Lake Hayes
State of Environment 2023

Prepared for the Friends of Lake Hayes Society Inc. by

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[This report was peer reviewed by David Kelly (PhD)]

Executive summary

Lake Hayes is possibly the most researched lake in New Zealand, with scientific measurements going back to the late 1940s. Problems with eutrophication of the lake seem to have begun in the 1960s when major drainage works were undertaken to convert wetlands in the mid-catchment to pasture. In recent years, the Otago Regional Council improvedwater quality monitoring of the lake and its main tributary, Mill Creek. This includes a continuous record of monthly water quality sampling since late 2016 and the deployment of a profiling lake monitoring buoy in 2019. The quality and quantity of information that has accumulated as a result of these monitoring programmes allows for a significant update to the assessment of water quality and ecological condition of the lake and its tributary.

Mill Creek exhibits high, and increasing, nitrate concentrations, breaching the regional water plan standard. The phosphorus phosphorus load from Mill Creek to the lake is partly dependent on the occurrence of floods. The phosphorus load to the lake in 2020 was estimated at 2.3 tonnes, which exceeded the load of 2.1 tonnes estimated in 1983/84. However, in 2021 and 2022, the P load decreased substantially. *E. coli* counts in Mill Creek often exceed regional standards and threaten recreation at the northern end of the lake, which is popular for bathing.

Water clarity in Lake Hayes has declined markedly since 2006, apart from two summers (2010/11 and 2017/18) when water clarity was remarkably clear. The reason for high water clarity in occasional summers is unknown, but research on the pelagic food web of the lake suggests that this could be related to a trophic cascade triggered by reduced perch recruitment in those years.

In the summer of 2022/23, Lake Hayes exhibited high temperatures, high pH, high chlorophyll *a* and a long duration of seasonal thermal stratification. The lake began its thermal stratification unusually early, before dissolved oxygen in the lake water had become fully equilibrated with the atmosphere. Therefore, the lake was already somewhat oxygen depleted when thermal stratification began in the spring of 2022. This led to particularly intense deoxygenation of the hypolimnion over the summer. Anoxic hypolimnia have been a feature of Lake Hayes since at least the 1970s and these summer dead zones occupy more than half the lake volume, potentially precluding fish from accessing deeper, cooler waters to escape the summer heat that affects the surface waters. This may be associated with occasional fish kills reported since the mid-2000s.

Intense anoxia results in the release of phosphorus that is bound to lake bed sediments, which is thendiffused into the water column. Estimates of this internal phosphorus load suggest that it is a substantial contributor to the lake's annual phosphorus budget and that it has not declined in magnitude between 1983/84 and 2012/13. As a result, the trophic state of the lake has not improved much over the past decades and appeared to increase from meso-eutrophic to eutrophic-supertrophic between 2017 and 2021.

In summer, high pH and high temperatures in the surface waters together with anoxia in the bottom waters constitutes a situation of significant potential physiological stress for brown trout. Furthermore, this situation is beneficial for cyanobacterial blooms and blooms of potentially toxic cyanobacteria have caused the Otago Regional Council to announce warnings about recreation on the lake in three of the past six years. Occasionally high *E. coli* counts and cyanobacterial blooms both impact on recreational use of the lake.

A survey of the macrophyte community in 2020 found that the community was in a moderate condition, similar to the condition found in a 1992 study.

A phosphorus budget for the lake in 2020 indicated that the retention efficiency of phosphorus in the lake was slightly higher than in 1983/84, possibly reflecting a small decrease in the internal phosphorus load and/or the longer water residence time of the lake in 2020.

The nitrogen to phosphorus ratio of the lake is substantially lower than that of the main inflow, Mill

Creek. This reflects the substantial internal phosphorus load that the lake receives annually and suggests that denitrification may also occur in the lake.

While a substantial knowledge base exists about the lake in relation to its degraded state, we identify some knowledge gaps that, if filled, could help our understanding of the lake.

1. **Rates of dissolved oxygen decline in the hypolimnion:** An analysis of the dynamics of dissolved oxygen in the hypolimnion during summer and the development of a hypolimnetic dissolved oxygen budget could help elucidate how the supply of oxygen to the hypolimnion could reduce internal P loads.

2. **Contribution of willows to hypolimnetic dissolved oxygen, nitrogen and phosphorus budgets:** The organic matter in willow leaves contributes to deoxygenation while the phosphorus and nitrogen contained in the leaves may become bioavailable after mineralisation by bacteria and fungi in the lake bed. No investigations to date have been carried out on the potential contribution of lakeshore willows to the hypolimnetic oxygen, phosphorus, and nitrogen budgets of the hypolimnion.

3. **Nitrogen and phosphorus limitation of algal growth:** It could be useful to investigate the relationship between algal blooms in the lake and the availability of nitrogen and phosphorus prior to, and during the blooms. Furthermore, an investigation into the dynamics of dissolved reactive phosphorus in Mill Creek could also shed some light on the current, relative importance of nitrogen and phosphorus to algal productivity in the lake.

4. **Weather effects:** Investigations into how spring weather may affect drivers of lake water quality could help our understanding of the variability in water clarity from year-to-year.

5. **Quantifying uncertainty in nutrient load estimates:** Investigations into the main sources of potential errors and bias in load estimates and nutrient budgets would allow confidence intervals to be placed around the estimates.

6. **Understanding the lake foodweb using eDNA:** As more and more eDNA samples are analysed for Lake Hayes, this information should be collated into a database which could be used to track species of interest in Lake Hayes.

In addition to revealing the knowledge gaps above, the analyses carried out in this report suggest three specific recommendations to help improve the monitoring of Lake Hayes.

1. **Nutrient profiles to help calculate internal nutrient loads:** Profiles should include at least four samples in the hypolimnion and should be repeated at least monthly between December and May.

2. **More samples in Mill Creek at high flows:** It would be helpful to carry out some event sampling to help understand the drivers of nutrient and sediment loads to the lake.

3. **Key indicators of degradation and recovery:** In addition to the statutory water quality, ecolsystem health, and public health attributes which must be monitored and reported, the analyses in this report reveal additional useful indicators of the condition of Lake Hayes. To faciliate future lake assessments, we recommend that the following indicators be measured at regular intervals and reported on:

    - The internal phosphorus and nitrogen load (requiring monthly summer and autumn nutrient profiles)
    - The nutrient concentrations in the lake during the isothermal period

- The duration of the stratified period
- The depth limit of the euphotic zone (i.e., the depth of photosynthetic activity and dissolved oxygen production)
- The concentrations of *Daphnia* in the water column (as an indicator of grazing pressure)

Contents

1 Background and scope

Lake Hayes is one of the most studied lakes in New Zealand. The earliest scientific water quality data we found is from the late 1940s. Detailed scientific investigations on the lake began in the 1970s, when the lake's eutrophic status was first reported (Cook 1973). Since then, numerous studies have elucidated ecological processes, water quality, the aquatic food web as well as management and restoration options. Complex ecological models have also been developed. Despite this, recovery from a eutrophic state has been elusive, and the lake and its main inflow still fail some regional and national water quality standards.

The Otago Regional Council (and its predecessor, the Otago Catchment Board) have monitored water quality in the lake for decades, but it was only in 2016 that the Otago Reginal Council (ORC) began a monitoring plan in which the lake has been monitored at monthly intervals with few gaps in the sampling regime. The ORC publishes occasional reports on the water quality of the lake and Mill Creek focused on how the water bodies perform specifically in relation to regional and national water quality standards. Occasionally the ORC have also commissioned assessments of the macrophyte community of the lake.

In 2017, The Friends of Lake Hayes (FOLH) commissioned a report from Hydrosphere Research Ltd., which synthesised information about how the lake functions and provided recommendations for improving monitoring and restoring the water quality of the lake. As a result of this work, the FOLH together with the ORC improved sampling of the Mill Creek inflow.

In July 2019, the ORC deployed a depth-profiling water quality monitoring buoy which provides high resolution data both in terms of depth and time intervals sampled. This information can provide important new insights regarding how the lake functions and how water quality responds to various drivers.

In light of recent improvements to the monitoring of the lake and Mill Creek, the FOLH have commissioned this report which provides an updated analysis of the water quality and ecological condition of Lake Hayes. The ORC undertakes periodic assessments of lake and river water quality and ecological condition as part of its State of the Environment (SOE) monitoring programme. The main aims of the OCRs SOE reports are:

- to provide a better understanding of the current condition and trends,
- to provide a clear baseline of water quality attributes to help quantify future changes in response to restoration actions such as replacement of the culvert at the outlet of the lake, the augmentation of inflows using surplus irrigation water, the implementation of catchment mitigations described in the Vision Lake Hayes, etc.,
- to calculate nutrient and sediment loads to the lake,
- to update water balance and nutrient and sediment budgets for the lake, and
- to update recommendations for lake and catchment monitoring and management.

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2 Planning framework

2.1 NPSFM/NOF (2020)

In 2020, the New Zealand Minister for the Environment, David Parker, introduced a comprehensive set of freshwater management objectives and standards in the National Policy Statement for Freshwater Management (NPSFM; Ministry for the Environment 2020). The appendices to the NPSFM are called the National Objectives Framework (NOF) and these provide national standards for water quality, ecosystem health, and human recreational safety attributes which will have statutory standing once they have been implemented by regional councils into regional land and water plans. The NOF numerical attribute states are standards that must be met. If they aren't met, then regional councils are required to place limits on resource use or to implement action plans to ensure the standards are achieved. In the NOF, each attribute is graded from A ("excellent") to D ("unacceptable"). The NOF lake attributes assessed in this report are presented in Appendix A and include:

- Chlorophyll *a* (phytoplankton biomass)
- Total nitrogen
- Total phosphorus
- Cyanobacteria
- *E. coli*
- Submerged plants (native and invasive condition)
- Lake-bottom dissolved oxygen

In this report, we present monitoring data on Lake Hayes and Mill Creek and, where appropriate, we indicate the relevant national bottom-line numeric attribute state, for comparison. However, the bottom line may not be the appropriate state and regional councils and communities are empowered by the NPSFM to set numeric attribute states that stricter that the national bottom lines, if desired.

The Regional Land and Water Plan is currently being drafted by the ORC and so the NPSFM/NOF numeric attributes states are not yet statutory. However, a recent ORC SOE report included assessments for Lake Hayes and Mill Creek in relation to the NOF standards (ORC 2021).

2.2 ORC Water Plan (2014)

The ORC developed a water plan in 2014 (ORC 2014). The plan included Schedule 15, in which water quality standards were set for lakes and rivers, including Mill Creek and Lake Hayes (Table 1). The standards refer to a threshold met by 80% of samples collected at a site over a five-year period. For river sites, data included in the calculation of river condition should only include samples taken when flows are at, or below, the median river flow. Hence, results from high flow events are disregarded when assessing compliance with the river water standards in Schedule 15.

Table 1. ORC Water Plan Schedule 15 limits for Mill Creek and Lake Hayes.

	Total Nitrogen (g/m^3)	Nitrate-nitrite nitrogen (g/m^3)	Total Phosphorus (g/m^3)	Dissolved reactive phosphorus (g/m^3)	Ammoniacal nitrogen (g/m^3)	*E. coli* (cfu/100 ml)	Turbidity (NTU)
Mill Creek		0.75	-	0.1	0.1	260	5
Lake Hayes	0.55	-	0.033	-	0.1	126	5

2.3 District Plan (2018)

The Queenstown Lakes District Council (QLDC) Proposed District Plan (QLDC 2023) mostly regulates development in the Queenstown Lake District. It sets policies, rules, and expectations regarding runoff and stormwater from sites of earthworks and other land development. In response to a submission from the FOLH, the proposed district plan includes Policy 24.2.4.2, which prohibits developments in the Lake

Hayes catchment that could diminish the water quality of Lake Hayes. This would likely include any developments that contribute contaminated runoff or stormwater to the streams and creeks flowing into Lake Hayes. This policy also likely applies to developments that could raise the nitrogen concentration of groundwater and of springs entering the creeks flowing to the lake or springs flowing into the lake itself (such as the springs at Rutherford Rd.).

2.4 Lake Hayes Strategy (1995 to 2003)

In 1995, The Lake Hayes Strategy was implemented jointly by the ORC and the QLDC, which aimed to improve the water quality of Lake Hayes. The Strategy stated that "the conservation of the Lake Hayes resource is of regional and national importance both economically, recreationally and for its intrinsic and scenic values" (ORC 1995). Among a series of goals listed was a 20% reduction in the phosphorus load to Lake Hayes.

The Lake Hayes strategy was rescinded by the councils in 2003, without providing a rationale. The water quality of the lake subsequently deteriorated.

2.5 Other notable, non-statutory management guidance

Since 1973, numerous management and restoration actions and plans have been proposed to improve the water quality of Lake Hayes (e.g., Cook 1973; Roberston 1988; Bayer et al. 2008; Bayer & Schallenberg 2009; Schallenberg & Schallenberg 2017; Gibbs 2018; Goldsmith & Hanan 2019). Many different restoration actions have been analysed and discussed in these reports, including in-lake remediations and catchment mitigations.

In 2021, the FOLH and e3 Scientific developed a catchment management plan to reduce contaminant loads to Lake Hayes, called the Vision Lake Hayes (FOLH 2021). This plan aimed to map contaminant sources in the catchment and set out areas for mitigations including the enhancement of existing wetlands, the development of extensive riparian vegetation corridors along creek banks, the installation and maintenance of sediment traps, and the construction of a wetland just back from the north shore of the lake. Some of the catchment mitigation work outlined in Vision Lake Hayes is currently being implemented.

The Otago Land and Water Plan is currently being drafted by the Otago Regional Council as is a strategy for the restoration of Lake Hayes. It is likely that a lake water quality target of mesotrophic will be set for the lake.

3 Data sources

This report collates data from a wide range of sources including theses, technical reports, and the peer-reviewed scientific literature. We collated and analysed:

- ORC monitoring data from Lake Hayes including lake levels, monthly water quality monitoring data, data from the lake monitoring buoy, and continuous flows, turbidity and recreation surveillance data;
- ORC monitoring data from Mill Creek including monthly water quality monitoring data, continuous flow, turbidity and nitrate data;
- FOLH data on Mill Creek water quality;
- NIWA data on submerged macrophytes;
- New Zealand Freshwater Fish database (NZFFD);
- Data from Land Air Water Aotearoa (LAWA);
- GIS data;
- Published peer-reviewed studies;
- Unpublished University of Otago data.

The Otago Regional Council carries out monthly State of the Environment (SOE) water quality testing of Mill Creek which flows into the northern end of Lake Hayes, and of Hayes Creek (the lake outflow) at SH6 (**Figure 1**). Monthly water quality sampling is carried out in Lake Hayes at the mid-lake site (**Figure 1**). In addition, the ORC has a water monitoring buoy which records depth profiles of numerous water quality attributes twice daily.

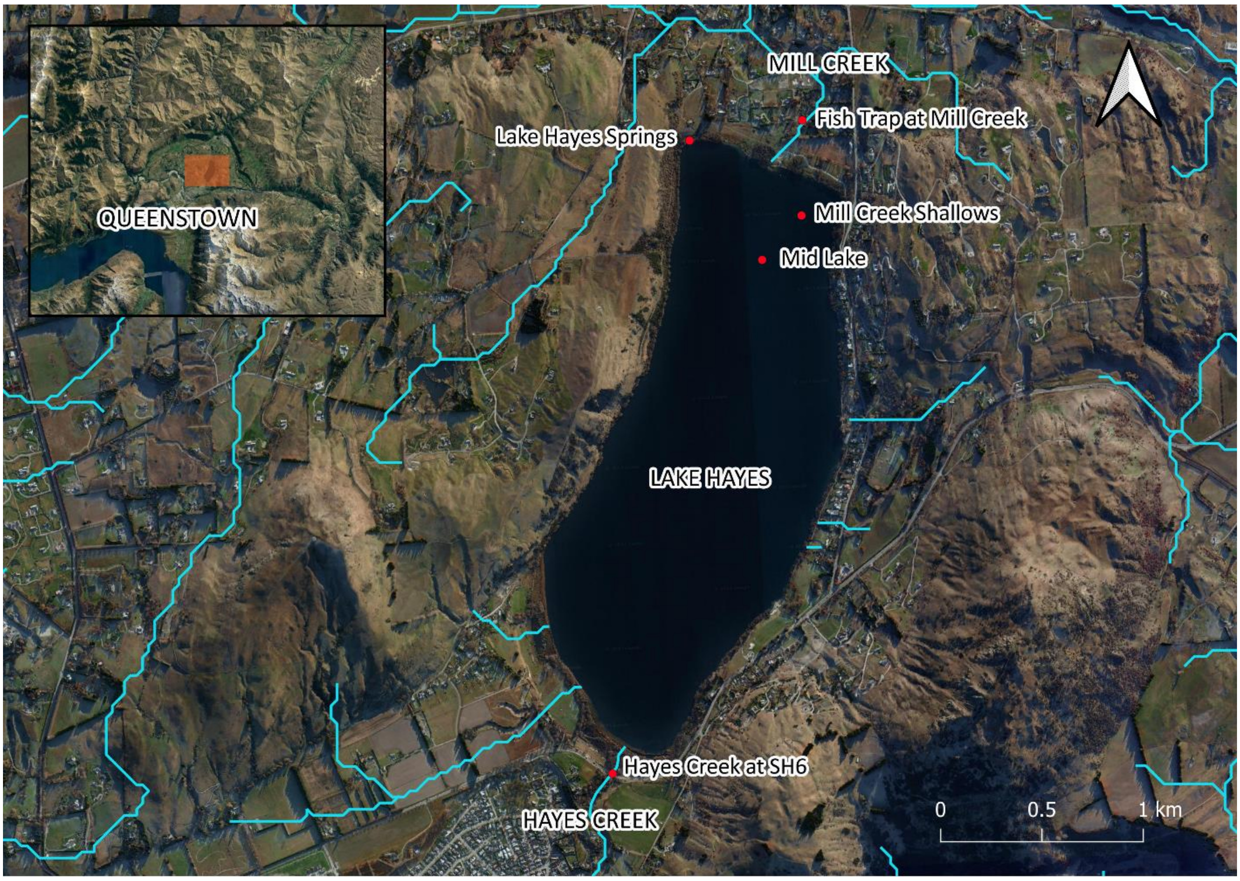

Figure 1. Lake Hayes and surrounding monitoring sites.

The FOLH data included spot measurements of water quality at the Fish Trap at Mill Creek including samples collected during floods.

Fish observational records were sourced from the New Zealand Freshwater Fish Database for Lake Hayes, Mill Creek, and its tributaries. These data were accessed on 18 December 2022.

Otago Fish and Game provided data on angler hours spent at Lake Hayes.

The LAWA website archived data on Lake Hayes and Mill Creek supplied by the ORC. We specifically used LakeSPI (macrophytes) and Trophic Level Index (TLI; an indicator of lake trophic state) data, accessed 28 January 2023.

The NIWA report on submerged macrophytes (Burton 2021) provided information on the impact of invasive macrophytes on the lake macrophyte community and on the condition of the native macrophyte community of the lake.

4 Pre-History and history

4.1 Catchment

The Lake Hayes catchment (Figure 2) is 44 km in area, primarily extending to the north-west along its main inflow, Mill Creek. Mill Creek is fed by a number of high-country streams including O'Connell Creek, Station Creek, and McMullan Creek. The lower catchment (below tree line) was predominantly forested prior to the arrival of humans to New Zealand, c. 700 years ago (McWethy et al. 2014). Mid-catchment, a wetland extended through the western reaches of Mill Creek and a number of smaller wetland swamps also existed in the catchment to the west and north of the lake (Robertson 1988).

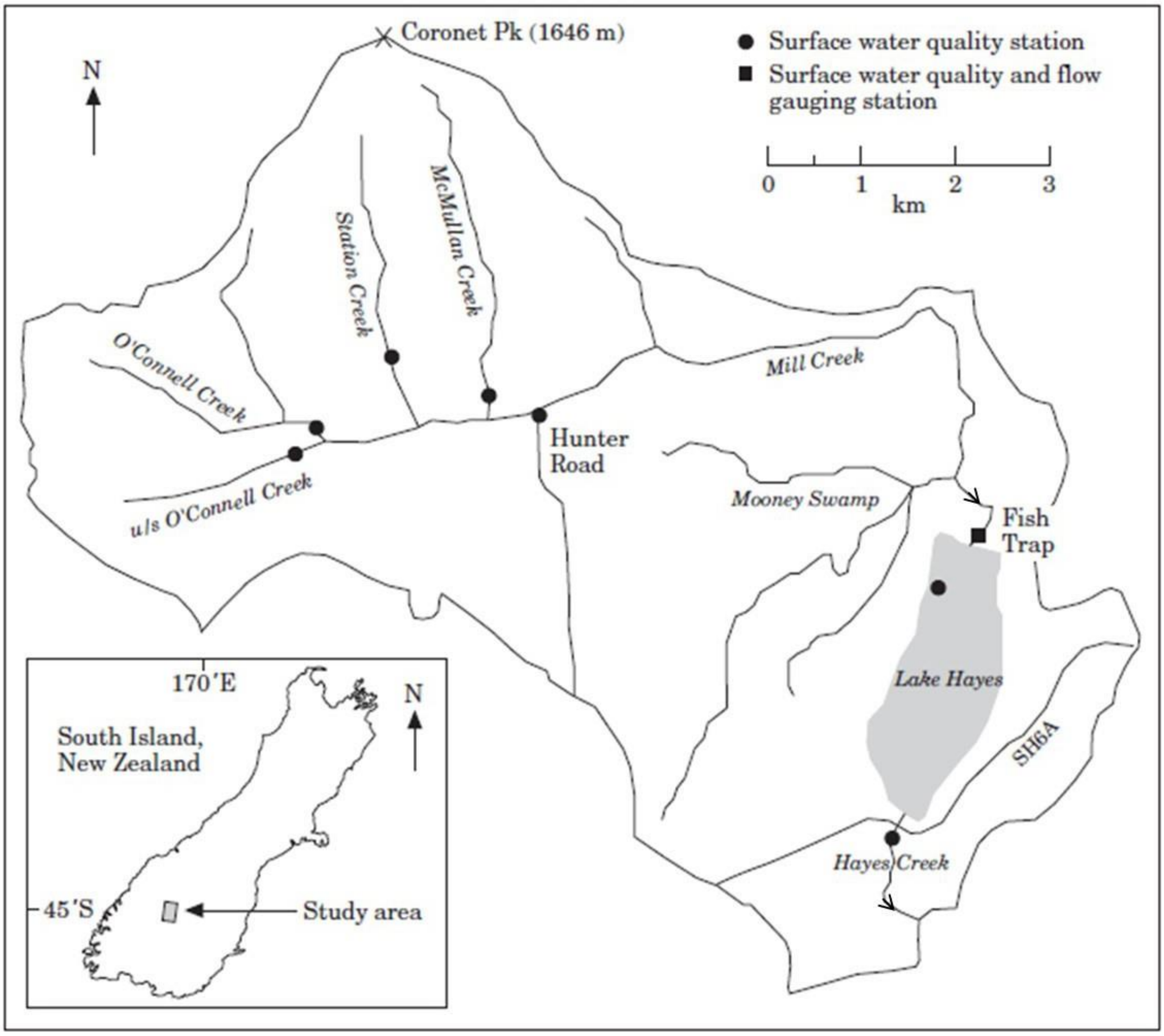

Figure 2. The Lake Hayes catchment (adapted from Caruso 2001).

Smaller wetland areas also existed adjacent to the mid-reaches of Mill Creek including Mooney Swamp, which acted as wildlife habitat, flood mitigation and a sediment and nutrient sink. Any sediment and nutrient losses from the catchment to waterways would have been largely intercepted in the wetlands before reaching Lake Hayes. Thus, it is likely that prior to large-scale burning, deforestation, and wetland drainage, lower concentrations of nutrients and sediment would have been transported to Lake Hayes by Mill Creek and its tributaries.

Extensive fires soon after human arrival converted much of New Zealand's forests into stable grass- and scrub-lands (McWethy et al. 2014) and as a result, the extent of wetlands in deforested catchments may have increased. Immediately following widespread forest burn offs, soil erosion in steep catchments likely increased, but rapid recovery by bracken fern, scrub and grasslands would have stabilised soils again. Thus, transient pulses of sediment and nutrient loads to the lake were expected to occur as a result of human-induced burning.

Upon the arrival of European settlers to the Wakatipu Basin in the 1860s, European techniques of land cultivation and exploitation began in the Wakatipu Basin, starting with gold mining and extensive sheep farming. In the 1870's, brown trout were introduced to Lake Hayes and the lake was known as a legendary trout fishery, which likely altered the foodweb of the lake, resulting in a rapid reduction of native fish (e.g., kōaro and bully) abundance in the lake. European perch were likely introduced soon after, perhaps in the 1880's. Through predator effects on lower trophic levels (e.g., on zooplankton), it is possible that the rapid change in the lake foodweb due to the introduction of these fishes also affected algal abundance and water clarity in the lake, although we haven't found any written records of this.

As settlement of the region continued from the 1880's to the mid 1900's, land in the Lake Hayes catchment was increasingly utilised as sheep pasture and lower-lying areas were converted from sheep to beef and dairy pasture. In 1912, a nearby cheese factory began discharging effluent into Mill Creek and in the 1940s, the Wildlife Service constructed a fish trap near the mouth of Mill Creek where thousands of spawning brown trout were collected annually to produce fertilised ova and trout fry for trout aquaculture. The slow pace of development in the catchment likely had relatively minor effects on the lake during this era, although discharges from the cheese factory potentially contributed 1000 kg of P per annum to the lake via Mill Creek until the factory closed in 1955 (Robertson 1988). Leftover whey from cheesemaking was fed to pigs, which likely also contributed to nutrients flowing into Lake Hayes (Roberston 1988).

Major effects on the lake likely began in the 1950s when areal topdressing with P fertilisers began, enabling higher rates of pasture growth and increased stocking rates on farms in the catchment. In 1953, a topdressing plane was lost in the lake (Robertson 1988), although it is neither clear whether the plane was recovered nor how much of its payload ended up in the lake. In palaeolimnological studies carried out in developed countries, apparent rates of lake water quality degradation often coincide with the post-WWII period and the advent of industrialised farming. In the environmental change literature, the post-WWII period is sometimes referred to as the Great Acceleration (e.g., McNeil & Engelke 2014).

In 1961-62, the Otago Catchment Board undertook drainage and channeling works in the catchment, which drained many remaining wetlands and artificially channelized parts of tributaries of Mill Creek, into new, high-producing pastures. In 1961, these works drained 80-120ha of wetland in the upper catchment, producing a significant amount of sediment discharge to Mill Creek and into Lake Hayes (Robertson 1988). Locals recorded the first sighting of "brown water" flowing into the lake in 1961 which continued sporadically throughout the remainder of the drainage and channelisation works over the next few decades (Robertson 1988). This significant land conversion and resulting sediment immobilization due to the destruction of wetlands in the early 1960s has been touted by some elderly locals who remember that time as the period when water quality in the lake became noticeably degraded. Robertson (1988) estimated that the existing wetland would have intercepted 80% of the P transported in Mill Creek. Given the other ecosystem services provided by wetlands, these ones would also have trapped much suspended sediment, particulate N, and *E. coli* as well as converted some nitrate in Mill Creek to inert N_2 gas. The supply of irrigation water from the Arrow River Irrigation scheme via the Hayes Race to, at times, >1000 ha of the mid-catchment farmland also increased the productivity of pastures in the catchment. Thus, this conversion of wetlands into high-producing pasture reduced the natural sediment and nutrient processing potential of the catchment's wetlands. This would have shunted any eroded soils, particulate phosphorus, *E. coli*, nitrate and other contaminants into Mill Creek which feeds into the lake.

By the late 1960s, algal blooms were being reported in Lake Hayes and by the 1970s, research on the lake mentioned its degraded state (e.g., Cook 1973), with the lake exhibiting a seasonally anoxic hypolimnion (bottom waters), substantial internal phosphorus load, and cyanobacterial blooms (Mitchell & Burns 1972; Burns & Mitchell 1974;. Mitchell & Burns 1981). By this time, the number of houses and baches around the lake was increasing and the ORC recognised the potential for septic tank seepage to affect the lake's water quality. Therefore, in the late 1990s, the septic tanks in proximity to the lake were decommissioned and sewage was reticulated out of the catchment at the Queenstown sewage plant on the Shotover River floodplain.

Rapidly increasing land prices in the 1990s resulted in a gradual loss of more intensive farming in the catchment in favour of residential property development, holiday homes, lifestyle blocks and resorts/golf courses. As land prices have continue to rise in the past decade, population growth, developments, and associated earthworks have also accelerated to the point where these developments may be further contributing to the degradation of water quality in the lake (FOLH 2018). For example, in February/March 2018, March 2020, and November 2022, the ORC warned recreational users of Lake Hayes to avoid contact with the lake due to conspicuous blooms of potentially toxic cyanobacteria.

4.2 Historical inferences on lake condition from sediment cores

Khan (2021) examined sediment cores obtained from the deepest site in Lake Hayes to understand how algal and zooplankton abundances have changed in the lake over time, in response to introductions of brown trout and perch. These fish introductions in the late 19th century had comparatively little effect on the algal pigments (i.e., abundance), although few samples representing sediment laid down prior to 1850 were available for analysis. On the other hand, the concentrations of algal pigments in the sediments showed four apparent phases of lake productivity: (1) the period from c. 1853 to c. 1945 when inferred algal biomass was low, (2) the period from c. 1946 to c. 1986 when inferred algal biomass increased markedly, (3) the period from c. 1987 to c. 2006 when inferred algal biomass declined again, and (4) the period from c. 2007 to 2017 when variability in algal pigment concentrations was high, indicating periods of alternating high and low inferred algal biomass.

This study also showed that the native grazing zooplankter, *Daphnia thomsoni*, was almost entirely displaced by the invasive *Daphnia pulicaria*, in c. 1980. This invasive species is known to prefer warmer temperatures and to reach higher densities than the native *Daphnia*, suggesting that grazing pressure on algae has increased in the past approximately four decades. This could have a positive effect of reducing planktonic algae concentrations in the lake, but may be subject to seasonal and inter-annual variability.

In summary, the inferences from sediment core data suggest that the main phase of eutrophication of Lake Hayes began after WWII (i.e., around 1950), reaching the highest algal biomasses in the 1980s. This is consistent with reports of cyanobacterial blooms in the lake in the 1970s (Burns & Mitchell 1974) and probably persisting into the 1980s and 1990s. Thus, the lake water quality appears to have degraded substantially from its condition in the late 1800's and early 1900's and this degradation happened during the era of agricultural intensification, wetland drainage, irrigation, increasing residential development near the lake, and increasing development further up the catchment. It is difficult to isolate the effects of these different pressures using the methods in Khan (2021). However, the study did show that the introduction of trout and perch in the late 19th century appeared to have little impact on algal biomass. However, after eutrophication had occurred and invasive *Daphnia* had colonised the lake, food web dynamics may have caused the lake to become less stable whereby it has been alternating between high and low levels of algal biomass in different summers.

5 Lake Hayes Inflows and Outflow

5.1 Mill Creek state and trends

In this section, we examine the water quality of the inflows and outflow of Lake Hayes. We focus on the Mill Creek inflow because it drains the majority of the Lake Hayes catchment and because Mill Creek is an ORC SOE site and has been the focus of Lake Hayes catchment water quality monitoring that has been carried out by the ORC and the FOLH. Furthermore, historical analyses of catchment contaminant loads to Lake Haye have focused on Mill Creek (e.g., Robertson 1988; Caruso 2000).

Less data are available on the water quality of the springs at Rutherford Road and on the outflow of Lake Hayes (Hayes Creek). We were unable to find any water quality measurements or calculations of contaminant loads from the minor tributaries draining into Lake Hayes.

Our analysis utilised measurements of monthly water samples from ORCs Fish Trap monitoring site (**Figure 1**). We analysed the following suite of variables:

- Total nitrogen (TN);
- Total phosphorus (TP);
- Dissolved reactive phosphorus (DRP);
- Nitrate-nitrite nitrogen (NOx-N);
- Total suspended solids (TSS);
- Turbidity; and
- *E. coli.*

In addition, high-frequency stage (i.e., water level) measurements are recorded by ORC at the Fish Trap and these are converted to discharge (henceforth referred to as "flow") by the ORC, based on numerous flow gaugings done at the site. In addition, high frequency turbidity and nitrate measurements are also available for the Fish Trap site and these were used to calculate sediment and nutrient loads to the lake. The FOLH have carried out water sampling at the Fish Trap during flood flows and their data enabled us to make more realistic estimates of contaminant loads because we could extend calibration curves to include flood conditions.

Most of our analyses cover the period since the revised ORC lake monitoring programme commenced at the end of 2016. In some cases, we present more historical data to provide important historical context, but this report focuses mainly on the assessment of water quality and ecosystem health of the Lake Hayes hydrosystem in recent times.

5.1.1 Temperature

A continuous temperature of Mill Creek has been recorded and logged at the Fish Trap site by the ORC at 5-minute intervals since December 1, 2020. The daily average, maximum, and minimum temperatures are plotted in Figure 3, showing that temperatures reach just over 20˚C in summer and descend to 1 to 3˚C in winter. The range of temperatures measured throughout a 24 h period is greater in summer than in winter. Although this record is too short to assess trends in temperatures, it provides a baseline from which changes in future temperatures can be quantified.

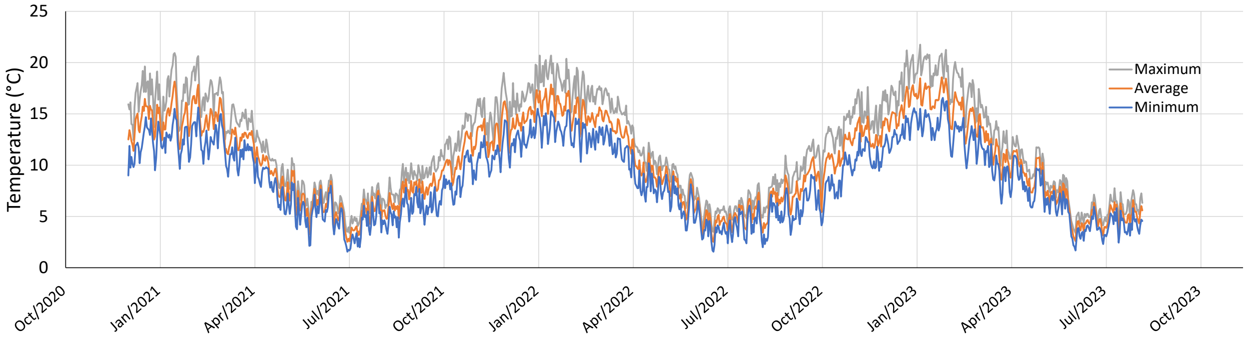

Figure 3. Daily maximum, average, and minimum temperatures from measurements made at 5-minute intervals

at the Fish Trap site.

5.1.2 Flow

The flow (discharge) of Mill Creek averages approximately 0.4 m^3/s, with highest flows generally observed in spring (Figures 4 and 5). Robertson (1988) and Caruso (2000) discuss in detail the seasonality of flows in Mill Creek. In Figure 4, we present a time series of average daily flows going back to 2006, to illustrate the cyclical nature of seasonal flows and the irregular patterns of interannual differences in flows, with multi-year periods when no significant floods occurred (e.g., 2010 to 2012) and multi-year periods with more significant floods (e.g., 2018 to 2021). Although we haven't analysed the drivers of this inter-annual variability, it is likely that large scale weather/climate patterns such as the southern oscillation and the interdecadal Pacific oscillation drive these variations. The magnitudes of spring floods are likely related to snow accumulation over winter and the dynamics of the spring thaw. High runoff and overland flows that occur during intense thawing and rainfall events can mobilise soil, sediment, total phosphorus and *E. coli*, as well as contribute to high stream flows which can result in bank erosion. The magnitude, frequency and dynamics of flood flows also influences the load of contaminants to Lake Hayes. This will be explored in more detail in Section 5.1.3, below.

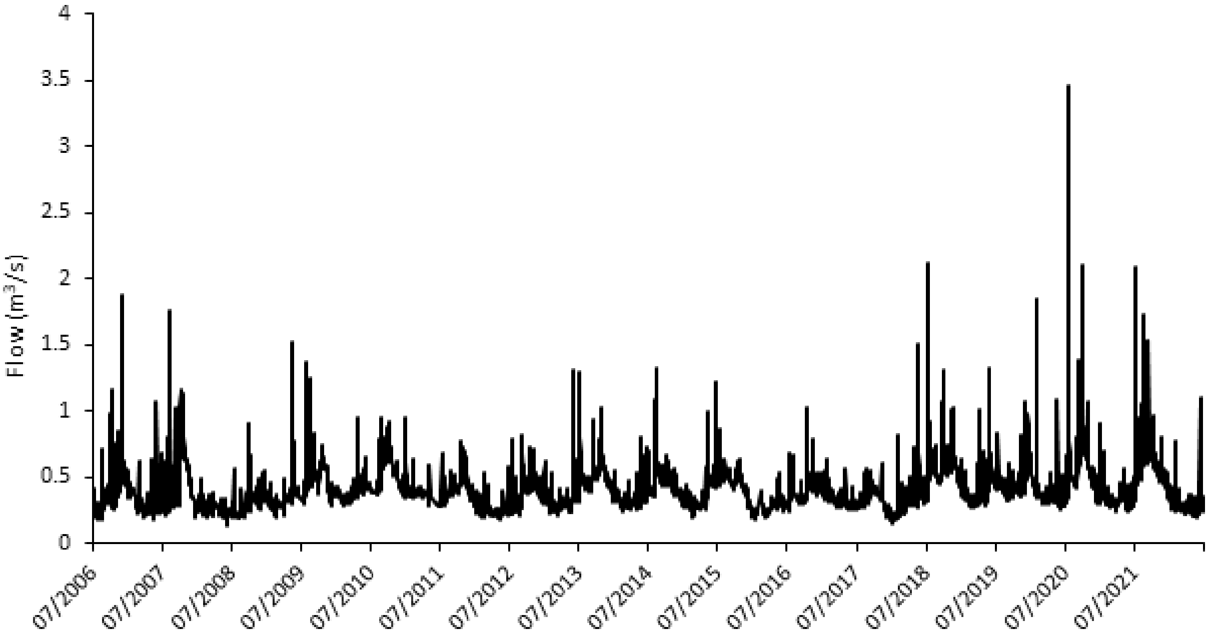

Figure 4. Mill Creek average daily flow from 2006 to 2022.

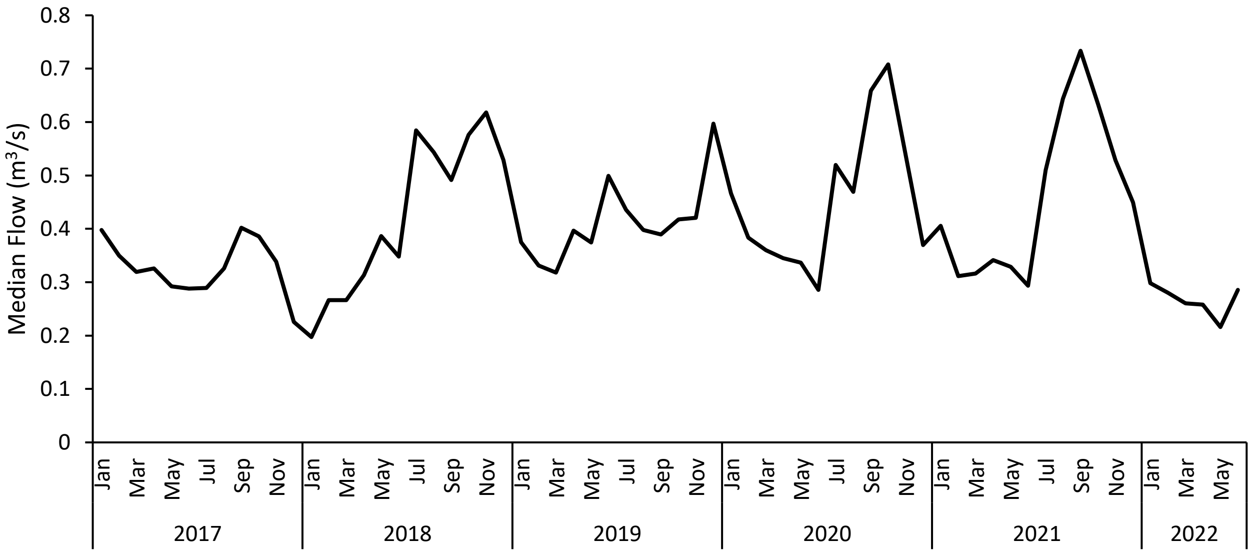

Figure 5. Mill Creek median monthly flow from 2017 to 2022.

5.1.3 Nutrients

The concentration of TN within Mill Creek has increased since 2017 and shows a strong seasonal pattern, with higher concentrations occurring in June through August (Figure). NOx-N, henceforth referred to as "nitrate", displays a similar seasonal pattern, constituting around 60% of the TN in the samples collected at median flow or lower. The 2020 ORC SOE assessment of water quality in Mill Creek at fish trap reported a five-year average nitrate-N concentration of 0.36 g/m^3 which exceeded the water quality standard for Mill Creek of 0.075 g/m^3 (80th percentile) in Schedule 15 of the Otago Water Plan (ORC 2014; ORC 2020; Figure).

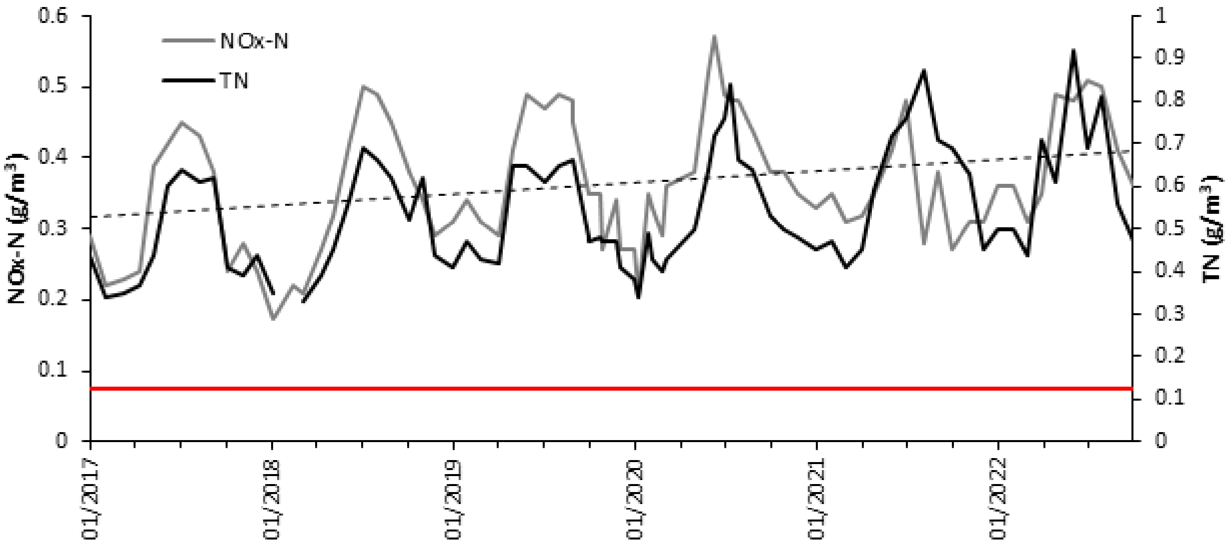

Figure 6. Mill Creek monthly total N (TN) and nitrate+nitrite-N (NOx-N) measurements, 2017 - 2022. Red line indicates ORC Water Plan standard for nitrate-N (approximated by NOx-N). The dashed line is the linear trend for NOx-N.

The ORC assessment of water quality in Mill Creek reported a five-year average DRP concentration of 0.007 g/m^3, (July 2015 to June-2020) which is below the water quality standard of 0.010 g/m^3 (80th percentile) in the Otago Water Plan (ORC, 2020). Since 2020, both TP and DRP concentrations have increased in Mill Creek (Figure 7). DRP constituted around 15% of the total phosphorus concentrations in the ORC samples.

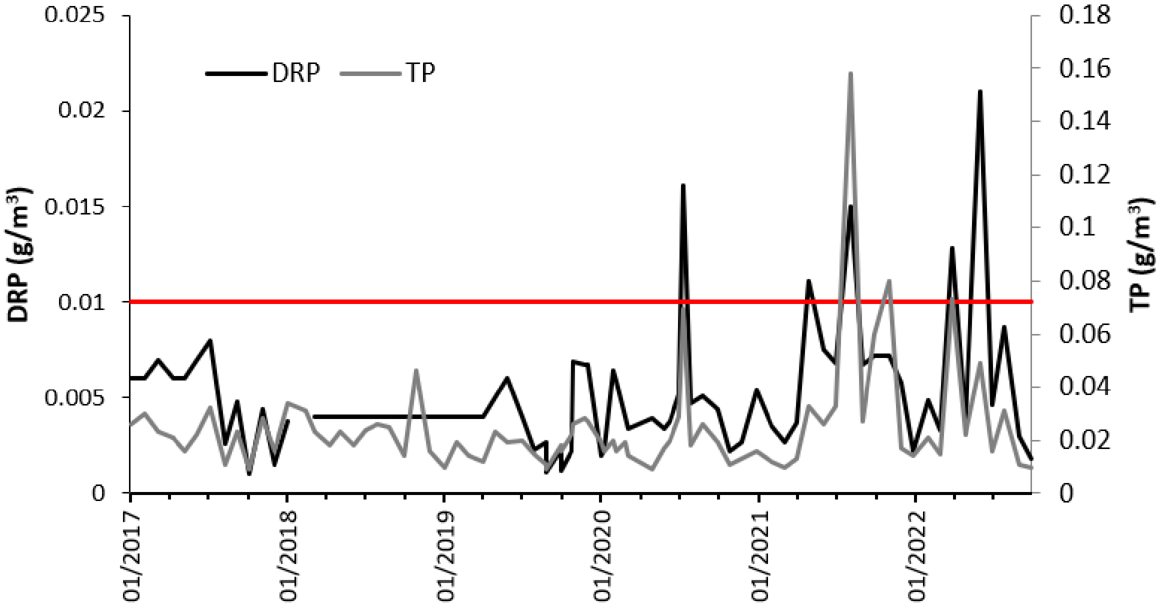

Figure 7. Mill Creek monthly dissolved reactive phosphorus (DRP) and total phosphorus (TP) measurements, 2017 - 2022. Red line indicates ORC Water Plan standard for DRP.

Concentrations of TN and TP were only weakly correlated and only at higher concentrations (.

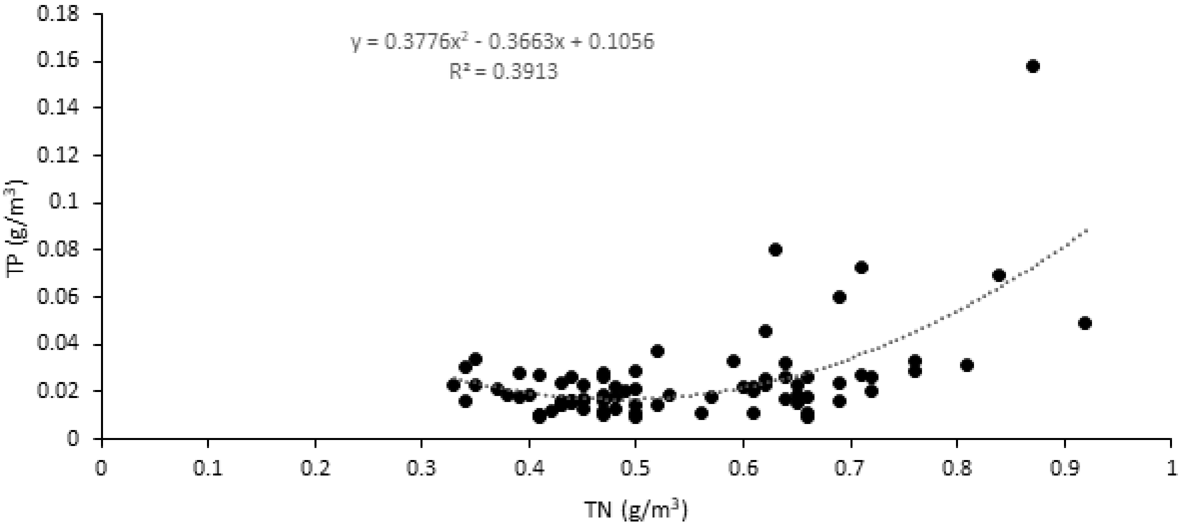

Figure). Note that the variation was approximately 18-fold for TP concentrations and only approximately 3-fold for TN concentrations for these samples, which are collected at lower flows.

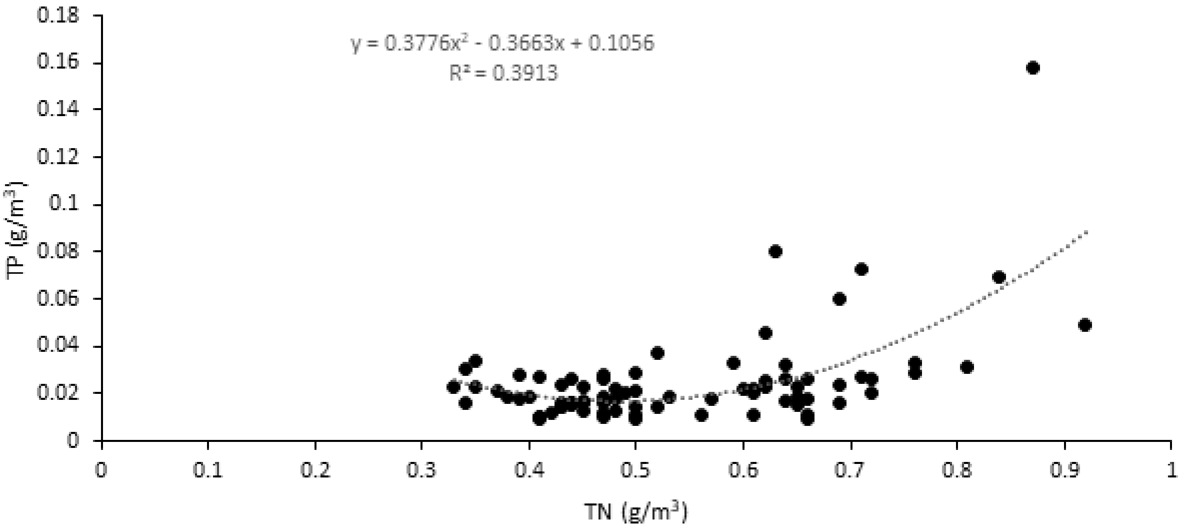

Figure 8. Mill Creek monthly measures of total nitrogen (TN) against total phosphorus (TP).

The NOF contains periphyton standard that specify nutrient thresholds to control periphyton growth. The ORCs assessment of Mill Creek nutrient data in relation to the DPR and TN in the periphyton standard indicates that the Mill Creek falls into the C-band (ORC 2021). The bottom of the C-band is the national bottom line, which demarcates the acceptable nutrient threshold.

Nutrient loads can be estimated using flow, nutrient concentration, and turbidity data, which can then inform lake nutrient budgets (see Section 7.2, below). The ORC installed a logging nitrate sensor at the fish trap site in Mill Creek to provide hourly nitrate data. Using the ratio of nitrate to TN obtained from grab samples (September 2018 to June 2020), hourly TN concentrations were estimated from nitrate concentrations by multiplying nitrate-N by 1.4. The ORC installed a logging turbidity sensor at the fish trap site in Mill Creek. Grab samples obtained by the Friends of Lake Hayes at a variety of flows were used to derive a linear relationship between turbidity and TP (R^2 = 0.97, n = 21) and this was used to estimate hourly TP concentrations in Mill Creek. Hourly nutrient concentrations were multiplied by ORC's hourly discharge data to calculate mass fluxes, which were integrated over time to determine annual nutrient loads from Mill Creek to the lake between 2020 and 2023 for phosphorus and for 2020 for nitrogen. Small gaps in the data were linearly interpolated. Larger gaps in turbidity data in 2022, were derived from the relationship between flow and turbidity (R^2 = 0.98, n = 17). Flow and turbidity data were also available for the calandar years 2020, 2021 and 2022, whereas nitrate data were only available for 2020. Measured and regression-derived high-frequency data for 2020 are shown in Appendix A.

The derived nutrient load estimates from Mill Creek and other inputs are presented in Table 2, together with TP loads from the years 1983/84 (Robertson 1988) and 1997 (Caruso 2000). Nutrients in rainfall were obtained from annual precipitation data from Queenstown airport and nutrient data for South Island rainfall from Verburg et al. (2018). The ORC and Friends of Lake Hayes provided data on nitrate, TN, TP and flow for the springs at Rutherford Road on four dates in 2020. These parameters were averaged and the average nutrient concentrations were multiplied by the average daily flow. The resulting daily load was then multiplied by 365 to yield annual nutrient load estimates for the springs. Total P concentrations were almost always below analytical detections limits and the TP values were, therfore, set to half of the detection limit. We could not estimate the nutrient inflows from the portions of the catchment not drained by Mill Creek as no flow or nutrient data are available for the small streams were not available, many of which flow only intermittently.

Table 2. Total phosphorus, total nitrogen and nitrate-N loads along with annual water discharges to Lake Hayes estimated for 1983/84 (Robertson 1988), 1997 (Caruso 2000) and 2020, 2021 and 2022 (this study).

Year	Inflow	Total P	Total N	Nitrate-N	Water	Reference
		kg yr^{-1}	kg yr^{-1}	kg yr^{-1}	$m^3 \times 10^6$	
1983/84	Mill Creek	2,100			22.0	Robertson (1988)
	Rutherford Road Spring	20*			1.5	
	Rainfall	10-20*			2.9	
	Other tributaries	200*			1.6	
1997	Mill Creek	395			12.5	Caruso (2000)
2020	Mill Creek	2,278	42,533*	30,359	15.6	This study
	Rutherford Road Spring	2	1,318*	1,234	1.2	
	Rainfall	64*	637*		1.8	
	Other tributaries					
2021	Mill Creek	491			15.6	This study
2022	Mill Creek	535			13.3	This study

** values were estimated, not measured*

The load estimates in Table 2 suggest that annual TP loads from Mill Creek are much higher than other sources of TP to the lake, ranging from 395 to 2,278 kg yr^{-1}. This high variability from year-to-year is not strongly related to the annual water discharge to the lake, which varied from 12.5 to 22.0 $\times 10^6$ m^3. The 5.8-fold variation in TP load only corresponds to a 1.8-fold variation in annual water discharge. Other factors also influence annual TP loads, such as rainfall dynamics and land use practice. This is highlighted by the large difference in TP loads in 2020 and 2021, two years when the annual water discharge was identical. In 2020, TP (and turbidity from which it is derived) was much higher than in 2021, despite the discharge of water down Mill Creek being equal. This points to non-weather-related sources of turbidity as the driver of the roughly 4-fold higher TP load in 2020.

Our calculations of annual nitrate and TN load in 2020 show that Mill Creek is by far the largest contributor of TN to Lake Hayes, with the Rutherford Road springs only contributing 4% of the nitrate load, significantly less than previous estimates (e.g., Bayer et al. 2008). Nutrient loads are discussed in more detail in Section 7.2.

5.1.4 Turbidity and suspended sediment

Suspended sediment is the main contributor to turbidity in Mill Creek therefore, the close association between monthly turbidity and TSS concentrations is not surprising (Figures 9 and 10). Based on ORCs monthly samples from 2017 to 2023, the median turbidity of Mill Creek was 3.95 NTU (Fig. 9) and the 80th percentile was 4.26 NTU (ORC 2020), both were not far below the ORCs Schedule 15 water plan standard of 5 NTU (ORC 2014). However, the 10- and 20-year trends in turbidity are both worsening (ORC 2021), suggesting that suspended sediment concentrations in Mill Creek have been increasing over time and that they may soon breach the standard in the regional water plan.

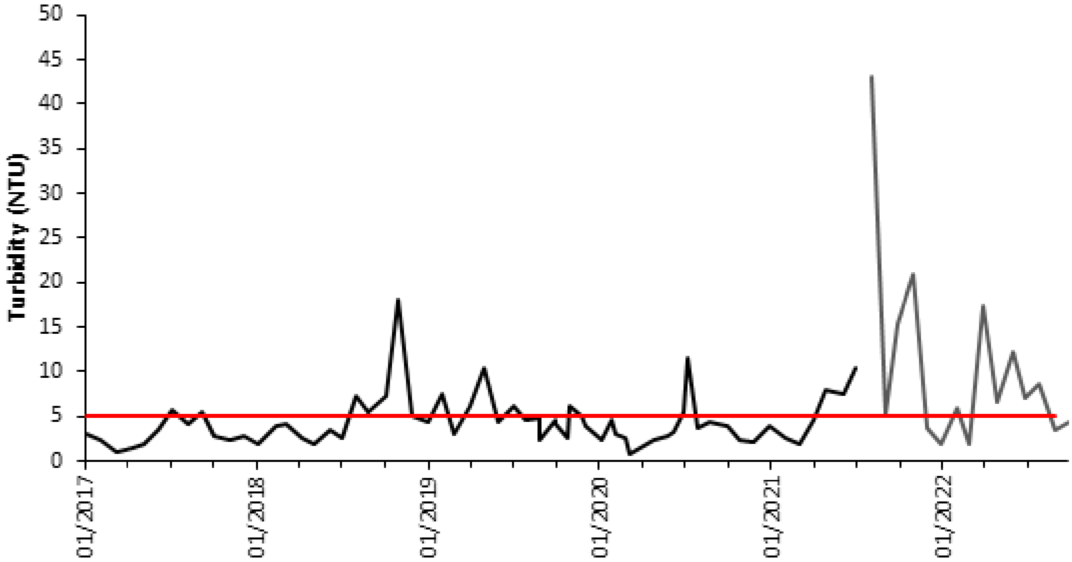

Figure 9. Mill Creek monthly turbidity measurements from 2017 to 2023. Black line indicates results reported in NTU, while gray line indicates results reported in FNU. The red line indicates the 5 NTU ORC water plan standard for the 80th percentile.

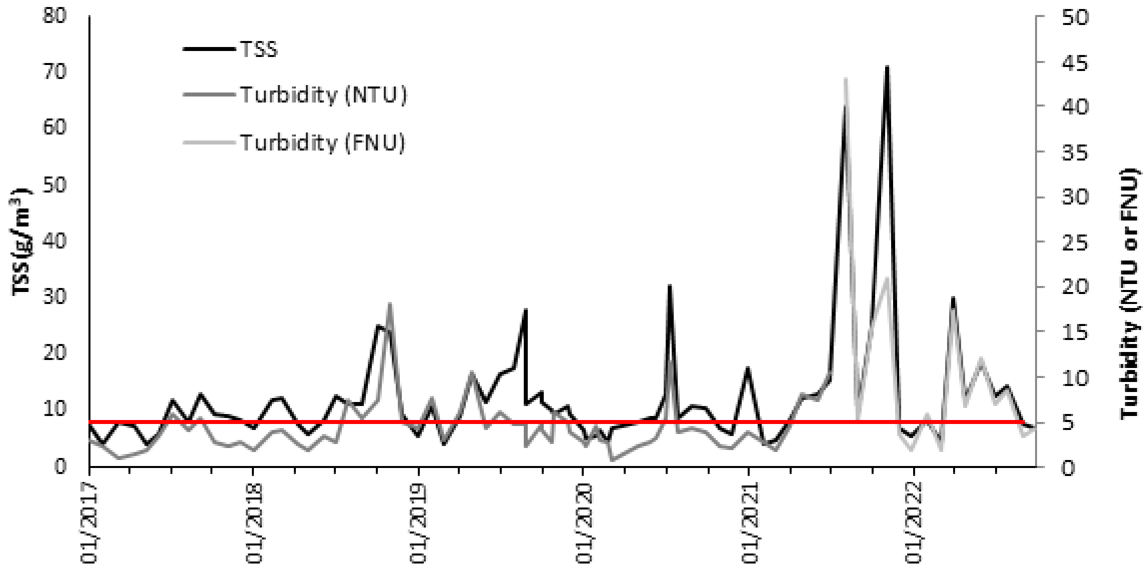

Figure 10. Mill Creek monthly total suspended solid (TSS) and turbidity measurements from 2017 to 2023. The red line indicates the 5 NTU ORC water plan standard for the 80th percentile of turbidity.

The NOF specifies suspended sediment standard for rivers. The ORCs assessment of total suspended sediment in Mill Creek indicates that Mill Creek is in the D-band and breaches the national bottom line for this attribute (ORC 2021).

The FOLH sampled TSS during a wide range of flood conditions, enabling the calculation of robust calibration curves for estimating hourly TSS concentrations from hourly turbidity data. After summing up for the calendar years, annual sediment loads for 2020, 2021, and 2023 were calculated. These are shown, together with the annual sediment load estimated for the year 1983/84 by Robertson (1988), in Table 3.

Table 3. Annual suspended sediment loads and water discharge for the years 1983/84 (Roberston 1988) and 2020 to 2022 (this study).

Year	Inflow	TSS	Water	Reference
		t yr^{-1}	m^3 ×10^6	
1983/84	Mill Creek	723	22.0	Robertson (1988)
2020	Mill Creek	2,302	15.6	This study
2021	Mill Creek	497	15.6	This study
2022	Mill Creek	540	13.3	This study

The pattern of sediment loads is similar to that for TP loads. This is because both are highly correlated with (and are derived from) turbidity. Note that the variation in annual water discharge does not explain the variation in annual TSS load and that factors other than rainfall and runoff drove the approximately 4-fold higher sediment loads in Mill Creek in 2020. Sediment loads are discussed in more detail in Section 7.3.

5.1.5 E. coli

While it is generally thought that E. coli are mobilised from land to water with flood events and associated overland flow, evidence for this is not compelling (e.g., Snelder et al. 2016; Whitehead et al. 2019). *E. coli* contamination of water can be associated with point sources of faecal pollution discharging to waterways, with waterfowl abundance, as well as with high runoff events.

The ORCs assessment of water quality at the fish trap in Mill Creek reported a five-year *E.coli* concentration of 300 cfu/100 ml (80th percentile in samples collected from median, or lower, flow conditions), which exceeded the ORCs Schedule 15 standard of 260 cfu/100 ml for Mill Creek (ORC, 2020). With highly episodic data such as those presented in Figure 11, it is difficult to quantify meaningful trends, and these are likely to be driven at least partly by changes in hydrology over time (see Section 6.1.1). The NOF specifies four different metrics for *E. coli* in its standard and Mill Creek is in the A- or B-band for all *E. coli* metrics. However, sampling tends to be carried out when the creek is below median flow, when high *E. coli* episodes are not likely to occur.

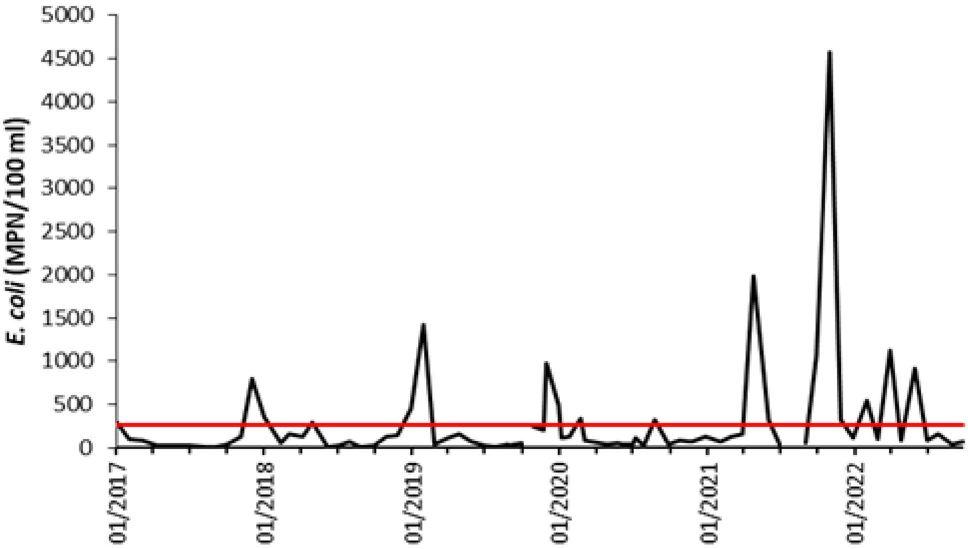

Figure 11. Mill Creek monthly *E. coli* results from 2017 to 2022. The ORC water plan standard of 260 cfu (colony forming units) per 100 mL is the red line. Note that the *E.coli* data presented are in MPN (most probably number) per 100 mL. Thus, the data and standard are not directly comparable. Despite this, the ORC assessed the *E. coli* counts in Mill Creek to exceed the Schedule 15 *E. coli* standard (ORC 2020).

5.2 Springs at Rutherford Road

Groundwater from the mid-Mill Creek aquifer emerges in springs at the north end of Lake Hayes (at

Rutherford Road). The discharge from the springs into the lake have been estimated to be approximately 25 L s^{-1} (ORC 2001). Monitoring of the water quality of the spring discharge carried out by the ORC and the FOLH indicates that the springs contain fairly high levels of nitrate (Figure 12). However, the TP (Figure 13), turbidity, and TSS (Figure 14) levels in the spring water are very low. Concentrations of nitrate in the spring water has fluctuated over the years. Roberstson (1988) reported levels ranging from 600 to 1600 µg L^{-1} in samples taken from 1983 to 1985. In the 1990s, nitrate ranged from 448 to 1539 µg L^{-1} (mean = 897 µg L^{-1}; Rosen & Jones 1998) while Bayer et al. (2008) measured levels of 448 to 1080 µg L^{-1} (mean = 791 µg L^{-1}) between March and July 2006. Nitrate concentrations were very stable from 2018 to 2020 at around 1000 µg L^{-1}. The natural background level for groundwater in the Wakatipu Basin was estimated to be around 70 µg L^{-1} (Rosen & Jones 1998). Thus, the concentrations shown in Figure 18 are within the range of previously reported values going back to the 1990's, but are likely much higher than the natural levels expected for the area.

The annual nitrate-N load estimate of 1,234 kg N yr^{-1} from the springs (Table 2) constituted only 4% of the measured nitrate load to Lake Hayes in 2020, suggesting that mitigation efforts to reduce nitrate loads from groundwater viat the springs would likely have a minimal impact on the total nitrogen load to the lake.

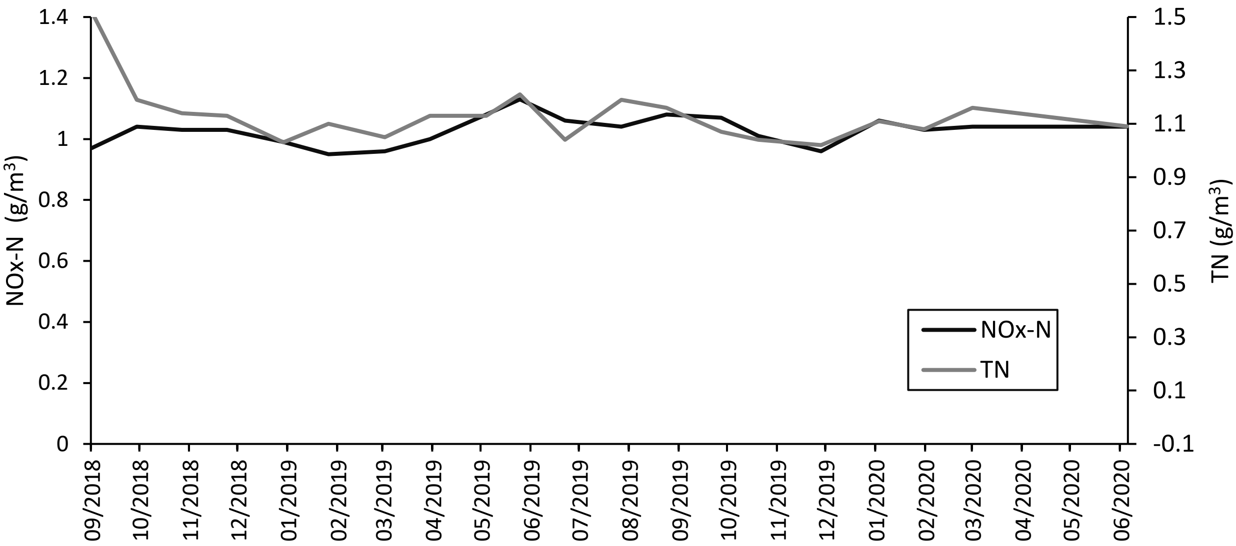

Figure 12. Nitrate nitrogen (NOx-N) and total nitrogen (TN) in Lake Hayes at the Rutherford Road Spring from 2018 to 2020.

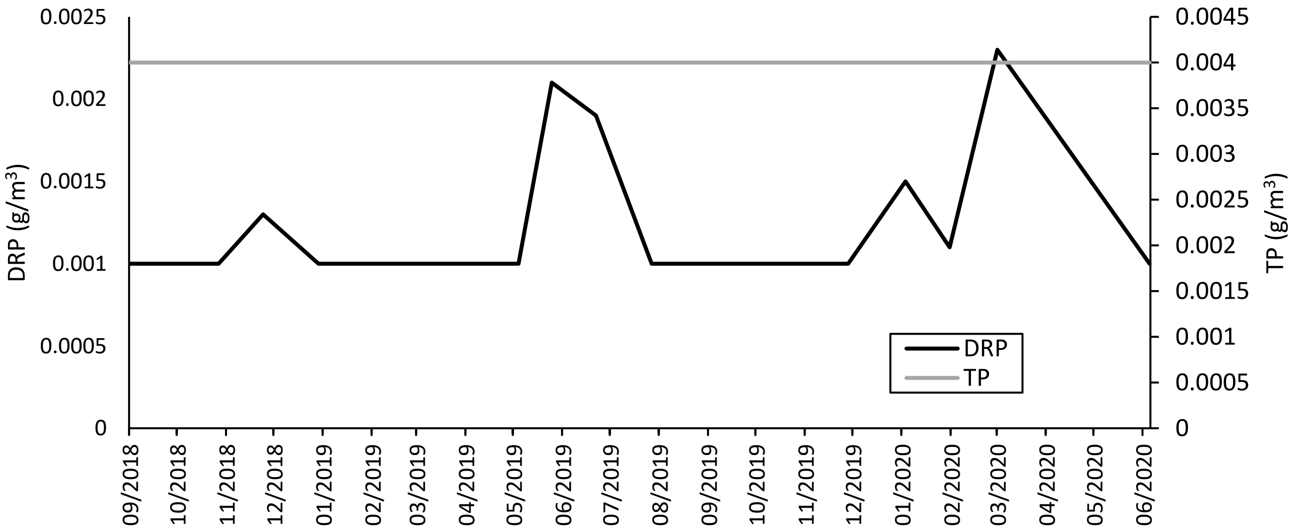

Figure 13. Dissolved reactive phosphorus (DRP) and total phosphorus (TP) in Lake Hayes at Rutherford Road Spring from 2018 to 2020.

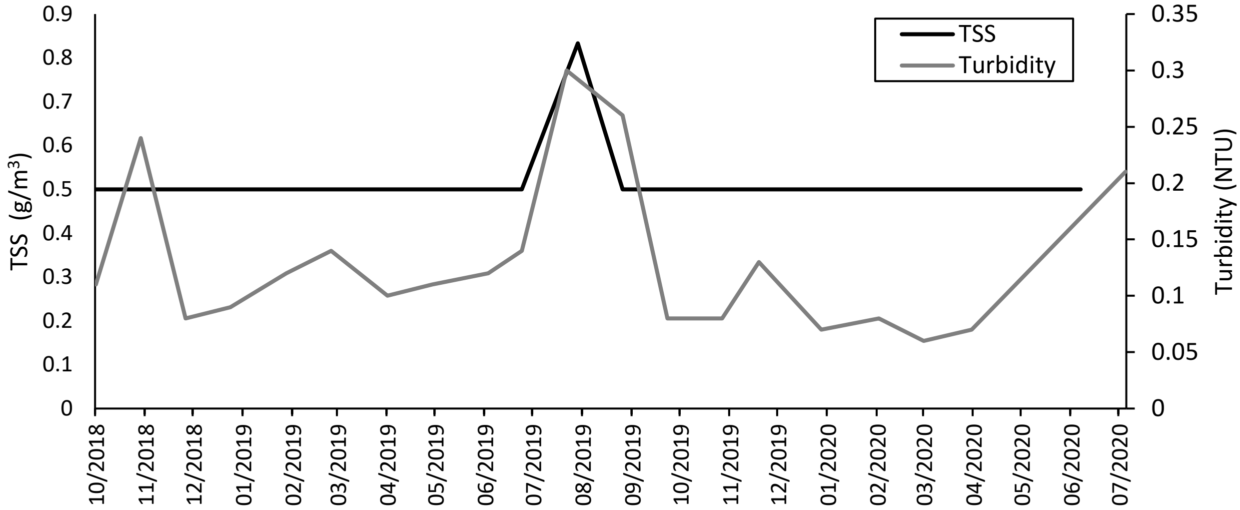

Figure 14. Total suspended sediment (TSS) and turbidity in Lake Hayes at Rutherford Road Springs from 2018 to 2020.

5.3 Outflow at Hayes Creek

Hayes Creek is the outflow of Lake Hayes, located at the southern end of the lake. ORC monitored this site but unfortunately a gap in monitoring occurred between August 2018 and June 2020. Therefore, we did not analyse the data from Hayes Creek. However, turbidity and TSS concentration are provided in Figure 15. Hayes Creek is lake-fed and, therefore, the water quality of the creek should approximate the surface water quality of the lake. Lake data are presented in the next section of this report.

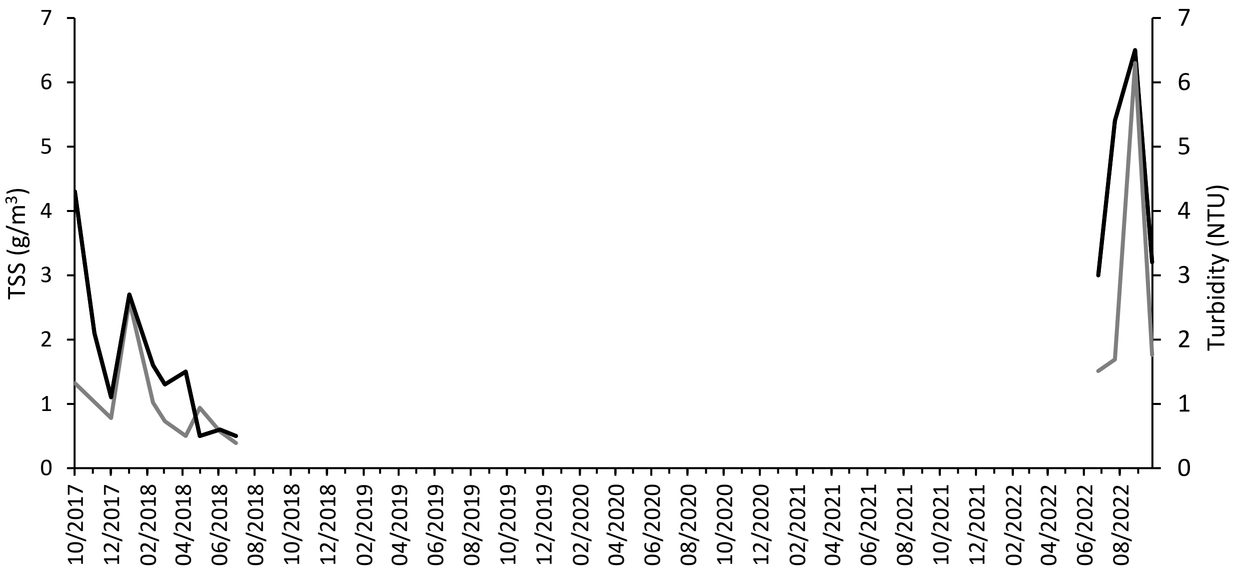

Figure 3 Total suspended sediment (TSS) and turbidity in Hayes Creek from 2017 to 2022. Turbidity is reported in NTU however the units were changed to FNU in 2022. No data was available from August 2018 through to June 2022.

6 Lake Hayes State and Trends

Lake Hayes is one of the most thoroughly monitored and studied lakes in New Zealand. A combination of interest from university academics since the 1940s coupled with the Otago Catchment Board/Otago Regional Council monitoring of the lake which began in the early 1980s, provides one of the best datasets in the country from which to interpret and understand lake conditions and trends.

6.1 Water level and culvert height

Lake water level fluctuations can influence many aspects of lake functioning. For example, patterns of shoreline erosion, aquatic plant distributions, and fish spawning habitats can be directly affected by lake water level variation. In addition, cycles of flooding and draining of shoreline wetlands and terrestrial soils can result in inputs of dissolved organic matter and nutrients as flooded areas are drained. Such effects have not been investigated in relation to water quality in Lake Hayes.

The measurement of water level at Lake Hayes was re-established by the ORC in 2019 (Figure 16). Since the end of 2019, the water level has varied by approximately 50 cm. Plotting lake levels together with monthly flow data from Mill Creek revealed a tight coupling between lake level and river flow. This relationship showed strong seasonality, with both flow and lake level increasing in winter and spring, and decreasing in summer. The pattern in lake levels appears to be driven both by predictable seasonal patterns in runoff and by more stochastic, large rainfall events. However, Figure 16 shows that individual rainfall events were able to rapidly raise the lake level numerous times by up to approximately 10 cm.

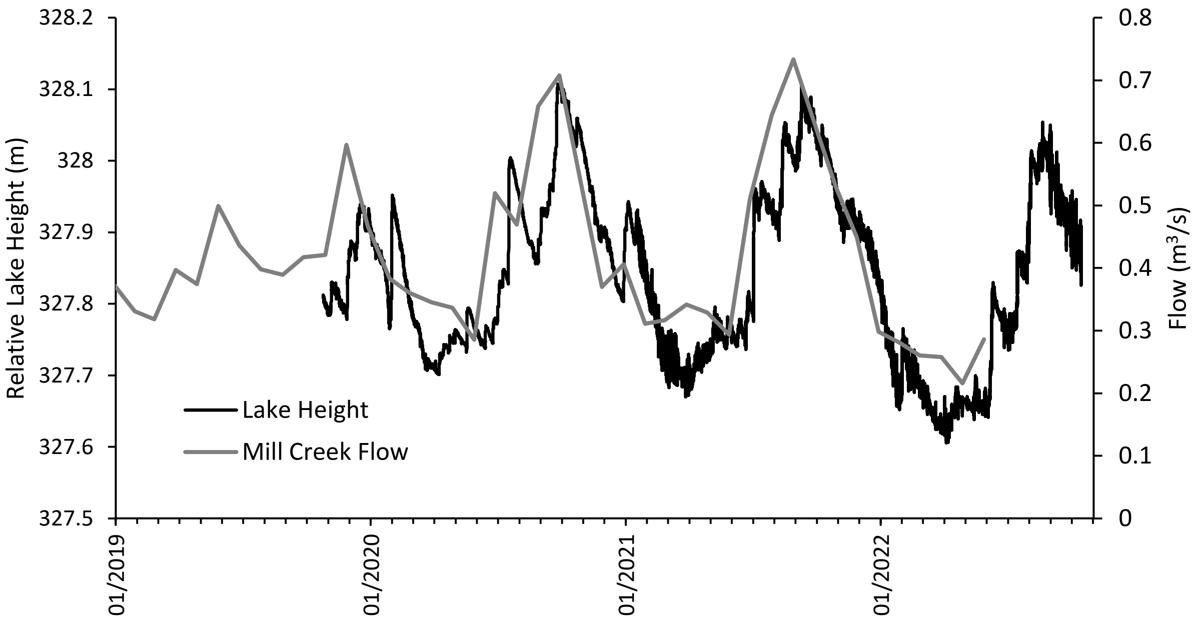

Figure 4 Lake Hayes water level relative to sea level and Mill Creek average monthly flow, 2019-2022.

6.2 Water temperature and stratification

When water is above 4°C, it's density decreases as it's temperature increases. During the winter months of June to August, the water column of Lake Hayes is isothermal (the same temperature), indicating that no density gradient exists and that the entire water column is mixed as a result of relatively low amounts of wind energy.

As the weather warms in early spring, periods of transient stratification can occur as a result of extended calm periods. However, wind-induced mixing can overcome the buoyancy forces caused by any small difference in water density and, thus, re-mixing of the lake water occurs. This happens until the rate of warming exceeds the rate of wind-induced mixing and buoyancy forces (and stratification) persist. Once stratification has established, further warming of the surface water increases the density differential between top and bottom waters until no amount of wind energy can fully mix the water column. This point in time marks the onset of seasonal density stratification, where mixing, temperature and gas exchange with the atmosphere only occurs in the upper, warmer water layer (the epilimnion). The deep layer (the hypolimnion) becomes isolated from the atmosphere by the warmer, less dense epilimnion. At this point, any processes occurring in the lake-bed that involve gases or solutes will influence the gas and solute composition of the hypolimnion. Such processes, including microbial respiration, denitrification, methanogenesis and sulphate reduction, can affect the gases and reduction-oxidation potential of the hypolimnion. This is why water from the hypolimnion in summer may lack oxygen and may smell of rotten eggs (hydrogen sulphide). The change in reduction/oxidation towards a more chemically reducing environment results in the dissolution of metal oxyhydroxides, which often contain iron and manganese and which have a high affinity for phosphorus. Thus, the dissolution of metal oxyhydroxides in an anoxic hypolimnion also dissolves sediment-bound phosphorus, which begins to diffuse upwards into the water column.

As the lake surface cools in autumn, the density differential between the cold, anoxic hypolimnion and the warmer epilimnion decreases and wind events are again able to deepen the mixed layer due to the weakening of the stratified water column. Ultimately, in June, the lake surface cools to the extent that mixing of the entire water column is inevitable. The low oxygen, high phosphorus bottom waters then circulate into the entire water column, diminishing the oxygen content and increasing the phosphorus content of the mixed layer. This process of deepening autumn mixing occurs from around March to May, with the full mixing of the water column usually occurring in June. Thus, an internal load of phosphorus is provided to the mixed layer from March until June. The above-mentioned processes are fundamental to the ecological functioning of seasonally stratified lakes, such as Lake Hayes, and are driven by the annual cycle of solar radiation, mediated by wind energy.

How this all plays out in Lake Hayes is well illustrated by the temperature data collected by the ORCs lake monitoring buoy (Figures 17 to 20). Unfortunately, there are some gaps in the data due to equipment malfunction, but there are enough data from 2019 to 2023 to illustrate the cycle of stratification.

From Figures 17 and 18, it is clear that the summer of 2022/23 produced a warmer and shallower epilimnion, probably due to calmer conditions. This created a more stable water column than in previous years (Figure 19). From the daily temperature profiles measured by the monitoring buoy it is possible to calculate the dates of onset and breakdown of thermal stratification as well as the duration of the stratified period in 2019/20 and 2022/23 (Table 4). Stratification began earlier and ended later in 2022/23 than in 2019/20 such that the lake was stratified for an additional 32 days. This would also contribute to the greater stability of stratification observed 2022/23. The predicted consequences of longer stratification would included higher temperatures (Figures 17 and 18), more intense oxygen depletion in the bottom waters, and high internal phosphorus loads. The latter is predicted both because the period of anoxia should be longer and because the area of lake-bed undergoing anoxia should therefore be greater.

Temperature is a key ecological variable affecting all physical, chemical and biological processes. For example, brown trout are sensitive to temperature and 25°C is generally considered to be a lethal temperature for this species (Elliot & Elliot 2010), which probably experiences some thermal stress at temperatures over 21°C. In addition, the risk of cyanobacterial blooms increases with increasing water temperature (Rigosi et al. 2015). Thus, we present tallies of the number of days per summer when the average temperature of the epilimnion exceeded 21°C, 22°C, and 23°C (Table 5). The temperature data is from night-time profiles collected by the lake monitoring buoy. The data show that the heat stress experienced by fish in 2022/23 was far greater than in 2019/202 and 2021/22. To escape the warm water, fish can only move to deeper waters.

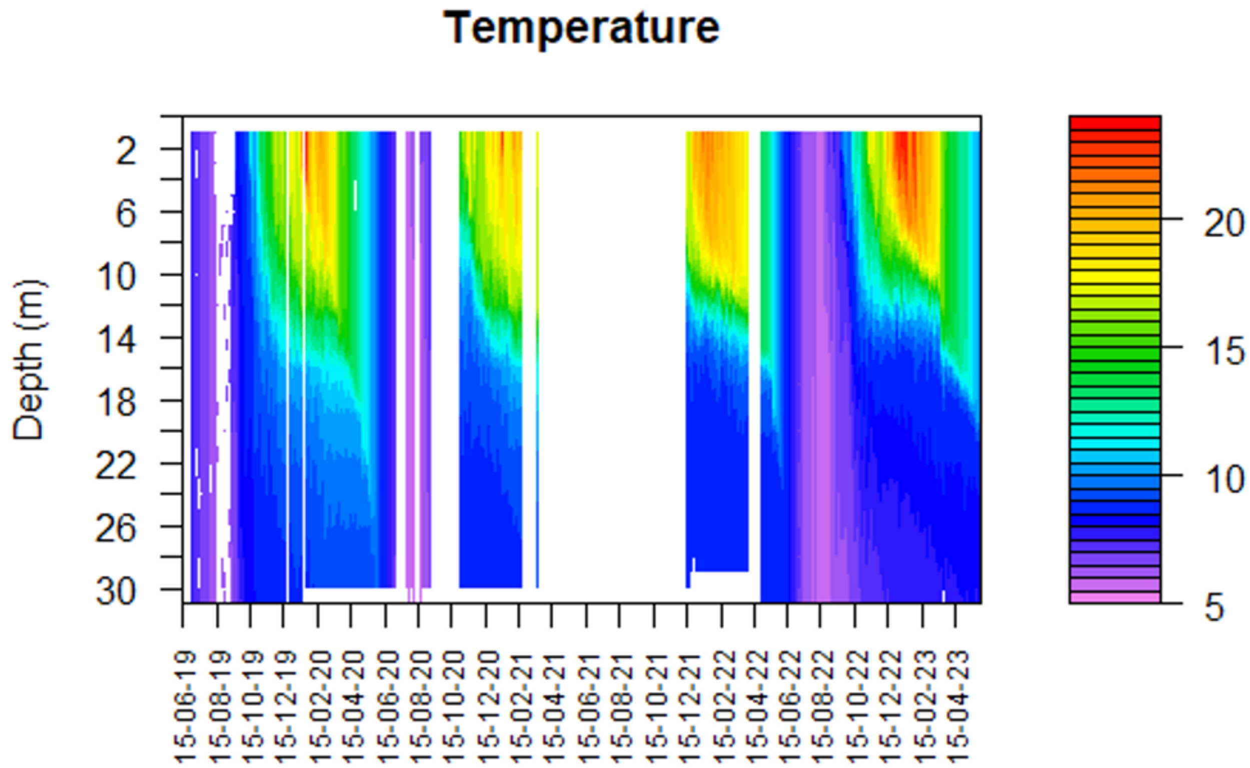

Figure 17. Temperature of Lake Hayes from July 2019 to June 2023 in degrees C as recorded by the lake monitoring buoy.

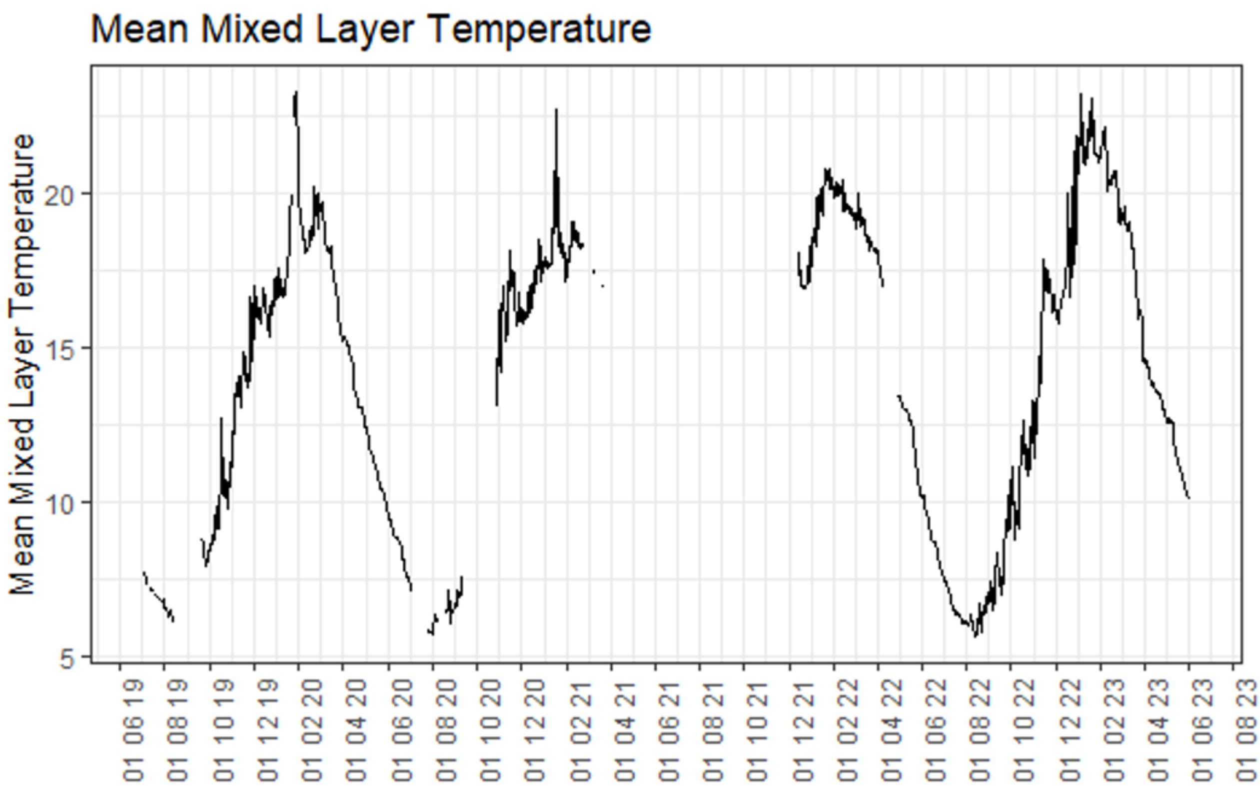

Figure 18. Daily temperature averaged throughout the mixed layer in degrees C from July 2019 to June 2023 as recorded by the lake monitoring buoy.

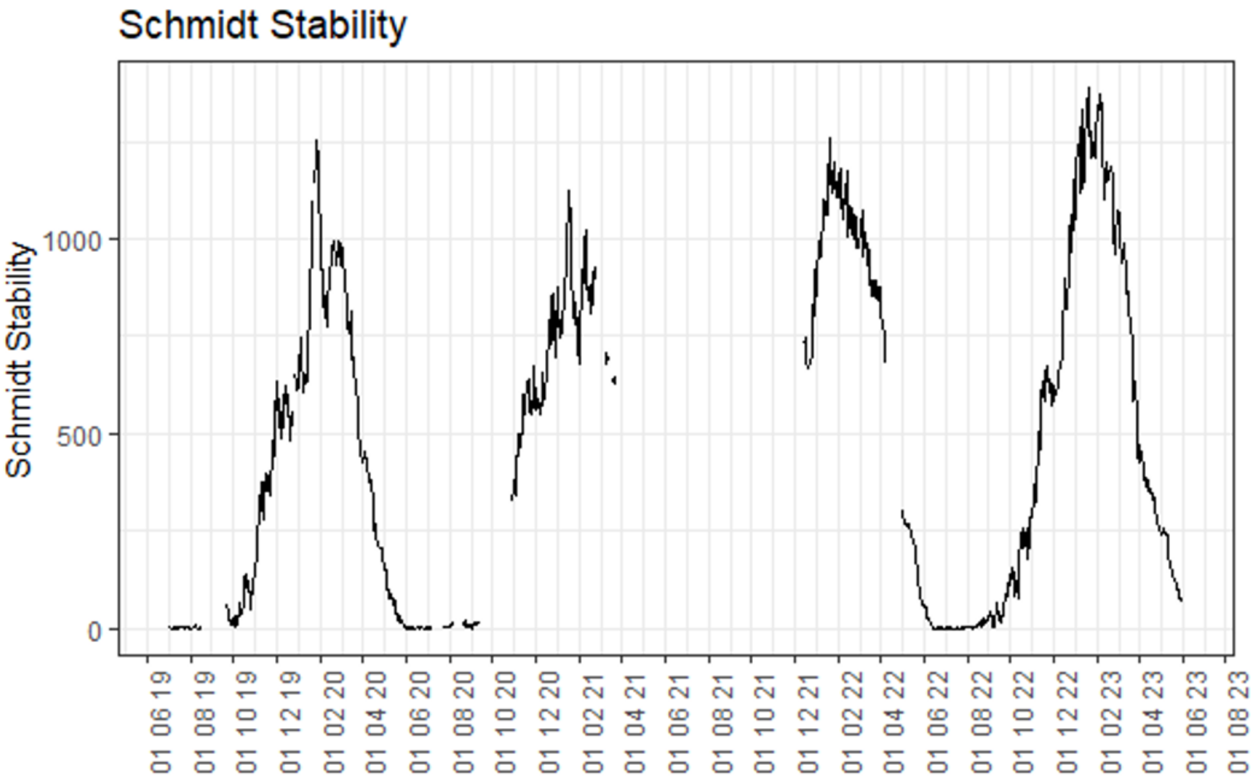

Figure 19. Schmidt stability (i.e. energy needed to mix the entire water column) from July 2019 to June 2023 calculated from temperature data recorded by the lake monitoring buoy.

Table 4. Dates of onset of stratification and water column turnover and the duration of seasonal stratification for the summers of 2019/20 and 2022/23. Calculated from the Schmidt stability (Fig. 17).

Date of onset	Date of turnover	Days of seasonal stratification
26 Oct 2019	13 May 2020	200
11 Oct 2022	31 May 2023	232

Table 5. Number of days when the epilimnion exceeded different temperature thresholds during four summers. Note that insufficient data were available to the summer of 2020/21 due to equipment malfunction.

	2019/20	2020/21	2021/22	2022/23
Temperature threshold	**Days exceeding threshold**	**Days exceeding threshold**	**Days exceeding threshold**	**Days exceeding threshold**
21˚C	9	n/a	14	45
22˚C	n/a	n/a	3	25
23˚C	3	n/a	0	7

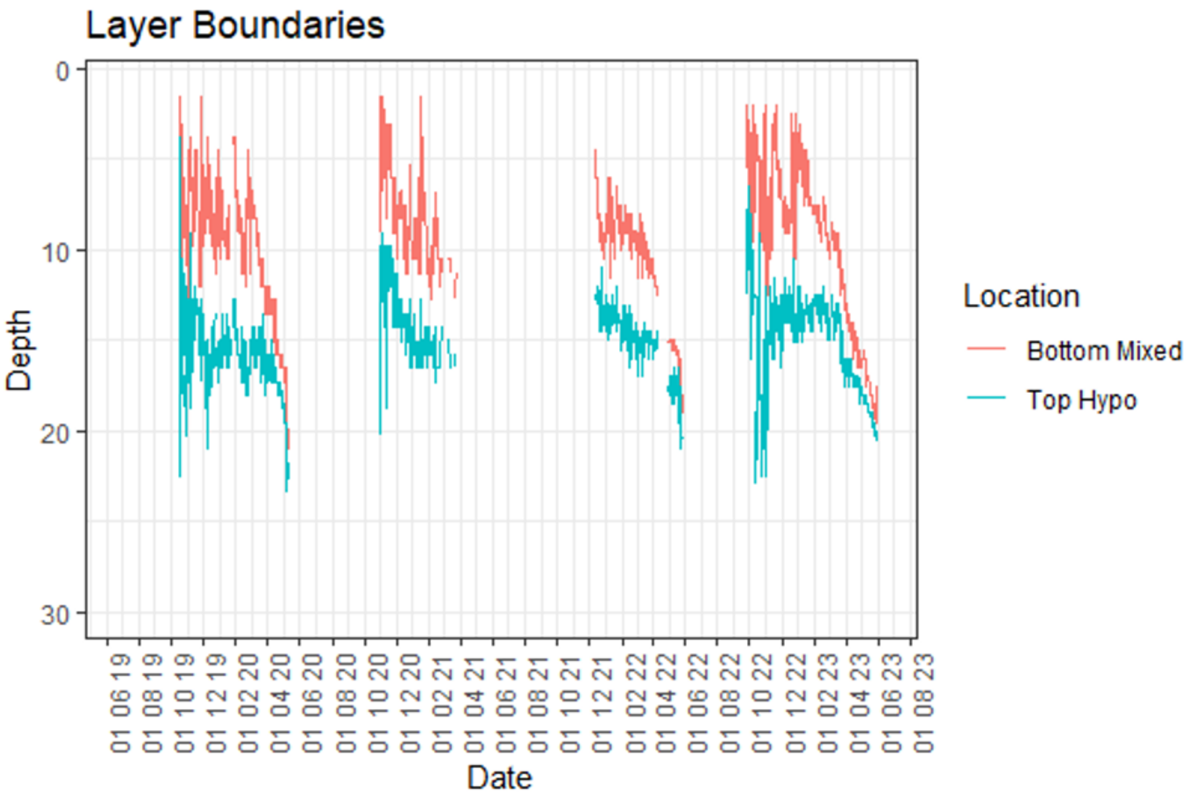

Figure 20. Calculations of the bottom of the epilimnion (mixed layer) and top of the hypolimnion during the stratified period.

6.3 Water clarity and phytoplankton

The trophic state of a lake is a composite measure which includes its degree of nutrient enrichment. Nutrient enrichment usually manifests as increases in algal biomass. In seasonally stratifying lakes, algal biomass is a major component of water clarity. Water clarity is the aspect of water quality most easily and commonly perceived by people engaging with lakes. Secchi disk depth and turbidity are both measures of water clarity. The Secchi disk measures transparency from the surface of the lake down to the Secchi disk, whereas turbidity measures the light-scattering property of lake water and can be measured in any parcel, or sample, of water. Suspended particles are the main components of lake water that influence both transparency and light scattering. In seasonally stratifying lakes like Lake Hayes, phytoplankton are likely to be the most common particles found in lake water at the mid-lake site, where effects of lake bed resuspension by wind and suspended sediment inputs from inflowing tributaries are likely to be negligible compared to the algal biomass in the water.

Fortunately, Secchi disk depth has been measured in Lake Hayes since the late 1940s thanks to the pioneering work of limnologist, Hilary Jolly (Jolly 1952). A dataset of summer Secchi disk readings assembled from various studies, reaching back to the late 1940s, shows how water clarity/transparency has changed in Lake Hayes since that time (Figure). The combined dataset shows that there was a substantial decline of approximately 2 m between the late 1940s and the 1970s, although the variability in Secchi disk readings remained roughly the same. On average, the summer water clarity dropped from around 5 m to around 4 m (a 20% decrease) over this time period and by the 1970s the lake was showing signs of eutrophication such as cyanobacterial blooms, anoxic hypolimnion, and significant internal phosphorus loading (Mitchell & Burns 1972; Burns & Mitchell 1974; Mitchell & Burns 1981). From the 1970s through to the 1990s and early 2000s, the average summer water clarity remained roughly constant, but the variability among individual measurements appeared to increase. By the 1990s, the summer water clarity was varying between just over 1 m to almost 7 m. By the early 2000s, the average water clarity was beginning to decline. However, the lake began a new dynamic since 2006, when the dinoglagellate, C*eratium hirundinella*, began to bloom in the lake. *Ceratium* blooms consistently reduced water clarity to an average of approximately 3m (range was approximately 1 m to 4.5 m), except in two summers (2010/11 and 2017/18) when *Ceratium* didn't bloom and water clarity averaged approximately 9 m (range was approximately 7.5 to just under 12 m).

Schallenberg & Schallenberg (2017) discussed this dynamic in detail and provided hypotheses as to how such a dynamic could develop in Lake Hayes. We will not reiterate that discussion here, but this bifurcating behaviour, in which the lake alternates between clear water years and highly turbid years, warrants further study as insights into the drivers of algal blooms in Lake Hayes could be revealed. Unfortunately, the lake was not monitored regularly in 2010/11, and so monitoring data are only available for one clear water year, 2017/18. With a regular monitoring programme in place, better data will be available should the lake have another clear water summer.

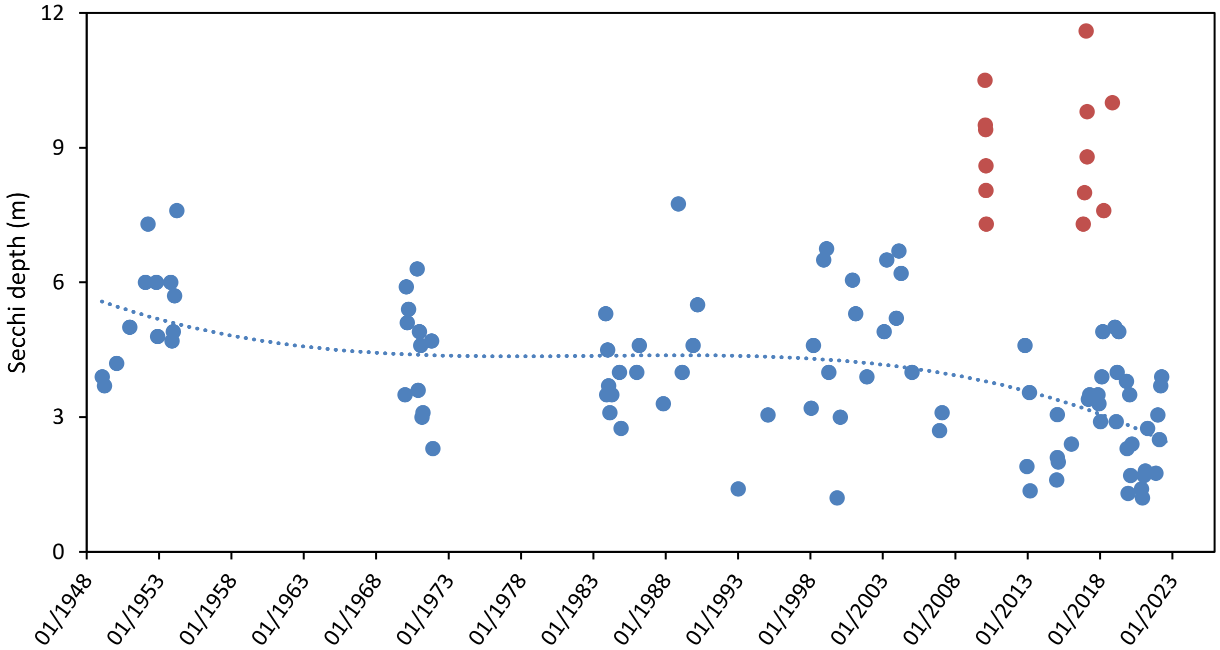

Figure 21: Historical summer (November to April) water clarity (Secchi disk depth) measurements in the open waters of Lake Hayes. Blue dot Secchi data from 1984-2015 are from unpublished Otago Regional Council (ORC) data, M. Schallenberg unpublished data, and Caruso (2001). Red dots are from M. Schallenberg, unpublished data (2011) and from the ORC (2017/18).

Turbidity is a measure of particulate material in the water that has a standard value of 5 NTU (80th percentile), as is listed in Schedule 15 of the Otago Water Plan (ORC 2014). Figure 22 shows the turbidity of the lake as measured by the lake monitoring buoy. While turbidity did occasionally exceed the standard value in 2021/22 and 2022/23 (Table 6), the 80th percentile turbidity is below the ORC standard.

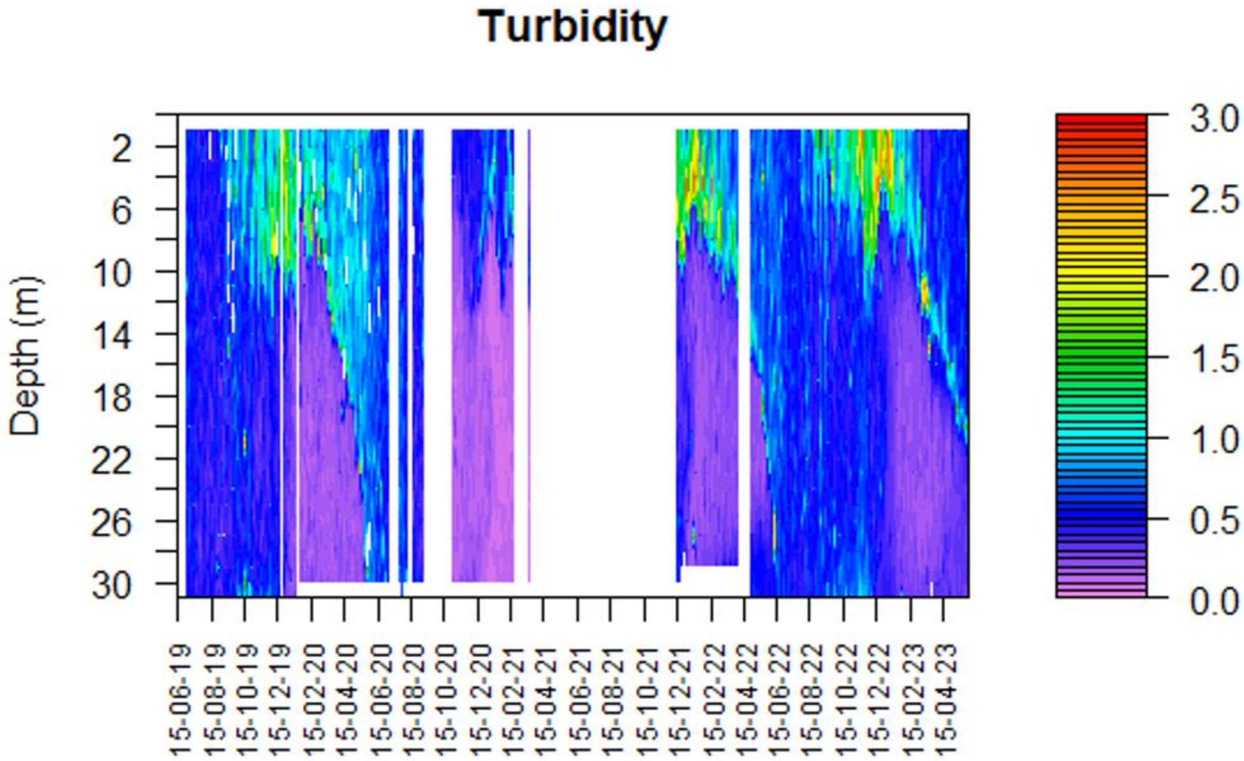

Figure 22. Turbidity in Lake Hayes as measured by the lake monitoring buoy.

Table 6. Maximum and median turbidity values from the lake monitoring buoy as well as the number of days when turbidity exceeded the 5 NTU standard for the four years of monitoring.

	2019/20	**2020/21**	**2021/22**	**2022/23**
Maximum NTU	3.0	2.5*	10.8*	8.3
Median NTU	0.5	0.2*	0.5*	0.5
Days > 5 NTU	0	0*	8*	5

*large data gaps exist for these years. Therefore, the data are approximate.

Chlorophyll *a* is a measure of algal biomass based on the concentration or fluorescence of the main photosynthetic pigment in algal cells. The ORC SOE report (ORC 2021) indicates that chlorophyll *a* at the mid-lake site breaches the national bottom line in the NOF (Appendix C), meaning that chlorophyll *a* must be improved, ideally through a limit-setting process as described in the NOF (MfE 2020).

The Lake Hayes monitoring buoy measures fluorescence of chlorophyll *a* in the lake water (Fig. 23) and this together with the temperature data, make it possible to calculate the average chlorophyll *a* concentration in the mixed layer (Table 6). The *in vivo* (in live cells) fluorescence measured by the buoy gives an indication of the chlorophyll *a* concentration but is just an estimate of the concentration measured by the standard chlorophyll a methodology (i.e., measured in the lab after chemical extraction of the chlorophyll *a* from a concentrated sample of lake algae) - the method that the NOF standard has been developed from. Thus, the lake buoy chlorophyll *a* readings shouldn't be used to assess lake chlorophyll *a* in relation to the standard. However, the *in vivo* fluorescence data from the buoy provide excellent depth and temporal resolution of chlorophyll *a* concentration, which improve understanding of phytoplankton dynamics in the lake.

As with temperature and turbidity, the chlorophyll *a* readings from 2022/23 were much higher than those for the previous three years indicating that the summer of 2022/23 had usually high algal biomasses, turbidities and water temperatures (Figures 23 and 24). The lake buoy data (we used readings measured at night) clearly show strong vertical structuring of algal biomass, with peaks in biomass often occurring near the thermocline, especially in spring and early summer of 2019/20 and 2022/23, but also in autumn of 2022/23 (Figure 23). This is a well-known feature of some algae, including the motile *Ceratium*, which may descend to the thermocline to obtain nutrients. During calm days in summer, dense, brown patches of *Ceratium* can often be seen at the surface of the lake, where it can benefit from high amounts of solar radiation (Schallenberg & Schallenberg 2017). The strong vertical structuring of chlorophyll *a* concentrations during summer in this lake highlight the importance of measuring chlorophyll *a* at many depths (i.e., at short depth intervals) to accurately estimate the amount of phytoplankton biomass in the lake at any particular time. This vertical structuring also indicates that the relationship between Secchi depth and phytoplankton biomass may be weaker during the summer and autumn, when algae may be concentrated at depth in the lake.

The chlorophyll *a* data in Figure 23 shows elevated levels of chlorophyll *a* in the hypolimnion during spring and early summer, which is particularly evident in 2022. This represents algae which have dropped out of the mixed layer and are settling to the bottom. These hypolimnetic peaks are usually associated with high algal biomass in the mixed layer. The sedimenting algae decompose in the hypolimnion, contributing to the development of summer hypolimnetic anoxia.

Another interesting feature in the data in Figure 23 is the elevated chlorophyll *a* in the water column of the lake during the isothermal period of 2022 (June to August), when the whole water column was mixed to the bottom. This contrasts with the much lower phytoplankton biomass in the winter of 2019. Elevated winter phytoplankton biomass could result in higher phytoplankton biomass in the subsequent summer as the phytoplankton biomass winter baseline is higher going into the growing season. Why chlorophyll *a* concentrations in winter are so variable from year to year is a question worth investigating further.

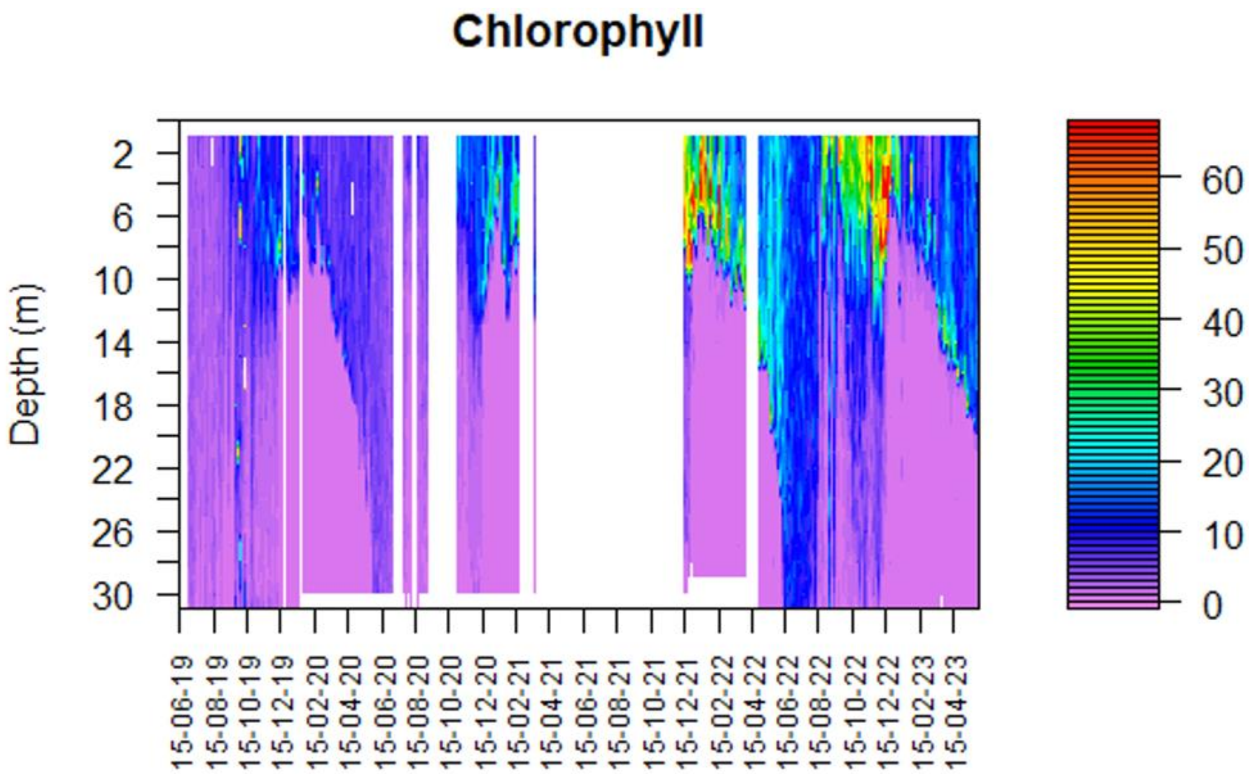

Figure 23. Chlorophyll *a* profiles measured by the lake monitoring buoy. The units are in vivo fluorescence units scaled to give an approximate estimate of chlorophyll *a* concentration in µg L^{-1}.

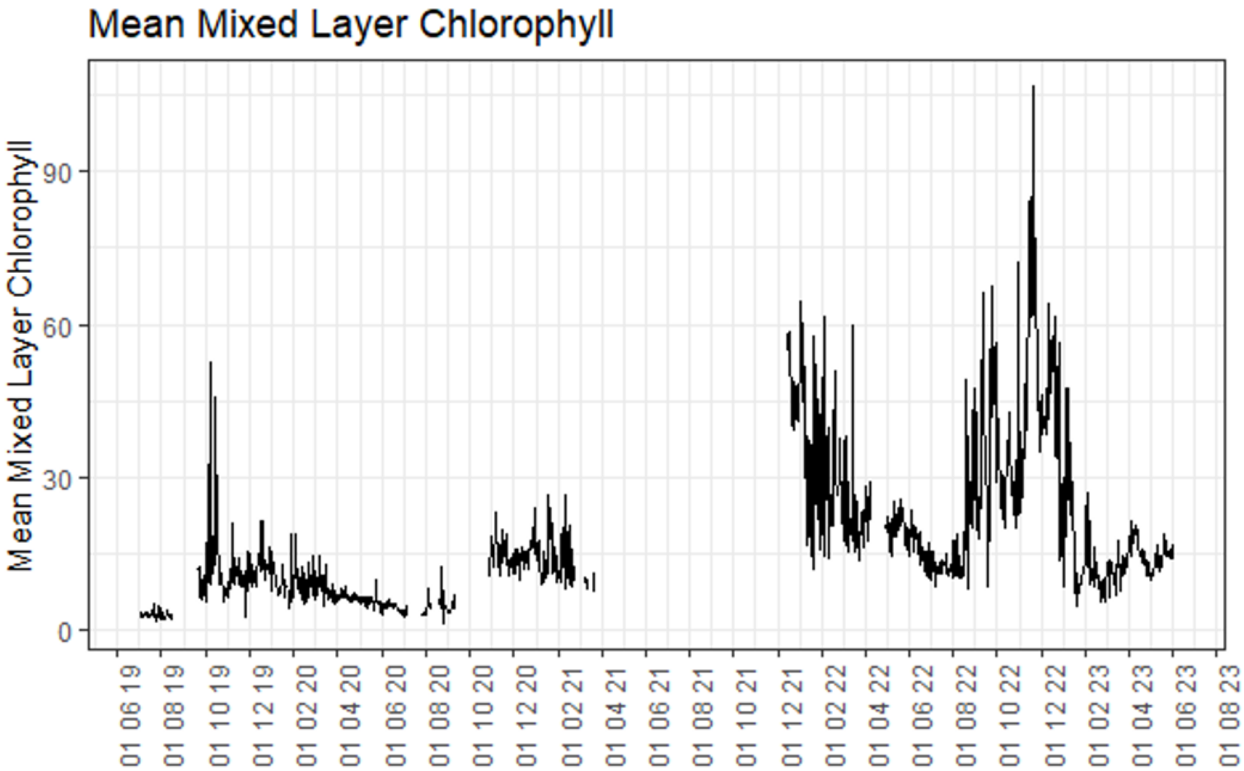

Figure 24. Average (mean) **c**hlorophyll *a* value in the mixed layer. Calculated from profiles measured by the lake monitoring buoy. The units are in vivo fluorescence units, scaled to give an approximate estimate of chlorophyll *a* concentration in µg L^{-1}.

The temperature data together with the chlorophyll *a* data enable the calculation of time-integrated maximum and median chlorophyll *a* concentrations in different parcels of lake water. Maxima and medians for the mixed layer and the maxima for the entire water column are shown for the four years of monitoring data in Table 7. In 2019/20 and 2022/23 the maxima occurred in the thermocline, below the mixed layer. As expected from the data in Figures 22 and 23, the chlorophyll *a* concentrations in Table 7 were much higher in 2022/23 than in 2019/20.

Table 7. Chlorophyll *a* maxima and medians for different layers of water. Calculations are based on chlorophyll *a* and temperature readings from the lake monitoring buoy. The units are in vivo fluorescence units, scaled to give an approximate estimate of chlorophyll *a* concentration in µg L^{-1}.

	2019/20	**2020/21**	**2021/22**	**2022/23**
Maximum in mixed layer	49	107*	197*	163
Median in the mixed layer	8	13*	22*	15
Maximum in the water column	139	107*	197*	177

*large data gaps exist for these years. Therefore, the data are approximate.

6.4 Bottom water dissolved Oxygen

As discussed in Section 7.2, the hypolimnion of Lake Hayes has undergone de-oxygenation since the 1970s, and probably since the 1960s, when water quality was first reported to have deteriorated. De-oxygenation of the cold, hypolimnetic waters in the summer precludes this cool water habitat from being used by biota, such as brown trout, which are sensitive to temperatures above around 20°C. In effect, heating of surface water together with deoxygenation of bottom water limits the habitat for most aquatic animals and plants to the warm shallow surface water layer during summer. At this time, the majority of the lake's volume and its cooler waters become uninhabitable to temperature-sensitive fish and other organisms. Zones like this in polluted marine habitats, such as off the Mississippi delta in the Gulf of Mexico, are known as "dead zones". Thus, Lake Hayes has exhibited substantial summer dead zones since at least the 1970s.

Dead zones occur when the rate of oxygen depletion exceeds the rate of oxygen replenishment, eventually leading to hypoxia (reduced dissolved oxygen concentrations) and eventually anoxia (an anaerobic or de-oxygenated environment). Once seasonal stratification develops in Lake Hayes (usually in October; see Table 3) the cooler, denser bottom water layer loses its ability to exchange gases with the atmosphere. This cuts off the external oxygen supply and unless the water is very clear, allowing for photosynthesis (oxygen production) below the thermocline, the oxygen in the hypolimnion will decrease due to the microbial decomposition of organic matter in the water column (sedimenting algae) and sediments. The more organic matter that is available in the hypolimnion, the more oxygen will become depleted.

The dissolved oxygen data measured by the lake buoy shows the development of the anoxic bottom waters that begins soon after seasonal stratification occurs (Figures 25 and 26). The rate of oxygen depletion is rapid, with the first signs of anoxia occurring above the sediment in November, at which time roughly half the oxygen in the hypolimnion has been used up (Figure 26). This appears to be driven mainly by algae settling out from the spring bloom, but other types of organic matter deposited on the lake-bed (e.g., willow leaves, dead zooplankton, organic sediments from the catchment) will also contribute to deoxygenation.

As long as the water clarity and ability of light to penetrate into the lake is restricted by turbidity, there won't be enough light for significant photosynthesis by algae and submerged plants to occur in the upper hypolimnion and deoxygenation will proceed until the entire hypolimnion is anoxic. This occurred in December 2022/January 2023, when the lake was anoxic from the bottom to approximately 5m depth, a volume of lake water equivalent to 77% of the total volume of Lake Hayes. A general rule of thumb is that plants can carry out enough photosynthesis to sustain themselves down to a depth at which 1% of the light reaching the surface of the lake can penetrate. At depths and light levels lower than this, most plants won't grow. The depth limit for plant growth (the 1% light level) is approximately 2 × the Secchi disk depth. Therefore, in years when turbidity is high in summer and the resulting Secchi disk depth (Figure 21) is less than half the mixed layer depth (Figure 20), plants including most algae will not be able to grow vigorously in the hypolimnion and will not be able to supply oxygen to the hypolimnion. In contrast, during clear water years when the Secchi disk depth can exceed 10 m (Figure 21), photosynthesis can potentially occur to a depth of 20 m, likely alleviating oxygen depletion of the hypolimnion to some extent.

It appears that the greater intensity of deoxygenation observed in the summer of 2022/23 is likely related to the lack of complete re-aeration of the lake in the winter of 2022 (Figure 26). It appears that the early stratification of the lake in the spring of 2022 prevented re-equilibration of the gases dissolved in the lake water with the atmosphere and the lake only achieved barely 70% oxygenation during the period of mixis. Any future climatic changes that result in earlier onsets of stratification will exacerbate deoxygenation problems during the summer months.

The other major impact of hypolimnetic anoxia is that metal oxyhydroxide minerals commonly found in lake sediments dissolve under low reduction-oxydation potentials (i.e., when dissolved oxygen is depleted). In well oxygenated sediment, the minerals bind P, whereas under hypoxic and anoxic conditions the minerals dissolve, releasing phosphorus into the sediment porewater, which diffuses into the hypolimnion. As thermal stratification breaks down in autumn, this phosphorus is mixed into the upper water column where it can be used by algae to photosynthesise and grow, in the presence of light. Thus, hypolimnetic dissolved oxygen is a doubly key element in Lake Hayes, determining whether the cooler hypolimnetic waters are available to plants and animals during summer and also mediating internal phosphorus loading to the lake.

The linked processes described above constitute one of the many feedback systems that operate in Lake Hayes. Figure 27 is a simplified feedback model linking algal growth, light penetration (water transparency), dissolved oxygen production, and hypolimnetic phosphorus release. The recognition of such feedbacks reveal potential interventions that could break the feedback cycle.

The NOF includes two dissolved oxygen standards for lakes: (1) a standard for minimum mid-hypolimnetic dissolved oxygen concentration (national bottom line is 4.0 mg L^{-1}) to protect cool water habitats in lakes, and (2) a standard for minimum oxygen concentration above the lake-bed sediment (national bottom line is 0.5 mg L^{-1}) to protect the phosphorus binding potential of the lake bed sediments. The lake buoy data show that in Lake Hayes the entire hypolimnion becomes anoxic and, therefore, breaches both of the NOF dissolved oxygen national bottom line standard for lakes.

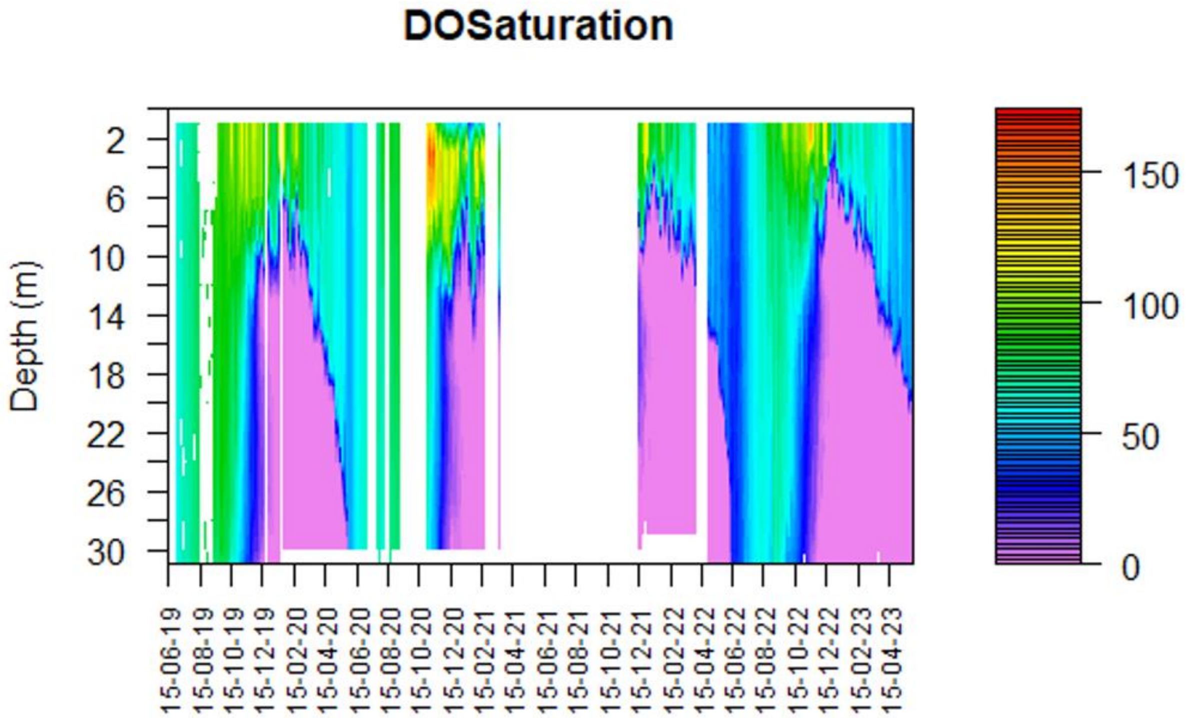

Figure 25. Dissolved oxygen, measured as percent of saturation, by the lake monitoring buoy.

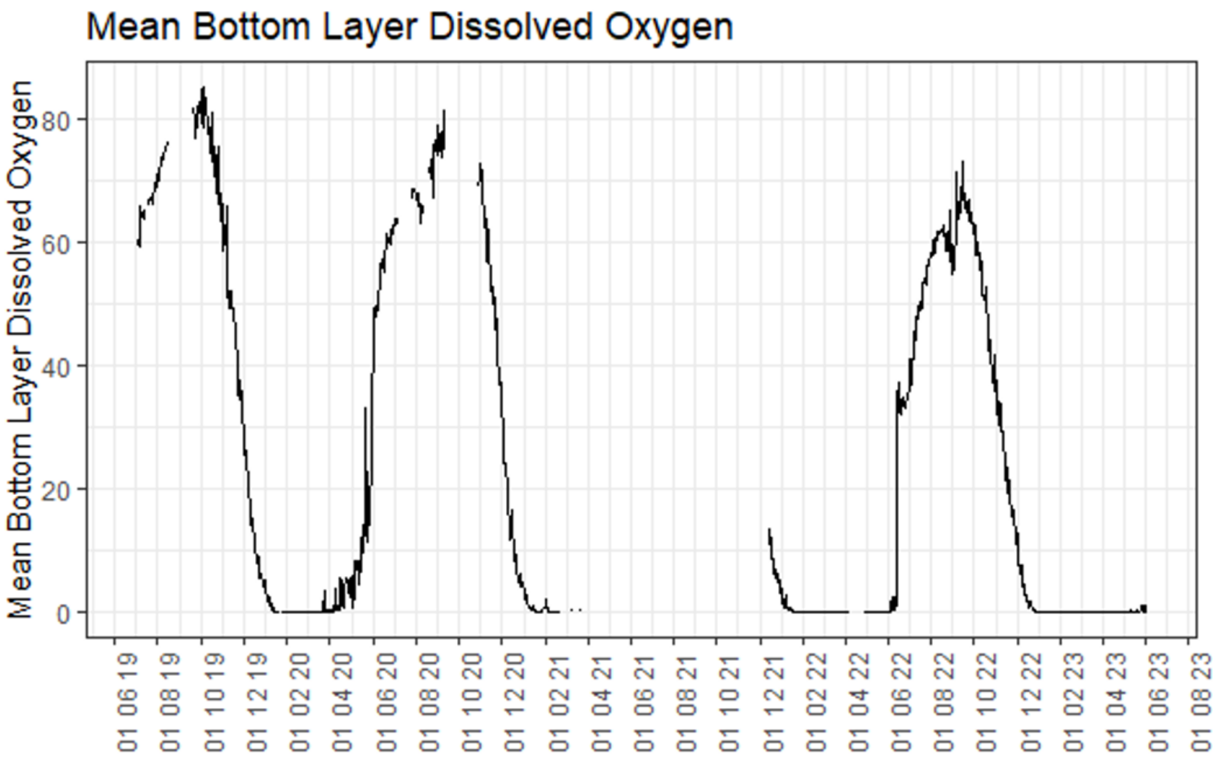

Figure 26. Calculated average (mean) bottom-water dissolved oxygen percent saturation.

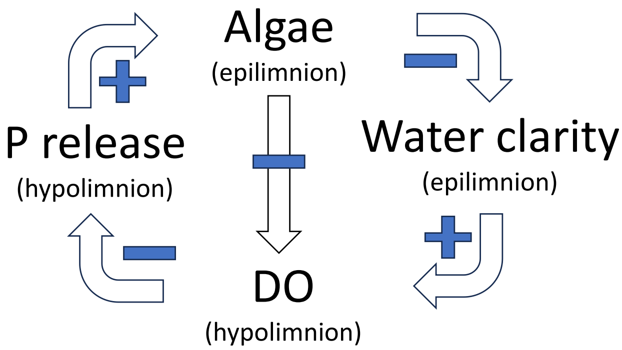

Figure 27. A simple feedback system describing how algae growing in the epilimnion can induce a feedback that facilitates algal proliferation. + and – indicate positive and negative interactions respectively, in the direction of the arrows. The recognition of such feedbacks reveals possibilities for interventions that can break the feedback cycle.

6.5 Nitrogen, phosphorus and trophic state

Total nitrogen (TN) and total phosphorus (TP) are NOF attributes for lakes (MfE 2020). The ORCs SOE report indicated that TP was in the C-band (fair) and TN was in the B-band (good) (ORC 2021). However, TP breaches the ORCs water plan limit for Lake Hayes, while TN and nitrate concentrations do not breach the regional water plan standards (ORC 2020). Thus, according to the standards, phosphorus concentrations are a concern for Lake Hayes.

Concentrations of TP (at 25 m and at 10 m depth) and dissolved reactive P (DRP at 25 m depth) from 2016 to 2023 are shown in Figure 28. The data from the bottom water show strong annual cycles of P concentrations with peaks occurring in March to May, prior to turnover, and minima occurring after the onset of stratification, from October or November through to December (Figure 28b and c). The data show that the majority (generally around 80%) of P in the bottom water during the stratified period is in the form of DRP. The seasonal pattern of phosphorus concentrations in the bottom water is clearly driven by anoxic P release from the sediment, which provides a large pulse of P to the water column in late summer and autumn. The data from 2017 to 2023 show a evdience of a slightly increasing linear trend in the average bottom water P concentrations for both DRP and total P.

The data from 10 m depth show more muted seasonal variations, with some peaks occuring in September and October, when bottom water concentrations tend to be near the annual minima. Other peaks at 10 m occur in June, at the time of turnover, but these temporal patterns are not evident in every year. On average, the total P concentrations at 10 m were similar to those in the bottom waters. However, from 2017 to 2023, there was an apparent rise and then decline in P at 10 m – a pattern that contrasts with the slight monotonic rise in P in the bottom waters of that time period. While the concentrations of P at the two different depths do influence each other, there are other factors that cause the phosphorus dynamics at the two different depth to differ.

The bottom water peaks will be influenced by sediment P release, which in turn is influenced by decomposition of previous algal blooms and the mineralisation of particulate phosphorus inputs from Mill Creek (both current and historical catchment loads). The release of P from these sources is mediated by anoxia, which is influenced by the duration of stratification and by the dissolved oxygen budget of the hypolimnion. All of these factors combined to result in slightly increasing average P concentrations at 25 m over the past 6 years. In contrast, the P concentrations at 10 m will be mainly influenced by phosphorus inflows from Mill Creek and by the *Ceratium* P pump, whereby the motile *Ceratium* translocates P from the hypolimnion to the mixed layer (Schallenberg & Schallenberg 2017). *Ceratium* was conspicuously in low abundance in the lake in the summers of 2017/2018 (a clear water summer; Fig. 21) and again in 2022/23 (M. Hanff, personal observation), potentially driving the unimodal relationship in total P observed over the past 6 years at 10 m (Fig. 28 a). A large flood in Mill Creek in August 2020 delivered over 40% of the annual P load to the lake, but does not appear to have immediately influenced P concentrations at the mid-lake site. However, in December 2019, a large peak of total P was observed at 10 m and, at this time, P at 25 m did not decline as much as in other years, indicating that the P "event" at 10 m was also influence P in the bottom waters, though to a lesser extent. It's not clear what the source of this significant P input in December 2019 was, but, the fact that it occurred 4 months after a large input of P from Mill Creek occurred suggest that the flood may be associated with the the significant spike in P at the mid-lake site 4 months later.

Lake Hayes is generally isothermal (i.e. fully mixed) from June to August. Thus, the concentration of P is similar thoughout the water column at this time. The nutrient concentrations in the lake during the isothermal period reflect the stock of nutrient available to algae in the lake at the beginning of the stratified period and the beginning of the spring and summer growing season. In Figure 28c, the P concentrations during the isothermal periods are were extracted from the dataset, revealing a similar unimodal pattern to the TP data at 10 m depth and contrasting with the multi-year trend observed in the full P datasets from 25 m depth. If the P available in the water column during the isothermal period is related to the size of subsequent algal blooms then Figure 28d suggests that floods may have a persistent effect on nutrient availability and algal blooms many months after the flood occurred.

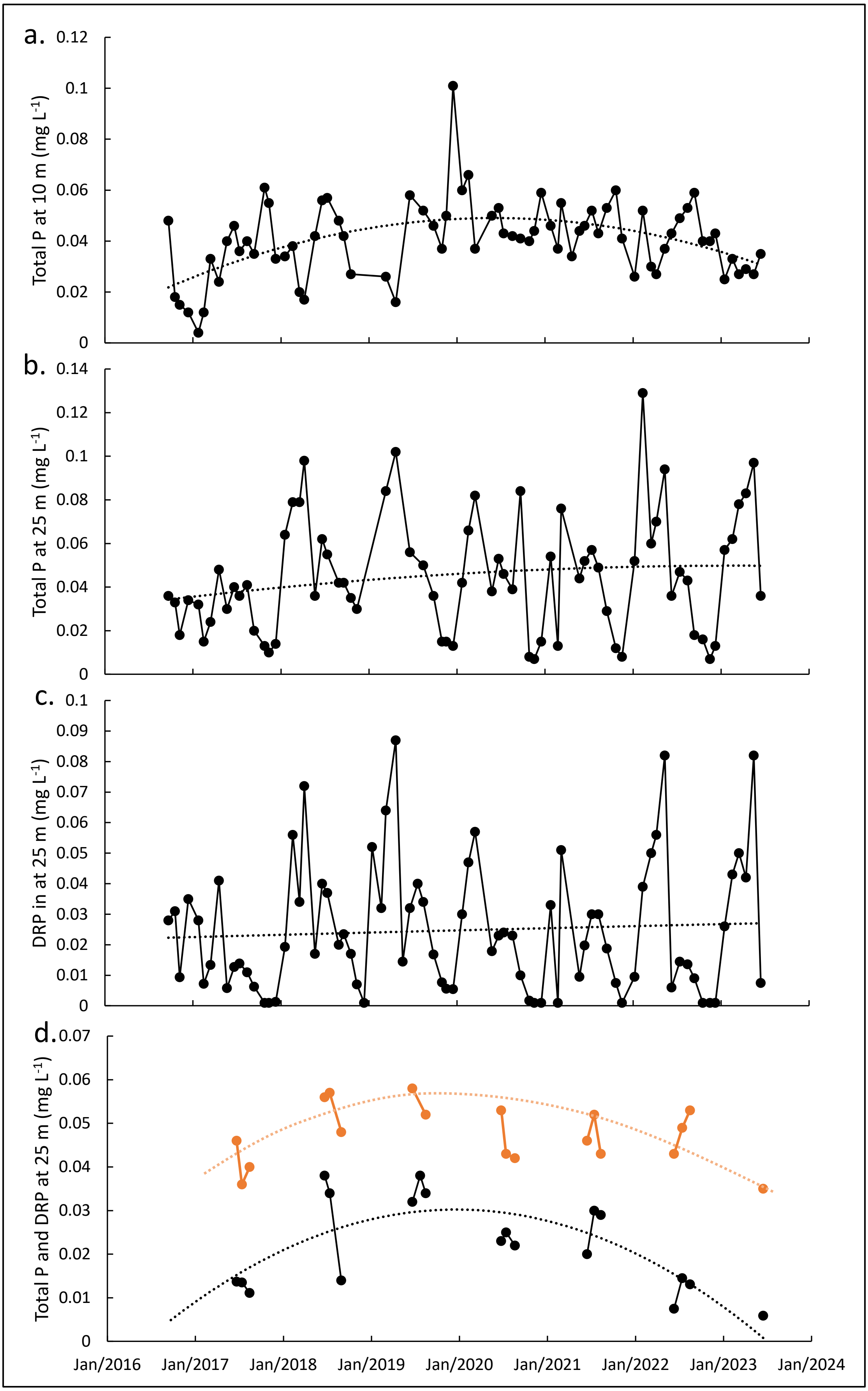

Figure 28. Concentrations of total phosphorus at 10 m (a.) and bottom water (25 m) concentrations of total phosphorus (b.), dissolved reactive phosphorus (c.), and total phosphorus (orange) and dissolved reactive phosphorus (black) in the months of June, July and August (i.e., the isothermal period) [d.] in Lake Hayes from 2016 to 2022. Lines are second-order polynomials fitted by least squares regression.

Internal loads can be estimated more accurately with the use of phosphorus depth profiles collected regularly over the stratified period. Such profiles are available for four years: 1983/84 and 1984/85 (Robertson 1988), 1994/95 (M. Schallenberg, unpublished data), and 2012/13 (M. Schallenberg, unpublished data). These data were linearly interpolated to estimate total phosphorus concentrations at each metre depth. These profiles were then multiplied by the volume of water at each metre depth using a hypsographic curve (M. Schallenberg, unpublished data), allowing the estimation of the phosphorus contained in each 1 m horizontal slice of lake water. Setting the top of the hypolimnion at 13 m depth allowed the summation of the phosphorus in the lake from 13 m to the bottom for each P profile. The summed hypolimnetic P burden for each profile was then plotted against days post December 1 for each profile sampling date (i.e., days since December 1) to allow for a calculation of the rate of P augmentation of the hypolimnion during the period when P is released from the sediments.

The raw data used in this analysis of internal P loading is shown in Figure 29. The rise of P in the hypolimnion in each of the summers is shown in Figure 30a. The relationship for 2012/13 is quite linear (R^2 = 0.92) whereas the releases of P in 1983/84 , 1984/85, and 1994/95 showed slower initial release rates and then very rapid releases near the end of summer (R^2 = 0.79, 0.80, and 0.57, respectively). On a linear basis, the release rates varied between 10.2 and 14.2 kg P d^{-1}, although by the end of summer, the release rates of P into the hypolimnion were quite similar in the different years. Thus, it's likely that the overall amounts released during the stratified period varied less than the summer-long release rates in the different years.

It's not clear which factors influenced the different P release rates in the different years, although different dates of the onset of stratification may have played a role. In addition, the influence of floods in advance of the stratified period may have resulted in differences in available organic matter and/or sediment-bound P in the different years. The P release relationships in Figure 30a do not show a clear trend in slope or intercept over time. Thus, it appears that internal P loads are more-or-less at equilibrium with the external loads to the lake at this time. However more data is needed to determine whether large differences in external load from year-to-year (e.g., due to unusual flood events) eventually lead to years with higher internal loads.

Although nitrogen species don't exhibit the same kind of particle adsorption/desorption dynamics as some phosphorus species do, ammonium and dissolved organic N could accumulate in the hypolimnion under anoxic conditions. We calculated the rates of change of hypolimnetic N in the years for which we could obtain total nitrogen profiles (Figure 30b). In the summer of 2012/13, the expected increase in TN was observed with a linear release rate of 26 kg N d^{-1} (R^2 = 0.73). While this rate is roughly double the rate of P release into the hypolimnion, the ratio of N:P release is far below the Redfield ratio (the ratio in which N and P are utilised by algae), which is approximately 7 (by mass). Therefore, in 2012/13, the release of nutrients into the hypolimnion provided a large excess of P over N, in relation to of the demands of phytoplankton. At turnover this would have provided a nutrient stimulus at an N:P ratio of approximately 4, resulting in an N-limited phytoplankton community. This is not to suggest that the growth of phytoplankton in Lake Hayes is N-limited, only that the internal load of nutrients provides a surplus of P in relations to the N:P demands of phytoplankton.

In 1994/95, there was no accumulation of N in the hypolimnion. In contrast to 2012/13, TN in the hypolimnion decreased from an initially much higher amount at the start of summer. It's not clear what the source of this much higher N was that occurred in the lake at the start of the summer. Apparently, much of this was denitrified as hypolimnetic oxygen depletion progressed during the summer.

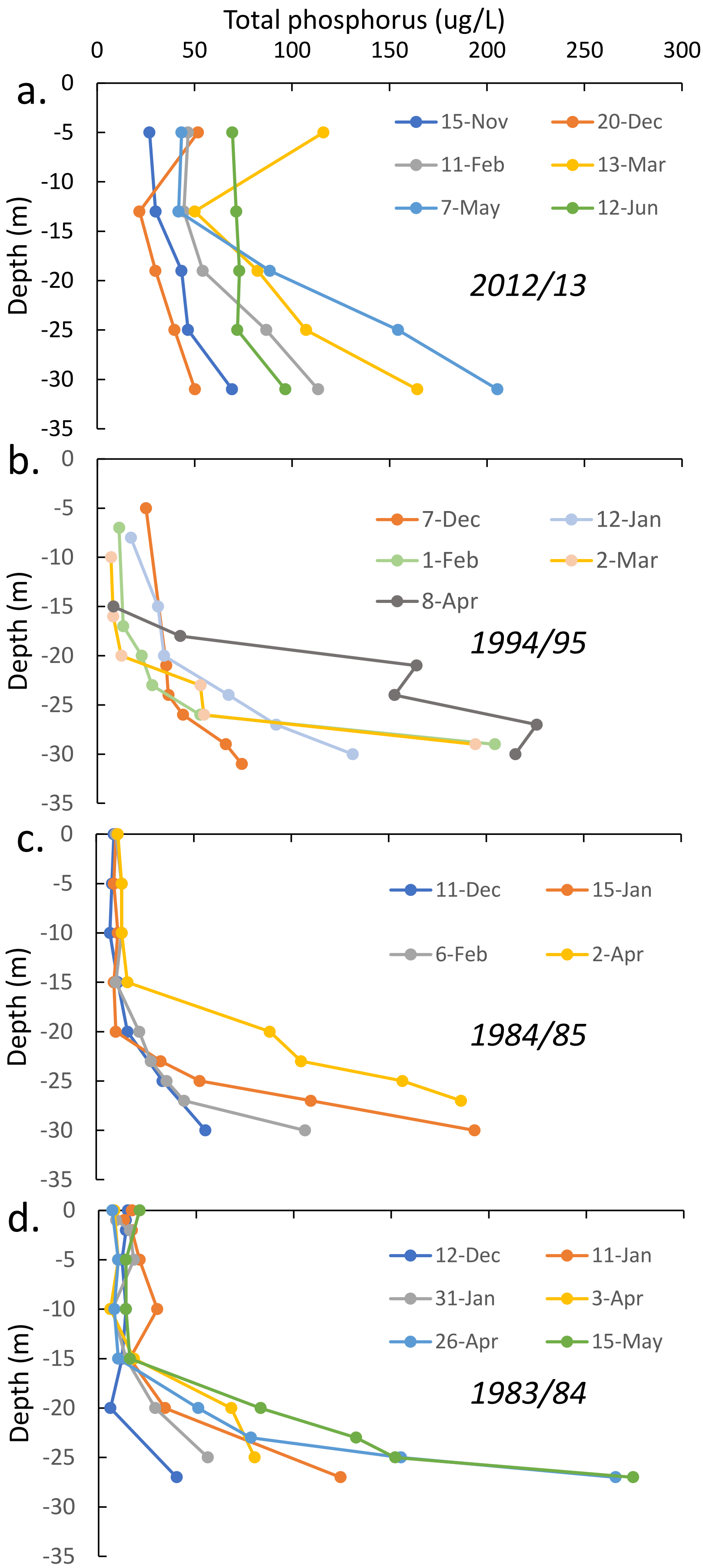

Figure 29. Total phosphorus depth profiles used to estimate internal P load. Data from 1983/84 and 1984/85 are from Robertson (1988) and data from 1994/95 and 2012/13 are from M. Schallenberg, unpublished data.

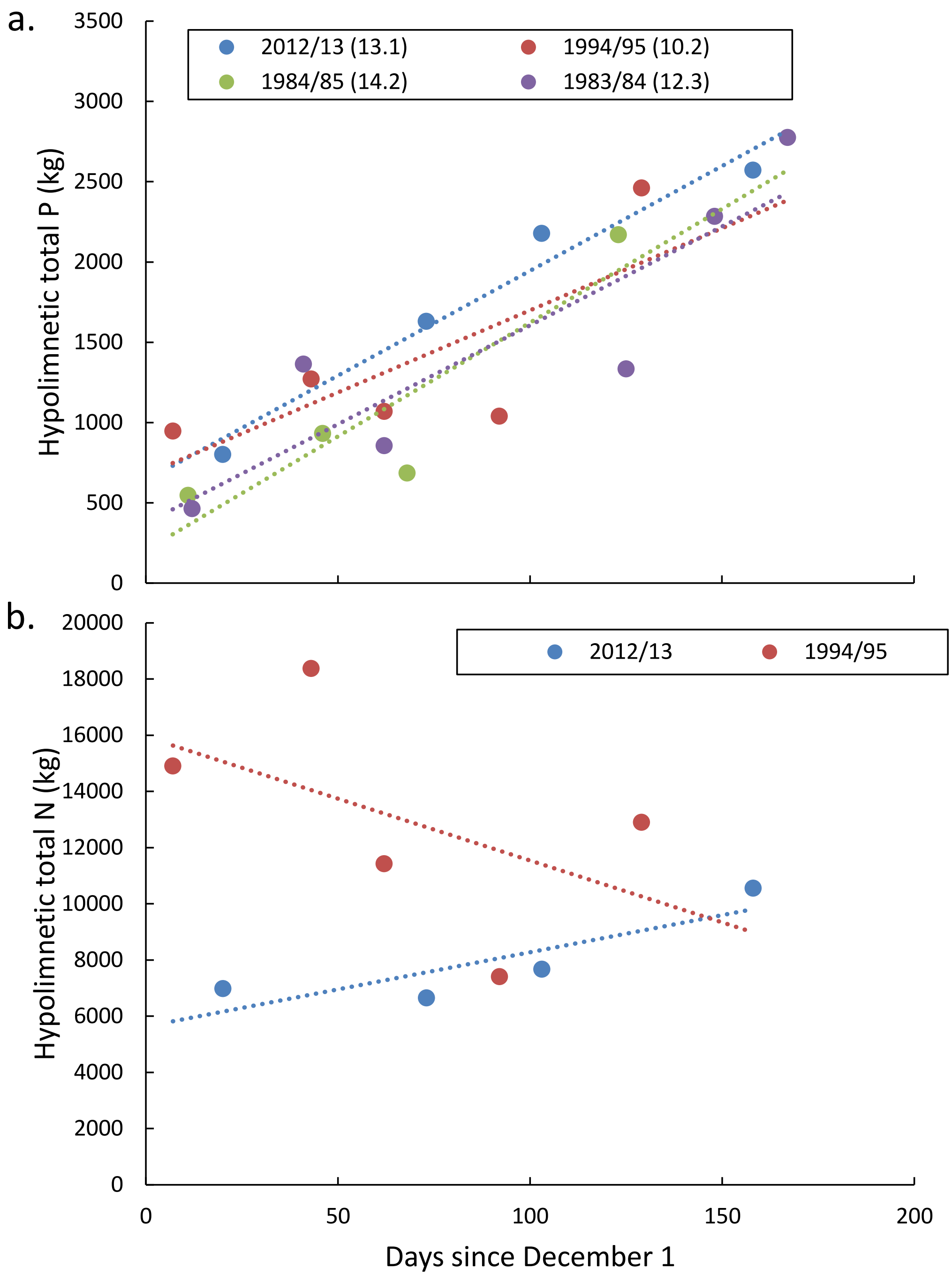

Figure 30. Development of the internal hypolimnetic phosphorus load (a.) and changes in hypolimnetic total N concentrations (b.) in summers for which nutrient depth profiles exist. The lines are least squares linear regressions. The numbers in parentheses in the legend are the linear slopes in kg d^{-1}. Data from 1983/84 and 1984/85 are from Robertson (1988) and data from 1994/95 and 2012/13 are from M. Schallenberg, unpublished data.

Nitrogen and phosphorus are important attributes of water quality because their availability is often related to the algal biomass and to algal blooms, anoxia, and further degradation of important lake values such as fisheries, recreation, and scenic values. This generalised trajectory of degradation can be described as a trajectory of nutrient enrichment effects, which has been defined as the concept of the trophic state of lakes. This normative concept is highly intuitive to lake users and lake managers and has prompted the development of indices to quantify the trophic state of lakes. These in turn have become so popular that environmental authorities often monitor and report on lake trophic state as a measure of lake health. To a lake, there is no relevant concept of lake health. But to humans, a lake that provides lots of ecosystem services and is perceived to be of high value is identified as a healthy lake. To humans, healthy lakes generally have a low trophic state (i.e., few symptoms of nutrient enrichment). As lakes become "enriched", symptoms of nutrient enrichment appear, indicating that the lake has a high trophic state.

Trophic Level Index (TLI) was developed by Noel Burns and colleagues as an index indicating nutrient enrichment in New Zealand lakes (Burns et al. 2000) and this index has been used by the Ministry for the Environment and Regional Councils to monitor, and report on, "lake health" (e.g., https://www.lawa.org.nz/learn/factsheets/lake-trophic-level-index/). The TLI is calculated from four, equally-weighted water quality attributes: (1) total nitrogen, (2) total phosphorus, (3) Secchi disk depth, and (4) chlorophyll *a*. Each unit of the trophic level scale corresponds to a class of trophic state from pristine-like (ultra-microtrophic; TLI score of < 1) to highly enriched and degraded (hypertrophic; TLI score of > 6).

The TLI for Lake Hayes is reported on the LAWA website (www.lawa.org.nz), showing a gradual rise since 2017 when it was near the mesotrophic-eutrophic boundary (TLI = 4; Figure 31). By 2021, the lake was approaching the eutrophic-supertrophic boundary (TLI = 5). In the summer of 2017 and spring of 2018, Lake Hayes exhibited very large Secchi disc depths (Figure 21) and no algal blooms, but unfortunately the condition of the lake has deteriorated since then. TLI data on the LAWA website have not been updated since 2021.

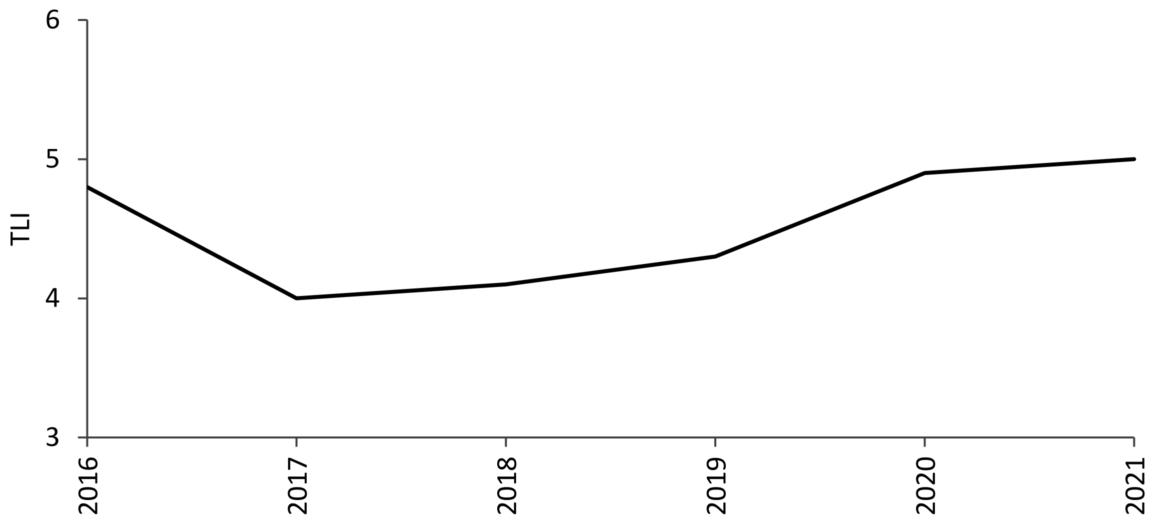

Figure 5. Lake Hayes Trophic Level Index (TLI) score over 5 hydrological years, 2016 – 2021. Dates indicate beginning of hydrological year (July - June). A TLI score of 3 to 4 indicates mesotrophic (i.e., "good" water quality), 4 to 5 indicates eutrophic (i.e., "poor" water quality), and 5 to 6 indicates supertrophic (i.e., "very poor" water quality). Data are from the LAWA website (Land Air Water Aotearoa; https://www.lawa.org.nz/explore-data/otago-region/lakes/lake-hayes/).

6.6 Water pH

pH is a fundamental physico-chemical attribute of water indicates how acidic, neutral, or alkaline lake water is. pH influences chemical and geochemical processes in lake waters as well as physiological processes in aquatic biota.

pH data collected from the lake monitoring buoy shows that the lake water is generally circum-neutral or alkaline (Figure 32). The main driver of alkaline conditions is high rates of photosynthesis which removes dissolved CO_2 and bicarbonate (HCO_3^-) and produces hydroxide OH^+. During algal blooms, the pH becomes quite alkaline, rising the pH above 9 and sometimes even above 10. At such high pH, any ammonium ions convert to toxic ammonia. Fish and other organisms can become stressed and some forms of bound phosphorus in lake sediments can also become mobilised, and diffuse into the lake water. The highest pHs are generally observed in the surface waters, which may interact with shallow, marginal lake-bed sediments to cause a type of internal P load that is distinct from the internal P load generated by the development of anoxia in sediments located in the profundal zones of lakes.

The data in Figure 32 show that the pH of surface waters in the summer of 2022/23 reached very high levels at which point stresses to fish and sediment P mobilisation would be expected to occur. Table 8 shows the number of days per year which exhibited pHs above three alkaline thresholds (pH = 9, 9.5 and 10; the pH range where dissolved ammonium ion is mostly transformed into toxic ammonia gas). The data show that the summer or 2022/23 was a particularly bad year for high pH in the surface waters of the lake, with pHs above 10 occurring on 37 days.

It is notable that the bottom water pH was also higher in 2022/23 than previous years and this could be a sign that the pH sensor has been drifting upward over time. If true, there would have been fewer breaches of the thresholds in 2022/23 than indicated in Table 8.

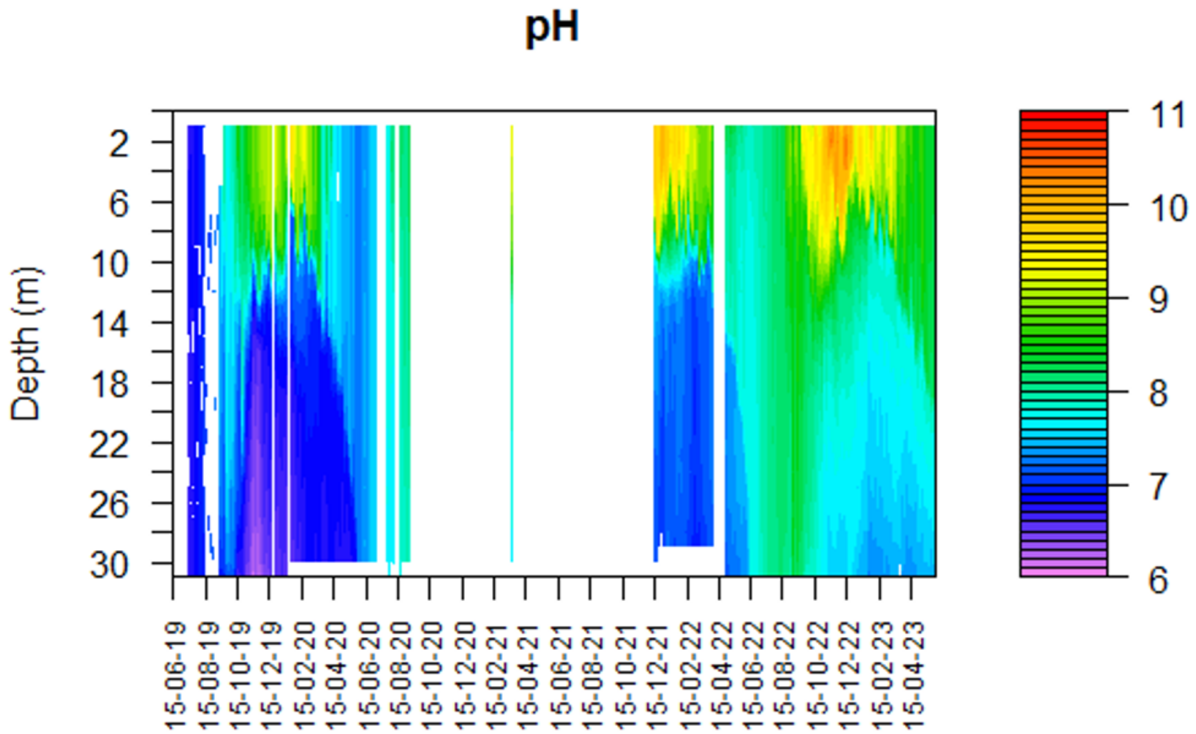

Figure 32. pH in Lake Hayes from 2016 to 2023 as recorded by the lake monitoring buoy. The data suggest that the pH sensor may have drifted upward in 2022/23.

Table 8. Number of days when pH exceeded three alkalinity thresholds (pH = 9, 9.5 and 10).

	2019/20	**2020/21**	**2021/22**	**2022/23**
pH threshold	**Days exceeding threshold**	**Days exceeding threshold**	**Days exceeding threshold**	**Days exceeding threshold**
9.0	75		95*	188
9.5	6		56*	97
10.0	0		4*	37

*data are underestimates due to data gaps as a result of equipment malfunction.

6.7 Fisheries and salmonid stress

In the past, Lake Hayes was a notable brown trout and perch sports fishery. However, the lake also contains natives fish species including common bully, kōaro, and longfin eel. Fish sampling by the University of Otago indicates that bullies are common in the lake, but kōaro and longfin eel are uncommon. A query of fish records for the lake, its tributaries, and Hayes Creek showed where different fish species have been observed (Figure 33, Table 9).

Occasional observations of dead fish have been reported in Lake Hayes and Mill Creek since the mid-2000s (M. Trotter, Otago Fish & Game, pers. obs.; M. Hanff, Friends of Lake Hayes, pers. obs.) suggesting that at times the lake becomes an unfavourable habitat for some fish species. Brown trout appear to be the most commonly reported dead fish. Both high temperatures and high pH are known to stress fish and brown trout are probably the most sensitive fish species in the lake to these stressors. In summer, when the hypolimnion becomes anoxic, the trout are not able to seek refuge in cooler bottom waters and must then seek refuge elsewhere, potentially in Mill Creek.

In an attempt to quantify the theoretical stress to fish from high temperatures and high pH, we have combined these two variables into a multiplicative index of fish stress. For each day that the lake monitoring buoy was operating, the maximum temperature and pH values were scored 1, 2, or 3 according to the thresholds presented in Tables 5 and 8 and the scores for temperature and pH were then multiplied together to produce a fish stress score (minimum = 1, maximum = 9), as is presented in Figure 34 and Table 10.

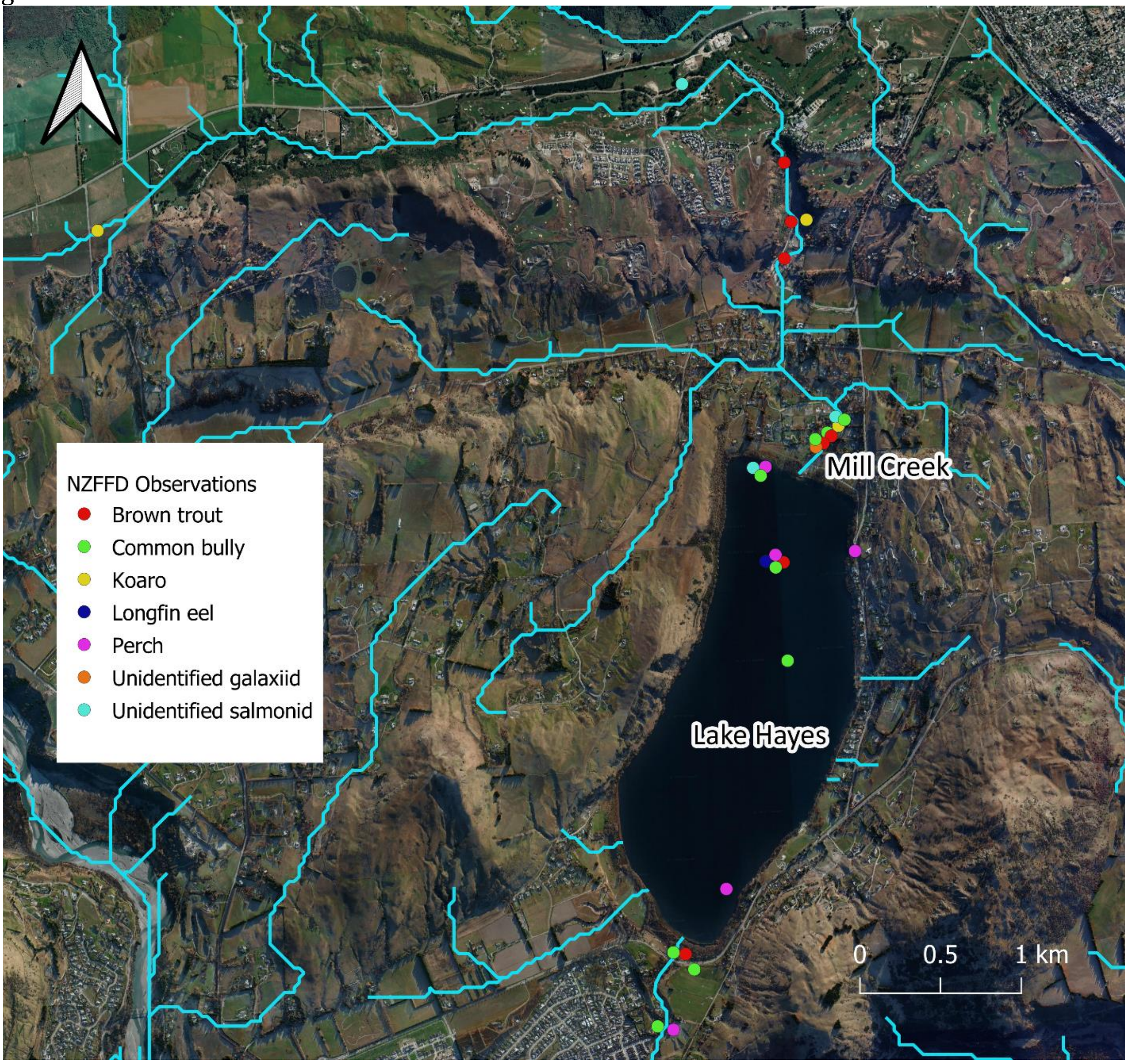

Figure 33. Reported locations of fish species records from Lake Hayes, Mill Creek, other tributaries, and Hayes Creek retrieved from the New Zealand Freshwater Fish Database on December 12, 2022.

Table 9. Fish species reported for Lake Hayes, Mill Creek, other tributaries, and Hayes Creek retrieved from the New Zealand Freshwater Fish Database on December 12, 2022. Threat status for these species is from Dunn et al. (2017) and Grainger et al. (2018). Occasional kōaro have also been observed in the lake (M. Schallenberg, personal observation).

Species		**Threat status***	**Lake Hayes**	**Mill Creek**
Kōaro	*Galaxias brevipinnis*	At Risk - Declining	-	✓
Longfin eel	*Anguilla dieffenbachii*	At Risk - Declining	✓	-
Brown trout	*Salmo trutta*	Introduced and naturalised	✓	✓
Perch	*Perca fluviatilis*	Introduced and naturalised	✓	-
Common bully	*Gobiomorphus cotidianus*	Not Threatened	✓	✓

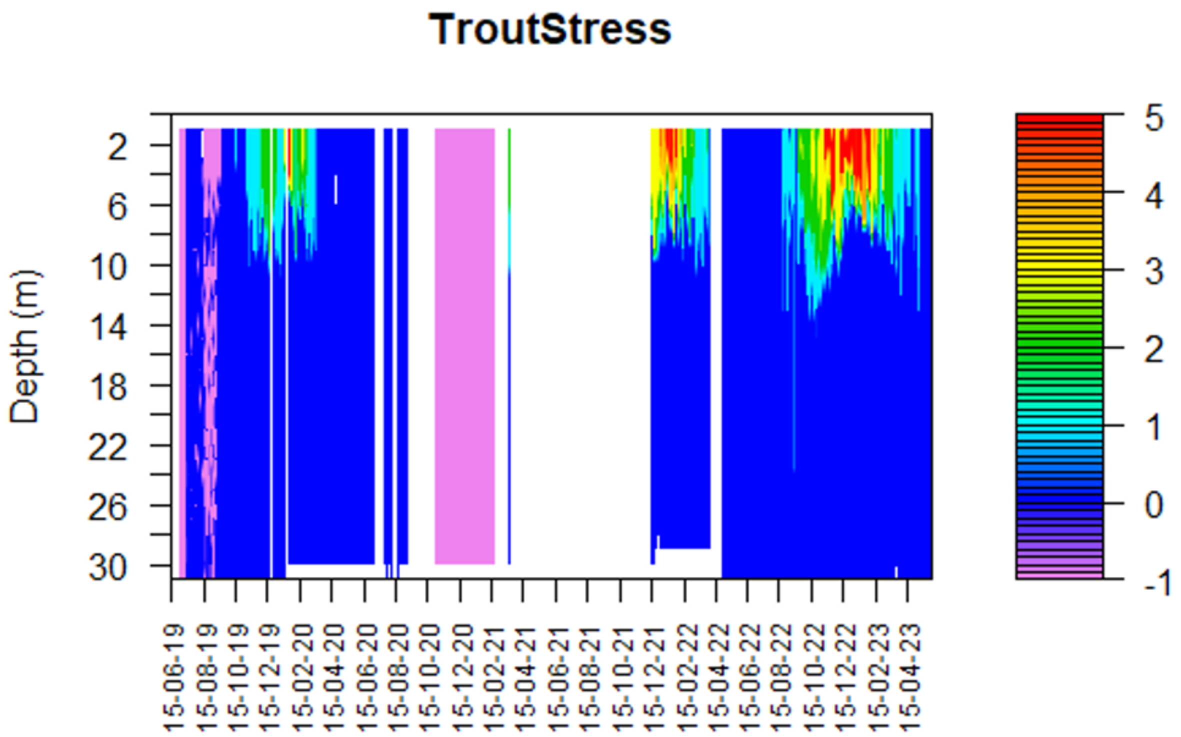

Figure 34. Trout stress score calculated as a multiplicative index of temperature and pH. See text in Section 7.8 for details.

Table 10. Days per year with trout stress scores exceeding three thresholds (score: > 3, > 6, and = 9).

	2019/20	**2020/21**	**2021/22**	**2022/23**
Stress threshold	**Days exceeding threshold**	**Days exceeding threshold**	**Days exceeding threshold**	**Days exceeding threshold**
> 3	75		95*	188
> 6	9		38*	81
9	3		0*	8

*data are underestimates due to data gaps as a result of equipment malfunction.

6.8 *E. coli*, cyanobacteria, and lake closures

The information in this section in based on recreational surveillance sampling that occured at the north end of Lake Hayes. *E. coli* was monitored weekly from December to March by the Otago Regional Council at the Lake Hayes site identified as Mill Creek Shallows (**Figure 1**ure 35). High *E. coli* numbers at this site have have been anecdotally attributed to large numbers of waterfowl that are sometimes present at the site and may also be due to sediment resuspension due to southerly winds. However, Mill Creek may also contribute to pulses of *E. coli* (see Section 5.1.5) potentially impacting the Mill Creek

Shallows, especially following rain events. Overall, the 5-year (long-term) assessment of recreational water quality of Lake Hayes is reported as "poor" on the LAWA website (https://www.lawa.org.nz/explore-data/otago-region/swimming/lake-hayes-at-mill-creek-shallows/swimsite), however, since 2020 no breaches of recreational *E.coli* standard have been reported (Figure 35).

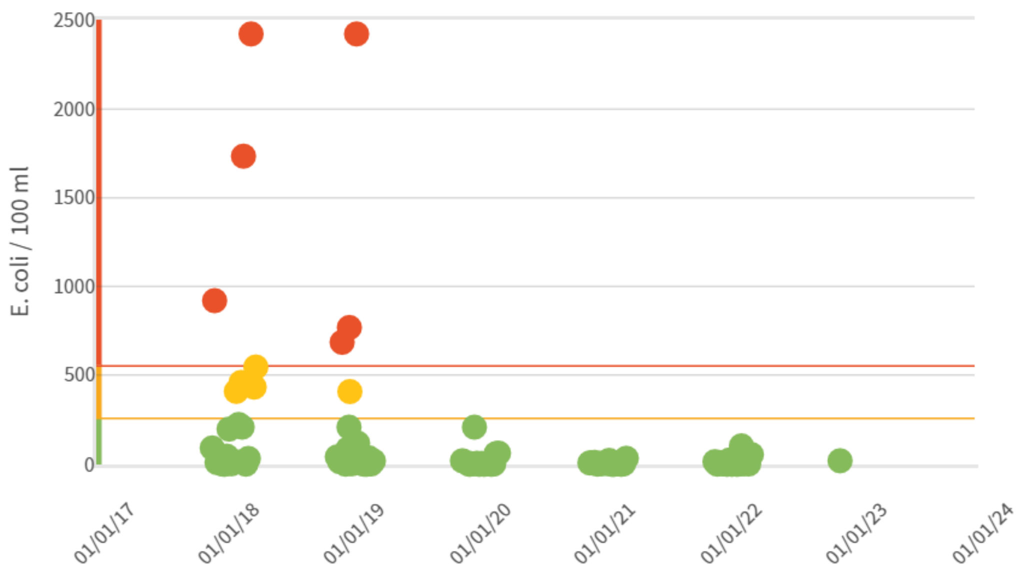

Figure 35. *E. coli results from surveillance monitoring of the bathing site at the north end of Lake Hayes. Data is from the* LAWA *website. E. coli counts exceeding the yellow line indicate "caution advised". E. coli counts exceeding the red line indicate "unsuitable" for swimming.*

Since 2018, Lake Hayes has had water advisories notified against contact recreation due to high biomasses of potentially toxic cyanobacteria on three occasions: in February/March 2018, March 2020 and in November 2022. Thus, over the last six years, there have been notified restrictions to recreational use due to cyanobacteria in three of the years.

6.9 Macrophytes

A 2020 LakeSPI (Submerged Plant Indicator) assessment of ecological condition indicate Lake Hayes to be in 'Moderate' condition based on submerged macrophyte community structure and composition (Burton 2021). This condition ranking remains generally consistent with the previous LakeSPI assessment carried out in 1992, however, the LakeSPI Index score has improved slightly from 32% to 40%.

Since 1992, the diversity of native submerged plants observed within Lake Hayes has increased from 5 to 7 vegetation types and the Native Index score has increased from 25.6% to 29.3%. The 2020 Native Index score indicates the lake to be in moderate ecological condition, indicating native submerged plant communities are moderately impacted, meeting a C band in the NOF. Native vegetation observed include charophyte species, emergents (reeds), *Isoetes*, native milfoils, native pondweeds, turf community. Submerged vegetation within Lake Hayes is often coated in filamentous algae. Vegetation was observed to grow to a maximum depth of 3.4 m, however charophyte meadows were restricted to 1.8 m depth. The maximum depth of vegetation in 2020 was shallower than the 1992 assessment which found native vegetation growing to a maximum depth of 8 m.

As of 2020, *Elodea canadensis* is the only invasive plant species observed within Lake Hayes and was found growing in up to 4.9 m of water. Although *Ranunculus trichophyllus* was present in the 1992 survey, it was not observed in 2020. The Invasive Index has improved from 65% in 1992 to 45% in 2020, reflecting the reduced diversity of invasive weeds present.

6.10 Sediment nutrients

From November 2012 to June 2013, a number of surficial sediment samples were collected from the deepest site (33 m) in Lake Hayes for analysis of sediment P geochemistry (M. Schallenberg, unpublished data). In November 2012, surficial sediment samples were also collected from two different sites; Site 2 which was at 20m depth and Site 3 at 11 m depth. The samples were put through a P fractionation protocol designed to quantify the amount of P in different fractions of the sediment with a view to quantifying the amount of P in the sediment that can be released under periods of anoxia.

Results show that a fairly consistent 30% of P in the sediment was either loosely bound, redox extractable, or was bound to metal oxides which can be solublised under anoxic conditions (Figure 36a). These fractions amount to approximately 2 g P per m^2 of sediment (Figure 36b). Experiments were carried out to measure the release rates of P from sediment samples incubated under a nitrogen atmosphere (i.e., under anoxia). P, iron, and manganese were found to have dissolved into the sediment pore waters under anoxia, indicating that anoxic conditions dissolved iron- and manganese-oxyhydroxides along with P (Figure 36c). The amount of P released was more strongly correlated with the manganese released than with iron released and this may be due to the presence of sulphur in the lake sediments which may scavenge iron to produce iron sulphide under anoxic conditions. While there was no clear trend in the amount of P released under anoxic conditions from the sediment samples collected through the summer at the deepest site, there appeared to be a trend of increasing P released from sediments at greater water depths (Figure 36c).

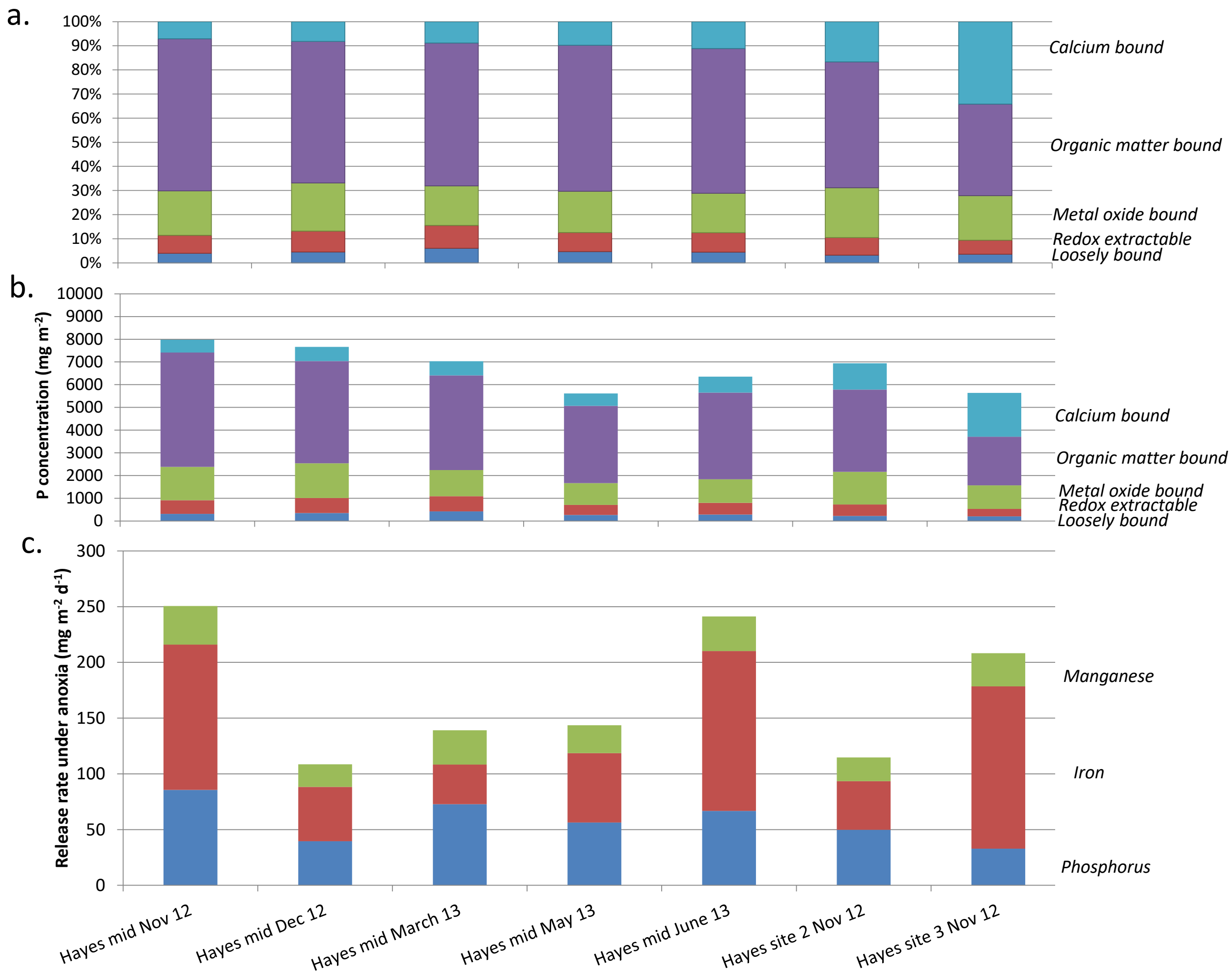

Figure 36. Sediment geochemistry study of Lake Hayes sediment. Panel (a.) shows the percentage of sediment P found in different fractions of sediment. Loosely bound P, redox extractable P and metal oxide P are fractions likely to be released from sediments specifically during anoxia. Organic matter P dissolves as sediment organic matter decomposes. Panel (b.) shows the amounts of P in the different fractions. Panel (c.) shows the release rates of P, iron and manganese in a laboratory experiment where the sediments were incubated under anoxic conditions. Samples were collected from the mid-lake site (33 m), site 2 (20 m) and site 3 (11 m) from November 2012 to June 2013.

Thus, these geochemical studies show that there is P in the lake sediments that can be released under anoxic conditions and fractions associated with manganese oxyhydroxides likely contribute to the hypolimnetic internal P load throughout the summer. This supports the observations made for high hypolimnetic P concentrations during anoxic periods (see Section 6.5, below). In addition, the large fraction of organic P in the sediments likely also contributes to the internal P load, but probably more so when the sediments are aerated during the winter period of full water column mixing.

6.11 The Lake Hayes foodweb

Schallenberg & Schallenberg (2017) suggested that the food web of Lake Hayes may contribute to the summer water clarity of the lake. They suggested that the predatory effects of new cohorts of juvenile perch in spring and summer on *Daphnia* could suppress *Daphnia* densities, thereby allowing algal biomass to increase, facilitating algal blooms and low water clarity. They suggested that year-to-year variation in perch spawning success would cause variation in juvenile perch densities and when spawning success was low, *Daphnia* could persist in sufficient densities over the summer to suppress *Ceratium* and other algae.

In the intervening period since the Schallenberg & Schallenberg (2017) report, a Master's (Trotter 2022) and a PhD (Khan 2021) thesis on the Lake Hayes food web have been completed. Both of these theses studied the potential for the perch-Daphnia-algae trophic cascade to operate in the lake and to produce the occasional clear water summers illustrated in Figure 20. Notable findings included:

(1) confirmation that *Daphnia* were able to feed on *Ceratium*,
(2) confirmation that juvenile perch feed on *Daphnia*,
(3) confirmation that as juvenile perch numbers increase in spring/summer, *Daphnia* densities in the lake decline sharply, and (
4) confirmation that *Daphnia* persisted over the clear water summer of 2017/18.

Both Khan (2021) and Trotter (2022) suggested a number of options for intervening in the trophic cascade to try and enhance summer *Daphnia* abundance, thus helping to improve summer water clarity in the lake. Their work on the food web of Lake Hayes is important because it highlights that water quality in the lake is not only influenced by nutrient availability, but also by top predators in the lake and their effects on lower trophic levels (e.g., zooplankton and algae). Both authors were sceptical that predators could exert a predominant control over the lake's water quality, but their work suggested that, at least in some years, predator control contributes to water quality outcomes, highlighting the importance of considering a holistic approach to the management of Lake Hayes.

7 Water balance, nutrient and sediment budgets

7.1 Water balance

Mass budgets are useful tools for managing and restoring lakes because they provide a quantitative, holistic perspective on flows and stocks in the lake and its catchment. By considering all or most of the flows (in- and outflows) in relation to standing stocks, turnover times, relative sizes of flows, retention coefficents, and other important aspects of a water balance, nutrient and sediment budgets can be calculated and considered. This can help identify the critical sources and flows of water, nutrients and sediments through the system.

The starting point for calculating mass budgets is the water balance, which attempts to account for significant inflows, outflows and lake volume over an annual time frame. Robertson (1988) calculated a water balance for the Lake Hayes hydrosystem for the period June 1983 to May 1984. Using ORC flow and climate data, we also calculated a water balance for the calendar year 2020 (Table 11).

Table 11. Water balance for Lake Hayes calculated for the periods June 1983 to May 1984 (Robertson 1988) and for the calendar year of 2020 (this study).

Water Flows and Stocks	Component	Robertson (1988) 1983/84 $10^6 m^3 yr^{-1}$	Robertson (1988) 1983/84 $10^6 m^3$	This study 2020 $10^6 m^3 yr^{-1}$	This study 2020 $10^6 m^3$
Load	Mill Creek	22		15.6	
	Other tributaries	1.6		?	
	Spring at Rutherford Rd	1.5		1.2	
	Direct rainfall	2.9		1.8	
Total load		***28.0***		***18.6***	
Export					
	Evaporation	2.1		3.1	
	Hayes Creek outflow	28		16.9	
Total export		***30.1***		***20***	
Lake					
	Lake Volume		55.1		55.1
	Change in lake volume	n/a		-0.3	
Theoretical residence time (yr)		***1.8***		***2.8***	

The water balances show that the total inflows and outflow to the lake were approximately 50% greater in 1983/84 than in 2020 and, as such, the water residence time was only 1.8 years compared to 2.8 years in 2020. The water balances suggest that Mill Creek is the predominant source of water to the lake, although the inflows from other tributaries were not measured in either water balance, although they were estimated by Robertson (1988). Contributions by the springs at Rutherford Road were relatively minor as was the input from rainfall directly on the lake (measured daily at Queenstown Airport in 2020).

7.2 Nutrient budgets

Nutrient budgets allow the calculation of net retention rates of nutrients by lakes and this can give an indication of whether the lake is a net sink or source of nutrients (Verburg et al. 2018). To calculate nutrient loads to the lake from Mill Creek, we used data from ORCs high frequency nitrate sensor and high frequency turbidity sensor, both located at the Fish Trap site. TP was derived from turbidity data collected at a variety of flows by the ORC and the FOLH (Figure 37a). TP concentrations were also highly correlated with flow in Mill Creek (Figure 37b) such that one large flood in July 2020 contributed approximately 45% of the annual phosphorus load from Mill Creek to Lake Hayes (Figure 38). In contrast, nitrate loads were not driven by floods, indicating that the base load to Lake Hayes was less dependent on flow, although loads were greater in spring than at other times of year. Using these data, we were able to calculate the main phosphorus and nitrogen loads from Mill Creek to the lake.

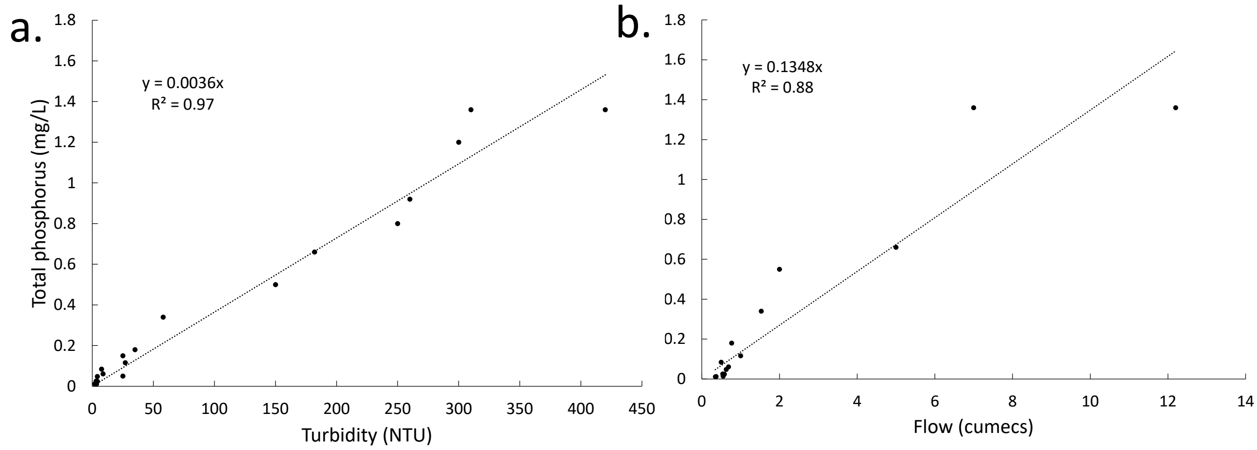

Figure 37. Calibration curves showing the relationships between total phosphorus concentration and (a.) turbidity, and (b.) flow of Mill Creek. Data are from the Otago Regional Council and the Friends of Lake Hayes.

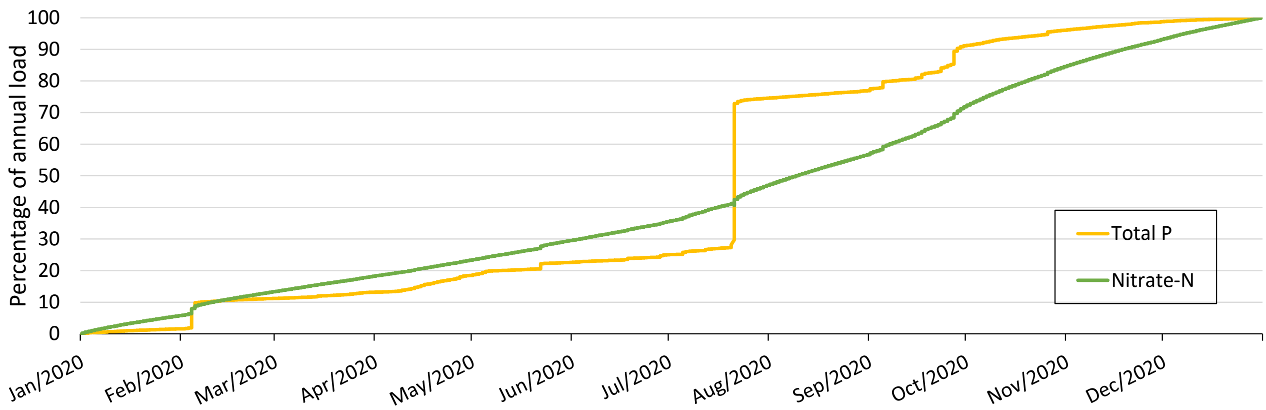

Figure 38. Cumulative percentage of the annual loads of total phosphorus and nitrate-Nitrogen to Lake Hayes from Mill Creek.

7.2.1 Phosphorus budget

The phosphorus budgets for 1983/84 and 2020 are presented in Table 12. While Robertson (1988) measured the load from Mill Creek, the other sources of P in his budget were estimated. We used the data from Mill Creek described above to calculate the Mill Creek load. We also used some sparser data from the FOLH to calculate the load from the Rutherford Road spring. Data from the spring is likely to be relatively accurate due to the much greater stabilities of nitrate concentrations and flows at the spring than at Mill Creek. We estimated phosphorus loads from rainfall directly on the lake using average phosphorus concentrations in rainfall for the South Island from Verburg et al. (2018). Phosphorus export from Hayes Creek was calculated from daily flows provided by the ORC and some phosphorus concentration data provided by the FOLH, which was linearly interpolated to daily values.

The phosphorus budgets of Robertson (1988) and this study were quite similar, except that the overall load was 13% larger in 2020 while the export via Hayes Creek was 7% lower, resulting in a higher P retention coefficient. This indicates that the lake was a stronger P sink in 2020 than it was in 1983/84. This is consistent with a reduction in the internal load and the lake achieving a greater state of equilibrium in relation to high legacy P loads (i.e., from historically higher P loads), but could also be explained by the longer water residence time calculated in the 2020 water balance.

The P load from Mill Creek was higher in 2020 than in 1983/84 despite the fact that 71% of the annual water 1983/84 annual discharged flowed down Mill Creek in 2020. This raises concerns about the relatively high P yield from the catchment in 2020 as compared to 1983/84.

The data in Figure 30a allow an approximation of the internal P load for the lake of around 2000 kg P per summer. Taking this estimate for 2020, the anoxic, hypolimnetic P load would have increased the total P load to the water column from 2600 kg P to 4600 kg P, an increase of 43%. The external P load from Mill Creek to the lake was much lower in 2021 and 2022 (Table 12), suggesting that the internal P load was probably the main source of phosphorus to the lake in those years. The calculated P inputs, outputs, and retention by sedimentation for the years 1983/84 and 2020 are summarised in Table 13, highlighting the remarkable similarity in the overal lake P budgets in those two years. However, in 2020, the lake itself contained over twice the amount of P that it contained in 1983/84 (Table 12).

7.2.2 Nitrogen budget

We also calculated a Nitrogen budget for the year 2020. The Mill Creek TN load was estimated as 1.4 × the measured nitrate load, based on the TN and nitrate concentrations of numerous grab samples of Mill Creek water analysed by the ORC and the FOLH. Estimates of the N load from the Rutherford Road springs were also based on grab samples and estimates of flow provided by the ORC and the FOLH. Average TN in rainfall for the South Island from Verburg et al. (2018) was again used to estimate nutrients in rainfall at the lake. Hayes Creek N export was estimated from daily mean flows provided by the ORC and the TN concentration in grab samples. Linear interpolation was used to interpolate to daily TN concentrations.

The lake retains nitrogen far more than it retains phosphorus, which highlights the sizeable internal phosphorus load of the lake. The N:P ratio of the loads in 2020 was 17 while that of the outflow was 9 and that of the average lake standing stock was 7.6. Thus, in-lake processes reduced the amount of N relative to P in the loads. These processes include internal P loading, but may also include denitrification. Nitrogen processes have not been studied in Lake Hayes to date although the presence of heterocysts in *Dolichospermum* blooms suggests that N-fixation may also occur at times in the lake.

Table 12. Phosphorus and nitrogen budgets for Lake Hayes. See text for details.

Total Phosphorus Flows and stocks	Component	Robertson (1988) 1983/84 $t\ yr^{-1}$	Robertson (1988) 1983/84 t	This study 2020 $t\ yr^{-1}$	This study 2020 t
Load	Mill Creek	2.1		2.3	
	Other tributaries	0.2		0.2	
	Spring at Rutherford Rd	0.02		0.002	
	Direct rainfall	0.015		0.062	
Total load		***2.3***		***2.6***	
Total export	***Hayes Creek outflow***	***0.8***		***0.75***	
Lake					
	Lake stock		2.0		4.4
	Change in lake stock	n/a		0.2	
Retention coefficient		***0.65***		***0.71***	
Total Nitrogen					
Load	Mill Creek			42.5	
	Other tributaries			?	
	Spring at Rutherford Rd			1.3	
	Direct rainfall			0.6	
Total load				***44.4***	
Total export	**Hayes Creek outflow**			**7**	
Lake					
	Lake stock				33.6
	Change in lake stock			18.6	
Retention coefficient				***0.84***	

Table 13. Summaries of annual nutrient budgets of Robertson (1988) and this study. All units are kilograms.

Year	External load	Hypolimnetic Internal load	Output	Sedimentation
1983/84	2400	2000 to 2600	640	3760 to 4360
2020	2600	2000	750	3850

7.3 Sediment Budget

Strong correlations between turbidity and suspended sediment concentrations in Mill Creek (R^2 = 0.98, N = 11) allow an approximate sediment budget for the lake to be calculated (Table 14). From this, we estimated that in 2020 at least 2,302 tonnes of suspended solids entered Lake Hayes from Mill Creek. The export of suspended solids was a small fraction of the load, resulting in a retention efficiency of 95%. Most of the inflowing suspended solids would have been inorganic sediment and soil particles mobilised from the catchment during floods, whereas most of the exported suspended solids would have been algal material.

Table 14. Approximate sediment budget for Lake Hayes for 2020.

Total suspended sediment		**2020**	**2020**
Flows and stocks	**Component**	**t yr^{-1}**	**t**
Measured load	Mill Creek	2302	
	Other tributaries	?	
	Spring at Rutherford Rd	0	
	Direct rainfall	0	
Total measured load		***2302***	
Total measured export	**Hayes Creek outflow**	**104**	
Lake			
	Lake stock		?
	Change in lake stock	?	
Load - Export		***?***	
Retention coefficent		***0.95***	

8 Summary

Lake Hayes is possibly the most researched lake in New Zealand, with scientific measurements going back to the late 1940s. Problems with eutrophication of the lake seem to have begun in the 1960s when major drainage works were undertaken to convert wetlands in the mid-catchment to pasture. In recent years, the Otago Regional Council improved water quality monitoring of the lake and its main tributary, Mill Creek. This includes a continuous record of monthly water quality sampling since late 2016 and the deployment of a profiling lake monitoring buoy in 2019. The quality information that has accumulated as a result of these monitoring programmes allows for a significant update to the assessment of water quality and ecological condition of the lake and its tributary.

Mill Creek exhibits high, and increasing, nitrate concentrations, breaching regional water plan standards. The phosphorus phosphorus load from Mill Creek to the lake is partly dependent on the occurrence of floods. The phosphorus load to the lake in 2020 was estimated at 2.3 tonnes, which exceeded the load of 2.1 tonnes estimated in 1983/84. However, in 2021 and 2022, the P load decreased substantially. *E. coli* counts in Mill Creek often exceed regional standards and threaten recreation at the popular northern end of the lake.

Water clarity in Lake Hayes has declined markedly since 2006, apart from two summers (2010/11 and 2017/18) when water clarity was remarkably clear. The reason for high water clarity in occasional summers is unknown, but research on the pelagic food web of the lake suggests that this could be related to a trophic cascade triggered by reduced perch recruitment in those years.

In the summer of 2022/23, Lake Hayes exhibited high temperatures, high pH, high chlorophyll *a* and a long duration of seasonal thermal stratification. The lake began its thermal stratification unusually early, before dissolved oxygen in the lake water had become fully equilibrated with the atmosphere. Therefore, the lake was already somewhat oxygen depleted when thermal stratification began in the spring of 2022. This led to particularly intense deoxygenation of the hypolimnion over the summer. Anoxic hypolimnia have been a feature of Lake Hayes since at least the 1970s and these summer dead zones occupy more than half the lake volume, potentially precluding fish from accessing deeper, cooler waters to escape the summer heat that affects the surface waters. This may be associated with occasional fish kills that have been reported since at least the mid-2000s.

Intense anoxia results in the release of phosphorus that is bound to hypolimnetic lake bed sediments, which is then diffused into the water column. Our estimate of this internal phosphorus load suggests that in 2020, this internal source of mostly DRP may have raised the P load to the lake by 43%. The contribution to the lake's annual phosphorus budget was probably much higher in 2021 and 2022 because the external P load from Mill Creek in those years was much lower. Our calculations suggest that the internal P load has not declined substantially between 1983/84 and 2012/13. As a result, the trophic state of the lake has not improved much over the past decades and appeared to increase from meso-eutrophic to eutrophic-supertrophic between 2017 and 2021.

In summer, high pH and high temperatures in the surface waters together with anoxia in the bottom waters constitutes a situation of significant potential physiological stress for brown trout. Furthermore, this situation is beneficial for cyanobacterial blooms and blooms of potentially toxic cyanobacteria have caused the Otago Regional Council to announce warnings about recreation on the lake in three of the past six years. Occasionally high *E. coli* counts and cyanobacterial blooms both impact on recreational use of the lake.

A survey of the macrophyte community in 2020 found that the community was in a moderate condition, similar to the condition found in a 1992 study.

A phosphorus budget for the lake in 2020 indicated that the retention efficiency of phosphorus in the lake was slightly higher than in 1983/84, possibly reflecting a small decrease in the internal phosphorus load and/or the longer water residence time of the lake in 2020.

The nitrogen to phosphorus ratio of the lake is substantially lower than that of the main inflow, Mill Creek. This reflects the substantial internal phosphorus load that the lake receives annually and suggests that denitrification may also occur in the lake.

While a substantial knowledge base exists about the lake in relation to its degraded state, we identify some knowledge gaps that, if filled, could help our understanding of the lake.

9 Knowledge Gaps

This report analyses and summarises recent data on Lake Hayes, Mill Creek, the springs at Rutherford Road, and Hayes Creek. While a substantial knowledge base exists about the lake in relation to its degraded state, we are still deficient in key knowledge in some areas.

1. **Rates of dissolved oxygen decline in the hypolimnion:** An analysis of the dynamics of dissolved oxygen in the hypolimnion during summer and the development of a hypolimnetic dissolved oxygen budget could help elucidate how the supply of oxygen to the hypolimnion could reduce internal P loads.

2. **Contribution of willows to hypolimnetic dissolved oxygen, nitrogen and phosphorus budgets:** The organic matter in willow leaves contributes to deoxygenation while the phosphorus and nitrogen contained in the leaves may become bioavailable after mineralisation by bacteria and fungi in the lake bed. No investigations to date have been carried out on the potential contribution of lakeshore willows to the hypolimnetic oxygen, phosphorus, and nitrogen budgets of the hypolimnion.

3. **Nitrogen and phosphorus limitation of algal growth:** It could be useful to investigate the relationship between algal blooms in the lake and the availability of nitrogen and phosphorus prior to, and during the blooms. Furthermore, an investigation into the dynamics of dissolved reactive phosphorus in Mill Creek could also shed some light on the current, relative importance of nitrogen and phosphorus to algal productivity in the lake.

4. **Weather effects:** Investigations into how spring weather may affect drivers of lake water quality could help our understanding of the variability in water clarity from year-to-year.

5. **Quantifying uncertainty in nutrient load estimates:** Investigations into the main sources of potential errors and bias in load estimates and nutrient budgets would allow confidence intervals to be placed around the estimates.

6. **Understanding the lake foodweb using eDNA:** As more and more eDNA samples are analysed for Lake Hayes, this information should be collated into a database which could be used to track species of interest in Lake Hayes.

10 Recommendations

In addition to revealing the knowledge gaps above, the analyses carried out in this report suggest three specific recommendations to help improve the monitoring of Lake Hayes.

1. **Nutrient profiles to help calculate annual internal nutrient loads:** We were only able to calculate internal nutrient loads and lake nutrient standing stocks for years for which nutrient profiles were available. Profiles should include at least four samples in the hypolimnion and should be repeated at least monthly between December and May.

2. **More samples in Mill Creek at high flows:** Most ORC data for Mill Creek were obtained at times when the creek was flowing at its median flow or lower. This made it difficult to estimate the nutrient load contributions of floods and accurately quantify annual loads. It would be helpful to carry out some event sampling to help better understand nutrient and sediment loads to the lake.

3. **Key indicators of degradation and recovery:** In addition to the statutory water quality, ecosystem health, and public health attributes which must be monitored and reported, the analyses in this report reveal additional useful indicators of the condition of Lake Hayes. To facilitate future lake assessments, we recommend that the following indicators be measured at regular intervals and reported on:

    - The internal phosphorus and nitrogen load (requiring monthly summer and autumn nutrient profiles)
    - The nutrient concentrations in the lake during the isothermal period
    - The duration of the stratified period
    - The depth limit of the euphotic zone (i.e., the depth of photosynthetic activity and dissolved oxygen production)
    - The concentrations of *Daphnia* in the water column (as an indicator of grazing pressure)

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Appendix A: Hourly data (discharge, turbidity, nitrate) and derived variables (total P, total suspended solids) for Mill Creek from January 1 to December 31, 2020.