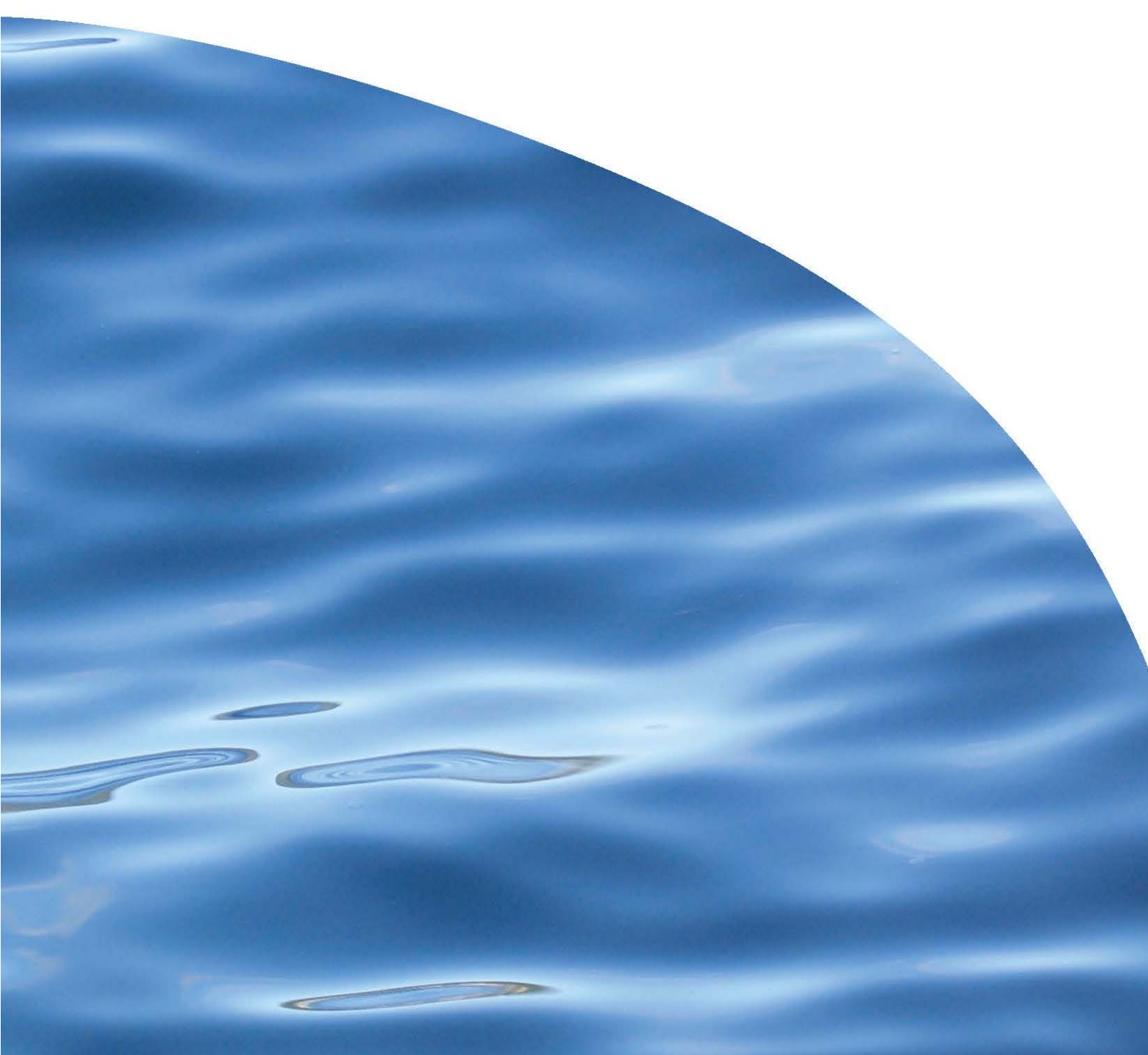**CAWTHRON INSTITUTE**

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NUTRIENT LOADING TO SHALLOW COASTAL LAKES IN SOUTHLAND FOR SUSTAINING ECOLOGICAL INTEGRITY VALUES

NUTRIENT LOADING TO SHALLOW COASTAL LAKES IN SOUTHLAND FOR SUSTAINING ECOLOGICAL INTEGRITY VALUES

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EXECUTIVE SUMMARY

The implementation of the National Policy Statement for Freshwater Management (Ministry for the Environment) has placed a strong imperative for regional authorities to establish and implement limits to protect water quality and ecological values of waterbodies in their region. Environment Southland (ES) is considering management targets for nutrient load limits for the region's shallow coastal lakes. These tend to be located in highly modified landscapes and as a result are under threat of degradation from land-use intensification.

This study consists of two main parts:

1. A literature review of national and international sources documenting nutrient loading to shallow lakes, and the effects on water quality characteristics (*e.g.* chlorophyll-*a*, water clarity) and ecological integrity (aquatic macrophytes, and other communities).

2. A modelling approach to assess nutrient loading to shallow lakes in Southland and the South Island, fitting Vollenweider regression models to nutrient loading and water quality data. We used the national CLUES (Catchment Land Use for Environmental Sustainability) model to predict loads of total phosphorus (TP) and total nitrogen (TN) and this was then related to in-lake water quality and ecological condition data for up to 19 lakes.

Review of the international literature indicates that managing loads to control phosphorus (P) concentrations is the most significant management action for protecting the ecological integrity (EI) of shallow freshwater lakes. In the extensive work conducted in European shallow lakes (UK, Denmark, Netherlands), managing catchment nutrient loads to maintain (or restore) shallow lakes to a clear-water macrophyte-dominated state, in-lake P concentration and sediment P content was nearly always the most immediate management objective. Patterns emerged in the literature around the importance of external and internal sources. For lakes that had experienced historically high loads or point source pollution, internal nutrient loading became a significant management objective, whereas moderately degraded systems responded more rapidly to catchment load reductions.

In European lakes (Denmark, Netherlands, UK), a critical threshold for P occurred around 100 mg TP/m^3, with very few lakes above this threshold having macrophytes covering > 50% of the lake bed. In contrast, in the warmer North American lakes (Florida) this threshold was closer to 50 mg/m^3. The nitrogen (N) threshold occurred around 1000 mg TN/m^3 in both European and North American lakes, above which few lakes had macrophyte cover exceeding 50%. From the study data available, lakes that experienced P loading in excess of 80 kg P/ha/y were likely to have accumulated P rich sediments that could result in internal loading.

The modelling work on the South Island shallow coastal lake-set provided significant relationships between N and P loading and several EI indicators. This was particularly evident for CLUES P loading corrected to in-lake TP concentrations using Vollenweider models. The strongest relationships were evident for chlorophyll-*a* and the trophic level index (TLI), with 79% and 86% of their variation explained by the TP loading variable, respectively. These relationships were linear, so for management purposes TLI targets for lakes could be directly related back to nutrient load targets for the lake. The strong model performance for this study is possibly related to the relatively short water residence times of the lakes, minimising the potential for in-lake nutrient processing. However there is a risk when using such correlational data, that modifying loads (either by load reductions or allowed increases) to particular lakes would not result in expected changes in retention or in-lake concentration due to factors such as internal loading or short-term blooms causing shifts in aquatic plant dominance (regime shifts).

From our modelling work, there was consistently less variation accounted for in TLI, chlorophyll-*a*, and other EI variables (*e.g.* macrophytes, macroinvertebrates) by TN loading models. However in several instances models were still significant suggesting that nitrogen loading is related to chlorophyll-*a* and TLI in some lakes. In-lake nutrient ratios of TN:TP suggested that many of the lakes in the data set are likely to be phosphorus limited (16 of 19 for TN:TP), with a smaller number being co-limited by both N and P. This could be possibly related to N-fixation when systems become highly -limited, but it was beyond the scope of this study to evaluate. A recent international meta-analysis study looking at the importance of nitrogen to water quality in shallow lakes reported that while chlorophyll-*a* and trophic status was more often controlled by P, macrophyte species composition and (in some cases) cover was related to in-lake TN concentration, with some species clearly having TN tolerances or preferences. This emphasises the importance of co-management of N and P for managing aquatic macrophyte communities in shallow lakes.

External loading estimates for South Island lakes were considerably less than those documented in most overseas studies, being maximum 32 kg P/ha/y. This made interpretation from the literature around risk levels difficult to interpret in the New Zealand context. However, based on negative phosphorus retention coefficients (suggestive of internal loads) for three of the more eutrophic NZ lakes, this indicated some risk when external P loads exceed 17 kg P/ha/y, which could ultimately generate internal loading issues. However further data collection around calculating nutrient retention coefficients (*i.e.* inflow/outflow monitoring) would provide greater certainty in these findings.

Results from the set of NZ lakes, relating nutrient loading to other key shallow lake EI variables (such as macrophyte and macroinvertebrate communities), also indicated that P loading was the more proximal variable relating to benthic communities. These results tend to indicate lower thresholds than results from overseas studies, with NZ lakes having loads resulting in mean summer TP concentrations of >50 mg/m^3 (and nearly always having little or no macrophyte cover). Macroinvertebrate community richness was even more sensitive to nutrient loading and declined linearly by nearly 42% (from approximately 26 to 15 taxa)

compared with reference condition lakes at the 50 mg/m^3 TP upper limit for macrophyte cover. So if this variable was used as an indicator of shallow lake EI, related to biodiversity values, we would suggest a more conservative target to minimize losses in biodiversity values. We would also suggest the loads, back-calculated using Vollenweider models, be no greater than levels to achieve a TP (and possibly N) concentration in the mid-eutrophic range (Burns *et al.* 2000), or 32 mg/m^3 TP (and possibly 531 mg/m^3 TN). This would be based on the combined relationships of P loading with macrophyte cover, macroinvertebrate richness and euphotic depth and be associated with a meaningful scale for in-lake nutrient concentrations, in this case TLI. It is important to recognise, because most of the CLUES loading data could not be thoroughly validated by inflow monitoring data for most lakes, some caution is suggested in directly applying CLUES load statistics, and further collection of inflow data would be useful in validating annual load predictions.

Overall, results from modelling work provide some valuable information in terms of relationships of N and P loading with key variables related to the EI of shallow lakes. The nature of these relationships can be used to help inform the decisions on loading rates for specific lakes to achieve water quality outcomes for the lake. The nature of these outcomes can then be discussed at a community planning level, taking into account the community's aspirations, and with water quality targets developed for specific waterbodies. This investigation was predominantly focused on ecological indicators related to EI, and did not consider other freshwater uses such as recreational and amenity values as they relate to nutrient limits. As such, this is identified as a gap and possibly an area for further work.

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1. INTRODUCTION

The implementation of the National Policy Statement for Freshwater Management (Ministry for the Environment) has placed a strong imperative for regional authorities to establish and implement limits to protect water quality and ecological values of waterbodies in their region. In Southland, as in many parts of New Zealand, the primary land uses are tied to agriculture. Therefore the management of nutrients are expected to be a critical component of maintaining ecological and other (recreational, aesthetic) values of lakes. As such, understanding ecological responses of waterbodies to nutrient loading is essential in a regional planning context for setting load limits to lakes.

Management of nutrient inputs to waterbodies has been an ongoing priority for management agencies in New Zealand and internationally for decades. However the current focus of establishing load limits to protect ecological values is likely to challenge management authorities because it relies on a thorough understanding of relationships between loading and water quality, and how these water quality values ultimately mediate the ecological integrity (EI) of these waterbodies. Whilst there is reasonable understanding of relationships between loading and in-lake nutrient concentrations, for shallow lakes there is considerable complexity (*e.g.* Moss & Lijklema 1994; Søndergaard *et al.* 2003; Abell *et al.* 2011), including important interactions with key biological components (*e.g.* macrophytes, zooplankton, fish) (Jeppessen *et al.* 2007; Moss 2013).

Coastal lowland areas in Southland tend to be used extensively for agriculture and also contain a number of low-lying shallow coastal lakes within this landscape. Some of these were formed along coastal dune swales that became isolated from the sea centuries ago, others formed more recently as river barrier-bar lagoons. Most of the lakes tend to be located near the terminal end of their drainage areas before they discharge to the sea. As such, they receive nutrient runoff from substantial catchments, and thus are a sink for pollutants eroding and leaching from land. These waterbodies are naturally productive lakes, and often contain highly valued plant and animal species. Because they are located in proximity to urban areas, they are also important for their recreational and cultural values.

This study investigates relationships between nutrient loading and shallow lake ecological values in the Southland area, based on information and scientific knowledge gained nationally and internationally.

1.1. Shallow lake ecology

Shallow lakes respond to nutrients in a different manner to deep lakes, and thus have their own ecology and management challenges (reviewed in Scheffer 1984). As a

result of the high rates of water column mixing, there are frequent interactions between the lake water column and sediments. Phosphorus is generally recognised to be the key nutrient in controlling primary production, because substantial amounts of nitrogen can be lost through denitrification at the sediment surface (Scheffer 2004). Alternatively, P which is stored in lake sediments, can be released to the water column during periods of anoxia or where wave action disturbs pore water from deeper anaerobic sediment layers (Jensen *et al.* 1992a). This process is mediated by water column temperatures, with nutrient (phosphorus) concentrations in shallow lakes often being highest during summer (*e.g.* Gibbs 1994) — the opposite of that usually observed in stratified lakes due to thermal stratification (Scheffer 2004).

Much of the literature relating to understanding the effects of nutrient loading on the ecology of shallow lakes is focused on the restoration of lakes that have experienced historical eutrophication through years of either point source (*e.g.* urban, industrial) or diffuse sources (*e.g.* agricultural) pollution. Thus loading rates and in-lake concentrations in published studies are often high, with much of the knowledge gained from European (England, Denmark, and the Netherlands) and North American (*e.g.* Moss, 1986; Lauridsen *et al.* 2005; Jeppesen *et al.* 2007) lakes. Phosphorus is often considered the primary nutrient to be controlled in shallow lakes as it can be absorbed by the lake sediment and released to the water column sometime later. This internal loading can cause a delay of many years in the equilibration of lake phosphorus concentrations with reductions in external loading (Jeppesen *et al.* 1991, Knuuttila *et al.* 1994, Perrow *et al.* 1994, Søndergaard *et al.* 2005). In some instances, increases in importance of internal P loading relative to external loads can increase in the incidence of cyanobacterial algal blooms when nitrogen concentrations were significantly lowered in relation to phosphorus (Noges *et al.* 2007). Thus, shallow highly polluted lakes can have long recovery periods following a reduction in external phosphorus loading (Bostrom *et al.* 1985, Cullens & Forsberg 1988, Jeppesen *et al.*1991).

1.2. Alternative stable states theory and the importance of macrophytes

Submerged macrophytes are very important structuring elements in lakes (Kelly & McDowall 2004), and may markedly affect the environmental conditions of a lake by their ability to stabilise the clear-water state (Scheffer & Jeppesen 1998). Thus the aim of shallow lake management is often to sustain or restore macrophyte communities to maintain the clear-water state of shallow lakes.

Loss of macrophyte communities from nutrient enrichment of lakes can occur via a number of mechanisms including; phytoplankton blooms, epiphyton growth, or intensive growths of tall macrophytes. All of these mechanisms generally result in shading of macrophyte communities causing light limitation of plants and their ultimate collapse, in a process termed 'flipping' (Schallenberg & Sorrell 2009). In some cases

the loss of macrophytes can be permanent, as internal loading of nutrients from sediments and re-suspension of lake bed materials generating turbidity stabilise the ecosystem in its new turbid-phytoplankton dominated state. In many cases management actions which lead to further reductions in external nutrient loading are counteracted by these internal lake processes, causing an inertia to re-oligotrophication and to the re-establishment of a macrophyte-dominated state (Carpenter 2004, Scheffer 2004).

1.3. Purpose of this report

Environment Southland is considering management targets for nutrient load limits for the region's waterbodies. Shallow coastal lakes in the region tend to occur in highly modified landscapes, and as a result are under threat of degradation from land-use intensification. This report comprises a preliminary assessment of loading rates to shallow lakes in New Zealand, and internationally, in relation to their key ecological indicators.

This study consists of two main parts. The first part consists of a literature review of national and international sources documenting nutrient loading to shallow lakes. The aim of this part of the study was to document ranges of nutrient loading to shallow lakes and their associated ecological conditions. Key in-lake condition indicators were documented (where possible) including; nutrient status, other water quality characteristics (*e.g.* chlorophyll-*a*, water clarity), aquatic macrophytes, and other biological data if relevant (*e.g.* exotic fish, herbivorous fish).

The second part of the study consisted of a modelling approach to assessing nutrient loading to shallow lakes in Southland. A Vollenweider modelling approach was taken, fitting regression models to nutrient loading and water quality data. Limited data availability on morphometry, inflows, and seasonal variability meant it was beyond the scope of this study to develop system-specific deterministic models for the southland lakes with more complex models such as DYRESM. Data sets were confined to non-saline, coastal shallow lakes on the South Island so as to be as similar as possible to Southland's shallow coastal lakes. We used the national CLUES[1] (Catchment Land Use for Environmental Sustainability) model (Woods *et al.* 2007) to predict loads of total phosphorus (TP) and total nitrogen (TN) and this was then related to in-lake water quality and ecological condition data for up to 19 lakes, depending on data availability.

The last section of this report brings together the two main components of the study (literature review and modelling) identifying the key findings, and recommends approaches to setting nutrient load limits for shallow coastal lakes in Southland.

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[1] http://www.mpi.govt.nz/environment-natural-resources/water/clues

2. LITERATURE REVIEW OF NUTRIENT LOADING TO SHALLOW LAKES

2.1. Data sources reviewed

We searched for articles from peer-reviewed journals using several scientific search engines including 'Google Scholar' and the 'Web of Science'. We also searched the references sections of relevant papers and reports.

The key words used in these searches included: polymitic, shallow lake, nutrient loading, phosphorus, nitrogen, macrophytes, and chlorophyll-*a.*

The following sections compile information on various scientific studies for the purpose of understanding relationships and identifying (where they exist) key thresholds for nutrient loading to shallow lakes and their ecological integrity.

2.2. Alderfen Broad

2.2.1. Background

Alderfen Broad — in Norfolk Broadland UK.

2.2.2. Condition

Alderfen Broad is one of 50 or more small lakes (broads) connected by rivers that also transport the sewage of about 400,000 individuals out to the North Sea. This and draining of the farmland has caused extensive eutrophication if the formerly low-nutrient broads. Total phosphorus (TP) concentrations were in the range of 300–500 mg/m^3, and chlorophyll-*a* concentrations in the range of 30–90 mg/m^3.

In 1979, the entire inflow was diverted away from the Alderfen Broad resulting in a series of changes that were observed over a period of 12 years. In the first three years following diversion, TP was reduced to between 30-50 mg/m^3, and chlorophyll was as low as 25 mg/m^3.

2.2.3. Key findings

- After diversion of nutrient-rich flows, Alderfen Broad initially returned to a clear-water state, with the establishment of submerged macrophytes. They mainly comprised very prolific species such as hornwort.

- Internal P then increased — possibly stimulated by senescence of submerged macrophyte beds and a lack of flushing.

- A summer fish kill occurred — probably due to changes in water level (lack of inflow) and increases temperature and lowered dissolved oxygen (DO).

- Macrophyte cover and biomass was cyclical over mutli-year periods, with TP returning to before diversion concentrations (250–750 mg/m^3) during periods of macrophyte dieback. Internal loading of TP was a major factor.
- Decreases in lake water level may have led to further concentration of internal P load.

2.2.4. References

Perrow *et al.* (1994), Moss (1986)

2.3. Danish lakes phosphorus loading reductions

2.3.1. Background

The study focused on monitoring the outcomes of catchment P reductions in eight shallow Danish lakes. These included the lakes; Orn, Byrup, Sogard, Gundsomagle, Arereso, Damhus, Bagsvaerd, and Vesterborg.

2.3.2. Condition

Shallow lakes were between 1.9 and 10.5 m in maximum depth, had areas of between 21 and 4,000 ha, and had residence times between 0.05 and 2.2 y^{-1}.

Nitrogen and phosphorus loads to the lakes were initially very high (1989–92), on average 83 kg P/ha/y and 1,900 kg N/ha/y. This corresponds with supertrophic conditions with 556 mg/m^3 TP and 7,650 mg/m^3 TN (on average). There were no macrophytes present in any of the eight lakes.

Phosphorus loads were reduced by 50% to 41 kg P/ha/y on average, whereas N loading rates remained more or less constant.

2.3.3. Key findings

- The 50% reduction in P loading decreased in-lake TP by almost 75% to 126 mg/m^3 and mean summer chlorophyll was halved from approximately 150 mg/m^3 to 75 mg/m^3. This is still considered very high by comparison to New Zealand lakes, and represents hypertrophic conditions. Total nitrogen concentrations decreased slightly to on average 5,600 mg/m^3.
- Reductions in in-lake TP were considerably greater than predicted by mass balance equations, and internal loading of sediment-bound P was found to be a major factor for internal loading of P during mid-summer periods.
- No recovery of macrophytes was recorded in the lakes at these concentration ranges, even some 10 years following loading reductions.

- Lakes that were biomanipulated to enhance grazing cladoceran zooplankton populations (usually by adding piscivorous fish populations) tended to be the only lakes in which macrophytes were re-established. Maintenance of euphotic depth of > 2 m was an important driver of macrophyte re-establishment.

2.3.4. References

Søndergaard, Jensen and Jeppesen (2005), Lauidsen *et al.* (2003)

2.4. Lake Søbygaard, Denmark

2.4.1. Background

Lake Søbygaard is a small, shallow, hypertrophic lake (mean depth 1.0 m; area 40 ha) in Denmark. Mean hydraulic residence times in the lake vary from 15-20 days in summer and 25-30 days in winter. Up until 1982, Lake Søbygaard received large amounts of wastewater resulting in the accumulation of P-rich sediments.

2.4.2. Condition

Søndergaard *et al.* (1993) reported on the loadings in the lake eight years after 1982, when external P loading was reduced by 80–90%. Jensen *et al.* (1992a) indicated that Fe:P ratio can influence the release of P from sediments. Iron (Fe) can suppress P release when the ratio (by weight) is more than 15–20.

2.4.3. Key findings

- External loads of P were historically (1978-82) between 250–330 kg/ha/y, resulting in a hypertrophic lake with serious algal blooms.
- External P loads were reduced between 1983–90 to between 30–60 kg/ha/y.
- Following external load reductions, P release from sediments mainly occurred during summer (April–October).
- Net internal P release initially was 80 kg/ha/y but decreased to 20 kg/ha/y by 1990, as sediment P was depleted.
- Maximum sediment P concentration was 11.3 mg P g^{-1} dw at a depth of 14–16 cm in 1985. In 1991 this had decreased to 8.6 mg P g^{-1} dw at a depth of 16–18 cm.
- Highly negative (-8 to -2) P-retention coefficients were observed after the major decline in external P loading, indicating an export of sediment P from the lake.
- No macrophyte regrowth in the lake has been reported.

2.4.4. References

Jensen *et al.* (1992a) and Søndergaard *et al.* (1993)

2.5. Danish shallow lake mesocosm experiments on nutrient loads

2.5.1. Background

Experiments were conducted in mesocosms in shallow lakes, where they were enriched with P, N , or combined N and P at varying levels of enrichment. This was to determine the effects on water quality, chlorophyll-*a*, and macrophyte biomass under a gradient of enrichments. The results were then compared to a broad-scale assessment of nutrient concentrations in Danish lakes (as above) to assess whether nutrient treatment effects were realistic and whether mesocosm responses were scaleable to lake-level observations.

2.5.2. Condition

The experiment was conducted in Lake Stigholm (Denmark); a shallow lake with an area of 21 ha and a mean depth of 1.2 m. Total phosphorus and TN were 100 and 2,000 mg/m^3, respectively (on average) and aquatic macrophytes in the lake were variable from year to year.

Mesocosm TP concentration was either initiated at background concentrations (100 mg/m^3) or increased to 200 mg/m^3, and TN was either 2,000 (low), 4000 (intermediate), or 10,000 (high) mg/m^3 at the initiation of the experiment.

2.5.3. Key findings

- Only combined N and P nutrient addition treatments had pronounced effects on chlorophyll-*a* biomass, water clarity or macrophyte biomass. This resulted in significant declines in macrophytes and an overall increase in chlorophyll-*a* and TP and TN concentrations.
- Most importantly, the shift in ecology of the mesocosms from a clear macrophyte dominated state, to a turbid phytoplankton-dominated state, occurred at TN concentrations > 2000 mg/m^3 and at TP concentrations > 130 mg/m^3.
- These results aligned reasonably well with broad-scale survey data of macrophyte dominance in Danish lakes (Søndergaard, Jensen & Jeppesen 2005).

2.5.4. References

Gonzales Sogrario *et al.* (2005)

2.6. Lakes Peipsi and Vortsjärv

2.6.1. Background

Lake Peipsi is a large, shallow lake (mean depth 7.1 m; area 355,500 ha) that borders Estonia and Russia. The mean hydraulic residence time in the lake is approximately two years.

Lake Vortsjärv is a large, shallow (mean depth 2.8 m; 27,000 ha), polymictic eutrophic lake in Estonia. The mean hydraulic residence time in the lake is approximately one year.

2.6.2. Condition

Lake Peipsi is made up of three sub-basins: Lakes Peipsi, Lämmijärv, and Pihkva. Peipsi sub-basin is an polymictic eutrophic lake (mean depth 8.3 m; area 261,100 ha), Lämmijärv is a dystrophic lake (mean depth 2.5 m; area 23,600 ha), and Pihkva is a eutrophic-hypertrophic lake (mean depth 3.8 m; area 70,800 ha). On a 'whole lake annual mean' basis, TN and TP concentrations in the Lake Peipsi were 768 and 42 mg/m^3, respectively.

More than 80% of N and P compounds entering Lake Peipsi are from rivers carrying biologically-treated wastewaters for two townships (a Russian town ~ 200,000 inhabitants, and an Estonian town ~100,000 inhabitants). Monitoring of inflow concentrations was conducted monthly for multiple years (no flood chasing; loading calculations were according to Grasshoff *et al.* 1982).

Loadings between 1994 and 2002 were approximately 8,000 t N/y and 275 t P/y corresponding to inflow concentrations of 1,200 mg/m^3 TN and 55 mg/m^3 TP.

Changes (reductions) of riverine loadings of nutrients (nitrogen in particular) have occurred in both lakes as a result of the collapse of intensive agriculture in the 1990s.

2.6.3. Key findings

- In Lake Vortsjärv both N and P concentrations followed the decreasing trends of loading, which showed the high sensitivity of these large shallow lakes to catchment processes. Total nitrogen concentrations have decreased from 1,000 to 750 mg/m^3 over a 5-year period. Total phosphorus concentrations declined from 80 to 50 mg/m^3 over the same period.
- The study showed a positive relationship between P content in sediments and the relative depth of Lake Vortsjärv.
- The N:P ratio decreased in both lakes.
- High chlorophyll and cyanobacterial blooms that were common in both lakes at the beginning of the 20th century disappeared during heavy nitrogen loading in the

1980s but started again in Lake Peipsi in recent years together with the decrease in the N:P ratio. The present ecological status seems to be mostly controlled by climatic factors via changes of water level.

- The N:P ratio was higher and the ecosystem was more stable in Lake Vortsjärv, although the share of N_2-fixing cyanobacteria has increased since the 1990s.

- Nutrient loading has decreased and water quality has improved in Lake Vortsjärv. The most important measure to improve water quality in L. Peipsi was the reduction of phosphorus loading from both Estonian and Russian sub-catchments

2.6.4. References

Noges *et al.* (2007)

2.7. Review of nutrient loading reductions to shallow lakes in the Northern Hemisphere

2.7.1. Background

This study (Jeppesen *et al.* 2007) focused on nutrient loading in relation to key ecological variables in lakes, including chlorophyll concentrations and macrophyte cover. It examined critical questions around the importance of time-lags following reductions in external loads, and the importance of N *vs.* P, reductions; the latter is normally the focus of nutrient load management.

2.7.2. Condition

A range of conditions, with data for hundreds of lakes synthesised in the review.

2.7.3. Key findings

- Nitrogen loading is usually considered to be less important for load management to shallow lakes, though there is some recent evidence suggesting the importance of nitrogen in regulating the abundance and species richness of macrophyte communities when P is already at moderately high levels.

- The authors plotted macrophyte cover in 44 Danish lakes (over several years, 246 lake years) showing macrophyte cover in relation to in-lake TN concentrations (Figure 1). This analysis suggests that lakes with TN concentrations greater than 1,000 mg/m^3 would have almost never have macrophyte cover > 50%, and lakes > 2000 mg/m^3 never had macrophyte cover > 5%.
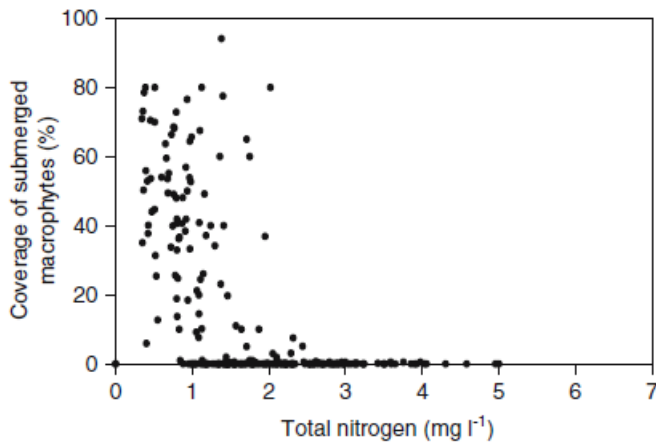
Figure 1.   Macrophyte cover of the lake bed in relation to in-lake total nitrogen (TN) concentrations in 44 lakes (246 lake years). All lakes were < 5 m deep and had lake areas of > 5 ha. (From Jeppesen *et al.* 2007).

- The cover of macrophytes in relation to in-lake TN, TP and chlorophyll-*a* concentrations was compared between two sets of shallow lakes from Denmark (cool temperate) and Florida (warm temperate). Findings suggest that there were differences in the range of N and P concentrations related to macrophyte cover in the two bio-climatic regions. In warmer climates, macrophyte cover was greater for given TN and TP range (Figure 2).

- In Denmark, a critical threshold for TP occurred around 100 mg/m^3, such that very few lakes above this threshold had macrophytes covering > 50% of the lake bed, whereas in the Florida lakes, this threshold was closer to 50 mg/m^3.

- For TN, a threshold occurred around 1,000 mg/m^3 for both the Danish and Florida lakes, above which few lakes had macrophyte cover exceeding 50%.
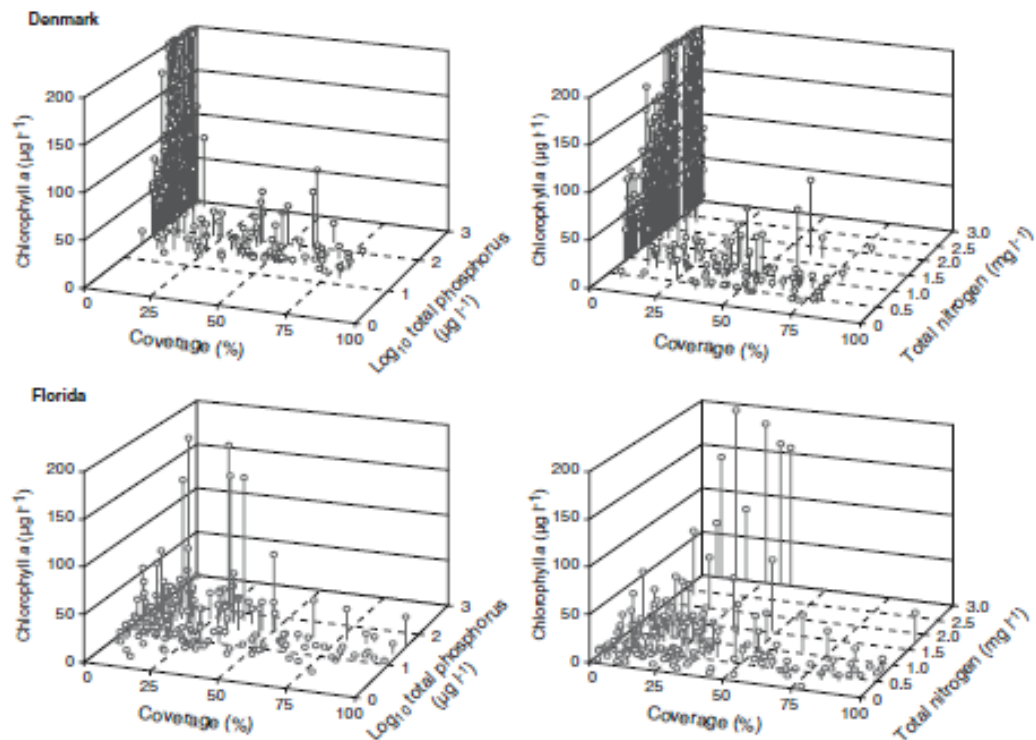
Figure 2.   Three-dimensional plots showing the relationship between macrophyte cover, chlorophyll-a, and total nitrogen (TN) and total phosphorus (TP) in a lake-set of both Danish and Florida lakes (From Jeppesen *et al.* 2007).

2.7.4. References

Jeppesen *et al.* (2007)

2.8. Lake Horowhenua

2.8.1. Background

Lake Horowhenua is a small, shallow (depth of < 2 m; area 2,900 ha), hypertrophic coastal dune lake on the west coast of the North Island in New Zealand. Mean annual residence time in the lake is approximately 47 days.

2.8.2. Condition

Lake Horowhenua receives runoff from intensive agriculture within its catchment and until 1987, treated sewage effluent from the town of Levin (population ~15,000). Consequently the lake is highly enriched with an annual cycle of algal phosphorus limitation in winter and nitrogen limitation in summer.

External inputs of P and N, based on spot nutrient samples, were estimated to be 3.1 and 203 t/y, respectively, with internal loading from sediments estimated to be 5.8 and 30 t/y respectively. This corresponded to inflow concentrations ranging between 35–500 mg TP/m^3 and 5,800–8,000 mg TN/m^3.

Lake conditions are highly turbid and phytoplankton-dominated with considerable cyanobacterial blooms. High growth rates and biomass of aquatic macrophytes are primarily driven by high spring TN concentrations, often with a collapse of macrophytes in summer when biomass reaches its maximum (Gibbs 2011).

2.8.3. Key findings

- Considerable denitrification results in almost 50% of the N load being lost to the atmosphere.
- Total nitrogen concentrations are highest in winter with winter in-lake concentrations reaching 4,000 mg/m^3, summer concentrations were between 1,200—3,000 mg/m^3.
- Total phosphorus concentrations due to internal loading peak in summer at 600-800 mg/m^3, and winter concentrations are close to 100 mg/m^3.
- Management of external and internal P loading sources is viewed to be critical to controling summer cyanobacterial blooms in the lake, as these blooms are capable of fixing N if it is limiting at the time.

2.8.4. References

Gibbs & White (1994), Gibbs (2011)

2.9. Lakes Kotojärvi and Villikkalanjärvi

2.9.1. Background

Lake Kotojärvi is a small, shallow (mean depth 2.5 m, maximum depth 4.1 m; area 30 ha) lake in southern Finland. The theoretical retention time is 160 days. Agricultural land accounts for 22 % of the total drainage basin area (565 ha).

Lake Vilikkalanjärvi is a 710 ha, shallow (mean depth 3.2 m, maximum depth 10 m), lake in southern Finland. The theoretical retention time is 67 days. Agricultural land accounts for 33 % of the total drainage basin area (41,000 ha).

Of the total anthropogenic nutrient input into the lakes, 90% of P and 85% of N is of agricultural origin.

Loading rates are based on mass balance calculations of measured nutrient loads, which are based on inflows and outflows from routine monthly monitoring with some high flow samples included. Loading rates were estimate to be as follows: Lake Villikkalanjarvi (TP — 10,950 kg P/ha/y, TN — 200,750 kg N/ha/y) and Lake Kotojarvi (TP — 2,950 kg P/ha/y, TN — 32,850 kg N/ha/y).

This almost seems erroneously high and possibly the authors used incorrect units (*i.e.* it was reported in $g/m^2/d$ in the manuscript).

2.9.2. Condition

Most of the agriculturally impacted Finnish lakes are small, shallow and characterised by high turbidity due to erosion of clay-sized particles from agricultural land. Suspended solids may influence the availability of phosphorus through buffering and reductions to the euphotic depth. Both lakes have high turbidity and cyanobacterial blooms.

Nutrient concentrations were incredibly high, with summer dissolved reactive phosphorus (DRP) concentrations > 40 mg/m^3 and winter nitrate concentrations > 3,000 mg/m^3 in Lake Villikkalanjarvi. In Lake Kotojarvi DRP concentrations were as high as 20 mg/m^3 and winter nitrate concentrations > 800 mg/m^3.

2.9.3. Key findings

- About half of the TP and a third of the TN load was retained in Lake Kotojärvi.
- About 24% of the TP and 19% of the TN load was retained in Lake Vilikkalanjärvi.
- These differences were probably due to the difference in theoretical retention times between the lakes.
- The mean TN and TP was higher in Lake Vilikkalanjärvi than Lake Kotojärvi, however the reverse was true for mean chlorophyll-*a* concentration. This was considered to be due to differences in the internal P load, which was considered to be as much as two-fold the external load in Lake Kotojärvi, but up to only 50% of the external load in Lake Vilikkalanjärvi.
- In lake Kotojärvi, the high internal P load coupled with a low DIN:DRP ratio resulted in a strong cyanobacterial bloom in the summer of 1988, while in Lake Vilikkalanjärvi only small amounts of cyanobacteria were observed.

2.9.4. References

Knuuttila *et al.* (1994)

3. TESTING OF MODEL PREDICTIONS OF NUTRIENT LOADING TO SOUTHLAND COASTAL LAKES

Considerable scientific investigation has been conducted into relating in-lake water quality conditions with catchment loading (Vollenweider 1976; Vollenweider 1982). Vollenweider's models relate the inflow nutrient loads with in-lake water quality, taking into account properties known to be related to nutrient cycling including the lake volume, depth and water residence time for the lake. Nutrient budget studies are generally conducted over multiple years to inform nutrient inflow and outflows from lakes, and their seasonal variability. These studies are often conducted over multiple years to account for variability in climate and runoff, and thus provide quite detailed examinations of nutrient loads to shallow lakes.

Presently, the extent of water quality data to inform models of nutrient loads to smaller shallow coastal lakes in Southland (and their inflowing tributaries) is limited. Moreover, the catchments of these systems are often highly modified and thus considerable nutrient enrichment of runoff occurs. Whilst there are several coastal lakes for which water quality and ecological data (*e.g.* chlorophyll-*a*, macrophyte cover, macroinvertebrates) has been collected to document ecological condition, there is very limited data on inflow nutrient concentration and flows to inform the calculation of annual N and P loads. Therefore an initial approach of testing the use of a broad-scale predictive tool such as the CLUES nutrient loading model, in relation to in-lake water quality monitoring data, was thought the best approach to examining relationships between nutrient loading and ecological integrity in Southland lakes.

3.1. Calculating nutrient loads to lakes

Annual loads of TN and TP to the lakes were estimated from a nutrient transport model combined with the regionally-based hydrological regression model, CLUES (Woods *et al.* 2007). Total nitrogen and TP loadings generated by this model reflect the effects of various land uses such as production forestry, low-intensity grazing, high-intensity grazing, dairy farming, horticulture and urban development and taking into account upstream retention by lakes and wetlands.

Values of TN and TP in tonnes per annum were the main output of the CLUES model, which were summed for each tributary inflow of each lake. Mean annual inflow TN and TP concentrations were also calculated by flow weighting of each of the tributary inflows by the mean annual flow obtained from the CLUES model hydroedge function. Mean annual aerial loads of TN and TP were calculated by dividing the total annual load by the area of the lake.

Unfortunately CLUES modules have not been developed for Stewart Island, so other options were pursued to investigate relationships between in-lake nutrient

concentrations versus N and P loading for this area. This included flow weighting mean annual TN and TP loading rates from other 'reference condition' lakes, but this resulted in significantly higher than expected loading rates. After evaluating several options, we used the average catchment area yield of N and P from other reference lakes in the data set multiplied by the catchment area to calculate a mean annual loading rates. This was then divided by the mean annual inflow volume data (from the Freshwater Ecosystems of NZ [FENZ]) geo-database) to determine inflow concentrations. These results provided reasonable correlation between loading rates and in-lake concentrations, but have been calculated in a different manner than other mainland lakes and thus need to be considered in this context.

3.1.1. *Calculating relationships of nutrient loading with in-lake nutrient concentrations*

To test and verify the likely accuracy of the predicted nutrient loadings from the CLUES model and their applicability to lowland lakes in the Southland region, the model predictions were correlated against in-lake water quality conditions in 18 shallow coastal lakes on the South Island for which water quality data was available. This included six lakes from the Southland region and a further 12 lakes for which monitoring data was collected as part of a national coastal lake survey (Drake *et al.* 2010). These lakes were monitored on a single occasion between 2004 and 2006, with a consistent survey approach (seasonality) and sampling methods.

Vollenweider type models were used to relate the predicted inflow nutrient loading rates to in-lake TN and TP concentrations. Vollenweider (1982) found that annual average TP and TN concentrations in lakes (TP_{lake} and TN_{lake} in mg m^{-3}) could be estimated from lake flushing rates and inflow concentrations:

$$TP_{Lake} = a\, [TP_{inflow} / [1+ (\tau)^{0.5}]\,^{b}$$

$$TP_{Lake} = a\, [TP_{inflow} / [1+ (\tau)^{0.5}]\,^{b}$$

Where TP_{Inflow} and TN_{Inflow} are the annual average inflow concentrations of P and N, respectively (mg m^{-3}), and τ is the hydraulic retention time of the lake (y). TP_{inflow} and TN_{inflow} were derived from the flow-weighted average nutrient concentrations derived from the CLUES catchment model (see above) which was also used to derive an annual mean discharge to the lake. The multiplier (a) and exponent (b) terms for the functions were optimised for the 18 South Island lakes in a non-linear regression model in R using the measured values of TN_{Lake} and TP_{Lake} from monitoring data (averaged for multiple years). Residuals were calculated between predicted annual average lake nutrient concentrations and observed values as an indicator of internal nutrient loading (*i.e.* positive residuals may reflect the ability of the water body to

buffer external nutrient loading whereas negative residuals indicate that net internal loading occurs).

For TP, an alternative previously published Vollenweider-type model was also considered from (Brett and Benjamin 2008) that included a term for lake mean depth, as below.

$$TP_{Lake} = TP_{inflow} / [1+ (v\ \tau\ / Z_{mean})]$$

Where v is a constant optimised to fit the TP_{lake} data for the 19 South Island lakes.

Parameters used in calibrating Vollenweider models, including lake volume, hydraulic residence time (τ), mean depth (Z_{mean}), and fetch were obtained from the FENZ geo-database (Leathwick *et al.* 2010), and were modified where more accurate data (usually depth and volume) were available.

3.1.2. Nutrient retention

Net nutrient retention rates (*i.e.* the portion of nutrient inflows retained in the lake basin) are often calculated by using estimates of particle settling times and water residence time. The retention factor for phosphorus (R_P) was calculated using the retention model of Vollenweider (1976):

$$R_P = \frac{v_p}{(v_p+q_a)}$$

where v_P is the apparent settling velocity of P (m y^{-1}) and q_a is the areal water load in m y^{-1}. However because we were lacking information on settling rates for any of the study lakes, we estimated retention using relationships between inflow and outflow nutrient concentrations. Thus we used the predicted retention by the equation

$$R_{TP} = [Mean\ TP_{inflow}]/ [TP_{Lake \times} Q_{out}]$$
$$R_{TN} = [Mean\ TN_{inflow}]/ [TN_{Lake \times} Q_{out}]$$

Where mean TP_{inflow} and TN_{inflow} is the flow-weighted inflow concentration calculated in CLUES, TP_{lake} and TN N $_{Lake}$ are the surface TN or TP concentrations of lake water calculated as the mean of measured data for lake surface samples,
Q_{out} is the lake surface outflow calculated using the sum of the data of the 'MEANQ' field in the 'HYDROEDGE' layer in CLUES for the lake outlet stream reach.

3.2. Lake ecological integrity response variables tested with loading

A number of physico-chemical and EI indicators were selected as a basis for assessing relationships with nutrient loading (Schallenberg *et al.* 2011). Physico-chemical data for the lakes were obtained from monitoring of 19 South Island lakes

between 2004 and 2013 (Drake *et al.* 2010; Schallenberg & Kelly 2012, 2013; Figure 3); and included concentrations of total phosphorus, total nitrogen, chlorophyll-*a*, and depth of the euphotic zone (1% irradiance) measured as 4.6 divided by the diffuse light (PAR) attenuation coefficient (k_d). Mean values were calculated for measurements taken between 2004 and 2013 (Table 1).

The TLI (Burns *et al.,* 2000) has been widely used in New Zealand to determine the trophic state of lakes. Values of TLI were determined from annual mean surface water concentrations of chlorophyll-*a*, total nitrogen and total phosphorus (TLI3) and excluded secchi depth.

Macroinvertebrate and submerged macrophytes community data were compiled from 18 of 19 lakes from sampling surveys between 2004–2013 from Drake *et al.* (2010) and Schallenberg and Kelly (2012, 2013). This included approximately 26 species of submerged macrophytes and 70 macro invertebrate taxa from the 18 lakes. Species counts were extracted from the latest or most complete survey results available for each lake.

Table 1.   Lake morphometric data, CLUES nutrient loads, and total nutrient concentrations for 19 shallow coastal South Island lakes used in the nutrient loading modelling study.

Lake	Number of samples (n)	Lake Area (ha)	Max. depth (m)	Hydraulic Retention time (τ) (y^{-1})	CLUES P-Load (kg/ha/y)	CLUES N-Load (kg/ha/y)	Mean in-lake total-P (mg/m^3)	Mean in-lake total-N (mg/m^3)
Southland								
George	4	91	2.0	0.052	6.1	49.6	57	490
Murihiku	2	6	1.3	0.078	20.2	214.9	235	2093
Vincent	4	17	5.0	0.134	3.7	158.8	21	600
Brunton	1	26	3.3	0.026	32.5	488.6	27	595
Reservoir	4	36	5.0	0.152	7.6	86.8	30	576
Sheila	2	14	6.6	0.082	1.6	39.9	11	253
Calder	1	4	6.7	0.188	1.7	41.7	7	220
West Coast								
Mahinapua	1	394	10.0	0.175	2.5	15.7	10	323
Poerua	1	213	7.8	0.097	12.3	47.5	8	245
Ryan	1	3	3.0	0.041	17.2	75.4	66	696
Ship Creek	1	10	3.0	0.022	11.2	86.0	6	260
Maori	1	37	0.6	0.004	5.5	63.4	4	228
Otago								
Tuakitoto	2	132	3.0	0.046	10.4	94.5	54	952
Waihola	48	608	2.2	0.324	2.6	11.4	19	251
Waipori	26	184	1.0	0.005	40.7	421.7	17	255
Canterbury								
Coopers	2	43	3.0	0.497	0.1	0.9	16	1381
Rotorua	2	2	3.2	0.858	1.6	29.5	270	3672
Tasman								
Kaihoka (2)	1	7	10.2	1.191	2.1	12.7	7	151
Otuhie	1	85	2.1	0.053	4.3	53.6	7	235
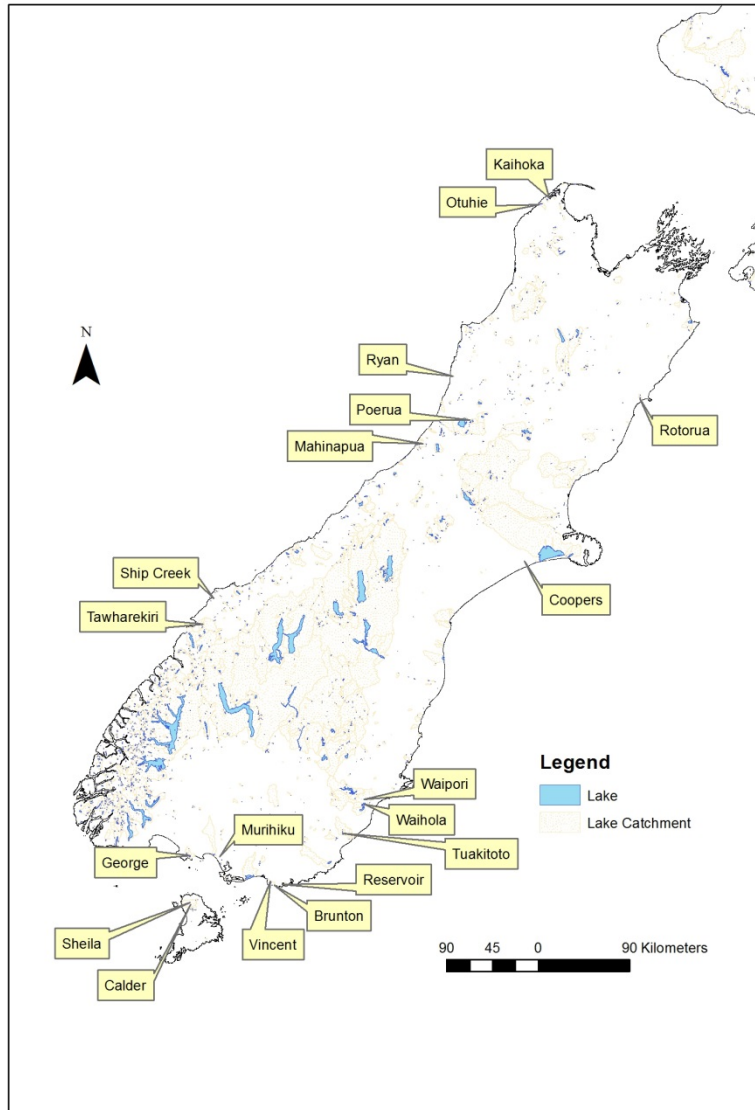
Figure 3.   All South Island lakes (>1 ha in area) and their catchments showing the 19 shallow
coastal lakes included in the nutrient loading model analyses.

3.3. Catchment Land Use for Environmental Sustainability model water quality comparisons

3.3.1. Nutrient loading water quality relationships

Catchment Land Use for Environmental Sustainability (CLUES) model predictions of
flow-weighted TP inflow concentrations were highly correlated with in-lake TP
concentrations in the 19 lake data set (Figure 4a). In this regression, predicted inflow
TP concentrations mostly exceeded in-lake phosphorus, as demonstrated by most
points falling below the 1:1 fit line. This was evident for all of the Southland lakes
except for Lake Murihiku, which had TP concentrations at least 3-fold higher than any

other lake. Higher loading to in-lake concentrations occurred indicating some degree of in-lake processing of phosphorus and a small degree of P retention. Three lakes had in-lake TP concentrations which greatly exceeded the predicted tributary inflow concentrations (Lakes Ryan, Taukitoto, and Waipori), which had some of the highest TP concentrations in the lake-set. This could be related to either an under-prediction of TP export by CLUES in catchments of lakes with higher TP concentrations, or the contribution of internal P loading (*e.g.* resuspension from sediments). Two of three of these lakes (Lakes Ryan and Tuakitoto) also had TN concentrations well in excess of the predicted CLUES inflow, suggesting that the CLUES catchment model may have under-predicted actual loads. Lake Rotorua, in North Canterbury, was removed from the regression model as an outlier due to its much greater in-lake P concentrations than predicted. Further work on this lake has revealed obvious diurnal fluctuations in dissolved oxygen concentrations (D. Kelly unpublished data), which could result in P release from lake sediments during low DO periods.

Similar trends were observed in regression of predicted tributary TN concentrations with in-lake TN concentrations, however, this relationship was weaker (Figure 4b). As with the CLUES-based tributary TP predictions, CLUES-based TN tributary predictions in most cases exceeded in-lake TN concentrations, suggesting the potential for dentrification to have occurred. A similar number of lakes also fell above this 1:1 fit line, which suggests either that in-lake nitrogen fixation is important in these lakes, or that the CLUES model under-estimated the tributary TN concentration. The weaker performance of the N model is not entirely unexpected, as in-lake processing of N (*e.g.* bacterial denitrification and N-fixation) is more complex than for P, and could contribute to the weaker correspondence between predicted tributary N concentrations and in-lake N concentrations.

On an overall basis, the level of statistical significance of the TN and TP regression relationships would lend confidence to predictions being made by the CLUES model. Such a direct correlation between in-lake TP and TN concentrations and the CLUES-derived inflow concentrations was somewhat unexpected. This suggests relatively little in-lake processing of N and P in the lakes included in this study. However because nutrient removal processes such as uptake by aquatic plants, bacterial denitrification, and particle settling can be counteracted by internal nutrient flux processes such as sediment resuspension and nitrogen fixation, it is possible that these sets of processes balance themselves to some degree. The very short residence times of these small, shallow study lakes, being on average 0.21 years, indicate that in these lakes there is little time for internal nutrient processing interactions to alter in-lake concentrations greatly from the inflow concentrations.

a) CLUES P Loading

b) CLUES N Loading
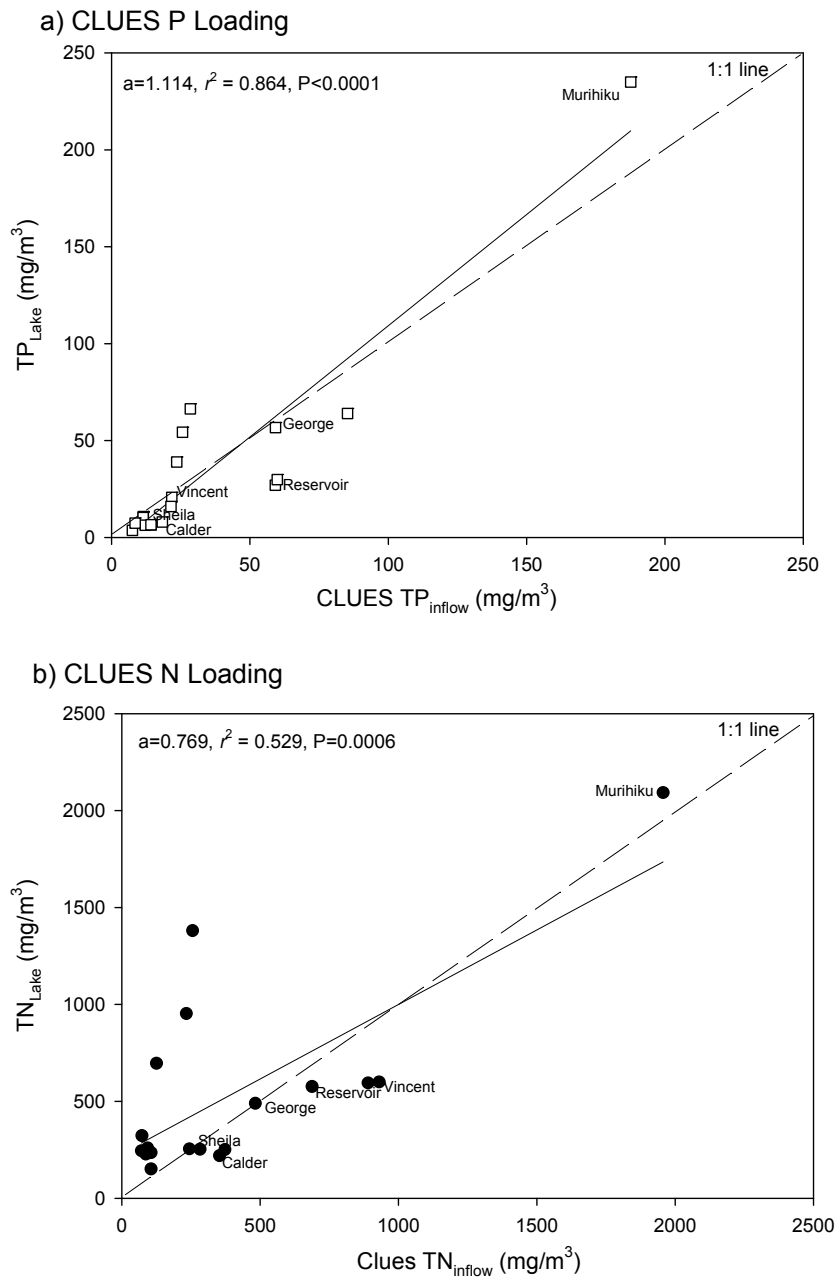
Figure 4.   Linear regressions of predicted mean CLUES flow-weighted inflow total nitrogen (TN) and total phosphorus (TP) concentrations compared with in-lake TN and TP measurements in 18 South Island shallow coastal lakes. Note that one lake (Lake Rotorua — Kaikoura) was removed from the plot due to it being an outlier in the regression model.

Vollenweider models that related CLUES TP_{inflow} concentrations to in-lake TP concentrations moderately improved the relationships between predicted loading and in-lake concentrations. Although all models were significant, even the most-simple model in which loading was corrected by the square-root of water residence time provided relatively good correlation. Other more complex models using exponent and multiplier terms (sensu Vollenweider 1982) had stronger regression r^2 coefficients, but

in some cases slopes were usually steeper than 1:1, indicating some slope bias and a tendency to under-predict in high TP lakes and over-predict in low TP ranges (Figure 5b). A model previously published by Brett and Benjamin (2008) that also incorporated mean depth performed reasonably well in predicting in-lake TP (Figure 5c). A significant polynomial multiple regression model incorporating TP$_{inflow}$ concentration and residence time was found to have a very strong correlation (Figure 5d). As the model was fit using a polynomial regression to our data, its applicability to lakes outside of this data set is uncertain.
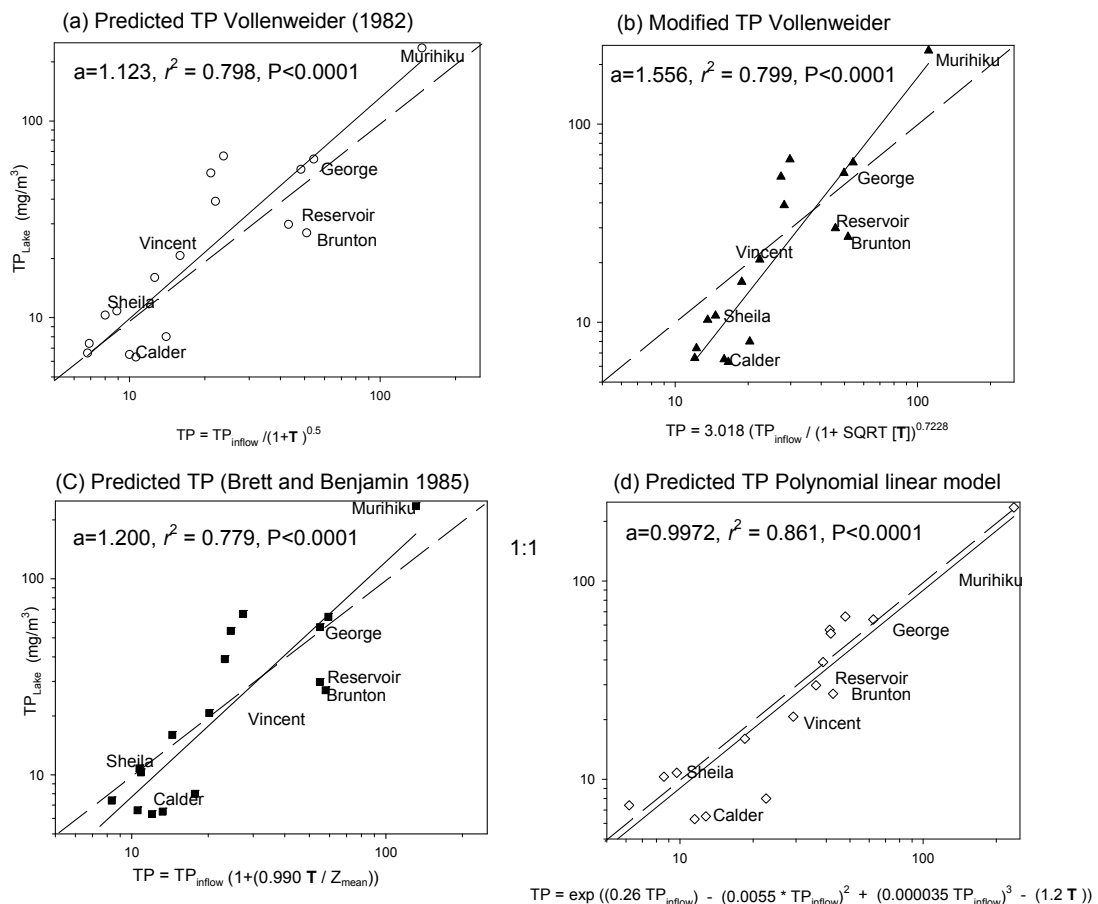
Figure 5.   Relationships between in-lake total phosphorus (TP) concentrations and four Vollenweider models predicting TP concentrations based on CLUES model TP inflow concentrations for 18 South Island shallow lakes. Note that one lake, Lake Rotorua, was omitted from the analyses as an outlier.

Vollenweider models relating CLUES predicted TN loading to in-lake TN concentrations did not perform as well as TP models. Using previously published models (*e.g.* Vollenweider 1982) tended to under-predict in-lake TN concentrations (Figure 6a), suggesting either that the degree of in-lake N uptake and denitrification was much lower for this lake-set, or that the CLUES model is under-

predicting N loss from catchments. The optimized Vollenweider model showed a moderate improvement in variance explained by the model (r^2 =0.360), but had considerable slope bias (slope=2.046) which under-predicted loading at higher TN ranges, and over-predicted at low TN. A linear model was fitted to the data that incorporated TN_{inflow}, residence time and mean depth, was highly correlated to in-lake TN (Figure 6c). Given that this multiple linear regression model was developed with the test data only, its wider applicability is uncertain.
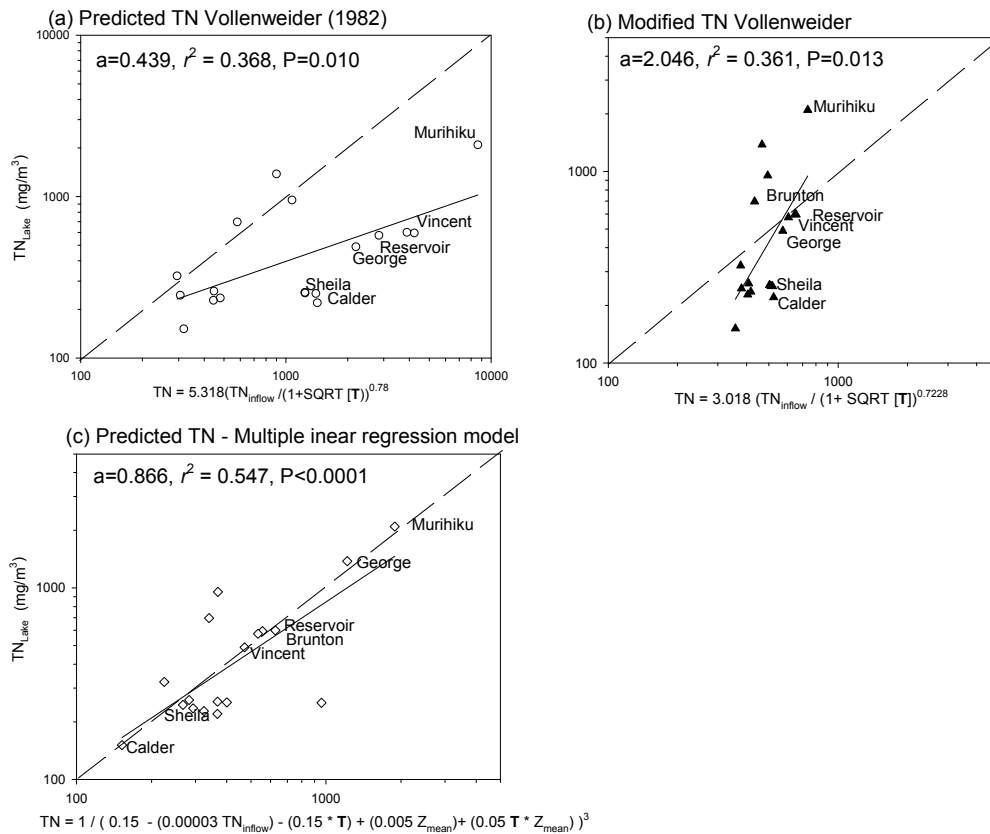
Figure 6.   Relationships between in-lake total nitrogen (TN) concentrations and three Vollenweider model predicted TN concentrations based on CLUES model TN inflow concentrations for 18 South Island shallow lakes. Note that one lake, Lake Rotorua, was omitted from the analyses as an outlier.

As discussed previously, the very short residence times of the small, shallow lakes in this study would tend to reduce in-lake processing of N, which could account for some of the difference between previously published models which included some deep lakes with much longer residence (Figure 6a). For three of the Southland lakes, we also compared predicted CLUES inflow TN and TP concentrations to inflow nutrient data where data were available. Total nitrogen inflow concentrations were more often than not under-predicted by the CLUES model (see Table 2), in some cases as much as 70% lower than measured values. This would suggest that TN loading may be

higher than is being accounted for in the CLUES model, keeping in mind that this hypothesis is based on a very small number of sampling occasions. In contrast, CLUES TP predictions were, more often than not, higher than measured values. As these are mean loading values, direct comparison to spot measures is somewhat more complex, but it would be probable that TP would be influenced by sediment transport during floods. Thus it is probable that CLUES estimates should exceed measured TP values, and this was the case for three or five tributary sites. The east (and main) inflow to Lake Vincent was the most obvious to diverge from this trend, where predicted mean loads were only 33% of measured values, thus CLUES has probably underestimated TP and TN loading significantly for this lake. It is important to recognise that all of these findings are based on a very small number of tributary monitoring data, and thus provide only a preliminary indication of CLUES model performance.

Table 2.   Mean inflow total nitrogen (TN) and total phosphorus (TP) concentrations and their associated predicted CLUES TN and TP concentrations for five coastal lakes tributaries in the Southland region.

Site	Sampling events (n)	Mean TN (mg/m^3)	Mean TP (mg/m^3)	CLUES model TN (mg/m^3)	CLUES model TP (mg/m^3)
L. Vincent East inflow	6	1760	60	914	20
L. Vincent North inflow	1	540	30	817	38
L. George North Inflow	2	350	20	390	40
L. George South inflow	1	1360	90	1022	57
The Reservoir North Inflow	2	2210	50	624	72

3.3.2. Nutrient retention

Modelled net phosphorus retention coefficients for the 19 South Island lakes ranged between -0.7 and 0.85, with median and mean rates of 0.46 and 0.26, respectively (Figure 7a). Lake Rotorua was not included in the analysis due to it being considered an outlier, with an estimated retention coefficient of -316. Such a large negative value, indicating the lake has much greater TP concentration than inflows, indicates that the lake is likely to be affected by internal P flux from sediments.

Comparison of P retention coefficients from this study to previously published findings suggest very comparable results (Figure 7), with values mostly ranging from -0.7 to 0.8, and median values around 0.5 (Brett & Benjamin 2008, Kiov *et al.* 2011). These studies considered a much larger number of lakes (including deeper lakes), so it is encouraging that retention coefficients calculated from CLUES predictive models are

in-line with those, which were often derived from more in-depth nutrient budget studies. As expected, retention values tended to be lower in more eutrophic lakes, indicating as lakes become more eutrophic they retain a lower proportion of nutrients. However there was considerable variation, with low retention coefficients modelled for some low nutrient lakes, such as Lake Mahinapua (Westland).

Phosphorus retention coefficients for Southland lakes were all >0 indicating moderate retention which generally decreased with increasing nutrient status of lakes. Positive moderate retention coefficients for the Southland lakes (0.04-0.73) suggest that most in-lake P could be accounted for by external sources, and thus internal loading sources are less likely to comprise a significant component of the loading. Other hypertrophic South Island lakes such as Lakes Rotorua, Ryan, and Tuakitoto had negative retention coefficients indicating internal load sources are likely to be significant. Lake Sheila had a lower retention than would have been expected (0.061) by comparison to other more eutrophic lakes. Because the CLUES model was not available for Stewart Island lakes, we used areal loading rates from other West Coast reference lakes to calculate mean annual loading rates. However, as a result of this, P loading may have been underestimated, which resulted in a lower than expected retention coefficient. Alternatively Lake Calder was estimated in a similar fashion, but would have P retention coefficients more in line with expectations at 0.64.

Nitrogen retention coefficients were lower overall than for P, and had a greater number of lakes with negative values, indicating these lakes to be overall net sources of N. Median and mean values for the 18 lake data set (excluding Rotorua) were 0.04, and -0.55 respectively. Retention coefficients can be affected by N fixation in lakes, thus any lakes with cyanobacterial dominated phytoplankton communities could contribute to lower or negative retention coefficients. For example, N retention for Lake Rotorua was -235 and had a cyanobacterial bloom at the time of sampling, showing how sensitive these ratios can be to such occurrences. Other lakes which had highly negative (< -1) values included more eutrophic lakes such as Lakes Tuakitoto and Ryan, but also Coopers Lagoon (TLI = 3.9) and Lake Mahinapua (TLI = 3.4). Although Lake Mahinapua is in a highly unmodified catchment, it had in-lake TN concentrations of 330 mg/m^3, predominantly consisting of dissolved organic nitrogen (DON). Thus there is likely to be more difficulty in interpreting N retention because in-lake TN concentrations can be affected by a number of N processing factors both within lakes (denitrification, fixation) and their catchments (wetland processing of organic material).

Comparisons with previously published data indicate that this data set has relatively lower N retention coefficients (Table 3) than found in other studies, including shallow lakes (*e.g.* Windolf *et al.* 1996). In most instances published values have been positive, predominantly ranging between 0.1–0.7. The higher degree of negative N retention coefficients found in this data set could indicate the CLUES model is under-predicting actual N loading rates. The lower correlation between in-lake TN and both

CLUES loading and Vollenweider TN predictions would also suggest that CLUES N loading predictions are either less accurate or far more affected by in-lake processing than can be accounted for by our modelling. The limited (modelled) data on which to base calculations of retention coefficients, in the absence of inflow-outlfow monitoring data, warrant a degree of caution in interpreting the precise value of nutrient coefficient. Thus we would interpret the retention coefficients to be indicative only in terms of the whether nutrient processes are likely to be dominated by internal loading and/or N-fixation, or in lake processing (*e.g.* denitrification, P-uptake by aquatic plants).

a) Phosphorus retention
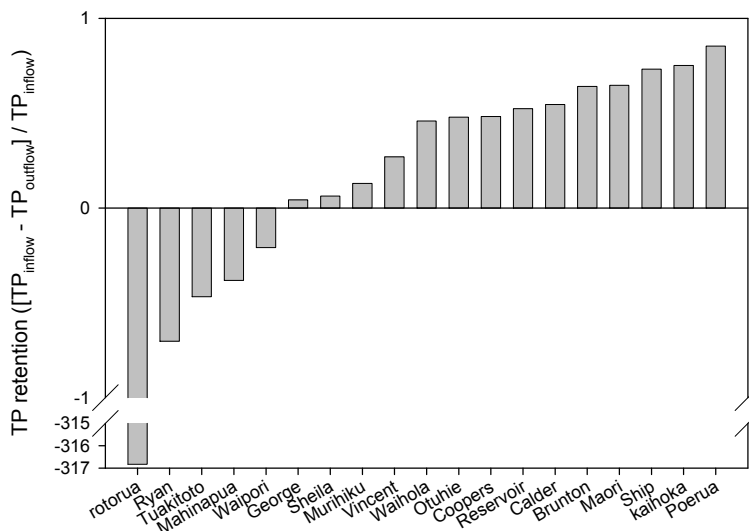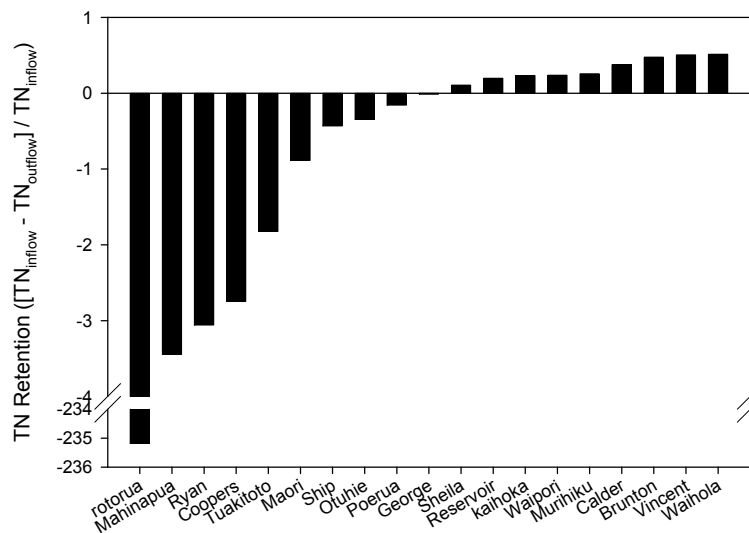

b) Nitrogen Retention

Figure 7.   Rates of nutrient a) phosphorus (P) and b) nitrogen (N) retention for 19 South Island shallow coastal lakes as estimated from ratios of predicted inflow concentrations and outflow nutrient concentrations.

Table 3.   Nitrogen (N) and phosphorus (P) retention coefficients for lakes collected from published nutrient budgeting studies. Sources are listed in the reference column.

Retention parameter*	Published values	n	Notes	Reference
RTP	Range: *c.* -0.7–*c.* 0.9. Median = 0.45.	305	Analysed published TP mass balances (mainly North America). RTP was < 0 for 12% of lakes.	(Brett & Benjamin 2008)
RTP	Range: *c.* -0.5–0.93 Median: 0.5	54	Analysed published TP mass balances. RTP was < 0 for 17% of lakes.	(Koiv *et al.* 2011)
RTP	0.55	1	Long-term (17 yr) study of an Australian lake	(Cook *et al.* 2010)
RTN	Range: 0.11–0.72	16	From mass balances for shallow, eutrophic lakes in Denmark	(Windolf *et al.* 1996)
RTN	Mean: 0.07	1	Long-term (17 yr) study of an Australian lake. The low value was attributed to N–fixation.	(Cook *et al.* 2010)
RTN	Mean: *c.* 0–0.28.	8	Norwegian study over 3 yr. RTN high in eutrophic lakes, low in oligotrophic lakes. High inter-annual variability with some annual RTN values < 0.	(Berge *et al.* 1997)

*RTP = Retention of total phosphorus, RTN = Retention of total nitrogen.

3.4. Nitrogen and phosphorus loading relationships with lake ecological integrity indicators

3.4.1.   Physico-chemical parameters

There were strong relationships between CLUES predicted loading and in-lake chlorophyll-*a* concentrations, particularly with TP loading (Figure 8). Loading values that were transformed using Vollenweider models provided stronger relationships than raw loading (in T/y) or loading corrected by lake area (t/ha/y). Given the connection between TP and chlorophyll concentration (*e.g.* Dillon & Rigler 1974; Rigler 1982), it is not surprising that loading values calibrated to nutrient concentration would provide a strong fit with in-lake chlorophyll-*a*. Southland lakes appear to fit the linear regression line well, with the exception of Lake Brunton, which had chlorophyll-*a* concentrations much lower than predicted. Because the lake's barrier-bar had breached just before the time of sampling in 2012 and the lake was quite saline, it is not surprising that the chlorophyll-*a* concentration was apparently not in equilibrium with the catchment nutrient load.

There was a poorer fit of the Vollenweider TN loading model data to in-lake chlorophyll-*a*, although the regression model was still significant. This indicates that either that the chlorophyll-*a* concentration was less dependent on N loading to the lakes in our data set, that inaccuracies in the TN loading model have resulted in a

poorer model, or that in-lake nitrogen transformation processes play an important role in determining N availability to phytoplankton.

To assess this, we looked at in-lake nutrient ratios of dissolved inorganic nitrogen (DIN) to TP and TN:TP to provide an indication of the likely nutrient limitation status of the lakes at the times the lakes were sampled (Figure 9). DIN:TP ratios that near 1, and TN:TP ratios near 7 (Redfield ratio by mass) indicate that supply of N and P are roughly balanced in relation to the demands of plant and algae growth. Increasing departures from these thresholds could suggest that primary productivity in the systems increasingly limited by either N (<< than the thresholds) or P (>> than the thresholds), with single nutrient limitation is more likely to occur with ratios exceed double or half the balanced ratios (*i.e.* for DIN:TP, p-limited when ratio >2, N limited when ratio <0.5. For TN:TP, P-limited when ratio >14, N-limited when ratio <3.5).

Looking at both ratios for the data set, all 19 lakes had nutrient TN:TP ratios indicating some degree of P limitation, with 16 of 19 lakes having ratios indicative of strong P limitation, being >14. For Southland lakes, Lakes George and Murihiku had ratios close to the nutrient thresholds (*i.e.*, TN:TP = 14, DIN:TP = 1), suggesting the lakes are more likely to be potentially co-limited by both N and P. For DIN:TP ratios, a greater number of lakes would be considered co-limited by both N & P (9 of 19), and 2 lakes were considered possibly N limited (Lakes Ryan and Tuakitoto). Lakes Vincent, Sheila, Calder and Brunton all had N:P ratios indicative of stronger P limitation. For lakes that were either co-limited or N limited, a stronger fit to the N loading chlorophyll model was observed. All Southland lakes, save for Lake Brunton, fit the P loading chlorophyll model reasonably well, suggesting P loading is likely to be an important controller of chlorophyll-*a* biomass in these lakes.

Inferring nutrient limitation is complex, and as discussed in previous sections, is likely to vary seasonally as processing and internal loading rates vary seasonally with changes in water temperature. Concentrations of soluble nutrient fractions (DIN and SRP) are important in determining nutrient bioavailability, but can vary greatly over relatively short periods. Thus TN:TP is more often used as an indication of nutrient status because it incorporates both dissolved and bound nutrient fractions, and is more stable. Overall these nutrient ratios would suggest a greater likelihood for P or co-limitation in the shallow Southland lakes, which is supported by stronger correlations between TP loading and in-lake variables. The prevalence of P limitation in New Zealand lakes has also been reported in other studies (Abell *et al.* 2010). Notwithstanding, the close proximity of these lakes to coastal estuarine areas would suggest N control to also be an management objective, as marine systems are more often limited by N.

Figure 8.   Relationships between mean annual CLUES total phosphorus (TP) and total nitrogen
            (TN) loading transformed using Vollenwider models and in-lake chlorophyll concentration
            for 18 South Island lakes. Note that Lake Rotorua was excluded from the plot due to it
            being considered an outlier.
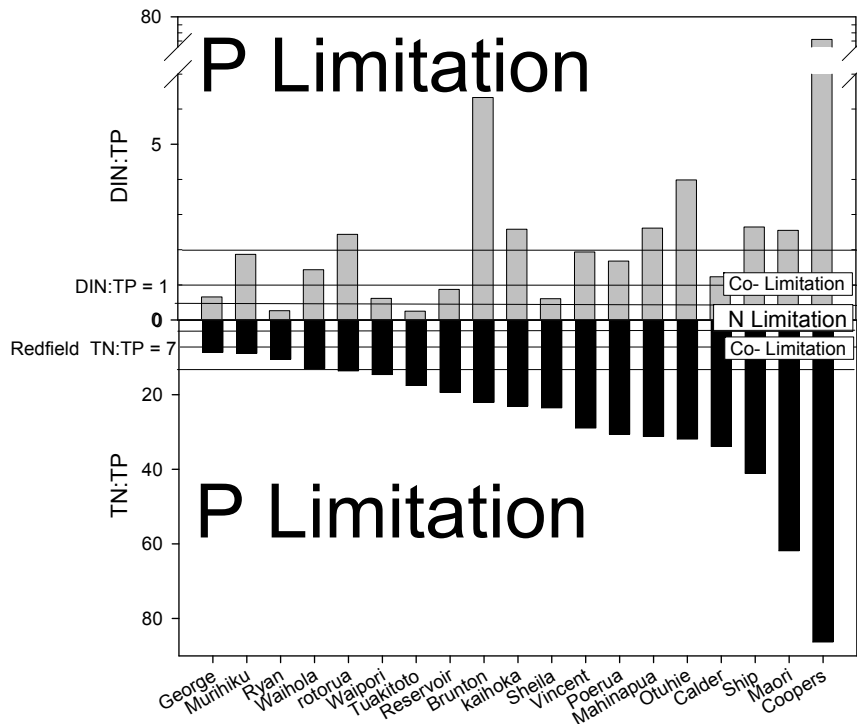
Figure 9.   Nutrient ratios of in-lake DIN:TP and TN:TP for 19 South Island coastal lakes. Nutrient
            limitation status, either by P or N limitation, is indicated by DIN:TP ratios diverging from
            1:1 (< 0.5 = N limitation 0.5 - 2 = co-limitation; >2 = P limitation) or TN:TP ratios diverging
            from 7 (< 3.5 = N limitation; 3.5 - 14 = co-limitation; >14 = P limitation).

A similar trend was observed in the TP and TN loading models with lake TLI (Figure 10). There was a strong fit in TP loading with TLI with 86% of the variation explained by the model. There was considerably more scatter in the relationship between Vollenweider corrected TN loading and TLI, with the model explaining approximately 36% of the variation. For lakes that would tend to indicate P limitation such as Lakes Sheila, Calder, Vincent and Brunton, the lakes were poorly related to the best fit line of the TN plot. These findings further support the observations of the importance of TP loading in controlling chlorophyll-*a* and TLI in these lakes.
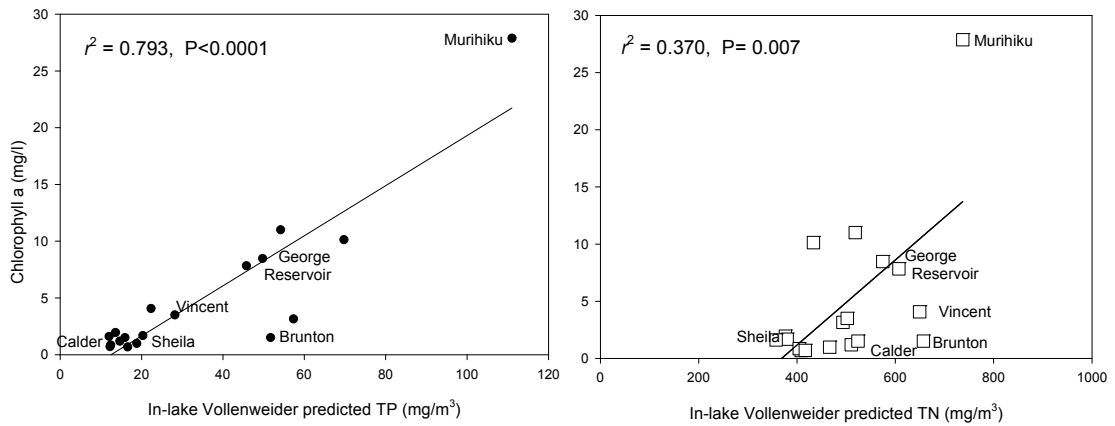

Figure 10.   Relationships between mean annual CLUES total phosphorus (TP) and total nitrogen (TN loading transformed using Vollenwider models and in-lake trophic level index (TLI) for 18 South Island lakes. Note that Lake Rotorua was excluded from the plot due to it being considered an outlier.

3.4.2. *Other lake ecological integrity indicators*

Significant inverse relationships were observed between CLUES TN and TP loading and lake euphotic depth (*i.e.* depth of 1% surface irradiance) (Figure 11). Inverse polynomial first-order equations were best fit to the models, with euphotic depth decreasing in a non-linear manner with loading. This non-linear relationship was expected, as light transmission is known to change in an exponential manner with increasing water column suspensoids (Wetzel 1983).

There was considerable scatter in the relationships, particularly around lower levels of TN and TP loading. Most of this is attributed to differences in water column clarity caused by either coloured dissolved organic materials (CDOM, *i.e.* humic acid staining), or turbidity related to wind-driven resuspension events. The light transmission environments in shallow lakes is highly variable (Hamilton & Mitchell 1997) due to their susceptibility to resuspension, so the limited number of sampling events to quantify water clarity for these lakes would be highly susceptible to being affected by short term variation. However, our analysis also demonstrates that CDOM in lakes with more intact catchments (and lower nutrient loading) strongly affected this indicator. As such, water clarity is likely to be a difficult indicator to relate to nutrient

loading in isolation, and probably acts in conjunction with CDOM (influenced by wetland areas in the catchment) and the lakes susceptibility to wind-driven resuspension. Of interest was that $Z_{euphotic}$ more strongly correlated with TN loading, unlike chlorophyll-*a* and TLI. This may be related to organic nitrogen content of humic acids and other dissolved organic matter, whereby CDOM concentration may reflect dissolved organic nitrogen concentrations.

There were significant relationships between euphotic depth and areal load of TN (Figure 11), although the areal TP load model was not significant. Both would suggest a similar rapid decline in euphotic depth with loading up to 5 kg P/ha/y P and 50 kg N/ha/y. As with the Vollenweider model TP and TN, there was considerable variability in euphotic depths of lakes with lower areal TN and TP loads, most likely related to variation in CDOM between the reference condition lakes.
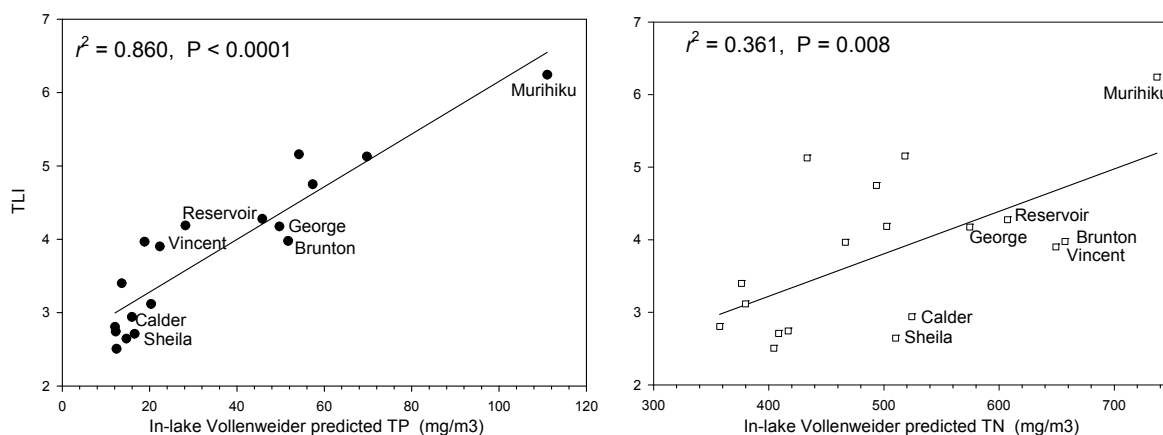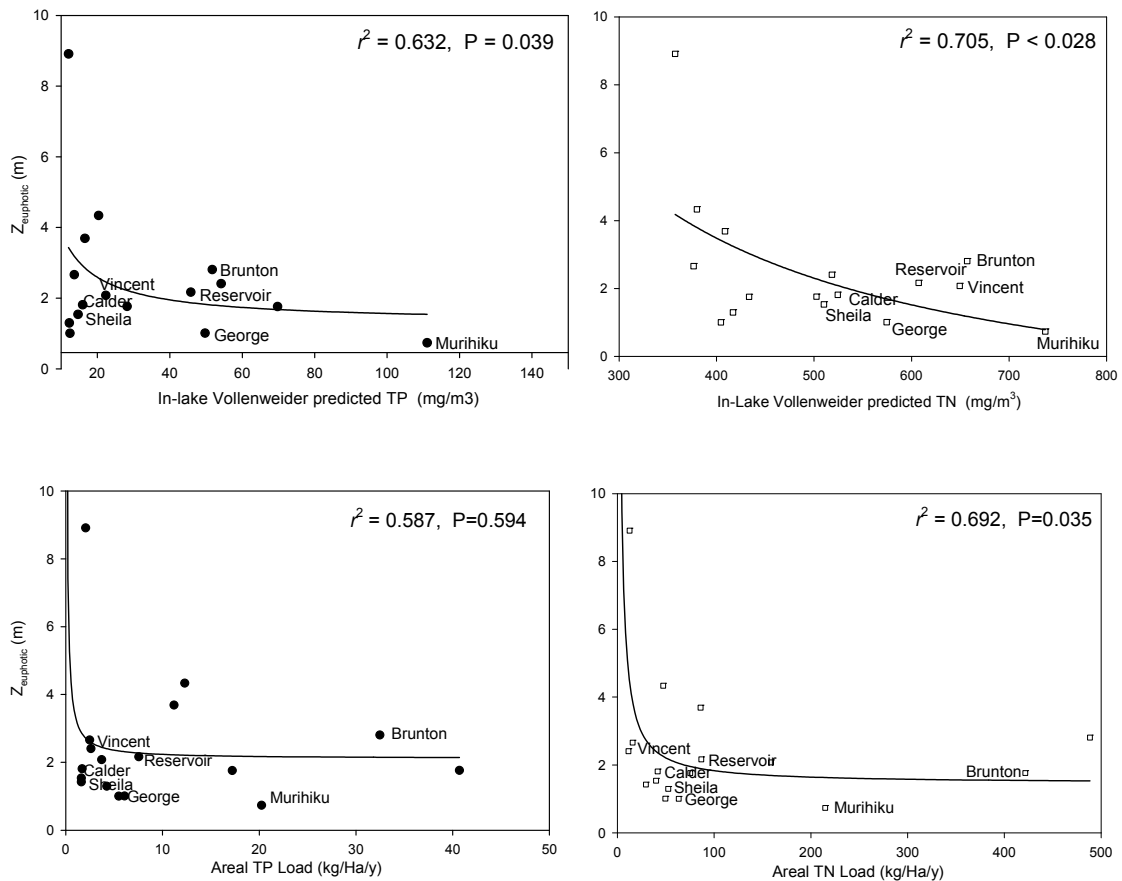

Figure 11.   Relationships between mean annual CLUES total phosphorus (TP) and total nitrogen (TN) loading transformed using Vollenwider models or expressed as areal loads and in-lake euphotic depth ($Z_{euphotic}$) in 18 South Island lakes. Note that Lake Rotorua was excluded from the plots due to it being considered an outlier.

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Aquatic macrophyte communities are considered to be an important ecological component for shallow lakes, which can have plant communities over their entire area due to their shallow depths (Scheffer 2004). We observed a general decreasing trend of macrophyte cover with both TP and TN loading (Figure 12), however these relationships had a great deal of noise, and thus there were no significant relationships. The strongest relationship (although non-significant) was observed between TP loading corrected to in-lake concentration using the TP Vollenweider model. Although there is sound ecological reasoning behind the possibility for a relationship between total areal loading of nutrients and their lake bottom communities such as macrophytes, there was no clear pattern of areal TP or TN loading in this data set.

Relationships observed general suggest that macrophyte cover is in lakes with greater loading of TP, but it is difficult to be certain where major changes (*e.g.* flipping) of macrophytes may occur (Figure 12). It is possible that loads for most lakes were below levels that would cause macrophyte collapse, and this is supported by overseas studies which suggest concentrations >100 mg TP/m^3 and between 1000-2000 mg TN/m^3 (Jeppesen 2007). Most lakes with loading that results in in-lake TP concentrations in excess of 40 mg/m^3 TP had low (<30%) macrophyte cover. However, Lakes George and Brunton were exceptions to this. Lake George has undergone some considerable fluctuations in its macrophyte cover since 2004 when it was first sampled (Drake *et al.* 2009) and has only recently been recolonized by macrophytes. It is anticipated for both lakes that sediment concentrations associated with P inputs could likely be as important a factor relating to macrophyte cover than P concentration, and difficult to entangle in terms of the proximal causes of macrophyte collapse. Although macrophyte cover has previously been observed to be related to in-lake TP concentration as in our study (Jeppesen *et al.* 2007; Lauridsen *et al.* 2003), recent evidence suggests that macrophyte richness can be strongly influenced by nitrogen concentration, possibly related to between-species differences in N concentrations for optimal growth (Moss et al. 2013). This would suggest the importance of co-management of nutrients for maintaining macrophyte community richness.

Some of the lakes in our data set had low macrophyte cover despite low TP and TN loading, for example Lake Otuhie and Lake Calder. Both of these lakes have high CDOM concentrations, making much of their lake bed deeper the depth of euphotic zone, limiting macrophyte growth potential. Lake Otuhie also had considerable peat influences in the immediate riparian area, and this was thought to contribute to lake bed sediments that are organically rich and not particularly conducive to aquatic macrophyte colonisation. Lake Sheila on Stewart Island also had iron-manganese oxide crus material over a portion of its lake bed that limited macrophyte growth (Schallenberg & Kelly 2013). Thus there are likely to be several other factors controlling the extent of macrophyte community cover, and it is unlikely that nutrient

load predictions will entirely account for macrophyte communities in these waterbodies.

Figure 12.   Relationships between CLUES total phosphorus (TP) and total nitrogen (TN) loading, expressed either as Vollenweider model concentrations or as areal loads, and littoral macrophyte cover in 18 South Island Lakes. Note that one lake was not included in the analysis due to it not having been sampled for macrophyte cover (Waipori).

Littoral macroinvertebrate community richness was observed to have a significant inverse relationship with TP loading (and in-lake TP concentration) in the 18-lake data set (Figure 13). As with relationships with aquatic macrophyte cover, areal loading rates of TN and TP were poorly correlated to invertebrate taxon richness, and all regression models were insignificant. The TN Vollenwider model was marginally non-significant, but trends were evident as with the TP model.

Although the macroinvertebrate community is likely to reflect littoral habitat values such as aquatic macrophyte cover, the relationships of macroinvertebrates with TN and TP loading were stronger than for macrophytes, suggesting that they may incorporate other environmental aspects related to loading or in-lake nutrient concentrations. This could include aspects previously discussed, such as sediment

quality, or the composition of the epiphyton community, which was not sampled as part of this investigation. Recent work by Hickey (2009) examining nitrate toxicity to river macroinvertebrates has observed chronic toxicity for *Deleatidium* at around 1000 mg/m^3, just within the range of nitrate concentrations observed in our lake data set (0.2 - 1061 mg/m^3). Unfortunately there is no knowledge at present on nitrate sensitivity of any littoral macroinvertebrate taxa such as Odonata, Tricoptera, or Mollusca, which were the main taxa responding to respond to nutrient loading. Macroinvertebrate communities may also be less sensitive to factors affecting light climate that correlated with macrophyte community metrics. Unlike our macrophyte cover estimates, which included sampling at multiple points along transects from the shoreline, macroinvertebrates were sampled closer to the lake margin (at 5-10 m distance from the shore) so would not have incorporated the dynamics of a changing light environment over depth as the aquatic macrophyte cover metrics did. The stronger relationship of macroinvertebrates with P loading could suggest that greater P rich sediment (silt) input is related to such community shifts, as this is evident for riverine benthic communities (Clapcott *et al.* 2011). Overall, it was evident from our data set that macroinvertebrate communities are responsive to environmental gradients related to nutrient loading, but mechanisms behind such responses are still relatively unknown.
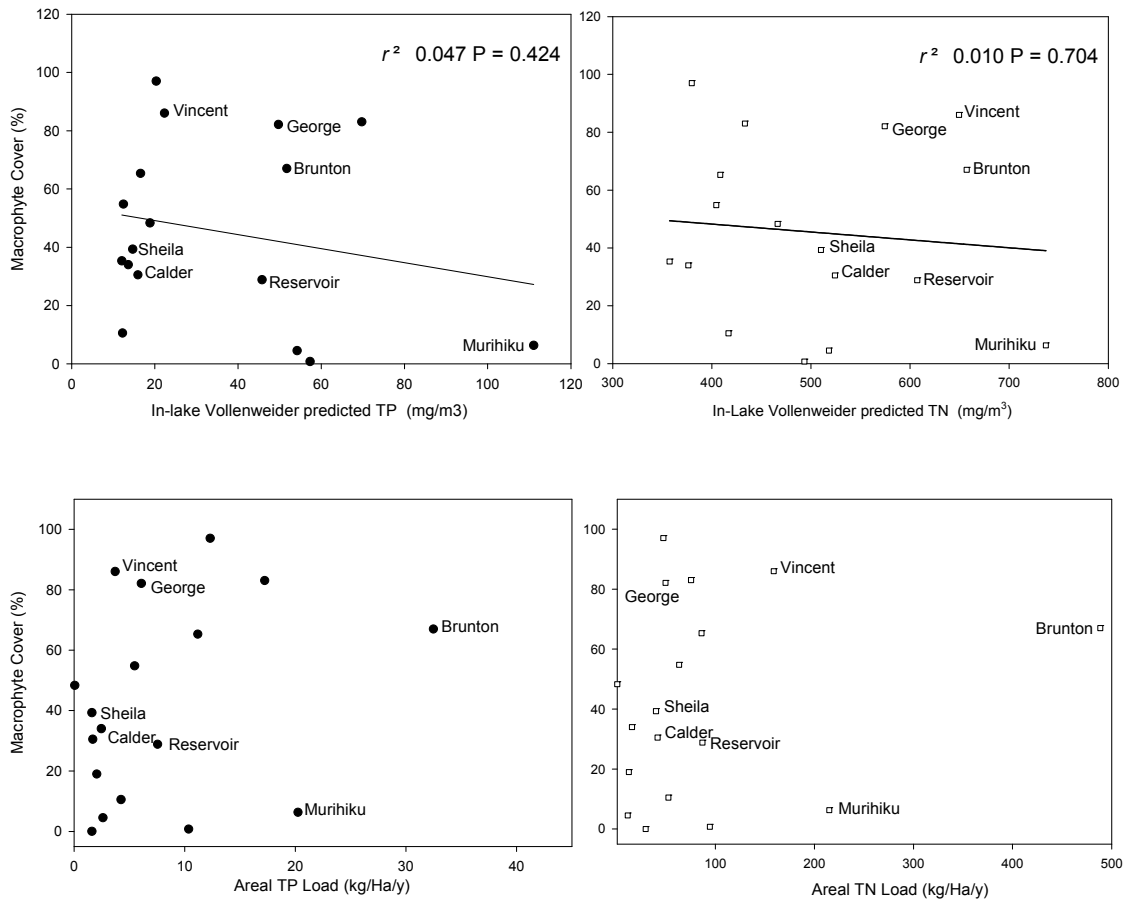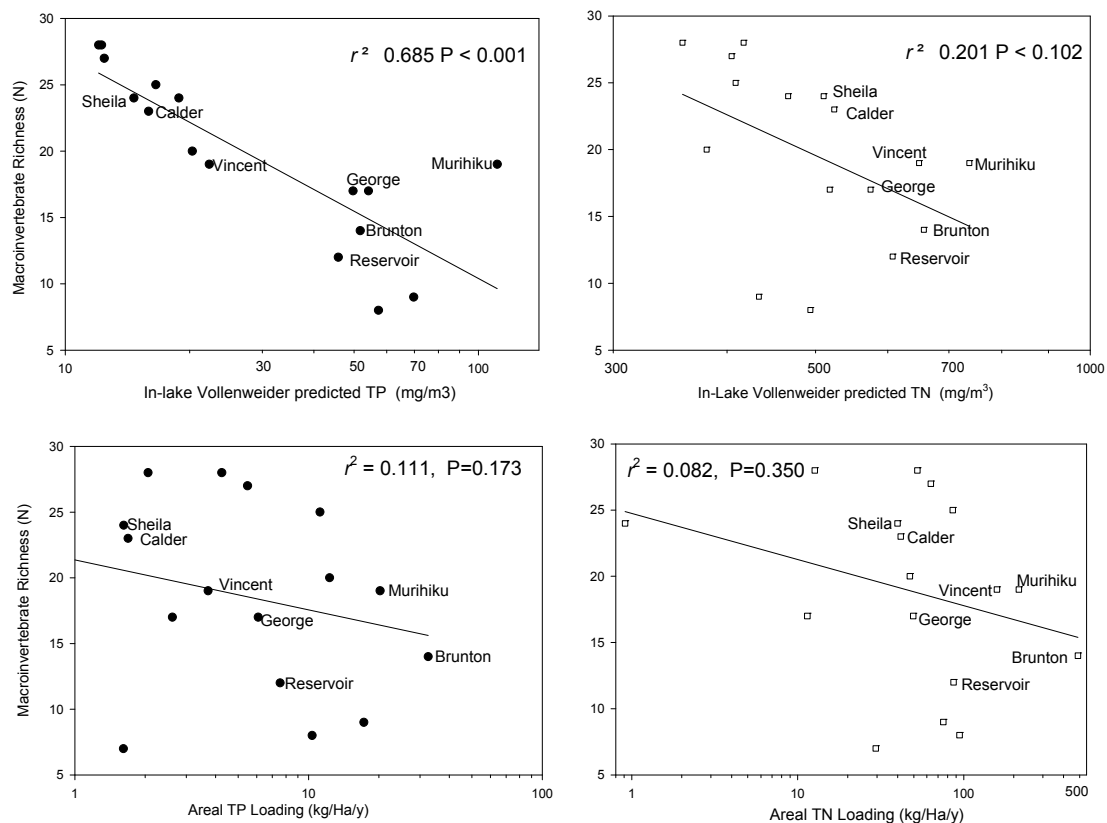

Figure 13.   Relationships between CLUES total phosphorus (TP) and total nitrogen (TN) loading, expressed either as Vollenweider model concentrations or as lake areal loads, and littoral macroinvertebrate taxonomic richness (N) in 17 South Island Lakes. Note that two lakes were not included in the analysis, one due to it not having been sampled (Waipori) and one due to it being considered an outlier from sample processing error (Mahinapua).

4. CONCLUSIONS AND RECOMMENDATIONS

Review of the international literature indicates that managing loads to control P concentrations is the most significant management action for protecting the ecological integrity of shallow freshwater lakes. In the extensive work conducted in European shallow lakes (UK, Denmark, Netherlands) managing catchment nutrient loads to maintain (or restore) shallow lakes to a clear-water macrophytes-dominated state, in-lake P concentration was nearly always the most immediate management objective. Achieving TP concentrations for these objectives, however, varied in terms of whether this could be achieved only through managing catchment loads, or if managing loading from internal sediment sources or biomanipulation of the food chain was necessary to achieve such results (Søndergaard *et al.* 2005; Jeppessen *et al.* 2007, Gibbs 2011).

Patterns emerged in the literature around the importance of external and internal sources (Figure 14). For lakes which had experienced historically high loads or point source pollution, internal nutrient loading became a significant management objective, whereas moderately degraded systems responded more rapidly to catchment load reductions. In most cases external loads and in-lake TP and TN concentrations were far greater than observed in the data set for South Island shallow coastal lakes, which adds some complexity in relating this information to management objectives for Southland shallow lakes. From the study data available, lakes that experienced P loading in excess of 80 kg P/ha/y were likely to have accumulated P rich sediments could result in internal loading. However, the highest CLUES loading rates prediction for the New Zealand lake-set was 34 kg P/ha/y, well below this overseas value. While this might suggest lesser potential for accumulations of P rich sediment for lakes in this study, negative retention coefficients (suggestive of internal loads) were observed for three of the more-eutrophic lakes, Lakes Rotorua, Ryan, and Tuakitioto. For these lakes, P loading was between 10-17 kg P/ha/y, excluding Lake Rotorua which we believe to have been an outlier in the CLUES model prediction. For Southland lakes, both Lakes Brunton and Murihiku exceeded these lakes in terms of predicted areal loads, with values of 32 and 22 kg P/ha/y, respectively. This suggests some risk that external loads are at levels that could ultimately generate internal loading issues, although more data around calculating nutrient retention coefficients (*i.e.* inflow / outflow monitoring) would provide greater certainty in these findings.

Relationships between several ecological integrity indicators and N and P loading were evident, particularly for CLUES loading corrected to in-lake TP concentration using optimised Vollenweider models. The strongest relationships with loading were evident for chlorophyll-*a* and TLI, with 79% and 86% of their variation explained by the TP loading variable, respectively. These relationships were linear, thus for management purposes TLI targets for lakes could be directly related back to nutrient load targets for the lake, correcting for inflow concentration and lake water residence time as done in normal Vollenweider modelling. It is important to note that this is

primarily correlational, based on relating loads with environmental patterns of water quality in the lake data set, and there is likely to be some degree of error. There are a number of complex in-lake processes linked with nutrient processing and resuspension that ultimately control the retention or export of nutrients from catchment loads (Scheffer 2004), and this approach largely ignores this complexity by relating retention predominantly to water residence time. The strong model performance for this study is possibly related to the relatively short water residence times of the lakes, minimising the potential for in-lake nutrient processing. However there is a risk when using such correlational data, that modifying loads (either by load reductions or allowed increases) to particular lakes would not result in expected changes in retention or in-lake concentration based on the models. Factors such as internal loading (*e.g.* risk in Murihiku, and Brunton) or short term blooms causing shifts in aquatic plant dominance (regime shifts) can be important factors, as widely cited in the literature (*e.g.* Perrow *et al.* 1994, Søndergaard *et al.* 2005). Thus, this form of nutrient load target setting would need to be in conjunction with in-lake monitoring. Lakes that appear to be poorly correlated with model predictions and suggest atypical nutrient loading - response dynamics could be the subject of further study.

This investigation, and to large extent the international literature, suggests lesser importance of N loading to shallow lakes. This is predominantly related to the view that if lakes become N limited, conditions will favour dominance by N fixing phytoplankton species, thereby making up for any deficit of N in the system (sensu Schindler 2008). Thus it is largely P loads that control nutrient availability and productivity, and thus has been the main focus of managing nutrient loads to shallow lakes. Our modelling would support this to some extent, as there was consistently less variation accounted for in TLI, chlorophyll-*a*, (and other EI variables, *e.g.* macrophytes, macroinvertebrates) by TN loading models. However in several instances models were still significant suggesting that N-loading is related to chlorophyll-*a* and TLI. For instance euphotic depth (water clarity) was more strongly correlated with TN concentration. Søndergaard *et al.* (2005) reported that a euphotic depth of > 2 m was the important variable for maintaining macrophyte communities in a restoration study of Danish lakes. Our data would indicate this threshold occurs around 550 mg TN/m^3.

In-lake nutrient ratios of TN:TP and to a lesser extent DIN:TP both suggested that many of the lakes in the data set are likely to be phosphorus limited (16 of 19 for TN:TP), but several were close to the Redfield, most likely indicating co-limitation. This could be possibly related to N fixation when systems become highly P limited, but it was beyond the scope of this study to evaluate. In a recent international meta-analysis study looking at the importance of N to water quality in shallow lakes, Moss *et al.* (2013) concluded that while chlorophyll-*a* and trophic status was more often controlled by P, macrophyte species composition and (in some cases) cover was related in in-lake TN concentration, with some species clearly having TN tolerances or preferences. Because macrophytes are considered critical in the ecology of shallow

lakes in maintaining clear-water conditions, this was thought to indirectly affect water quality, thus management of both nutrients *is* clearly intertwined (Moss *et al.* 2013).

Results from our lake-set relating nutrient loading to other key shallow lake EI variables such as macrophyte and macroinvertebrate communities, also indicated that P loading was the more proximal variable relating to benthic communities (Figure 13). While it was originally thought that areal loads would be the most relevant load statistic, as this could be related to total inflow of particulate material per area of the lake bed, this was not the case. Loading corrected to in-lake concentration was the stronger predictor, however all models for macrophyte cover were weak, being only marginally significant for Vollenweider corrected P loading. Values from the literature suggested there is considerable variability in the 'threshold' values of in-lake TP concentration linked with either significant decline or collapse of macrophyte communities. From the range of studies available, in European (Danish, Netherlands) lakes with in-lake TP concentrations between 100–130 mg/m^2 seemed to be most often related to the transition between lakes with some and no macrophyte cover (Gonzales Songrario *et al.* 2005; Søndergaard *et al.* 2005). However, Jepessen *et al* (2007) reported for a Florida lake-set that this threshold was much lower, at around 50 mg/m^3 in-lake TP concentration. Results for this study on South Island shallow lakes would tend to indicate a lower threshold, with lakes having loads resulting in mean summer TP concentrations of > 50 mg/m^3 nearly always having little or no macrophyte cover. However the extent of our data set was limited to draw strong conclusions in relation to specific threshold loads or concentrations.

Macroinvertebrate community richness declined linearly with P loading and, to a lesser extent, with TN loading (Figure 13). At the previously mentioned 50 mg/m^3 TP upper limit as proposed from macrophyte cover, macroinvertebrate richness would have declined by near 42% (from approximately 26 to 15 taxa) compared with reference condition lakes. Thus if this variable was used as an indicator of shallow lake EI related to biodiversity values, this would suggest that there is considerable decline in value as this upper TP limit is approached. As such, we would suggest a more conservative target to minimize losses in biodiversity values, and be more moderately below the threshold at which macrophyte cover rapidly declines. This could be considered on a case-by-case basis based on retaining in-lake water clarity at sufficient levels to ensure the euphotic depth was below the mean depth of the lake (see Figure 11). However, if a single load threshold was required for all shallow coastal lakes in the Southland Region for planning purposes, we would suggest the loads, back-calculated using Vollenweider models, be no greater than levels to achieve a TP (and possibly N) concentration in the mid eutrophic range (from Burns *et al.* 2000), or 32 mg/m^3 TP (and possibly 531 mg/m^3 TN). This is based on the combined relationships of P loading with macrophyte cover, macroinvertebrate richness and euphotic depth, and is also tied to a meaningful scale for in-lake nutrient concentrations, in this case TLI. Because calculation of loads in our study were based

entirely on model outputs, further monitoring around validating loads through inflow-outflow monitoring is recommended.

Overall, results from modelling work provide some valuable information in terms of relationships of N and P loading with key variables related to the ecological integrity of shallow lakes. The nature of these relationships can be used to help inform the decisions on loading rates for specific lakes to achieve water quality outcomes for the lake. The nature of these outcomes can then be discussed at a community planning level and with water quality targets developed for specific waterbodies taking into account the communities aspirations. While these modeled relationships provide a means of developing load limits based on ecological values and water quality objectives, and are generally in agreement with those reported in the literature, it is important to remember that models only approximate environmental patterns. Thus information derived from the models should be the starting point of discussion, with further monitoring supporting and validating these decisions. This investigation was predominantly focused on ecological indicators related to EI, and it did not consider other values such as recreational and amenity uses as they relate to nutrient limits. As such, this is identified as a gap and possibly an area for further work.
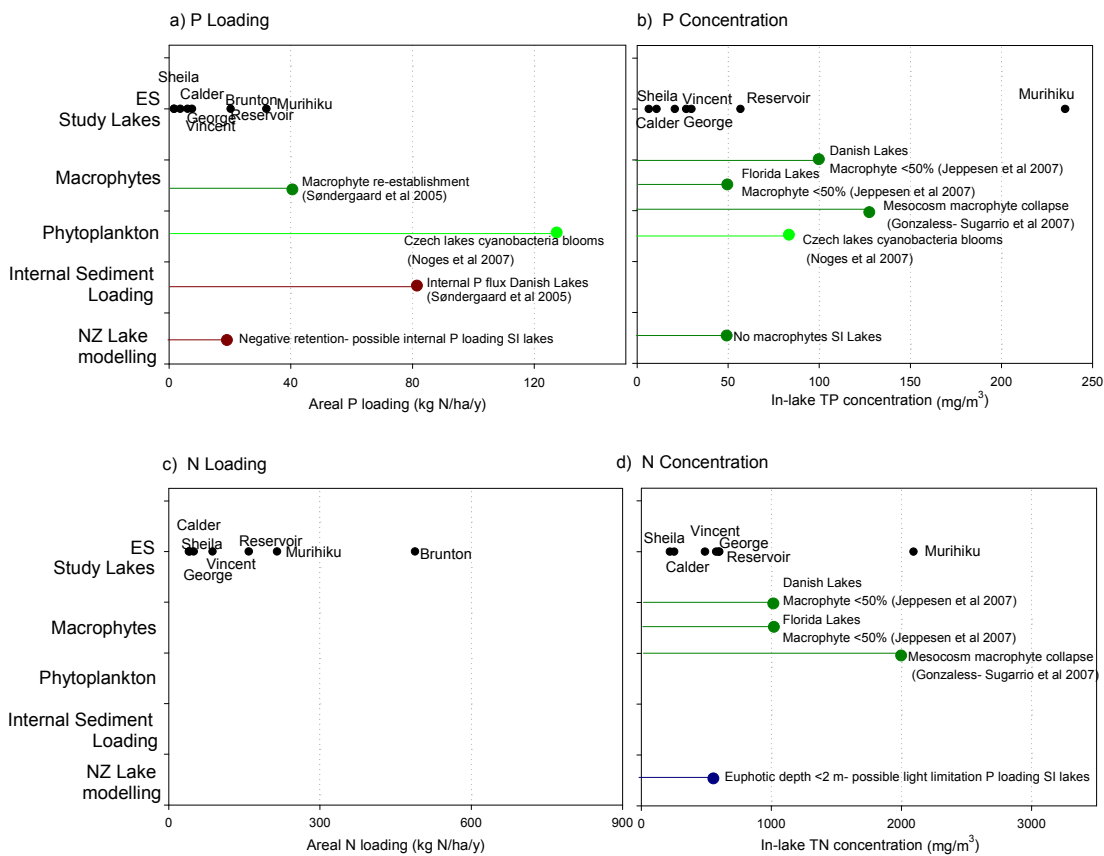

Figure 14.   Threshold nutrient load and nutrient concentration responses summarised in this report for a) phosphorus (P) loading, b) P concentration, c) nitrogen (N) loading, and d) TN concentration and various in-lake ecological integrity response indicators.

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6. APPENDIX

Appendix 1.   Summary of data compiled in literature review for shallow lakes.

Lake	Mean depth (m)	Surface area (ha)	Volume (10^6 m^3)	Residence time	P loading (t/y or t/ha lk area/y)	N loading (t/y or t/ha lk area/y)	Chlorophyll-*a* range mg/m^3	Nitrogen: NO$_3$, TN, DIN mg/m^3	Phosphorus: DRP, TP mg/m^3	Macrophytes: dominant species	Fish: dominant species	References
Søbygaard (Denmark)	1.0 Z_{mean}	40	N/A	15-20 days (winter) 25-30 (summer)	300 kg P/ha/y (pre 1982) 50 kg P/ha/y (post 1982)	N/A	Mean = 800 (1984-5) Mean = 200-300 (1987 - publication time)	N/A	TP (inlet) 0.1 - 0.3 TP (outlet) 0.1 – 0.3 (winter) 0.5 – 1.0 (summer)	N/A	N/A	Søndergaard *et al.* (1993) Jensen *et al.* (1992)
Eight shallow lakes (Denmark)	1.9-10.5 Z_{max}	21-4000	N/A	0.05-2.2 y	83 kg P/ha/y (1989-92) 41 kg P/ha/y (1993-2001)	1900 kg N/ha/y (1989-92) 1897 kg N/ha/y (1993-2001)	Summer 150 (1989-92) 90 (1998-2001)	Winter TN 7650 mg/m^3 (1989-2001)	Summer TP 330 mg/m^3 (1989-92) TP 170 mg/m^3 (1998-2001)	None	N/A	Søndergaard *et al.* (2005)
Peipsi[1,2]	7.1 Z_{mean}	355,500	25.07	~ 2 years	0.83 kg P/ha/y 1980-1991 0.73 kg P/ha/y 1992-2004	33.1 kg N/ha/y 1980-1991 20.0 kg N/ha/y 1992-2004	18	TN 0.768	TP 0.042	N/A	N/A	Noges *et al.* (2007)

Lake	Mean depth (m)	Surface area (ha)	Volume (10^6 m^3)	Residence time	P loading (t/y or t/ha lk area/y)	N loading (t/y or t/ha lk area/y)	Chlorophyll-*a* range mg/m^3	Nitrogen: NO$_3$, TN, DIN mg/m^3	Phosphorus: DRP, TP mg/m^3	Macrophytes: dominant species	Fish: dominant species	References
Vortsjärv[2]	2.8 Z$_{mean}$	27,000	0.75	~ 1 year	1980-3.5 kg P/ha/y (1991)　2.3 kg P/ha/y (1992-2004)	134.1 kg N/ha/y (1991)　65.3 kg N/ha/y- (1992-2004)	24	TN 1.600	TP 0.054	N/A	N/A	Noges *et al.* (2007)
Alderfen Broad	0.8 Z$_{mean}$	4.7	N/A	N/A	N/A	N/A	N/A	NO$_3$ 0-2.1	DRP 0-0.425 TP 0.07-0.5	*Ceratophyllum demersum*	*Rutilus rutilus* (roach)	Perrow *et al.* (1994) Moss (1983)
Horowhenua	1.3 Z$_{mean}$	2,900	3.8	47	External 10.7 kg P/ha/y Internal 2.0 kg P/ha/y	External 700 kg N/ha/y Internal 103 kg P/ha/y	150-400	NO$_3$ 0-2.5	DRP 0-0.4	*Potamogetan crispus*		Gibbs & White (1994), Gibbs (2011).
Kotojärvi[3]	2.5 Z$_{mean}$	30	0.7	160 days	2190 kg/ha/y		Mean chl. *a* 26	Mean TN 0.99	Mean TP 0.067	N/A	N/A	Knuuttila *et al.* (1994)
Villikkalanjärvi[3]	3.2 Z$_{mean}$	710	23	67 days	10950 Kg/ha/y		Mean chl. *a* 20	Mean TN 1.7	Mean TP 0.12	N/A	N/A	Knuuttila *et al.* (1994)

Notes:

[1]This lake consists of three subbasins to it – Lakes Pihkva, Lämmijärv and Peipsi s.s. I used the information for all three combined in the table.

[2]The lake N and P loadings given in the table are the loading from external sources. The paper also provides in-lake concentrations of N and P in units of T/Y.

[3]The TP and TN loadings are from agricultural land.