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ECOLOGICAL ASSESSMENT OF LAKE TUAKITOTO AND TOMAHAWK LAGOON AND OPTIONS FOR LAKE REHABILITATION

ECOLOGICAL ASSESSMENT OF LAKE TUAKITOTO AND TOMAHAWK LAGOON AND OPTIONS FOR LAKE REHABILITATION

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Prepared for Otago Regional Council

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EXECUTIVE SUMMARY

Lake Tuakitoto and the Tomahawk Lagoons are significant shallow coastal lakes in the Otago Region. They are highly valued, having outstanding wildlife, biodiversity, and recreational resources and amenities. In response to concerns about degrading water quality, the Otago Regional Council (ORC) has engaged Cawthron Institute and the University of Otago to conduct a study into ecological health and processes that control water quality in Lake Tuakitoto and the Tomahawk Lagoons. The overall aim of the study and this report is to provide an understanding of restoration options that could be employed to improve water quality and safeguard the ecological health of the lakes. This report presents the recommendations developed through:

- identifying the values and restoration aspirations held by stakeholders
- summarising relevant water quality and ecosystem health data for the lakes and their catchments
- determining key processes and limits that drive the ecological health of the lakes
- presenting recommendations on options for improving lake ecological health and enhancing values for stakeholders (note that this does not include quantitative assessments of costs or benefits).

Review of existing environmental data for the lakes indicated that ecological values are currently being compromised by continued degradation in water quality, which impacts the ecological health and condition of the lakes.

Lake Tuakitoto – current state

For Lake Tuakitoto, increasing total nitrogen and total phosphorus concentrations in the lake are contributing to extensive macroalgae and phytoplankton blooms, which now exceed the regional and national guidelines for the protection of ecosystem health. Undesirable feedback processes that drive internal nutrient loads from lake sediments appear to have become established in the lake, leading to high water column pH and possibly low, transient dissolved oxygen concentrations. Such poor water quality conditions have flow-on effects to sensitive biota such as native fish and, in particular, kākahi, which have declined by 85% in biomass since observations were first made in the 1990s. Unfortunately, this means that the important ecosystem services kākahi provide to the lake by filtering lake water and removing algae have greatly declined. Thick macroalgae mats were seen to cover around 70% of the lakebed; these mats contribute to highly variable water quality conditions, smother kākahi habitat and drive the transfer of phosphorus contained in lakebed sediments back into the water column.

Lake Tuakitoto – rehabilitation options

The greatest priority for managing ecosystem health in Lake Tuakitoto should be focused on reducing nutrient loads to the lake, which would have flow-on effects for several of the other management priorities (e.g. enhancing kākahi populations, controlling macroalgae). In particular, reconnection of the major inflows to the wetland at the top of the lake is likely to

improve the attenuation of sediment and nutrients in the tributaries before they discharge into the lake. This includes Stony Creek, which is presently diverted directly to the lake via a diversion race, thereby circumventing the wetland into which it naturally flowed. Alternatively, realignment of the diversion race to bypass the lake and discharge directly to the lake outlet canal could be another option for removing some of the nutrient and sediment load to the lake. Catchment stream care initiatives focused on improving riparian areas and developing farm plans are also needed to reduce contaminant loads from inflow streams to the lake.

Managing internal nutrient loading in Lake Tuakitoto, which is driven mainly by macroalgae mats, is likely to be complex due to limited knowledge on controlling macroalgae. We recommend trialling sediment capping and macroalgal harvesting options at smaller spatial scales to determine their possible effectiveness before a wider lake-scale intervention can be considered. Other rehabilitation priorities recommended for the lake are to manage European perch, improve native fish populations (i.e. hosts for the parasitic kākahi larval stage), and address fish passage issues caused by the outlet weir and tidal gates near the outlet's confluence with the lower Clutha River (Kaitangata locks).

Tomahawk Lagoon – current state

Monitoring data for the Tomahawk Lagoons collected since 2016 indicate that both lagoons and their inflow creeks breach regional and national water quality guidelines. The lagoons are subject to occasional algal blooms, often caused by cyanobacterial species that can produce toxins harmful to people and animals, especially dogs. Historical data from the 1960s and 1970s shows that Upper Tomahawk Lagoon underwent repeated shifts between a macrophyte-dominated state and a phytoplankton-dominated state. High concentrations of nitrate, dissolved reactive phosphorus and suspended sediment in inflows are the most important contributors to this variability. Grazing by black swans can also contribute to these shifts by reducing macrophytes. The use of empirical catchment and lake models suggests that nitrogen loads will need to be substantially reduced in order for the lagoons to meet total nitrogen guidelines. The lakebed of the lower lagoon also contains substantial amounts of phosphorus, which can be mobilised to the water column by sediment resuspension, anoxic conditions, high pH conditions and microbial organic matter mineralisation. The presence of a self-sustaining population of European perch in the lagoons may also mediate water clarity due to the zooplanktivorous diet of juvenile perch.

Tomahawk Lagoon – rehabilitation options

Restoration options for the Tomahawk Lagoons are somewhat constrained due to the encroachment of urban areas and the fact that the lagoons are managed as a wildlife reserve. The greatest priority will be to manage high nutrient and sediment concentrations in the inflow creeks, which highlights the need for reductions in contaminant flows from land to waterways. This could be achieved by fencing and planting riparian buffer zones along waterways, ensuring forestry blocks provide an adequate buffer zone along waterbodies, and encouraging wetland protection and enhancement in the catchment. In addition, we suggest that the shallow lake area near the inflow from Lagoon Creek could be engineered to be a wetland. The internal loads of nutrients from the lakebeds to the water column are more difficult to control, as dredging and phosphorus capping are likely to have undesired side effects.

Although artificial openings of the sand barrier between the lower lagoon and Tomahawk Beach alter the hydrology of the lagoons, the need for flood management to protect houses and roading infrastructure negates consideration of alteration of the water level regime of the lagoons.

Management of exotic species – including black swans, European perch and exotic zooplankton species – could assist in mediating shifts between turbid and clear water states in the lagoons. However, when swan culling has been undertaken elsewhere by Fish & Game, recolonisation by swans from the larger South Island population has been rapid. Furthermore, the Tomahawk Lagoons are a wildlife refuge, making it unlikely that approval for a long-term programme of swan culling could be obtained.

Monitoring

A range of monitoring recommendations was also made for the lakes. Improving monitoring could assist decision-making regarding rehabilitation options. Monitoring recommendations included monitoring: (1) the inflowing tributaries, (2) wetland function, (3) physico-chemical fluctuations within the lake, (4) kākahi recruitment dynamics, and (5) native fish (both in the lakes and at potential barriers to fish passage).

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GLOSSARY

Term	Definition
Adsorb	To gather a dissolved substance on a surface in a condensed layer.
Anoxia	A complete absence of dissolved oxygen in its free O_2 chemical form.
Bioavailable (nutrient)	The degree to which nutrients are available for absorption and utilisation by living organisms. Nutrients may exist in 'bioavailable' forms, which may be adsorbed by organisms, or in forms that are not bioavailable and hence may not be adsorbed by organisms.
CLUES	Catchment Land Use and Environmental Sustainability (CLUES) model is a catchment model that predicts nutrient run-off to waterbodies based on soil, rainfall and catchment land use. It has been used as a tool to estimate annual nutrient inflows to lakes.
Conductivity	A measure of the concentration of ions in water. Measured as the degree to which a specified material conducts electricity, calculated as the ratio of the current density in the material to the electric field that causes the flow of current.
Eutrophic lake	A lake with high phytoplankton productivity due to excessive inorganic nutrients, especially nitrogen and phosphorus. Eutrophic lakes often experience low dissolved oxygen in their bottom waters due to greater amounts of heterotrophic (e.g. decomposition) processes occurring in their hypolimnion. They are prone to algal and cyanobacterial blooms, which can promote greater respiration by bacteria living in sediments.
Hypolimnion (hypolimnetic)	The dense, bottom layer of water in a thermally stratified lake or reservoir. It is the layer that lies below the thermocline and has little exchange of dissolved gases and solutes with the overlying epilimnion.
Internal loading	Process by which dissolved inorganic phosphorus (phosphate) and nitrogen (ammonium) flux occurs from lakebed sediments into the bottom waters (hypolimnion) of lakes during conditions of low dissolved oxygen. Typically, this results from changes in oxidation–reduction (redox) conditions at the sediment–water interface that favour reduction of particulate-bound nutrients to dissolved (phosphate, ammonium) forms, and results in their solubilisation into the water column.
Kākahi	Freshwater mussel, *Echyridella menziesii.*
LakeSPI	Lake Submerged Plant Index. A monitoring protocol for aquatic macrophytes that evaluates the health of the macrophyte community based on native and invasive macrophyte condition.
Macroalgae	Algae or cyanobacteria that grow attached to the bed and are visible, typically forming mats or long filaments. Can grow to thick levels (> 5 cm) that can cause nuisance issues such as smothering of biota and dissolved oxygen fluctuations.
Macrophyte	A species of aquatic plant that is rooted in the lakebed and generally remains below the waterline. Invasive macrophytes, or weeds, are invasive species that occur in Aotearoa New Zealand (e.g. curly pondweed, *Potamogeton crispus*).
Mahinga kai*	The production and gathering of all foods and other natural resources, and areas where they are sourced.

Mesotrophic lake	A lake with an intermediate level of productivity. Mesotrophic lakes are commonly clear water lakes and reservoirs with medium levels of nutrients and phytoplankton production.
Phytoplankton	Autotrophic (i.e. able to photosynthesise) components of the plankton community. They include algae, cyanobacteria and some mixotrophic groups such as dinoflagellates.
Rohe	District, region or territory.
Solubilise	Make a substance soluble or more soluble.
Sorbtion	A physical and chemical process by which one substance becomes attached to another.
Supertrophic	A water source that has very high algal / cyanobacterial productivity due to excessive inorganic nutrients. These waters are highly prone to regular algal and cyanobacterial blooms and are likely to experience oxygen depletion due to greater levels of heterotrophic (e.g. decomposition) processes occurring in their hypolimnion.
Thermocline	In a thermally stratified waterbody, the location in the temperature gradient where the temperature change per unit distance is maximal. This can persist for months in seasonally stratifying lakes, or for shorter periods of time (e.g. hours) in shallow, polymictic lakes.
Tuna	Eels, including the longfin eel (*Anguilla dieffenbachii*) and shortfin eel (*Anguilla australis*).

Notes

Most te reo Māori definitions in this glossary are from the Te Aka Māori–English, English–Māori Dictionary and Index, which is indexed to the Te Whanake Māori language series.

* This is the Waitangi Tribunal definition for mahinga kai used during the Kāi Tahu claim settlement process.

1. INTRODUCTION

1.1. Coastal lake management in Otago

The Otago Region contains significant coastal lake and wetland environments (Otago Regional Council 2022). In response to concerns about degrading water quality, the Otago Regional Council (ORC) has engaged Cawthron Institute (Cawthron) and the University of Otago to conduct a study into ecological health and processes that control water quality in Lake Tuakitoto and Tomahawk Lagoon. The aim of the study is to provide an understanding of restoration measures that could be employed to improve water quality and safeguard the ecological health of the lakes.

The implementation of the National Policy Statement for Freshwater Management (NPS-FM; MfE 2022) has mandated regional authorities to establish and implement limits on resource use to protect water quality and ecological values of waterbodies. In Otago, as in many parts of Aotearoa New Zealand, agriculture is a widespread land use. Therefore, management of nutrients is expected to be a critical component of maintaining ecological and other (e.g. recreational, aesthetic) values of lakes. While in-lake nutrient processing is well researched, there is considerable complexity (e.g. Moss 1983; Søndergaard et al. 2005; Abell et al. 2011), including important interactions with key biological components (e.g. macrophytes, zooplankton, fish; Jeppessen et al. 2007; Moss 2013). These ecological interactions are often lake specific, and gaining an understanding of the local conditions is helpful when designing lake rehabilitation plans (Howard-Williams and Kelly 2003).

Previous community workshops held in 2018 with the communities of South Dunedin (Tomahawk Lagoons) and Kaitangata (Lake Tuakitoto) identified a need for further understanding of the ecology of coastal lake ecosystems. Specifically the objectives identified for this study were to:

> *Investigate the balance between the needs of human interaction with the wildlife (hydrological function, ecology, wildlife, flood hazard etc). Include assessment of what the limits are for the system in this catchment (tipping point). What are the key stressors and how resilient is the catchment. What actions do we need to undertake to make the catchment more resilient.*

1.2. Background on shallow lake ecology

1.2.1. Eutrophication of shallow lakes

Primary productivity in lake systems is strongly limited by nutrient availability (Søndergaard et al. 2005), and while numerous nutrients are critical, the dominant role of the macronutrients nitrogen (N) and phosphorus (P) in controlling phytoplankton biomass in lakes is well established (Wetzel 2001; Abell et al. 2011).

Phosphorus is mainly bioavailable in the form of the soluble orthophosphate ion (PO_4^{-3}), and the concentration of this ion in natural waters is commonly reduced by its affinity for binding to particulate sediment. Bioavailable N generally occurs in lake waters in much higher concentrations than bioavailable P, and hence phytoplankton biomass may be limited by the lower availability of dissolved P. In addition, N limitation may be circumvented due to the fixation of atmospheric N by various bloom-forming cyanobacteria (Wetzel 2001; Søndergaard et al. 2005). Many studies have demonstrated N limitation in lake systems (e.g. White et al. 1986), and such systems may also be co-limited, experience a change in the limiting nutrient over time, or be limited by light (Burger et al. 2007; Moss et al. 2013).

When catchments are undisturbed and covered by longstanding native vegetation, nutrient inputs to a typical lake tend to be low. In comparison, when a catchment is disturbed, the nutrient and sediment losses to the lake are usually high. In the undisturbed condition, shallow lakes are more likely to have well-developed macrophyte (aquatic plant) communities, which outcompete planktonic algae, stabilise the lake's bed and suppress algal blooms. As nutrient and sediment loads increase, excess nutrients will be taken up by macroalgae that then compete with macrophytes for light. If nutrient and sediment loads continue to increase, macrophytes may collapse, resulting in a destabilised lakebed, more wind-induced sediment resuspension and high levels of available nutrients, which in turn lead to algal blooms. This is a highly simplified, but often useful, conceptual model of shallow lake eutrophication (Figure 1).

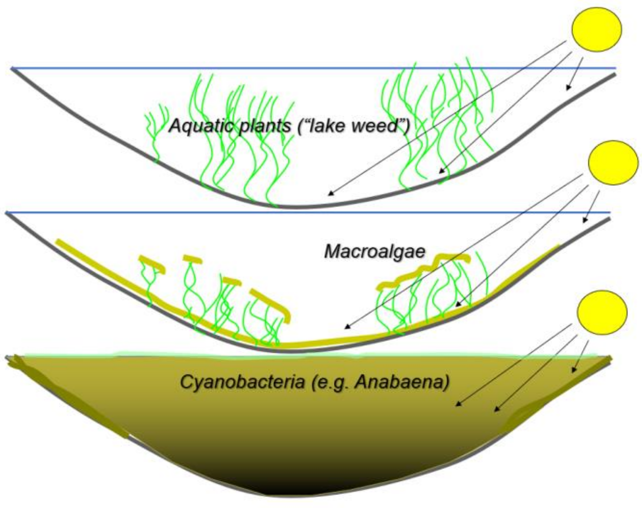

Figure 1. Conceptual model of the shallow lake eutrophication process. When nutrient and sediment loads are low, shallow lakes tend to have abundant macrophyte communities. As nutrient and sediment loads increase, macroalgae use excess nutrients and begin to compete with macrophytes for light. At high rates of nutrient and sediment loading, macrophyte communities collapse, resulting in algal blooms. See text for detailed explanation.

However, rather than a gradual progression, the manifestation of eutrophication in shallow lakes tends to result in sudden changes and shifts in productivity as a result of ecological feedbacks. Negative feedbacks resist change, while positive feedbacks accelerate change. This leads to tipping points, where small changes in a stressor (e.g. nutrient load) result in large changes in degradation (Figure 2). An example of such a feedback mechanism is the effects of macrophytes on the damping of turbulence, lakebed stabilisation, light availability and nutrient availability – all of which favour macrophyte growth over algal growth. Once the collapse of macrophytes occurs (perhaps mediated by macroalgal proliferation), a tipping point is reached and these negative feedbacks disappear, resulting in a shift in conditions favouring algal proliferation (e.g. high nutrient availability, low light penetration, wind-induced sediment resuspension).

Algae grow much faster than macrophytes and are able to benefit more from excess nutrient availability. Some algae (e.g. some cyanobacteria, dinoflagellates) can move vertically in the water column, enabling them to rise up to compete with macrophytes for light. High algal biomass can also reduce light penetration to the lakebed, suppressing macrophyte growth. The loss of macrophytes from a shallow lake allows greater wind-induced turbulence, greater sediment resuspension and higher levels of turbidity, further impeding light penetration into lakes.

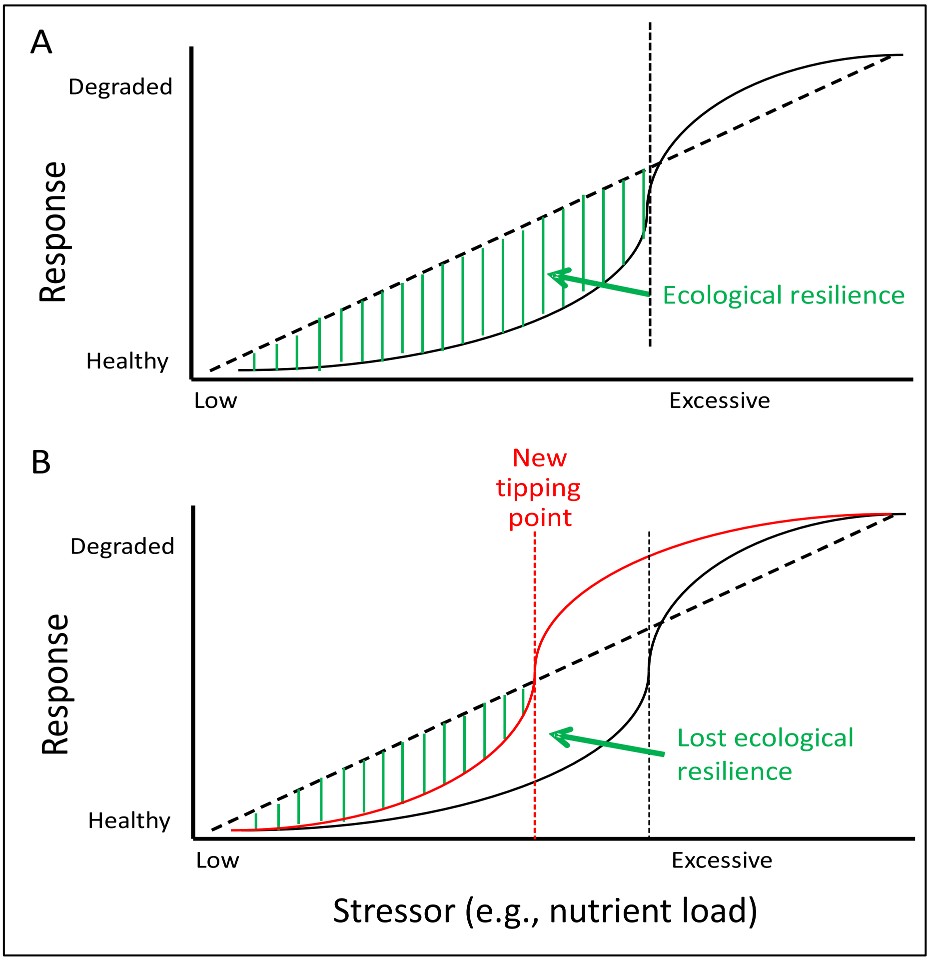

Figure 2. Conceptual model of eutrophication of shallow lakes showing ecological resilience, tipping points and hysteresis. The black dashed line shows a linear stressor-response relationship. The stressor level (nutrient loading rate) relates to a response in the condition of the lake (in terms of algal blooms). The curved black line is the degradation trajectory (as nutrient loads are increased) with feedbacks conferring ecological resilience and resulting in a tipping point (dashed vertical line). The red line is the recovery trajectory (as nutrient loads are reduced). The difference between the tipping points illustrates hysteresis in the system.

This dynamic of alternating stable states is a phenomenon that has been reported for many shallow lakes (Scheffer 2004). In fact, this was observed for Upper Tomahawk Lagoon from data collected in the late 1960s and early 1970s (Mitchell 1989); the data indicated alternating states of dominance by macrophyte biomass and phytoplankton biomass, whereby the states were not just seasonal but lasted over multiple years before shifting to the alternative state. To our knowledge, this dynamic has not been reported for Lower Tomahawk Lagoon, although this may indicate a lack of data rather than the absence of the dynamic.

1.2.2. *Internal loading processes in lakes*

The transport of P from a lake's catchment (external load) – predominantly associated with fertiliser application, eroded sediment or organic material – is the primary source of P in most lake systems. However, once P is in the lake, a significant proportion may settle to the bottom of the lake and be bound in the lakebed sediments, constituting a major reservoir of P (Wetzel 2001). Hence, a key aspect to consider in the management of lake systems is the potential for the recycling of these accumulated 'legacy' nutrients from the sediment back into the water column (Figure 3). This process, referred to as 'internal loading', can greatly increase annual loads of P to the water column, over and above the load from external nutrient sources (Søndergaard et al. 2005; Gibbs 2011). Lake sediments can therefore be both a source and sink for P, and internal loading may delay lake recovery following reductions in catchment loads (Cooke et al. 2005). Knowledge of the forms in which P is bound within sediment and an understanding of the complex chemical and biological interactions that govern nutrient fluxes between the sediment and the water column are critical for identifying eutrophication in a given lake system.

Shallow lakes (i.e. lakes with maximum depths < 10 m) tend to respond to nutrients in a different manner to deep lakes, and thus have their own ecology and management challenges (Scheffer and van Nes 2007). Deep lakes often undergo significant periods of thermal stratification. Shallow lakes are generally well mixed, and when stratification events do occur, they tend to be short-lived and spatially discrete, and often require an additional factor to stabilise the water column, such as the presence of macrophyte beds, salinity incursions or prolonged periods of calm weather (Waters 2016). Even short stratification events may result in rapid deoxygenation of the lake bottom water, and anoxia may result in nutrient release from the sediment.

Excessive algal growth or dense macrophytes may also produce high water column pH. When pH is > 9, there is the possibility of desorption (release) of P from lake sediments and, hence, pH-driven recycling to the water column (Jacoby et al. 1982). In addition, shallow lakes experience frequent resuspension of the bed sediments into the water column, which may release P contained in sediment pore water and / or from the sediment itself (Cyr et al. 2009). The relative importance of these internal loading processes is lake specific. Understanding the occurrence and frequency of bottom-water deoxygenation, high pH and P sorption / desorption dynamics, as well as the manner in which P is bound to the sediments, reveals the key drivers of potential internal nutrient loading.

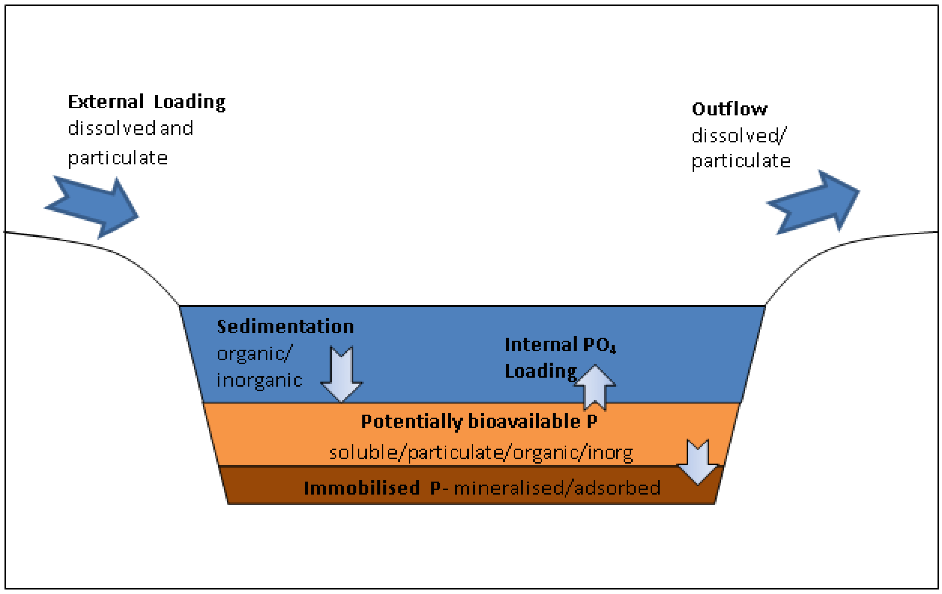

Figure 3. Simplified diagram of lake phosphorus (P) dynamics (adapted from Søndergaard 2007). The blue colour represents water and the brown shades represent lakebed sediments.

Many shallow lakes, with large legacy phosphorus reservoirs, have resisted restoration efforts (in the form of decreased external loads) due to compensatory internal P loading from the lake sediments (Cooke et al. 2005; Schindler 2012). Such internal loading may typically last 10–15 years after reductions of external loads (Jeppesen et al. 2007), and in some instances, may last more than 20 years (Kangur et al. 2013). Results of long-term, whole ecosystem studies indicate that the management of P is the key to controlling eutrophication and remediating lake systems (Schindler 2006, 2012). The identification of lake-specific sources and processes that result in internal P loading will also inform successful restoration programmes and allow targeted and cost-effective interventions.

1.2.3. Coastal lake management issues

Otago’s coastal lakes have high wildlife (waterbird), biodiversity, mahinga kai and sports fishing values. They also provide scenic values within their rural and urban settings, and thus local communities have strong connections to the lakes / lagoons and closely monitor their condition. Key issues with respect to managing water quality, ecology and biodiversity for the coastal lakes included the following points that were identified in workshops:

Lake Tuakitoto

Water level management:

- flooding of land / infrastructure, higher water level regime
- kākahi habitat reduced.

Siltation:

- silt infilling the lake.

Water quality:

- causes of poor water quality.

Biodiversity and aquatic habitats:

- fish passage an issue
- mussel breeding
- mahinga kai and native fish habitat (īnanga, eels, kākahi)
- pest species – Canada geese, willow, other noxious weeds.

Tomahawk Lagoons

Water level management:

- water level regulation of the lagoons
- artificial openings of the lagoons.

Siltation:

- soil erosion and sediment infilling of the lagoons.

Biodiversity and aquatic habitats:

- drivers of occasional algal / cyanobacterial blooms
- stormwater drainage into the lagoon.

A range of environmental monitoring data has been collected by ORC and other organisations such as Ecotago to understand the state of the Otago coastal lakes. Subsequent sections of the report will document and analyse these data, evaluate state and trends relative to regional and national guidelines, and make recommendations for rehabilitation options for Otago lakes.

1.2.4. Otago Regional Council Water Plan standards and national guidelines

To assess ecological state, some comparator attributes and attribute levels are needed. The most appropriate of these are the shallow lake attributes listed in the NPS-FM (MfE 2022) and in Schedule 15 of the ORC Water Plan (Otago Regional Council 2022) (Table 1).

Table 1. Lake condition attributes applying to Lake Tuakitoto and the Tomahawk Lagoons. NPS-FM – National Policy Statement for Freshwater Management (MfE 2022). ORC Water Plan – Regional Plan: Water for Otago (Otago Regional Council 2022).

Attribute	Compulsory value	How measured	How calculated	Type or attribute
NPS-FM				
Chlorophyll-*a*	Ecosystem health (Aquatic life)	• Water column • Monthly samples	• Annual median • Annual maximum • For intermittently open / closed lakes, calculate only for closed periods	Requiring limit setting
Total nitrogen	Ecosystem health (Water quality)	• Water column • Monthly samples	• Annual median • For intermittently open / closed lakes, calculate only for closed periods	Requiring limit setting
Total phosphorus	Ecosystem health (Water quality)	• Water column • Monthly samples	• Annual median • For intermittently open / closed lakes, calculate only for closed periods	Requiring limit setting
Ammonia toxicity	Ecosystem health (Water quality)	• Water column • Monthly samples	• Annual median	Requiring limit setting
Nitrate toxicity	Ecosystem health (Water quality)	• Water column • Monthly samples	• Annual median • Annual 95th percentile	Requiring limit setting
E. coli	Human contact	• Water column • Monthly samples	• Four statistical parameters calculated on monthly data measured over 5 years	Requiring limit setting
Cyanobacteria	Human contact	• Water column • Monthly samples	• 80th percentile	Requiring limit setting
Dissolved oxygen	Ecosystem health (Water quality)	• Continuous monitoring near lakebed or monthly profiling	• Annual minimum	Requiring action plan
Macrophytes (invasive impact)	Ecosystem health (Aquatic life)	• LakeSPI survey every 3 years	• % of maximum potential score	Requiring action plan
Macrophytes (native condition)	Ecosystem health (Aquatic life)	• LakeSPI survey every 3 years	• % of maximum potential score	Requiring action plan

Attribute	Compulsory value	How measured	How calculated	Type or attribute
Otago Regional Council Water Plan				
Total nitrogen	Water quality	• Water column • Monthly samples	• 80% of samples over 5 years meet the threshold	n/a
Total phosphorus	Water quality	• Water column • Monthly samples	• 80% of samples over 5 years meet the threshold	n/a
Ammoniacal nitrogen	Water quality	• Water column • Monthly samples	• 80% of samples over 5 years meet the threshold	n/a
E. coli	Water quality	• Water column • Monthly samples	• 80% of samples over 5 years meet the threshold	n/a
Turbidity	Water quality	• Water column • Monthly samples	• 80% of samples over 5 years meet the threshold	n/a

The NPS-FM classifies the condition of waterbodies into four classes: A, B, C and D. These can be interpreted as indicating excellent, good, fair and unacceptable conditions, respectively. The threshold between the C and D classes is the 'national bottom line'. If a waterbody falls below this, Regional Councils must either identify the lake as one that could not meet the national bottom line for intrinsic reasons, or councils must set in place policies or plans aimed at achieving at least the national bottom line.

1.3. Purpose of this report

To further understand the ecological functions of Lake Tuakitoto and the Tomahawk Lagoons and provide options for the rehabilitation of the lakes, we assess current ecological condition and make recommendations on future management actions to restore the water quality, ecological health of the lakes. Specifically, this report:

- identifies the values and restoration aspirations held by stakeholders
- summarises relevant water quality and ecosystem health data for the lakes and their catchments
- determines key processes and limits that drive the ecological health of the two coastal lakes
- presents recommendations on options for improving lake ecological health and enhancing values for stakeholders (note that this does not include quantitative assessments of costs or benefits).

This report is intended to support the subsequent development of an action plan to improve management and rehabilitation of the lakes.

2. LAKE TUAKITOTO

2.1. Background on Lake Tuakitoto

Lake Tuakitoto is a freshwater wetland situated in the lower Clutha River catchment in South Otago. It has an open-water area of 131.8 ha and a mean depth of around 0.7 m. The lake is modified and is a remnant of a much larger wetland system, which included Lake Kaitangata. Lake Tuakitoto has three main inflowing tributaries: Lovells Creek, Stony Creek and Frasers Stream. Large areas surrounding the lake have been drained and reclaimed for farming purposes, and modifications to the drainage network mean that significant portions of flow from Lovells Creek and Stony Creek now bypass the wetland via a drainage channel that enters the lake directly at its northern end (Figure 4). Control of flows entering the lake via the diversion is achieved by a control gate at the head of the lake, which has been recently upgraded to be electronically controlled. A sill, which is in place downstream of the lake in the outlet canal, controls lake level, particularly at the lower levels. A gate (Kaitangata locks) at the junction of the lake outlet canal with the Clutha River manages flows exiting the lake; this gate is closed during flood periods in the lower Clutha River to minimise flooding of upstream areas. Lake Tuakitoto and the surrounding wetlands perform a valuable hydrological function as a flood ponding area integral to flood management of the lower Clutha River.

The lake and its associated wetlands have high values including:

1. listed as a Significant Wetland in the Clutha District Plan. Described as a rush and sedge swamp, lowland lake, with an artificial water level.
2. habitat for nationally or internationally rare or threatened species, including feeding and breeding habitat for the threatened Australasian bittern / matuku-hūrepo (*Botaurus poiciloptilus*) and banded dotterel / tūturiwhatu (*Charadrius bicinctus bicinctus*). Also breeding area for the marsh crake / kotoreke, (*Porzana pusilla affinis*), spotless crake / pūweto (*Porzana tabuensis plumbea*) and South Island fernbird / mātātā (*Poodytes punctatus*). Habitat for threatened giant kōkopu (*Galaxias argenteus*). The threatened nettle (*Urtica linearifolia*) and *Isolepis basilaris* are present on the swamp margin.
3. regionally and nationally important habitat for waterfowl, waders and swamp birds, including a significant proportion of the national population of mallard (*Anas platyrhynchos*), New Zealand shoveler / kuruwhengi (*Anas rhynchotis variegata*), grey teal / tētē moroiti (*Anas gracilis*) and black swan (*Cygnus atratus*). All these species breed here.
4. kākahi (*Echyridella menziesii*) populations within the lake, which have been found to filter the lake, enhancing water clarity.
5. supporting a commercial eel fishery, as well as recreational fisheries for European perch (*Perca fluviatilis*) and brown trout (*Salmo trutta*). Giant kōkopu (*Galaxias argenteus*) are found in two of its tributary streams.

6. highly valued by Kāi Tahu for cultural and spiritual beliefs, values and uses, including wāhi taoka. Wetland highly valued by Kāi Tahu for its historical associations and as a gathering area.

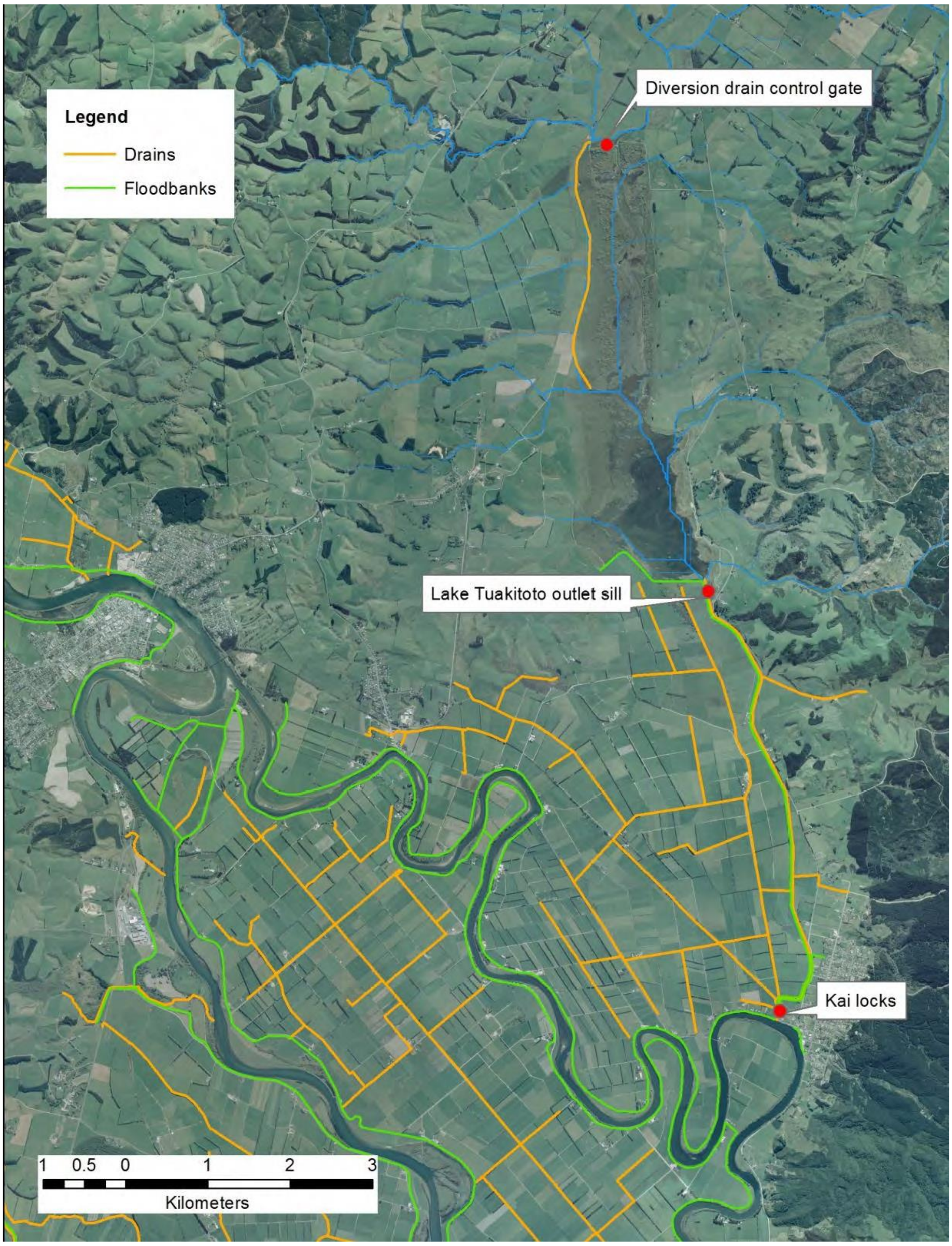

Figure 4. Lake Tuakitoto showing drains (yellow lines), floodbanks (green lines) and locations of drainage scheme structures that may influence lake level (red circles). Source: Ozanne (2014).

2.2. Lake Tuakitoto catchment

The Lake Tuakitoto catchment is 143 km^2 in total area, with the main inflows to the lake coming from the sub-catchments of Lovells Creek, Frasers Stream and Stony Creek. Catchment vegetation is primarily modified pastoral farmland used for intensive grazing (75%) and plantation forestry (15%) (Figure 5; Table 2). Small areas of indigenous forest occur in the upper portions of Frasers Stream, but the most significant remaining area of native vegetation comprises wetland vegetation (3.7 km^2) lying immediately north and east of the lake. The wetland has had significant encroachment by willows (*Salix* sp.).

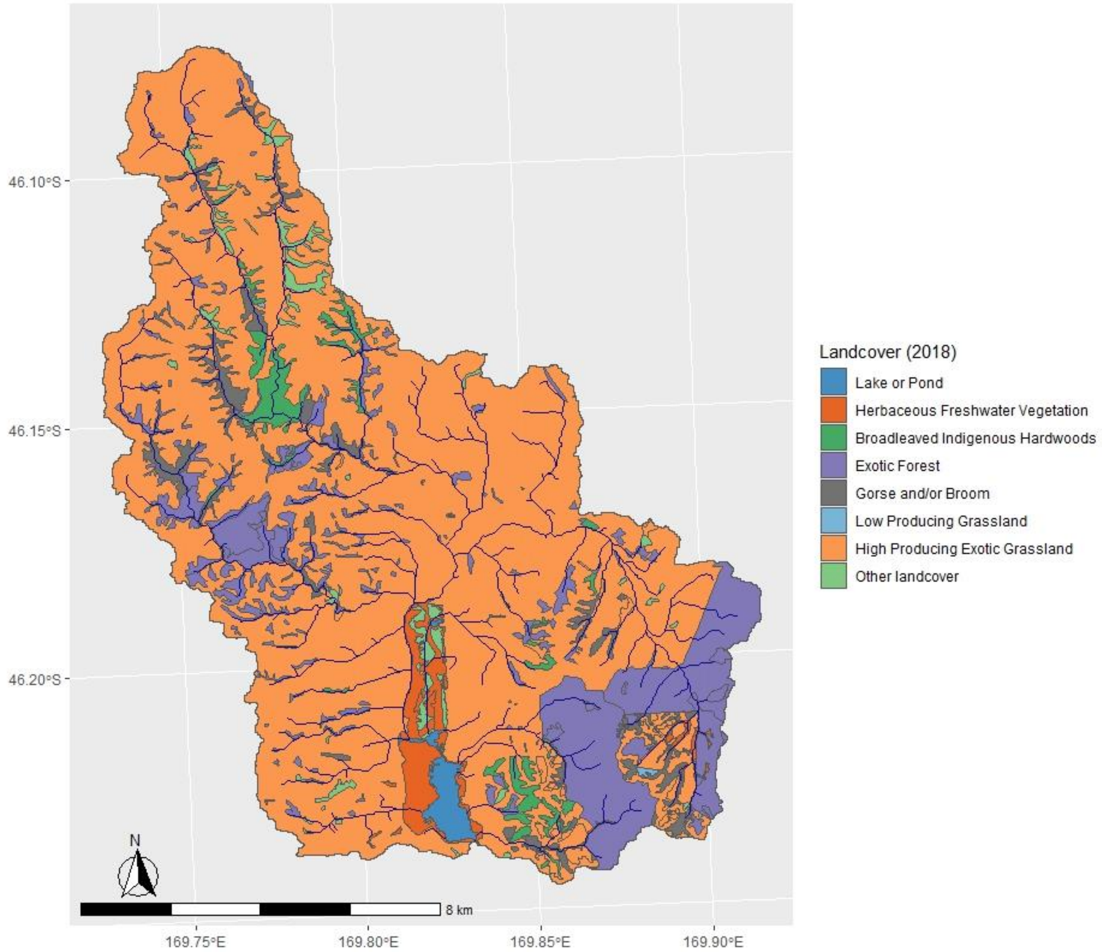

Figure 5. Map of the Lake Tuakitoto catchment and landcover classes based on LCDB5. Source: Landcare Research (2018).

Table 2. Landcover (LCDB5; Landcare Research 2018) for the Lake Tuakitoto catchment.

Landcover category	Area (km^2)	Percent of catchment (%)
Broadleaved Indigenous Hardwoods	3.8	2.6
Deciduous Hardwoods	0.8	0.6
Manuka and / or Kanuka	0.9	0.6
Matagouri or Grey Scrub	0.2	0.1
Gorse and / or Broom	5.0	3.5
Herbaceous Freshwater Vegetation	3.7	2.6
High Producing Exotic Grassland	107.2	74.9
Low Producing Grassland	0.1	0.1
Exotic Forest	21.2	14.8
Built-up Area (settlement)	0.2	0.1
Total	143.0	

2.3. Stream inflows

Intensification of land use in the catchment has affected water quality, particularly in the lower part of Lovells Creek and Stony Creek (Table 3). Several monitoring sites in these streams exceeded ORC Water Plan limits (Schedule 15) for nitrate and *E. coli* as evidenced in detailed monitoring conducted in 2012–13 (Ozanne 2014). The 2012–13 monitoring data also indicated very high concentrations of total nitrogen (TN) and total phosphorus (TP) at sites in the lower portions of Lovells and Stony Creeks. Concentrations of TN (80th percentile) were in some cases 2–4 times higher than lake ecosystem health NPS-FM national bottom-line limits (800 $mg.m^{-3}$ for polymictic lakes) and up to five times greater than NPS-FM national bottom-line values for TP (50 $mg.m^{-3}$). Therefore, it is with high certainty that concentrations of total nutrients to Lake Tuakitoto in inflows are well in excess of meeting NPS-FM national bottom-line values in the lake.

ORC have recently added sites within the Tuakitoto catchment to its stream state of the environment monitoring programme. It is expected that data from this monitoring will provide updated information on water quality in lake inflows to inform the management options that are recommended in this report (Rachael Ozanne, ORC, pers. comm., 26 May 2023).

Table 3. Water quality parameters (80th percentiles) for Lake Tuakitoto river inflow sites (2012–13 data) with receiving water quality limits in plan change 6A. Values that exceeded the limit are in bold type. Values were calculated using samples collected when flows were at or below median flow (0.143 $m^3 s^{-1}$), as this is when Schedule 15 limits apply. Data source: Ozanne (2014).

Site name	TN ($mg.m^{-3}$)	NNN ($mg.m^{-3}$)	NH_4-N ($mg.m^{-3}$)	DRP ($mg.m^{-3}$)	TP ($mg.m^{-3}$)	*E. coli* (100 ml^{-1})
ORC Schedule 15 limit	none	444	100	26	none	260
Lovells Creek						
West Branch (Hillend Rd)	3380	**1956**	30	5	262	**670**
East Branch (Fallaburn Rd)	1078	**700**	5	20	56	84
NW Branch (Fallaburn Rd)	1020	**468**	5	16	105	**3260**
Bloxham Rd	746	384	5	11	40	**568**
Station Rd	792	366	6	15	45	**1300**
West Branch (Hillend Rd)	3380	**1956**	30	5	262	**670**
Frasers Stream						
Elliotvale Rd	334	41	5	13	31	200
Station Rd	592	201	9	11	47	**372**
Stony Creek						
Hillend Rd	2018	**1074**	39	23	156	**754**
Stony Creek (Hillend Rd)	1384	**1032**	5	20	55	**406**
Stony Creek at SH1	924	**480**	5	20	83	**1420**

2.4. Lake Tuakitoto palaeohistory

As part of the New Zealand Ministry of Business, Innovation and Employment research programme – Our lakes' health: past, present, future (Lakes380; C05X1707), sediment cores were collected from Lake Tuakitoto to examine the palaeohistory of the lake. This work involves using sediment indicators of historical water quality and other characteristics that provide information on conditions in the lake dating back to pre-European settlement and, for some lakes, prior to arrival of Māori.

2.4.1. Sediment cores

Pine pollen was detected in the sediment core to a depth of 10 cm, with all records beyond this sediment sub-sample depth (13 cm) containing no pollen from exotic taxa. This suggest that approximately 10–13 cm of sediment has been deposited in the lake in the last 180 years following the arrival of Europeans in the area (Figure 6).
The pollen results from the deepest sample in the sediment core indicate that podocarp forest was prevalent in the region prior to European settlement. It is likely that the landscape was also modified prior to European settlement given the presence

of bracken fern (*Pteridium esculentum*) pollen (common following vegetation disturbance) and charcoal (a proxy for fire) in sediments below 13 cm.

There was also a notable rise in algal pigments post-European arrival, with a steady increase from about 6 cm. The algal signal is difficult to interpret and may have been influenced both by macroalgae and phytoplankton growth in the water column.

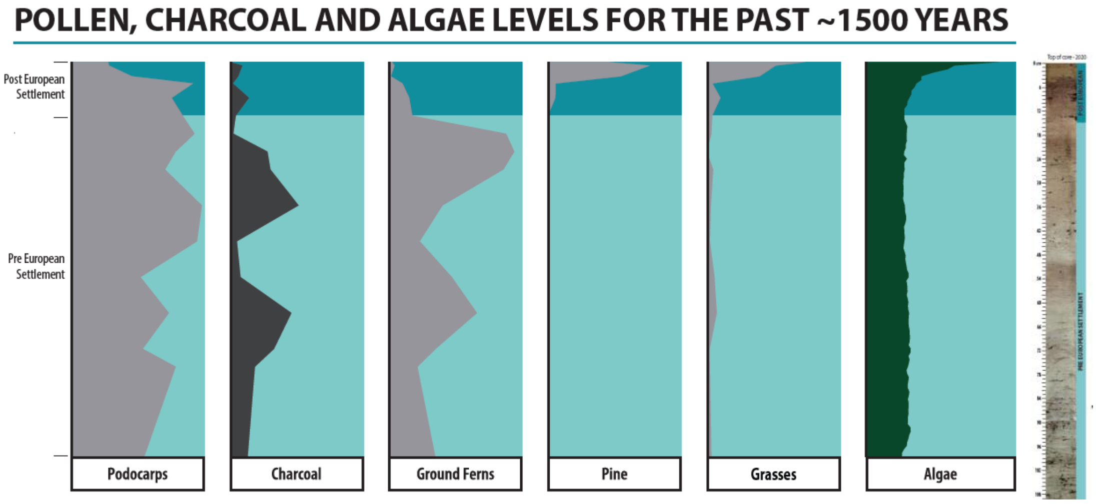

Figure 6. History of catchment vegetation in sediment core records for Lake Tuakitoto showing arrival of exotic pine pollen at approximately 12 cm sediment depth. Also shown is the prevalence of charcoal prior to European arrival, indicating fires in the landscape. Algal pigments are also shown indicating a recent increase in algae associated with benthic macroalgae and / or phytoplankton biomass increases.

2.5. Lake water levels

Water levels in Lake Tuakitoto are managed by the outlet sill as well as the diversion race at the head of the lake, which diverts inflow water directly to the lake during high flow periods. Two main policy measures related to managing water levels in the lake are now in place:

- The Local Water Conservation (Lake Tuakitoto) Notice 1991 set the boundary of the lake area at 101.42 m above datum. This level was set to ensure land outside of the Lake Tuakitoto area was not significantly adversely affected by manipulation of lake water levels. The intent of this was to protect grazing land, which requires good land drainage and flood mitigation.
- A minimum lake level of 100.77 m above datum (0.77 m above sea level) was set for the lake for the period beginning 30 September in any year and ending 16 May in the following year. The intent of this was to protect the regionally significant recreational and wildlife features of the lake. This level was adopted by the ORC Water Plan (Otago Regional Council 2022).

Water level monitoring for Lake Tuakitoto indicates the lake experiences high variation in water levels, with annual variation typically between 100.7 m and 102.5 m (relative to datum), indicating approximately 1.8 m of water level fluctuation (Figure 7). This suggests a relatively large range of variation given that normal water levels provide a mean lake depth of 0.7 m. Increases in minimum water levels were apparent in the dataset after 2013 when the minimum water level was formally adopted into the Otago Regional Water Plan (Otago Regional Council 2022). Prior to this time, minimum water levels, which usually coincide with mid-summer periods, were around 100.6 m between 2002 and 2012, almost 20 cm lower than recent years (2013–23). The recent raising of lake level under the new levels adopted in 2013 will have reduced the areas around the lake margin that regularly underwent drying and will have improved habitats for lake edge-dwelling biota.

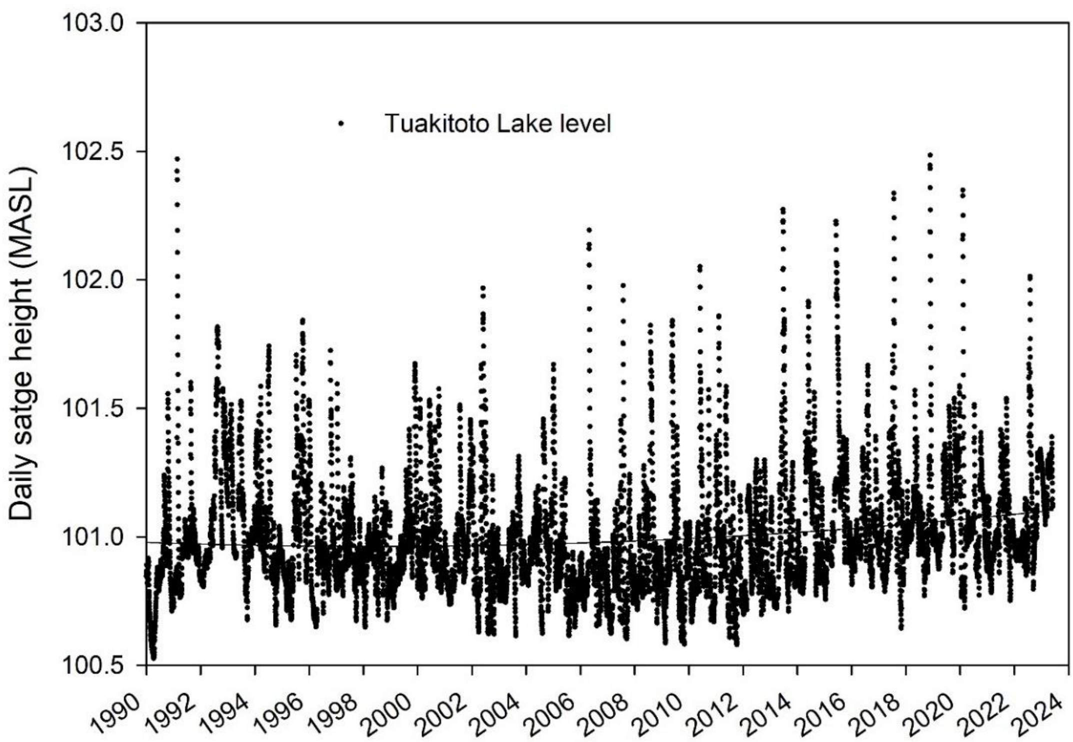

Figure 7. Water levels in Lake Tuakitoto between 1990 and 2023. Data expressed in metres above datum (0.77 MASL).Note that the height of 100.77 is set as a minimum water level for the lake in the Otago Regional Council Water Plan. Data source: Otago Regional Council.

2.6. Lake Tuakitoto water quality

Water quality conditions in Lake Tuakitoto have been monitored by ORC at the lake outlet since 1994, with more regular monthly monitoring occurring since 2013. This includes monitoring of physico-chemical conditions, nutrients, phytoplankton biomass and water clarity. The following sections outline the state and trends of water quality in the lake.

2.6.1. Physico-chemistry

Physico-chemical conditions for Lake Tuakitoto at the lake outlet have been collected as spot-monitoring events since 1994, with readings taken at the time of water sampling at the site. Results from water temperature monitoring indicate that lake temperatures mostly ranged between 4 and 22 °C annually (Figure 8). Based on seasonal Mann–Kendall analyses (which normalises for the month the data was collected), temperature increased over the last 10 years of monitoring (2014–23; Table 4). Therefore, a slight warming of the lake outflows has occurred in recent time, possibly associated with climate variation. However, it is worth noting that spot measurements of surface water temperature are poorly suited to detecting long-term temperature trends because data trends can be influenced by normal daily variation caused by collecting measurements at different times of day. Long-term data of lake water temperature using thermistor loggers would be desirable and provide a better understanding of whether lake temperatures are trending and how this might affect temperature-sensitive biota or metabolic processes in the lake.

Surface water conductivity at the lake outlet has varied considerably over time. Typically, the lake outlet water varied between 150–220 μs.cm^{-1}, but on some monitoring occasions it dropped below 100 μs.cm^{-1}. These low conductivity events are probably associated with floods, when water from the lower Clutha River propagated upstream via the outlet canal that joins the river at the Kaitangata gates. Backflows of water into the lake are complex and depend on flow and tidal conditions in the lower Clutha River, as well as the opening of the control gates that are operated by ORC. Importantly, there were no instances of brackish water being transferred to the lower Clutha River through the canal into Lake Tuakitoto, which remained below 300 μs.cm^{-1} on all occasions. There was a trend for increasing conductivity over the past 10 years of monitoring (2014–23; Table 4), with a median concentration of 183 μs.cm^{-1}. This increasing trend could be associated with increasing nutrient concentrations, or by higher lake temperatures that would promote greater rates of evapotranspiration.

Surface water pH at the lake outlet indicates that pH has been increasing in the lake (Figure 9), and more importantly, at times the pH exceeds 9, the level at which particulate P appears to solubilise (Waters et al. 2020). Although such high water column pH events are relatively infrequent, it is probable that higher pH occurs near the sediment–water interface, which is related to benthic algal primary productivity. The lakebed is known to have extensive coverage of filamentous green algal communities (Ozanne 2014), which can mediate pH shifts near the bed and recycle sediment P to the water column (Goa et al. 2012; Vadeboncoeur et al. 2021). Concurrent increases in water column dissolved reactive phosphorus (DRP) concentrations indicate that this could potentially be driven by high pH from photosynthesis by benthic (and planktonic) algae. Further monitoring of daily pH fluctuations both in surface waters and near the bed of the lake (associated with benthic algae) would provide important information to inform how benthic algal communities may be driving nutrient recycling in the lake.

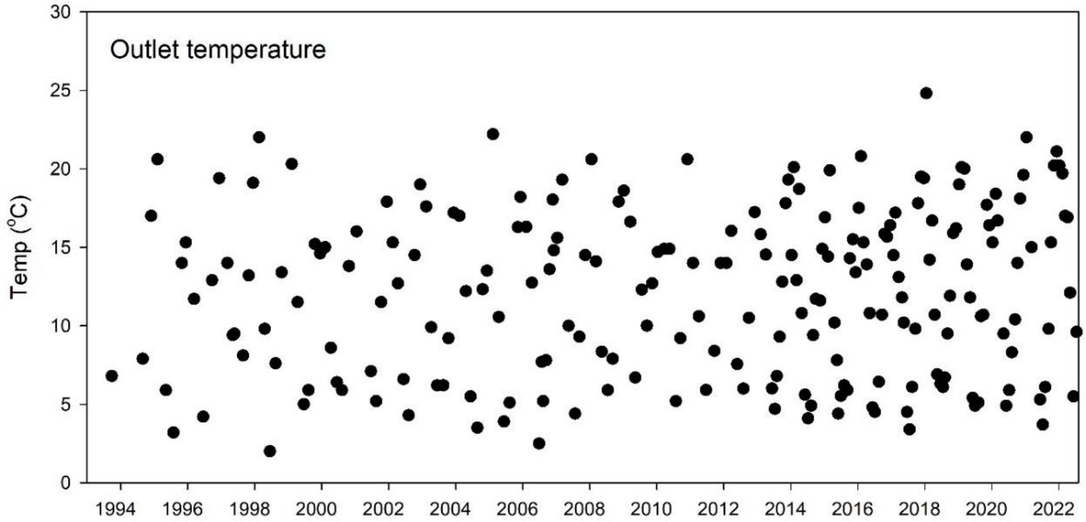

Figure 8. Lake outlet water temperature from spot monitoring between 1994 and 2023. Data source: Otago Regional Council.

Table 4. Trends in water quality parameters in Lake Tuakitoto between 2014 and 2023. Data source: Otago Regional Council.

	Median (2014–23)	Seasonal Mann–Kendall test stat	P-value	Trend
Temperature (°C)	12.80	2.449	0.0143	Increasing
Dissolved oxygen ($mg.l^{-1}$)	8.93 (82.8%)	2.434	0.0149	Increasing
Conductivity ($\mu s.cm^{-1}$)	183.0	2.841	0.0107	increasing
pH	7.60	3.639	0.0003	Strongly increasing
Ammonium-N ($mg.m^{-3}$)	39.00	-1.731	0.0835	Moderately decreasing
Nitrate / nitrite-N ($mg.m^{-3}$)	57.00	-0.764	0.4446	No trend
Dissolved reactive-P ($mg.m^{-3}$)	37.50	3.706	0.0002	Strongly increasing
Chlorophyll-*a* ($mg.m^{-3}$)	5.00	1.840	0.0658	Moderately increasing
Total phosphorus ($mg.m^{-3}$)	103.00	1.934	0.0531	Moderately increasing
Total nitrogen ($mg.m^{-3}$)	1045.00	1.642	0.1000	Moderately increasing
Turbidity (NTU)	6.00	-1.336	0.1814	No trend
Total suspended solids ($g.m^{-3}$)	5.25	-2.661	0.0078	Strongly decreasing

Monitoring data for dissolved oxygen (DO) collected in spot measurements since 1994 (Figure 9) suggest that water column concentrations at the outlet were consistently high (median 8.93 $mg.l^{-1}$) and did not occur at levels associated with oxygen stress to sensitive biota (NPS-FM limit of 4 $mg.l^{-1}$). However, it should be noted that daytime surface water measurements of DO will reflect lake conditions when production of DO is highest and when the water column is expected to show high DO levels. Night-time measurements may differ considerably, particularly for a lake such as Tuakitoto where there is considerable biomass of primary producers. Furthermore, there may be short periods when DO may decline around the benthic mats or at the lakebed, and this would not be measured in outflow. Hence, the spot-monitoring data are poorly suited for understanding oxygen dynamics in the lake and do not necessarily identify if DO conditions are stressful to aquatic life. We would recommend continuous monitoring of DO at multiple levels (i.e. surface, near-bed) over weeks to months during the summer period to gain a better understanding of DO variation in Lake Tuakitoto. Given the extent of benthic algal mats in the lake, it is quite possible DO conditions could vary considerably over depth and time, and this could result in oxygen stress to fish and other species.

Another related factor is that low DO conditions at the lakebed (associated with algal mats) can promote particulate P solubilisation, which occurs under anoxic conditions (Wood et al. 2015). Both DRP and ammonia release could increase water column dissolved nutrients and stimulate algal metabolism and biomass accrual. This is discussed further in Section 2.3.4.

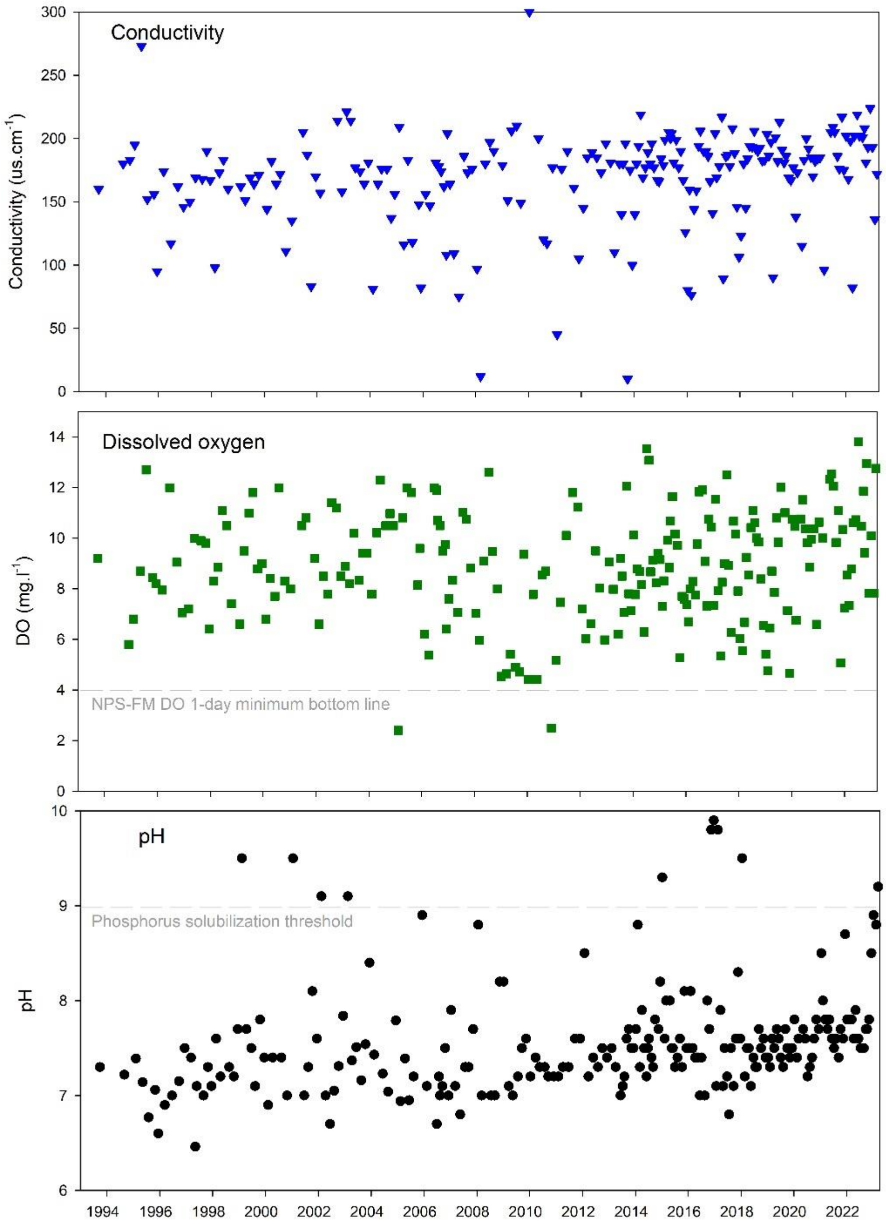

Figure 9. Surface water conductivity, dissolved oxygen (DO) and pH between 1994 and 2023 in Lake Tuakitoto (at outlet). Data source: Otago Regional Council.

2.6.2. Trophic level index

Water quality outcomes for the lake are strongly associated with the nutrient status and phytoplankton biomass in the lake. Burns et al. (2000) developed a monitoring protocol for assessing the trophic status of lakes in Aotearoa New Zealand called the trophic level index (TLI), which is based on measurements of TN, TP, chlorophyll-*a* (chl-*a*), and Secchi disc depth (Table 5).

Table 5. Ranges of total nitrogen (TN), total phosphorus (TP) and chlorophyll-*a* (chl-*a*) concentrations and Secchi disc depth for trophic level states of Aotearoa New Zealand lakes according to Burns et al. (2000).

Trophic state	**TLI**	**Chl-*a* ($mg.m^{-3}$)**	**TN ($mg.m^{-3}$)**	**TP ($mg.m^{-3}$)**	**Secchi (m)**	**Algal and cyanobacteria bloom risk**
Microtrophic	1–2	0.33–0.8	34–73	1.8–4.1	15–25	Very low
Oligotrophic	2–3	0.8–2	73–157	4.1–9	7–15	Low
Mesotrophic	3–4	2–5	157–337	9–20	2.8–7	Intermediate
Eutrophic	4–5	5–12	337–725	20–43	1.1–2.8	High
Supertophic	5–6	12–31	725–1,558	43–96	0.4–1.1	Very high
Hypertrophic	6+	> 31	> 1,558	> 96	< 0.4	Very high

For Lake Tuakitoto, monitoring at the outlet site included three of the four TLI parameters, thus TLI was calculated using only the TN, TP and chl-*a* components, termed TLI3. This has been routinely carried out for other lakes in Aotearoa New Zealand, particularly those monitored by water sampling from helicopters, which does not permit dropping a Secchi disc to look at lake visual clarity. In Lake Tuakitoto, TLI3 mostly fluctuated within the supertrophic range, meaning a very high level of nutrient status for the lake (Figure 10). While we did not test for trends in the annual TLI index, there have been moderately increasing trends for total nutrient and chl-*a* levels in the lake over the past 10 years (Table 4).

In the past year, ORC has moved the monitoring of all TLI parameters in Lake Tuakitoto to a mid-lake site to better align with the national TLI monitoring protocols (Burns et al. 2000); however, insufficient data were available to calculate annual TLI for the lake site.

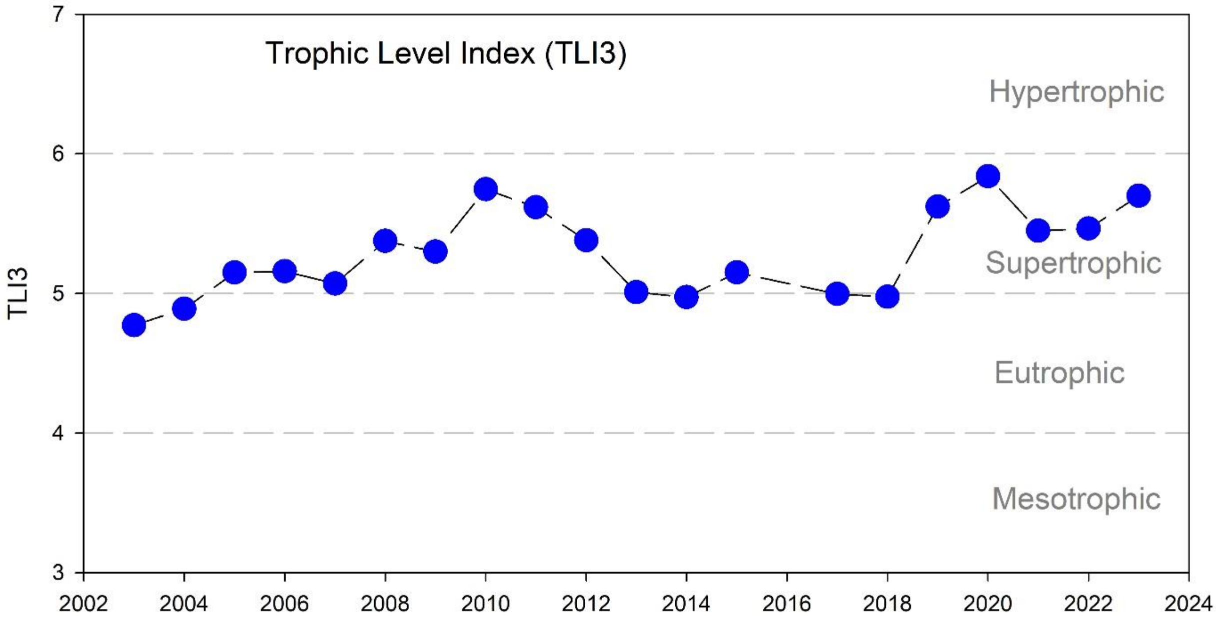

Figure 10. Lake trophic level index (TLI3) for Lake Tuakitoto based on surface water sampling of the lake outlet between 2002 and 2023. Data source: Otago Regional Council.

2.6.3. Total nutrient concentrations

Concentrations of TN and TP have consistently been high at the lake outlet site in Lake Tuakitoto. Over the past 10 years of monitoring, the median concentration of TN was 1045 $mg.m^{-3}$ and the median TP was 103 $mg.m^{-3}$, which are within the supertrophic range (Burns et al. 2000). Both parameters were in excess of the NPS-FM national bottom lines (NPS-FM limits: TP = 50 $mg.m^{-3}$; TN = 800 $mg.m^{-3}$) for the protection of aquatic health over the entire monitoring record between 1996 and 2023 (Figure 11). Based on seasonal Mann–Kendal tests between 2014 and 2023, both TN and TP showed moderately increasing trends (Table 4).

These results show there is a high risk of promoting algal blooms and benthic algal growth, and potentially promoting cyanobacterial-dominated conditions. The very high DRP concentrations (also discussed in Section 2.3.5) would further contribute to risks of cyanobacteria. Because some cyanobacteria can produce toxins and provide limited benefit for animal grazers (zooplankton, kākahi, macrobenthos), there is further risk to the food web of the lake.

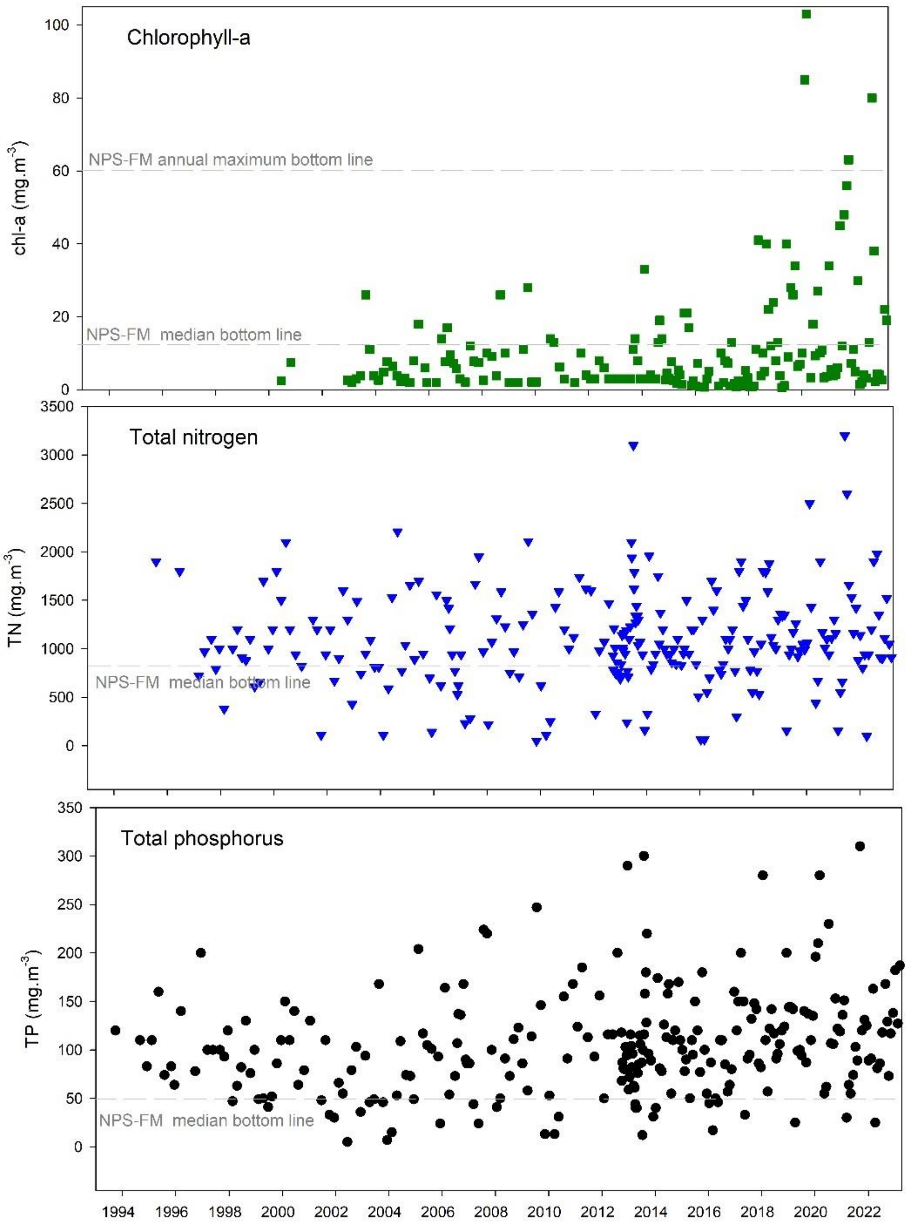

Figure 11. Surface water concentrations of phytoplankton chlorophyll-*a* (chl-*a*), total nitrogen (TN) and total phosphorus (TP) between 1993 and 2023 in Lake Tuakitoto (at outlet). Also shown are national bottom lines for the NPS-FM for the three ecosystem health lake attributes. Data source: Otago Regional Council.

2.6.4. Phytoplankton biomass

ORC have monitored phytoplankton biomass (measured as chl-*a*) at the lake outlet since 1995, and more recently (since 2013) at monthly intervals (Figure 11). Historically, the high nutrient concentrations in the lake have not resulted in associated algal blooms, with a corresponding 10-year median chl-*a* concentration (5 mg.m^{-3}) indicative of meso-eutrophic conditions. Previous research on Lake

Tuakitoto has demonstrated that water filtration by kākahi populations results in high rates of filtration of phytoplankton, thereby maintaining relatively lower algal biomass in the lake and limiting chances for phytoplankton blooms (Ogilvie and Mitchell 1995, Ozanne 2014).

More recent phytoplankton biomass data (since 2018) indicates that the mid-summer chl-*a* can exceeded the annual maximum NPS-FM guidelines of 60 $mg.m^{-3}$, doing so in each of the last four monitoring years. This recent increase in phytoplankton biomass is indicative of changes in lake-wide processes that are likely to negatively affect water quality and are considered in relation to other water quality and ecology variables below.

2.6.5. Dissolved nutrients

Concentrations of dissolved nutrients have been monitored by ORC at the lake outlet since 1995, more recently (since 2013) at monthly intervals. Associated with the timing of increasing phytoplankton biomass in 2018, there have been concurrent increases in DRP in Lake Tuakitoto (Figure 12). Trends in DRP indicate strongly increasing concentrations over the monitoring record, with a 10-year median of 37.5 $mg.m^{-3}$. Normally DRP comprises only a small fraction of the TP pool in shallow lakes (often at undetectable levels) because it is so readily taken up by primary producers. High concentrations of in-lake DRP are often associated with internal nutrient cycling processes, which can occur when particulate-bound P is solubilised under either low oxygen or high pH conditions (Waters et al. 2020). The timing of very high DRP events (in excess of 90 $mg.m^{-3}$) can be seen in obvious peaks, which have occurred during the summers (December to March) of 2015, 2018 and 2020, with smaller peaks in 2019 and 2021. Summer maxima suggest that primary production by algae, which is greatest in warm summer months, is likely to be driving these peaks by increasing daytime water column pH and potentially reducing DO during night-time periods. The dissolution of DRP during such conditions is likely to be accelerating the growth of phytoplankton and benthic algae in the lake in a positive-feedback process.

Concentrations of dissolved inorganic nitrogen (DIN) have been in intermediate ranges, with median concentrations of ammonia and nitrate / nitrite of 39 $mg.m^{-3}$ and 57 $mg.m^{-3}$, respectively. However, at times concentrations of nitrate+nitrite can regularly approach 1000 $mg.m^{-3}$ in winter and spring when water levels are high and uptake by aquatic plants in the lake is lower. Both parameters have had either no trend (ammonia) or a slight reducing trend (nitrate+nitrite) in the past 10 years of monitoring (Table 4). Limits for nitrate and ammonia toxicity to sensitive aquatic life are set in the NPS-FM; however, these national bottom-line guideline concentrations are very high with respect to levels that would promote ecological effects. There were no instances in which the ammonia-N toxicity bottom line was exceeded and only one instance in 2008 when the nitrate toxicity bottom line was exceeded.

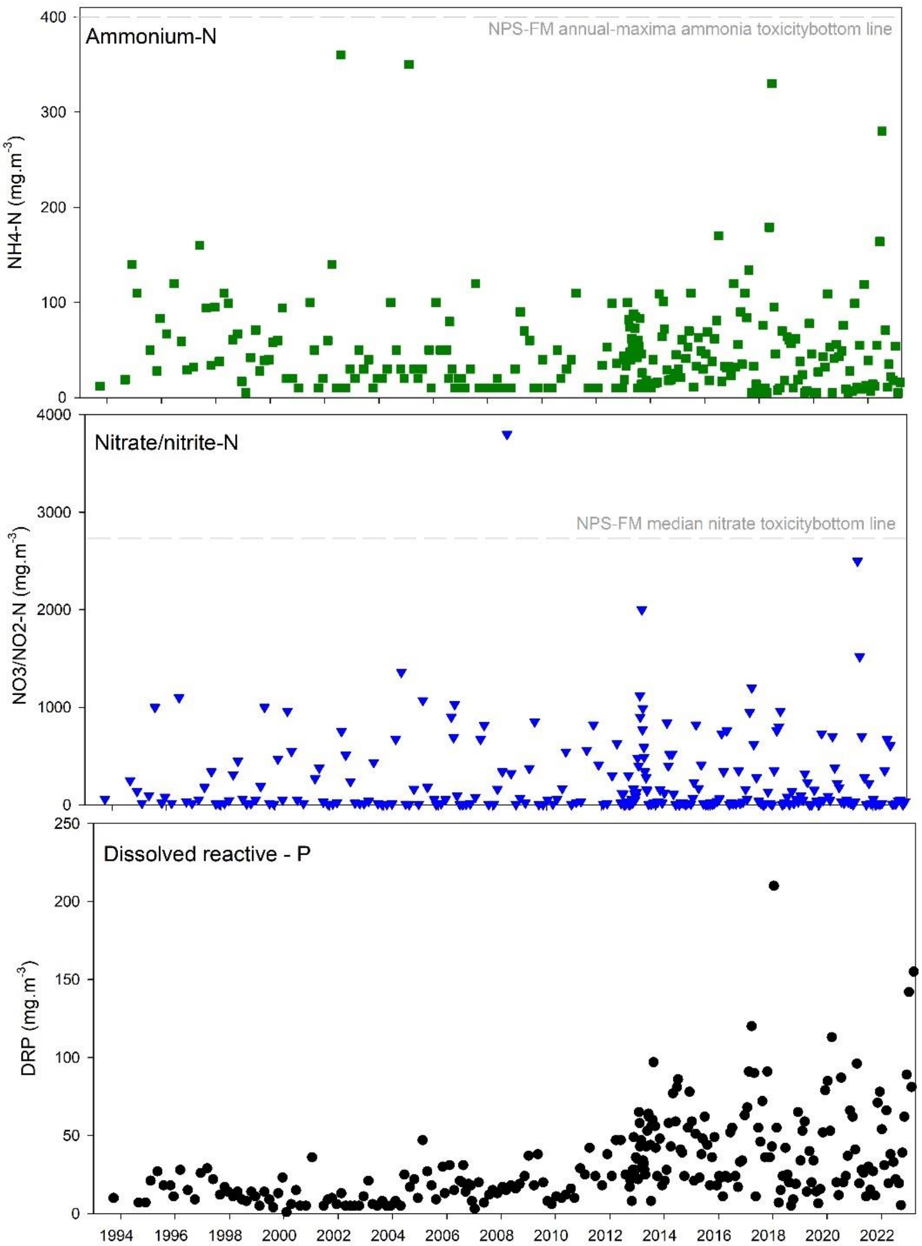

Figure 12. Surface water concentrations of nitrate / nitrite and dissolved reactive phosphorus (DRP) between 1993 and 2023 in Lake Tuakitoto (at outlet). Also shown is the NPS-FM nitrate toxicity median national bottom line indicating severe acute stress to sensitive aquatic species above this threshold. Data source: Otago Regional Council.

Nutrient ratios can be useful for investigating potential nutrient limitation of algal growth in lakes and providing information for restoration measures focusing on the limiting nutrient. DIN:TP ratios 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 a system is increasingly limited by either N (<< than the thresholds) or P (>> than the thresholds), with single nutrient limitation more likely to occur when 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. For Lake Tuakitoto using median data (whole season), ratios of DIN:TP were approximately 0.91 and TN:TP were approximately 9.9, both of which suggest that nutrient ratios are within the co-limitation ranges by N and P. This finding indicates that both nutrients are important for maintaining healthy waterbodies and are occurring in balance with those required for algal growth. Hence, mitigation strategies for managing both N and P should be considered to reduce algal productivity and blooms in the lake.

2.6.6. Lake clarity

Water clarity at the outlet of Lake Tuakitoto has been assessed since 1994 using three different water quality indicators (Figure 13). Black disc clarity, which is a measurement of horizontal sighting distance, was available between 1998 and 2005. Data for black disc clarity suggest that water clarity of lake outlet water was typically quite low, usually less than 1 m, but on few occasions was up to 3.6 m.

Measurements of other water clarity parameters included turbidity and total suspended solids (TSS), and these have been recorded over the entire monitoring record since 1994. The 10-year (2014–23) median values for TSS and turbidity were 5.25 g.m^{-3} and 6.0 NTU, respectively. Both values suggest moderate concentrations of suspensoids (particulate matter) in the outlet water, which limits clarity of lake water. Interestingly, TSS has been significantly decreasing over the past 10 years (no significant trend for turbidity) of the monitoring record, indicating that the clarity of outflows has increased (Table 4). Although causes of this decline in TSS are not certain, we surmise that the increasing prevalence of benthic mats covering the lakebed may be preventing resuspension of sediment during windy periods, which is contributing to lower TSS and increasing the water clarity.

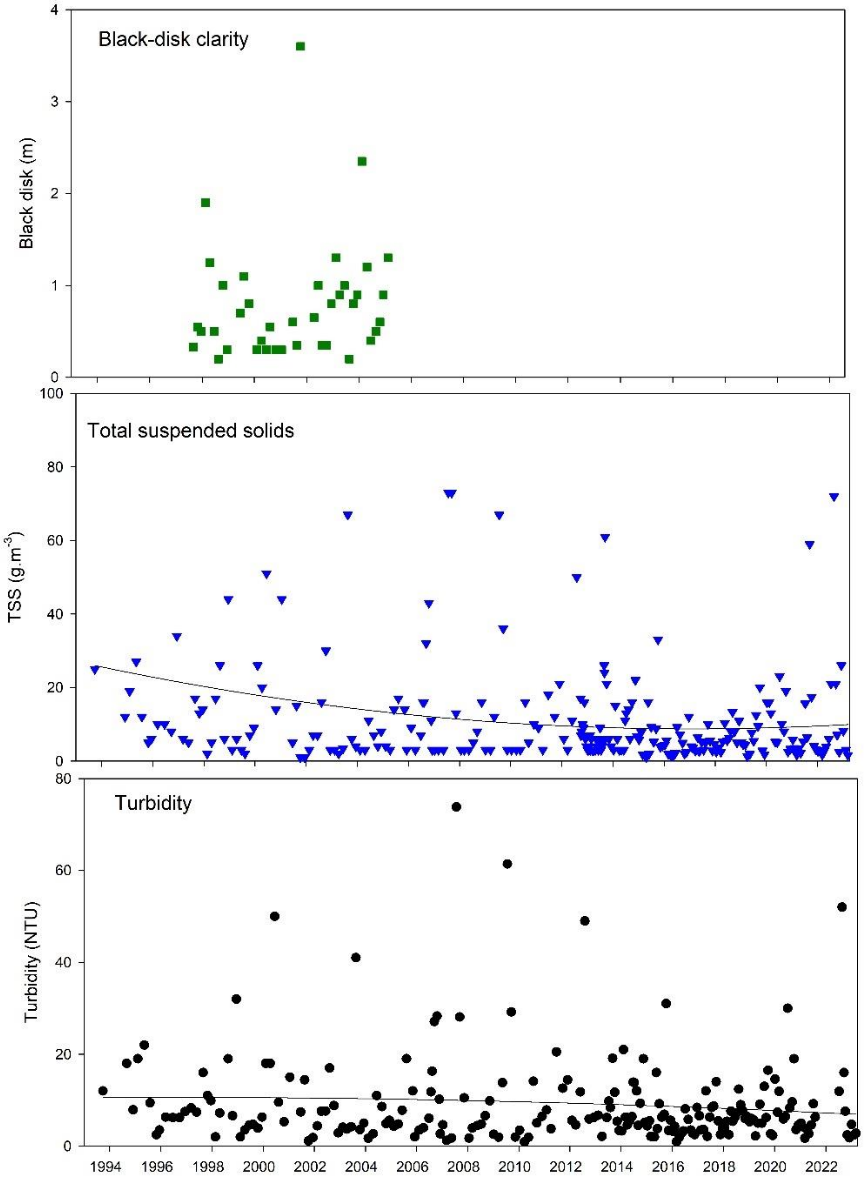

Figure 13. Monitoring results for lake clarity parameters in Lake Tuakitoto between 1994 and 2023. Also shown are polynomial regression lines for total suspended solids (TSS) and turbidity data indicating moderate declines in these variables over the monitoring record. Data source: Otago Regional Council.

2.6.7. Sediment chemistry

Sediment geochemistry data was analysed in six surface sediment samples collected from Lake Tuakitoto (Methods in Appendix 2). A short sediment core (0–24 cm) was also collected from a mid-lake site (K12) using a Uwitec gravity corer. The sediment was sectioned into 2 cm (0–16) and 4 cm (16–24) sections, collected in 50 mL centrifuge tubes and sent to Analytica Laboratories (Hamilton) for total recoverable P analysis. In addition, a surface sediment sample (0–2 cm) was collected in June 2020 as part of the Lakes380 programme. This sample was analysed for P fractionation by sequential chemical extraction (Figure 14) and quantification of P anoxic release rate (Figure 15). The sequential extraction followed the methods outlined in Waters et al. (2023). The release rates were determined by slurry experiments conducted in the Cawthron Laboratory under anoxic conditions and as such represent maximum potential release rate rather than environmentally realistic rates.

This data provides a snapshot of the general sediment geochemistry and is particularly focused on the question of legacy nutrients, especially P. To place the Lake Tuakitoto sediment geochemistry in a national context, we have compared the data to a multi-lake dataset compiled under the Lakes380 programme, including data from 83 shallow (< 10 m) lakes from around Aotearoa New Zealand. When interpreting the Lakes380 Tuakitoto data, it should be noted that geochemistry parameters can display high spatial variability, and hence single samples may not be representative of the whole lakebed.

The Lake Tuakitoto sediments had high bulk densities and low organic matter content relative to the national dataset (Table 6). The exception to this was the southernmost site (K2), which had high organic matter (51%). TN was also low relative to nationwide sediments. These parameters may reflect repeated sediment resuspension, which is common in shallow lakes such as Tuakitoto and can result in decomposition of organic material in the water column, rather than sequestration in the sediments.

Copper (Cu) and zinc (Zn) concentrations in sediments were elevated relative to the national sediment dataset but were below the low trigger values for sediment quality guidelines (ANZECC and ARMCANZ 2000), as were lead (Pb) and cadmium (Cd). Iron (Fe), manganese (Mn), aluminium (Al) and calcium (Ca) are all associated with mineral phases known to bind P in sediments. The Lake Tuakitoto sediments were low in Ca, high in Mn and slightly elevated in Fe and Al relative to the national dataset.

The P content of the Lake Tuakitoto sediments was similar to the national median (Table 6; Figure 14), while the average was lowered due to the low TP content in the southernmost sample (K2 = 460 $mg.kg^{-1}$ dw). When this sample was excluded, the Lake Tuakitoto mean TP (1,408 $mg.kg^{-1}$ dw) was also similar to national shallow lake average. There appeared to be a southward decrease in TP content of the sediment, although this is based on a limited spatial survey. There was a strong correlation between Fe and P content in the sediment (R^2 = 0.89) likely indicating an important role of Fe mineral phases, such as Fe oxyhydroxides, in binding P (Dzombek and Morel 1990; Wang et al. 2013).

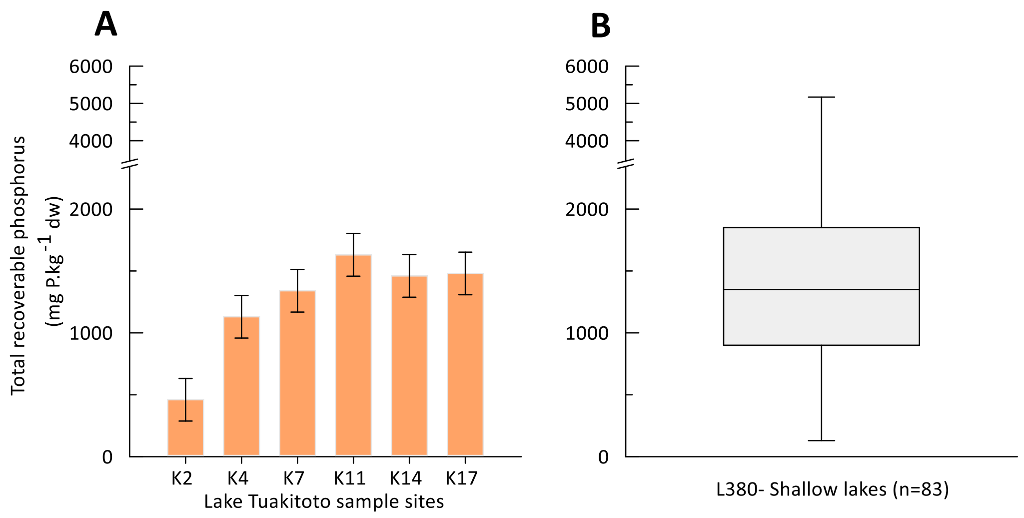

Figure 14. Total phosphorus in the six sediment samples taken in Lake Tuakitoto in this study (plot A). The error bars denote the standard error of the six samples. The box and whisker plots (plot B) show summary statistics (range, 25th and 75th quartiles and median) for total phosphorus contents in the top 0–2 cm of sediment in 83 shallow lakes (< 10 m depth) sampled during the Lakes380 national-scale study. The sample sites in plot A are arranged from south (left) to north (right).

The P fractionation analysis conducted on the sediment sample collected during the Lakes380 programme resulted in a somewhat higher TP than that obtained from total recoverable analyses (Figure 15). This finding is common and is the result of a more robust chemical extraction procedure combined with a degree of 'carry over' whereby a small proportion of P may be reabsorbed and hence reanalysed (Waters et al. 2020). The analysis indicated a very high proportion of redox-sensitive P (56% of TP compared with the shallow lake average and median of 24% and 21%, respectively). This confirms the likely importance of redox-sensitive P-binding minerals such as Fe oxyhydroxides and indicates a strong susceptibility to P release under anoxic conditions. While the pH-soluble fraction appears relatively minor, it is about average for shallow lakes nationwide (Tuakitoto = 16 % of TP relative to a national shallow lake mean and median of 18% and 16%, respectively). It should also be noted that much of the bound P in the redox-sensitive P fraction may also be susceptible to release by elevated pH (> 9.2). The anoxic release rate obtained by laboratory glove-box, slurry experiments indicated Lake Tuakitoto had a high potential release rate, near the 75th percentile of the national dataset (Figure 16).

Table 6. Key statistics for the bulk sediment geochemistry parameters for six surface sediment (0–2 cm) samples collected from Lake Tuakitoto, and the data samples collected during the Lakes380 programme. For comparison, the summary statistics for 83 shallow lakes sampled during the Lakes380 programme are also presented.

		Lake Tuakitoto this study (*n* = 6)				**Lakes380 shallow lakes** (*n* = 83)	
	Unit	Min.	Max.	Mean	Median	Mean	Median
Bulk density	$g.cm^{-3}$	0.18	0.43	0.275	0.241	0.1536	0.12
Organic matter	%	4.90	51.0	15.8	8.6	32.2	28.0
Total P*	$mg.kg^{-1}$ dw	460	1,630	1,250	1,400	1,494	1,340
Total N*	$mg.kg^{-1}$ dw	0.36	0.70	0.55	0.56	1.45	1.39
Fe*	$mg.kg^{-1}$ dw	15,800	40,700	30,883	31,500	31,847	25,300
Mn*	$mg.kg^{-1}$ dw	377	1,610	1,021	994	705.7	433
Al*	$mg.kg^{-1}$ dw	11,300	26,800	20,167	20,300	19,445	17,800
Ca*	$mg.kg^{-1}$ dw	2,430	4,580	3,688	3,815	13,007	5,855
Pb*	$mg.kg^{-1}$ dw	14.3	34.3	26.2	26.7	19.1	16.65
Cu*	$mg.kg^{-1}$ dw	26.9	40.9	34.7	35.8	23.0	17.9
Cd*	$mg.kg^{-1}$ dw	0.11	0.24	0.17	0.17	0.195	0.13
Zn*	$mg.kg^{-1}$ dw	72.5	167	123	123	75.6	60.9
S*	$mg.kg^{-1}$ dw	880	1,900	1,208	1,080	6,654	3,980

* Total P = total recoverable phosphorus, Total N = total recoverable nitrogen, Fe = iron, Mn = manganese, Al = aluminium, Ca = calcium, Pb = lead, Cu = copper, Cd = cadmium, Zn = zinc, S = sulphur, dw = dry weight.

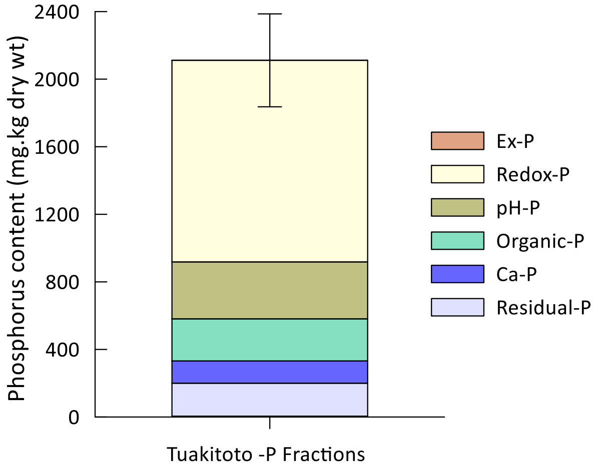

Figure 15. Sediment phosphorus fractions in the surface sediment sample (0–2cm) taken in Lake Tuakitoto during the Lakes380 national-scale study. The error bar denotes a 13% error associated with analysis and sampling.

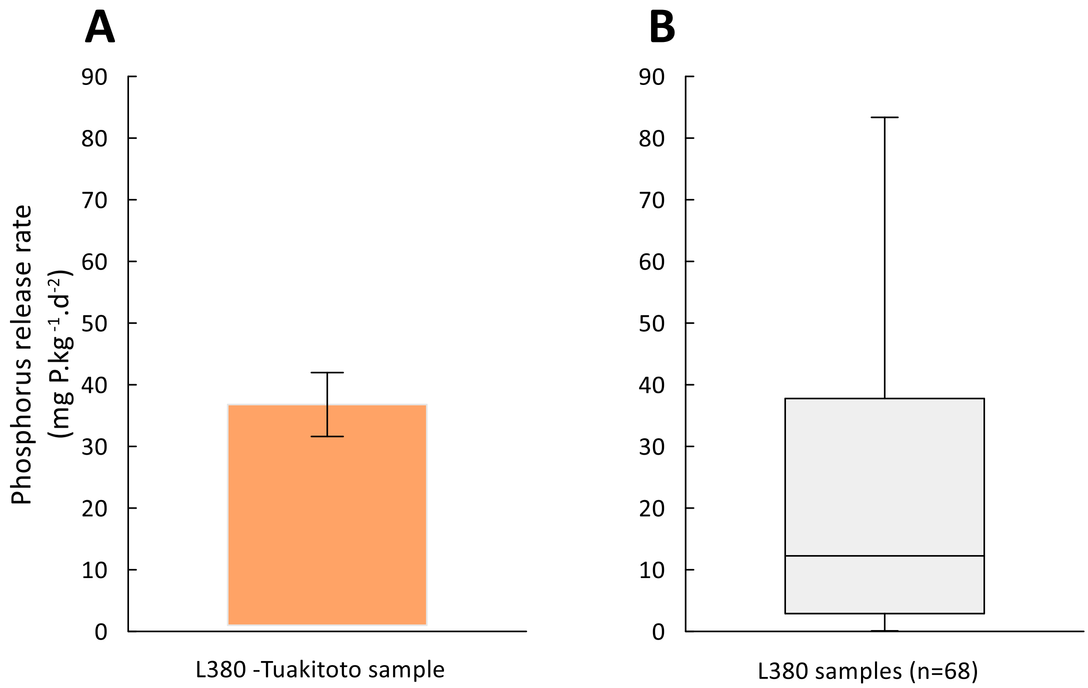

Figure 16. Phosphorus release rates determined by slurry experiment in A) a single surface sediment (0–2cm) sample from Lake Tuakitoto, and B) summary statistics (range, 25th and 75th quartiles and median) on release rates determined for Lakes380 (L380) shallow lake sediment samples (n = 68). The error bar on plot A is the standard error derived from the 68 lake dataset.

The TP profile in the short core from the mid-lake site of Tuakitoto showed a strong downcore decrease in P content, with a particularly steep gradient in the upper 6–7 cm (Figure 17). This pattern may be a result of rapid increases in recent sedimentation of P, but may also be indicative of a flux of P from deeper sediment towards the sediment–water interface, which is associated with anoxia and mobilisation in the deeper sediments (Waters et al. 2020).

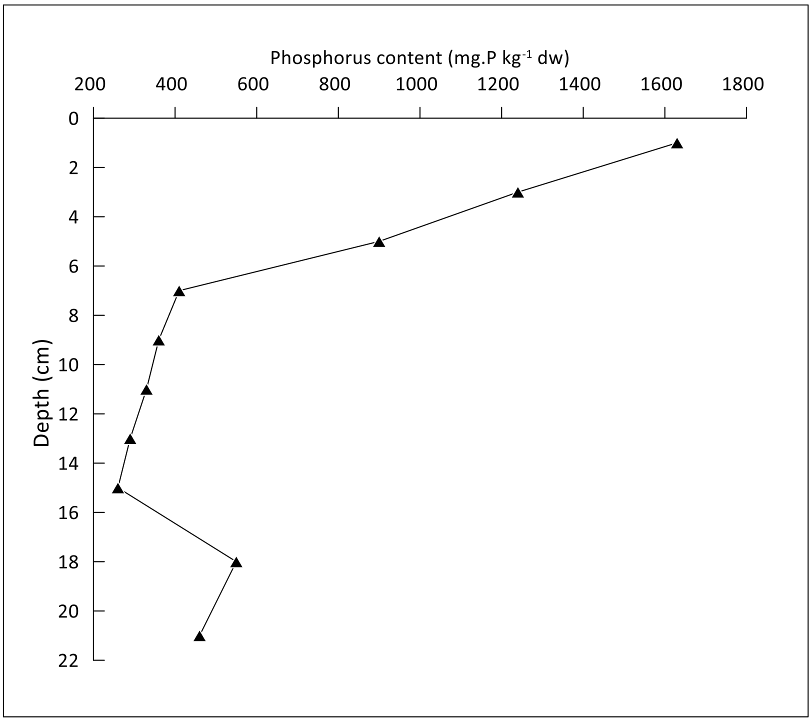

Figure 17. Phosphorus content profile in a short sediment core from the mid-lake sample site K12.

The relatively high P content of the Lake Tuakitoto sediments combined with the very high proportion of the redox-sensitive P fraction, the high potential release rates and the downcore pattern in TP content all indicate that significant internal P loading is likely to be occurring in the lake. This may be driven by high pH (> 9.2), which is observed in the water quality record (see Figure 4) and / or anoxic conditions at the sediment–water interface. Although such anoxia is not evident in the water quality data, it is likely that anoxia as well as high pH conditions exist near the bottom of the benthic algal mats proliferating in the lake (discussed in the next section).

2.6.8. Macrophytes and macroalgae

Hamilton (1990) and McKinnon and Mitchell (1994) reported very low macrophyte biomass in Lake Tuakitoto compared to other shallow lowland lakes that they sampled. In 2006, aquatic plants were surveyed at three shoreline locations as part of a national coastal lakes study (Drake et al. 2010). These surveys of macrophytes and macroalgae were made along transects extending 50 m out from the lakeshore. The study used underwater viewers (shallow areas) and sediment grabs (deep areas), and recorded observations of lakebed cover by different species over the transect. They observed low-moderate coverage of the bed by the macrophytes *Lilaeopsis ruthiana* and *Glossostigma* sp., predominantly in shallow areas along the shoreline. There were also records of filamentous green macroalgae covering portions of the transects, in some cases covering up to 80% of the lakebed (Drake et al. 2009).

During fieldwork conducted in March 2023, aquatic plant surveys were conducted in the same manner as in 2006, along 50 m transects at the same three shoreline sites. There were no submerged macrophytes observed on any of the three transects, with only some *Carex* sp. (sedge) and *Juncus* sp. (rush) emergent plant species observed along the lake edge (Figure 18). Cover by filamentous macroalgae was extensive, with thick mats covering most of the bed area and in some cases extending upwards of 0.5 m from the lakebed. Overall, macroalgae covered 69% of the lakebed, averaged across all the transects. There were obvious growths of macroalgae to the surface of some parts of the lake, and in some cases, mats formed large floating rafts of decaying material that could be expected to provide significant inputs of dissolved nutrients to the lake water column (Figure 19). The floating rafts of macroalgae likely formed as accumulations of material that had originated in areas where mats become buoyant with oxygen bubbles and float off the lakebed. We also observed areas where mats had partly lifted off the lakebed.

The finding of very low macrophyte biomass and cover in Lake Tuakitoto in our March 2023 survey was confirmed by de Winton et al. (2023), who attempted to carry out a systematic macrophyte survey of the lake in June 2023. They reported that the lake had < 10% cover of macrophytes and 30–80% cover of filamentous green algae, which also exhibited as surface-floating mats. Isolate sprigs of the macrophytes *Elodea canadensis*, *Ruppia polycarpa* and *Potamogeton ochreatus* were found. The authors reported that the dense macroalgal cover likely impacted other benthic organisms. They also suggested that a deepening of the lake could benefit the macrophyte communities of the lake. The LakeSPI (Lake Submerged Plant Index) score reported for the lake was 0 (functionally devegetated).

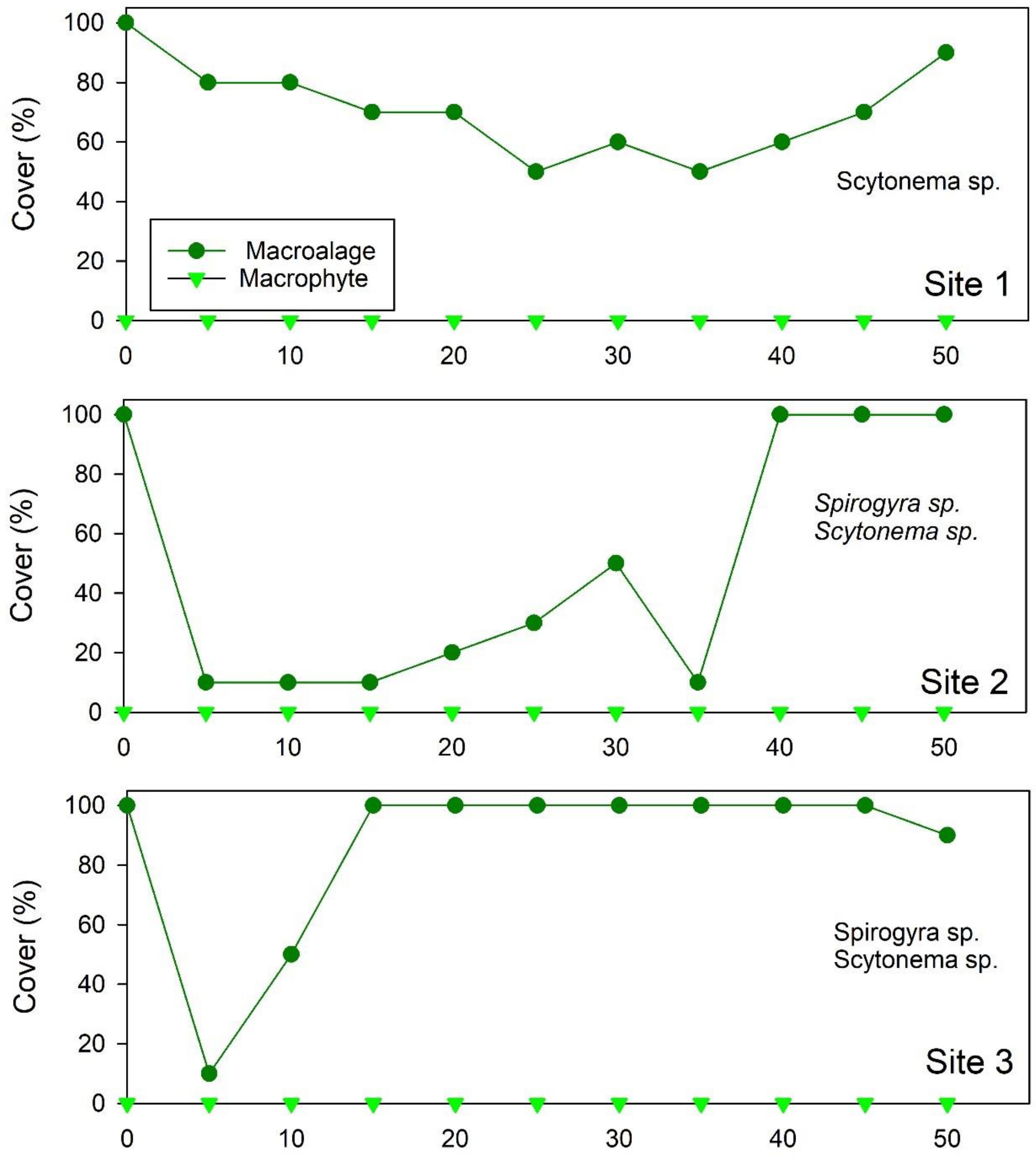

Figure 18. Cover of the lakebed by macroalgae based on surveys of three sites in Lake Tuakitoto in March 2023.

Although it was not the intention of the field survey to focus in detail on benthic macroalgae, a preliminary investigation of their coverage and species composition was conducted due to their prevalence in the lake. Mats were comprised mainly of two types: long, bright green filaments of the genus *Spirogyra* (Chlorophyta); and thick olive-brown filamentous mats, which predominantly comprised the cyanobacteria taxa *Scytonema* sp. (Figure 19). The filamentous green alga *Spirogyra* is known to form nuisance growths in lakes, is highly resistant to grazers and can modify food web pathways to higher trophic levels such as fish (Stewart et al. 2021; Vadeboncoeur et al. 2021). *Scytonema* mats appear to comprise a much greater proportion of macroalgae cover in Lake Tuakitoto, with some species (e.g. *S. cripsum*) known to produce saxitoxin, a cyanotoxin (Smith et al. 2012). Benthic mats dominated by cyanobacteria are found in many aquatic habitats (Scott and Marcarelli 2012); they have been documented in lakes (e.g. Smith et al. 2012; Wood et al. 2012) and alpine tarns (Novis and Visnovsky 2011), including several Canterbury high country lakes (Smith et al. 2012). The taxonomic composition of *Scytonema* mats in Lake Tuakitoto and other associated cyanobacteria-inhabiting mats were not explored in detail during this study, and there is poor knowledge of the seasonal variability in cover and biomass in the lake. In order to better understand these dynamics, we highly recommend a more detailed monitoring of macroalgae, including quantifying its seasonal cover, biomass and species composition.

Figure 19. Images of benthic macroalgae prevalent in Lake Tuakitoto during the March 2023 Cawthron field surveys. *Spirogyra* sp. is a long filamentous green algae that can dominate benthic habitat in some lakes (e.g. Lake Baikal) (Vadeboncoeur et al. 2021). *Scytonema* sp. is a filamentous cyanobacteria species that has been recorded in other South Island lakes, with some species capable of producing saxitoxins (e.g. Smith et al. 2011, 2012). Also noteworthy was the prevalence of large floating rafts of macroalgae, which likely accumulated from areas of the bed where macroalgal mats become buoyant and float to the surface.

2.7. Lake food web

2.7.1. Kākahi

Historical surveys have been conducted on kākahi (freshwater mussel) populations in Lake Tuakitoto (Ogilvie 1993), including the quantification of kākahi filtration rates due to the large populations that were observed in the 1990s (Ogilvie and Mitchell 1995). It was concluded that the lake water filtration by kākahi was a major driver of water clarity in Lake Tuakitoto, with the entire volume of the lake filtered in only a 32-hour period (Table 7). This rapid filtration rate had a stabilising effect on phytoplankton biomass in the lake, reducing phytoplankton biomass relative to the lake's nutrient status.

As part of a wider monitoring investigation into the Lake Tuakitoto catchment, ORC reconducted the kākahi survey at the same 26 sites in 2013 (Ozanne 2014). The 2013 survey showed that mussel biomass was reduced by 52% compared to the 1991 survey, and lake filtration rates declined from 32 hours to 102 hours. These changes were thought to result from both a decline in overall density of kākahi and an increase in the portions of the lake devoid of mussels. The absence of mussels at previously occupied habitats was associated with extended periods of low water levels in the lake. Since the 2013 monitoring, water level management of the lake has been modified to a provide higher summer minimum water levels, which should benefit kākahi.

In March 2023, kākahi surveys were reconducted at the same 26 sites (see Appendix 1 for sites); however, concerns regarding high *E. coli* levels and extensive cyanobacteria meant that dive surveys were not undertaken by snorkellers, and instead abundances were estimated from video surveillance of multiple quadrats on the lakebed. Surveillance of kākahi by video quadrats has been used in other monitoring investigations, and some research has been undertaken into understanding the differences in effectiveness between the two methods. It was reported that monitoring data using the two different methods followed similar trends, but the video surveillance reported slightly lower counts of mussels because some were buried in the sediment. The density estimates of mussels were roughly 20% lower using video quadrats compared with finger sifting by snorkelling (Nuri et al. 2022).

Observations of kākahi at 22 sites in Lake Tuakitoto during March 2023 (four sites could not be assessed due to high turbidity) indicated that significant declines in the abundances of kākahi have occurred since the last survey in 2013 (Figure 20). Mean abundance of kākahi in 2023 was 0.75 per m^2, approximately 65% lower than in 2013 and 86% lower than in 1991 (Table 7). After adjusting the values upwards by 20% to account for fewer mussels being detected using video surveillance (Nuri et al. 2022), these declines are marginally lower, approximately 59% lower than in 2013 and 84%

lower than in 1991. Because kākahi were considerably larger in size in 2023, this meant that declines in the overall biomass of kākahi were slightly lower, which was estimated to be on average 57% lower in 2023 than in 2013. The most obvious declines in abundances were for populations located in central portions of the lake, which historically had abundances of kākahi of up to 27 individuals per m^2, whereas recent monitoring of the central areas found highest densities were between 1–2 kākahi per m^2. Significantly lower abundances of kākahi at shallow sites, noted during 2013, did not occur in 2023. Few sites in the 2023 survey had shallow depths, with only two sites having water levels less than 0.5 m (in 2013 monitoring, 18 sites had water depth < 0.5 m). Hence, it did not appear that shallow water depth was a significant factor contributing to lower kākahi abundances.

The prevalence of thick macroalgal mats appeared to have a negative effect on kākahi abundance. While the macroalgae made it much harder to detect kākahi in video quadrats, considerable time was spent by the field team scooping bed materials using kick-nets in macroalgal covered areas, but this did not yield any kākahi detections. Kākahi seemed to avoid areas of the lakebed covered with thick macroalgae. Effort was made to take video images in areas with low macroalgae abundance to improve chances of detecting kākahi, which likely introduced some sampling bias into our survey. However, even with this bias towards detecting more kākahi in clearer parts of the lakebed, there were still strong declines in abundances compared with previous surveys.

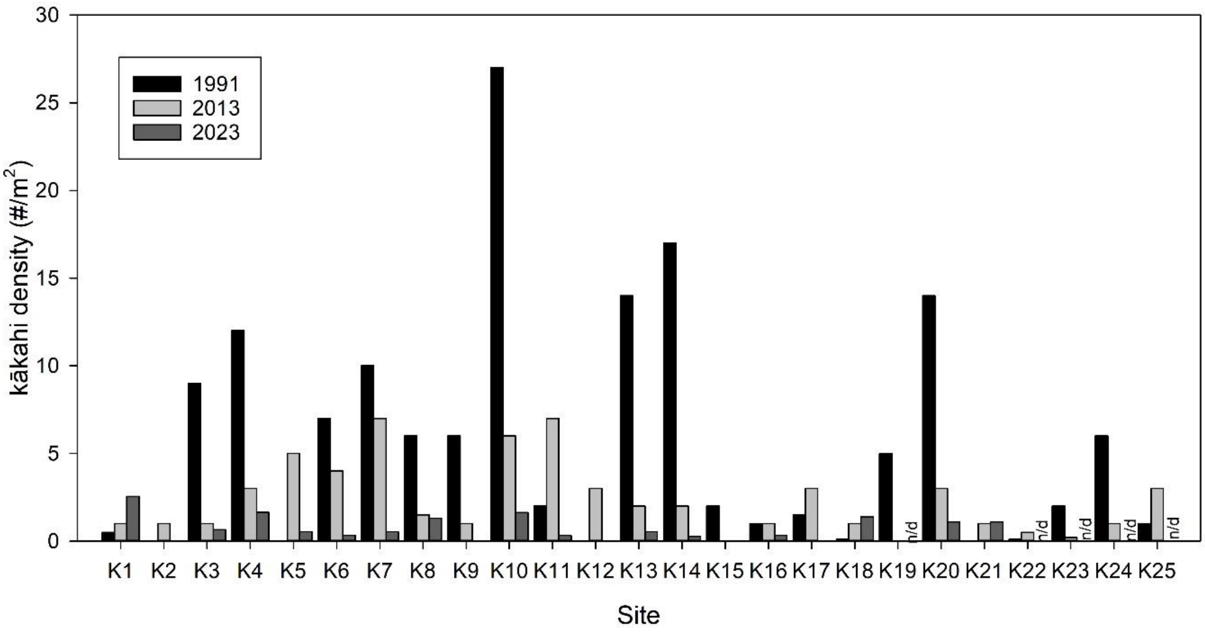

Figure 20. Abundances of kākahi observed at 26 sites in Lake Tuakitoto in 1991 and during a re-survey in 2013 and 2023. Data source: Ozanne (2014).

Table 7. Comparison of mussel filtration calculations between 1991, 2013 and 2023 kākahi surveys. Assumptions on filtration rates were based on Ogilvie and Mitchell (1995). Data source: Ozanne (2014).

Statistic	**1991**	**2013**	**2023**
Area (km)	1.18	1.18	1.18
Depth (m)	0.7	0.7	0.7
Volume (m^3)	826,000	826,000	826,000
Stations with no mussels	2	4	4
Stations less than 0.5 m depth	4	18	2
Mean mussel abundance per site (m^2)	5.5	2.2	0.75
Mean biomass per site ($g.m^{-2}$)	12.3	5.86	2.51
Mean filtration rate ($l.hr^{-}1.g^{-1}$)	1.91	1.26	0.54
Active filtering (%)	0.93	0.93	0.93
Volume water filtered ($m^3.hr^{-1}$)	21.85	6.87	2.94
Time to filter entire lake volume (d)	1.3	4.3	9.9

Declines in biomass of kākahi in 2023 had flow-on effects on predictions of lake water filtration by the kākahi population. Filtration rates from the original 1991 surveys (Ogilvie and Mitchell 1995) were used to calculate filtration rates that took into account mussel size distribution across the 22 sites surveyed. Using the 2023 population data, we estimated that lake filtration rates have further reduced to 2.94 m^3/hr, meaning that the whole lake water column is now filtered in approximately 10 days. This filtration rate is approximately 2.3 times longer than in 2013 (4.3 days) and 7.6 times longer than in 1991 (Ozanne 2014). While the 2023 filtration rate indicates relatively rapid clearance of phytoplankton, filtration capacity is now considerably lower than historical estimates, which means that phytoplankton have a greater period for biomass accumulation and there is potential for blooms when phytoplankton growth rates become very rapid (Ogilvie 1993). Because phytoplankton can double in a single day during high-growth periods (i.e. blooms), kākahi filtration would now not be capable of controlling phytoplankton growth in Lake Tuakitoto. This is consistent with recent increases in the lake's phytoplankton biomass observed in ORC monitoring data.

Because kākahi are so effective at stabilising phytoplankton, further declines in populations would be expected to result in direct increases in phytoplankton biomass and productivity. Because algal productivity can have a self-reinforcing (positive-feedback) effect on nutrient concentrations through influencing pH and DO, it is important to maintain kākahi populations. There are currently high concentrations of water column total and dissolved nutrients in Lake Tuakitoto, and large sedimentary pools of P could be solubilised if algal productivity is high enough to cause pH and DO shifts. There is still a relatively poor understanding of whether recent trends of increasing pH and high DRP are driven predominantly by benthic algae (acting at the sediment–water interface), phytoplankton (acting over the whole water column); or a combination of both. Our observations of the extensive coverage of macroalgae mats would suggest this is the most important factor controlling these dynamics.

Kākahi age structure and recruitment

Kākahi shell lengths were measured at multiple sites by collecting lakebed material using a 500 µm kick-nets from the boat. In some instances, kākahi could be observed on the bed and were directly scooped up in kick-nets, but more passive towing of nets over the lakebed was also effective in collecting mussels. In total, 48 individuals were collected for measurement of shell length to better understand the population size structure.

Shell lengths of kākahi varied between 46 mm and 108 mm, with the greatest number of individuals measured between 70–90 mm shell length (Figure 21). This indicated that the population consisted mostly of large mature adult individuals, with no individuals within the juvenile size range of less than 38 mm (James 1985). Based on size distribution data from the 2013 kākahi survey (Ozanne 2014), individuals of greater than 60 mm length were only found in Lake Tuakitoto, suggesting a highly skewed age structure towards older kākahi. A considerably higher proportion of individuals in smaller size ranges would be expected if kākahi populations were successfully breeding and good growing conditions were available for young kākahi. However, this was not the case for the Lake Tuakitoto population, which had no immature individuals and was comprised mostly of old mature mussels. This raises concern that conditions in the lake are not resulting in successful breeding, or that habitat conditions are not supporting the growth of juvenile kākahi and their development to early adult stages.

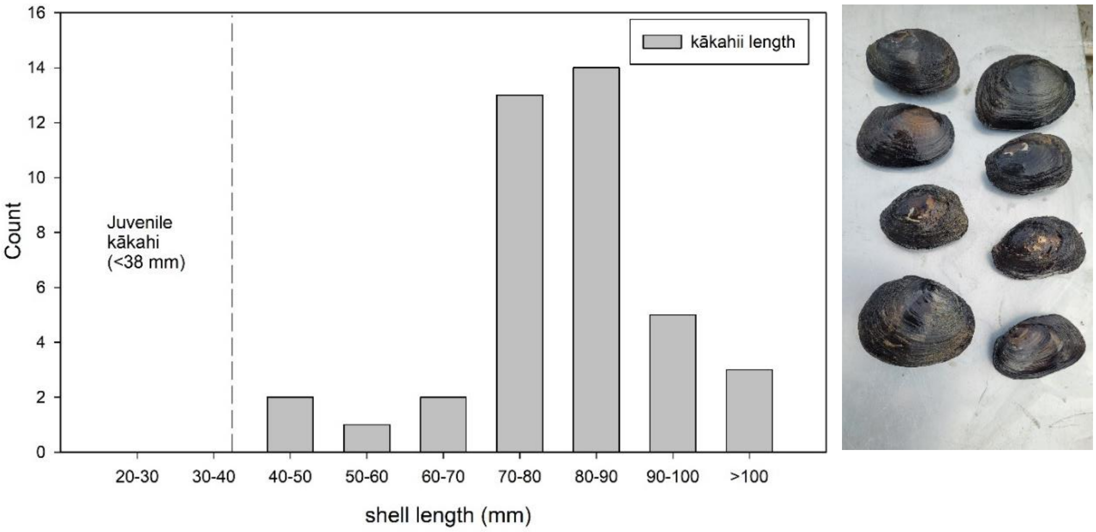

Figure 21. Measurements of shell length of kākahi in the March 2023 survey at 22 sites and picture showing the large kākahi size range, up to 108 mm shell length. Photo: David Kelly, Cawthron.

General declines in once abundant populations of kākahi have been observed across many Aotearoa New Zealand lakes (Grainger et al. 2014). The decrease in kākahi populations has been attributed largely to declining water quality and potentially to changes in the productivity and composition of phytoplankton, the main food source for these filter feeders (James 1985). Less is known about the importance of early life-history stages to kākahi population dynamics and how this can affect recruitment of kākahi populations. Kākahi have complex life histories, which include larval stages and different habitat requirements for juvenile and adult mussels. Glochidium larvae are released from kākahi parasitise fish. The larvae attached to the gill lamellae or skin of a fish before releasing as juveniles, moving to soft sediments of the littoral zone to filter feed until maturity. It is known that adult kōaro (*Galaxias brevipinnis*) and common bullies (*Gobiomorphus cotidanus*) are hosts for kākahi glochidia (juveniles). However, since the introduction of exotic piscivorous fishes, such as European perch and brown trout, to many Aotearoa New Zealand lakes, native host species can be greatly reduced in abundance. The observation that kākahi populations in Lake Tuakitoto are decreasing and lack significant recruitment raises serious concerns for the population and suggests that intervention may be prudent (discussed in Section 2.6)

2.7.2. *Native fish communities*

Fish communities have been monitored in Lake Tuakitoto since the 1980s, and data are available in the New Zealand Freshwater Fisheries Database (Table 8). Catch data from these historical fish surveys indicate that the most abundant species reported were shortfin eel, European perch (*Perca fluviatilis*) and brown trout, with occasional records of common bullies (toitoi, *Gobiomorphus cotidianus*), giant kōkopu

(*Galaxias argenteus*), shortfin eel (*Anguilla australis*) and longfin eel (*Anguilla dieffenbachii)*. Somewhat unusual for the reported catch data in Lake Tuakitoto were the relatively low occurrences of catches (and low abundances) of smaller native fish species such as common bullies, īnanga (*Galaxias maculatus*) and kōaro (*Galaxias brevipinnis*), which were rarely or never recorded in the catch records. These species often comprise the greatest proportions of fish catch records for other lowland coastal lakes (Drake et al. 2010).

Table 8. Summary of fish catch data from the New Zealand Freshwater Fisheries Database for Lake Tuakitoto between 1984 and 2020. Source: NIWA – NZFFDB.

Year / month	**Organisation**	**Method**	**Species caught**	**Mean relative abundance (fish/net/d)**
1984–85	Fish & Game NZ	Fyke nets	Shortfin eel	31.7
May–July		Gee-minnow traps	Longfin eel	2
			Brown trout	2.7
			Perch	17.2
			Bully (unidentified)	1
1995	Private individuals	Fyke nets	Giant kōkopu	7
December			Shortfin eel	2
			Longfin eel	4
			Perch	1
			Brown trout	1
2002	Department of Conservation	Fyke nets	Shortfin eel	2
June			Perch	11
			Common bully	–
			Brown trout	13
2013	Department of Conservation	Fyke nets	Giant kōkopu	1.8
June			Shortfin eel	1
			Perch	1
2020 March	Department of Conservation	Fyke nets	Eels	5.1
		Multi-panel gill nets	Galaxiids	3.3
			Perch	22.4
			Common bully	1
			Brown trout	5.3

A previous study of shallow coastal lakes collected fish catch data from 46 lakes across Aotearoa New Zealand using a standardised fishing effort (Drake et al. 2010). This included catch records for Lake Tuakitoto in March 2006, which employed a combination of fyke nets and gee-minnow traps at three sites spread across the lake (Drake et al. 2010). In March 2023, this survey methodology was repeated in Lake

Tuakitoto at the same sites to gain a better understanding of the present status of fish communities in the lake. The catches in 2023 were also compared to the earlier survey and to a range of South Island lakes included in the original lowland lake study.

Fish caught during the 2023 survey included shortfin eels, longfin eels, īnanga, common bully and perch (Figure 22). Results of the 2023 fish catches suggested shortfin and longfin eels were abundant, but other species such as common bully, īnanga and European perch were in very low abundance. Comparison of the catch data with previous assessments conducted in Lake Tuakitoto in 2006 and in 22 South Island lakes suggest that both eel species are currently very abundant in the lake, but other taxa that are generally more abundant, such as common bullies, are very low (only two individual fish caught in 2023). This pattern of very low abundances of both common bully and īnanga was also present in the previous 2006 survey. There were relatively high abundances of perch caught in Lake Tuakitoto, particularly in 2006 when European perch were caught at the highest rate of all 22 South Island lakes.

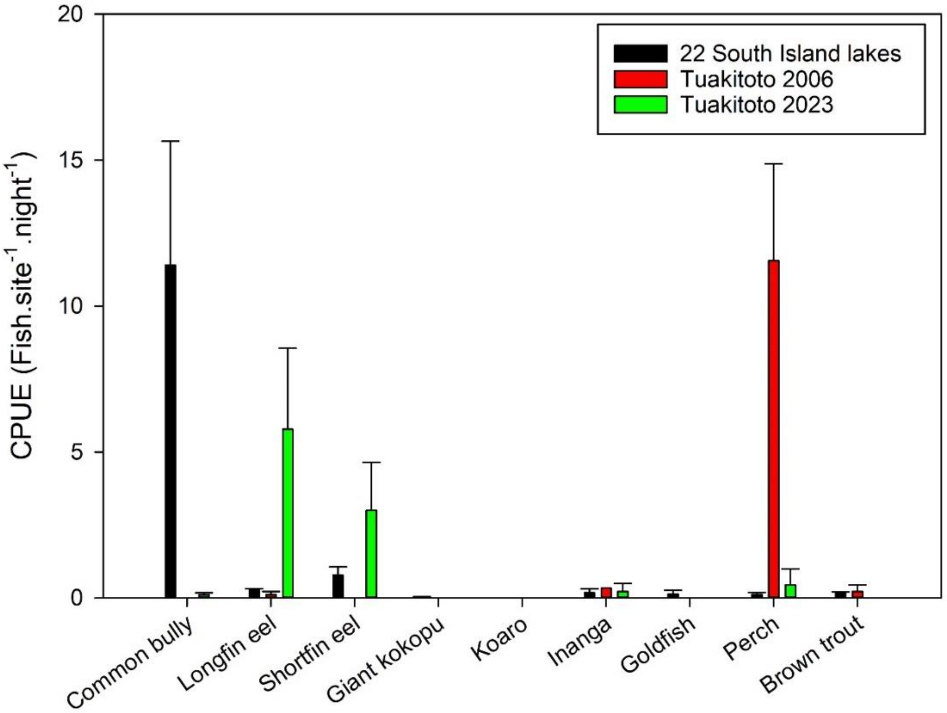

Figure 22. Mean (± standard error) catch per unit effort of fish species in Lake Tuakitoto during fish surveys from 2006 and March 2023. Also shown are mean catch data from 22 South Island shallow coastal lakes fished with the same fishing effort. Data source: Drake et al. (2010).

Native fish surveys data from 22 lowland coastal lakes in Aotearoa New Zealand between 2006 and 2009 indicated a strong inverse relationship between catch rates for perch and for common bullies (Drake et al. 2010). For Lake Tuakitoto, no common bullies were caught, and the lake had the highest perch catch rates (2.96 fish.trap^{-1}.hr^{-1}) of all lakes in the dataset (Figure 23). Catch rates of European perch in 2023 were significantly lower than in 2006; although the reasons for this decrease are not clear, anglers have anecdotally reported that fish catches have been poor in the lake over the past 3 years (Ian Hadland, Otago Fish & Game, pers. comm., 28 May 2023). The combination of very low abundances of fish observed in catch records and by angers indicates a wider recent decline in fish habitat in the lake. The extent of benthic algal mats that can grow to the lake surface during low water periods may be driving major changes in habitat quality and affecting daily variability in DO and pH. The required high-frequency monitoring data to quantify physico-chemical dynamics within the lake was not available; therefore we suggest this is a high priority for further investigation.

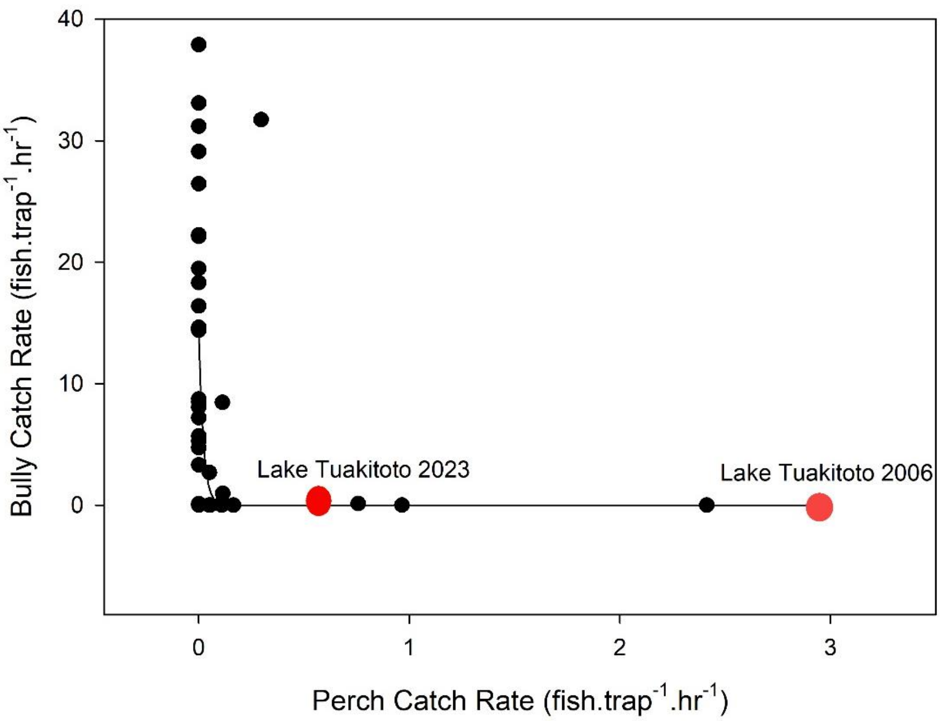

Figure 23. Catch rates of common bullies and European perch in 38 shallow coastal lakes in Aotearoa New Zealand. Data source: Drake et al. (2010).

Previous investigations have identified fish passage barriers to Lake Tuakitoto (Mitchell 1977). The outlet canal from Lake Tuakitoto has two barriers that can periodically block fish passage: a sill downstream from the lake outlet that controls lake levels, and the Kaitangata floodgates, which protect the surrounding reclaimed swampland when the Clutha River floods. The sill affects the passage of fish at low lake levels when outflow water levels recede below the level of the sill. The floodgates can block fish passage when the gates are completely closed to prevent Clutha River floodwaters from entering the lake; however, it is expected this would be intermittent because one of the gates is normally left open to facilitate passage (ORC workshop, pers. comm., 26 May 2023). It is currently unclear if other aspects associated with these barriers, such as outflow velocities or the duration of low water levels, are impeding fish passage over the sill for significant time periods, and it was beyond the scope of this report to determine such aspects. Our review of fish catch records indicate that migratory species with limited or poor climbing ability, such as īnanga and common bullies, are in very low abundances in Lake Tuakitoto, which does indicate a fish passage issue. A further assessment and monitoring of fish passage around these two barriers is recommended to better understand how, and if, these barriers are continuing to significantly impede fish migration.

Figure 24. Aerial picture of Lake Tuakitoto (25 January 2023) during waterbird population monitoring showing the areal extent of surface growing benthic algal mats covering the lake. Source: Ian Hadland, Otago Fish & Game.

2.7.3. Waterbirds

Populations of waterbirds are cited as an important value for Lake Tuakitoto, which has highly abundant waterfowl populations that provide recreational hunting opportunities on the lake. The lake also provides habitat for other rare waterbirds such as bittern, marsh crake / kotoreke and spotless crake / pūweto. Waterbirds can also have important effects on ecological dynamics in shallow lakes because some species are grazers and can recycle plant material and input nutrients back into the lake. In particular, black swans can have an effect on the standing stocks of aquatic plants by grazing (Mitchell et al. 1988). For very shallow lakes like Lake Tuakitoto, swans are able to graze over most of the lake surface and thus can have a strong effect on aquatic plant production when swan densities are high.

Otago Fish & Game have monitored black swan populations (as well as other game birds) in Lake Tuakitoto since the 1970s, with a regular January flight survey of bird numbers undertaken annually. Median abundances of black swans over the past 10 years of monitoring (2014–2023) were 130 birds (min. 15 birds, max. 450 birds; Figure 25). Interestingly, bird count data suggest black swan populations have been declining in the lake, with an annual decline of approximately 0.5% over the 50-year record. This could be related to decreases in the availability of palatable aquatic plants such as macrophytes in the Lake Tuakitoto, which were more abundant in the lake historically (Drake et al. 2010). Swan densities are on average 0.98 birds/ha, with maximum densities (in the past 10 years) of around three birds/ha. These densities are not considered high for South Island coastal lakes, but even at these moderate densities, swans can have a significant effect on aquatic plant production. Because aquatic macrophytes in Lake Tuakitoto are presently so sparse, swans could present some barrier to macrophytes re-establishing over larger areas; however, they are unlikely to be the cause of the decline. We suggest that it is more likely the dominance of macroalgal communities across the lake has resulted from increasing nutrient supply to the lake, rather than from swan grazing removing macrophytes.

It is unclear if macroalgae growing in Lake Tuakitoto provides a significant food source for black swans. However, given its widespread coverage of the lakebed, it seems to be the most readily available food source. The cyanobacteria species *Scytonema*, which is a highly abundant macroalgae in the lake, can produce compounds that may be toxic to swans. Although we do not regard controlling swans as a necessary step to improve the lake health of Lake Tuakitoto, reducing swan numbers could improve the success of macrophyte recolonisation. Establishing such a programme would require discussion amongst wildlife management authorities (Department of Conservation and Fish & Game New Zealand) and ORC.

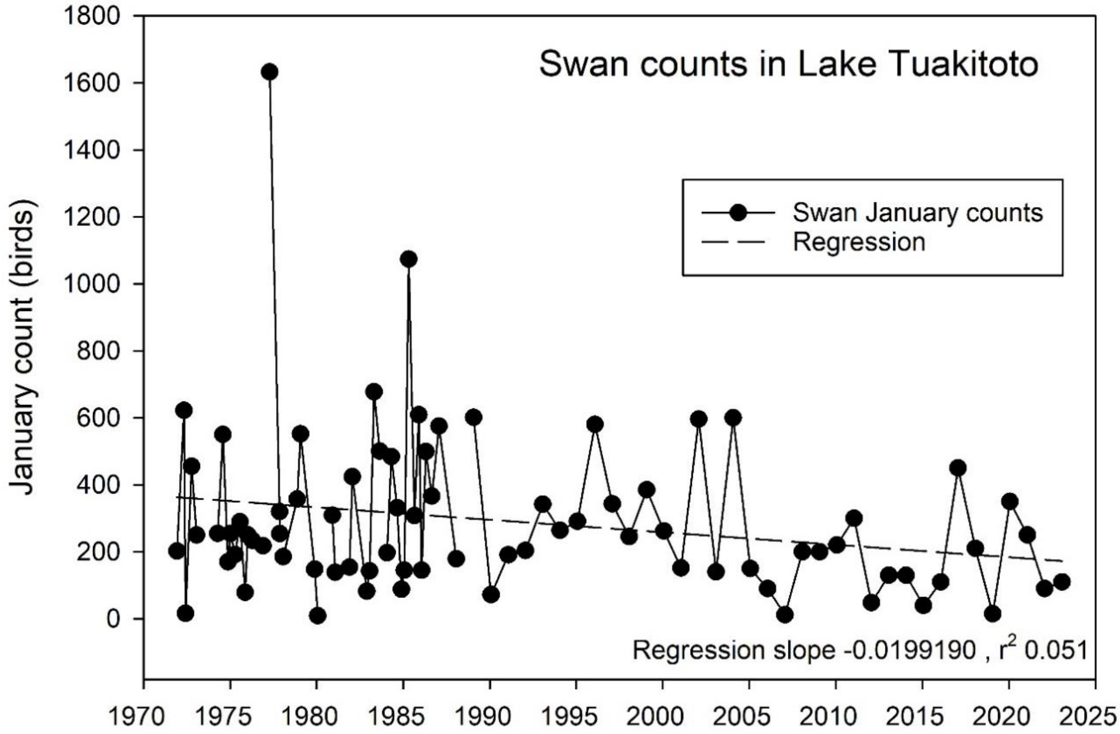

Figure 25. Annual counts of black swans in Lake Tuakitoto between 1973 and 2023. Also shown is a linear regression of counts over time suggesting approximately a 2% decline in numbers across the 50-year record.

2.8. Summary of key water quality and ecology findings

The following points summarise the key findings of historical water quality analyses for Lake Tuakitoto:

- Based on lake core data, sediment accumulation in Lake Tuakitoto has been low, with only 10–13 cm of deposited sediment over the past 180 years of European settlement.
- Water level fluctuations are high (> 1.8 m annually) relative to the lake's shallow mean depth of 0.7 m, but this is consistent with other shallow lakes with large catchments, and biota have likely adapted to such fluctuations. There was a trend indicating higher summer water levels in the past decade (2014–23) compared with previous records.
- Lake nutrient concentrations have been high for several decades and have increased moderately in the last decade. TN and TP concentrations are currently well in excess of the national bottom-line values set in the NPS-FM for the protection of ecosystem health. This is a result of high nutrient loads from inflowing tributaries (Lovells and Stony Creeks) that drain intensive farmland.
- More recently (since 2018) phytoplankton biomass has trended upward in the lake and now breaches the NPS-FM annual maximum guideline for chl-*a*. It is likely

that current growth rates of phytoplankton exceed the capacity of kākahi populations to filter algae from the water column. The relatively higher increase in chl-*a* has resulted both from increasing nutrient supply (bottom-up process) and further declines in kākahi populations (top-down process) in the lake.

- The relatively high P content of Lake Tuakitoto sediments, combined with the very high proportion of the redox-sensitive P fraction, the high potential release rates, and the downcore pattern in TP content, all indicate that significant internal P loading is likely occurring in the lake. This may be driven by high pH (> 9.2), which is observed in the water quality record and possibly the anoxic conditions near the lakebed.
- Increasing concentrations of DRP in Lake Tuakitoto, particularly during summer, is a worrying trend because it is a self-reinforcing (positive-feedback) process and can cause cyanobacterial blooms that could be harmful to lake ecological values and kākahi populations.
- Aquatic plant surveys in 2023 indicated limited or no submerged macrophytes species in the lake, and the benthic community is now dominated by thick growths of macroalgae, which is poor habitat for benthic biota. Based on a limited survey, macroalgae covered 69% of the lakebed and was dominated by the filamentous species *Spirogyra* sp. (Chlorophyta) and *Scytonema* sp. (Cyanobacteria), which are both known from other shallow Aotearoa New Zealand lakes and can proliferate to nuisance biomass (Vadeboncoeur et al. 2021). Extensive coverage by macroalgae is having a negative impact on benthic biota such as kākahi and is likely to be driving variations in pH and DO, which impact sensitive species such as fish and promote internal nutrient loading. Proliferation of macroalgae is likely to be primarily related to increasing nutrient loads to the lake, but grazing swans can also have a negative influence on aquatic macrophytes and can hinder macrophyte re-establishment.
- Surveys of kākahi indicate that abundances are continuing to decline in comparison to historical levels, with kākahi biomass in 2023 being 57% lower than in 2013, which had already declined from levels observed in the 1990s. The progressive trend for increasing kākahi shell size from 1991 to 2023 points to an ageing kākahi population that is not experiencing significant recruitment (breeding success) and will ultimately disappear unless conditions causing poor recruitment are not ameliorated. Poor recruitment is likely to be related to both poor lake water quality, a lack native host fish and macroalgae overgrowing suitable mussel habitat. Because of their very important role in filtering phytoplankton, kākahi are a critical to maintaining the water quality in Lake Tuakitoto, with the filtration rates declining to 43% of that in 2013 and now taking approximately 10 days to filter the lake water column.
- Fish surveys in 2023 found low abundances and diversity of native fish in Lake Tuakitoto and suggested that severe water quality conditions are preventing some species (common bullies, īnanga) from inhabiting the lake. Shortfin and longfin eels were the only species to occur at reasonable densities in Lake Tuakitoto,

indicating poor fish community health by comparison to 22 other lowland coastal lakes. The low non-eel fish abundance results in a scarcity of hosts for the parasitic glochidia larvae of kākahi. Low abundances of native fish are also likely to be impacted by high European perch abundances in Lake Tuakitoto, with historical fish surveys indicating perch abundances were high compared to other lowland coastal lakes in Aotearoa New Zealand.

2.9. Proposed management options for Lake Tuakitoto

Previous community consultation around the values and aspirations for Lake Tuakitoto was facilitated by ORC in 2018. From information gathered during these consultation meetings, several lake restoration priorities were identified and written as objectives within the Tuakitoto management plan.

Objectives related to water quality and ecological health of the lake included:

- improving the water quality and meeting the national freshwater and ORC land and water plan standards in Lake Tuakitoto and the upstream catchment
- improving biodiversity within the catchment
- supporting a healthy ecosystem that sustains and enables mahinga kai
- improving water quality to support recreational fishing
- preserving and protecting the wetlands, rivers and streams, their margins and the saline environment to prevent further loss or degradation
- encouraging and supporting soil conservation to minimise sedimentation
- maintaining and enhancing public access around Lake Tuakitoto
- ensuring that the existing kākahi beds present in the lake are enhanced and managed effectively
- promoting Lake Tuakitoto and encourage people to visit and use the lake
- ensuring the management of the lake is influenced by good-quality science
- managing flood risk and land drainage for adjacent land.

Workshops with the community and stakeholders identified a range of rehabilitation priorities:

- riparian enhancement of catchment for water quality improvement
- excluding stock from lake margins
- creating wetlands and sediment traps (capturing silt)
- investigating fish passage options (Kaitangata locks fish ladder), spat ropes for culverts
- lake dredging – sediment removal and creating deeper refuge areas

- maintaining minimum summer water level – kākahi protection
- improving native fish habitat throughout the catchment – giant kōkopu
- researching freshwater mussel breeding
- enhancing birdlife and waterfowl
- creating a pest management strategy for Canada geese, willow, glyceria, predators.

Based on our review of historical data, and investigations and understandings of ecological processes in Lake Tuakitoto, we propose four goals for managing stressors that would improve water quality and the ecological health of Lake Tuakitoto (Table 9). Some of these goals require multiple actions. While we have done our best to make recommendations using the best data available, in some cases further investigations need to be undertaken to support specific management actions. However, based on the reasonably good historical data availability and an understanding of appropriate restoration actions for shallow lakes (Abell et al. 2020), there was confidence in the recommended actions and their potential to achieve the goals.

2.9.1. Management Goal 1: reducing stream nutrient loads in inflows

Nutrient loads to Lake Tuakitoto are high, and it is unlikely that the ecosystem health national bottom-line guidelines under the NPS-FM for TN (median 800 $mg.m^{-3}$) and TP (median 50 $mg.m^{-3}$) will be met unless significant stream remediation is taken in the inflow catchments to reduce nutrient loads. These understandings are based on 2012–13 monitoring data, and it is probable that loads are higher in 2023. At a workshop held at ORC (26 May 2023), it was signalled that further water quality monitoring in the inflow catchments by ORC is planned. Concentrations of TN and TP in lower portions of Lovells and Stony Creeks were high and are likely to be a major source of nutrient loads to Lake Tuakitoto. We see two options for reducing loads to the lake from the inflows:

- Divert greater proportions of combined flows from Lovells and Frasers Creek (yellow channel) into the wetland area to provide increased attenuation of nutrients prior to inflowing to Lake Tuakitoto. It is also recommended that the flows of Stony Creek be diverted through to the wetland area instead of their current flow directly to the lake.
- Increase riparian vegetation and enhance vegetation along upper portions of Lovells and Stony Creeks to reduce nutrient losses to the streams, potentially in conjunction with landowners developing farm plans.

Load-reduction targets for stream inflows were harder to ascertain because there was limited inflow data available to guide the prediction of the required load reductions. We estimated loads using two catchment models that predicted mean annual nutrient concentrations and flows for all stream reaches in Aotearoa New Zealand (Table 10).

The national CLUES model (V10.6; Woods et al. 2006) predicted relatively high loads of TN and TP to Lake Tuakitoto and suggested that current loads would need to be reduced by between 51% (for TP) and 69% (for TN) to meet NPS-FM bottom-line guidelines. These predictions are likely high given they overpredict current in-lake nutrient concentrations. The NZ River Maps model operated by NIWA predicted inflow concentrations that were more in-line with 2013 ORC monitoring 80th percentiles. The model also indicated around a 53% reduction in TN load is required to meet NPS-FM bottom-line lake standards, but that TP loads already meet these standards. The relatively low TP load predicted by NZ River Maps may be accurate, indicating that most of the TP in Lake Tuakitoto in excess of the 50 $mg.m^{-3}$ guideline results from internal loading processes. Load predictions using 2013 stream monitoring data collected by ORC for the lower Lovells, Frasers and Stony Creeks indicated potentially even lower load reductions may be required; however, this would not account for the high TN levels in Lake Tuakitoto, which are less likely to result from in-lake nutrient loading processes. The differences between the load models suggests a moderate level of uncertainty for predicting inflow loads, and therefore nutrient and flow monitoring of inflow streams in their lower reaches (i.e. close to the lake) is recommended to better understand load-reduction targets.

Table 9. Predicted inflow nutrient concentrations by the CLUES (10.6) and NIWA NZ River Maps models, as well as the predicted in-lake values based on a mass balance model previously derived in a study of shallow coastal lakes for the South Island (Kelly et al. 2013). Also included in the table are reductions in loads necessary to meet NPS-FM lake bottom-line standards for TN (median 800 $mg.m^{-3}$) and TP (median 50 $mg.m^{-3}$).

	Inflow concentrations		**Predicted in-lake concentrations**		**Estimated load reductions required to meet NPS-FM C band**	
	TN ($mg.m^{-3}$)	TP ($mg.m^{-3}$)	TN ($mg.m^{-3}$)	TP ($mg.m^{-3}$)	TN reduction to meet NPS-FM 800 $mg.m^{-3}$	TP reduction to meet NPS-FM 50 $mg.m^{-3}$
CLUES model inflow mean	2,265	156.8	2,556	102.4	69%	51%
NIWA NZ River Maps inflow mean	1,508	57.3	1,702	49.1	53%	-2%
ORC Inflow monitoring 80th perc. 2012–13 (downstream sites)	868.1	50.8	868	50.8	8%	2%
ORC lake outflow median 2018–23			1,171	122.4	32%	59%

Flow diversions

We see the current bypass of stream flows from Lovells and Frasers Creek via the drain as an opportunity to utilise ecosystem services the catchment already has available by diverting flows into the wetland (Figure 8). This may require altering drainage of the wetland so as not to encroach on adjacent pastoral land or potential purchases of pastoral land adjacent to wetland areas that could be affected. High water levels that exceed the upper limit of 101.4 m (above datum) more often occur in winter or spring. We anticipate that spring and summer will be the most important periods for diverting flows into the wetland, as these seasons are important for macroalgal growth. Limiting the extent of macroalgal development by reducing nutrients of inflows may be the only way to control macroalgae. Upgrading of the existing diversion gate at the head of the wetland was an important step for this process, as it allowed more automated control of water diverted into the wetland.

Flows from Stony Creek presently bypass the wetland area and are diverted directly to the lake via the diversion race. We would recommend this drainage is changed so that flows can be diverted, along with Lovells Creek and Frasers Stream flows, through the wetland area. Lower Stony Creek had high nutrient loads, suggesting diversion to the wetland could potentially reduce loads to the lake. An alternative to this option would be to re-align the existing diversion race to bypass the lake and input drainage of the race to the lake outlet canal. This would have the added value of being able to bypass high flows to the lake that would not normally be diverted to the wetland during higher flow periods because they would inundate neighbouring farmland.

When this option was initially considered, there was no nutrient data collected from within the wetland to quantify potential nutrient attenuation rates by the 365 ha wetland. We recommend monitoring water quality (specifically dissolved and total N and P and DO) at sites within the wetland (e.g. entrance and exit of flow) to better understand how effectively the wetland attenuates nutrients in inflows. The use of continuous monitoring sensors such as nitrate sensors could be highly effective for conducting such monitoring, given the difficult access to the site.

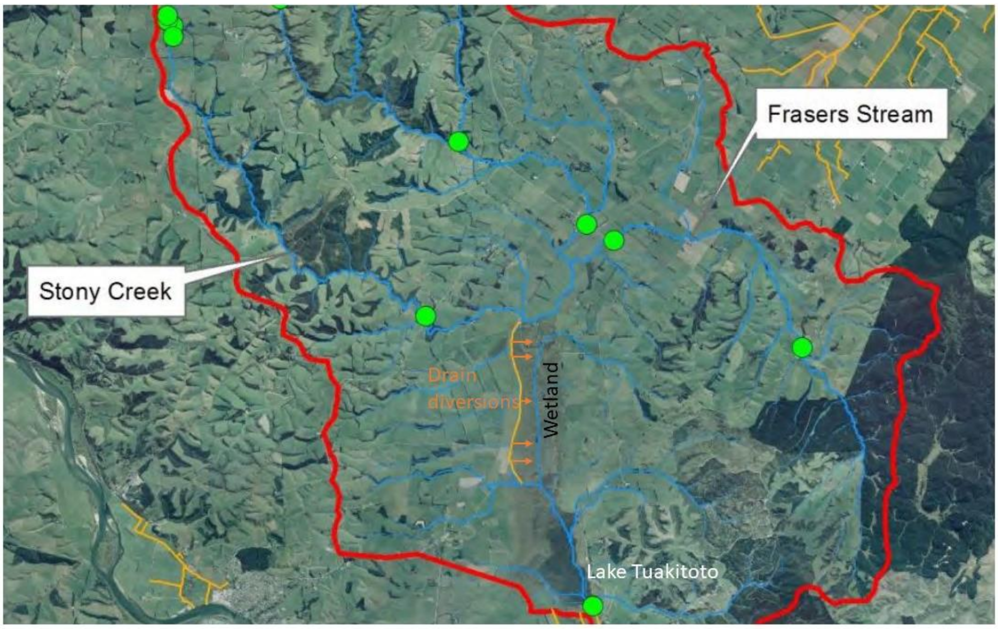

Figure 26. Lake Tuakitoto catchment showing the existing diversion of Lovells Creek into a drainage channel (yellow line) and potential points of flow diversions (orange arrows) into the wetland area located north of Lake Tuakitoto.

Riparian management

Replanting of riparian zones along the lake tributaries could limit the extent of overland flow of P and sediment inputs to the lake. Stream water quality monitoring data indicated that the upper portion of Lovells Creek (particularly West Basin) and Stony Creek (upstream of Station Road) had some of the highest nutrient yields in the catchment. Actions could include:

- fencing of riparian margins of streams and the lake edge to discourage stock from directly accessing waterways
- enhancing stream and lakeside vegetation (e.g. toetoe or harakeke) along steeper banks to trap sediment prior to it entering the waterways
- replanting low-growing riparian vegetation along lake margins to encourage nutrient uptake of overland flow and shallow groundwater
- replanting native trees (in place of willows and pines) where shading or cover is desired, using species that are not seasonal in their growth and regeneration patterns (e.g. mānuka or kānuka) and do not generate large amounts of autumn leaf litter.

It is not expected that riparian enhancement alone would achieve the reductions in nutrient loads necessary to meet water quality guidelines for Lake Tuakitoto. From experience, riparian planting is likely to make only small improvements in inflow nutrient concentrations, most significantly for sediment and associated sediment-bound P. However, this is an important intervention to reduce erosion and the impact of heavy stock accessing the margins of tributaries and the lake.

Land-use management by farm plans

It is recommended that farm plans are developed or modified in order to identify ways to minimise nutrient losses to the lake. The plans should also consider inflow tributaries for farms bordering the lake and its tributaries. Freshwater farm plans will provide farmers with the flexibility to find the right solution for their farm and catchment, as well as encourage actions to reduce a farm's impact on fresh water, including calculations of nutrient losses to waterways through tools such as Overseer (Overseer Limited, https://www.overseer.org.nz). Freshwater farm plans will bring together many existing requirements and allow for better recognition of on-farm efforts to improve fresh water. Such plans can demonstrate how farms are meeting other regulatory requirements, including those below:

- National Environmental Standards for Freshwater 2020
- nitrogen-cap regulations
- stock exclusion regulations
- intensive winter grazing regulations
- regional plans and consent requirements.

The extent of nutrient reductions to the lake required to meet plan standards is likely to be moderate (See Table 9). This may preclude consideration of converting some land parcels to other uses with lower nutrient losses, such as forestry, which already occurs in a small proportion of the catchment area.

2.9.2. Management Goal 2: controlling internal phosphorus recycling

Although reducing external nutrient loads to the lake (Management Goal 1) may reduce in-lake nutrients, legacy nutrients currently stored in lake sediments could potentially provide a source of P to the lake for decades, as observed in other catchment restoration projects (Søndergaard et al. 2013). Since about 2018, water quality conditions in Lake Tuakitoto have shown sharp rises in dissolved P consistent with internal recycling of P into the water column as DRP during summer. The timing of these events is associated with periods of high phytoplankton and benthic algal biomass in the lake, and generally occurs over peak productivity between December and March. However, mats die-off, and decomposition later in the summer / autumn period could also yield considerable inputs of dissolved nutrients to the lake. Limiting the extent to which DRP is recycled into the water column would further reduce summer algal blooms and lessen the risk of cyanobacterial blooms that could impact kākahi.

Sediment capping

Sediment-capping agents have been used to reduce internal nutrient loads by strongly binding sedimentary P pools and reducing nutrient solubilisation during high pH or low DO events (Gibbs and Hickey 2018). Sediment capping involves creating a boundary layer of capping agent between the bed sediment and the water column. The use of *active* capping agents requires the formation of only a thin (e.g. 5 mm) layer of capping agent at the sediment surface, which inactivates P via metal binding (e.g. with iron, aluminium, lanthanum). The use of such agents has achieved good results internationally, especially in shallow lake systems. Active capping agents consist of more or less granular pellets, which are applied evenly over the lakebed to provide a 1–2 mm thick layer (Gibbs and Hickey 2018).

Passive capping involves the formation of a thick (1–8 cm), inert boundary layer on the surface of the bed sediment that reduces the flux of contaminants from sediment to water column to the rate of molecular diffusion (Hamilton et al. 2018). Numerous capping agents have been used including sand, gravel and clay and more recently, local soils modified with various additives such as chitosan (Pan et al. 2012). The disadvantage of these types of capping agents is the potential effect on benthic fauna from depositing so much material on the lakebed (Hamilton et al. 2018). In addition, multiple applications of passive and active capping agents may be required, as continued sedimentation buries capping layers and, in the case of active agents, binding capacity becomes saturated.

However, the use of geochemical capping agents in Lake Tuakitoto raises several concerns regarding how effective they might be for controlling internal nutrient loading. This includes:

1. Shallow lake depth throughout could mean that the cap is regularly disturbed by wind-wave resuspension and thus have inconsistent effects.
2. Thick benthic algae could reduce the effectiveness of capping agents in forming a cohesive capping surface over the sediment.
3. Capping agents could potentially harm kākahi both from direct contact and ingestion.

It should be noted that all the P-inactivation techniques described relate to controlling P release to the water column with a view to managing planktonic algae. No research literature was found on the use of P-inactivation agents for the control of benthic algal mats. In addition, the effectiveness of various P-inactivation agents can be influenced by other water chemistry parameters such as alkalinity (Gibbs and Hickey 2018). Further field investigation possibly using mesocosm trials could aid understanding the effectiveness of capping agents in Lake Tuakitoto. Further information would also be needed on sediment properties (grain size, P content, sediment oxygen-demand) as well as on DO and pH dynamics in the lake during periods of high risk of internal loading (January to March). This data would provide a better understanding of processes that drive such internal recycling and control interventions. The effects of capping agents on kākahi populations in the lake are uncertain and further investigation on this may also be required (Tempero 2015).

Controlling macroalgae

Our analyses of water quality data suggests that macroalgae are likely to be the main factor controlling internal recycling of nutrients from sediments into the water column. This is driven by creating physico-chemical conditions that drive solubilisation of P (Wood et al. 2015), but it also occurs when mats are dislodged (or float) from the bed and accumulate at the water surface where they decompose. We suggest that reducing the extent of macroalgae cover and biomass would reduce the magnitude of internal loads.

There is limited research on the control of benthic mats in temperate lake systems in Aotearoa New Zealand (Wood et al. 2012). Work on urban-lake or farm dams has shown benthic algae can be controlled using herbicides (Sink 2014). However, at a larger scale, the use of herbicides is probably impractical. Nutrient load reduction is likely the most practical method for controlling macroalgal proliferations. However, mechanical methods to reduce macroalgal biomass could be employed on a short-term basis. This could consist of suction dredging areas of the bed (Figure 27), or mechanically raking the material off the bed. Collection (or suction dredging) of floating macroalgal material would be beneficial because as this material decomposes it directly contributes nutrients to the water. Mechanical harvesting or suction dredging over larger areas of the lake would be a large undertaking and potentially require a harvester specially designed for the lake. A separate dewatering and composting facility would also need to be constructed.

A better understanding of macroalgae seasonality, biomass / cover, and accrual rates are needed to better evaluate this option and determine how rapidly mats would regrow in the lake should harvesting occur. Our expectation is that macroalgae mats represent several years growth, but this is based only on observations from other deep lakes (Kelly et al. 2017).

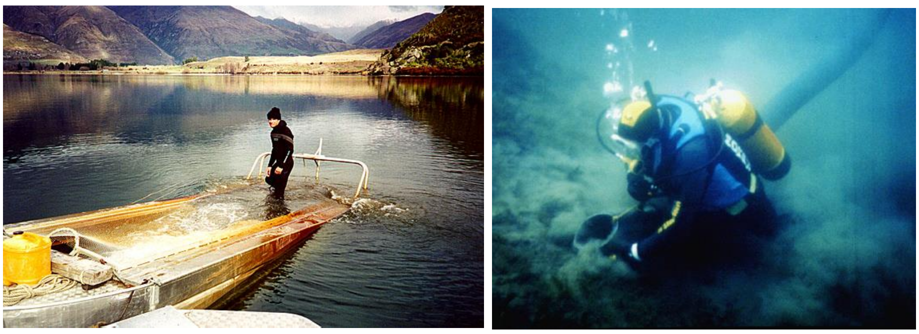

Figure 27. Floating suction dredge barge operating on Lake Wanaka (left) – used for removing invasive plants (*Lagarosiphon major*); divers operating the dredge at the lakebed (right). Photos: David Kelly, Cawthron.

2.9.3. Management Goal 3: improving kākahi recruitment

Kākahi populations in Lake Tuakitoto enhance water quality by grazing on phytoplankton biomass. Based on monitoring data collected between 1991 and 2023, kākahi abundances have declined by over 85% in Lake Tuakitoto. Water quality data indicate that phytoplankton biomass has markedly increased over this period, suggesting a reduction of kākahi grazing. Declines in kākahi appears to be mostly related to poor recruitment (i.e. reproductive success), with its current population comprised of large-shelled, mature individuals. A previous hypothesis that kākahi declines were related to unusually low summer water levels between 2002–2012 is unlikely to explain the reduction in kākahi (Ozanne 2014). We suggest that the combination of degraded water quality with high variation in DO and pH, the smothering of benthic habitat by macroalgae and the low abundances of hosts for glochidia are the most likely causes of continued declines in kākahi. It is not certain which of these factors is most affecting kākahi and a more detailed population study is required to answer such questions.

While Management Goals 1 and 2 address the water quality and macroalgae issues that are likely to negatively affect kākahi, we recommend that actions are taken to improve bully and galaxiid glochidia host abundances. High abundances of European perch are likely to be substantially reducing native fish populations through predation and competition (McDowall 1987). Perch appears to have a strong effect on native fish populations (common bullies, īnanga, kōaro) in Lake Tuakitoto, with no bullies or galaxiids found in a survey conducted in 2006, and very low abundances of these found in 2023. We suggest that controlling perch populations in the lake could improve kākahi recruitment by improving native fish populations. This could consist of a mix of measures including gill netting of adult perch and juveniles. Perch spawn on hard substrates on lake margins, and the use of removable hard structures to attract perch spawning has been experimentally tested as a way of reducing perch spawning success. We suggest that a range of measures are undertaken as part of an enhancement plan for Lake Tuakitoto.

Monitoring of both native fish communities and kākahi in Lake Tuakitoto would clarify and confirm the need for perch control. However, given that perch are a sport fish species managed by Fish & Game New Zealand, it could be timely to discuss such options as part of a wider fish management plan. The Department of Conservation may also provide input on ways to enhance native fish species and control exotic perch populations.

2.9.4. Management Goal 4. Fish passage

Previous investigations have identified fish passage barriers to Lake Tuakitoto (Mitchell 1977). The scarcity of native fish hosts (mainly common bully and īnanga) for kākahi glochidia could be an important factor contributing to the decline of kākahi in Lake Tuakitoto. However, the extent to which operations of the existing control structures affect fish passage in and out of Lake Tuakitoto is uncertain. For instance, the outlet sill that regulates lake levels can hinder fish passage in and out of the lake when lake levels are low and may restrict the passage of poor climbing species such as īnanga and common bullies (Kelly and McDowall 2004). The Kaitangata locks, which protects the surrounding reclaimed swampland when the Clutha floods, can block fish passage intermittently when the gates are completely closed, but a gate is normally left partially open to facilitate passage. However, it is unclear if other aspects associated with the lock, such as outflow velocities around the gates, are impeding fish passage for poorer swimming species over longer periods.

We recommend a detailed review of the fish passage structures to explore potential mitigations if the barriers are impeding fish migration. This would likely include installation of a passage structure around the sill for low water level passage. The use of hydroacoustic cameras that can effectively monitor fish moving around, or over, structures could be beneficial (e.g. Kelly et al. 2019). This review should also investigate key migration periods when special management of fish passage infrastructure would be beneficial.

3. TOMAHAWK LAGOON

3.1. Background on Tomahawk Lagoon

The Tomahawk Lagoons comprise a coastal aquatic system encompassing a small catchment on the southern side of the base of the Otago Peninsula (Figure 28). Two standing waterbodies are referred to as the Tomahawk Lagoons: (1) the eastern or Upper Tomahawk Lagoon, which is a freshwater lake, and (2) the western or Lower Tomahawk Lagoon, which is a brackish, impounded, intermittently closed estuary that also resembles a shallow lake / lagoon (Table 11). Upper Tomahawk Lagoon is also sometimes referred to as Tomahawk Lagoon #2. The two lake / lagoon systems are connected by a short channel, which discharges to the lower lagoon via a weir and restricts drainage of the upper lagoon. Background morphological information on the lagoons is presented in Table 11.

Lagoon Creek drains the eastern part of the catchment into the upper lagoon, while an unnamed creek drains the western catchment into the northern end of the lower lagoon. Other creeks and drainage features in the catchment are ephemeral.
The outlet of the lower lagoon discharges to the sea across Tomahawk Beach. Due to variations in outflow discharge and sand accretion on the beach, the outflow is often blocked by sand, causing the level of the lower lagoon to rise. The outlet is sometimes artificially opened to prevent the flooding of land and infrastructure. This allows for migration of fish and invertebrates into and out of the lagoon system. When the lagoon is open to the sea, saline intrusions can occur whereby high tides and storm surges flow into the lagoon, which cause a rise in the lagoon's salinity and often result in the deposition of kelp and other seaweeds.

The lagoons are managed by the Department of Conservation as a Wildlife Management Reserve. Fringing wetland areas in both lagoons are protected under a QEII covenant. Under the Dunedin City Council's District Plan, the area is described as a lowland lake with reed swamp, being of local and regional significance. ORC recognises the lagoon area as a Schedule 9 Regionally Significant Wetland due to:

- it being a habitat for nationally or internationally rare or threatened plant species or communities (e.g. *Isolepis basilaris*), and naturally uncommon plant species (e.g. *Althenia bilocularis*, *Lilaeopsis novae-zelandiae*, *Myriophyllum triphyllum*, *Limosella lineata*)
- it exhibiting a high degree of wetland naturalness
- it being unique in Otago in terms of its ecological or physical character
- it being a regionally significant wetland habitat for waterfowl, waders and native fish (https://www.orc.govt.nz/managing-our-environment/water/wetlands-and-estuaries/dunedin-district/tomahawk-lagoon).

Otago Fish & Game manages Upper Tomahawk Lagoon as a brown trout (*Salmo trutta*), rainbow trout (*Oncorhyncus mykiss*), and European perch (*Perca fluviatilis*) fishery by stocking the lake with mature trout. Perch are self-sustaining in the lake. As such, the lake is a valued sports fishery in the Dunedin region.

Over the years, the lagoons have been shaped and constrained by various activities including:

- the infilling of wet shoreline areas (especially on the south side of the upper lagoon)
- the dredging of part of the upper lagoon to increase water depth
- the installation of the weir between the two lagoons to maintain a higher water level in the upper lagoon
- the physical constraining of the outlet of the lower lagoon by roading
- the building of houses and infrastructure on the floodplain of the lagoons.

Table 10. Background information on the Upper and Lower Tomahawk Lagoons and their catchments. Data sources: Google Maps™, the Freshwater Environments of New Zealand (FENZ) database and Marc Schallenberg's personal observations.

	Upper Tomahawk Lagoon	**Lower Tomahawk Lagoon**
Northing	-45.90126	-45.90066
Easting	170.55093	170.54279
Altitude (MASL)	approx. 1.0	< 1.0
Type of waterbody	Eutrophic shallow lake (sand dune formation)	Eutrophic intermittently closed estuary (sand dune formation)
Surface area (ha)	10.2	18.7
Maximum depth (m)	approx. 1	< 1.0
Catchment area (ha)	185	243
% catchment area urban	2.9	14.4
% catchment area grassland	74.1	66.1
% catchment area shrubland	3.7	0.9
% catchment area exotic forest	13.5	10.0
% catchment wetland	0	0.7
% catchment area water	5.8	8.0

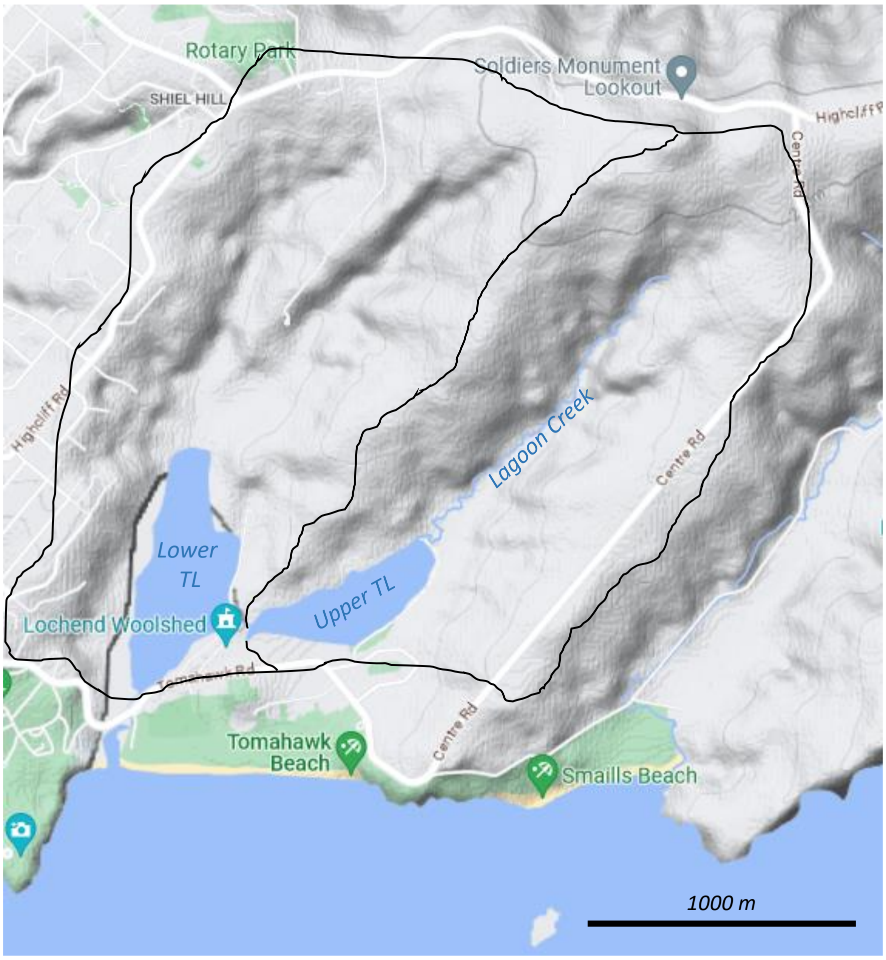

Figure 28. Topographic map of the Tomahawk Lagoon system and its catchment. Upper Tomahawk Lagoon drains into Lower Tomahawk Lagoon.

3.1.1. Management issues

A range of management issues have been identified for the Tomahawk Lagoons (previously listed in Section 1.2). Algal blooms in the Tomahawk Lagoons have been linked to catchment nutrient loads. Land use within the catchment has been altered over time with the removal of native vegetation and an associated increased in soil erosion within the catchment. Algal blooms were reported in Upper Tomahawk Lagoon as early as the late 1960s (Mitchell 1989).

Sediment infilling was recognised as a problem by Otago Fish & Game, which deepened the upper lagoon by dredging with the aim to improve the sports fishery of the lake. In addition, some concern has been expressed regarding the proliferation of *Typha orientalis* (raupō) at the western end of the upper lagoon, which has been associated with sediment infilling upstream of the weir.

Stormwater inflow from urban areas contains contaminants that could negatively impact the ecology of the lagoons. ORC issued a resource consent to the Dunedin City Council relating to stormwater discharge from urban areas to the lagoons. The resource consent requires measurement of the chemical composition of the stormwater effluent to the lagoons as well as the assessment of potential impacts on the lagoons of stormwater discharges.

Locals have described significant historical water level variations in the lagoons. For example, at times in the past, the Ocean Grove Domain Board Hall has been flooded. On the other hand, during dry summers, the water levels of the two lagoons can recede, such that a substantial amount of the southern end of the lower lagoon can become stranded above the normal lake lagoon water level. The presence of such water level variations has drawn attention to the regulation of water levels in the system, which is accomplished by the artificial opening of the lower lagoon's mouth when rising water levels threaten to flood land.

The above issues have prompted ORC to undertake community consultations to help identify values, issues and potential solutions. The outcomes of these consultations were summarised in three unpublished documents (ORC pers. comm.). One of the priority actions identified by the community was to undertake an ecological assessment of the Tomahawk Lagoon system. The remainder of this report provides: (1) a scientific perspective on the ecological condition of the Tomahawk Lagoons and (2) a list of recommended ecological restoration actions.

3.2. Tomahawk Lagoon catchment

The Tomahawk Lagoon system drains a catchment that primarily comprises hilly low intensity agricultural land, which is mainly used for sheep grazing (Figure 29). There is some exotic forestry in the catchment of both lagoons, although it is more prevalent in the catchment of the upper lagoon. Some native bush and scrub is also found in these catchments, but the dominant land cover is grassland. Urban areas cover 14% of the catchment of the lower lagoon and 3% of the catchment of the upper lagoon. Each of the lagoons has one main inflow and a number of ephemeral streams that carry water to the lagoons mainly during substantial rain events. Mean annual rainfall at the Musselburgh weather station near to the lagoons is 787 mm and exhibits no seasonality. However, strong seasonality in evaporation results in catchment run-off being generally higher in winter than it is in summer.

Figure 29. Satellite image of the Tomahawk Lagoon system and its catchment, showing variation in land cover.

Drainage water from urban areas is collected by stormwater systems, which have two main discharge points, one on each of the lagoons. There are numerous smaller stormwater discharges evident along the south shore of the upper lagoon.

The low intensity agriculture in the catchment employs some fertiliser top-dressing (Marc Schallenberg, pers. obs.). In addition, piggery waste is imported and spread into the upper catchment of the upper lagoon. ORC deems this a permitted activity, although discharge regulations state that such waste cannot be distributed within 50 m of a waterbody. Presumably, Lagoon Creek qualifies as a waterbody and, therefore, discharge regulations should apply to this activity.

3.3. Tomahawk Lagoon palaeohistory

When developing a lake restoration plan, it is useful to determine appropriate restoration targets. To understand the range of potential restoration targets and trajectories, it can be helpful to understand the time frame and trajectory of lake degradation that has occurred. This can also reveal information on key drivers of degradation – stressors that may need to be mitigated to improve the condition of the lake. The Lakes380 research programme (led by Geological and Nuclear Sciences and Cawthron) was designed to infer the historical conditions of lakes based on information obtained from palaeoecological analysis of sediment cores. Sediment cores were obtained from both the Upper and Lower Tomahawk Lagoons. The cores were dated using pollen preserved in the sediments as well as the radionuclides ^{210}Pb and ^{14}C. In addition, a variety of analyses were undertaken to help infer and reconstruct historical conditions in the lagoons.

This report discusses inferences based on the palaeolimnological data that reveal historical dynamics of lake productivity, submerged aquatic macrophyte occurrence over time and sediment infilling rates. Information on all three attributes is available for Upper Tomahawk Lagoon, whereas only information on the dynamics of sediment infilling is available for Lower Tomahawk Lagoon. The Lakes380 factsheet produced for the upper lagoon illustrates a pattern that was commonly observed by the Lakes380 programme in lake sediments from regions of Aotearoa New Zealand that had undergone deforestation and conversion of land cover from native vegetation to productive grassland pasture (Figure 30).

The general pattern, also illustrated by the upper lagoon in Figure 30, shows a decline in native trees (e.g. podocarps) and forest plants (e.g. tree ferns) initiated soon after the time of Māori settlement and corresponding with an increase in charcoal and bracken fern spores, which indicate that fires were common in the landscape. Subsequently, pollen from grasses becomes dominant, showing the landscape transitioned to largely agricultural areas following the arrival of European settlers. Algal pigments provide an indication of the algal productivity in the lake. However, interpretation of the algal pigment data must be carried out with caution because pigments likely degrade with age of deposition, and in lakes as shallow as Upper Tomahawk Lagoon, our analysis of algal pigments could not distinguish between the presence / abundance of phytoplankton and benthic microalgae. In Upper Tomahawk Lagoon, the algal pigments show three peaks over time: one at the sediment surface (most recent sediments), one near the beginning of the time of European settlement, and one near the beginning of Māori settlement. It is interesting to speculate why algal pigments increased during these eras, and examination of the historical dynamics of submerged macrophytes would be a useful parallel analysis.

The data in Figure 31 show the historical dynamics of four different macrophyte indicators: (1) the pollen *Potamogeton* sp., (2) the pollen of *Myriophyllum* sp., (3) the seeds of *Ruppia* sp., and (4) the oospores (propagules) of charophytes. These macrophyte indicators show two historical periods of macrophyte abundance in the lake, one period during the time of Māori settlement and one period during the period of European settlement from the 1950s to the early 2000s. The periods of macrophyte abundance occurred when algal abundance was relatively low, suggesting an alternating abundance of macrophytes and algae in the lake, with at least two long cycles of alternating states going back to the early stage of Māori settlement. Periods when macrophyte abundance was low corresponded to periods when algal pigments rose above baseline levels.

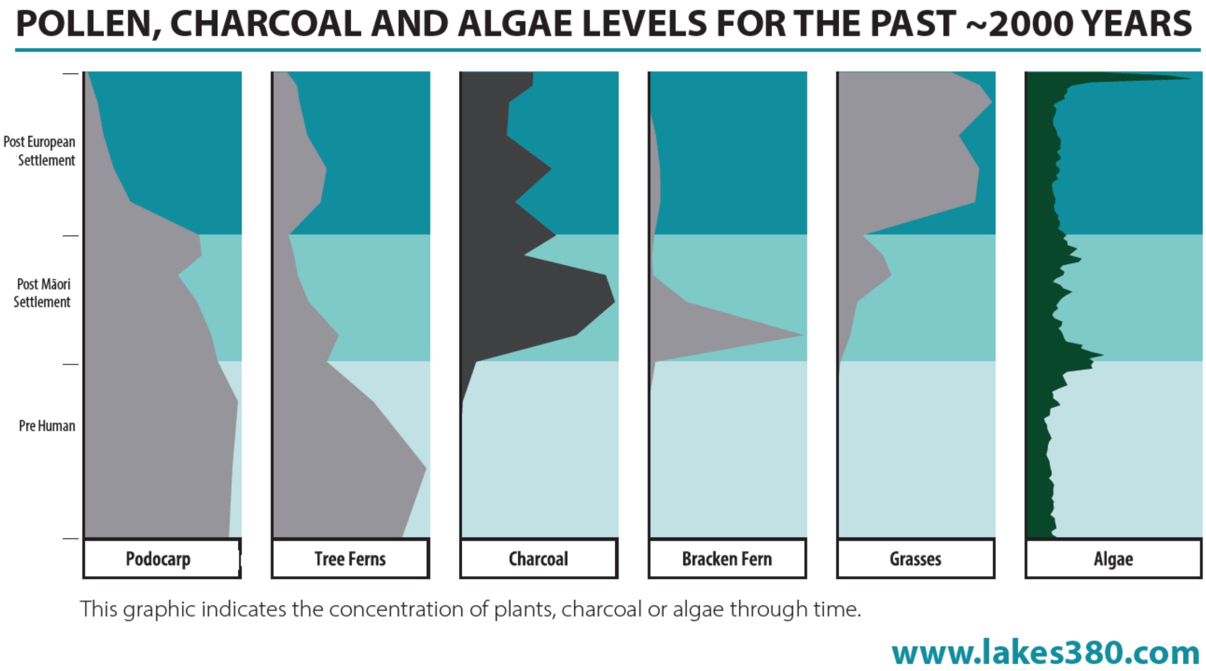

Figure 30. Selected palaeolimnological data from Upper Tomahawk Lagoon, showing patterns of catchment and in-lake changes as a result of Māori and European land-use practices.

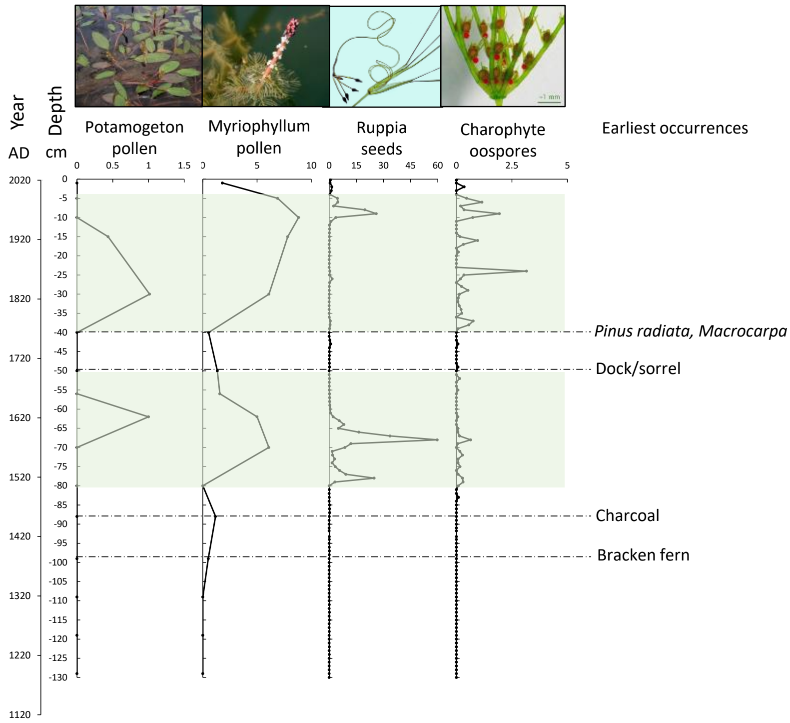

Figure 31. Submerged macrophyte indicator abundances going downcore and back in time. The abundance numbers are % of total pollen for the pollen data and seed and oospore counts per unit sample fresh mass. Green shading represents the inferred historical periods of macrophyte abundance in the lake. The dashed horizontal lines show when terrestrial plant indicator species first appeared and where charcoal and bracken ferns show substantial upward increases in the sediment record.

The most recent sediments show another peak in algal pigment concentration, which also seems to correspond with a recent decline in submerged plant indicator abundance, further supporting the hypothesis that periods of algal dominance and macrophyte dominance alternated twice in this lake over the time period represented in the sediment cores. Alternative stable states have been described for many shallow lakes in Aotearoa New Zealand (Schallenberg and Sorrell 2009) and elsewhere (Scheffer 2004) and were identified as a dynamic for Upper Tomahawk Lagoon in the late 1960s and early 1970s (Mitchell 1989).

The rate of sediment infilling in Upper Tomahawk Lagoon has increased in recent times to 5.2 mm yr^{-1} from a prior mean sediment infilling rate of 1.1 mm yr^{-1} (Figure 32). The recent high sediment infilling period occurred from c. 1937 to the present, whereas the mean rate is estimated to be 1.1 mm yr^{-1} from c. 1407 to c. 1937. Prior to c. 1407, the sediment infilling rate was calculated to be 1.5 mm yr^{-1}. This suggests an increase in sediment transport from the catchment to the lake began just before WWII. However, the estimated date of change is somewhat uncertain, and could reflect the post-war period of rapid intensification of agriculture due to the onset of top-dressing and the use of efficient tractors and other agricultural technologies. This post-WWII increase in soil erosion and fertiliser use initiated a period known as the great acceleration when anthropogenic pressures on the environment increased rapidly in Aotearoa New Zealand and in many other parts of the world (Steffen et al. 2015).

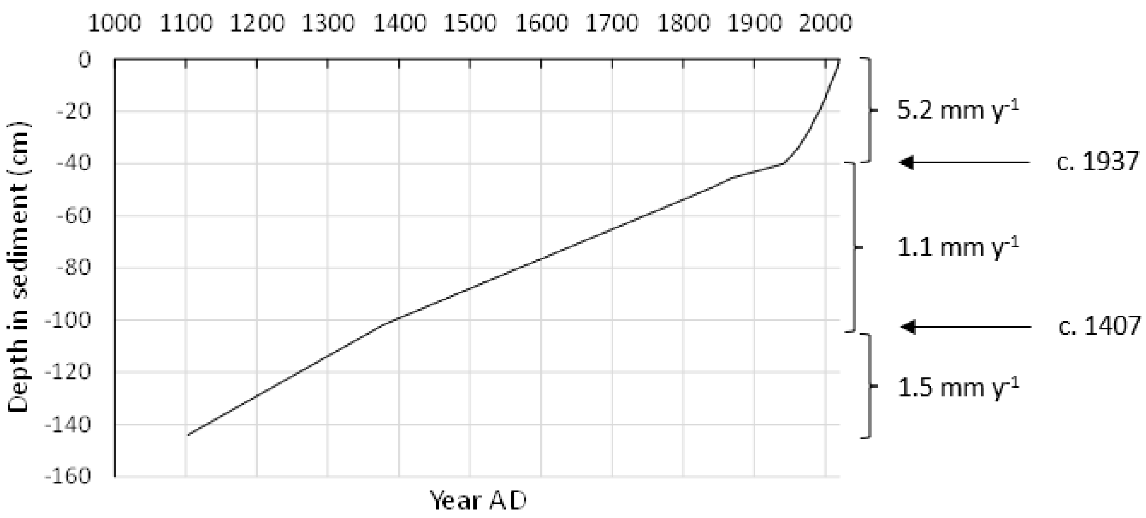

Figure 32. Age-depth model for Upper Tomahawk Lagoon showing inferred average sediment infilling rates. Rates have not been corrected for dewatering and, therefore, somewhat exaggerate increases in particulate matter deposition in recent times.

Lower Tomahawk Lagoon

Calculated sediment infilling rates in Lower Tomahawk Lagoon did not show significant changes over historical time (Figure 33). The rates calculated throughout the core were approximately the same as the rates in Upper Tomahawk Lagoon prior to c. 1937, falling between 1.0 and 1.9 mm yr^{-1} (uncorrected for dewatering).

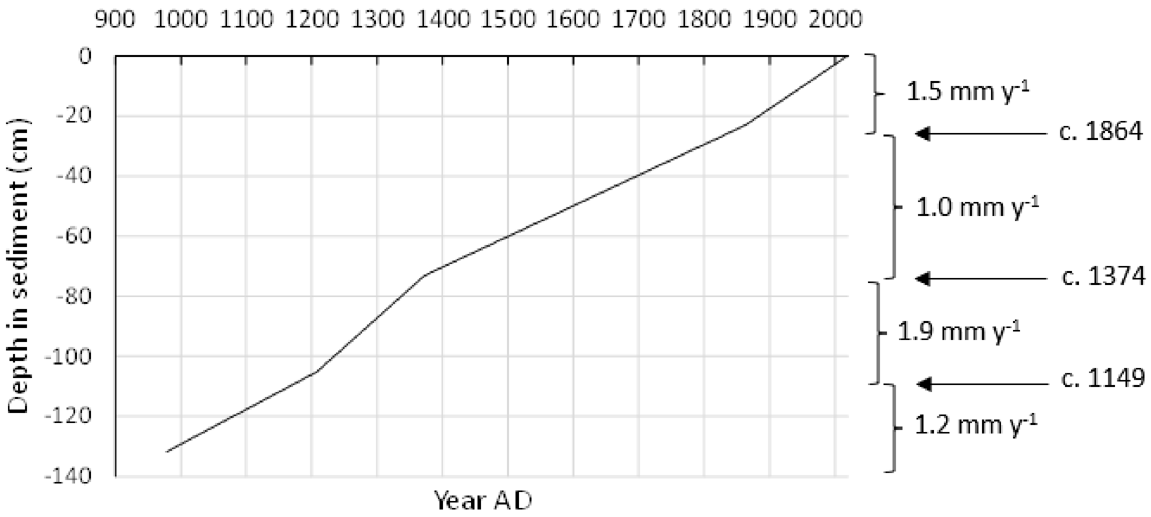

Figure 33. Age-depth model for Lower Tomahawk Lagoon showing inferred average sediment infilling rates. Rates have not been corrected for dewatering.

3.3.1. Palaeohistory analyses conclusions

In summary, the palaeolimnological data seem to indicate that the upper lagoon underwent historical transitions between phases of macrophyte abundance and phases of phytoplankton or benthic algal abundance. The macrophyte phases lasted 200 to 300 years, and the earliest recorded phase in the sediment record began after Māori settlement. During the period of early European settlement, macrophyte abundance declined, but then increased again in the 1950s and has persisted until the most recent sediment stratum was deposited. In the most recent sample, the abundances of all indicators of macrophyte declined while algal pigments increased. This could signal the beginning of another shift towards algal dominance. The age-depth model suggests that following a fairly stable and low rate of infilling over many centuries, sediment infilling has increased markedly since the 1950s.

The age-depth model for the lower lagoon did not show the same recent increase in sediment infilling rate, suggesting that the increasing erosion rate affecting the upper lagoon did not impact the lower lagoon. Alternatively, the increase in infilling in the upper lagoon may reflect the result of dredging and / or land reclamation activities, which were not undertaken at the lower lagoon.

3.4. Current ecological condition

The assessment of the condition of the Tomahawk Lagoons benefits from a wealth of studies and monitoring that has been undertaken since the 1960s. Most of the ecological studies are components of major, multi-year research efforts (Figure 34).

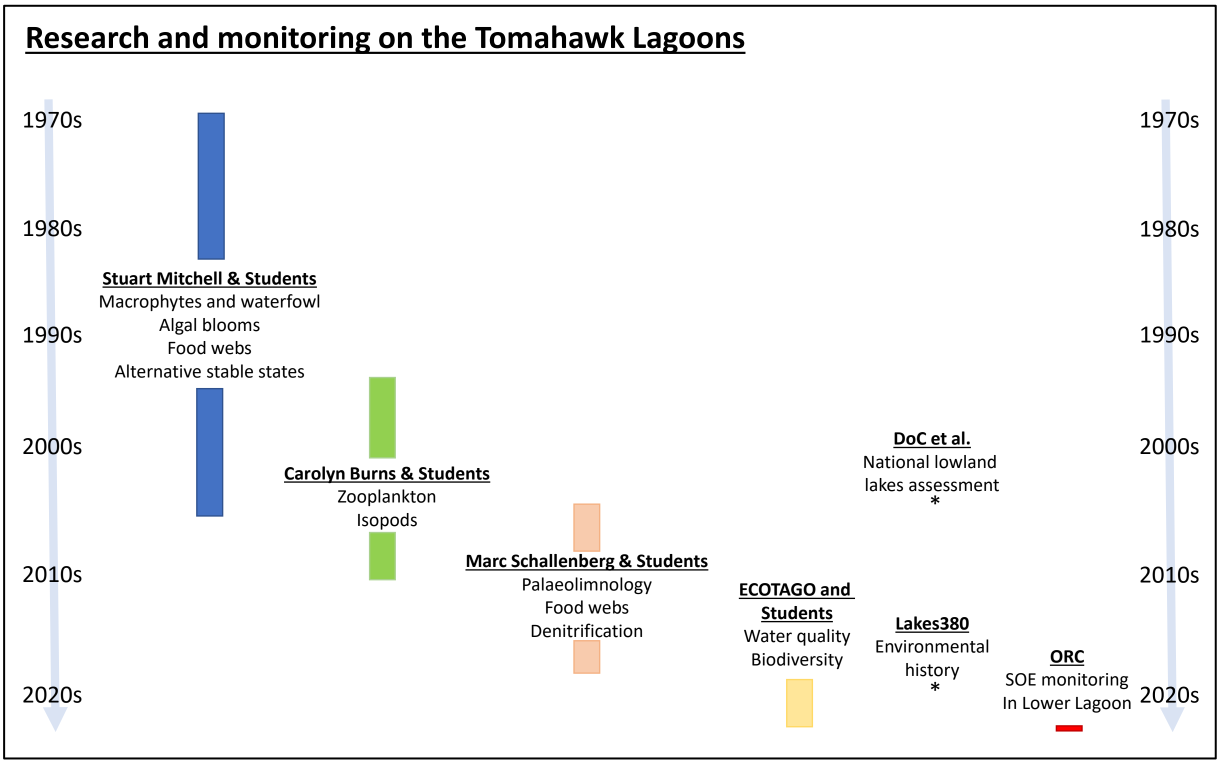

Figure 34. Historical summary of major research and monitoring programmes undertaken on the Tomahawk Lagoons. The asterisk (*) indicates one-off sampling of the lagoons as part of multi-lake studies.

Previous studies on both the upper and lower lagoons have identified these lakes as 'nutrient enriched' (Crawshaw et al. 2018), 'hypertrophic' (Mitchell et al. 1988; Crawshaw et al. 2019) and 'highly eutrophic with frequent algal blooms' (Mitchell 1989). McKinnon and Mitchell (1994) described the trophic state of Upper Tomahawk Lagoon: 'extreme variations in apparent trophic status of Tomahawk Lagoon No. 2 (i.e. the upper lagoon) can be seen from the wide range of annual average phytoplankton chlorophyll *a* and euphotic depths (Z_{eu}) in different years.' This highlights the characteristic alternation between a clear water, macrophyte-dominated state and a turbid, phytoplankton-dominated state. It is difficult to assess the ecological condition because the water quality, food webs and the influences of other stressors are affected by the large variation in macrophyte abundance that characterises the upper lagoon, and possibly also the lower lagoon (although this dynamic has not been studied as thoroughly in the lower lagoon).

3.5. Water quality

ORC has not historically monitored the Tomahawk Lagoons as part of its state of the environment monitoring programme for lakes. The only available datasets for which the ecological condition of the lagoons can be compared to the ORC Water Plan guidelines and NPS-FM limits (see Table 2) is the dataset belonging to the Ecotago citizen science project. The Ecotago monitoring programme was not designed to thoroughly assess the condition of the lagoons in relation to the guidelines, but some of the attributes monitored can be used to provide an assessment against some of the guidelines. Ecotago have produced two report cards that assess data from multiple sites in both lagoons and in inflowing creeks against the NPS-FM and ORC Water Plan guidelines. The results of these assessments are reproduced in Figure 35.

b. 2019

Results						
Site	Nitrogen*	Phosphorus	Algae	Max Algae	Turbidity	*E. coli*
Creek (Site 3a)	n/a	Fail	-	-	Fail	Fail
Creek (Site 6)	n/a	Fail	-	-	Fail	Fail
Upper Lagoon (Site 1)	n/a	Fail	C	C	Fail	Fail
Upper Lagoon (Site 2)	n/a	Fail	C	C	Fail	Fail
Lower Lagoon (Site 4)	n/a	Fail	C	C	Fail	Fail
Lower Lagoon (Site 5)	n/a	Fail	C	C	Fail	Fail

* Some extremely high numbers at all sites. This means that excel couldn't calculate the 80th percentile. The data was too far away from the norm.

All results were compared with the Otago Water Plan Receiving Water limits, except for Algae and Max Algae which were graded based in the the guideline values found in the National Policy Statement for Freshwater Management (NPSFM).

c. 2018

Results						
	Nitrogen	Phosphorus	Algae	Max Algae	Turbidity	*E. coli*
Creek (Site 3)	Fail	Fail	-	-	Fail	Fail
Upper Lagoon (Site 2)	Fail	Fail	C	D	Fail	Fail
Upper Lagoon (Site 1)	Fail	Fail	C	D	Fail	Fail
Lower Lagoon (Site 4)	?*	Fail	C	C	Fail	Fail
Lower Lagoon (Site 5)	?*	?**	C	C	Fail	Fail

All results were compared with the Otago Water Plan Receiving Water limits, except for Algae and Max Algae which were graded based on the guideline values found in the National Policy Statement for Freshwater Management (NPSFM).

* We are unable to determine a result as Nitrate-N was measured whereas limits contained in the Otago Water Plan are for Total Nitrogen.

** We are unable to determine a result as Dissolved reactive phosphorus was measured whereas limits contained in the Otago Water Plan are for Total Phosphorus.

Figure 35. Report cards for the Tomahawk Lagoons and inflow creeks from data collected in 2019 (b) and 2018 (c). Source: https://tomahawkcitizenscience.com/results/report-cards

These report cards show that the lagoons and creeks all breached some of the water quality guidelines in the 2 years they were assessed. All sites consistently failed to meet the *E. coli* and turbidity guideline thresholds in the ORC Water Plan. The upper lagoon failed the NPS-FM 95th percentile phytoplankton (i.e. chl-*a*) guideline in 2018, but both lagoons achieved the 'fair' category (i.e. C) for the annual median in both years, while the lower lagoon also achieved the fair category for the 95th percentile in 2018. For N and P, the Ecotago team measures DRP and nitrate, whereas both the ORC Water Plan and the NPS-FM guidelines stipulate thresholds for TN and P. However, the ORC Water Plan thresholds for DRP almost always exceeded the TP guideline at all sites, allowing an assessment of 'fail' against this guideline. Nitrate levels did not exceed the ORC Water Plan guidelines for TN; therefore, Ecotago was unable to assess the water quality based on TN guidelines.

Ecotago's assessments clearly support the previous qualitative work undertaken by experts, which demonstrated that these lakes exhibit signs of eutrophication. The assessments also support the ongoing efforts made by the community and ORC to develop a restoration plan for these lakes and their catchments.

3.5.1. Dissolved nutrients

Lagoon dissolved nutrient concentrations have been monitored in both the Upper and Lower Tomahawk Lagoons by Ecotago since 2016. Figure 36 shows the Ecotago data for DRP in the lagoons and inflow creeks. The 80th percentile of these data exceeds the ORC Water Plan guideline, especially for the lower lagoon and for the inflow streams. TP concentrations in the lakes would have been even higher than the DRP data shown; therefore, the comparison of the lake data against the guideline is a conservative assessment of the situation with regard to TP.

In the lakes, the DRP concentrations seem to peak in summer and show a relatively consistent seasonality, with no apparent trend over the entire time series. Some seasonality is also apparent in the creek data, whereby levels in winter are generally lower than levels in summer.

Ecotago's data on nitrate-N is shown in Figure 37. The ORC guideline for lakes is based on TN. Therefore, an assessment of lake levels against the guideline cannot be made because the TN levels would have been higher than the nitrate levels measured. However, the data for the inflow creeks shows that nitrate-N concentrations in the inflows far exceed the nitrate guideline for rivers (80th percentile of the data). In particular, Lagoon Creek, which drains into the upper lagoon, often far exceeds the ORC nitrate guideline, sometimes over 50-fold. In fact, nitrate-N levels in this creek sometimes exceed the NPS-FM nitrate toxicity national bottom lines (the guideline annual median is 2400 $mg.m^{-3}$ and the 95th percentile is 3500 $mg.m^{-3}$), highlighting that nitrate levels in this creek are often extremely high. In general, nitrate-N levels in Lagoon Creek far exceed those in the creek flowing into the lower lagoon (Figure 13).

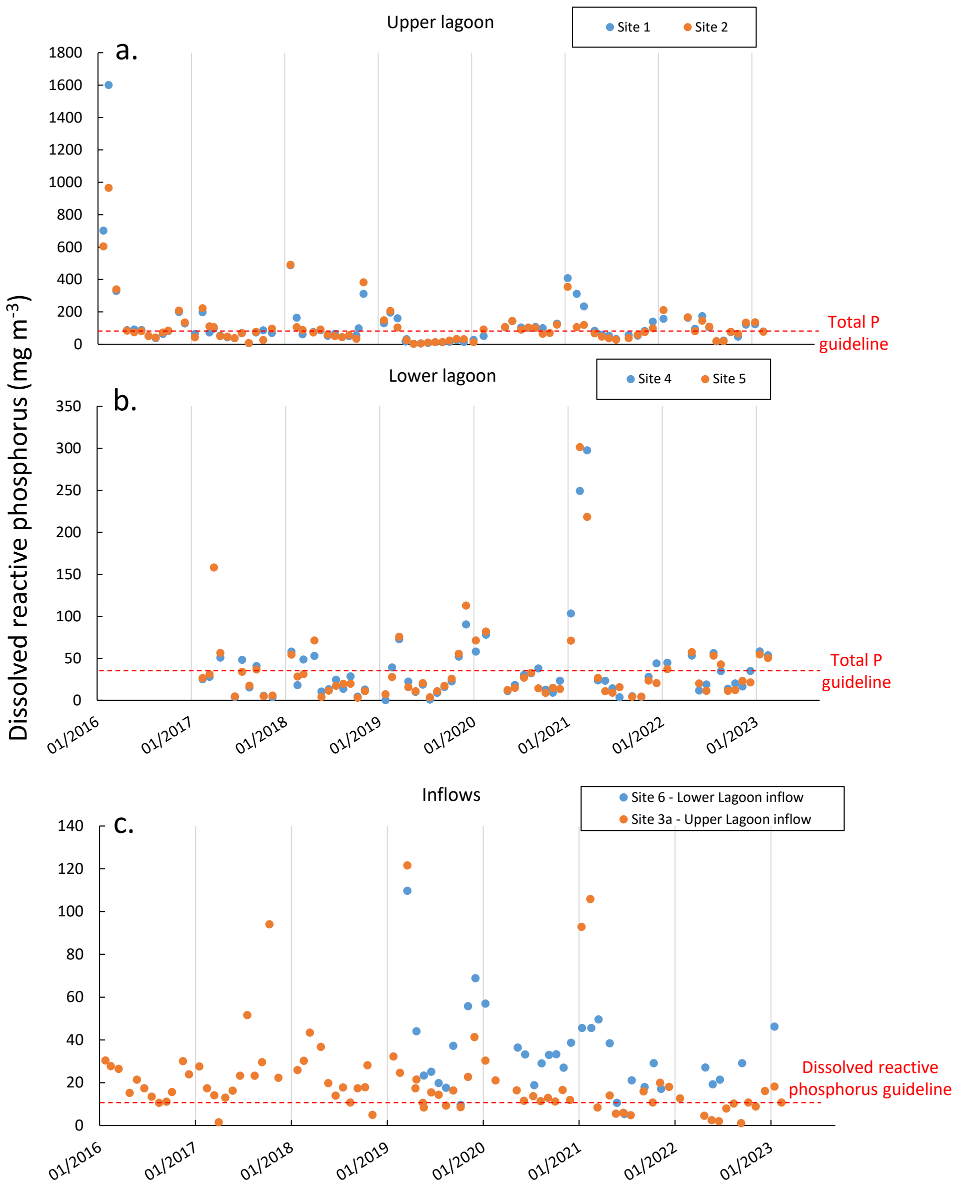

Figure 36. Dissolved reactive phosphorus concentrations in the waters of the Upper (a) and Lower (b) Tomahawk Lagoons and the inflow creeks (c). The horizontal red lines are the Otago Regional Council Water Plan guidelines (Otago Regional Council 2022) for total phosphorus (a and b) and for dissolved reactive phosphorus (c) (80th percentile). The NPS-FM total phosphorus national bottom line for these lakes is 50 mg.m^{-3} (not shown), which is higher than the Otago Regional Council Water Plan guideline of 33 mg.m^{-3}.

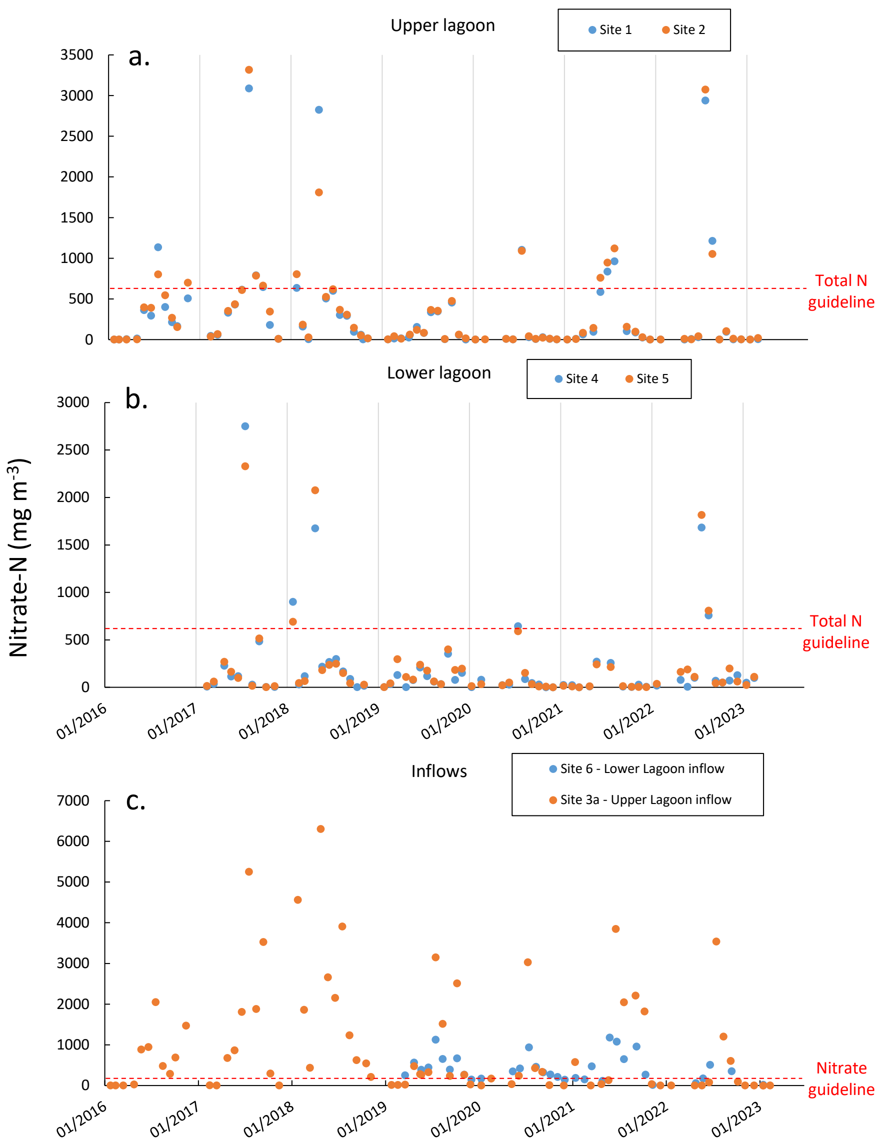

Figure 37. Nitrate-nitrogen (N) concentrations in the waters of the Upper (a) and Lower (b) Tomahawk Lagoons and the inflow creeks (c). The horizontal red lines are the Otago Regional Council Water Plan guidelines (Otago Regional Council 2022) for total N (a and b) and for nitrate-N (c) (80th percentile). The NPS-FM nitrate-N national bottom lines (toxicity) for rivers are 2,400 mg.m^{-3} (annual median) and 3,500 mg.m^{-3} (95th percentile). The creek flowing into the upper lagoon sometimes exceeds these values. The NPS-FM total N national bottom lines for these lakes are 750 mg.m^{-3} for the lower lagoon and 800 mg.m^{-3} for the upper lagoon (not shown), which are higher than the Otago Regional Council Water Plan guideline of 550 mg.m^{-3} for these lakes.

3.5.2. Phytoplankton biomass

Phytoplankton biomass has been monitored as chl-*a* in both Upper and Lower Tomahawk Lagoons by Ecotago since 2016. Ecotago's data from 2016 to 2023 indeed show that between these years, phytoplankton biomass varied greatly in the lake, with generally low levels in Upper Tomahawk Lagoon; however, for some periods, the levels did exceed the annual maximum threshold defined in the NPS-FM (Figure 38). While some samples showed elevated levels of chl-*a*, there was little evidence of a prolonged period of high algal biomass during this period, apart from during the spring and summer of 2019/20. At this time, there was no public notification of algal blooms, suggesting that cyanobacterial taxa were not dominant (there are no phytoplankton cell counts available for this period). However, the 2019/20 event was a seasonal bloom and did not persist for multiple years, unlike earlier reports of Upper Tomahawk Lagoon (e.g. Mitchell 1989).

Chl-*a* levels were generally higher in the Lower Tomahawk Lagoon than the upper lagoon, although the algal bloom did not achieve levels as high as those in the upper lagoon (Figure 38). Again, these data show no evidence of persistent blooms lasting multiple years. As shown in the Ecotago report, both lakes exceeded the NPS-FM annual maximum national bottom-line guideline in some years, but the annual median national bottom-line guideline seems to have been met in most years.

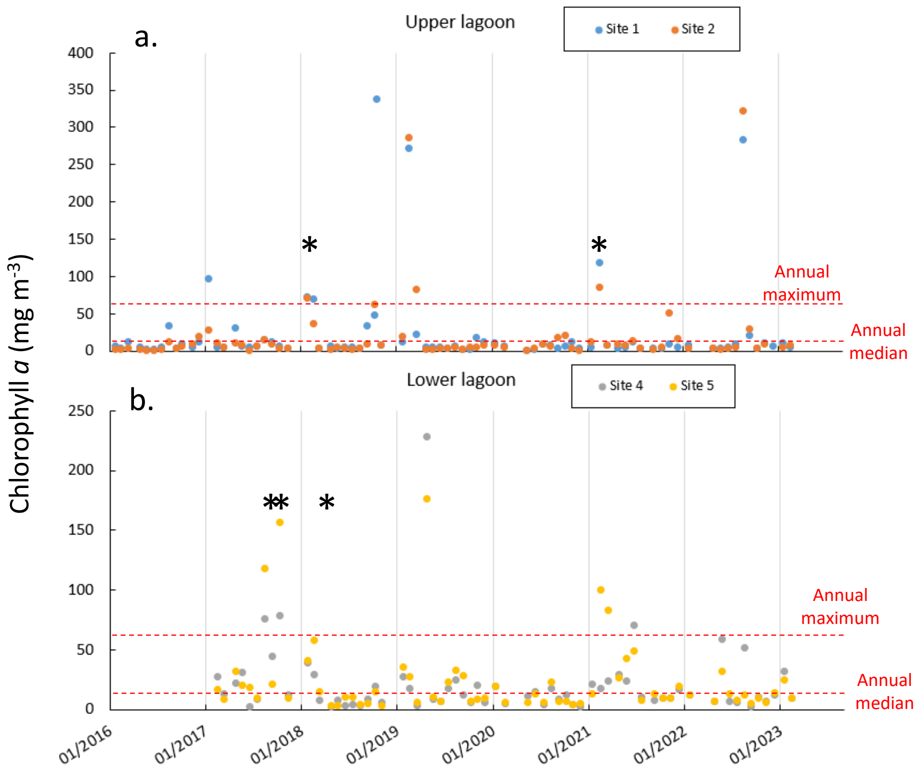

Figure 38. Chlorophyll-*a* concentrations in the waters of the Upper (a) and Lower (b) Tomahawk Lagoons. The horizontal red lines are the national bottom-line guidelines from the NPS-FM (Ministry for the Environment 2022). Asterisks are times when public notifications of cyanobacterial blooms were issued by the Otago Regional Council.

3.5.3. *Water clarity*

Lake clarity has been monitored as turbidity in both the Upper and Lower Tomahawk Lagoons and their inflows by Ecotago since 2016. Figure 39 shows the turbidity data time series for the upper and lower lagoons and for the inflow creeks. Turbidity exceeded the ORC guidelines at all the monitored sites. The turbidity data for the lagoons showed multi-year, alternating phases of low and elevated turbidity (e.g. mid-2017 to mid-2019 in the upper lagoon) interspersed by phases of clearer water (e.g. mid-2019 to early 2021). Turbidity in the creek entering the upper lagoon appears to have decreased markedly after 2018.

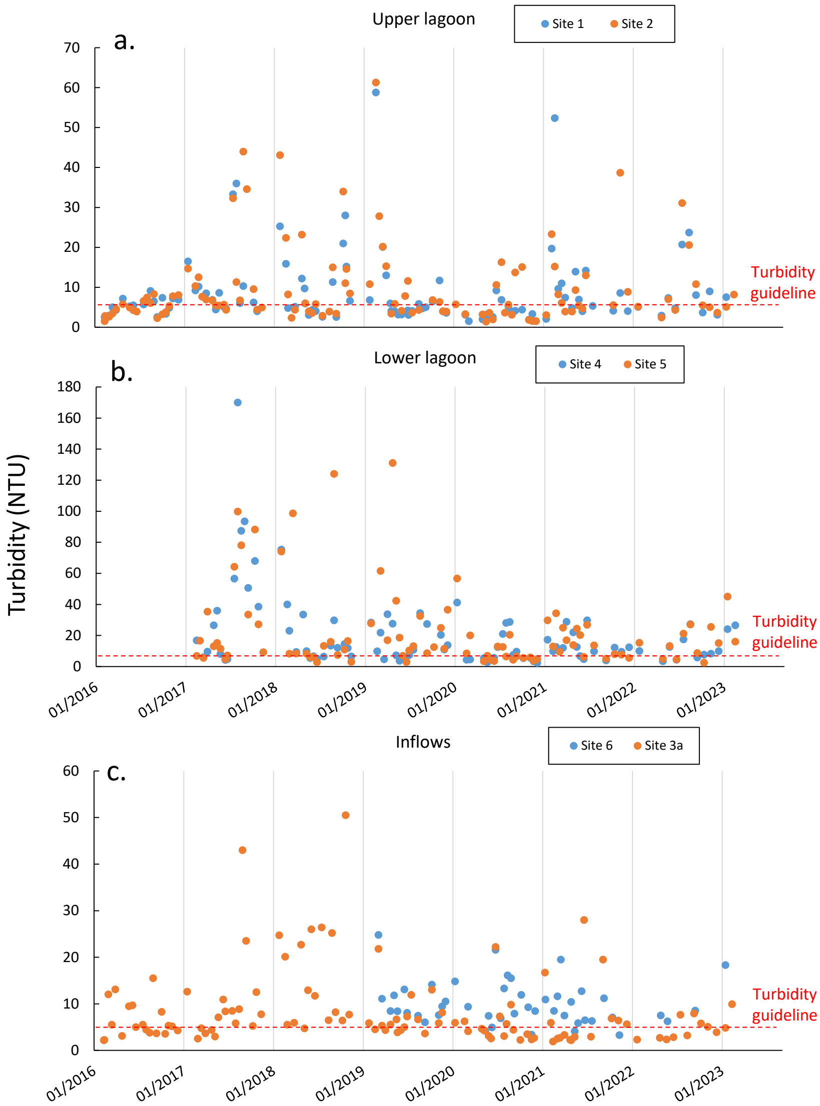

Figure 39. Turbidity in the waters of the Upper (a) and Lower (b) Tomahawk Lagoons and the inflow creeks (c). The horizontal red lines are the Otago Regional Council Water Plan guidelines (Otago Regional Council 2022) for turbidity (80th percentile).

3.5.4. *Sediment geochemistry*

Historical nutrient loads from the catchment can be deposited in the lakebed due to the binding of P to particulate matter and its sedimentation. Nutrients can then desorb or be mineralised from the sediment and may diffuse back into the water column under certain conditions (e.g. under anoxia, during microbial mineralisation, and / or under high pH conditions). In addition, bioturbation and wind-induced sediment resuspension can resuspend particle-bound nutrients and entrain nutrients dissolved in pore water. Furthermore, macrophyte roots can obtain nutrients from lakebed sediments, translocating N and P to the plant tissues. These legacy nutrients are recycled back into the water column if the plant tissues are eaten and excreted or decomposed. Thus, the bed of the lake contains a pool of nutrients that can continue to influence nutrient availability in the water column, and through such mechanisms, historically high nutrient loads may elevate lake productivity, even if external loads have reduced over time.

The surface sediment of Lower Tomahawk Lagoon was sampled by the Lakes380 programme to determine the contents of different P fractions on a per dry weight basis (Figure 40). This analysis was not undertaken for the upper lagoon.

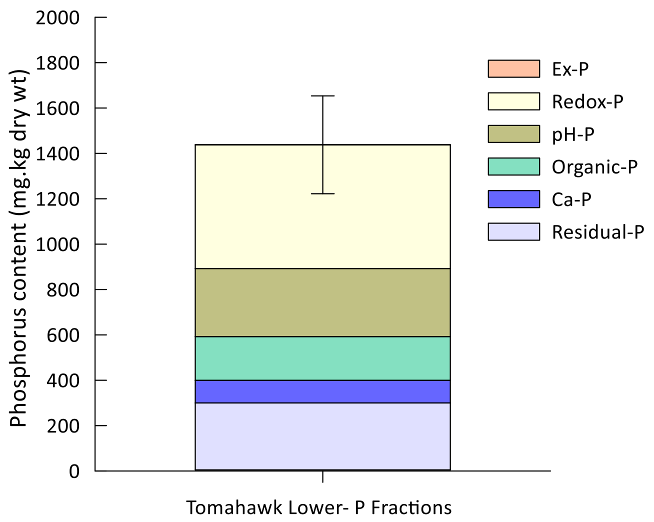

Figure 40. Sediment phosphorus (P) fractions in the surface sediment sample (0–2cm) taken in Lower Tomahawk Lagoon during the Lakes380 national-scale study. The error bar denotes a 13% error associated with analysis and sampling.

The redox-P fraction can be solubilised during anoxia, the pH-P fraction can be solubilised at high pH (e.g. over about 9.0 pH), and the organic-P fraction can be mineralised by microbial activity. Thus, in the lower lagoon, approximately 70% of the P sequestered in the sediment can potentially be resolubilised. Placing the surficial sediment samples in an anoxic environment releases the redox-P, and such an experiment was carried out with sediment from the lower lagoon (Figure 41). This experiment showed that the sediment P release rate under anoxic conditions was higher than the average release rate measured in the multi-lake dataset collected by the Lakes380 programme. Together, these findings show that there is a large pool of potentially soluble P in the sediment of the lower lagoon.

Figure 36 shows that peaks in DRP in the water column of the lower lagoon tend to occur in summer, when water temperatures are higher and when conditions of high pH and low DO are more likely to occur. This suggests that in some summers (e.g. 2020/21 and 2021/22), a recycling of sediment P back into the water column probably contributed to elevated DRP levels in the lake. Unfortunately, the available data do not indicate which of the many potential mechanisms are responsible for high summer DRP in the lake.

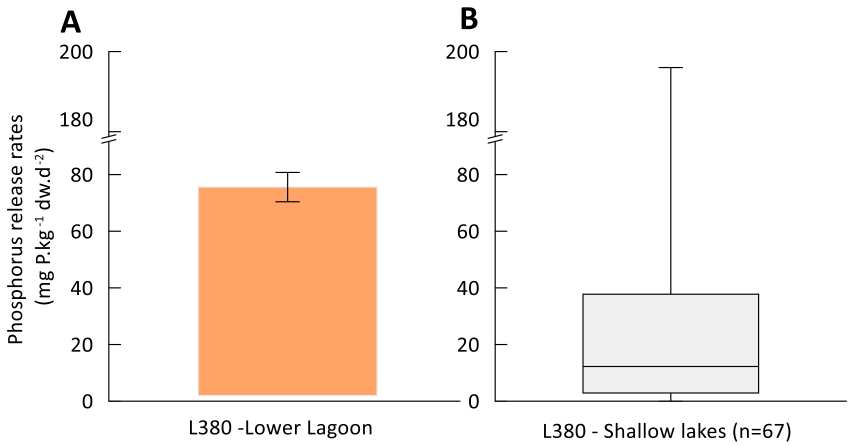

Figure 41. Phosphorus release rates determined by a slurry experiment on a single surface sediment (0–2cm) sample from Lower Tomahawk Lagoon during the Lakes380 national-scale study (A), and summary statistics (range, 25th and 75th quartiles and median) of release rates determined on 67 shallow lake sediment samples (Lakes380 dataset; B). The error bar on B is the standard error derived from the Lakes380 dataset.

3.5.5. Dissolved oxygen

Dissolved oxygen concentrations in the lagoons and creeks were measured on a monthly basis; however, these measurements were carried out in the daytime, when DO concentrations are often enhanced by the photosynthesis of aquatic plants. Oxygen sags are more likely to occur at night, highlighting the importance of measuring oxygen using sensors that can log DO concentrations automatically at high frequency, allowing the measurement of oxygen minima.

In 2017, Dirk Van Walt (Van Walt Ltd) loaned Ecotago an oxygen sensor, which was placed 10 cm above the lakebed of the upper lagoon. The deployment lasted from February 22 until April 17, a time of year when water temperatures are near annual highs and oxygen sags are expected. The data, presented in Figure 42, show large daily peaks and nightly sags over the time period. The NPS-FM national bottom-line guideline is also shown, indicating that DO in the lagoon did not breach this guideline. The results indicate that DO did not decrease sufficiently to allow the dissolution of iron- and manganese-bound P. However, the sags, which went below 4 $mg.l^{-1}$ at times, were low enough to affect some aquatic organisms, including fish (Franklin 2013; MfE 2023). This is consistent with occasional observations of common bully die-offs, which have been observed in the upper lagoon (Marc Schallenberg, pers. obs.).

In addition to P solubilisation and oxygen stress on sensitive biota, anoxia near the sediment surface can also result in ammonium diffusion (which can be toxic if pH is high) and toxic hydrogen sulphide diffusion into the water column. The data collected from the upper lagoon between September 2022 and March 2023 as a part of ORC's new monitoring programme, indicated that ammoniacal N levels were far below the toxicity threshold during the sampling period, with the highest measurement being 197 $mg.m^{-3}$. The data from the DO sensor, shown in Figure 42, indicated that sediment anoxia is unlikely to have occurred during the sensor deployment, but the significant oxygen sags measured suggest that transient anoxia might occur in this lake at times.

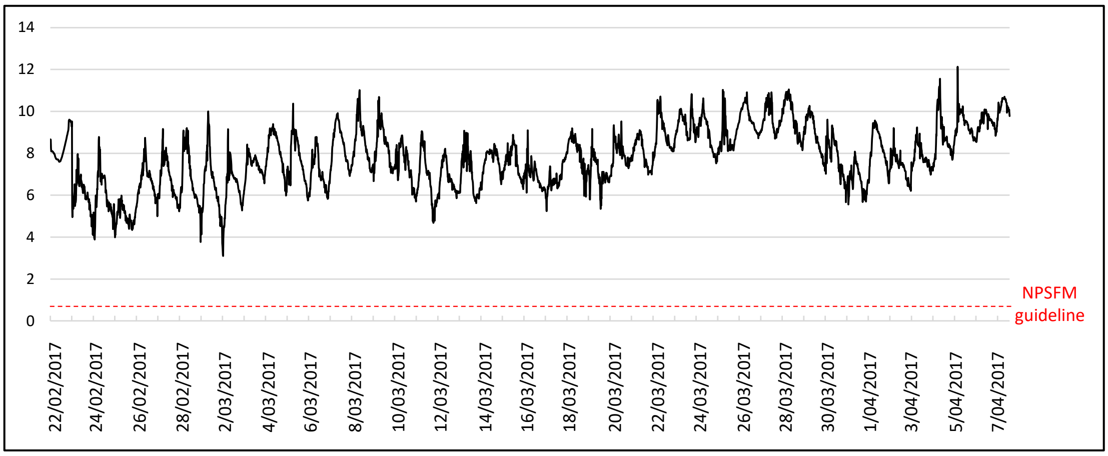

Figure 42. Dissolved oxygen concentrations ($g.m^{-3}$) in Upper Tomahawk Lagoon measured at 5-minute intervals from 22 February to 7 April 2017. The sensor was 10 cm off the bottom of the lake. The NPS-FM guideline shown in red is the national bottom line for annual minimum dissolved oxygen concentration.

3.5.6. *E. coli*

A graphical presentation of the Ecotago data for *E. coli* was not possible because a large proportion of the samples were above the upper measurement threshold of the method (2,429.6 cells per 100 mL). This was observed for all monitored sites, indicating that the *E. coli* levels at these sites breach the NPS-FM national bottom lines and the regional council guidelines. Thus, the waterbodies in the catchment of the Tomahawk Lagoons have concentrations of *E. coli* so high that the waterbodies should not be considered safe for contact recreation.

3.5.7. Phytoplankton and macroalgae

Both Tomahawk Lagoons are known to exhibit cyanobacterial and dinoflagellate blooms that instigate action by ORC, including sampling to determine phytoplankton biomass and species identifications. Signs warning the public of the potential toxicity of the algal material are sometimes erected at access points to the lakes. At times, both lakes exhibit signs of severe eutrophication, including algal blooms and low water clarity, but these phases alternate with periods of higher water clarity and an abundance of aquatic plants. This ongoing dynamic means that it is challenging to assess and monitor the health of the lakes. The outcome of a lake health assessment depends on when it is carried out and the period of time over which the assessment takes place.

A detailed analysis of phytoplankton community structure is beyond the scope of this report. However, Ecotago produced monthly phytoplankton data for the two lagoons from July 2021 to December 2022. In addition, media reports of occasional lagoon closures and other observations are available to provide an overview of the key taxa that can dominate the phytoplankton in these lakes. Nuisance cyanobacterial blooms do occasionally occur in both lakes and comprise taxa such as *Dolichospermum lemmermannii* and picocyanobacteria. In addition, occasional dinoflagellate blooms occur in both lakes caused by *Gymnodinium* sp. and / or *Peridinium* sp.

Dolichospermum blooms can be toxic, resulting in signage being put up on the publicly accessible lake shores. Since 2012, two cyanobacterial blooms have been notified in the lower lagoon (one in February 2014 and one in October / November 2017). Four cyanobacterial blooms have been notified in the upper lagoon (one in December 2012 / January 2013, one in November 2013, one in February 2017 and one in January 2018).

Both lagoons can also occasionally exhibit blooms of benthic macroalgae, such as *Enteromorpha* sp. and *Ulva* sp. Filamentous algae can also be important. In March 2023, we observed the conspicuous presence of *Ulva* sp. in both lagoons, as did de Winton et al. (2023) in their macrophyte survey of June 2023.

3.5.8. Macrophytes

The macrophyte cover, biomass and dynamics of Upper Tomahawk Lagoon have been extensively studied (e.g. Mitchell 1989; McKinnon and Mitchell 1994; Drake et al. 2009; de Winton et al. 2023). Macrophyte abundance and species composition have varied greatly over time in this lake, with *Ruppia* sp., *Stuckenia pectinatus*, *Myriophyllum* sp., *Potamogeton* sp., *Elodea canadensis* and charophytes appearing frequently.

In March 2023, we conducted a survey and found low amounts of macrophyte biomass but many small shoots of what appeared to be *Ruppia* sp. In June 2023, de Winton et al. (2023) carried out a LakeSPI assessment of the upper lagoon, which reported that the macrophyte community was in a moderate condition. *Ruppia polycarpa* was the dominant macrophyte species, followed by *Elodea canadensis*, which was described as sub-dominant. *Potamogeton ochreatus*, *P. cheesemanii*, *Althenia bilocularis* and the charophytes *Nitella hyalina* and *Chara globularis* were also present in small quantities.

De Winton et al. (2023) reported some *Ruppia polycarpa* in the lower lagoon, but the salinity of the lagoon was high enough to restrict the utility of the LakeSPI index. Therefore, the macrophyte status of the lower lagoon was not assessed as thoroughly as that of the upper lagoon.

Observations and studies of macrophytes undertaken in both lagoons indicate that macrophyte biomass is generally lower in the lower lagoon than in the upper lagoon (this study; de Winton et al. 2023; Marc Schallenberg, pers. obs.). *Elodea canadensis* and *Ranunculus trichophyllus* are the only non-native submerged macrophyte reported from the lagoons. These species are given an invasive impact score of 3 and 1 out of 7, respectively (Clayton and Edwards 2006), indicating that, in general, they are only moderately to mildly invasive. It is not clear whether these macrophytes threaten the native macrophyte communities of the upper lagoon. Due to its higher salinity, the lower lagoon is not likely to be impacted by these invasive species.

3.6. Tomahawk Lagoon food webs

3.6.1. Fish community

Fish surveys of the Tomahawk Lagoons have been carried out multiple times, including in 2006 (Drake et al. 2010) and, more recently, by Ecotago, who have been collecting fish data seasonally since July 2021. The New Zealand Freshwater Fish Database also contains additional entries for the Tomahawk Lagoons. All these sources have been examined to compile the species list for the two lagoons, shown in Table 11. The native species assemblage is typical of lowland South Island lakes. However, it is notable that shortfin eel are not well established in this catchment, having been reported only on one occasion in Upper Tomahawk Lagoon by the Ecotago team.

Fish sampling in 2006 and 2023 was undertaken with standardised sampling effort and methodology (as detailed in Drake et al. 2009), allowing for the comparison of catch rates between lakes and between sampling dates. Figure 43 shows the catch rates for fish in Lower Tomahawk Lagoon in 2006 and 2023 as well as the average catch rates for 22 South Island shallow, lowland lakes. The lower lagoon showed substantially higher catch rates of common bully, longfin eel and īnanga than have generally been reported for South Island lakes. The fish populations in the lower lagoon were quite similar in 2006 and 2023. Catch rates in Upper Tomahawk Lagoon (Figure 44) were substantially lower, but a similar species assemblage to Lower Tomahawk Lagoon was observed.

Table 11. Fish species reported for the Tomahawk Lagoons and tributaries based on multiple sources of information (see text). * indicates non-native fish, self-sustaining population; ** indicates non-native fish population sustained by stocking.

Upper Tomahawk Lagoon	Lower Tomahawk Lagoon	Tributaries
Perch*	Perch	
Rainbow trout**		
	Brown trout**	
Common bully	Common bully	Common bully
Longfin eel	Longfin eel	Longfin eel
Shortfin eel		
Īnanga	Īnanga	Īnanga
	Flounder	
		Banded kōkopu
		Redfin bully
		Kōaro

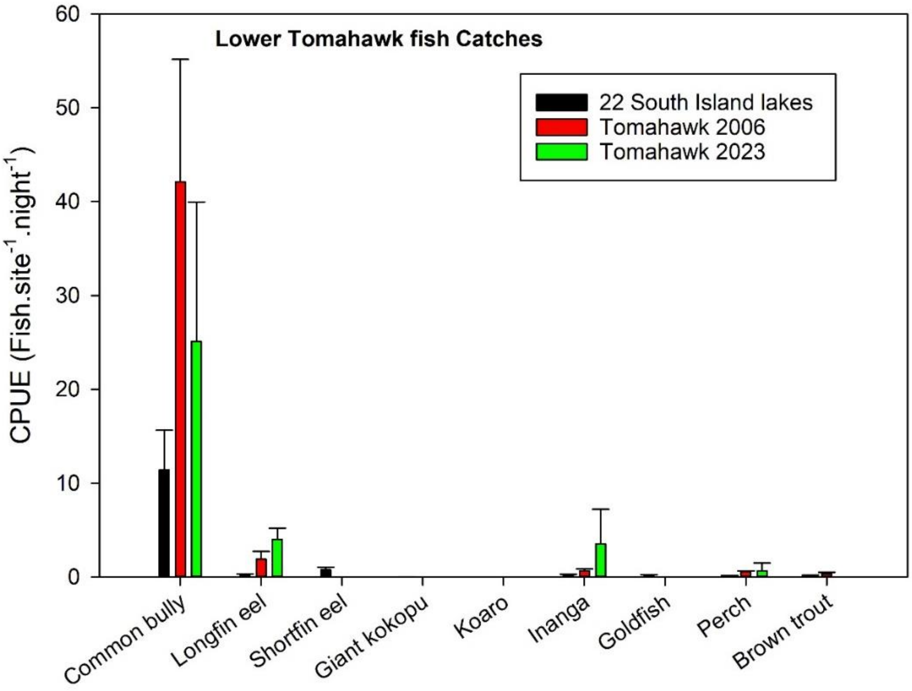

Figure 43. Fish catch per unit effort in Lower Tomahawk Lagoon in March 2006 and 2023. The average catch rates for 22 South Island lakes are also shown for comparison. Data source: Drake et al. (2009).

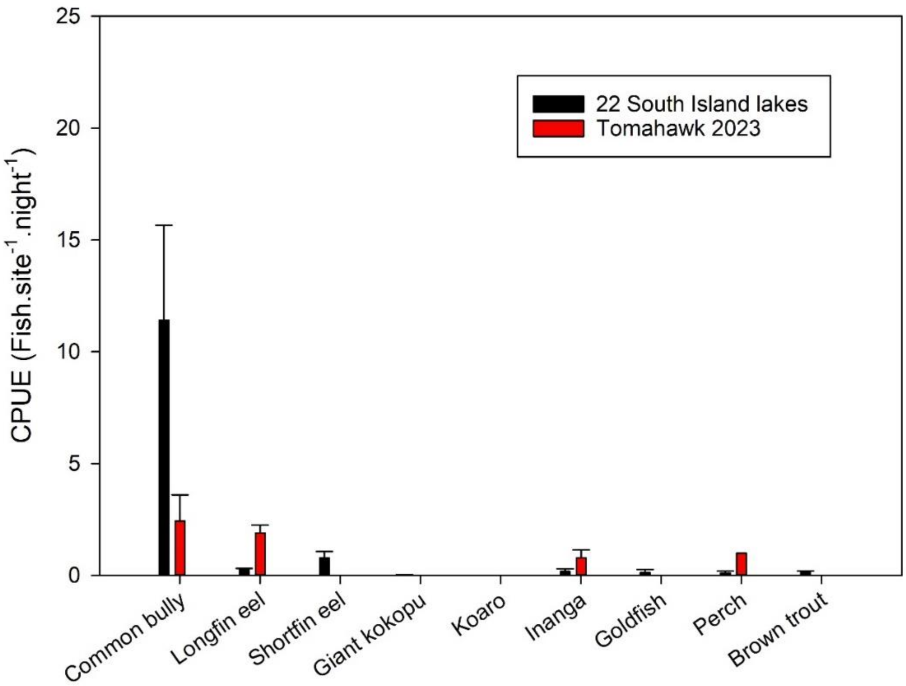

Figure 44. Fish catch per unit effort in Upper Tomahawk Lagoon in March 2023. The average catch rates for 22 South Island lakes are also shown for comparison. Data source: Drake et al. (2009).

3.6.2. Invertebrates

A thorough analysis of invertebrates of the Tomahawk Lagoons is beyond the scope of this report. Ecotago undertook some semiquantitative analyses of invertebrates, and the 2006 survey carried out by Drake et al. (2009) sampled invertebrates in the lower lagoon. Professor Carolyn Burns and Associate Professor Marc Schallenberg and their students have also undertaken studies of invertebrates, including zooplankton, in the lagoons (e.g. Dufour et al. 2007; Crawshaw et al. 2018, 2019).

High densities of the invasive *Daphnia pulicaria* and the native *D. thomsoni* are often observed in the upper lagoon. These are filter feeders and high densities of these zooplankters are often associated with high water clarity in the lagoons. Both are sensitive to salinity and are, therefore, much more important in the upper lagoon.

Freshwater crayfish / kōura (*Paranephrops* sp.) have been recorded in the lagoons and inflow streams. The lagoons also support mysids, amphipods, snails, isopods, polychaetes, caddisfly larvae, odonate larvae, chironomids and other invertebrates typical of coastal lakes and lagoons.

3.6.3. Waterbirds

Since February 2019, Ecotago, with the help of Mary Thompson, have been collecting information on waterbird numbers at the lagoons. Black swan numbers fluctuate between 0 and 467 birds per count. Other birds of note that use the lagoons are marsh crakes / kotoreke, white herons / kōtuku, shags, royal spoonbills / kōtuku ngutupapa, New Zealand shovelers / kuruwhengi, grey teal / tētē moroiti, Canada geese and gulls.

Otago Fish & Game monitors the number of black swans on the Tomahawk Lagoons (Figure 45). These data confirm high variations in black swan numbers over time and suggest a rise in swan numbers in the 1990s, followed by a decline in the 2010s. However, there was significant variation in counts. Black swan densities are likely to have tracked macrophyte cover in the lagoons to some extent (McKinnon and Mitchell 1994).

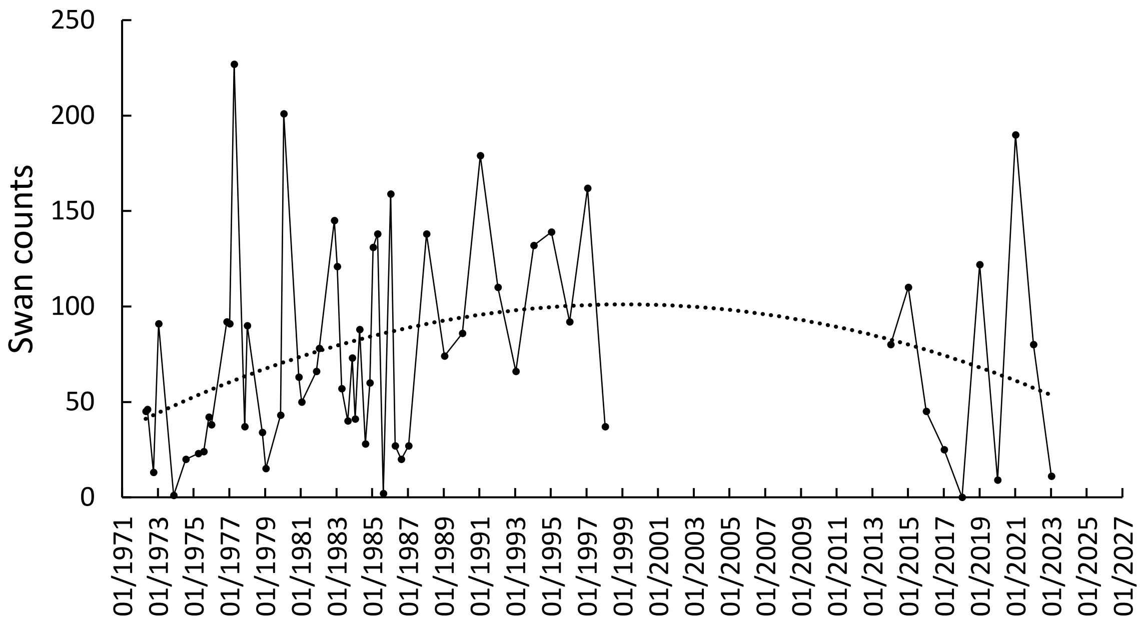

Figure 45. Swan counts at the Tomahawk Lagoons. Data source: J. Couper, Otago Fish & Game.

3.7. Summary of key water quality and ecology findings

The following points summarise the key water quality and ecological health findings for the Tomahawk Lagoons:

- The Tomahawk Lagoons and inflow creeks fail to achieve national and regional water quality guidelines for P, turbidity and *E. coli*. At times, the lake national bottom line for chl-*a* was also breached. The creeks breach the regional nitrate guideline, and it is likely that the lakes would breach TN guidelines, although only nitrate levels have been measured.
- The turbidity time series in the lagoons supports the previous observation that the upper lagoon is a flipping lake, alternating between multi-year clear water and turbid phases. The chl-*a* time series data do not appear to support this inference as strongly.
- The lakebed sediment in the lower lagoon has a substantial component of potentially mobile P, which could be mobilised by conditions of anoxia, high pH, high wind, and / or microbial mineralisation. When experiencing anaerobic conditions, lakebed sediments of the lower lagoon released substantial amounts of P into pore waters.
- High-frequency DO data were only available for a 6-week period but showed that concentrations did not breach the national bottom line, which was set to prevent anoxic releases of P from the lakebed. However, the DO data did show occasional oxygen sags below 4 $mg.l^{-1}$, reaching oxygen levels likely to cause physiological stress to fish.

- *E. coli* counts were very high at all sites, breaching both the national and regional guidelines for contact recreation.
- Occasional phytoplankton blooms occurred in both lagoons. These blooms were usually due to cyanobacterial taxa that are capable of fixing N and which may potentially produce cyanotoxins. Picocyanobacteria, dinoflagellates and macroalgae also occasionally bloom in the lakes.
- Observations indicate that macrophyte biomass and cover fluctuates substantially in the upper lagoon. The lake has flipped back and forth between turbid, algal-dominated and clear water, macrophyte-dominated phases for decades and possibly even centuries. Macrophytes appear to be more abundant in the upper lagoon. *Elodea canadensis* and *Ranunculus trichophyllus* appear to be the only non-native macrophytes in the upper lagoon and neither appear to be a large threat to the macrophyte community or lake health. The macrophyte community was recently assessed as being in moderate condition, but this was only based on a single assessment in June 2023.
- The fish communities in the lakes are typical of those found in other shallow South Island lakes. The lakes contain both introduced trout and perch and , for shallow, lowland lakes, unusually low numbers of shortfin eels.
- The invertebrate assemblages are also typical for shallow, coastal freshwater and brackish lakes. The upper lagoon sometimes has high densities of *Daphnia* sp., a zooplankter that is an effective grazer of algae.
- The Tomahawk Lagoons attract a wide range of birdlife. Black swan densities are highly variable and can reach high densities at times, which may result in significant grazing on macrophytes as well as nutrient cycling. If swans also feed on pasture grasses, their use of the lagoon may also increase external loads of nutrients to the lagoons.

3.8. Tomahawk Lagoon management options

A conceptual model of how shallow lakes undergo eutrophication is presented In Figure 2. In assessing the data on the Tomahawk Lagoons, both stressors (e.g. nutrients, turbidity) and responses (e.g. phytoplankton biomass, fish communities) were considered, as well as factors that mediate lake responses to stressors (McDowall et al. 2018). These mediators increase or decrease ecological resistance and resilience to eutrophication, potentially shifting tipping points and altering hysteresis in the lake systems. Some of the stressors and mediators can potentially be managed by mitigations (e.g. reducing contaminant inputs) or interventions (e.g. intervening in the lake food web).

3.8.1. Management Goal 1: reducing contaminant stressors

Catchment nutrient loads

Run-off from the catchment results in significant nutrient loads to the lakes. Catchment models (based on land cover) together with lake input / output models can be used to estimate nutrient loads from the catchment to the lakes. The results of these models can help define the nutrient load reductions required to meet lake nutrient concentration guidelines in the NPS-FM and the ORC Water Plan (Table 13).

Table 12. Estimated mean annual inflow concentrations from catchment models to Upper Tomahawk Lagoon, predicted mean annual in-lake concentrations from input-output models, and estimated nutrient load reductions required to meet the NPS-FM lake national bottom lines for total nitrogen (N) and total phosphorus (P) concentrations.

	Inflow concentrations		**Predicted current in-lake concentrations**		**Estimated reductions in loads to meet the NPS-FM national bottom-line lake concentrations**	
	Total N (mg.m^{-3})	Total P (mg m^{-3})	Total N (mg.m^{-3})	Total P (mg.m^{-3})	Total N (%)	Total P (%)
CLUES V10.6 catchment model	1,195	102	1,348	75	41	34
NIWA NZ River Maps model	947	49	1,069	43	25	-17

The calculated load reductions required differ somewhat depending on which catchment model is used. The CLUES and NZ River Maps models indicate that the N load reductions required are between 41% and 25%, respectively. Using the CLUES catchment model suggests that P loads also need to be reduced by 34%, but using the NZ River Maps catchment model indicates that P loads are currently adequate to meet the lake NPS-FM national bottom line for lake TP concentration. Unfortunately, these estimates cannot be compared or validated with the Ecotago water quality data because total nutrient concentrations were not measured in the lakes. ORC has begun to measure TN and TP in the upper lagoon, but there is not enough data to estimate mean annual concentrations of these attributes at this time.

Some further insights can be gained from the Ecotago data on nutrient and turbidity levels in the inflow creeks. These data show generally elevated levels of turbidity (i.e. suspended sediment), P and N in the creeks, particularly in Lagoon Creek, which drains into the upper lagoon and has nitrate levels that are extremely high for this type of low intensity agricultural and forestry catchment. Given that Lagoon Creek is the only creek feeding the upper lagoon, the high nutrient levels coming out of the creek likely influence the lagoon's eutrophic status.

Reducing the high external nutrient loads should result in improvements to the water quality of the lakes; however, because of legacy nutrients, time lags and hysteresis, the improvements will likely occur gradually over the time scale of years to decades. Actions that can be undertaken to reduce external nutrient loads should focus on mitigating N and P mobilisation to the inflow creeks.

From a planning perspective, the Tomahawk catchment could be viewed as a nutrient-sensitive zone. In recognition of this, regulations and public education regarding land-use practices in the catchment could be employed specifically to reduce nutrient inputs to the land. For example, the discharge of piggery waste in the upper catchment of Lagoon Creek might be reviewed an activity that is not consistent with land uses permitted in a nutrient-sensitive catchment. Similarly, compliance with the National Policy Statement for Plantation Forestry could be mandated with regard to the management of forestry blocks in the catchment.

Pine forestry has been shown to limit water yields. Therefore, forestry blocks in the catchment may reduce the hydrological flows through the catchment and lakes, thereby slowing down the flushing of legacy nutrients from the lakes. While this land use change is likely to reduce nutrient loads, there will be a trade-off between reduced nutrient loading and reducing the water yield and flushing potential. If substantial new forestry blocks are developed in the catchment, some careful consideration of these effects would be warranted.

Other options include fencing off and / or planting of riparian buffer zones along creeks in the catchment and around the lake margins where stock can access the lakes. Appropriate riparian margins can help intercept overland flow, reducing sediment and P mobilisation to waterways during floods. These margins may also help attenuate the N mobilisation to creeks via shallow groundwater, where and when this occurs.

Protecting and reinstating wetlands in the catchment may also help reduce nutrient and sediment mobilisation to creeks and lakes. A land survey could be undertaken to identify potential wet areas in the catchment. If such areas are identified, the council could work with landowners (possibly by assisting with the development of farm environment plans) to facilitate an increase in the extent of wetland areas in the catchment.

In addition, a wetland could be engineered where Lagoon Creek enters the upper lagoon (Figure 46). The lake shallows near the inflow could be converted to wetlands by creating a bund to trap water and sediments and to protect the new wetland from wave action. The area could be planted with appropriate wetland plants (e.g. raupō). Once established, this area could act as a sediment trap, also attenuating N and P as well as facilitating denitrification of the inflowing waters. This type of intervention is likely to have a more immediate impact on the nutrient status and condition of the upper lagoon than mitigations distributed throughout the catchment.

Figure 46. Bund location for potential constructed wetland at the inflow to Upper Tomahawk Lagoon.

Internal legacy nutrient loads

Historical nutrient loads probably continue to contribute to nutrient availability in the lagoons due to internal nutrient loading facilitated by wind-induced sediment resuspension, microbial mineralisation, hypoxia / anoxia at the sediment–water interface, and elevated pH during algal blooms. Thus, by mediating the internal load of nutrients to the water column, these mechanisms result in hysteresis in the eutrophication response, if restoration actions are solely aimed at reducing external nutrient inputs to the lagoons. Nevertheless, over decades, reducing external loads should produce a durable reduction in the trophic state of the lagoons, as the internal pool of nutrients is decreased over time due to flushing, denitrification and deep burial in the lakebed.

A number of options could be explored for reducing the recycling of legacy nutrients and sediments from the lakebed to the water column. Two options are dredging sediments out of the lakes and using P binding and capping agents. From anecdotal information supplied by Otago Fish & Game, an attempt was made decades ago to deepen the upper lagoon by dredging a section of the lakebed. However, it is not known how successful this was in terms of deepening the lake or improving the fishery. The site of dredged sediment has now filled in.

Some disadvantages of dredging include the disruption of the lakebed and organisms living in it, difficulties disposing of dredge spoil in an acceptable and appropriate way, and dredged sediment being replaced over time by sediment resuspension and new sediment inputs. For these reasons, dredging (particularly if not done over the entire lakebed) is unlikely to result in durable improvements to the lakes.

Phosphorus binding and capping agents are chemicals applied to the lake to lock P into the sediments (Hickey and Gibbs 2009; Gibbs and Hickey 2018). Alum and phoslock are examples of materials that have been used to reduce P availability in lakes. However, given that the Tomahawk Lagoons are shallow lakes, often with dense macrophyte beds and frequent wind-induced sediment resuspension (Hamilton and Mitchell 1997), it is unlikely that such agents would provide durable improvements to the water quality of these lakes.

Thus, few appropriate options exist for managing legacy sediments and nutrients in the beds of these lakes, highlighting the importance of reducing sediment and nutrient loads to these systems.

Stormwater and heavy metals

Measurements of the heavy metal concentrations in the lakebed sediments of the lagoons by Cawthron scientists similarly showed that heavy metal contents (lead, copper, cadmium, zinc) are generally elevated in the lagoons compared to average contents in a sub-set of other Aotearoa New Zealand lakes. Lead (lower lagoon) and zinc (upper and lower lagoons) contents exceeded the default guideline lower trigger for sediment quality (ANZECC and ARMCANZ 2000; Figure 46), which represents the concentration below which there is a low probability of biological effects (Batley and Maher 2001). While these heavy metal contents measured in the lagoons are higher than in many other shallow lakes, elevated levels were also found in other lakes that receive urban stormwater.

The Spencer Street stormwater outfall is a consented outfall near the shore of the lower lagoon. The stormwater is derived from the urban area to the west of the lower lagoon. The outfall discharges 25 m from the edge of the lagoon to an area fitted with stormwater baffles, which lower water velocity, and a reno mattress, which helps infiltration of the stormwater into the soil. The area has been vegetated with native plants. Little data are available on the water quality of the stormwater discharging at this site; however, one sampling of the outfall during a minor flood event indicated slightly elevated levels of zinc and copper (Denmead 2021). It is not known to what extent this outfall impacts the concentration of heavy metals in the lower lagoon. Although prior to the installation of the infrastructure at this site, it is likely that stormwater would have had a greater impact on the lagoon.

There are numerous small stormwater outfalls to the upper lagoon, which appear to come from houses, but there is one larger outfall directly entering the upper lagoon at the domain. Denmead (2021) was unable to sample the outfall directly but did sample water from the lagoon in the vicinity of the outfall during the same minor flood event. Analyses did not definitively indicate contamination from the stormwater, although the electrical conductivity of the sample was very high, indicating either high salinity or significant (unidentified) contamination. More data on the contamination levels in these stormwater outfalls are required to assess the potential effects on the lagoons.

At this stage, insufficient information exists to assess the impacts of contaminants from stormwater discharges to the lagoons. Further research is needed before the importance of end-of-pipe treatment or diversion of stormwater discharges can be assessed. However, initiatives to reduce stormwater flows from urban areas are likely to decrease the harmful effects of stormwater entering the lagoons.

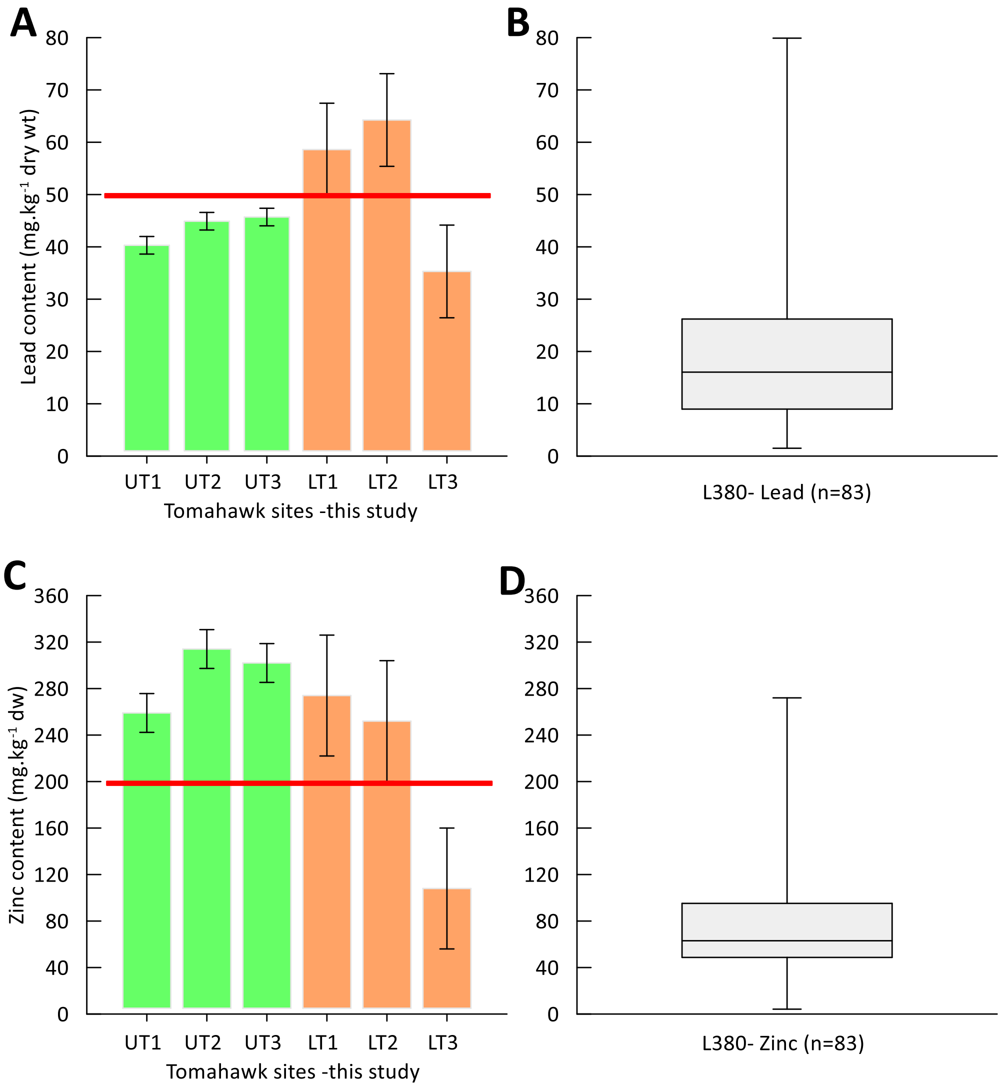

Figure 46. Heavy metal contents in the surface (0–2 cm) sediments of the Tomahawk Lagoons. Plots A and C show the lead and zinc contents, respectively, of samples analysed during this study in the Upper (UT – green bars) and Lower (LT – orange bars) Tomahawk Lagoons. The red lines represent the ANZECC lower trigger values for sediment quality. Plots B and C show summary statistics (range, 25th and 75th percentiles and median) for lead and zinc contents, respectively, of surface sediments from 83 shallow lakes, analysed as part of the Lakes380 national-scale study.

3.8.2. Management Goal 2: lagoon water level management

Active water level management is a current hydrological feature of both lagoons. The water level of the upper lagoon is constrained by a weir, which maintains water levels in the lagoon during dry periods. The water level of the lower lagoon is managed by artificial openings across Tomahawk Beach. Managed lagoon openings are facilitated by sand mining at Tomahawk Beach, whereby a contractor is consented to remove up to 7,100 m^3 of sand per year in the vicinity of the outlet of the lagoon. In recent years, the contractor has been removing < 50% of the consented amount of sand from the beach (Figure 47).

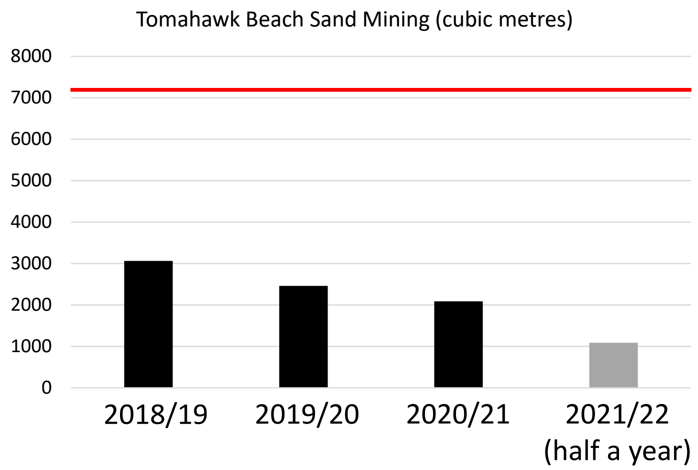

Figure 47. Consented extent of sand removal from Tomahawk Beach. The bars are the actual removal per year. The red line is the consented annual sand take. Source: Otago Regional Council.

The ecological effects of the weir between the lagoons are likely to be beneficial to the ecology of the upper lagoon because significant dewatering of the lagoon can occur in dry, hot summers. Maintaining water level during such times undoubtedly benefits fish, macrophytes and waterfowl. However, the presence of the weir probably elevates flood risk, and floods have occurred at the upper lagoon, damaging buildings near the lake edge. The weir may also inhibit the migration of aquatic organisms between the lagoons.

In contrast, the artificial openings of the lower lagoon are employed to prevent flooding of the lagoons and damage to property such as dwellings and roading infrastructure. The openings are dependent on water levels in the lagoon and have been reported on a quarterly basis since 2020 (Figure 48).

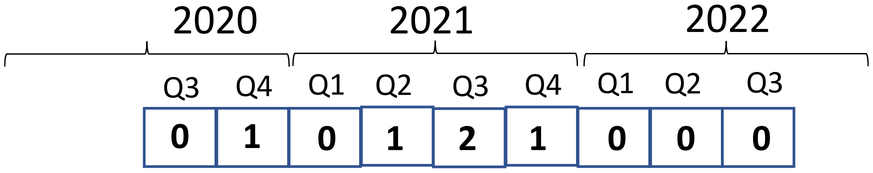

Figure 48. Artificial openings of Lower Tomahawk Lagoon reported by quarter. Source: Otago Regional Council.

The imperative to protect property and infrastructure undoubtedly affects the ecology of the lagoons via higher water levels, greater water level variation, and the reduced frequency of periods of marine influence. The natural opening regime would increase the time-averaged depth and reduce the time-averaged salinity of the Tomahawk system. Coupled with the sand accretion occurring at Tomahawk Beach, the lagoons would naturally become a larger, more freshwater dominated system (less saline influence), being increasingly separated from the sea by a growing sand dune barrier.

Despite the obvious effects of water level management on the 'natural' condition of the lagoons, the proximity of housing and roading to the lagoons necessitates flood protection by maintaining drainage to the sea. Artificial water level management is also carried out on many similar lagoon systems along the east coast of the South Island (e.g. Waituna Lagoon, Kaikorai Lagoon, Wainono Lagoon, Lake Ellesmere / Te Waihora, Lake Forsyth / Wairewa). Thus, the ability to allow a more 'natural' water level and flushing regime in these lagoons is highly constrained by asset management.

It appears the options to alter the opening regime of the Tomahawk Lagoons are limited, even if there is a focus on improving ecological values. However, rising sea levels may increase marine connectivity of this system, potentially raising water levels, although this outcome is also dependent on the management of sand accretion at Tomahawk Beach. Assuming the regional council will continue to mine sand and open the lagoons to maintain the current water level regime, the marine influence will likely increase over time in these lagoons as the sea level rises.

3.8.3. Management Goal 3: swans

Black swans are undoubtedly a key factor influencing the water quality of the Tomahawk Lagoons. With their long necks, swans can forage on macrophytes down to approximately 1 m depth, which enables grazing across the total surface area of both lagoons. Their occasional very high densities on the lagoons means that they have the potential to harvest a large amount of macrophyte biomass. Their faeces can also provide a nutrient stimulus to phytoplankton and add detritus to the lake, which consumes oxygen as it decomposes. If they are mainly feeding on the macrophytes, this increases the nutrient cycling and the transfer back into the water column of some nutrients taken up from the sediments by macrophytes. If the swans also feed on pasture grasses (which they often do), this represents an additional input of nutrients from the catchment into the lake. Thus, swans can be a strong mediator of eutrophication, specifically by facilitating the shift from a state of nutrient enrichment of abundant macrophytes to a state in which phytoplankton biomass is likely to prevail over macrophyte biomass. In the context of the flipping behaviour noted in Upper Tomahawk Lagoon (and possibly also occurring in the lower lagoon), substantial grazing of macrophytes by swans could initiate a flip to an algal-dominated state.

McKinnon and Mitchell (1994) showed that swan densities at Upper Tomahawk Lagoon were positively correlated with macrophyte biomass in the lagoon (R^2 = 0.63), suggesting that high macrophyte biomass attracts swans to the lagoon. Furthermore, Mitchell (1989) calculated that swans could consume 20% to 50% of the annual macrophyte production in the lagoon.

The maximum swan density reported by McKinnon and Mitchell (1994) was approximately 6.5 swans per ha (the lagoon is 10.2 ha in surface area). However, swan counts undertaken by Otago Fish & Game and Ecotago show that swan densities are often much higher than that in the lagoons (e.g. often over 15 swans per ha and up to 47 swans per ha). Thus, the figures produced by Mitchell (1989) on swan grazing are likely to be very conservative estimates in relation to the grazing that can occur in the lagoon at high, often observed, swan densities.

The interactions between swans and water quality described above are complex. A hypothesis of how swans could affect water quality in the Tomahawk Lagoons is presented in Figure 49.

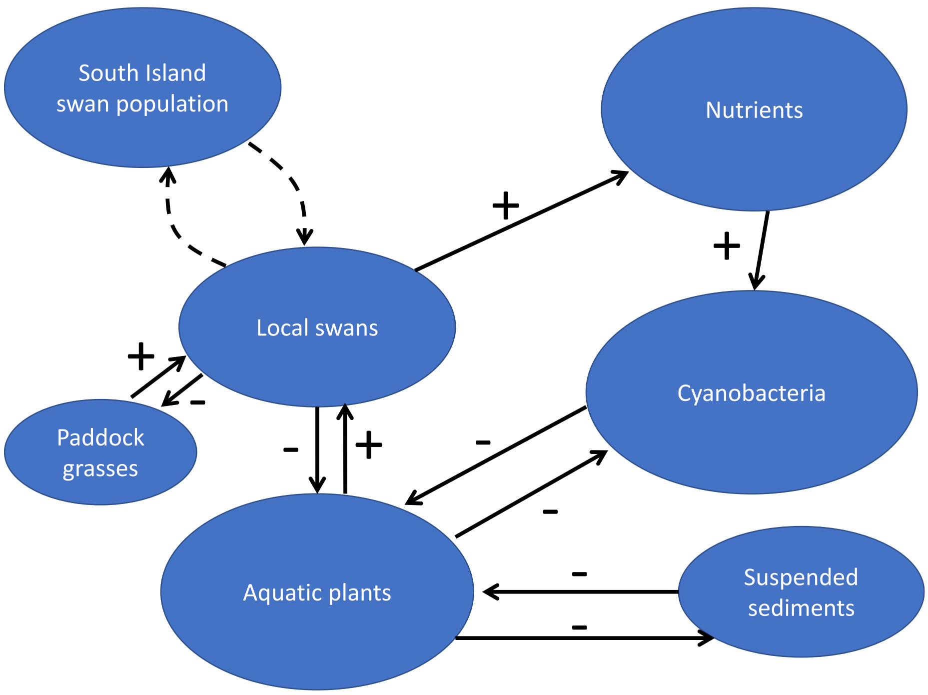

Figure 49. A hypothesis of swan interactions with water quality in the Tomahawk Lagoons. The solid arrows show hypothesised causal relationships. + indicates a positive effect. – indicates a negative effect. The dashed arrows indicate exchange of swans between the local and larger South Island populations.

Paddock grasses and the presence of macrophytes in the lagoons attracts swans to the area, where they forage and set in motion the various interactions illustrated and discussed above. This conceptual model suggests that by controlling swan numbers, the macrophyte biomass and water quality of the lagoons could be improved. However, Otago Fish & Game have undertaken swan culls at Lake Waihola and other locations and have reported that local culls do little to reduce swan populations because of the much larger surrounding swan population that rapidly replaces the culled swans. Thus, swans appear to be a major mediator of water quality that is difficult to control or manage. Furthermore, waterfowl are identified as an important key value of the Tomahawk Lagoons and, therefore, swan culls or swan deterrents may not be feasible or acceptable to the community.

3.8.4. Management Goal 4: invasive species

Daphnia pulicaria

The Tomahawk Lagoons are host to a number of invasive species that have the potential to mediate the symptoms of eutrophication. The invasive zooplankter, *Daphnia pulicaria*, has colonised the upper lagoon and often reaches very high densities. *D. pulicaria* is an effective grazer of algae and anecdotal evidence suggests that high densities of *D. pulicaria* in the lagoon are associated with clear water phases. Thus, this invasive species potentially confers some resilience and resistance to eutrophication due to its ability to graze on algae.

European perch

The Ecotago data indicate that European perch can also reach high densities in Upper Tomahawk Lagoon. Perch have been associated with poor water quality in some Aotearoa New Zealand lakes, where juveniles feed on *Daphnia* (Persson and Hansson 1999) and thereby appear to suppress grazing on algae (Hicks et al. 2013). Perch recruitment is also known to be changeable, showing high year-to-year variations in some lakes (Farmer 2013). Although no work has been done on perch-*Daphnia*-algae interactions in the Tomahawk Lagoons, it is possible that perch may mediate the eutrophication response, such that algal blooms are also influence by perch recruitment success as well as by nutrient availability in the upper lagoon.

The hypothesised interactions between perch, *Daphnia* and algae in shallow lakes suggest that if juvenile perch numbers could be controlled in Upper Tomahawk Lagoon, there may be a reduction in phytoplankton biomass due to resulting increases in *Daphnia* biomass. The small size of the lagoon could make the intensive netting of small perch a feasible management action; however, data are currently insufficient to assess to what extent perch removals could improve the ecological condition of the lagoon. Furthermore, when macrophyte biomass and cover is high, the efficiency of attempts to remove a substantial proportion of the juvenile perch by netting will be greatly reduced.

Another option to reduce perch recruitment could be to temporarily introduce perch spawning substrates to attract perch egg-laying. These substrates could then be removed after the perch spawning season is over and before the eggs hatch. Pilot studies could be used to fine-tune the parameters of such an approach and assess its likely effectiveness at reducing perch recruitment.

Finally, perch are considered a game fish by Fish & Game New Zealand, and they would need to approve any proposed intervention to reduce perch numbers.

Elodea canadensis

The invasive macrophyte *Elodea canadensis* is found in Upper Tomahawk Lagoon. Anecdotal evidence indicates that this species can constitute a large proportion of the macrophyte biomass. While *E. canadensis* may have negative impacts on macrophyte native biodiversity (Kelly and Hawes 2005), its presence in the lagoon may make the lagoon more resilient to flipping. However, this will depend on whether *E. canadensis* enhances biomass and cover or whether it merely replaces native macrophytes in the lake. *Elodea canadensis* is not as invasive as many other problematic invasive species, but its potential to proliferate should also be considered in any project to manage macrophytes in Upper Tomahawk Lagoon. At present there is inadequate information to assess the impact of *E. canadensis* on water quality in the lake.

More information on macrophytes would facilitate the assessment of whether *E. canadensis* is an asset or a threat to the ecological condition of Upper Tomahawk Lagoon.

4. CONCLUSIONS AND RECOMMENDATIONS

Lake Tuakitoto and the Tomahawk Lagoons are highly valued coastal lakes, having significant wildlife, biodiversity, and recreational resources and amenities. However, these values are currently being compromised by degradation in water quality, which significantly impacts the ecological health of the lakes.

Management options for Lake Tuakitoto are summarised in Table 13. The trend of increasing total nitrogen (TN) and total phosphorus (TP) concentrations in the lake is contributing to extensive macroalgae and phytoplankton blooms, and phytoplankton now exceed the National Policy Statement for Freshwater Management (NPS-FM) national bottom lines for the protection of ecosystem health. Undesirable ecological feedback processes appear to have become established in the lake, driven by high primary productivity of macroalgae in summer. These processes appear to result in internal nutrient loads from lake sediments, high water column pH and possibly dissolved oxygen (DO) sags (DO data are insufficient to carry out a comprehensive evaluation). We suspect that poor water quality conditions have flow-on effects to sensitive biota such as native fish (e.g. īnanga, common bullies) and possibly juvenile kākahi, which have declined by 85% in biomass since kākahi surveys were first carried out in the 1990s. It is concerning that the important ecosystem services kākahi provide to the lake by filtering lake water and removing algae are declining, and this will likely continue without management intervention. There is some uncertainty around the necessary nutrient load reductions to improve these conditions in the lake. However, catchment models suggest that to achieve national bottom lines, N load reductions in the range 32–56% are needed (mostly via reducing N in inflows) and P load reductions of around 59% are required (mostly by reducing internal nutrient load sources).

Monitoring data for the Tomahawk Lagoons collected since 2016 indicate that both lagoons and their inflow creeks breach regional and national water quality guidelines. This situation confirms the submissions by members of the local community to the Otago Regional Council (ORC) advocating for the restoration of the lagoons. Both lagoons are subject to occasional algal blooms, often caused by cyanobacterial species that potentially produce toxins harmful to people and animals, especially dogs. Historical data from the 1960s and 1970s show that Upper Tomahawk Lagoon underwent repeated shifts between a macrophyte-dominated state and a phytoplankton-dominated state. Recent turbidity and chlorophyll-*a* data, together with anecdotal evidence, suggest that these regime shifts continue to occur in the lagoons. However, the triggers that shift the lake condition towards algal blooms are not confirmed, but it is likely that black swans, which feed on macrophytes, play a role in driving this unusual dynamic.

The inflow creeks contain high concentrations of nitrate, dissolved reactive phosphorus (P) and suspended sediment (measured as turbidity). In particular, Lagoon Creek, which flows into the upper lagoon, has extremely high nitrate concentrations. The use of empirical catchment and lake models suggests that nitrogen (N) loads will need to be substantially reduced in order for the lagoons to meet regional and national guidelines for TN.

Analyses of sediment P fractions indicate that the bed of the lower lagoon contains substantial amounts of P. This can be mobilised to the water column by sediment resuspension, anoxic conditions, high pH conditions and microbial organic matter mineralisation. The P in the sediment is likely a legacy from historically higher P loads to the lake, suggesting that attempts to improve water quality will encounter some hysteresis and time lags between the reduction of external loads and the improvement of water quality in the lagoons.

The water levels of the lakes are managed to protect low-lying dwellings and infrastructure around the shore of the lagoons. While this contributes to the lagoons departing from their natural condition, it is not feasible to allow water levels to rise naturally due to the constraint of flood management. The presence of a self-sustaining population of perch in the lagoons may also mediate water clarity because of the zooplanktivorous diet of juvenile perch.

4.1. Summary of rehabilitation options – Lake Tuakitoto

We recommend a range of management options aimed at reducing nutrient loads to the lake (the most important action), managing legacy (internal) loads, improving kākahi populations and rectifying fish passage issues.

The greatest priority for managing ecosystem health in Lake Tuakitoto is the reduction of nutrient loads to the lake, which would have flow-on effects for several of the other management priorities (e.g. kākahi, macroalgae). Directing inflows through the existing wetland could harness the existing ecosystem services of the wetland for attenuating sediment and nutrients before they flow into the lake. The inclusion of flows from Stony Creek, which has high loads of nutrients, into the diversion to the wetland could also be beneficial. If this were undertaken, then flood mitigation around farmland bordering the lake is required.

Alternatively, the diversion race to the lake could be realigned to discharge to the outlet canal, thereby bypassing the lake. Catchment stream care initiatives focused on improving riparian areas and developing farm plans in agriculturally dominated catchments are also needed to reduce loads to inflow streams.

The control of internal loads should be investigated to try to minimise the negative effects of macroalgae on the lake, particularly in summer. This would involve testing sediment-capping agents that bind P in lakebed sediments. Mechanical harvesting of macroalgae is another option, possibly initially focusing on the floating rafts of decaying macroalgae that occur over the lake. Management of macroalgae is a complex and emerging problem (Vadeboncoeur et al. 2021). The design of these interventions would therefore be more experimental and should first be trialled at small scales to better understand their effectiveness.

The age structure of the kākahi population suggests that there has been limited recruitment (breeding success) in Lake Tuakitoto for considerable periods (possibly decades). We suggest that limited native fish hosts for kākahi glochidia larval stages is likely to be an important contributor to poor recruitment. While water quality improvements could rectify this by improving fish habitat, European perch in the lake could have a negative effect on common bully populations, which are potential hosts for the parasitic larvae of kākahi. Management of perch abundances could be another way of aiding the recovery of kākahi by improving native fish populations and biodiversity values.

Fish passage was previously identified as an issue for Lake Tuakitoto and is related to the hydrological control structures at the lake outlet sill and the Kaitangata locks. Native fish species that possess poorer climbing and swimming abilities are currently observed at unusually low abundances in the lake. The extent to which these structures are, or could be, operated to facilitate better fish passage is uncertain and should be investigated. Furthermore, a fish passage management plan should be developed.

Table 13. Catchment and in-lake management options for improving ecological health of Lake Tuakitoto.

Goal	Management action	Priority	Timing	Uncertainty
Reducing stream nutrient loads in inflows	Divert more flows from Lovells / Frasers Creek to drain to wetland to increase nutrient removal before entering Lake Tuakitoto	Highest	Immediate	Low – monitor flow rates to evaluate nutrient removal and hydrology
	Evaluate adding Stony Creek inflow to wetland diversion or realignment of diversion race bypassing the lake to the outlet channel	Medium	Long term	Intermediate – hydrological uncertainties regarding diversions
	Riparian enhancement and farm plans in Upper Lovells and Stony Creeks to reduce catchment nutrient losses from agricultural land and forestry	High	Immediate	Low – good stream water quality data to suggest problem catchment areas
Controlling internal phosphorus loading	Apply phosphorus binding / capping agents (e.g. alum, Phoslock) to the lakebed to reduce phosphorus recycling	Trials needed	Immediate	Uncertainty on mechanism and spatial extent of P-recycling in the lake. Effects on kākahi
	Mechanical harvesting of macroalgal mats to reduce dissolved oxygen and pH variation that drive internal loads	Trials needed	1–3 years	High – mats could regrow rapidly. Understanding of accrual and seasonal dynamics needed
Improving kākahi recruitment	Pest fish control to reduce perch populations and spawning and enhance native fish that could serve as hosts for kākahi larvae	Medium	1 year	Medium – need to scope flow rates to evaluate if wetland residence time is long enough to be effective for nutrient removal
Fish passage	Monitoring of fish passage at outlet channel sill and Kaitangata locks to assess passage and operational requirements	High	Immediate	Low – can assess passage requirements with acoustic monitoring and design operational requirements or modify structures to allow passage

4.2. Summary of rehabilitation options – Tomahawk Lagoon

Restoration options for the Tomahawk Lagoons are somewhat constrained due to the encroachment of urban areas and the fact that the lagoons are managed as a wildlife reserve. Nevertheless, several restoration actions have been identified and assessed within the broad categories of reducing contaminant loads, and managing water levels, black swans and invasive species (Table 14).

High nutrient and sediment concentrations in the inflow creeks highlight the need for reductions in contaminant flows from land to waterways. The catchment of the Tomahawk Lagoons can be considered a nutrient-sensitive zone. Therefore, land-use practices in the catchment should strive to minimise nutrient and sediment losses from land to water. This could be achieved by fencing and planting riparian buffer zones along waterways, by ensuring forestry blocks provide an adequate buffer zone along waterbodies, and by encouraging wetland protection and enhancement in the catchment. In addition, we suggest that the shallow lake area near the inflow from Lagoon Creek could be engineered to be a wetland, where water could be held inside a bund in an area vegetated with wetland plants before being allowed to enter the lagoon. This modification would remove sediment and P from inflowing waters, and allow P and N uptake by plants and denitrification by microbes.

The internal loads of nutrients from the lakebeds to the water column are more difficult to control, as dredging and P capping / binding are likely to be ineffective in these lakes.

Evidence suggests that stormwater inputs to the lagoons may be associated with elevated heavy metal concentrations in the lakebed. More information on contaminant levels in stormwater outfalls is needed before we would recommend large investment in stormwater treatment. However, reducing catchment run-off from urban areas should be investigated.

Although artificial openings of the sand barrier between the lower lagoon and Tomahawk Beach alter the hydrology of the lagoons, the need for flood management to protect houses and roading infrastructure is a priority. Altering the water level regime of the lagoons to favour their ecology is currently not a feasible option.

Black swans may mediate the shift between a macrophyte-dominated clear water state and a turbid, phytoplankton-dominated state through their ability to graze macrophytes across the entire bed of both lakes. However, when swan culling has been undertaken in other areas by Fish & Game New Zealand, recolonisation by swans from the larger South Island population has been rapid. Furthermore, obtaining approval for swan culling would likely be difficult given that the Tomahawk Lagoons are a wildlife refuge and the shore of the upper lagoon is close to urban areas.

The presence of European perch in the lagoons may impact water quality due to the zooplanktivory of juvenile perch, releasing algae from the grazing pressure of *Daphnia*. Thus, methods for reducing perch recruitment in the lagoons could benefit water quality, but more research is needed to confirm the efficacy of this intervention.

Table 14. Summary of management actions considered for the improvement of the ecological health of the Tomahawk Lagoons.

Goal	Management action	Priority	Timing	Uncertainty
Reducing contaminants in the lagoons	Explore the possibility of zoning the catchment as a nutrient-sensitive zone in the Otago Regional Council Land / Water Plan	Highest	Immediate	Appropriate planning tools may not be available
	Undertake fencing and riparian planting; buffers in forestry areas	Highest	Long term	Getting buy-in from landowners
	Enhance existing wetlands in the catchment	High	5 to 10 years	Educating and getting buy-in from landowners
	Construct a wetland at the outlet of Lagoon Creek	Medium	Immediate	Getting buy-in from DOC, Fish & Game and the community
	Dredging to remove legacy phosphorus in lakebed sediment	Not recommended		Disruptive to lakebed plants and animals; likely to fill in again
	Use phosphorus capping agent such as Phoslock, Alum	Not recommended		Likely to be of low cost-effectiveness due to shallowness, macrophyte beds
	Stormwater treatment	Not recommended		More data on stormwater quality and impacts needed
Change water level regime	Alter management of the weir between the lakes to be more ecologically beneficial	Not recommended		Raising of water levels could enhance flood risk, endangering dwellings and infrastructure
	Alter management of lower lagoon openings	Not recommended		Raising of water levels could enhance flood risk, endangering dwellings and infrastructure
Black swan management	Culling black swans	Not recommended		Would negatively impact wildlife values; not likely to be accepted by locals
Invasive species management	Reduction of juvenile perch numbers	Not recommended		Uncertainty as to the strength of control that juvenile perch exert on *Daphnia*
	Control of *Elodea canadensis*	Not recommended		Macrophyte dynamics uncertain for *E. canadensis*

4.3. Further monitoring to assist decisions on options

A number of recommendations that are made in the report would benefit from increased monitoring data to aid in decision-making around the design and implementation of management interventions.

The following monitoring recommendations are made for Lake Tuakitoto catchment:

1. inflow monitoring of nutrient loads by major tributaries (lower tributary sites only) to update information on current nutrient inflows to the lake and validate load-reduction targets (minimum 1 year)
2. monitoring of water column nutrients and physico-chemical conditions (DO, pH) within the wetland flow path to the lake to better understand potential nutrient removal by the wetland (1 year, monthly – ideally using continuous nitrate sensors)
3. monitoring of seasonal dynamics of macroalgal cover, species composition, biomass and tests for saxitoxin
4. monitoring of continuous mid-lake pH and DO (monitoring buoy surface and bottom depths) to understand effects on sensitive species and timing of potential internal nutrient loading, as well as profiling through algal mats with microelectrodes if possible
5. kākahi recruitment study to improve understanding of the drivers of recruitment success / failure, including water quality, habitat destruction (macroalgae smothering) and glochidia host availability
6. native fish monitoring over a 3-year period to understand population dynamics, and to assess kākahi host availability in the lake and fish passage around the Kaitangata lock and outlet weir structures (using underwater acoustic video)
7. waterbird monitoring and, possibly, experiments to better understand the importance of swan grazing as a factor in controlling macrophyte establishment (e.g. waterbird exclusion plots, macroalgal exclusion), including undertaking eDNA analysis of faecal pellets to determine if macroalgae are consumed.

The following monitoring recommendations are made for the Tomahawk Lagoon catchment:

1. undertaking monthly monitoring of lower lagoon water quality
2. installing DO sensors near the lakebed in both lakes
3. continuing monitoring of macrophytes in both lagoons
4. monitoring heavy metals and nutrients in stormwater inflows.

In conclusion, we hope this report is useful in updating ORC regarding the ecological condition and stressors of Lake Tuakitoto and the Tomahawk Lagoons. We would be happy to provide further discussion on these recommendations and work with ORC to implement management actions.

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7. APPENDICES

Appendix 1. Sites for field surveys March 2023

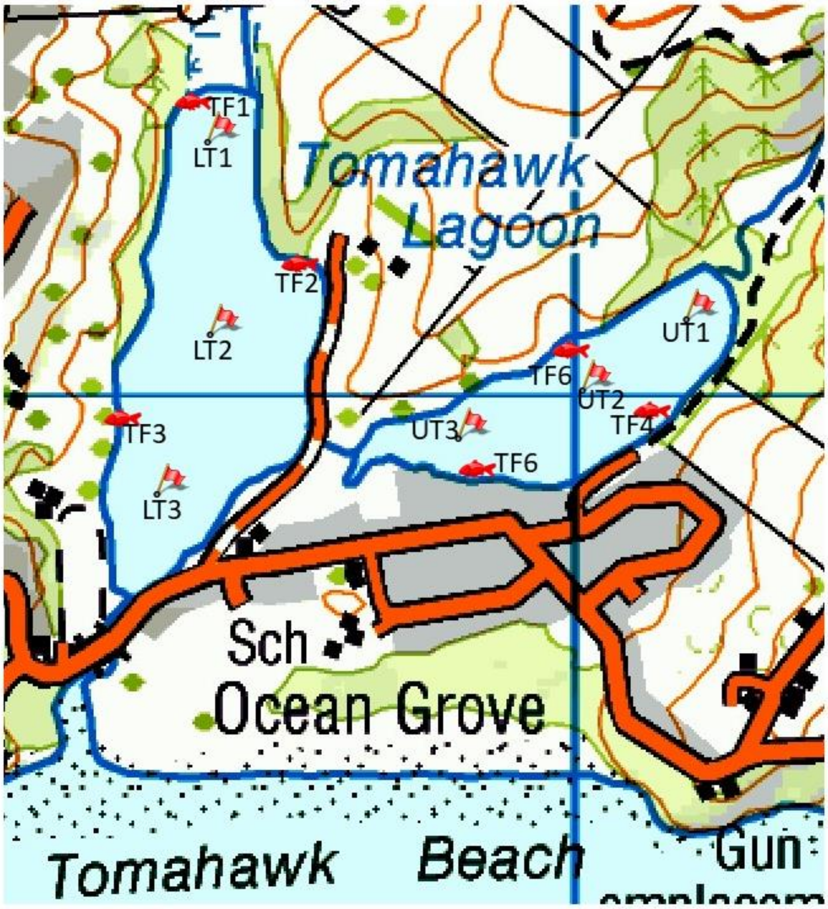

Figure A1. Sites sampled as part of the 2023 field surveys on 23–25 March 2023. Site coordinates and samples collected are shown in Table A1.

Table A1. Site coordinates and samples collected at Tomahawk Lagoon 23–25 March 2023. Coordinates are in NZTM.

Lake	Sample	Site	Easting	Northing
Lower Tomahawk	Sediment	LT1	1409395	4914100
Lower Tomahawk	Sediment	LT2	1409395	4914100
Lower Tomahawk	Sediment	LT3	1409307	4913834
Upper Tomahawk	Sediment	UT1	1410191	4914126
Upper Tomahawk	Sediment	UT2	1410005	4914012
Upper Tomahawk	Sediment	UT3	1409810	4913927
Lower Tomahawk	Fish	TF1	1409362	4914490
Lower Tomahawk	Fish	TF2	1409243	4913961
Lower Tomahawk	Fish	TF3	1409539	4914219
Upper Tomahawk	Fish	TF4	1355371	4875849
Upper Tomahawk	Fish	TF5	1355582	4875370
Upper Tomahawk	Fish	TF6	1355259	4874861

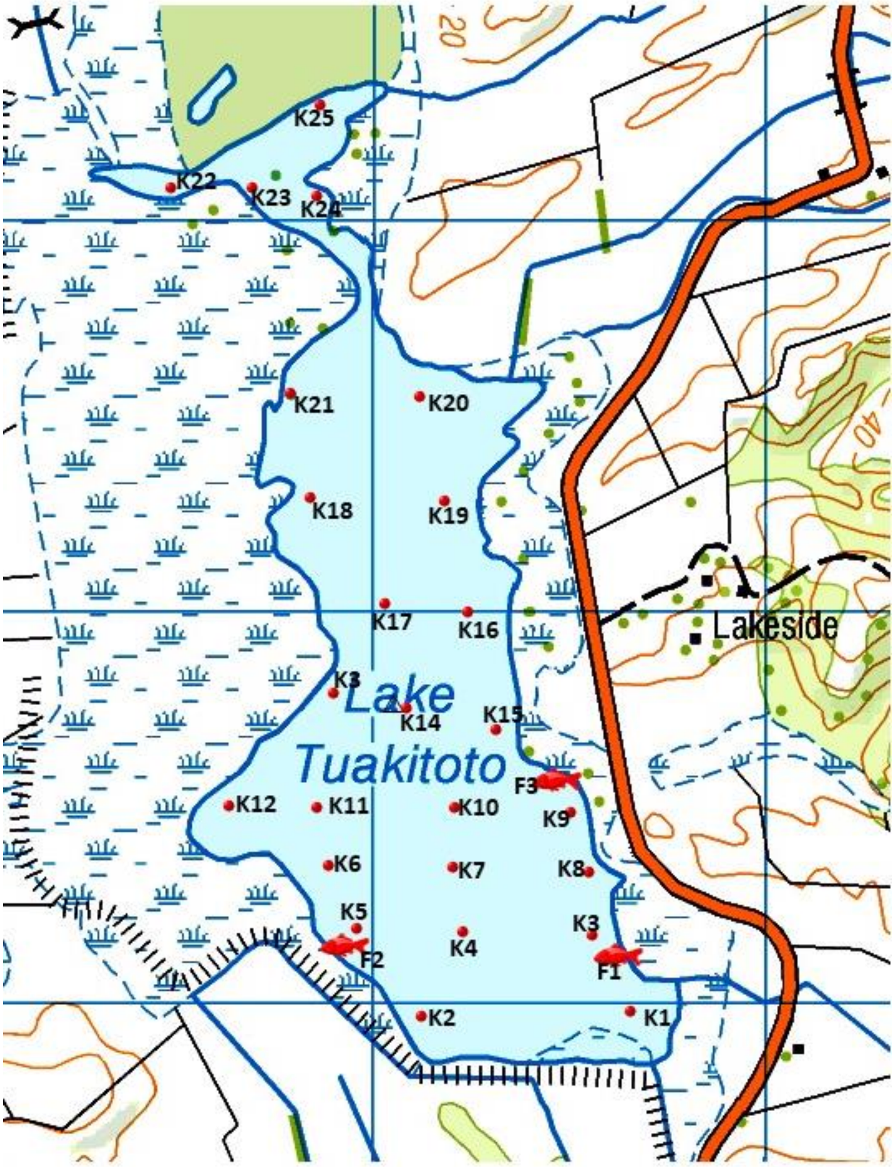

Figure A2. Sites sampled on Lake Tuakitoto as part of the 2023 field surveys on 25–28 March 2023. Sites coordinates and samples collected are shown in Table A2.

Table A2. Site coordinates and samples collected at Lake Tuakitoto 25–28 March 2023. Coordinates are in NZTM.

Lake	Sample	Site	Easting	Northing
Tuakitoto	Sediment	K2	1355659	4874978
Tuakitoto	Sediment	K4	1355232	4875181
Tuakitoto	Sediment	K7	1355207	4875346
Tuakitoto	Sediment	K11	1354860	4875498
Tuakitoto	Sediment	K14	1355088	4875752
Tuakitoto	Sediment	K17	1355016	4875998
Tuakitoto	Core	K14	1355088	4875752
Tuakitoto	Fish, macrophyte	F1	1355371	4875849
Tuakitoto	Fish, macrophyte	F2	1355582	4875370
Tuakitoto	Fish, macrophyte	F3	1355259	4874861
Tuakitoto	Kākahi	K1	1355127	4874965
Tuakitoto	Kākahi	K2	1355659	4874978
Tuakitoto	Kākahi	K3	1355562	4875172
Tuakitoto	Kākahi	K4	1355232	4875181
Tuakitoto	Kākahi	K5	1354962	4875189
Tuakitoto	Kākahi	K6	1354890	4875350
Tuakitoto	Kākahi	K7	1355207	4875346
Tuakitoto	Kākahi	K8	1355554	4875333
Tuakitoto	Kākahi	K9	1355507	4875486
Tuakitoto	Kākahi	K10	1355211	4875498
Tuakitoto	Kākahi	K11	1354860	4875498
Tuakitoto	Kākahi	K12	1354636	4875503
Tuakitoto	Kākahi	K13	1354902	4875790
Tuakitoto	Kākahi	K14	1355088	4875752
Tuakitoto	Kākahi	K15	1355317	4875697
Tuakitoto	Kākahi	K16	1355245	4875998
Tuakitoto	Kākahi	K17	1355016	4875998
Tuakitoto	Kākahi	K18	1354843	4876290
Tuakitoto	Kākahi	K19	1355185	4876281
Tuakitoto	Kākahi	K20	1355160	4876510
Tuakitoto	Kākahi	K21	1354792	4876557
Tuakitoto	Kākahi	K22	1354487	4877081
Tuakitoto	Kākahi	K23	1354694	4877081
Tuakitoto	Kākahi	K24	1354859	4877060
Tuakitoto	Kākahi	K25	1354868	4877293

Appendix 2. Lakes380 research programme sediment core analyses methods

Core sampling and sub-sampling

A sediment core was retrieved from the deepest point of Lake Tuakitoto on 14 June 2020 using a Uwitec gravity corer. After retrieval, the core was sealed and stored at 4 °C and in darkness until sub-sampling. The core was split, and sub-samples taken from the centre of the half-core using a sterile spatula at various depths for DNA (stored frozen) and pollen and charcoal analysis.

Pollen analysis

Pollen species were identified using microscopy as described in Short et al. (2022). Pine (*Pinus* spp.) and other non-native taxa (e.g. *Macrocarpa* spp.) were introduced by Europeans and are used to mark European activity in the region in this report.

Hyperspectral and Itrax scanning

The core was scanned using a hyperspectral scanner at GNS Science, with the spectral data from the RABD660-670 index used as a proxy for chlorophyll-*a* and its degradation products. Elemental abundance data were obtained using a Itrax µ-XRF Core Scanner at the University of Otago. The ratio of manganese (Mn) to Iron (Fe) was selected for plotting as a proxy for bottom-water oxygenation in the lake.

Environmental DNA analysis

DNA was extracted from the sediment and a region of the bacterial 16 S rRNA gene was amplified and analysed as described in Pearman et al. (2020). The cyanobacteria component of the data was extracted for plotting. The functional profiles of the bacterial community were inferred using the software paprica (Bowman and Ducklow 2015). Enzymes related to denitrification, dissimilatory nitrate reduction to ammonia and sulphate reduction were selected for plotting.