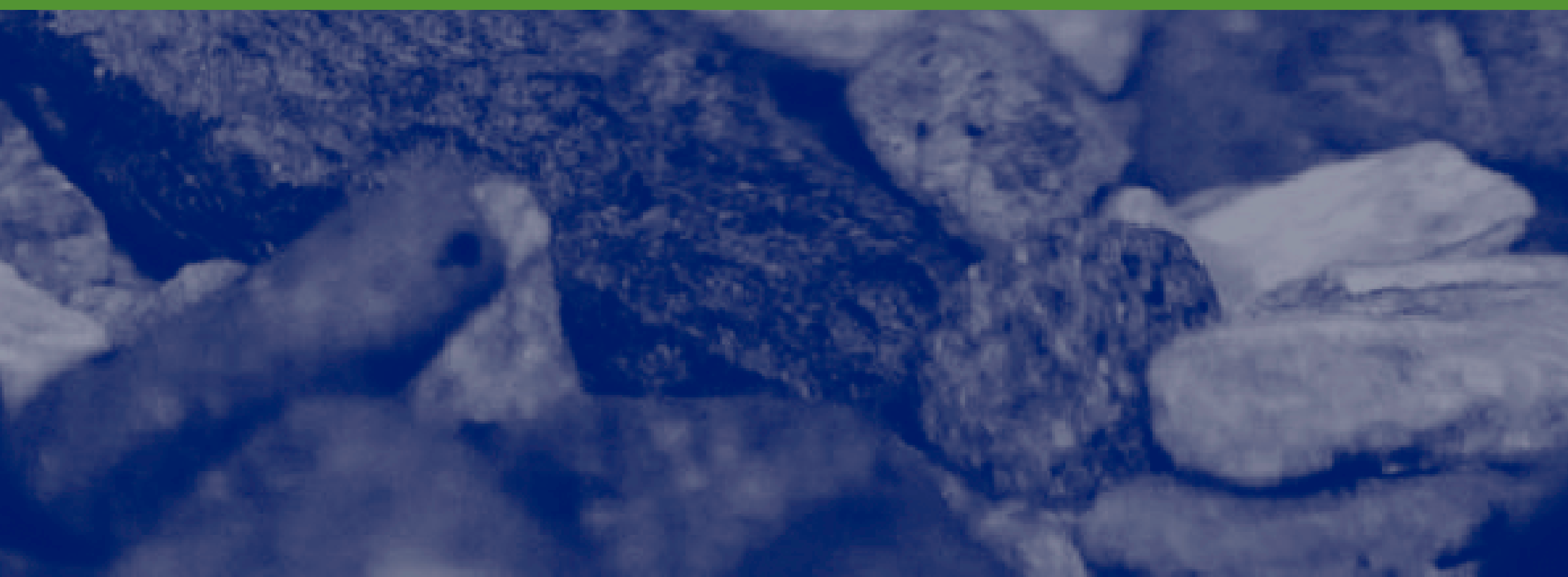
Ministry for the
Environment
Manatū Mō Te Taiao

LAKE MANAGERS' HANDBOOK

Land-Water Interactions



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SECTION 1

Background

The *Lake Managers' Handbook* was published in 1987 and is still widely used and highly regarded by lake managers and others involved in water management. The purpose of the present document is to update the information on land and water interactions provided in the *Handbook*, using the significant amount of knowledge and experience that has been accumulated since it was published.

'Land-water interactions' refers to the relationship between water quality in lakes and the processes in the surrounding catchment. This document describes methods for quantifying how human activities in catchments affect material and nutrient loads into lakes, and how this can affect lake water quality and lake ecosystems.

Some aspects of land-water interactions were discussed in Chapters 14 to 17 in the original *Lake Managers' Handbook*. With the increase in understanding of the catchment as an important component of lake ecosystems and the overriding effect catchment processes have on lake water quality, 'land-water interactions' has now become a large scientific discipline, encompassing a broad suite of processes – from rainfall and groundwater hydrology, to nutrient cycling and biotic responses. In the revised *Handbook* we update information on water and sediment sources to lakes, focusing on nitrogen and phosphorus sources. In doing so, we concentrate on modelling sediment and nutrient sources to lakes, determining loads, and suggest methods for reducing loads.

The modern modelling approaches discussed here were not available in the 1980s. These models have been applied with great success in recent years to land and water interaction problems in New Zealand, and their application in real management situations is relatively straightforward. Their development has reflected a change in the management imperative for lakes and a greater understanding of how sediment and nutrient inputs affect lake biota.

Effective lake management requires providing managers with the tools to think ahead and anticipate trends. Past management approaches have involved monitoring water quality and then managing the system if undesirable changes were observed. This kind of water-quality monitoring will always be an essential part of lake management, to determine whether management interventions are having their desired effects and whether mitigation or restoration action is needed. For those lakes that have already become severely nutrient-enriched, monitoring in-lake processes is essential for following the changes that occur as restoration progresses.

However, lakes and their catchments are known to have long residence times (especially if spring-fed), which means that the water in them at any one time will often reflect what was taking place a long time – up to decades – previously. Because of this time lag, lake management should, in addition to regular monitoring:

- use modelling techniques to predict the effects of land-use changes on inputs of suspended sediments and nutrients, so that decisions can be made on what changes are required

- understand, and therefore be able to predict, as many environmental consequences as possible within lakes in response to particular nutrient and sediment loads
- manage catchments, and especially the land–water interface (i.e. major input sites, both surface and groundwater) to protect the lake against long-term changes.

In this document we first review the current understanding of the sources of water and sediment (section 2) and nitrogen and phosphorus (section 3) for lakes in New Zealand. This is followed by an overview of the modelling techniques now available (section 4), the protocols for monitoring and measuring loads that are needed for modelling (section 5), and methods for reducing loads to lakes (section 6).

Understanding of the biotic responses to changes in particulate and nutrient loads has also advanced considerably in recent years. Some of the effects that are likely to occur in response to increases in catchment loads are:

- the loss of aquatic macrophyte communities due to reduced water clarity, and the favouring of phytoplankton over macrophytes
- changes in phytoplankton community composition from picophytoplankton to larger netplankton and colonial forms
- alterations in food web structure, often involving reductions in populations of zooplankton and increases in numbers of coarse fish
- the loss of larger predatory fish that use vision for feeding (in New Zealand trout are particularly at risk).

These problems have often occurred in New Zealand, and are discussed in this document in a series of case studies (see appendices 2 to 5). These illustrate how lake water quality and ecosystem values can deteriorate in response to changes in the catchment, and the management actions that have been attempted to redress the changes. Not all of these actions have been successful, so some of the case studies also provide examples of what not to do, or of how management may have been assisted by a better understanding of the catchment–lake interactions prior to intervention. Here is a brief summary of the case studies found at the back of this document.

- Lake Rotorua has a long history of well-publicised eutrophication problems and consequent attempts to improve water quality. The case study illustrates how recent modelling approaches have provided a better understanding of nutrient inputs, the effectiveness of amelioration actions such as riparian strips, and the effects of changing land use in the catchment.
- Lake Taupo is an example where protection of high lake water quality has been an important issue for a long time, and where recent application of models has been used to anticipate future trends and delayed responses to changes in catchment land use.
- Lake Alexandrina provides an example of how rapidly phytoplankton blooms can develop in response to increased phosphorus inputs into small, shallow lakes, and the difficulty of controlling high phosphorus concentrations once they have developed.

- The reduction in light availability in lakes when suspended sediment loads increase can cause sudden collapses in populations of aquatic macrophytes in the littoral zone, leading to nutrient releases and algal blooms. Lake Waahi provided a dramatic example in the 1980s (Tanner et al. 1993), and rapid switches between macrophyte-dominated and phytoplankton-dominated states triggered by changes in sediment and nutrient loading have been documented in many other lakes (e.g. Lakes Omapere and Tutira). Even moderate changes in sediment loads can affect macrophyte populations, and the mechanisms of these light–macrophyte interactions are illustrated in the Lake Coleridge case study. In this case, changes in the suspended sediment load are related to changes in the depth to which macrophytes can grow, providing information on likely loss and recovery times for anthropogenic changes in water clarity.
- Extensive changes in land use have occurred in the catchment of Lake Forsyth since the earliest days of European settlement, and this lake provides an example of how complex interactions between nutrient availability and turbidity serve to control cyanobacterial blooms.

Water and sediment sources

Water

Water movement is responsible for transporting sediment and nutrients from the catchment to the lake. The flow rate into the lake affects the flushing of the lake and the associated residence time of water in the lake (how long it stays there), which in turn affects how the lake responds to a given nutrient or sediment load. So it is worth looking at some relevant aspects of hydrology (how water behaves) before tackling the sources of sediment (section 2) and nutrients (section 4).[1]

Rainfall is the key driver of hydrology in New Zealand. Higher rainfall generally leads to more leaching of nutrients, more overland flow, and a greater potential for transporting sediment into streams[2]. Extreme rain events are also important for sediment movement.[3]

A rough estimate of the mean annual flow in a stream can be obtained from Figure 1. This map was primarily based on data from stream-gauging sites over a longer period. The map was developed for streams with a catchment greater than 50 km^2, and flows obtained from the map can be expected to be less reliable for smaller streams. Flood flows can be estimated from maps in Mosley (Mosley 1992 Figures 6.6 and 6.7). Alternatively, the average flow in streams can be estimated from the rainfall on the catchment minus the evapotranspiration.[4] Flows can also be estimated from more detailed hydrological modelling.[5]

Water is delivered into streams via two routes: surface and subsurface flow. When soils become saturated, water cannot infiltrate and so runs off as surface flow. Drainage of infiltrated groundwater provides streamflow after rainfall events.

[1] The book *Waters of New Zealand* (Mosley 1992) provides an introduction to hydrology in New Zealand, and more detailed information can be found in the *Journal of Hydrology (NZ)*.

[2] A map of annual average rainfall is contained in Mosley (1992, p.66).

[3] Maps of extreme rainfalls are given in Tomlinson (1984). Information on rainfall can be obtained from local rainfall records, NIWA's climate database (CLIDB, available online), and the HIRDS database (NIWA 1995).

[4] See Hoare and Spigel 1987 for more details.

[5] See Ibbitt and McKerchar (1992) for an overview of modelling in New Zealand.

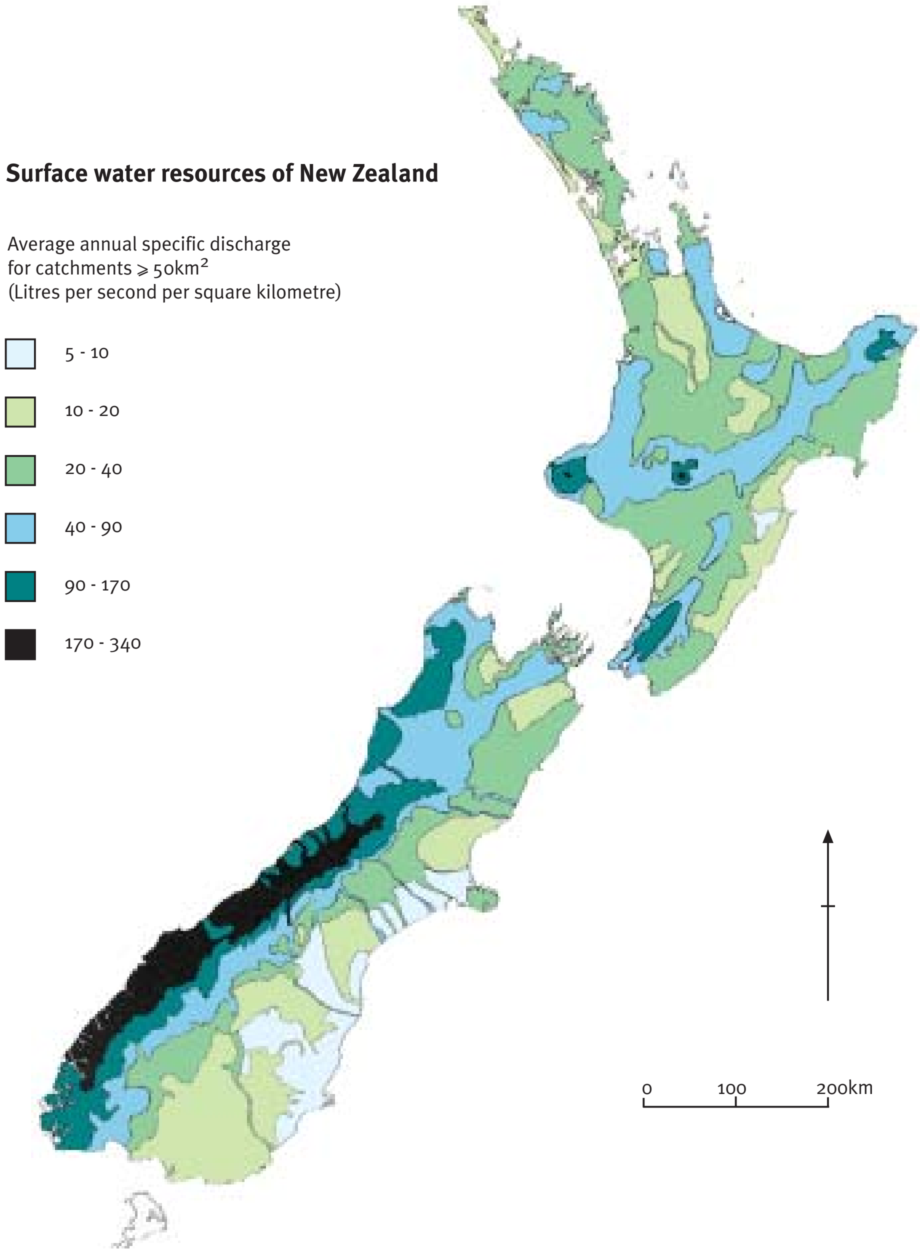


Figure 1: Map of water yields around New Zealand. The volume of runoff at a point in a stream is the yield at that point times the area of the catchment.

Source: Toebes 1972.

In many catchments in New Zealand the soils are sufficiently permeable for the rainfall to infiltrate during most rainfall events. In such catchments, surface runoff can be generated by 'saturation excess' (Pearce 1973). This occurs when water that has infiltrated through the soil moves as subsurface flow and re-emerges on hillsides near streams or in valley bottoms, giving rise to permanently, seasonally or temporarily saturated areas. When rain falls on the saturated areas, surface runoff is generated.

These saturated areas, which are often referred to as 'variable source areas' or 'critical source areas', have been observed in some catchments in New Zealand (Pearce 1973; McColl et al. 1985; Cooke and Dons 1988). This is important for nutrient generation, as surface runoff can occur even in catchments with permeable soils, and surface runoff carries nutrients from the nutrient-enriched surface soil. As the saturated areas usually occur near streams, exclusion of stock and vehicles from riparian areas is an important land management practice for reducing nutrient and sediment loads.

In larger storms or with less permeable soils (such as pasture areas), 'infiltration excess' surface runoff can occur (McColl et al. 1985; Pearce 1973). This happens when the soil is not sufficiently permeable for the rain to soak in. This results in surface runoff throughout the catchment, even if on a somewhat patchy basis. The resulting runoff and overland flow results in sediment and nutrients being transported into the stream, provided the flow does not reinfiltrate into more permeable soils on the way (Elliott and Ibbitt 2000).

Another storm runoff generation mechanism that occurs in many New Zealand catchments is rapid subsurface flow, which results in storm flow in the stream without surface runoff (Cooke and Dons 1988; McColl et al. 1985). This flow is caused by rapid movement of water directly to the stream through preferential subsurface flow paths (such as cracks in the ground) or the displacement of water already in the ground.

Some streams, such as those in the pumice country in the central North Island or near the coast in Central Canterbury, are dominated by baseflow, which results from groundwater re-emerging through springs, seeps (distributed along banks), or through the stream or lake bed. The groundwater may be from deep aquifers recharged some distance from the point of re-emergence, or from shallow groundwater systems (often perched over low-permeability material) with relatively local recharge. The groundwater catchment for some lakes can be significantly larger than the surface water catchment, giving apparently high inflows compared to the size of the surface catchment (Gibbs 1987).

Some lakes, such as the dune lakes in Northland, are fed predominantly by groundwater entering the lake directly through the lake bed (along with rain on the lake surface), rather than through streams (e.g. Kokich 1991). Unfortunately no existing summaries or classifications of New Zealand lakes show whether they are predominantly groundwater-fed or stream-fed: existing classifications (e.g. Lowe and Greene 1987) are based more on geological formation processes or biota rather than hydrology. One indication of a groundwater-dominated lake is if the lake level varies with the local groundwater level. Lakes with no surface outlet also indicate strong interactions with the groundwater.

The net input from groundwater (groundwater inflows minus groundwater outflows) can be estimated from a water budget based on measuring the lake's volume, stream inflows and outflows, and rain on the lake surface, combined with estimates of evaporation from the surface.[6] This can be used as a lower-limit estimate of the groundwater inflows. The groundwater gradients in the vicinity of the lake can be used to estimate the groundwater inflows using groundwater hydrology principles (Gibbs 1987). Groundwater inflows can also be measured directly (although such measurements are fairly imprecise), and such methods

[6] As outlined in the previous *Lake Managers' Handbook* (Hoare and Spigel 1987).

have been applied in New Zealand (Lock and John 1978; Gibbs 1987; Wells et al. 1988).

Hydrology is not only affected by rainfall and geology, but also by land use. Forests reduce transpiration and hence reduce water yield (Fahey 1994). Urbanisation increases surface stormflow runoff and can also reduce baseflow (McConchie 1992; Shaver 2000). Establishing pasture often leads to soil compaction and a reduction of infiltration capacity of the soil (e.g. McColl and Gibson 1979; Selby 1972), which results in more-frequent generation of infiltration excess surface runoff and increased storm flows. Treading damage from intensive grazing can also result in a short-term reduction in the infiltration capacity of the soil (Nguyen et al. 1998; Greenwood 1999). However, pasture improvement can help to reduce runoff (Yates 1971; Lambert et al. 1985).

Sediment

Rainfall and geological factors have a strong effect on the amount of sediment found in streams (Hick and Griffiths 1992). Regression equations for sediment yield (mass per area per unit time), as a function of precipitation and location, can be used for a first estimate of sediment load (mass per unit time) entering a lake (see Figure 2). A new set of maps of sediment yield, which use more recent data and take more detailed geology into account, are currently being prepared by NIWA. Higher rainfall increases sediment yield through increased rock weathering, mass movement and gullying, soil detachment and surface runoff, and increased sediment transport capacity in the stream. The variation between regions reflects the influence of rock type, geological uplift, slope, catchment relief and climate.[7]

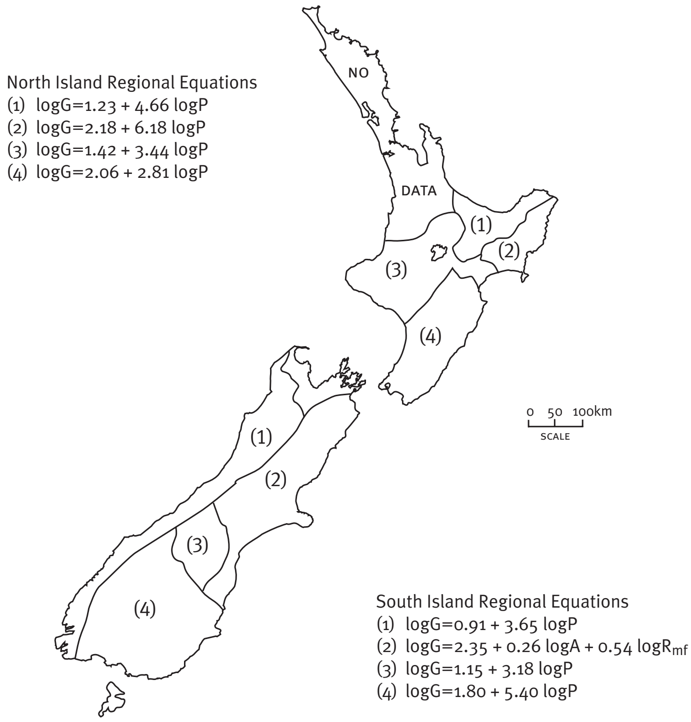


Figure 2: Sediment yields in New Zealand

Source: Based on the work of Hicks and Griffiths (1992), as given in Hicks (1992).

[7] Overviews of erosion types in New Zealand are contained in Hicks (1995), Hunter-Smith (2000, predominately North Island) and DSIR (1948, for a historical perspective).

Sediment yield is also affected by land use. Urban earthworks cause a large increase in sediment yields.[8] As urban areas mature following development, yields eventually reduce to low levels, although there can be a considerable adjustment period before the sediments are flushed out of the channels and the stream shape adjusts to increased flows.

Forest harvesting usually increases sediment yields, but established pine forests generally have lower sediment yields than other land uses (Dons 1987; Hicks 1990). Harvesting increases sediment yields through general soil disturbance due to:

- exposing soil at roads and landings
- pushing soil into streams
- creating unstable slopes near streams associated with road and landing construction
- general vehicle traffic and log hauling on slopes
- increased likelihood of slope failure once roots decay
- increased stream flows, with associated bank erosion (Wallis and McMahon 1994).

In a rotation-length study, Hicks and Harmsworth (1989) reported that 70% of the total sediment yield occurred during harvesting and re-establishment.

Land cultivation has the potential to produce high sediment yields by exposing bare soil. Control of sediment from such areas forms the main thrust of soil conservation efforts in the USA, but cropping is only a small part of land use in New Zealand (Williamson and Hoare 1987).

Pasture land use generally has a two-to-tenfold greater sediment yield than pine or bush catchments (Fahey 2000; Hicks 1990; Quinn et al. in prep). The increased sediment yield is associated with increased surface runoff and slope destabilisation (de Rose 1998; Quin et al in prep; Selby 1972) and treading damage from intensive grazing (Nguyen et al 1998). Grazing by heavy animals can also destabilise streambanks (Stassar and Kemperman 1997; Williamson et al 1992), especially for smaller streams. Ironically, good pasture protects and stabilises the soil surface, and can also stabilise stream banks of small streams in hilly areas (Davies-Colley 1997), but this apparently does not offset the increase in sediment due to other pasture-related factors.

The Universal Soil Loss Equation (USLE), or its more recent derivative (RUSLE), can be used to derive a preliminary estimate of sediment loss at the hill-slope scale (Renard et al 1997; Wischmeier and Smith 1978). These equations express the effect on sediment loss of rainfall intensity and energy, ground slope, vegetation cover, soil erodibility and slope length. They were developed primarily for application to cropping areas in the USA, but many of the principles are applicable to New Zealand. However, they do not account for landslips, in-stream erosion and deposition, riparian buffers, or deposition in flat areas near streams. Such effects can be important. For example, at a Pukekohe vegetable farm, Basher et al (1997) measured erosion rates of 56.8 t/ha/year at the plot scale, but only 0.49 t/ha/year at the catchment outlet.

[8] Yields from urban earthworks are given in Williamson (1993) and Hicks (1994).

SECTION 3

Nitrogen and phosphorus sources

This section looks at nitrogen and phosphorus loads deriving from point sources and various land-use types. The sources of nutrients in a catchment are summarised schematically in Figure 3.[9] We will also look at processes affecting nutrient loads to give managers a better understanding of the sources and pathways of nutrients. First, however, we will review the different forms of nutrients.

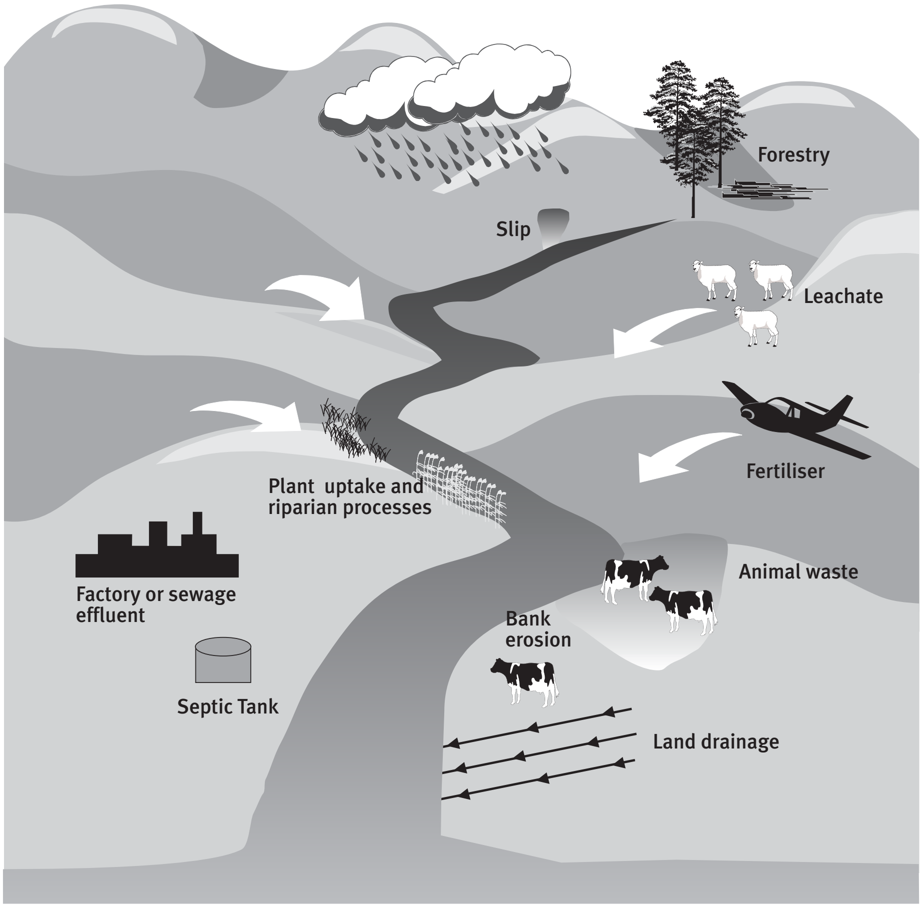


Figure 3: **Sediment and nutrient sources**

[9] Nutrient sources have been reviewed previously in Chapters 15 and 17 of the *Lake Managers' Handbook* (Vant 1987), and by Cooke (1979), McColl and Hughes (1981) and Rutherford et al. (1987). Point sources have been reviewed by Hickey and Rutherford (1986), while agricultural sources have been reviewed by Wilcock (1986) and McColl (1982).

Nutrients can take a number of forms, which vary in their mobility and bioavailability (see Table 1). Particulate inorganic and organic matter and their associated nutrients entering lakes may settle to the lake bed and only release bioavailable fractions under certain conditions, while many forms of particulate and dissolved organic matter are highly resistant to decomposition and only release bioavailable nutrients slowly. Indeed, developing methods to assess the bioavailable fraction of nutrients in New Zealand waters is an urgent priority. The topics that need most investigation are:

- bacterial rates of organic matter regeneration in New Zealand waters
- the role of organic chelation, iron and solubility equilibria with suspended particles in regulating inorganic nitrogen and phosphorus availability.

Table 1: Main forms of nutrients*

Nitrate (NO^3)	Oxidised, inorganic, mobile, readily bioavailable.
Ammonia (NH_3, NH^+_4)	Reduced, inorganic, semi-mobile, readily bioavailable.
DIN (dissolved inorganic nitrogen)	Nitrate, nitrite, and ammonia nitrogen, readily bioavailable.
DON (dissolved organic nitrogen)	Organic nitrogen that passes through a filter; includes cell fragments or organic macromolecules from the decay of organic material; semi-mobile, easily mineralised to inorganic forms.
Particulate organic nitrogen	Organic nitrogen that is trapped on a filter; plant fragments and detritus (fresh or decayed), organic coatings on particles; mobile with particles; availability for mineralisation is variable.
Particulate nitrogen	Total nitrogen retained on a filter; most particulate nitrogen is organic.
TKN (total Kjeldahl nitrogen)	Reduced organic nitrogen plus ammonia nitrogen, obtained from a Kjeldahl digestion; bioavailability predominantly as for particulate organic nitrogen.
TN (total nitrogen)	Total nitrogen in the sample; should be close to the sum of TKN plus nitrate and nitrite.
DRP (dissolved reactive phosphorus)	Dissolved orthophosphates and polyphosphates; inorganic, semi-mobile, readily bioavailable.
DOP (dissolved organic phosphorus)	Dissolved phosphorus in organic form; includes cell fragments or organic macromolecules from the decay of organic material; semi-mobile, largely bioavaliable.
Particulate P	Phosphorus retained on a filter; can be broken down into organic and mineral fractions; bioavailability of the inorganic fraction depends on the mineral form, chemical surroundings, and whether the phosphorus is on the surface or within mineral particles.
TP (total phosphorus)	All phosphorus, determined from a digestion; should be close to particulate P plus DRP plus DOP.

** Concentrations are usually expressed as mass of nutrient (N or P) per volume of water.*

Given the caveat that improved understanding of bioavailability needs urgent attention, and will certainly improve the accuracy of predictions in the future, total nitrogen and total phosphorus still serve as suitable bases for managing nutrient inputs for most situations, for the following reasons:

- Many organic forms that are not immediately bioavailable do eventually become available to phytoplankton after biological processing within the water column or sediments. This is why most assessments of eutrophication effects are based on total rather than inorganic forms (Horne and Goldman 1994).
- Totals will always be easier to predict than constituent forms. Dissolved inorganic forms are highly unstable and liable to be transformed in riparian areas, wetlands and streams. Also, more data are available on total nutrients, as the different forms are often not measured adequately in studies of nutrient sources.
- Using total nutrients is conservative, in that the values will be higher than for the readily bioavailable fraction (i.e. their use is a precautionary approach).

Fluxes of organic matter into lakes are an important supply of energy for secondary production as well as for nutrient release, and can be the major nutrient store in lakes (Wetzel 1995; Thomas 1997). The rate and type of organic carbon entering streams has a significant effect on microbial community respiration (Young and Huryn 1999), and hence nutrient regeneration. This is also likely to be the case for groundwater directly entering lakes. Forested catchments release significantly more organic matter into New Zealand streams than do pasture sites, supporting higher rates of community respiration (Young and Huryn 1999). Microbial respiration and exoenzyme activity also differ between land uses (Findlay et al. 1997). The accumulation and metabolism of organic matter within lakes is critical to understanding nutrient relations, trophic interactions, and trophic status, especially in shallow lakes (Wetzel 1990), and needs further study to be properly addressed in New Zealand.

Nutrient loads from point sources

The main point sources of nutrients to New Zealand streams and lakes reported by Hickey and Rutherford (1986) were sewage, meatworks, cowsheds and piggeries (all about equal inputs), while pulp and paper effluent and dairy factories made a smaller contribution.

Nationally, point source inputs of nutrients to lakes and rivers are considerably less significant than non-point sources. The total phosphorus load from non-point sources is roughly 50 tonnes/day (based on typical yields, see section 3), which is considerably greater than the 6.1 tonnes/day from point sources entering inland waterways (Hickey and Rutherford 1986). The total nitrogen load from non-point sources is about 400 tonnes/day, compared with Hickey and Rutherford's point-source estimate of 28.6 tonnes/day.

Typical loads from point sources are given in Table 2. These can be used to make a preliminary estimate of loads from specific point sources. Discharge monitoring can be used for a more refined estimate, and such monitoring is often performed as part of the discharge consent process. Factory waste tends to vary greatly depending on the type of process and degree of waste treatment, so estimates for a particular discharge should be obtained from monitoring records or specialist knowledge of the industry.

Table 2: Typical nutrient loads for some point sources

Point source type	Degree of treatment	Total N load	Total P load
Domestic sewage[1]	Untreated sewage and septic tank effluent[2]	4.2kg/person/year	1.5kg/person/year
	Conventional secondary treatment	5-40% removed	5-40% removed
	Enhanced nutrient removal	50-95% removed	70-85% removed
Dairy shed effluent	Untreated[3]	5.4kg/cow/year	0.66kg/cow/year
	Treated (dual pond)[4]	75% removed	60% removed
Piggeries[5]	Untreated	8kg/pig/year	2.7kg/pig/year
	Treated (anaerobic lagoon)	60% removed	40% removed

Notes: *1. From Potts and Ellwood (2000).*
2. Septic tanks effluent close to raw effluent (Gibbs 1979).
3. Environment Waikato and Selvarajah (1996b).
4. Based on concentration reduction in Selvarajah(1996a). Assumes pond inflow equals outflow.
5. Vanderholm (1984).

There is an increasing trend in New Zealand for point-source wastes to be applied to land. For example, effluent from 70% of dairy sheds in the Waikato is applied to land, which is very different from the situation 10 years ago. Obviously application to land reduces the load to the lake or stream compared with a direct discharge, but the degree of removal is variable and depends on the application rate, land management, climate, soil and proximity to the stream or lake. A very conservative estimate would be to assume that *all* the nutrients reach the stream or lake. If the resulting load from this estimate is a concern, then refined estimates based on monitoring or models of the soil can be generated.[10] There are several studies under way on the effects of dairy effluent irrigation, and some studies have already been completed (e.g. Greenwood 1999; Selvarajah 1996b).

For discharges directly to the lake, the concentration in the mixing zone close to the discharge can be affected by the concentration of the discharge (not just the load) because there is little dilution. Further away from the discharge, where there is more mixing with lake water, only the load from the discharge is important.

Export coefficients and concentrations for various land uses

Export coefficients are useful for obtaining preliminary estimates of loads entering lakes (see section 4), and can also be used as a check on the results of modelling studies. An export coefficient of a nutrient is the mass of that nutrient leaving the catchment, per unit area of catchment, per unit time (typical units are kg/ha/year). Often the terms 'yield' or 'specific yield' are used instead, but we will use the term 'export coefficient' to avoid confusion, and to make it clear that we are referring to the mass exported from the catchment after any attenuation within the catchment.

[10] Further information on land application of sewage and estimation of the resulting loads is contained in Whitehouse et al. (2000).

Export coefficients for various land uses have been summarised previously by Wilcock (1986) and Cooke (1979). Updated export coefficients are shown in Figure 4, and are summarised in Table 3. These were obtained from collating the results from catchment studies in New Zealand, many of which came after the earlier reviews. Most of the study sites were in the North Island.

Table 3: Export coefficients (kg/ha/year) for a variety of land uses

	Dairy	Hill	Low-intensity pasture	Native	Exotic
TN					
Number of studies	4	6	5	8	4
Mean	25.0	9.0	5.2	3.0	2.8
Median	26.9	8.1	5.1	3.1	1.1
Min	10.7	2.6	2.8	0.6	0.6
Max	35.3	19.5	8.8	5.8	8.5
TP					
Number of studies	4	5	6	10	5
Mean	1.00	1.98	0.46	0.39	0.35
Median	1.05	1.70	0.43	0.32	0.17
Min	0.60	0.60	0.30	0.12	0.07
Max	1.30	3.40	0.60	0.80	1.20

**Notes: TN = total nitrogen; TP = total phosphorus.*

There are some clear differences in export coefficients between different land uses. For nitrogen, dairying has the highest export coefficient (out of the land uses in the table), followed by hill pasture (which generally carries sheep and beef). Export coefficients from low-intensity pastoral grazing, native bush or scrub, and pine plantations are lower. The ranges of export coefficients for the different land uses overlap, which causes some difficulties when comparing the different land uses. However, the trend identified from averages from many catchments is generally supported by results from studies with different land uses in adjacent catchments (see Table 4). Environment Waikato (1999) have also demonstrated that total nitrogen export coefficients for streams in the Waikato are strongly correlated with the number of dairy cows per hectare.

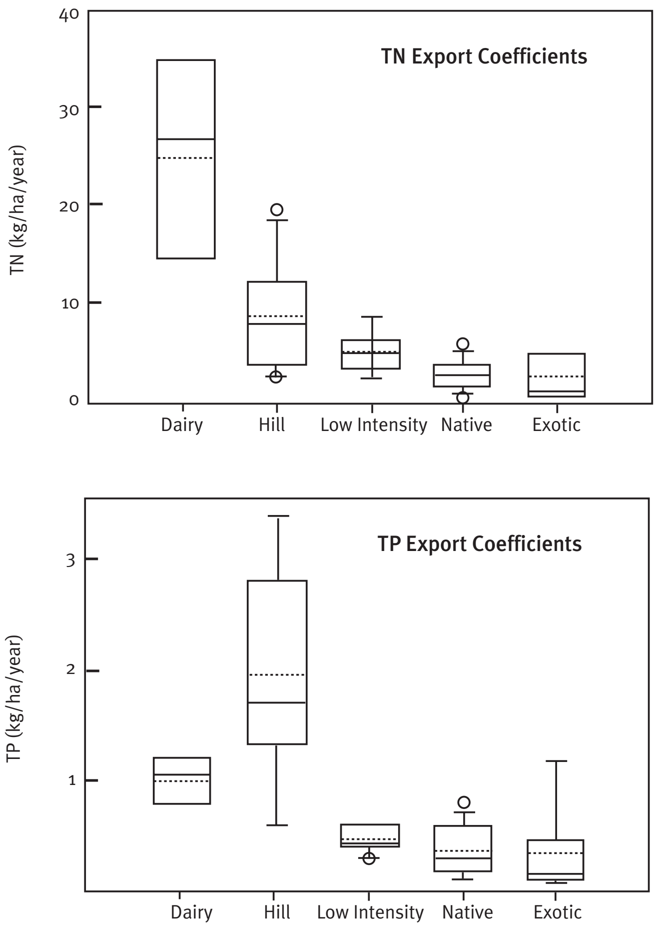


Figure 4: Nutrient export coefficients for various land uses

Notes: The central solid line is the median value, the dotted line is the mean value, the ends of the box are at the 25th and 75th percentiles, whiskers are at the 10th and 90th percentiles (where appropriate), and circles are outliers. TN = total nitrogen; TP = total phosphorus.

Table 4: Relative nutrient exports from adjacent catchments with different land uses

STUDY	RELATIVE EXPORT COEFFICIENT FOR DIFFERENT LAND USES*	
Whatawhata, hill country to west of Hamilton. Quinn et al (in prep).	N: Hill pasture → native P: Hill pasture → native	(11: 1) (5.6: 1)
Purukohukohu, central volcanic plateau. Cooper and Thomsen (1988).	N: Pasture → native → pine P: Pasture → native → pine	(9: 2.8: 1) (17: 1.3: 1)
Taita, Hutt Valley, hill country. (McColl et al. 1977)	P: Hill pasture → native → pine	(4.1: 2.8: 1)
Ballantrae, hill country NE of Palmerston North. Lambert et al. (1985).	N: Cattle pasture → sheep pasture P: Cattle pasture → sheep pasture	(1.4: 1) (2: 1)

** The value in parentheses is the ratio of yield compared to the lowest.*
Notes: N = nitrogen; P = phosphorus.

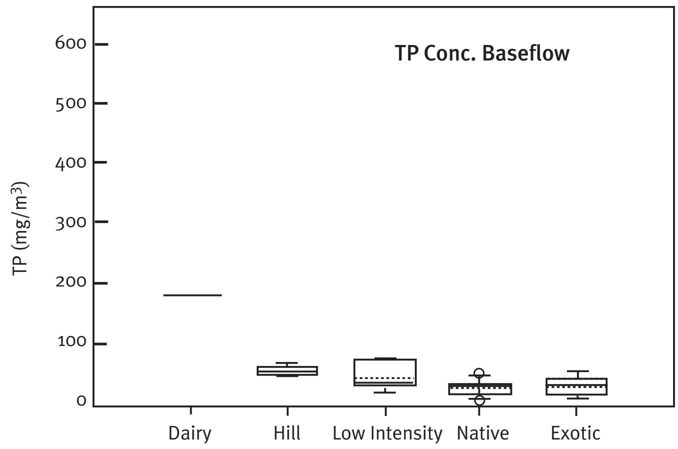
For phosphorus, hill pasture has the highest export coefficient, followed by dairying, low-intensity pastoral grazing, bush and pine. Greater erosion probably accounts for the higher phosphorus export from hill pasture compared to the more intensive dairy land use.

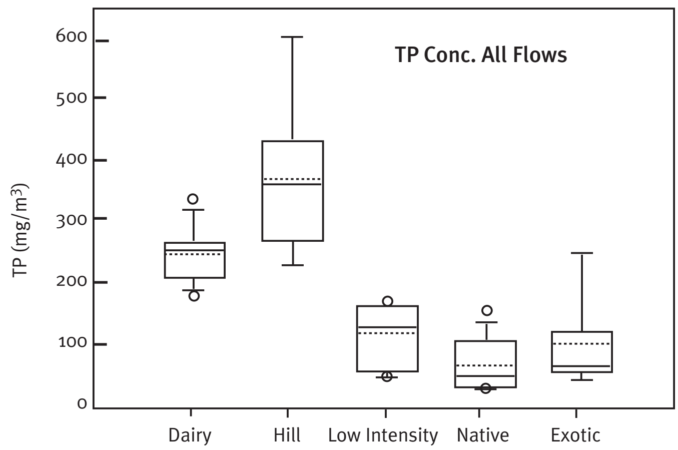
Representative nutrient export coefficients for urban areas in New Zealand are given by Williamson (1993). For total nitrogen the low, average and high export coefficients were 2.5, 8, and 11 kg/ha/year, while for total phosphorus the values were 0.4, 0.8 and 1.6 kg/ha/year respectively. For total nitrogen the values are comparable to those for hill pasture, while for total phosphorus the values lie within the range for the pasture land uses.

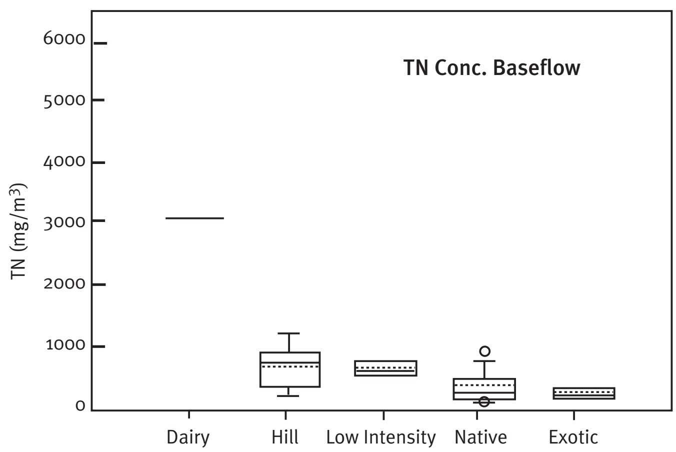
There is very little information on nutrient export from cropping land in New Zealand. An unpublished study by Williams and Tregurtha of leaching under a Pukekohe spinach crop (referred to in Lincoln Environmental 1997) found nitrogen leaching of 240 kg/ha/annum – higher than all other land uses. While such market gardening represents only a small fraction of the land use in New Zealand, it could have an important effect in some catchments.

The concentration of nutrients in streams for various land uses is also interesting, because they can be used to estimate the load entering the lake (see section 6). Also, near a stream mouth there may be little dilution with the lake water, so that the concentration in the lake near the stream reflects the concentration of the inflow rather than the concentration in the bulk of the lake. Finally, according to simple empirical models, the concentration of phosphorus in the lake can be predicted from the concentration of the inflows and the residence time of the water in the lake (Vant, 1987; Chapra 1997).

Figure 5 summarises concentrations in streams from various land uses, based on a review of available data from studies in over 30 catchments in New Zealand. The concentrations refer to temporally-averaged values, not flow-weighted values. As expected, the concentrations in Figure 5 follow the same trend as for export coefficients. An exception is the concentration of phosphorus in baseflow, which is highest for dairying, whereas the export coefficient is highest for hill pasture. This reflects the influence of high concentrations in storm events for hill pasture.


TP Conc. Baseflow
TP (mg/m3)
600
500
400
300
200
100
0
Dairy
Hill
Low Intensity
Native
Exotic


TP Conc. All Flows
TP (mg/m3)
600
500
400
300
200
100
0
Dairy
Hill
Low Intensity
Native
Exotic


TN Conc. Baseflow
TN (mg/m3)
6000
5000
4000
3000
2000
1000
0
Dairy
Hill
Low Intensity
Native
Exotic

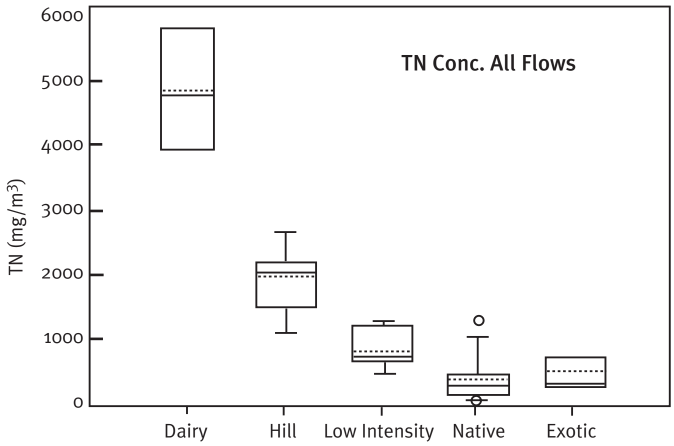


Figure 5: Concentration data for various land uses, summarised from a variety of data sources (see Appendix 1)

Notes: the central solid line is the median value, the dotted line is the mean value, the ends of the box are at the 25th and 75th percentiles, whiskers are at the 10th and 90th percentiles (where appropriate), and circles are outliers.

The nutrient export coefficients and concentrations for a given land use vary depending on a number of factors, including climate, soil, land management practices and uptake in wetlands, riparian areas and streams. Selecting a value to use in a particular study should recognise these relationships, and an appreciation of these factors will help the lake manager to refine export coefficient estimates. Understanding the factors will help the lake manager to identify and select measures to control the input of nutrients to lakes. Factors affecting the loss of nutrients in New Zealand have been reviewed previously (McColl 1983; Rutherford et al 1987), and are summarised in Table 5.

Table 5: Summary of factors affecting nutrient yield from non-point sources (excluding land-use effect)

Factor	Examples and references
Fertiliser loss	• Fertiliser lost due to application onto or near streams during aerial topdressing (Cooke 1988a; McColl 1983; McColl et al., 1975; Rutherford, et al. 1987). Typically 0.5-2% of applied fertiliser is lost for pasture; less for forest. Increased DRP in baseflow shortly after topdressing (< 1 day) (Fish 1969; Sharpley and Syers 1979a). • Increased nutrient loss in surface runoff and tile drains for 1-4 weeks after fertiliser application (McColl and Gibson 1979; Nguyen et al. 1998; Sharpley and Syers 1976; Turner et al. 1979). Typically 1-5% of applied fertiliser is lost.
Grazing	• Nutrient concentrations increase several-fold and loads increase for a few weeks after grazing on pasture with or without tile drainage (McColl and Gibson 1979; Sharpley and Syers 1976; Turner et al. 1979). • Treading damage increases runoff, sediment loss, and nutrient loss from hill pasture (Nguyen et al. 1998). Treading reduces infiltration capacities on dairy land in Southland (Greenwood 1999). • Grazing increases bank erosion for small streams (Stassar and Kemperman 1997; Williamson et al. 1992). • Wastes deposited in streams and riparian zones by cattle potentially increase nutrient load (Rutherford et al. 1987).
Nutrient enrichment of soils	• Build-up of phosphorus in surface soil increases nutrient loss from pasture (Sharpley et al. 1977; Wheeler et al. 2000). • Soil organic P and TN concentrations increase after establishment of pasture (Jackman 1964).
Irrigation	• Irrigation increases nitrate leaching (Fraser et al. 1994; Quin 1979; Turner et al. 1979).
Artificial drainage	• Drainage decreases loss of P in surface runoff (Sharpley and Syers 1976). • Riparian or wetland denitrification areas are bypassed by field drainage.
Forest harvesting	• Increased concentrations of nitrate in leachate after felling (Dyck and Cooke 1981), but weeds can reduce leaching (Parfitt et al. 2002). • Increased sediment loss (see text on sediment sources).
Stream erosion source of P	• Bank erosion and sediment re-suspension can account for a large fraction of phosphorus export (Sharpley and Syers 1979b).
Nutrients generated near streams	• Phosphorus concentration increased with fraction of catchment saturated (Cooke and Dons 1988). • Surface runoff may not reach streams if it is generated >30m from the stream (McColl and Gibson 1979).
Increased nutrient export lags change in land use	• Soil nutrients and organic matter take years to adjust after establishing pasture. Half-life of change ranged between two and 40 years (Jackman 1964). • Nitrate in spring flows may take decades to respond to changes in land use due to delays in groundwater (e.g. Williamson et al. 1996).

[11] Loss process in New Zealand streams have been reviewed by Rutherford et al. (1987).

Atmospheric sources	• Mean TN deposition 3.7 kg/ha/year (range 2.8-5); TP is 0.38 kg/ha/year (Rutherford et al. 1987).
Natural DRP sources in baseflow	• High DRP from dissolution of apatite in springs in central North Island (Timperley 1983; White and Downes 1977).
Removal and processing in streams, wetlands and filter strips	• Refer to main text.

Notes: DRP = dissolved reactive phosphorus ; N = nitrogen; P = phosphorus; TN = total nitrogen; TP = total phosphorus.

In-stream loss and transformation of nutrients

Several studies in New Zealand have shown that nutrients can be removed from streams. Such loss is termed in-stream attenuation, decay or loss. The loss occurs as a result of uptake by macrophytes, periphyton, sewage fungus, bacteria, sorption to sediments, and settling of particulate material.[11]

Such processes are important for several reasons. They reduce the total load exported from a catchment, and manipulation of the processes controlling in-stream losses is a potential way to reduce lake loadings. For example, drainage ditches for in-stream nutrient removal are being trialled as a new control method for nutrients in drainage from dairy farms, with encouraging initial results (Nguygen 2001). Also, in-stream processes can modify the form of nutrients, transferring nutrients from dissolved and readily available forms to less-directly available organic or particulate forms. It is important to consider such processes when interpreting measured loads of nutrients, especially dissolved nutrients.

In-stream processing is one cause of the variability of nutrient export coefficients within a particular land use. This relates not only to different degrees of processing in different streams (which may be affected by the riparian vegetation and land use), but also the effect of the size of the catchment. In-stream processing should cause the catchment export coefficient to be less for larger catchments (all other factors being equal), because in a larger catchment there is likely to be a greater stream length in which in-stream processing can occur. This makes it difficult to transfer an export coefficient measured for one catchment to one of a different size. Indeed, this is one of the main limitations of the export coefficient approach to load estimation.

The degree of shading of the stream channel and the type of stream vegetation also affect the stream attenuation rates. Cooper et al. (1987) noted that establishing forest at Purukohukohu decreased the in-stream removal rate due to shading-out of in-stream vegetation. In the Whangamata Stream, nitrate uptake rates increased following fencing of the riparian areas as dense in-stream vegetation established. Later, when larger, woody vegetation established, the nitrate removal dropped again (Downes et al. 1997), and now nitrate removal is negligible (Howard-Williams and Pickmere 1999).

Stream attenuation is particularly noticeable for dissolved nutrients such as nitrate and dissolved reactive phosphorus, which are more readily taken up by plants than organic or particulate nutrients. The uptake is strongest in summer, when stream plants are growing fastest and when temperatures are higher. For example, stream uptake is responsible for strong seasonal variations of nitrate in the Whangamata Stream, which enters Lake Taupo (Howard-Williams et al. 1986).

An important nutrient removal process is denitrification (conversion of nitrogen-oxide gas to nitrogen gas). Denitrification results in permanent removal of nitrogen, and accounted for 1–25% of nitrate removal in some studies (Cooke 1988b; Cooper 1990; Howard-Williams and Downes 1984).

Some of the material removed from a stream during baseflow may be remobilised during floods. For example, Cooke (1988a) found that a pasture stream acted as a sink for phosphorus at baseflow and in low-medium intensity floods, but was a source of phosphorus during higher-intensity floods.

In a modelling study for Waikato (Alexander et al. 2002; McBride et al. 2000), there was evidence of net attenuation over five years for both nitrogen and phosphorus, which indicates that remobilisation in moderate storms does not 'undo' all the in-stream loss. In fact loss and remobilisation processes take place on a range of time scales (Table 6), and there are long-term sinks in the stream system.

Table 6: Time scales of nutrient removal and remobilisation processes in streams

Time scale	Response
Ongoing removal and frequent floods	Sorption to bed sediment, denitrification in organic-rich zones, settling to channel bed.
Seasonal	Uptake by plants, dieback of plants.
Annual	Sloughing of vegetation, suspension of organic detritus, mobilisation of bed material.
Decades	Long-term accretion of bed and bank material between large floods; scour of bed and banks in large floods and during wet or stormy climate periods; succession of stream bank vegetation to more shading, less ground cover, and root stabilisation of higher stream banks.
Centuries	Reservoir and lake infilling, channel migration.

In principle, the fraction of nutrients sources delivered to the catchment outlet could be characterised by a delivery ratio (load exported from the catchment divided by sources to the stream). However, there are no well-established means for estimating the delivery ratio for nutrients.

Recently, in a modelling study of nutrient sources and exports in the Waikato River basin (Alexander et al. 2002; McBride et al. 2000), the relative export coefficients for representative catchments of different sizes were calculated (Table 7). These could be used to make a preliminary estimate of the effect of catchment size on nutrient export.

Table 7: Relative export of nutrients for representative catchments of various sizes

Catchment size (km^2)	Relative nitrogen export	Relative phosphorus export
2	100%	100%
8	82%	73%
20	60%	45%

Source: based on results from a catchment model of the Waikato basin by Alexander et al (2002).

Rutherford et al. (1987) have presented a method for estimating attenuation during baseflow, and this method has been incorporated into some catchment models. The authors expressed the loss of nutrients in baseflow as an exponential decay with distance:

$C = C_0 \exp(-k\, x)$

where C_0 is the concentration at the source, C is the concentration a distance x downstream of the source, and k is a nutrient uptake length constant. The k values are higher for streams with smaller flows (more nutrients removed for a given distance), as the removal is dominated by benthic (stream-bed) or near-bed processes, and a small stream has a larger ratio of bed surface area to volume. Rutherford and Williamson prepared a plot of k versus Q based on several studies of attenuation in New Zealand. Cooper and Bottcher fitted an equation to this plot, giving $k = 0.0001Q^{-0.7}$, where Q is in m^3/s and k is in m^{-1}.

In-stream processing is seasonal. Vegetation transforms nitrogen and phosphorus most effectively in summer, whereas it is a net source to the lake during winter die-back. Likewise, microbial processes are temperature dependent and will tend to be more effective in warmer weather. At present there is insufficient information to make generalisations on how to moderate processes seasonally.

Nutrient removal in wetlands and buffer strips

There is ample evidence that wetlands and riparian areas trap nutrients (Collier et al. 1995; Quinn et al. 1993; Schnipper et al. 1991). Key points from studies in New Zealand (which are consistent with results internationally) are summarised below.

Riparian and headwater wetlands

Moist, organic-rich soils can remove nitrate by denitrification of lateral, sub-surface seepage in riparian areas (Schipper et al. 1991), or in headwater wetlands (e.g. Nguyen et al. 1999). Concentration reductions of 90% are common (e.g. Cooper, 1990), although the removal efficiency in riparian wetlands is worse at higher flows. For example, Nguyen et al. (1999) found that removal efficiency of a wetland decreased with increasing flow rate, and for some of the larger flows sediment was scoured. At high flows, water can be forced up to the surface to flow over the ground in channelised surface runoff (Rutherford et al. 2000; Rutherford et al. 1999).

The overall effect of the denitrification by organic-rich soils will obviously depend on how much of the inflowing water moves through these soils. For example, Cooper (1990) found that in organic seepage zones the nitrate concentration was reduced by over 90%, but that the overall removal of nitrate from groundwater over the length of the stream was only 53% because not all the inflow passed through the organic seepage zone. With deep soils near the stream, groundwater may enter the stream through the stream bed without contacting the riparian soils (Hill 1996).

The removal of total phosphorus and total nitrogen is lower than that of nitrate and dissolved reactive phosphorus. For example, in a small wetland Nguyen et al. (1999) found that removal of nitrate and dissolved reactive phosphorus was partly offset by the wetland acting as a source for ammonia, and there was ineffective trapping for particulate nutrients. In a headwater seepage zone, Cooke (1988b) found that although the seepage zone had low nitrate concentrations, this was accompanied by high organic nitrogen concentrations.

There is some concern that the removal of dissolved phosphorus cannot be sustained in riparian zones and wetlands, as over time the soils can become saturated with phosphorus (Cooper et al. 1995), and during large storms wetlands may scour away and release much of the material trapped in preceding years (as occurred in 1998 at Whatawhata, a hill-country farm in the Waikato).

Vegetative filter strips

Dense ground vegetation or tree litter can remove sediment and nutrients from surface runoff. These vegetated areas, situated in riparian strips or in small ephemeral channels, not only remove sediment by reducing the erosive power and sediment transport capacity of the flow, but can also enhance the infiltration of surface runoff (Cooper et al. 1995; Smith 1992). Enhanced infiltration increases the contact with the soil, allowing for more removal of fine sediment, sorption of phosphorus, and soil denitrification.

Constructed grass filter strips have been investigated intensively overseas, especially for removing sediment around the margins of cultivated fields (Dillaha et al. 1989). Such filter strips appear to be effective on New Zealand hill pasture. On a farm in the Waikato, for example, suspended sediments and nutrients were removed by vegetative strips along small ephemeral watercourses (Quinn et al. 1993; Smith 1989). In principle, such filtering would apply to vegetated riparian areas, and indeed riparian retirement in the Ngongotaha catchment has led to reductions in sediment and nutrient loads (Wiliamson et al. 1996). Removing ground cover can have the opposite effect. Smith (1992) found that riparian planting with pines increased the sediment and nutrient loss from a pasture catchment at Moutere, which was partly attributed to the loss of close ground cover near the stream.

Modelling sediment and nutrient sources

Overview of predictive methods

Catchment models are mathematical representations of how water, sediment and nutrients are generated and transported in a catchment. A range of models using a variety of approaches and degrees of complexity is available, and several models have been applied in New Zealand (see Table 8). The features of some of these models are discussed in the following sections.

There are several benefits of modelling.

- The discipline of preparing model inputs and analysing outputs forces data to be organised and assembled in a coherent framework. This process can also spur the collection of further relevant data.
- Model outputs can be used to break down the sources into their components, which in turn can suggest suitable control measures. For example, if point sources are the main input to the lake, the control measures should probably be focused on treatment or diversion of the point sources.
- With some models, maps of nutrient sources can be prepared which highlight the main sources or hot spots. This can help to focus control actions on critical source areas.
- It can be difficult to understand complex catchments with a range of land uses, sources and processes without using a model.
- Models can be used in a predictive way to assess the loads entering the lake under various land-use, land management or other source-control scenarios. In other words, the model can be used to answer 'what if' questions; for example, will treatment of point sources be sufficient to reduce loads to the lake to an acceptable level?
- Models can be used to interpolate, extend and interpret measured loads. For example, if measured loads cover a large part of the catchment, a simple model could be used to fill in the remaining gaps. A model can also be used to predict the load in a catchment based on the load measured in another catchment, taking into account differences in factors such as topography and climate. Some models can also help to interpret and extend results from a relatively short monitoring record, which by itself may be biased through annual variability or storms.
- Some models can be applied over a long simulation period to provide quantitative risk assessment. For example, in New Zealand, models have been used to assess the risk from large slugs of sediment entering estuaries as the result of urban earthworks (Cooper et al. 1999).
- Model results, input and output maps can be used as communication tools and can help to focus discussion of the catchment.

Table 8: Summary and comparison of models used in New Zealand

Model Type	Examples	Description, strengths and weaknesses
Export coefficient models	Export coefficient model in this guide WinCMSS (CSIRO 1997; Davis and Farley 1997)	• Based on export coefficients and point source estimates. • Simple and quick to use for present situation and land-use change predictions. • Can serve as a quick check of more complex models. • Uncertainty in assignment of export coefficients. • Most techniques do not consider in-stream attenuation or wetland/buffer losses explicitly. • CMSS includes attenuation and effect of land management practices, but it has not been set up for New Zealand.
Nitrogen leaching models	OVERSEER (AgResearch 1999; Ledgard et al. 1999) Nitrogen Leaching Estimation (NLE) for dairy pasture (Di and Cameron 2000) GLEAMS (Knisel 1993; Webb et al. 2000)	• Simple budgeting models predict nitrogen leaching based on a balance of inputs and outputs. • Simpler models give only annual average leaching. • More complex leaching models account for variations in soil reservoirs of nutrients and a variety of physical processes. • Basic budgeting models are easy to use. • Generally supported by fertiliser industry. • Most leaching models do not address phosphorus or sediment loss, and loss in surface runoff. • Do not account for wetland, riparian, or in-stream loss. • Only set up for pasture and cropping.
Physically based catchment models	BNZ, GLEAMSHELL, WAM (Bottcher et al. 1998; Cooper and Bottcher 1993)	• Based on mathematical description of a range of chemical, physical and biological processes to predict water, sediment and nutrients. • Account for variability in soils, climate, topography, land use. • Include in-stream attenuation and real stream network. • Calibration data should be used. • Require specialist input to set up, run, and calibrate.

Regional regression	SPARROW (Alexander et al. 2002; McBride et al. 2000; Smith et al. 1997)	• Load prediction equations using measured loads to develop the prediction equations. • Methods usually statistically based, not physically based, so it is difficult to interpret equation parameters. • Has not been applied to nutrients in the past in New Zealand. • SPARROW is based on mass balance, includes in-stream attenuation, and allows for spatial variation in soils and land use. It has been applied successfully to the Waikato and is being applied to other NZ locations. • Limited use for prediction of effects of nutrient source controls.

Despite these positive aspects, modelling is not without its drawbacks. For one thing, all models are imperfect and approximate. There are techniques, such as sensitivity analysis, that attempt to identify the uncertainty in the model predictions. Even then, there remains a broader uncertainty about whether the model is a fair reflection of reality.

One way to assess the reliability of a model is to compare the models predictions against real ground truth data. For example, predicted loads can be compared against measured loads. Often part of the measured data is used for calibration: model parameters are adjusted until there is a reasonable agreement between the predicted and measured values. The remaining data can then be used as a separate check, or validation of the model. Even after such a process, uncertainty remains about model predictions for conditions outside the range of the calibration/validation conditions. Collection of field data often goes hand-in-hand with modelling, and it should not be overlooked when preparing for a modelling exercise. Also note that if the lake-outlet load and flow are available, the input load can be estimated using simple empirical relations between input load and output load (Vant, 1997, section 15.3; Chapra 1997), and this estimate can be compared with the load predicted from the model.

Many models use spatial information, such as the distribution of various land uses. To aid data input and mapping of model results, it is now common practice to link models with digital spatial databases, or geographic information systems (GIS). Various nationwide databases are available in New Zealand (see Table 9) in addition to local or regional databases.

Table 9: Summary of national databases

Database name	Contents	Source
Land Resource Inventory (LRI)	Digitised maps of vegetation (can be used for land use), land-use capability, soil type, rock type, slope class, erosion class.	Landcare Research
Digital Cadastral Database (DCDB)	Legal lot boundaries.	Terralink
NZTopo digital data	Digitised 1:50,000 National Topographic Map 260 Series maps; includes features such as contours, streams, roads.	Land Information New Zealand; distributed through various suppliers
Climate Database (CLIDB)	National climate database; rainfall, temperature and climate data; web-based access available.	NIWA
National Soils Database Extension	Physical and chemical properties for soils; extension to LRI.	Landcare Research
National Hydrometric Database	Stream level and flow records.	NIWA and regional councils
Landcover database (LUCD)	Satellite land-use maps, gridded; more recent than LRI.	Terralink or Ministry of Forestry
Land use derived from AgriBase	Land use with breakdown of agricultural land-use classes.	AgriQuality

Notes: There is usually a charge for data retrieval, and there may also be a royalty for access to or use of data. Local authorities often have access to national databases. There are various local and regional databases in addition to the national databases.

Models can never predict accurately how a specific catchment will perform, since all the variables cannot be foreseen. This is particularly true if they are dependent on climate – itself unpredictable. As a general principle, models are better at predicting *relative* effects of different land uses or land management rather than the absolute load of nutrients. Much more calibration and model testing is required for good predictions of absolute loads than is required to determine the relative effects of different management scenarios.

Finally, many models do not provide predictions of fluctuations in load through the year or from year to year. Such information is important for prediction of the impacts on lakes, and in such cases predictions of the impacts of nutrient loads must be based on observations of the inflows or lake.

We will now look more closely at the models listed in Table 8.

Export coefficient model

This simple approach predicts nutrient loads entering a lake, based on the land use and export coefficients for each land use. Point sources can also be added. Despite the limitations of estimating export coefficients, this method is still very popular and can serve as a quick screening method or a rough check of more detailed models.

The steps for this modelling are as follows.

Step 1. Determine catchment and sub-catchment boundaries.
Obtain digital or hard-copy maps of the topography (see Table 9 for sources of digital data). The GIS departments of local authorities can also provide help at this stage. Delineate the catchment boundary, either manually or using automated catchment delineation programs (such as those contained in good GIS packages). Delineate sub-catchment boundaries if you need to break down the loads to a sub-catchment level.

Step 2. Determine land-use areas.
Decide on land-use classifications to be used and determine the area of each land use in each sub-catchment. Usually this will involve obtaining maps of land use in the catchment. If digital maps are available, then determining the areas of each land use can be done easily using standard GIS operations. Otherwise the area can be obtained using a planimeter.

In some cases new land-use classes may need to be introduced, or finer-scale information used. Local knowledge or a local survey, cadastral boundaries or aerial photos may help in this case. City and district councils can be a useful source of information and expertise when refining land-use maps.

Check that the sum of the land uses equals the total sub-catchment area.

Step 3. Select export coefficients for each land use.
This is the most difficult part of the exercise. Use Figure 2, Figure 4 and Table 3 as a starting point, then modify these after considering the factors in Table 5. Factors to consider when modifying the export coefficients are: rainfall, slope, intensity of land use, amount of fertiliser used and method of application, land drainage, wetland and riparian conditions, vegetative cover, lithology, soil nutrient status, land management, and imperviousness of urban catchments. If measured catchment export coefficients for the study area or a similar area are available, then use them (with modifications if necessary). Consider measuring more export coefficients.

Step 4. Determine the load from point sources.
Obtain a list of the point sources in the study area, and assess the load for each source. There may be existing load information from resource consent applications or consent monitoring. Consider taking new measurements for suspected major point sources if existing information is not available or reliable. Table 2 may be used as a first estimate for some sources.

Step 5. Calculate the load entering the lake.
This is a straightforward calculation which can be performed with a spreadsheet. The area of each land use is multiplied by the export coefficient for that land use, and these loads are added together. Point sources are then added to arrive at the total load entering the lake. Separate sub-totals for each sub-catchment can be calculated if desired. Sub-totals can also be calculated for each land use, and non-point sources can be compared with point sources. Present the results as pie charts or maps.

Step 6. Refine the estimate.

Consider refining the load estimated for the main sources by taking measurements, reading the literature or consulting with experts. Estimate the sensitivity to error: re-run the model for possible values using a range for each of the parameters. Typically, modelling packages allow random allocation of parameters based on mean and error values. By running each model repeatedly, an idea of confidence in the estimate can be given.

Step 7. Repeat the above steps for various land-use, land-management, or point-source control scenarios as required.

For partial implementation of controls, first estimate the effect of full implementation, then scale back the change in load based on the degree to which the control practice will be implemented (Davis and Farley 1997).

The Catchment Management Support System (CMSS) is an extended form of the export coefficient model (CSIRO 1997; Davis and Farley 1997). It includes a simple travel-time method for in-stream attenuation. It also allows land-use change and management policies and their effectiveness to be defined, and can combine these into policy sets for scenario analysis. The programme is Windows-based and makes use of a simple mapping system. However, this system is not compatible with standard GIS data, and is not set up for New Zealand conditions (for example, the database of export coefficients is specific to Australian conditions). Nevertheless, CMSS demonstrates the potential for automating and extending this popular class of model.

Leaching models

Several models have been developed to assess nitrogen leaching from pasture and cropping areas in New Zealand. (Here we define leaching as the loss of soluble constituents from the soil by percolation of water beyond the soil root zone.) While leaching is only one loss pathway, it can be a dominant loss mechanism for nitrogen from pasture land use.

OVERSEER (AgResearch 1999; Ledgard et al. 1999) is a simple budgeting model which calculates leaching based on the balance of a variety of inputs and outputs to the soil system, as shown in Figure 6. The program is intended primarily as a means of matching fertiliser application to plant requirements, but has also been used to assess leaching.

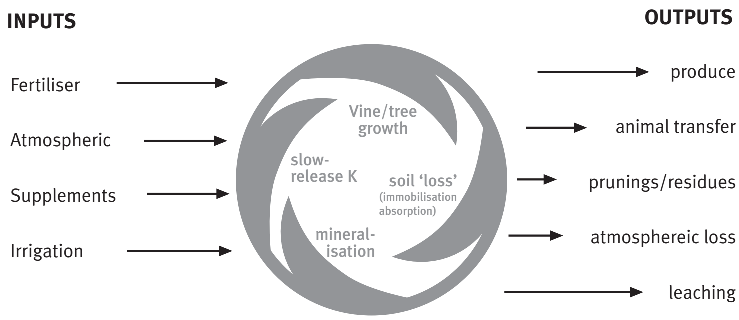


Figure 6: Schematic representation of the OVERSEER nitrogen leaching model

The Nitrogen Leaching Estimation model (NLE) is also a simple, semi-empirical, budget-based model for dairy pasture (Di and Cameron 2000). It uses a mass balance to determine the amount of potentially leachable nitrogen generated each year, and then determines the leaching as a function of the amount of potentially leachable nitrogen. Results are expressed as leaching rate per 100 ml of leachate, so an estimate of the volume leached is also required.

GLEAMS (Knisel 1993) is an example of a more complex physically based model which has been applied in New Zealand (Webb et al. 2000). Such models account for the nitrogen in a number of pools or compartments of nitrogen in the soil and plants, and the fluxes between the compartments. Even so, some of the pools are conceptual and cannot be measured directly. Also, the equations describing the transformations are often empirically based, and may not be applicable outside the range for which they were developed. For example, Webb (2000) found it was necessary to recalibrate an important mineralisation rate parameter for New Zealand conditions. Nevertheless, such models have the advantage that they can predict the daily or annual variation of leaching, and as our understanding of the processes improves they are likely to be used more often at a management level.

Physically based catchment models

In a physically based catchment model mathematical expressions are derived for the various physical, biological and chemical processes occurring in the soils and streams. These components are then combined temporally and spatially to derive the nutrient load. Because the model simulates the chemical and physical processes, it can (in principle) determine the effects of varying climate (rain and temperature), land management and soil properties. This allows a model developed for one area to be applied to another area — provided the range of conditions is not very different. This approach differs from other more empirically based approaches, which rely on knowledge of the likely yield based on measurement at similar sites. Physically based models also have the advantage of improving our understanding of why the catchment behaves in a certain way.

Usually such models are spatially distributed, the catchment is broken down into a number of sub-catchments or grid cells to account for spatial variability in land use, soils and climate, and stream conditions.

There are several physically based, spatially distributed models described in the literature. In New Zealand the related family of models (BNZ, GLEAMSHELL and WAM) (Cooper and Bottcher 1993; Bottcher et al. 1998) have been used in a number of applications, such as at Oteramika in Southland (Thorrold et al. 1998), Toenepi in the Waikato (Rodda et al. 1997a), Mahurangi in the Auckland region (Stroud and Cooper 1997) and Ngongotaha and Taupo (see case studies in the appendices). GIS interfaces have been developed for GLEAMSHELL (Rodda et al. 1997a) and WAM.

These models work by breaking up the catchment into a grid of cells, calculating the load from each cell, then combining the loads to derive the total load at each sub-catchment outlet (Figure 7).

The GLEAMS model (Knisel 1993) is used to calculate the load from each cell in GLEAMSHELL and WAM. GLEAMS was developed by the US Department of Agriculture to predict the leaching and surface runoff of nutrients and pesticides from agricultural land. It uses mathematical descriptions of the individual processes occurring in the soil, such as infiltration of rain, plant uptake of nutrients and water, decay of plant residue and soil organic matter, and soil erosion. The model then links and combines these processes to calculate the load of nutrients leached into the groundwater or washed off the land surface (either dissolved or attached to soil) on a daily basis. In BNZ the older model, CREAMS (Knisel 1980), is used instead of GLEAMS.

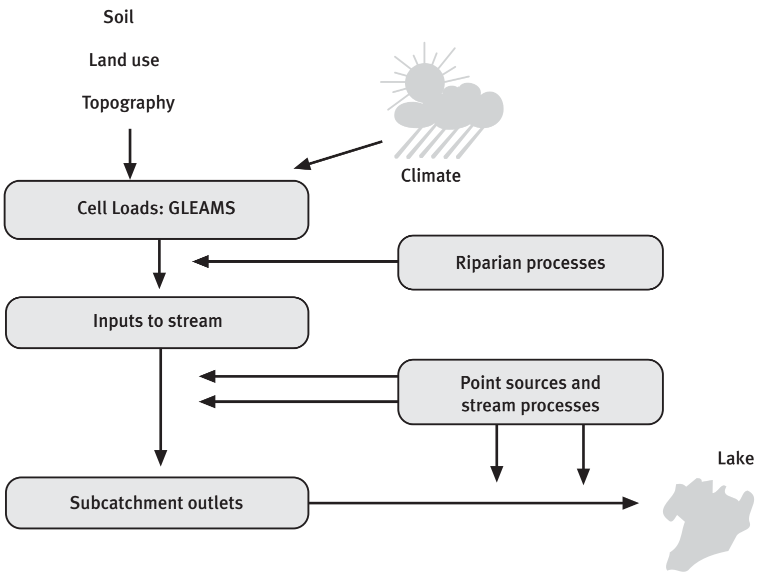


Figure 7: Components of the GLEAMSHELL model

The models then take the loads from each cell, route them to the sub-catchment outlet, and then down the stream network to the lake. During this routing, point sources are added where appropriate, and attenuation is taken into account. The result of these calculations is a time series of the flows and nutrient loads for each year for each sub-catchment.

Despite being physically based, such models usually require calibration, so a measurement programme should be considered in the budget for such a modelling programme if suitable data are not already available.

Regional regression models

In regional regression models, the load in streams in a region is regressed against explanatory variables such as catchment area, slope, and population density. The equations for sediment load given in Figure 2 are a simple example. The equations are developed for one region, and so cannot be readily applied to other areas.

These sorts of techniques are of limited use in scenario analysis, because the load usually varies as a function of only a few variables. There is also no guarantee that the explanatory variables are in fact the causative factors, which can introduce uncertainty when deciding on management practices. For example, a correlation between silt in streams and intensive pastoral land use may reflect naturally more silty conditions in lowland streams where intensive pasture occurs, rather than the effect of the pasture land use itself.

There are no existing conventional regional regression equations for nutrient load in New Zealand. Recently, the SPARROW regression technique (Alexander et al. 2002; McBride et al. 2000; Smith et al. 1997) has been applied successfully to the Waikato basin, and is now being applied to the entire country. SPARROW is an advanced technique, because it is based on mass balance, includes in-stream attenuation, allows for spatial variation in soils and land use, and routes loads down a drainage network. The present version is limited to the prediction of annual average loads.

Risk factor mapping

Risk mapping can be used to identify relative hot spots of sediment and nutrient generation, where managers can target source control measures, or sensitive areas in a catchment where land-use change could lead to large changes in load.

The general approach in such models is to identify risk factors for sediment or nutrients (for example, slope, rainfall, vegetative cover and soil erodibility for sediment). Maps of each risk factor are developed, and each risk factor may be broken into a number of categories. The factors are then overlaid or combined to form an overall index of risk (for example, multiplying slope, rainfall erodibility and vegetation factors). GIS is ideally suited to such methods.

The Land Resources Inventory erosion index (Eyles 1985) and a recent system developed by Summit-QuinPhos for phosphorus loss, are examples of risk mapping in New Zealand.

SECTION 5

Monitoring and measuring loads

Measuring loads is an important tool for water quality management. Here we can give only a brief introduction to monitoring, but readers requiring more detail are referred to a number of other publications.

Measuring is carried out in order to:

- gain direct data on inputs to the lake, without using modelling
- monitor the effectiveness of load-reduction programmes (which should include baseline or reference conditions)
- refine export coefficients for local conditions
- provide data for model calibration and validation.

When setting up a monitoring programme it is important to decide the location within the stream network where samples should be taken. In general, more attention should be given to large streams and streams draining critical source types. You should also consider a range of source types and land uses, especially for model calibration.

Another important consideration is the duration of sampling. For calibrating dynamic models (those that give a time series of outputs), a period of one or two years may be sufficient, although it is desirable to have a longer period both to reduce measurement uncertainty and to increase the likelihood of including dry periods and large storms. For measuring catchment export coefficients and direct load measurement, long-term records (at least over five years) are desirable to smooth out inter-annual variability.

Frequency of sampling is another important consideration. It is particularly important to sample storm events, as these can carry a disproportionately large fraction of the nutrient load. Ideally, some form of load-proportional sampling should be used, whereby the flow is monitored and samples are taken more frequently when the flow is greater. Usually an automatic sampler in conjunction with a flow-measuring device (such as a weir) is used for flow-proportional sampling. On the down side, this kind of sampling equipment requires considerable maintenance and is subject to failure. Another approach is to take samples on a regular basis (say, monthly), and over time build up enough samples from storm conditions. Another strategy is to have sampling parties on standby to collect samples during storm events.

The time lags within the catchment following a change in land use also need to be taken into account. Measuring the load straight after land conversion could give a false impression of the eventual load for that land use, because there are typically lags in soil fertility and groundwater delays. For monitoring pine areas, consideration should be given to the time since planting, because the loads from a pine catchment can change as the trees mature.

Measuring only concentrations (without flows and loads) gives a preliminary picture of variations between different land uses and sub-catchments, and is cheaper than load measurement. However, at some stage it will probably be necessary to measure the loads. Simply multiplying the mean concentration by the mean flow gives a biased estimate of load, since the concentration usually varies with flow rate.

A load estimate can be derived from concentrations and a flow record using various rating-curve methods. The idea is to derive a relationship between concentration and flow (and perhaps other variables, such as season), and then apply this relationship to a continuous flow record to arrive at the load estimate. If a flow record is not available, you can derive a surrogate record based on another flow recorder in the vicinity, either on a pro-rata basis by catchment area, or by using flow gaugings to derive a relation between the flows at the two sites.[12]

For some lakes, where groundwater inflows are a major proportion of the lake's water budget (or where you suspect there are near-lake subsurface nutrient sources that would not be picked up in stream sampling), measure the seepage of nutrients into the lakebed or shore zone. The original *Lake Managers' Handbook* (Gibbs 1987) describes techniques for measuring groundwater contributions to lakes. There are also techniques for measuring groundwater flux in boreholes, such as by tracer techniques or hot-wire probes, which can be used in conjunction with concentration measurements to estimate nutrient flows.

[12] More information on rating-curve methods can be found in Cohn et al. (1989), Crawford (1991), Ferguson (1986), Ferguson (1987) and Johnson (1979).

SECTION 6

Reducing loads to a lake

A typical process for deciding how to reduce sediment or nutrient loads into a lake is outlined in Figure 8. Generally, the process involves:

1. identifying the sources of sediment and nutrients
2. assessing options to reduce loads to target levels
3. selecting options
4. monitoring the implementation and effectiveness of the options.

This will usually involve a modelling and measurement effort. The selection of options and target levels depends on the social and political environment, as funds for the control measures must be raised and any measures to control sources will need to go through the normal resource planning process.

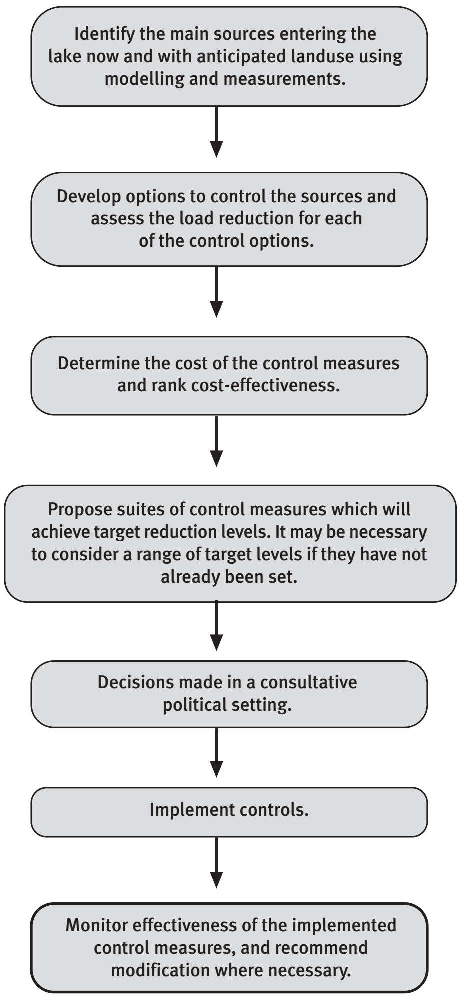


Figure 8: A typical process for controlling loads to the lake

A checklist of potential control measures, along with further references, is given in Table 10. There are methods available for assessing the effectiveness of some control measures, such as riparian buffer strips. For other control measures, the effectiveness must be judged. Physically based models can also provide guidance on the effectiveness of controls; indeed this was the case for the guidelines on riparian buffer strips (Collier et al. 1995). Catchment modelling is also an important tool for predicting the effectiveness of control measures such as changes in land use or land management.

If the proposed control measures alter the flows into the lake, then the concentrations of the inflows and the flushing rate of the lake should also be considered. Diverting a large, high-concentration inflow will increase the residence time in the lake, which will allow more time for in-lake processing. The average concentration in the inflows will also be reduced — an added plus. However, if the diverted inflow had low concentrations compared with the other inflows, the concentration in the remaining inflows will increase, which would offset the benefits of decreased loads and increased residence times (Vant 1987).

In this document, we have outlined a number of the recent developments in modelling approaches for studying effects of catchment land use on lake water quality, including principles of their application and potential problems of misapplication. Used wisely, these methods can have great benefit for lake management, while at the same time not removing the need for empirical studies of processes affecting nutrient inputs and in-lake responses. Together, the two approaches can be very powerful in allowing the prediction and mitigation of adverse effects, and in the application of suitable restoration programmes to redress lake degradation.

Table 10: Checklist of source control measures

Class	Examples and further references
Point sources	• Reticulation of septic tanks. • Improved waste treatment and disposal, including sewage, dairy shed waste and industrial waste. • Disposal of waste outside the lake catchment. • Relocation of industry.
Land use	• Reduce area of land uses with high nutrient or sediment yields (e.g. nitrogen leaching from market gardens). • Increase area of land uses with low yields (e.g. planting of forests). • Off-set new high-yield land use with new low-yield land use. • Restrict land use in sensitive locations.
Land management	• Grazing and stocking: reduce stocking intensity, remove stock from catchment over winter, change stock mix, avoid over-grazing, cut-and-carry feed, and avoid grazing heavy animals on sensitive soils in winter (Hunter-Smith 2000). • Fertilising: avoid over-fertilisation; use careful application avoiding streams, saturated areas and wet times of the year; apply two smaller dressing rather than a single large dressing (Code of Practice for Fertiliser Use, NZFMRA 1998).
Riparian and wetland management	• Use stream-bank planting to reduce bank erosion (Hicks 1992; Ritchie 2000). • Filter strips: quantitative methods for designing filter strips are in Collier (1995); an approximate quantitative method is in Williamson et al. (1996). • Use riparian buffer strips (Collier et al. 1995). • Exclude stock from streams and provide stream crossings; place troughs away from streams; provide shade away from streams (Collier et al. 1995; Hicks 1995). • Protect wetlands (Collier et al. 1995). • Construct wetlands for urban stormwater (Auckland Regional Council 1992). • New technologies: construct wetlands for dairy drainage ditches (Nguyen 2001); construct riparian denitrification walls (Downes et al. 1997).
Soil conservation measures	• General measures: tree planting on slip-prone areas, maintaining good ground cover, pasture improvement, re-grassing slips, debris dams (Hicks 1995; Kraayenoord and Hathaway 1986; National Water and Soil Conservation Authority 1973; Ritchie 2000). • Control urban earthworks (Auckland Regional Council 1995). • Control forestry earthworks (Vaughan et al. 1993).
Diversion of inflows	• Divert the flow from high-concentration inflows from entering the lake, and discharge them into a neighbouring or downstream catchment .

Appendices

Appendix 1: Estimation of the concentration of nitrogen and phosphorus under different land uses

Dairy

Toenepi, Waikato (Wilcock et al. 1999) (two separate years)
Oteramika, Southland (Thorrold et al. 1998) (two separate years)

Hill Pasture

Otaketake, Taupo (Schouten et al. 1981)
Manawatu (Bargh 1978)
Purukohukohu, Central Volcanic Plateau (Cooper and Thomson 1988)
Whatawhata, Waikato (Quinn et al. in prep)
Mahurangi, Auckland (Stroud and Cooper 1997)
Moutere, Nelson (Smith 1992) (Nitrogen only)

Low Intensity Pasture

Taita, Hutt Valley (McColl et al. 1977) (Phosphorus only)
Whareroa, Taupo (Schouten et al. 1981)
Alexandra, Auckland (van Roon 1983) (two separate years)
Kuratau, Taupo (Schouten et al. 1981)
Lucas, Auckland (van Roon 1983)

Native

Purukohukohu, Central Volcanic Plateau (Cooper and Thomson 1988)
Taita, Hutt Valley (McColl et al. 1977) (two catchments, Phosphorus only)
Big Bush, Nelson (Neary 1978)
Tararua, Manawatu (Bargh 1977)
Maimai (Neary 1978)
Waimarino, Taupo (Schouten et al. 1981)
Waimarino, Taupo (Schouten et al. 1981)
Waihaha, Taupo (Schouten et al. 1981)
Whatawhata (Schouten et al. 1981)
Bay of Plenty (Macaskill et al. 1997) (Concentrations only)

Exotic

Taita, Hutt Valley (McColl et al. 1977) (Phosphorus only)
Purukohukohu, Central Volcanic Plateau (Cooper and Thomson 1988)
Deacons, Auckland (van Roon 1983) (two separate years)
Mahurangi, Auckland (Stroud and Cooper 1997)
Bay of Plenty (Macaskill et al. 1997) (Concentrations only)

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Appendix 2: Lake Rotorua case study

Lake Rotorua is a shallow lake, 81 km^2 in area, on the central volcanic plateau of the North Island. The lake became eutrophic during the 1950s as a result of land development in the region. By the 1970s it had become clear that source controls were needed to control the eutrophication (White 1978).

An early nutrient budget for the lake (1965–70) (Fish 1975) was prepared based on measurements of nitrate, ammonia and phosphate in sewage, rainfall, and the spring-fed streams during baseflow. This identified that (at the time) sewage contributed 4.2% of the dissolved inorganic nitrogen and 13% of the phosphate load entering the lake. A nutrient budget for 1976–77 was also prepared by Hoare (1980), and this included particulate and organic nutrients and used a simple rating method for load estimation, including the effects of storm flows. This identified that sewage contributed 10.9% of the total nitrogen and 15% of the total phosphorus.

As a first step to control loads to the lake, sewage reticulation was extended during the late 1970s and the sewage treatment plant was upgraded, with chemical removal of phosphorus in 1976. Diversion of sewage disposal outside the lake catchment was also proposed, and it was predicted that this would reduce concentrations in the lake to the upper end of the mesotrophic range and avoid increased future population-related increases (Rutherford 1978). This recommendation was never implemented.

Despite these controls, the sewage load to the lake increased as a result of an increasing population. By 1984–85 sewage contributed 50% of the total phosphorus and 27% of the total nitrogen external inputs to the lake, and this was accompanied by a gradual increase in nutrients in the lake and more rapid deoxygenation in calm periods (Rutherford et al. 1989). In the late 1980s targets for nutrient concentrations into the lake were established, based on restoring chlorophyll concentrations to pre-1960s levels. Models for chlorophyll concentrations were used to establish target concentrations of phosphorus in the lake, and the associated reductions in external inputs were established from a simple calibrated lake mass balance model (Rutherford et al. 1989).

Assuming that inputs from streams would remain constant, a target load of phosphorus from sewage (to three tonnes/year, a reduction of over 90%) was established. A target for sewage nitrogen loads (30 tonnes/year, 80% reduction) was also established. In 1991 advanced sewage treatment, involving spray application to land with passage of the drainage water through wetlands, was introduced. This reduced sewage-related nutrient loads to below the target levels, although in recent years the nitrogen load has increased to close to the target, as a result of deteriorating performance of the land-treatment/wetland system (Kit Rutherford, personal communication).

It was noted during the 1970s that high concentrations of nitrogen and phosphorus in the lake coincided with periods of relatively prolonged calm periods, indicating that internal sources of nutrients were released from sediments during periods of anoxia in the bottom waters. This was confirmed using a mass balance approach (White et al. 1978). Using a simple model for the lake it was determined that internal loads contribute a large part of the load to the lake (Rutherford 1989). Such internal loads can provide nutrient loads to the lake long after the reduction of external inputs, and hence prolong the recovery time of the lake.

The reduction of phosphorus concentrations in the lake following reductions in sewage load was assessed using a sophisticated lake-mixing and water-quality model (Rutherford et al. 1989). While the lake water flushes out in a matter of months, the time scale for changes in the sediment is much longer (about 200 years for a sediment depth of 0.1 m). Nevertheless, the model indicated a noticeable reduction in lakewater total phosphorus concentration (roughly half the ultimate reduction) in the first 20 years following sewage load reduction. These predictions have not been tested.

It was also realised at an early stage that non-point sources of nutrients were important (Fish 1969). Erosion in the catchment of the lake was also a concern, with widespread gully, slip, stream-bank and sheet/rill erosion (White 1978). The implications for nutrient loads were unclear, but it was nevertheless considered appropriate to introduce soil conservation measures. These included riparian retirement and conservation planting, which were introduced during the 1980s as part of the Upper Kaituna Catchment Control Scheme (BOPCC 1975) (see Figure A1).

Figure A1: Ngongotaha stream reaches before and after riparian retirement

The benefits of the riparian retirement programme were estimated by Williamson et al. (1996). They determined from water quality and flow measurements that after implementation of riparian controls in the Ngongotaha sub-catchment there were load reductions of 85% for sediment, 27% for particulate phosphorus, 41% for particulate nitrogen and 26% for soluble phosphorus, but an increase of 26% for soluble nitrogen in the Ngongotaha Stream (see later for further discussion of nitrate).

These results were extrapolated to estimate the effect of riparian retirement on the phosphorus load entering the lake. They used the Ngongotaha results to estimate export coefficients from pasture with and without retirement. Then, using information on land use throughout the entire Lake Rotorua catchment and literature-based export coefficients for other land uses, they estimated that riparian retirement was responsible for a 21% reduction in total phosphorus load to the lake. They estimated that this would result in a 5 mg/m^3 reduction in total phosphorus concentration in the lake, which, in conjunction with control of sewage, would be sufficient in the long term to move the lake from a eutrophic state to a mesotrophic state. This provides a good example of a simple approach to load estimation and lake management based on export coefficients.

Williamson et al. (1996) also estimated the riparian buffer treatment efficiency by estimating the degree of treatment afforded by the riparian area for each stream reach (on the basis of a subjective classification), and then averaging over all the stream reaches. By this means they estimated that in this catchment the riparian buffers remove 55% of the nutrients in overland flow, which indicated that the riparian area should be reasonably effective in removing nutrients. This represents an alternative simple method to estimate the effectiveness of riparian controls

The increase in nitrate in the study by Williamson et al. (1996) was attributed to the delayed effects of pasture development in the catchment. They also noted that baseflow nitrate concentrations in the Ngongotaha stream had doubled over 20 years. Such delays were consistent with long groundwater residence times (50–100 years), which is comparable to the time since establishment of pasture. Tritium dating of larger spring flows (15–20 years) and stream baseflow (about 10 years) suggests a faster response in the groundwater (Fish 1975). This delayed response illustrates the importance of anticipating the effects of land-use conversion, and the long time scales involved. Nonetheless, the riparian retirement decreased total nitrogen and total phosphorus loads.

More sophisticated catchment modelling has been applied to the Ngongotaha sub-catchment (Cooper and Bottcher 1993). This was the first application of the BNZ family of models. The model was applied using over 1000 cells and including mapping of wetlands. The model predictions of the effects of riparian retirement were close to the measured effects (as discussed above). In the modelling study, increased nitrate concentrations after implementing riparian retirement were attributed to decreased in-stream attenuation resulting from shading of the stream, rather than to delayed increases in inputs.

The model was also used to evaluate the effect of buffer strips and removal of septic tanks. This indicated that vegetation in the buffer strips should be maintained with a dense ground cover to improve trapping in overland flow. Removing septic tank effluent did not have much effect, as the tanks were in the headwaters and in-stream losses were predicted to remove a large fraction of the nutrients from this source. For similar reasons, improving wetland denitrification had little effect on catchment export of nitrogen. It was also suggested that the most effective source control would vary in different parts of the catchment, depending on the predominant source type and distance from the catchment outlet, and the model was valuable in highlighting such spatial differences.

In a later scenario study (Rodda et al. 1997b), the effects of changes in land use were assessed, including conversion of land-use capability class VI land to forestry (predominantly the steeper areas), and expansion of deer farming or dairying to cover class IV areas only or both class IV and class VI areas. The model was also set up and run using GIS. The study predicted that dairying would increase nitrogen loads (up to a factor of two), and dairying on steeper land would increase phosphorus loss. Deer farming on steep land was predicted to increase sediment loss by a factor of approximately two, suggesting that deer farming should be restricted to class IV areas. The expansion of pine onto steep areas decreased water, sediment and nutrient export. The model results also showed that some soils produce little surface runoff, and that sediment loss from these soils was insensitive to land use. This BNZ modelling demonstrates the use of detailed modelling for scenario analysis and analysis of hot spots and sensitive areas.

Future management of the Rotorua lakes will be governed by the framework set up in Environment BOP's *Draft Regional Water and Land Plan*. The rules in the plan are still being developed, and are likely to include an assessment of changes in nutrient load associated with consents for landuse change, and promotion of nutrient source control measures.

Appendix 3: Lake Taupo case study

Lake Taupo is a large, oligotrophic lake, 622 km^2 in area, located in the central volcanic plateau. It is treasured for its clear blue water, and the tourism associated with the lake forms an important part of the local economy.

Protection of lake quality has been seen as an important goal for some time, and the Lake Taupo Catchment Control Scheme began in 1973 (see Figure A2). This scheme has involved extensive soil conservation, fencing, planting, riparian retirement, and the construction of stock access ways and bridges, with assets valued at $16.1 million.[13] At the time, the main reason for this control scheme was sediment control. This was in response to obvious land and river degradation, and reduction of sediment loads to the lake was really a precautionary measure rather than a response to deteriorating water quality (Waikato Valley Authority 1973).

Scientific studies on the lake catchment began in earnest in the 1970s with the establishment of an office of the DSIR at Taupo. Long-term monitoring and intensive studies of riparian retirement on the Whangamata Stream (see Figure A2) showed that soon after retirement the stream became choked with aquatic plants, and this resulted in uptake of nitrate from the stream. Over the decades, flax and woody vegetation became established, which provided better in-stream habitat but reduced the nutrient-trapping role of the stream (Howard-Williams and Downes 1984; Howard-Willams and Pickmere 1999; Howard-Williams et al. 1986).

During the 1970s there were also large monitoring programmes to measure the loads of nutrients entering the lake (Schouten 1983; Schouten et al. 1981; White and Downes 1977).

Early nutrient-related water quality concerns centred on near-shore weed growth around Taupo township, and this prompted investigations of the movement of leachate from domestic septic tanks around the lake shore (Gibbs 1979). In response to these concerns, sewage reticulation and treatment was introduced for Taupo township in the 1970s with the treated wastewater disposed of outside the Taupo catchment. Also, all the small townships around the lake now use wastewater treatment with application to land or wetlands.

[13] Environment Waikato website.

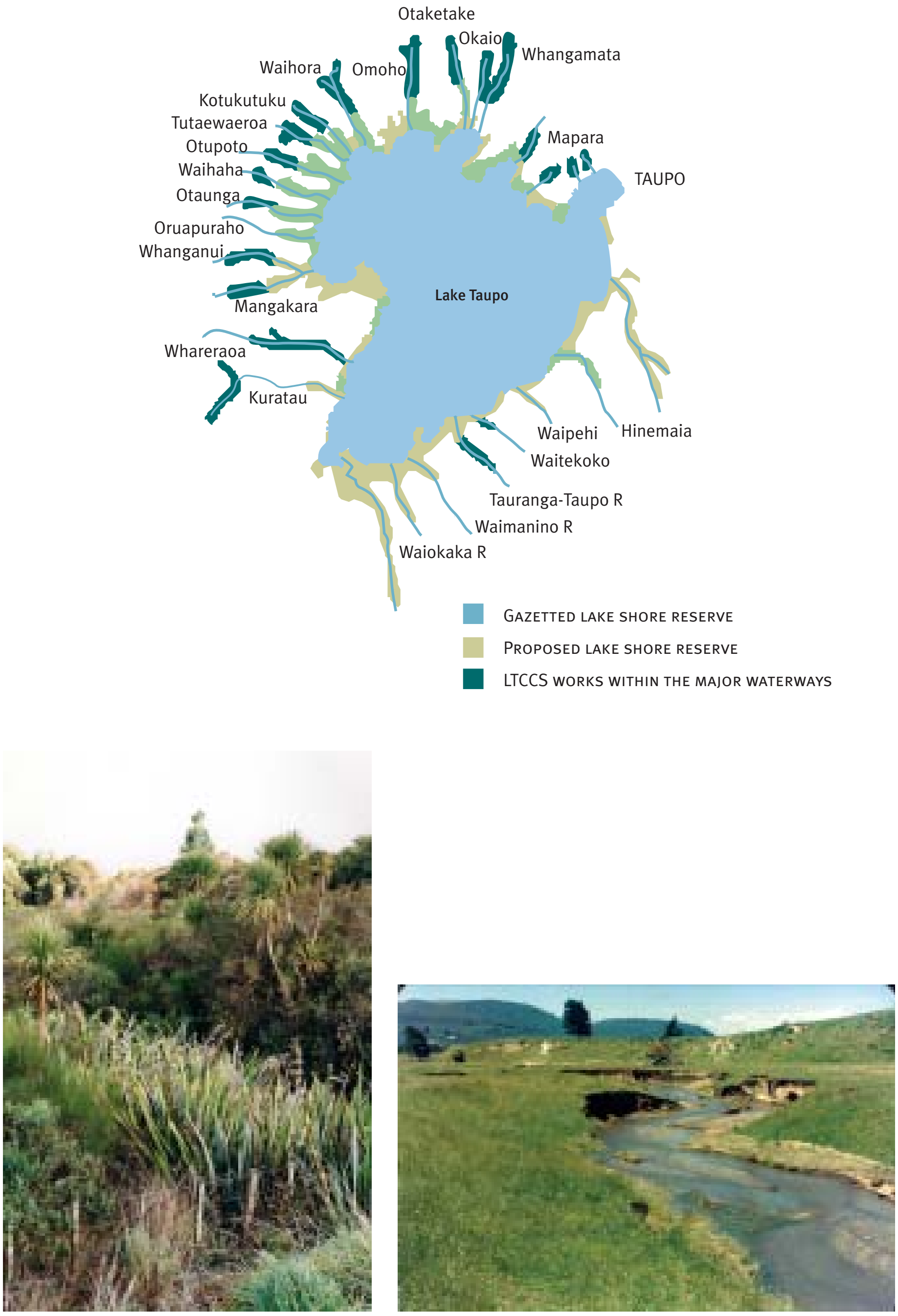
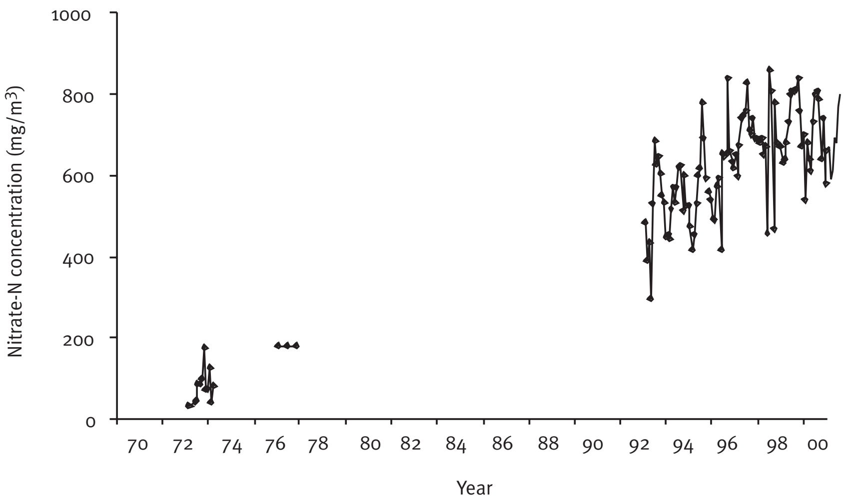


**Figure A2: Top: map of Lake Taupo showing areas with riparian protection
Bottom: re-vegetation of the Whangamata Stream**

Map and photographs provided by Environment Waikato.

More recently, concern has focused on the general degradation of water quality throughout the lake. Environment Waikato have raised concerns about changes in water clarity during the 1990s (see Figure A3). Ongoing monitoring in the Whangamata Stream and Mapara Stream, which have pasture catchments, showed increasing concentrations throughout the 1990s even though conversion to pasture occurred decades before (see Figure A3). It is suspected that this is a delayed response to the introduction of pasture into the catchment before 1975. The delays may be due to the long residence time of groundwater in the catchment and the gradual build up of soil fertility. This highlights the need to anticipate future trends and delays when managing lake water quality.



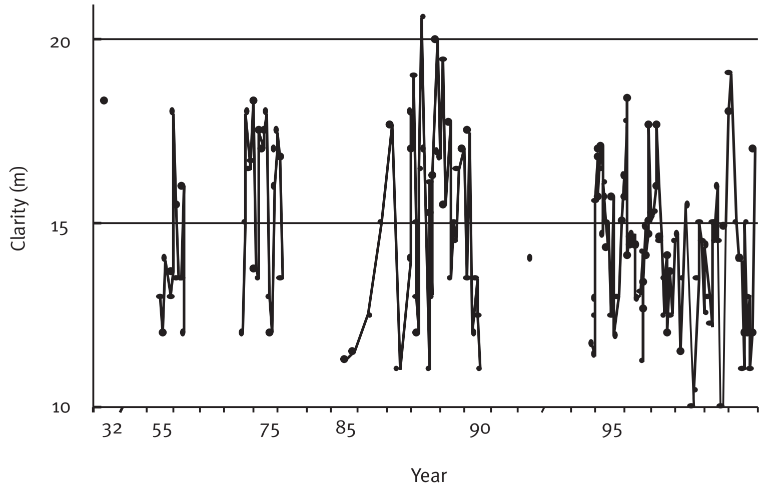


Figure A3: Top: changes in water clarity in Lake Taupo
Bottom: nitrate concentrations in the Mapara Stream

Concentration data and clarity figures provided by Environment Waikato.

The trends, in conjunction with the anticipated introduction of dairying into part of the catchment, have prompted a flurry of studies in the Taupo catchment, including:

- an export coefficient-based analysis of nutrient sources, including prediction of the effects of dairying
- OVERSEER modelling of nitrogen leaching from sheep and beef farming, dairy farming, and alternative land-management strategies to reduce leaching (see the main text for a description of the model)
- GLEAMSHELL modelling of the entire catchment, including scenario analysis for the assessment of the effect of dairying and the Tongariro Power Diversion, and linking to a lake mixing and ecosystem model (see the main text for a description of the model)
- more water-quality monitoring in pasture streams around the western bays, and groundwater quality measurements in conjunction with groundwater dating.

Environment Waikato have set up the planning and consultation process for controlling nutrient sources, and it is expected that some form of land-use control will be implemented. A range of goals (from do nothing to pre-development reversion) has been put forward for consideration (Environment Waikato 2000).

Appendix 4: Lake Alexandrina case study

Jonet Ward, Environmental Management and Design Division, Lincoln University

Lake Alexandrina is a relatively shallow lake, with a maximum depth of 27 m, situated in the Mackenzie basin (Figure A4). It is 7.2 km long and 0.9 km wide with an area of 640 ha, and shows strong visual signs of its glacial origin. It has high landscape, wildlife and recreational values (fishing, birds, aesthetic), and is situated in a catchment where farming has been the principal land use (Figure A5). Aquatic macrophyte communities grow down to 10–11 m with 100% cover, and *Chara corallina* continues sparsely down to 16 m.

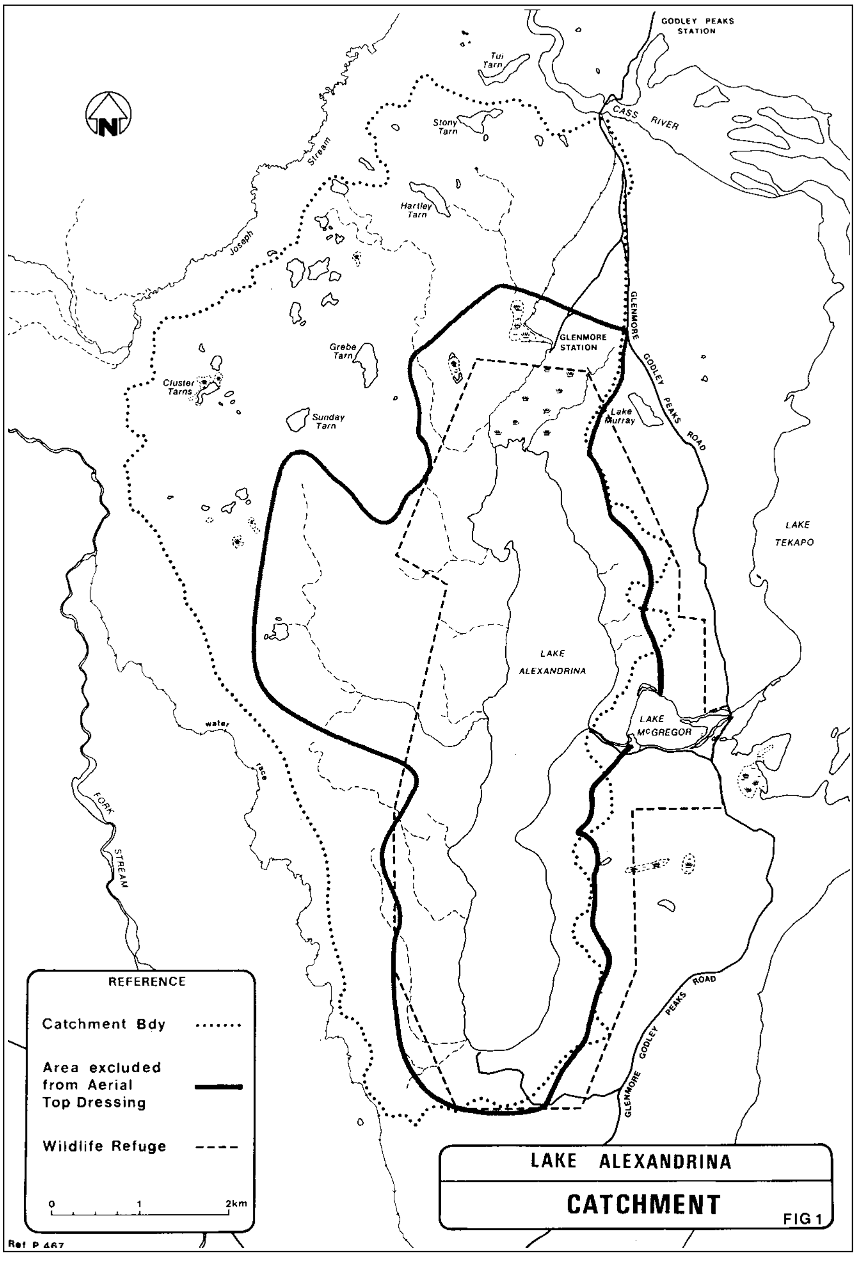


Figure A4: Map of Lake Alexandrina and its catchment, Mackenzie basin

Figure A5: View of L. Alexandrina, showing modified rural catchment.

Sources of water to the lake (cumecs/ year x 10^6) include:

- surface streams: 5.72 (27%)
- overland flows: 1.11 (5%)
- ground water: 10.1 (48%)
- precipitation: 4.11 (20%).

(Source: Waitaki Catchment Commission and Regional Water Board 1984)

Water is retained in the lake for approximately four years.

The lake has mesotrophic status (Vant and Davies-Colley 1984). Hayes (1980) recorded chlorophyll a levels of 0.4–3.8 mg/l between November 1978 and March 1979, and total phosphorus at 1 m of 0.009–0.015 mg/l in March 1979. Nutrients from the catchment enter the lake attached to soil particles, as organic matter and in solution from inflow steams, groundwater and diffuse sources. Nutrients originate from:

- access of stock to the trout-spawning streams
- grazing at the lake edge
- hut settlements
- aerial topdressing
- underground sources.

The main management concern for the lake has been the frequency of algal (*Anabaena*) blooms caused by increased nutrient levels in the lake, which began to cause concern after 1980 due to their potential effects on wildlife and the recreational fishery.

Hoare (1982) modelled the phosphorus in the lake and concluded that the nutrient status was probably the result of nutrient loads coming in with the bulk of the water rather than from sewage, which is a relatively minor contributor. Lovegrove (1985) estimated that up to 50% of the phosphorus entering the lake may come from groundwater, 32% from surface water inflows and only 3–9% from hut settlements. Water from springs entering the bottom of the lake was sampled in 1987 and found to contain phosphorus and nitrogen in higher quantities than in water nearby (Ward 1989).

The Lake Alexandrina Steering Committee was set up in 1984 following a meeting of interested parties. Phosphorus was identified as the major factor causing deterioration of water quality (Ward-Smith et al. 1985). Two main sources of phosphorus were identified: one from the hut settlements at the outlet and south end, the other from agricultural sources.

A report by the Taranaki Catchment Commission to the Waitaki Catchment Commission (1987) states that while inflows exceed outflows, much will be permanently lost to the sediments and the annual mass contribution to the lake is low. They recorded total phosphorus levels in the lake of 0.014–0.032 mg/l from June 1984 to June 1985. Concerns over the blooms and the potential effects on wildlife and the recreational fishery resulted in interim guidelines for management of the lake (Ward-Smith et al. 1985).

To control phosphorus from agricultural sources, the interim guidelines of the steering committee (Ward-Smith et al. 1985) recommended:

- a moratorium on top dressing – there should be no aerial top dressing within 500 m of the eastern shore or 800 m of the western shore, or around the water sources to the lake
- runholders restrict the use of fertiliser, and refrain from supplementary feeding of livestock along the lake and stream margins
- farmers be asked to minimise stock access to the lake and spawning streams, and a deer fence be installed around the north end wetland (the South Canterbury Acclimatisation Society also improved spawning streams)
- the Department of Lands and Survey should not allow earthworks, tracking or cultivation near the lake shore or near any watercourse that may lead to sediment entering the water; they should also ensure that no more than one paddock on a property is cultivated at any one time in the area excluded from top dressing, and then only when there is minimum risk of wind erosion
- the Waitaki Catchment Commision complete its study of the relationship between land use and lake-water quality. This study was considered essential to determine the source of the large percentage (48%) of groundwater entering the lake and its phosphorus content.

Subsequent studies suggest that the Cass fan to the northeast of the lake catchment is the source of the groundwater inflow (Waitaki Catchment Commission and Regional Water Board 1987).

In 1987 the Mt John Station run that occupies the southern part of the lake catchment was purchased by the New Zealand Defence Force (NZDF), so stock were removed from this land. However, the NZDF are in the process of negotiating with a runholder to exchange this land for land adjacent to their live-fire area, so it is likely that grazing will be re-established on the Mt John run. The

Department of Conservation, however, now have a reserve/buffer strip around Lake Alexandrina on the Mt John run, so stock intrusion to the southern lake margins is controlled. Around the northern margins, on the Glenmore run, stock intrusion continues to be a problem as sheep are drawn to the lake edge by the shade of the willow trees.

To control phosphorus deriving from the settlements, the interim guidelines (Ward-Smith et al. 1985) recommended that:

- the Department of Lands and Survey act immediately to maintain a moratorium on building at the hut settlements, but permit the installation of household effluent-holding tanks
- the Mackenzie County Council investigate a system for sewage disposal outside the catchment for each settlement; sewage tanks had already been installed at the south end settlement by 1989 (Ward and Stewart 1989).

Lake Alexandrina Management Guidelines, prepared by the Mackenzie District Council, were adopted in August 1999 following a consultative process with stakeholders at the lake. These provide assistance to applicants seeking resource consents in relation to building development. All liquid waste must be discharged into a holding tank, which is to be emptied by an approved septic maintenance contractor. The holding tank installation has been very successful, with 90% completed by June 2001 and a cut-off date for the remainder by January 2002 (A. Shuker, Outlet Hut Holders Committee Chairman, personal communication).

The Mackenzie District Council has included a Lake Side Protection Area around Lake Alexandrina in the *Proposed District Plan 1997*, to protect the visual amenity of the lake-side environment from inappropriate building development. It has discretionary status on new buildings and extensions to existing buildings around the lake.

Although management guidelines and controls in the lake catchment have been progressively implemented since 1985 to reduce phosphorus levels in surface inflows, an increase in total phosphorus from 1992 to 1996 of about 6% has been recorded (Burns and Rutherford 1998). Total nitrogen and chlorophyll a values showed little change over this period. However, the heavy algal blooms of the 1980s seem to have disappeared and are now seen on the lake only about every three years (A Shuker, Outlet Hut Holders Committee Chairman, personal communication).

Appendix 5: Lake Coleridge case study

Lake Coleridge is 33 km^2 in area, and is the eighth deepest lake in New Zealand, with a maximum depth of 200 m and a mean depth of 99 m. Together with other deep South Island lakes, it provides a vivid illustration of the importance of light attenuation in controlling the depth colonised by aquatic macrophytes of the littoral zone, and of sediment loads in particular.

Most of the large inland South Island lakes have relatively little urban development and low intensity of agriculture in their catchments. They nevertheless vary considerably in their optical properties, due to natural parameters such as high dissolved organic matter concentrations and high loads of suspended particulate matter in some lakes (Rae et al. 2001). Dissolved organic matter is derived mainly from decaying terrestrial plant material in the catchment, causing the characteristic tannin-stained colour of lakes such as Te Anau and Hochstetter. Suspended inorganic particulates, largely of glacial origin, cause the bright blue colour of alpine lakes such as Lake Tekapo. These optical properties of lakes were described in detail by Davies-Colley (Chapter 7 in the *Lake Managers' Handbook*).

The deepest-growing aquatic macrophytes in New Zealand lakes are often the characean algae, in the genera *Chara* and *Nitella*. Light limits the depth of these aquatic macrophytes via its effects on photosynthesis and growth. The important wavelengths of light are in the visible spectrum of photosynthetically active radiation (PAR) from 400 to 700 nm, and it is the available light in this range that can be related to the colonisation depth of the characeans. The decrease in available light with depth occurs exponentially, and can be described by an extinction co-efficient *K*d. Differences in *K*d between different lakes are closely related to the maximum colonisation depth of the macrophytes. Figure A6 presents a dataset from 22 South Island lakes, showing a very strong relationship between *K*d and the maximum depth of macrophytes. This strong relationship demonstrates the importance of light for limiting the depth of the macrophyte-dominated littoral zone, and suggests that these communities are likely to be highly susceptible to any changes in catchment management that add light-absorbing particulate or dissolved matter to lakes.

In managing lakes to avoid such changes, the time scale over which the plant communities respond to decreased water clarity is significant. In these lakes the bottom limits do not change seasonally, suggesting relatively long (several months) periods over which plants integrate their light environment (Howard-Williams et al. 1995).

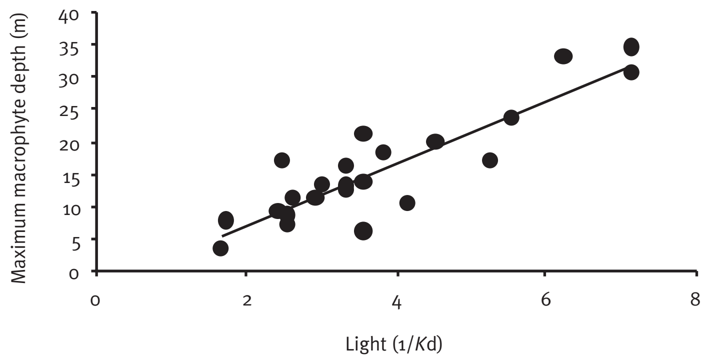


Figure A6: Relationship between available PAR, measured as the inverse of *K*d, and the maximum depth of the characean-dominated macrophyte community in 22 South Island lakes

Source: modified from Schwarz et al. (1996).

In 1993 Lake Coleridge provided a natural experiment to study the decline and recovery of a characean community following changes in light attenuation in the lake. An earthquake in the catchment triggered a large release of suspended particulate material into the lake, with the result that *K*d increased from 0.109 in August 1993 to 0.394 in July 1994. The subsequent loss of deeper biomass and reduction of the bottom limit from 30 m to less than 20 m in July 1995 occurred within about nine months. More recently, the lake has become clearer again as the suspended load settled. The recovery of the bottom limit was very slow, but has now stabilised again at 30 m. Further details of this event are described by Schwarz et al. (1996) and Schwarz and Hawes (1997). The fact that the decline occurred more rapidly than the recovery is an important issue to consider when managing lakes to protect littoral zones from reduced light intensity.

Although light is an important factor limiting the depth of aquatic macrophytes in most lakes, there are few other regions in the world where the relationship is as strong as in the South Island. This is because other factors that interact with light in controlling depth limits elsewhere are not important in the South Island, leaving light as the sole control of macrophyte depths. For example, temperature gradients and thermoclines are thought to be involved in macrophyte depth limitation in North America, and possibly Europe; but the thermoclines of New Zealand lakes are usually very deep, lying well below the littoral zone, due to our oceanic climate and exposure to wind, and do not usually interact significantly with macrophyte depth. There are also no animals that can exert a large enough grazing pressure on the characeans in the South Island lakes to affect their growth. In contrast, in North Island lakes the large freshwater koura *Parenephrops planifrons* occurs to depths of 100 m, and its grazing activity is thought to reduce charophyte biomass significantly in deeper water where growth is light-limited. Hence, although a reasonably strong relationship between *K*d and macrophyte depth does still occur in the North Island, it is modified by an interaction between light and koura grazing (Schwarz et al. 2000).

There is now also evidence that light attenuation controls not only the bottom limit of the littoral zone, but also species composition within the littoral zone. The ability of some characean species to grow deeper than others appears to be related to their different photosynthetic responses (Sorrell et al. 2001). There is also evidence that the limitation of the vascular macrophyte species to depths shallower than 10 m is due to their inability to support root growth when photosynthetic carbon fixation is limited by low light availability.

The management implication of these data is that the extent of lake littoral zones is likely to be highly susceptible to reductions in water clarity that result from changes in catchment management, whether this is due to suspended particles or phytoplankton growth due to eutrophication. Further details of the relationships between light and macrophyte depth limits, and how they may be applied to lake management, are discussed by Schwarz et al. (2000).

Appendix 6: Lake Forsyth case study

Malcolm Main, Environment Canterbury

Lake Forsyth (Wairewa) is a long, narrow lake in the lower part of the Little River Valley on the southwest corner of Banks Peninsula. It covers about 560 ha, is shallow (maximum depth about 4 m) and brackish. Freshwater flows into the headwaters of the lake from the Okuti and Okana rivers.

In early European times Lake Forsyth was an arm of the sea, and was known as Mowry Harbour. It was navigable by whale boats to its headwaters, and whalers felled trees in the valley, rafted the timber down the lake, and then out to their whaling stations in the eastern bays of Banks Peninsula. Sometime in the early nineteenth century, accretion of the gravel barrier along Kaitorete Spit caused the entrance to begin to close over, and by the mid-nineteenth century it was more or less permanently closed. It began to be opened artificially to the sea to allow drainage in about 1866, with subsequent openings in 1875, 1881, 1886, 1888 and 1895. Currently, it is opened about once every year, depending on the rainfall.

The Little River Valley was clothed in broadleaf-podocarp forest when the whalers arrived. They began to mill the timber by pit-sawing, and in 1863 a sawmill was established at Little River. By 1895 all the millable trees were gone, and soil erosion had occurred on a large scale. The streams were filled with silt during the latter half of that century and silt accumulated in the lake, so the lakebed had to be dredged to move the timber. After being taken down the lake it was transported by tramway to Lake Ellesmere and boated across to Timberyard Point.

Fires, both deliberate and accidental, were common in the Little River Valley in the nineteenth century. In 1886 they were so common that it was described as the 'Valley of a Thousand Fires'. These are likely to have been a source of nutrients to the lake. Another source was dairy farming, which was established after the timber was felled. In the 1920s there were 20 to 30 herds in the valley, with 20 to 40 cows each. For many years surplus whey from these farms was apparently discharged into the streams.

The lake has a large eel population, which is an exclusive Maori fishery, and – at least until recently – good populations of trout and perch were present. Inanga, common smelt, common bully and black flounder are also present in the lake.

In 1907 the first *Nodularia spumigena* bloom was recorded in Lake Forsyth. This hepatotoxic cyanobacteria has been recorded regularly since, and now blooms every summer. Sheep and dogs have died after drinking water from the lake, or after swimming in it. Fish kills have sometimes been associated with the blooms. It accumulates on the lake shore, and odours are produced when the masses of cells decompose.

The brackish nature of the lake is caused not only by the lake openings, but also by seawater over-topping the gravel bar, and the salinity regime makes it ideal for *Nodularia*. However, other factors also favour *Nodularia* blooms. Lake Forsyth is eutrophic, no doubt in part because of the historical discharges and subsequent nutrient release from the sediment. The nutrient concentrations in this lake are similar to those in Lake Ellesmere, but it differs from Ellesmere (which has *Nodularia* blooms infrequently) in some respects. Lake Forsyth is relatively clear, except when there is a *Nodularia* bloom (unlike Lake Ellesmere which is always turbid; see Figure A7), because it still supports extensive growths of *Ruppia* spp. and *Potamogeton pectinatus*. The macrophytes stabilise the lake bed, and Lake Forsyth is more sheltered than Lake Ellesmere, because it has hills on two sides.

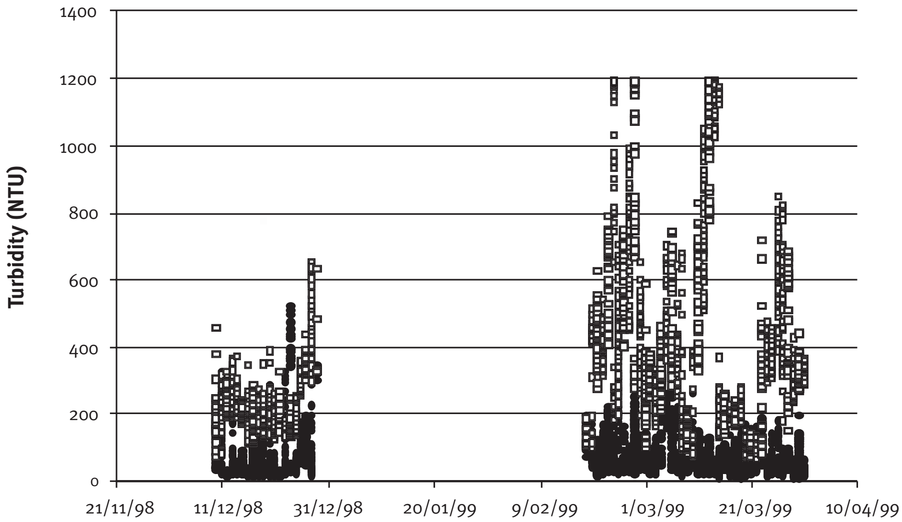


Figure A7: Continuously recorded turbidity measurements from Lake Forsyth (solid circles), and Lake Ellesmere (open squares) during two periods in 1998 and 1999

Nodularia blooms in Lake Forsyth are usually associated with increases of dissolved reactive phosphorus and dissolved nitrogen. This especially occurs after a period of calm weather (when daily mean wind speeds over the preceding week are less than about 3 m/s), and often during periods of low turbidity. However, the bulk of the dissolved nitrogen is ammonia, and at this stage it is unclear whether the ammonia actually aids the growth of the algae, or is produced by the decomposition of algal cells through the turnover of a large volume of cells during these bloom periods. In some years there is a series of blooms, and it appears that the later blooms are sometimes aided by the availability of soluble nutrients caused by the breakdown of earlier ones.

Environment Canterbury is continuing to collect data from the lake, and the addition of further data should enhance our understanding of *Nodularia* bloom development.

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About the Ministry for the Environment

The Ministry for the Environment works with others to identify New Zealand's environmental problems and get action on solutions. Our focus is on the effects people's everyday activities have on the environment, so our work programmes cover both the natural world and the places where people live and work.

We advise the Government on New Zealand's environmental laws, policies, standards and guidelines, monitor how they are working in practice, and take any action needed to improve them. Through reporting on the state of our environment, we help raise community awareness and provide the information needed by decision makers. We also play our part in international action on global environmental issues.

On behalf of the Minister for the Environment, who has duties under various laws, we report on local government performance on environmental matters and on the work of the Environmental Risk Management Authority and the Energy Efficiency and Conservation Authority.

Besides the Environment Act 1986 under which it was set up, the Ministry is responsible for administering the Soil Conservation and Rivers Control Act 1941, the Resource Management Act 1991, the Ozone Layer Protection Act 1996, and the Hazardous Substances and New Organisms Act 1996.

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