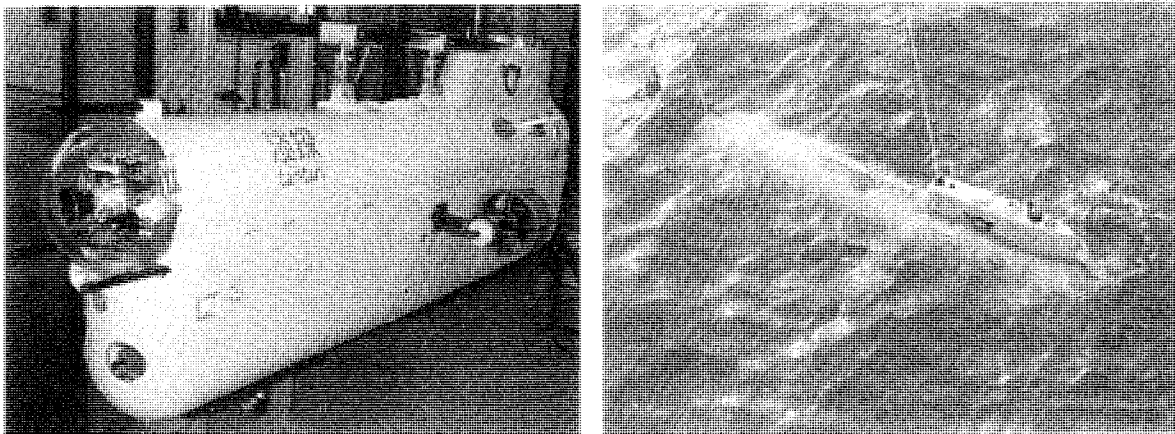


Figure 1.7 The autonomous underwater vehicle *Tantan*: (a) side view, and (b) a view under the water surface of Lake Biwa, Japan.

Photos: M. Kumagai

subject to increasingly comprehensive restoration works (Petch 2004; Dell 2004).

A successful new use of remote sensing is the measurement of riverbed topography by airborne LIDAR, providing a vertical resolution of ~20 cm every few metres over areas of several kilometres (Fig. 1.8). This allows very detailed hydrodynamic modelling of rivers and their habitats, especially for the complex topography of braided gravel-bed rivers (e.g., Hicks *et al.* 2002).

New Zealand researchers and management agencies have just begun to evaluate the effectiveness of remotely measured surface water colour in lakes to detect algal blooms and check water clarity. New generations of satellites will shortly provide colour images at 250-m-pixel size resolution (Behrenfeld and Boss 2003). The use of airborne remote sensing to detect and manage blue-green algal blooms in freshwaters has been investigated in Australia since the early 1990s (Jupp *et al.* 1993), and hyperspectral radiometry using satellite sensing is now under close study for water quality and riparian vegetation studies of inland and coastal waters (Environmental Remote Sensing Group 2003). In 2003 an exploratory survey of the Waikato River, using airborne hyperspectral radiometry, was carried out by CSIRO (Ian Hawes, pers. comm.), to assess the effectiveness of this method for detecting cyanobacteria in New Zealand waters.

Data networks

New Zealand has several institutions that maintain networks for freshwater monitoring—these provide standardised measurements at a relatively stable set of locations. This standardisation is very important for providing reliable baseline information for both long-term

science studies and freshwater management. The established networks for measuring climate, water levels (rivers, lakes, groundwater) and river flows have evolved through a shift to regionally-based water resource archives, in place of the former centralised system operated by the Ministry of Works and DSIR, and the movement of the Water Resources Archive into a Nationally Significant Database holding data collected by numerous organisations (Pearson 1998). Three new national freshwater networks have emerged since the late 1980s: for river water quality (Smith *et al.* 1996), groundwater quality (Rosen 1999), and soil moisture. The Freshwater Fish database has continued to grow and become more widely accessible through the internet. A more recent development, in response to management needs, is the growth in regional monitoring of water use, as pressures on water resources grow and resource management matures. Water-use monitoring includes close metering of water flows to agricultural as well as urban and industrial users, and, where appropriate, metering of the water leaving user premises.

A significant change is the improved timeliness and accessibility of data, through the use of telemetry and electronic data transmission (most recently using the internet). This has enabled much more complete data sets to be collected, with much less missing data, because instrument failures can be detected remotely and therefore repaired more rapidly.

The development of geographic information systems (GIS) has allowed improvements in the presentation of map-based data (e.g., Fig. 1.2), which aids in communication of the detailed information collected at many locations. More significantly, it has created opportunities to find cause-and-effect links between catchment properties and river parameters, most obviously in the development of spatially-distributed catchment simulation models such as Basin New Zealand (Cooper and Bottcher 1993), but also in many other types of analyses. For example, environmental mapping models such as SUSTANZA (FertScience 2003) depict, on a farm map, risks of fertiliser and nutrient loss, to allow the farmer to concentrate on mitigation methods or special fertiliser applications in restricted areas on the farm.

Conceptual ideas of cause and effect

Fundamental to the study of freshwater are the hydrological cycle (Fig. 1.9) and biogeochemical cycles (e.g., Fig. 1.10),

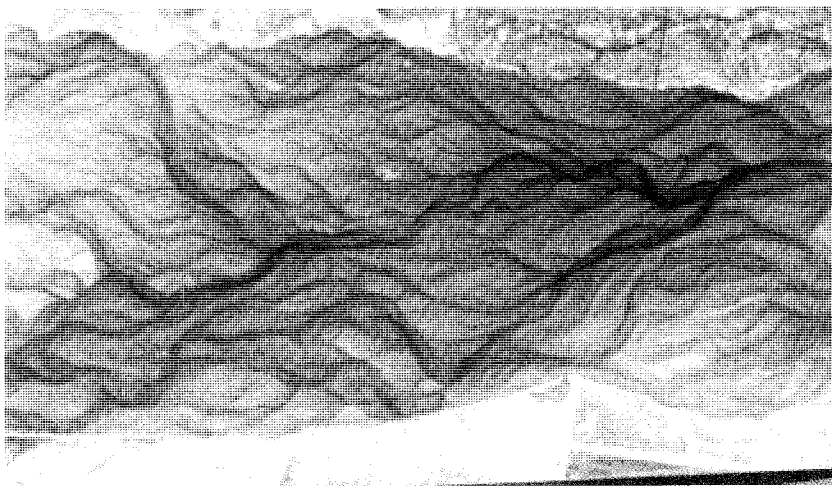


Figure 1.8 Digital elevation model of Waimakariri River bed, developed from LIDAR measurements of exposed river bed, soundings of wetted area, and relationships between water colour and depth. Reprinted with permission from Hicks *et al.* (2002).

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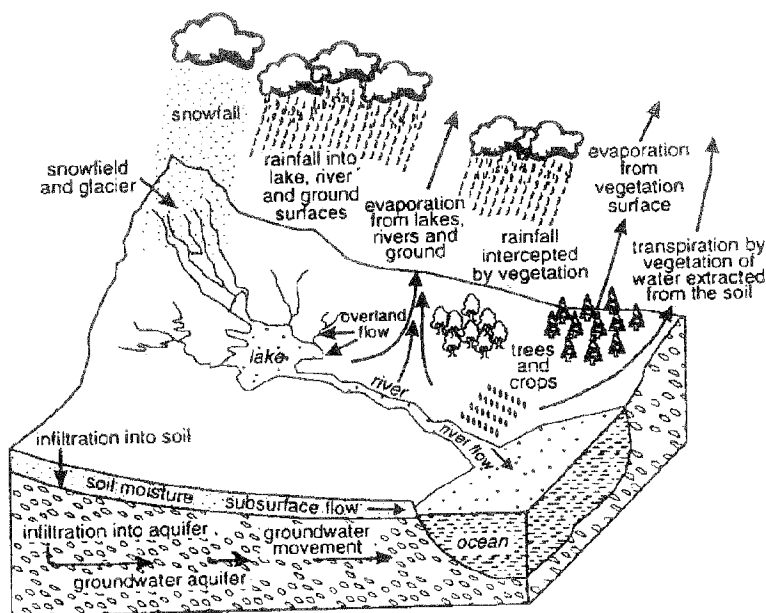


Figure 1.9 Generalised hydrological cycle

biological migration. More complex concepts such as competition, predation, and food webs involving pathways of matter and energy, growth, mortality and decay also all play important roles in structuring ecosystems. The development of tools and methods that have allowed the elucidation of microbial food webs has advanced our understanding of lake ecosystems (Laybourn-Parry *et al.* 1995). The role of the so-called nano-plankton (size range 2-20 μm , pico-plankton 0.2-2 μm , and now ultra-plankton < 0.2 μm) in carbon cycling has been quantified in some systems and it is now realised that natural free-living viruses (virus-like particles) in freshwaters are also abundant enough to play a measurable role in plankton biomass (Kepner and Wharton 1998).

Complexity

Advances are being made now in unravelling system complexity in areas such as temporal and spatial variability, and in the interactions between all the factors that influence freshwater ecosystems. The non-linear nature of natural system function is currently being addressed in the relatively new concept of “complex systems science” (Ziemelis 2001). Complex systems science is based on the behaviours of systems as whole, which can be studied in ways that provide more information than studies of individual parts. Although in its infancy, the science of complexity has grown very rapidly in the last few years and it is reasonable to predict that it will allow a better understanding of, for instance, freshwater ecosystems as whole systems. One example is the development of a so-called science of networks (e.g., Watts 2003). New knowledge about sudden changes in large complex networks, and the extent to which particular types of

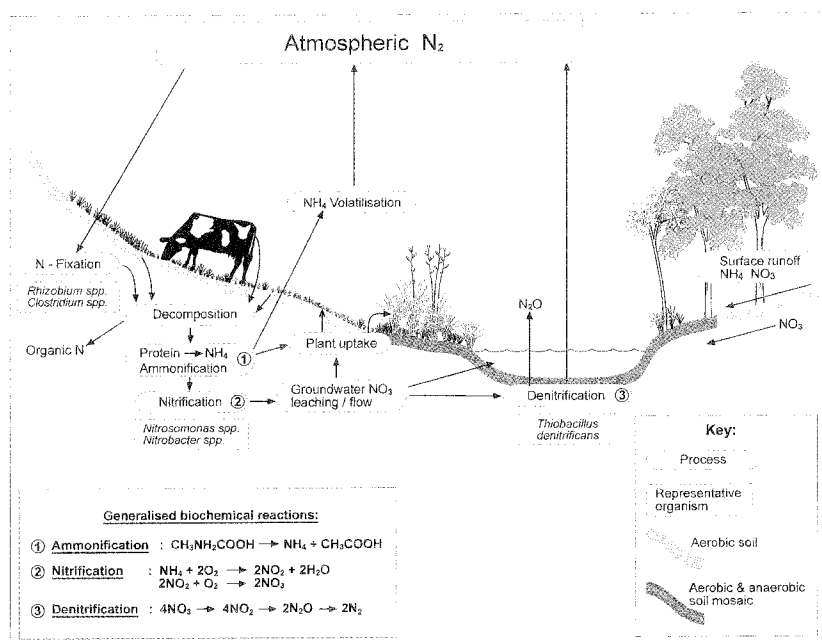


Figure 1.10 Generalised nitrogen cycle for a rural landscape.

which are based on the conservation law for mass. In their simplest form, these cycles are simply tools for making balanced budgets. However, they do depend on more complex ideas about transformation: the concepts of change of state of water (solid, liquid, gas), of biochemical transformation (chemical reactions, the nitrogen cycle), and biological change (evolution and biogeography). Equally important scientific principles are those that govern movement, such as Newton’s second law (force equals mass time acceleration), Fick’s law of diffusion, and

interconnections play a role in determining their susceptibility to system collapse, have application to ecosystem changes that may propagate through a nutrient cycle or a food web. Network science offers an alternative view of the problem, and may offer new insights into such difficult questions as river channel meanders, bedform patterns in sediment beds (Werner 1999) and even complex biological behavioural phenomena such as animal aggregations.

Spatial scale

Considerable effort has gone in to investigating the effect of spatial scale and spatial variability on the hydrology and ecology of freshwater systems. From a hydrological perspective, more sophisticated concepts have been developed (e.g., Sivapalan and Kalma 1995; Blöschl *et al.* 1997; Grayson and Blöschl 2002). The major conclusion of a widespread international research effort on physical hydrology is that no fundamental spatial scale exists, and that spatial variability in hydrological response depends on both spatial scale and on hydrological regime. Seyfried and Wilcox (1995) give an example of these ideas, showing changes in a cold arid mountain catchment, from the dominance of vegetation at scales of a metre or two (where the location of individual shrubs affects patterns of water balance), through scales of tens and hundreds of metres, where topography determines the lateral movement of blowing snow and flowing water, up to scales of kilometres where rainfall gradients are the dominant feature of the hydrological landscape. A crucial next step will be the adoption of an internationally agreed classification of hydrological systems, to allow the hydrological regime to be unambiguously identified, and thus provide a context for regional studies of spatial variability and scale.

An assessment of the importance of spatial scales has long been a major research direction in freshwater ecology (e.g., Townsend *et al.* 2004). Understanding the relative importance of differing factors that structure stream communities at differing scales is crucial to sound management of these systems.

Long-term variability

Research has shown the importance of inter-annual and decadal-scale climate variability as factors causing changes in freshwater quantity (Mosley 2000), quality (Scarsbrook *et al.* 2003) and biology (e.g., McKnight *et al.* 1996). The El Niño–Southern Oscillation (ENSO) phenomenon (changes in water temperature and atmospheric pressure over the Pacific Ocean) is responsible for El Niño and La Niña events lasting months or even years (Salinger and Mullan 1999), during which the frequency of floods and droughts changes markedly (McKerchar and Henderson 2003), with follow-on effects on water chemistry and biology. In addition, there are periods of a few decades where the relative frequency and intensity of ENSO events change—for example, the period 1978–1998 included an unusually large number of El Niño events, with conditions in the east of New Zealand being drier and warmer than normal (Mullan 1995). This longer-term cyclic weather pattern is known as the Inter-decadal Pacific Oscillation (IPO). This adds an extra degree of complexity to any study of trends in freshwater variables, and especially

studies of the effects of changes in factors such as land use or land management.

Ecosystem stressors

The effects of land use on aquatic ecosystems seldom, if ever, result from a single contaminant or “stressor” alone. The effects generally show up in response to several factors acting in concert. This is known as the “multiple stressor” framework and recent research has allowed us to apply it to freshwaters. This framework recognises that factors interact to produce an effect that any one of these alone does not provide. For instance, the effect of farming on nutrient enrichment of streams is exacerbated when riparian shade is removed to make more use of “productive land”. The result is more light for nuisance algal growth to utilise the added nutrients. In addition, removing shade results in higher stream temperatures (Rutherford *et al.* 1997; Chapter 11). Higher stream temperatures result in higher rates of growth and decay, so that nutrient transformations are more rapid, with even more non-linear ecosystem feedbacks. The recent work by Suren *et al.* (2003a, b) has shown that the effect of low flows on stream ecosystems is highly dependent on stream nutrient status. Ecosystems of enriched streams are more sensitive to flow reductions than those of non-enriched streams. Understanding and unravelling these combinations of multiple stressors is a currently an important research area and scientific challenge, particularly in the framework of water allocation.

Prediction

Over the last decade in New Zealand, our ability to model lake and stream processes at a catchment scale has advanced significantly. In part this has been due to the need to understand the consequences of changes in land use. Models such as Basin New Zealand (Cooper and Bottcher 1993) and GLEAMSHELL (Elliott *et al.* 2000) have been applied in several catchments where land development has intensified. Predictions of increased nutrient loss from the catchments draining into Lake Taupo as a result of increasing dairy farming have been made possible using complex catchment models such as GLEAMSHELL. In this case, the model simulated hydrologic, chemical and biological processes in almost 30,000 computational cells distributed through the catchment. The model used detailed climate, soil, topography, and land-use inputs specific to the Taupo catchment. Within the model, nutrients were transported from the computational cells to the lake, accounting for point-source loads and losses in the stream network. The outputs from the GLEAMSHELL model are being linked to lake ecosystem studies, using the one-dimensional hydrodynamic lake model DYRESM (Imberger and

Patterson 1981), combined with a lake water quality model CAEDYM (Hamilton and Schladow 1997). This will form the basis for a nitrogen cycling model of Lake Taupo to predict the effects of dairy conversions on lake water quality and clarity (R. Spigel pers. comm.).

There have also been advances in modelling the physical aspects of freshwater systems, include flood forecasting, water resource modelling, and river hydrodynamic modelling. The linking of weather models with catchment models significantly extends the flood prediction lead-time for New Zealand's short, rapidly responding rivers (Ibbitt *et al.* 2001). Such models are now being linked to two-dimensional floodplain inundation models (e.g., Beffa and Connell 2001), which rely heavily on high-resolution river topography data (e.g., Fig. 1.8). These same models are also being used to manage water resources—for example, the Topnet model has been used to develop a national hydrological model to compile a set of water accounts (John Gudgeon, Statistics New Zealand, pers. comm.). The effects of water allocation on the availability of instream physical habitat have been predicted using both one- and two-dimensional models (e.g., Jowett 1997; Carter *et al.* 1998).

Fewer attempts have been made to model biotic communities within freshwater systems, however, fish communities have been modelled by Joy and Death (2002, 2003), who used a River Invertebrate Prediction and Classification System (RIVPACS)-type model to assess physico-chemical conditions and macroinvertebrate communities.

New paradigms

Since freshwater science is generally conducted not as an integrated science but as several separate disciplines, a paradigm shift could occur by changing focus to the connections between disciplines, or alternatively an individual discipline could be revolutionised by breakthroughs in theory, computation or measurement. One example of a *potential* paradigm shift for environmental science is the development of the Gaia concept (Lovelock 2000, 2003), which views the earth as a living organism. This concept has the potential to radically change the practice and public perception of environmental science. However, it has not had widespread acceptance among the science community (e.g., Dawkins 1998), partly because of the difficulty of testing any predictions that are based on Gaia theory against appropriate measurements. However, it is becoming evident that as we understand more about systems and their linkages, the network widens towards a greater understanding that all systems in our biosphere are linked. Within the hydrological cycle (Fig. 1.9), the concept of "hydrological connectivity" (Pringle 2003) is gaining

momentum. Hydrological connectivity is used in a global context to describe how the hydrological cycle fits into the landscape. It is only in the last decade that surface water-groundwater linkages have been potentially viewed as a single entity (Winter *et al.* 1998).

A New Zealand example shows the extent of this connectivity, linking through to climate. Phytoplankton production drives the productivity of mussel farms in Pelorus Sound. This, in turn, is linked to inflows of freshwater, which affect the stratification of the water column, and hence the depth of the mixed layer for plankton growth. The freshwater inflows also import nutrients that foster plankton growth. The freshwater inflows are, in turn, linked to rainfall and land use in the Pelorus River catchment. Rainfall is linked to the wider patterns of Pacific climate. Global climate change research over the last decade has improved our understanding of interconnected regional and global systems, and this research will aid in further unravelling long-term variability in freshwater systems.

At the other end of the scale is the remarkably rapid advance in molecular science, which has changed our ability to distinguish levels of organisation within living organisms. This has an effect on the management of freshwater resources. For instance, advances in molecular genetics over the last decade have allowed the discrimination of nine new species of galaxiids in New Zealand (e.g., Waters *et al.* 2001, McDowall and Waters 2002, 2003;). This has increased the number of galaxiid species of our fish fauna by a massive 69% and the number of native fish species in the total fish fauna by 37%. For example, five new species of galaxiids have been described in rivers of the Otago peneplain and Southland, and two new species have been described for the McKenzie Basin tributaries (R. McDowall, pers. comm). These advances will have important consequences for river management, and for New Zealand's obligations to international treaties such as the international Convention on Biological Diversity.

Molecular genetics has also allowed us to better appreciate rates of evolution and how species adapt to separation and altered environmental conditions. In the last 100 years, for instance, the genetics of the original Chinook salmon stock introduced to New Zealand from the Kokanee River in California have changed significantly as a result of isolation and of our river environments, which differ from those of the North American west coast (Quinn *et al.* 2001; Unwin *et al.* 2003).

Science infrastructure

The comprehensive restructuring of government departments involved in freshwater science, and of the mechanisms to fund science, has seen radical changes in

the landscape of freshwater science in New Zealand since the publication of *Waters of New Zealand* (Mosley 1992).

In 1992, freshwater research functions in former government departments such as the Department of Scientific and Industrial Research (DSIR), Ministry of Works, Ministry of Agriculture and Fisheries, Forest Service and Meteorological Service were transferred to new entities called Crown Research Institutes (e.g., NIWA, Landcare Research, IGNS and others). These institutes are owned by the government, have a corporate structure, and their purpose is to carry out research while remaining financially viable and independent of direct government input. The principal national funding agency for freshwater research, the Foundation for Research, Science and Technology, moved from an annual funding of many relatively small programmes in 1992 to fewer longer-term (5-6 years) larger research programmes in 2004. In 2004, there is a requirement for agencies that use science and technology for management (for example, the Department of Conservation, Ministry for the Environment and regional governments) to support, or have a significant input into, Foundation research programmes. Although this change has had major implications for both universities and non-government science organisations, the structures of those entities have remained unchanged for the most part. However, universities have seen an increasing importance attached to attracting and maintaining student numbers, and this has made it more challenging to find academic research time. The most important external changes for universities has been the introduction of the Marsden Fund as a new source of basic or “blue skies” research funding.

MANAGEMENT

Decision-making processes

New Zealand’s decision-making and monitoring processes for water changed completely with the passing of the Resource Management Act (RMA) of 1991. Catchment Boards, the National Water and Soil Conservation Authority and the Planning Tribunal were all abolished, and Regional Councils, the Ministry for the Environment, the Parliamentary Commissioner for the Environment, and the Environment Court were all created. Under the RMA, considerable authority has devolved to regional councils: to write policy statements and resource plans, to make decisions on resource allocation through the processing of consent applications, and to enforce consent conditions. Change continues, with a review of the Resource Management Act under consideration at the time of writing. There is also scope for the development of national environmental standards and national policy statements, although none have been

completed that directly affect freshwater.

The Environment Court was established under the Resource Management Act to hear appeals relating to matters that have been considered by a Council (e.g., resource consent applications). A Court usually consists of at least one Environment Judge and one or more Environment Commissioners. Environment Commissioners often conduct mediation to facilitate the resolution of matters arising in cases before the Court. For example, one Environment Court hearing is widely applicable to freshwater management in New Zealand. The court found in 1998 that afforestation could have a significant effect on stream flow, so that the planting of forests in water-short parts of Tasman District is a discretionary activity, meaning that it can be restricted. This decision has influenced the development of regional water plans in Tasman District and also other parts of New Zealand.

The office of the Parliamentary Commissioner for the Environment independently reviews and provides advice on environmental issues and the system of agencies and processes established by the Government to manage the environment (see Foreword to this book). That office has produced reviews on several water issues, especially as they relate to urban environments, where environmental pressures are high and are subject to frequent change.

An additional responsibility for Regional Councils is to monitor the state of the environment. Although under way throughout New Zealand, this type of reporting is still in development, and in parallel, the Ministry for the Environment is developing environmental performance indicators, and is working with Statistics New Zealand to develop a national set of water accounts.

A major influence on the management of lakes and rivers has been the need for resource consent renewals for all the large hydroelectric schemes (affecting the Waikato, Tongariro, Whanganui, Waitaki, Clutha and Waiau Rivers). In every case this process has required the assembly of a large body of evidence on the environmental effects of these schemes, thus producing integrated studies of large freshwater systems. Taken jointly, these documents provide a significant contribution to New Zealand knowledge of large freshwater systems.

Demand for water

The major change in water demand since the 1980s has been an increase due to agricultural intensification—most obviously with increased demand for irrigation in the east of the South Island. There is approximately 500,000 hectares of irrigated land in New Zealand, 350,000 hectares of which is in Canterbury. The area of irrigated land has been increasing at around 55% each decade since the 1960s (Lincoln Environmental 2000). Based on weekly volumes, the total water allocation for irrigation in New

Zealand is in excess of 330 m³/s (Lincoln Environmental 2000). With the loss of government funding for irrigation schemes in the late 1980s, and the sale of existing schemes at that time to farmers, many individuals have developed their own water supplies since then. Recently, many new schemes have been proposed, in tandem with regional development initiatives. As the availability of water for "run-of-river" schemes diminishes, off-river storage is being considered, sometimes in tandem with hydro-electric development. The continuing growth in demand for electricity (due mainly to steady economic growth and population increase), in comparison to the modest growth in generation capacity, resulted in shortages of electricity in 1992, 2001 and 2003, although a number of other factors also contributed to those events.

Decisions regarding the use of water are complicated by the need to consider other natural resources used for electricity generation (coal, Maui gas), and their environmental impacts (e.g., emission of greenhouse gases). The best possible information, science, models, and tools are needed for such major decisions on water use and allocation.

To link the environmental, economic and social impacts of water management and water use, a group of central government agencies is developing national accounts for natural resources. The accounts include both stocks and flows, i.e., both changes in national inventory for each resource and details of how each resource is being used. Accounts for energy, fish and wood resources have recently been released by Statistics New Zealand. A first set of water accounts is due for publication in mid-2004. Components of the water accounts include surface water, groundwater, and snow and ice, as well as some aspects of water use. Future aspects of water use will be affected by developments in water metering. The concept of a "water market" is now being discussed, in which water becomes a tradeable commodity, like energy from electricity generation. Water markets exist in several countries, including parts of Australia, and close monitoring of water use is a pre-requisite for such economic and resource-use initiatives.

An interesting development resulting from the growth in demand is the emergence of direct competition among consumptive uses of water. Questions of water allocation are no longer just a matter of balancing the interests of the abstractor and the in-stream values. The use of the Resource Consent process (sometimes in the absence of an operating Regional Policy Statement or Regional Resource Plan) has in the past produced piecemeal decision-making. Where resources are not under pressure, this can work adequately, but in areas where water is short, it may not address competing abstractive uses, where factors such as irrigation efficiency, economic and social

benefit, and sustainable development become crucial.

The rapid intensification of agriculture, most notably the increase in dairy farming in Canterbury and Southland, has significantly increased the use of surface water and groundwater for irrigation, and in some places has placed pressure on these systems. In addition, the growth in irrigation has led to an increase in discharge to streams of irrigation bypass flow, containing additional nutrients. While the extra water may be beneficial to streams during low flow periods in summer, the additional nutrients may promote nuisance weed growth, and cause changes in the natural ecosystems that are adapted to summer low flows. The pressure on the groundwater ecosystem and complex food web that exists under the Canterbury plains was referred to earlier (see Chapter 29). Research is urgently needed to determine whether groundwater pollution from land use is adversely affecting these systems, which help maintain high quality groundwater in the first place.

Aquaculture productivity can be significantly affected by freshwater inflows from rivers to coastal water bodies (marine conditions can also play an important role). It is interesting to speculate whether the value of river water quality and quantity to coastal aquaculture will be sufficiently recognised that resource planning in the coming decade will include this interaction between freshwater and coastal marine systems.

Environmental challenges

Water quality legacy

Considerable recent attention has been given to the poor water quality of several of the Rotorua lakes, and the declining water quality of Lake Taupo (e.g., Dell 2004; Petch 2004; Chapter 21). A number of potential causes for these problems have been identified. Of critical importance in evaluating options for the remediation of these lakes is a scientific understanding of the linked groundwater-surface water interactions. Groundwater dating (Hadfield 2003; Gordon and Morgenstern 2004) has shown that the water in many of the aquifers supplying lakes Rotorua and Taupo is between 20 and 80 years old. Many of these aquifers are enriched with nitrate, thought to result from the rapid development of drystock farming in the 1960s. As agricultural practises have resulted in an intensification of land use since then, many New Zealand aquatic systems are faced with a long legacy of water quality problems, even if immediate steps are taken to minimise groundwater enrichment.

In both cases the science aspects are relatively uncontroversial, but management responses are still widely debated, and the political element of decision-making is very significant. Controls on land use and the conversion of agricultural land may be applied in the Lake Taupo catchments to reduce the nitrogen leached by pastoral

agriculture (Environment Waikato 2003). The concept of a “nitrogen cap”, i.e., a maximum annual load of nitrogen that can be permitted to leave a catchment, is currently under discussion. The development of land-use activities to keep nitrogen levels under the “cap” will demand innovative research in farming practices, new methods of maximising nitrogen retention (or alternatively loss to the atmosphere – see Fig. 1.10), and control of diffuse-source pollution. This is a challenge for combining science (hydrology, ecology, agronomy, forestry and animal husbandry) with economics and community involvement. An innovative start to this process, which signals a new approach to lake research underpinned by community values, is seen in the Taupo 2020 project (Huser 2002).

Biosecurity

New Zealand’s freshwaters have been increasingly invaded by aquatic plants and pest fish that significantly deplete the natural biodiversity, thus altering the natural life-supporting capacity of New Zealand’s aquatic ecosystems (Howard-Williams and Davies 1988; de Winton *et al.* 2003; Champion and Clayton 2003). The invasive organisms can alter ecosystem structure and function, and reduce recreation and landscape values. Agriculture, tourism, recreation and hydropower industries are all affected by invasive freshwater organisms (Howard-Williams 1993; Clayton 1996; Champion and Clayton 2003). New Zealand is home to some of the world’s worst aquatic weeds, and plants such as hornwort (*Ceratophyllum demersum*) and lagarosiphon (*Lagarosiphon major*) are spreading south to colonise previously unaffected water bodies in the South Island. The continuing spread of lagarosiphon, in spite of long-term monitoring and (limited) control of this plant in Lake Wanaka, has been a recent public concern.

Some coarse fish have been deliberately spread: e.g., koi carp (*Cyprinus carpio*) has already reached pest status. Others, such as rudd (*Scardinius erythrophthalmus*) and tench (*Tinca tinca*), have spread widely with a potential to become significant pests, altering ecosystem structure and natural biodiversity richness (de Winton *et al.* 2003; McDowall 1990; 2003). Because numerous agencies are responsible for aquatic weeds, for instance, fragmentation of responsibility is one of the stumbling blocks to effective control of aquatic pests.

Several significant invasions of aquatic pests have occurred in New Zealand waters in ways not recorded in other countries. One hundred and sixty years ago watercress (*Nasturtium officinale*) reached pest proportions in several New Zealand rivers. In 1857 the Canterbury Provincial Government offered a reward for methods to eliminate watercress, which was choking Canterbury rivers. Watercress, however, is no longer a problem and is now a

valued food crop throughout New Zealand, with plans for cultivation of this species as a managed crop. The most recent example of an unexpected invasion was the rapid spread through North Island lakes and rivers of the nuisance alga “water net” (*Hydrodictyon reticulatum*) (Hawes *et al.* 1991). The plant was first recorded in the field in 1988 and within two years had proliferated to an extent unrecorded in the international literature, with considerable economic consequences to the local tourist industry (Wells *et al.* 1999). Other such invasions can be expected.

Biodiversity

Following ratification of the international Convention on Biological Diversity in 1993, New Zealand’s Biodiversity Strategy (Department of Conservation and the Ministry for the Environment 2000) was released to provide a response to the national loss of ecosystems and species. New Zealand has a characteristic and distinctive aquatic flora and fauna with a high degree of endemism (Harding *et al.* 1997; Boothroyd 2000). Wallis (1994) pointed to inbreeding, loss of variation through drift, genetic differentiation among populations and identification of taxa as major aspects of biodiversity (population genetics) relevant to conservation in New Zealand. Recent government initiatives in biodiversity have resulted in several major advances. These have mostly been due to two factors: advances in molecular genetics, and the increased interest in sampling small, unusual, and often remote, habitats.

The last decade has witnessed a rapid advancement in molecular genetic techniques to differentiate species and sub-specific groupings, and in the application of these techniques in New Zealand. One of the most noteworthy has been the discovery of at least five new galaxiid fish species, referred to earlier in this chapter. Molecular genetics, in combination with classical taxonomy and biogeography, has also allowed an improved understanding of the place of New Zealand’s biodiversity in a regional or global framework (McDowall 2002). For instance, diadromous fish species exhibit a relatively uniform genetic structure across wide geographical ranges (Allibone and Wallis 1993; McDowall 2002), whereas non-diadromous fish species tend to show a genetic structure associated with their distribution patterns.

The focus on biodiversity has led to increased attention to small, previously little studied ecosystems and habitats, such as springs, seeps, mountain tarns, groundwaters and ephemeral waterways (Chapters 28–32). One dramatic example has been the revision of the taxonomy of hydrobiid snails in New Zealand by Martin Haase, who paid special attention to new collections in seepages and groundwaters. Previously, only 16 species in six genera were

known. The new revision has resulted in at least 63 species in 15 genera (Collier 2004). Similarly, it is clear that the knowledge of freshwater algal taxa in New Zealand is incomplete, with many undescribed species in remote habitats. It is estimated that there are about 7000 mountain tarns above 600 m altitude (H. Hurren pers. comm.) and up to half of the diatoms in one recently studied montane wetland system in the Southern Alps may be species new to science (Kilroy in press). This work, together with the discoveries of the new galaxiid species, has reinforced the need for conservation measures aimed at small aquatic habitats. Periphyton communities in wetlands and even large lakes have received scant attention, and this is a fertile area for biodiversity work in the future (Chapter 15). The remarkable fauna of ground waters and of the hyporheic zone of rivers (Chapters 29, 32) is a biodiversity area which is now gaining in importance with the realisation that surface waters and ground waters need to be managed as a single resource.

Freshwater management practice

A number of management practices are widely established in New Zealand. For example, minimum flows are set for rivers, based on mean annual low flows (Ministry for the Environment 1998), and flow sharing is used to maintain river flow variability (abstractions are reduced as flow diminishes). Best management forestry practices are used to reduce erosion, and riparian areas are managed (Boothroyd and Langer 1999). However, there have been some efforts to define values associated with rivers, and then determine the flows needed to support those values (Ministry for the Environment 1998). One such effort is the development of decision support tools to assist in setting low flows, such as LOWFAT (Low Flow Assessment Tool). The use of such value-based minimum flows is likely to continue to grow as the Resource Management Act becomes more widely implemented.

The positive effects on streams of riparian fencing on farms and keeping farm animals out of streams are now widely accepted and are rapidly being implemented in some areas (Collier *et al.* 1995; Ministry for the Environment 2001; Quinn 2003; Parkyn *et al.* 2003). While there are clear benefits to such techniques, their effectiveness varies from site to site. However, there is a need to temper expectations of the benefits of planting riparian areas by recognising appropriate time scales, and the length and spatial arrangement of the planted reaches. There must be time for canopy closure by the riparian vegetation, and protecting small streams and headwaters will have greater benefits on larger downstream rivers than protecting the banks of the large rivers themselves. Riparian protection is widely seen as a positive management practise by the Dairy Industry, and the 2003

“Dairying and Clean Streams Accord” between the industry, Regional Councils, and central government stresses the value of stream protection as an important sustainable farming practice. Sustainable farming is a vital part of New Zealand’s international marketing of agricultural products.

Sustainable farming practices for the maintenance of water quality require cooperation between the farming industry and science and resource managers. The development of the recent “Code of Practice for Fertilizer Use” promoted by the New Zealand Fertilizer Manufacturers Association (Furness 2003) is another example of an industry response to an environmental problem—deteriorating water quality—that is seen as harming the industry’s image and therefore potential markets.

Community

Two major movements have emerged that are influencing community views on freshwater and active participation in freshwater research. These are the Landcare Trust and associated landcare groups around the country, and the resource management infrastructures being established by Maori tribes as they become effective in managing their freshwater resources.

The New Zealand Landcare Trust has provided an organised grassroots movement that has developed local projects to restore and manage freshwater and associated environments. Of particular importance is the Integrated Catchment Management (ICM) initiative of the Trust. This links community groups, management agencies and science and education organisations, using a catchment focus. In particular this aims to enhance young people’s participation in environmental management and ecosystem restoration. Successful community-led Integrated Catchment Management projects are seen in the Taieri Trust and the Whaingaroa Harbourcare community group at Raglan. Integrated Catchment Management is currently being developed for the Lake Brunner catchment (Edgar 2004). Research-led Integrated Catchment Management was recently established in the Motueka catchment (e.g., Basher 2003), linking scientific and social research into catchment management, and involving a wide range of stakeholders. The Motueka catchment has a high diversity of land uses and extends to the sea, where land use may be affecting marine aquaculture in the coastal zone.

Through Treaty of Waitangi settlements of the last 15 years, a number of iwi have developed approaches to freshwater resource management within their tribal boundaries. An example is the Te Runanga o Ngai Tahu Freshwater Policy Statement (Te Runanga o Ngai Tahu 2000), which focuses on the management of freshwater

resources within the rohe, the environmental outcomes sought by Ngai Tahu, and the means by which these outcomes will be achieved.

Special decisions, such as the transfer of ownership of the bed of Lake Taupo to Ngati Tuwharetoa, have led to major changes in rights and responsibilities for Lake Taupo. Ngati Tuwharetoa are now actively involved in decisions regarding the management of Lake Taupo and the research that backs up that management. The Taupo 2020 project (Huser 2002), funded through the Ministry for the Environment, aimed to develop a long-term plan for Lake Taupo and its catchment to protect values identified by the community. The project works across three fronts: community, tangata whenua (in this case Ngati Tuwharetoa), and central and local government. The science in Taupo 2020 concerns science communication, and linking science to management action.

Maori values relating to freshwaters are being further defined to allow the recognition of these values in resource management (Tipa and Tierney 2003). The next stage is in the development of management tools that allow for an assessment of changes in land use or other changes to freshwaters on these values (C. Severne pers. comm.).

PROSPECTS

Scientific discoveries

New areas of research continually emerge as science techniques change, and as new issues dictate research needs. The study of global connectivity in the earth's biosphere has become more tractable, with global models of climate, and an appreciation of interactions of climate and freshwater resources. Global and local approaches to freshwater management are discussed in Kumagai and Vincent (2003). The next decade will see a greater appreciation of groundwater and surface waters as a unified system. A number of water-related questions in both the physical and biological sciences need to be addressed.

What are the impacts of flow variability (how low and for how long) and offsite water harvesting? What is the national role of ephemeral waters? We need to define the pathways taken by water through soil and shallow groundwater.

We need to study erosion at scales small enough to manage, but large enough to affect the health of receiving waters. Much of the existing research is either at the scale of experimental erosion plots, or on large catchments, where networks have monitored areas with a wide range of land uses.

Some ecological aspects include multiple stressors in freshwater systems, the interactions of light, nutrients and mixing hydrodynamics on lake phytoplankton, and the role of the microbial food web and free-living viruses in freshwater ecosystems.

Complex systems science may be a promising approach to providing models of freshwater systems, and we need to develop predictive models for assessing the impacts of land use on water quality and sediment transport.

The concepts of hydrologic connectivity (climate-rainfall-surface water-groundwater) and the transport, transformation and attenuation of contaminants as an interactive system will provide tools and realistic predictive models for water resource management. The linkage of the hydrological cycle at different spatial scales will also contribute to conservation biology (Pringle 2003) and to the information needed for the restoration of degraded freshwater ecosystems.

Economics and the environment

If the industrialisation of New Zealand continues as it has over the last 20 years, and our use of freshwater resources and services is similar to that of other countries with comparable economies, we can expect a substantial increase in direct day-to-day use and management of many rivers and lakes. At first there will be more abstraction and diversion of river water, and later more between-season storage of water for uses such as irrigation, energy generation, maintenance of physical habitat values and dilution of pollution during periods of low flow.

If existing values are to be maintained while this type of development proceeds, then diffuse sources of pollution will need extraordinary management. Diffuse source pollution is an almost inevitable consequence of agricultural intensification, which is likely to continue as a major economic activity in New Zealand. Control of diffuse source pollution will be a key to sustainable farming in the next decade.

We can reasonably expect a stronger connection to develop between economic benefits and environmental services: the development of new methods of bottom-line accounting will undoubtedly facilitate this. A rational discussion about the relative costs and benefits of economic activity can then begin. The increasing recognition of New Zealand as a relatively unspoilt place is likely to place additional pressure on our industries to maintain this position, leading to industry certification of environmentally-friendly practices in the production of goods for export.

Already the metering of water is growing rapidly. In future one might also expect the metering of pollution—at present it is hard to imagine how this might be done, but at the very least some kind of proxy measure of pollution generated by economic activity seems inevitable.

We already have international obligations with respect to atmospheric emissions of greenhouse gases. One day we might also have international agreements to reduce ocean pollution, and corresponding limitations on

emissions of nutrients from rivers to oceans. Perhaps in the next decade we will see the setting of nationwide minimum flows to estuaries and coastal embayments to provide adequate dilution or adequate water to ensure the maintenance of coastal stratification and other physical process that ensure sustainable coastal ecosystems. Will we see a nationwide maximum quota on nutrient export to the ocean? Will we see internationally and locally tradable nutrient discharge permits, in the same way that a market for tradable carbon credits is emerging? And finally, will we see a water market developing in New Zealand to offset increasing competition for water resources that cannot keep up with demand? There have been major changes in freshwater science over the last decade and we look forward to an integrative volume on "Freshwater Resources of New Zealand" in a decade from now, to highlight the rapid extent of the change that appears to be on the horizon.

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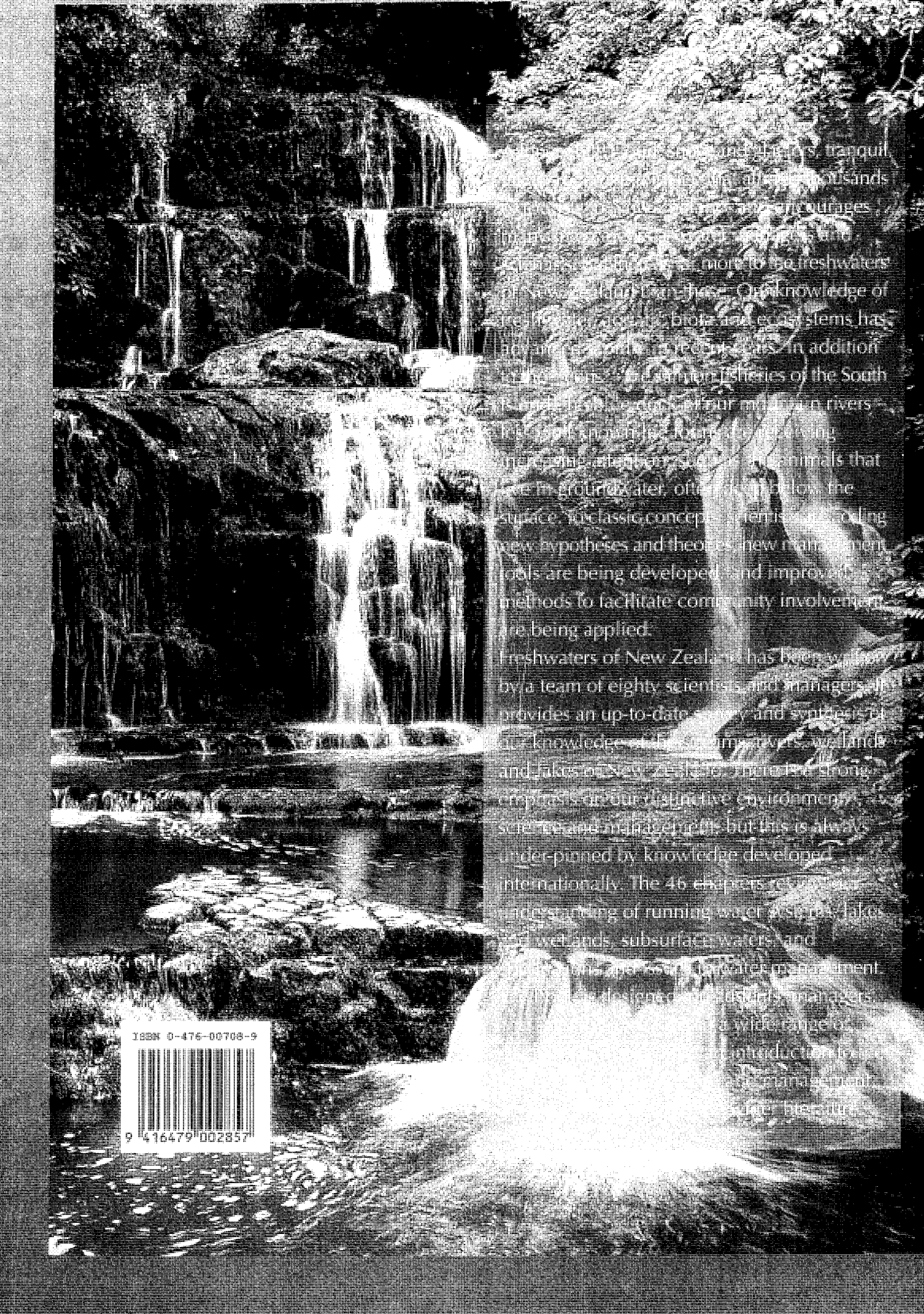
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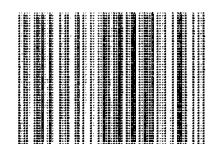
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freshwaters of New Zealand have been a tranquil and beautiful landscape for all the thousands of years since the first human outrages, and the first European explorers and colonists. But it is not only the freshwaters of New Zealand that have. Our knowledge of freshwater ecosystems, both natural systems and human-altered systems, is increasing. In addition to the growing scientific literatures of the South Island, the last decade of our millennium rivers has seen a number of important scientific findings, including the discovery of animals that live in groundwater, often far below the surface. New classic concepts, systems, and new hypotheses and theories, new management tools are being developed, and improved methods to facilitate community involvement are being applied.

Freshwaters of New Zealand has been written by a team of eighty scientists and managers, all of whom provides an up-to-date survey and synthesis of our knowledge of the country's wetlands and lakes to New Zealand, including a strong emphasis on the distinctive environmental science and management goals, but this is always underpinned by knowledge developed internationally. The 46 chapters cover the understanding of running water, water lakes, groundwater, subterranean water, and the management of water resources. Water management is discussed from the perspective of industry managers, government managers, and environmental managers, and includes a section on the management of water resources. The book is a comprehensive review of the current literature.

ISBN 0-476-00708-9



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