Lake Hayes Restoration and Monitoring Plan

Photo: David Hamilton

Prepared for the Friend of Lake Hayes Society Inc.

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Executive summary

Lake Hayes is a highly-valued lake that has suffered from algal blooms for many decades. Since 2006, the blooms have worsened and lake health and fishing has deteriorated markedly. This report provides relevant historical background to the Lake Hayes water quality story, analyses water quality and ecological data and information, and proposes a restoration strategy for lake recovery.

The worsening of algal blooms since 2006 has occurred when both external and internal nutrient loads to the lake had been either declining or stabilising. So, the reason for the blooms was not related to increased nutrient inputs to the water column. Rather, a change in the dominant algae species occurred and the new nuisance alga, a dinoflagellate called *Ceratium hirundinella*, possesses some particular adaptations that allow it to supplement its nutrition in unusual ways. So, the development of *Ceratium* blooms appears to have been facilitated by a decline in nutrient loads, which gave it a competitive advantage over other algae. The scientific literature contains reports of *Ceratium* and other dinoflagellates sometimes becoming dominant during a recovery from high nutrient loads.

While *Ceratium* seemed to have had a hold on Lake Hayes, two recent summers (2009/10 and 2016/17) have seen the lake exhibit extremely clear waters with very little algae biomass. This could indicate that the lake is approaching a recovery tipping point. How long it will take the lake to achieve consistently high water clarity is unknown. However, observations indicate that high densities of zooplankton (grazers of algae) in summer have been associated with high summer water clarity, suggesting that dynamics of the pelagic food web may play an important role in the lake's recovery.

This report evaluates the potential for many various restoration activities to accelerate the recovery of the lake. Four of these strategies have been selected to be the most promising and cost-effective. These are: (1) food web biomanipulation, (2) enhanced flushing by using surplus irrigation water from the Arrow River, (3) alum dosing to flocculate and bind phosphorus in the lake bed, and (4) a focus on land use activities in the catchment to further reduce nutrient and sediment losses from land to water. These strategies were scrutinised using the available data and some costing were determined. This allowed the development of a restoration strategy proposing the most promising strategies to use, potential timelines to achieve implementation, and suggesting a range of restoration targets by which to measure success. This report also discusses lake monitoring options to help track recovery of the lake and demonstrate effectiveness and cost-effectiveness of the strategies.

Contents

Scope

Lake Hayes is a treasured asset to Tangata Whenua, locals and tourists alike. Located within the Arrow Basin between Queenstown and Arrowtown, the lake is highly visible by road and frequented by lake users and campers year-round. Since the late 1960's the lake has been subject to severe algal blooms, initially as a result of increased nutrients entering through Mill Creek and from springs at the northern end of the lake, which are high in nitrate concentration (Bayer & Schallenberg 2009). From the 1960's through to 2010, lake water quality had steadily decreased to a eutrophic state with blooms of blue-green algae/cyanobacteria, green algae and dinoflagellates occurring in stages throughout that time. Currently the lake suffers from severe blooms of the dinoflagellate alga *Ceratium hirundinella* blooms almost yearly, prompting the Friends of Lake Hayes Society to investigate and instigate restoration measures with the aim of returning the lake to a healthier state.

Multiple reports have described the main issues affecting Lake Hayes as well as a range of potential lake restoration options for consideration. However, with the changeable nature of the lake's algal blooms and the fast-increasing trend towards further eutrophication, this restoration plan was commissioned by the Friends of Lake Hayes Society. The restoration plan aims to describe the current state of the lake, summarise the major issues affecting water quality, recommend and discuss realistic restoration options, provide a restoration strategy with timelines, and recommend useful monitoring strategies for monitoring lake status and recovery to a stable water quality state which aligns with community, stakeholder and tourism values.

1 Background

1.1 Community values, uses and importance

Lake Hayes has been described as one of the most scenically attractive landscapes of its type in New Zealand and it holds significant importance for recreation and tourism (Cromarty & Scott 1995). Surrounding the lake is a vegetated margin with patches of wetland areas supporting a high diversity of endemic, rare or threatened fauna including the koaro, longfin eel and breeding birds such as paradise ducks, New Zealand shovellers, marsh grebes, Australian coots and great crested grebes (Cromarty & Scott 1995). A popular shared-use trail navigates these margins. Parts of the lake surroundings have been granted recreational and wildlife management reserve status as well as belonging to a wider wildlife refuge area covering 354 ha, including the lakebed. The lake and its immediate surrounds are used by locals and tourists alike for a range of recreational activities including rowing, boating, fishing, swimming, running, biking, walking and picnicking.

The Lake is culturally important for its food gathering which has led to the lakes recognition as a treasured resource (Waahitaoka) (ORC 2009) and the Lake Hayes Management Strategy states 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).

1.2 Historical catchment development

Extending far to the north-west of the lake along Mill Creek, the Lake Hayes catchment (Fig. 1) was likely forested with kahikatea prior to 1740 and a large wetland extended through the western reaches of Mill Creek in the mid-catchment. A number of smaller wetland swamps also existed in the catchment to the west and north of the lake as well as extensive riverine marshes on the banks of Mill Creek and smaller streams (Robertson 1988).

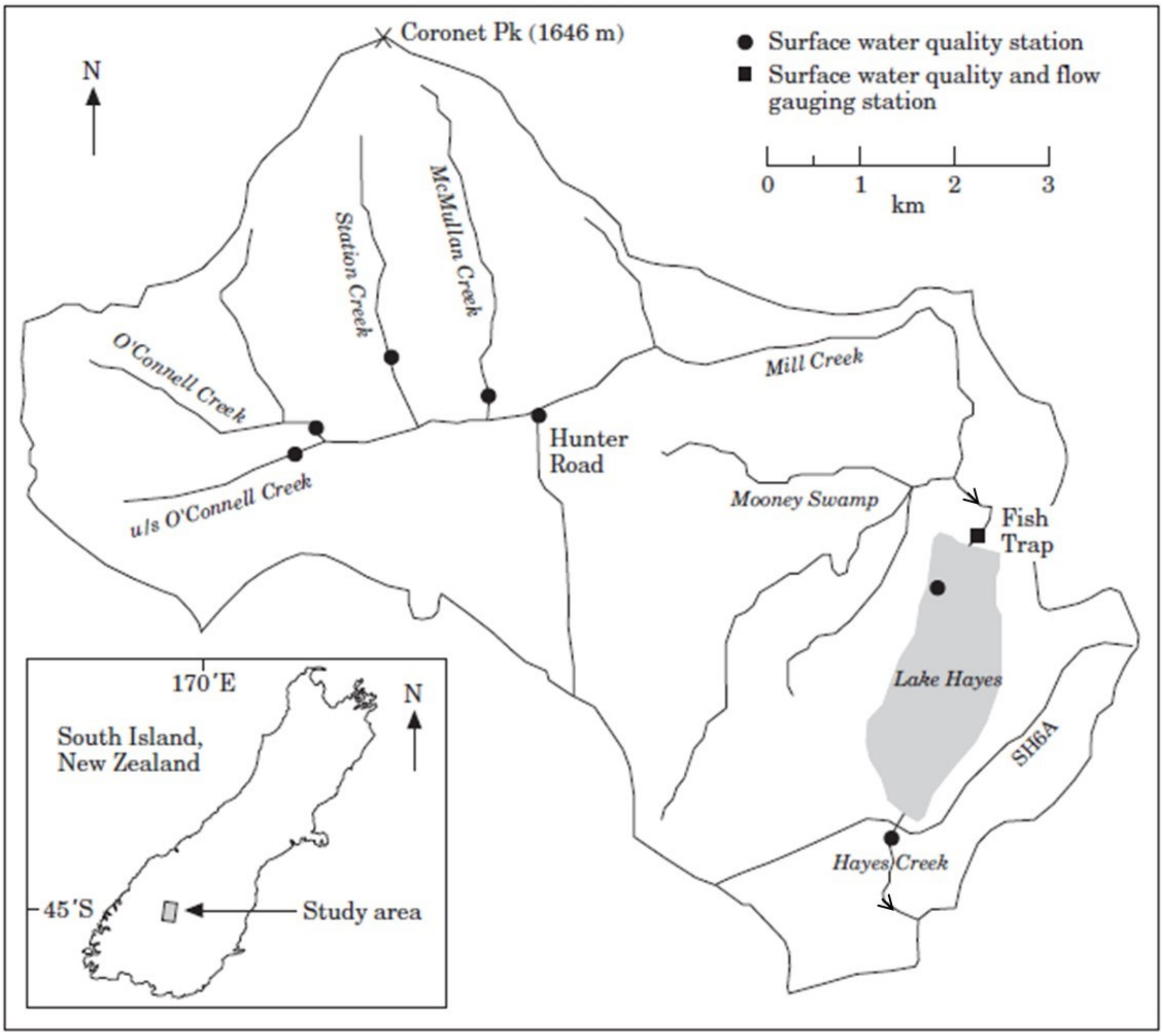

Figure 1. The Lake Hayes catchment (adapted from Caruso 2001)

Deforestation in the catchment began around 1740 when the Kahikatea forest was largely destroyed by fire, and further deforestation likely occurred through the late 1800's as miners and settlers harvested trees for shelter and firewood (Robertson 1988). After the deforestation of the Kahikatea forest, the Lake Hayes catchment comprised mostly native tussock grassland in the high country with swamps and wetlands dominating the lowland areas including the Arrow Basin (Robertson 1988). Mill Creek is the major tributary in the catchment, fed by a number of high country streams including O'Connell Creek, Station Creek and McMullan Creek, sediment and nutrients from which were once immobilised in the wetland before continuing on down Mill Creek towards Lake Hayes. 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. Relatively low concentrations of nutrients are expected to have been transported by Mill Creek and its tributaries through the early 1900's.

The early-to-mid 1900's saw land converted to sheep pasture and further conversions from sheep to cattle and dairy. Superphosphate fertilizer was introduced in the 1950's allowing cattle and dairy to intensify in the catchment and aerial topdressing was common on farms, particularly around the lake, within which a topdressing plane was lost in 1953 (Robertson 1988). From approximately 1912-1955 a local cheese factory operated to the north of the lake where it released whey effluent with a phosphorus (P) load of approximately 1000kg/yr (roughly equivalent to the annual effluent of 2000 cows) directly into Mill Creek (Robertson 1988). Remaining whey was fed to pigs which also contributed further effluent to the creek.

The Otago Catchment Board began major drainage and channeling works in 1961-62, which saw wetlands drained and artificial channelization through what was soon to be high producing exotic grasslands. The initial channel and drainage works in 1961 cut through 80-120ha of wetland in the upper catchment, bringing a significant amount of sediment through 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 sediment immobilization has been touted as a major turning point for lake ecosystem health.

With the conversion of wetlands into pastoral grasslands, the water quality buffering capacity of the catchment decreased. It is estimated that 80% of the P load in Mill Creek came from the tributaries above the large wetland (Robertson 1988) and when in its natural wetland state, sediment and sediment-bound nutrients such as P were trapped and nitrate was denitrified. Nutrient loads from the catchment to the lake via Mill Creek and the springs at the northern end of the lake are likely to have been very low. Robertson (1988) also notes the operation of the Arrow River irrigation scheme which at the time of writing in 1988, was taking 1.75m^3/s of water from the Arrow River and irrigating 1100ha in the middle of the Lake Hayes catchment. Through the 70's, 80's and the early 90's, multiple catchment stressors continued to affect lake water quality including the loss of wetland buffering capacity, the application of superphosphate fertilizers on new pastoral land, and continued catchment cutting and drainage works which delivered further pulses of sediments and nutrients to the lake.

1.3 Fisheries

1.3.1 Trout fishery

Brown Trout were introduced to Lake Hayes in 1870 and the fishery flourished from the late 1800's through the 1930's with fish up to 25lb caught (Fig. 2). In the 1940's, the Wildlife Service set up fisheries operations including the collection of brown trout ova (Robertson 1988; H. Trotter, Otago Fish & Game, pers. comm.) and the construction of a fish trap on Mill Creek near the inflow to Lake Hayes. From 1940-1960, 1000-4000 adult trout passed through the fish trap annually with up to 2 million ova collected annually for national and international fish stocking (Otago Fish & Game, unpublished data). Fish trapping operations slowed through the 1960's and 70's before ceasing in the late 1970's as demand for ova stock decreased and the water quality in Mill Creek declined (Robertson 1988; H. Trotter, Otago Fish & Game, pers. comm.).

Figure 2. Lake Hayes trout fishery, 147 brown trout, 1 October 1897.

1.3.2 Perch fishery

Perch were introduced shortly after trout, in the late 1870's, and quickly established, being the most commonly caught fish species by 1900. The perch population grew and in 1988 the Percy Perch Classic fishing competition was established, running until 1990. The event attracted over 1000 anglers to the lake and more than 13,000 perch were landed over two days in 1989; however around 97% of those adults caught were less than 20cm long, indicating a highly-stunted population (H. Trotter, Otago Fish & Game, pers. comm.). Such stunting can be a result of unrestrained population growth controlled only by competition within the species for food resources as opposed to predatory 'top down' population controls. The lack of predation on Perch continues to result in a high proportion of stunted adults confirmed by a survey in 2016 where 70 Perch were caught in one hour and 97% of adults were stunted (<20cm). (Otago Fish & Game, unpublished data).

The Lake Hayes trout fishery has long been recognized as a regionally important fishery and is highly regarded by anglers for both its recreational and amenity values (H. Trotter, Otago Fish & Game, pers. comm.), however its popularity among anglers has decreased rapidly since the mid 2000's. Annual angler days (a measure of angler effort) were at around 1500 days in the mid 1990's and early 2000's (Fig. 3) but dropped dramatically in 2006/2007 (Otago Fish & Game, unpublished data). Fish & Game received numerous complaints from anglers regarding the "muddy, brown colour" of the lake water and the poor condition and scarcity of trout during the 2006/2007 season, followed by two fish kills observed in March and April 2007 (Otago Fish & Game, unpublished data). In March, Mill Creek was running high and carrying brown sediment into the lake where few trout were seen, which were all in very poor condition and 6 dead trout were found around the creek mouth. In April, emaciated trout were observed in the lower reaches of Mill Creek and there were reports of around 30 dead trout floating near the mouth of Mill Creek. Lake Hayes itself had a bloom of the dinoflagellate *Ceratium hirundinella* during this time and all remaining trout observed were described as emaciated and in very poor condition.

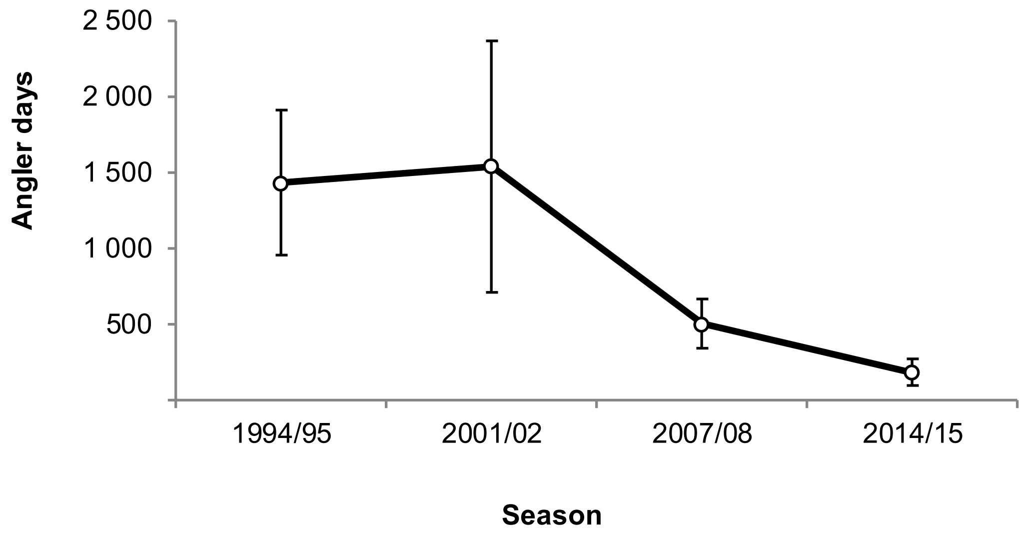

Figure 3. Annual total angler effort estimates for Lake Hayes taken from the National Angling Survey (Otago Fish & Game, unpublished data).

Due to increasingly poor fishing conditions, angler days dropped to around 500 in the 2007/2008 season (Fig. 3) and although no further fish kills have been reported, angler effort continued to decline to a record low of 180 angler days in the 2014/15 season. Over this time, public concerns arose again regarding the low numbers of poor quality trout spawning in Mill Creek, however a Fish & Game monitoring programme set up in response found the condition of trout in Mill Creek had improved over 2013-2015 compared with those found in 2007. While there have been reports of trout in good condition being caught in recent years (H. Trotter, Otago Fish & Game, pers. comm.), anglers remain concerned about the lower numbers of fish caught and the degraded water quality of the lake.

2 Water Quality

2.1 Background

The information presented above describes the situation, whereby Lake Hayes has become a eutrophic lake, with generally relatively low water clarity, poor water quality and frequent algal blooms. Since 2006, the trophic level index (indicating nutrient enrichment) has deteriorated markedly, and in 2015 the water quality of the lake was very poor (supertrophic) (Fig. 4).

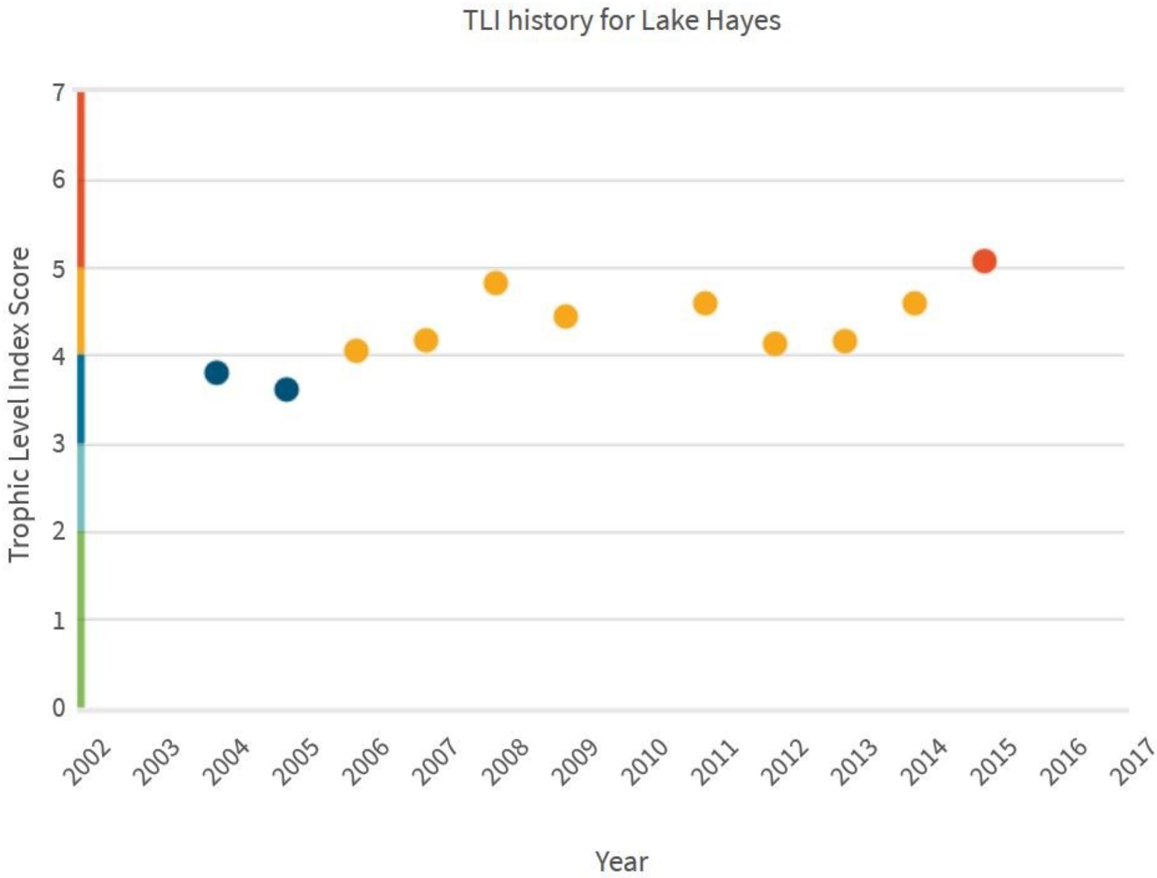

Figure 4. The trophic level index (TLI) score for Lake Hayes from 2004 to 2015. The TLI aggregates total phosphorus, total nitrogen, chlorophyll *a* (an indicator of phytoplankton biomass) and Secchi disk depth (a measure of water clarity) data. TLI between 3 and 4 is mesotrophic (good water quality). TLI between 4 and 5 is eutrophic (poor water quality). TLI between 5 and 6 is supertrophic (very poor water quality). Data and graph are from the LAWA website.

Publicly available data from the LAWA website (Land Air Water Aotearoa; https://www.lawa.org.nz/explore-data/otago-region/lakes/lake-hayes/) only go back to 2004. However, Lake Hayes has been studied since the late 1940s, beginning with the work of Jolly (Jolly 1959). Comparing lake data back as far as Jolly's time provides a useful context for our analysis of the historical and current condition of Lake Hayes.

Figure 5 presents the Lake Hayes Secchi disk depth data, showing how water clarity in the lake has changed over time. Since the 1950s, when the lake's bottom waters were oxygenated in summer (Jolly 1959), water clarity has been variable, but has often been quite poor (e.g., eutrophic or water clarity below 3.6 m) due to algal blooms.

From 1970 onward, nitrogen-fixing cyanobacteria (e.g., *Anabaena* sp.) have often been part of the phytoplankton community, sometimes occurring as the dominant species of phytoplankton (Burns & Mitchell 1974; ORC 1995). Nitrogen-fixing cyanobacteria may outcompete other phytoplankton when excess phosphorus is available because the cyanobacteria are able to harvest nitrogen from the atmosphere (from air dissolved in the lake water). The bottom waters of Lake Hayes have become anaerobic (with a complete loss of dissolved oxygen) since at least 1970 (Burns & Mitchell 1974) and summer deoxygenation of the bottom waters has been recorded, whenever it has been measured, since that time (Robertson 1988; ORC 1995; Bayer et al. 2008; Bayer & Schallenberg 2009; M. Schallenberg, unpublished data; ORC, unpublished data). The loss of dissolved oxygen from the bottom waters not only excludes trout, zooplankton and many invertebrates from the cooler bottom waters of the lake, but it also causes biogeochemical changes in the lake sediments, releasing sediment-bound phosphorus into the water column (Bayer et al. 2008). When the surface of the sediment is oxygenated, the oxygenated minerals (e.g., iron and manganese oxyhydroxides) in the sediments bind a large proportion of sediment phosphorus, preventing its release back into the water column. Deoxygenation of the water and sediment converts the sediments from P sink to a P source. Since at least 1970, the summer bottom waters have been releasing significant amounts of phosphorus into the lake water (Mitchell & Burns 1981; Robertson 1988; Bayer et al. 2008; Bayer & Schallenberg 2009; M. Schallenberg, unpublished data; ORC, unpublished data), recycling historically accumulated and immobilised P back into the lake ecosystem and further fuelling algal and cyanobacterial blooms. Severe blooms eventually settle to the lake bed delivering more P to the sediments as dead phytoplankton cells, where they decompose and consume oxygen, contributing to the next year's summer deoxygenation. This internal anoxia-phosphorus-algae feedback cycle has contributed to maintaining Lake Hayes in a eutrophic state since at least 1970 despite the fact that external nitrogen and phosphorus loading from Mill Creek and the springs at the northern end of the lake decreased into the 1990s and early 2000s (Caruso 2000; Bayer et al. 2008; Bayer & Schallenberg 2009).

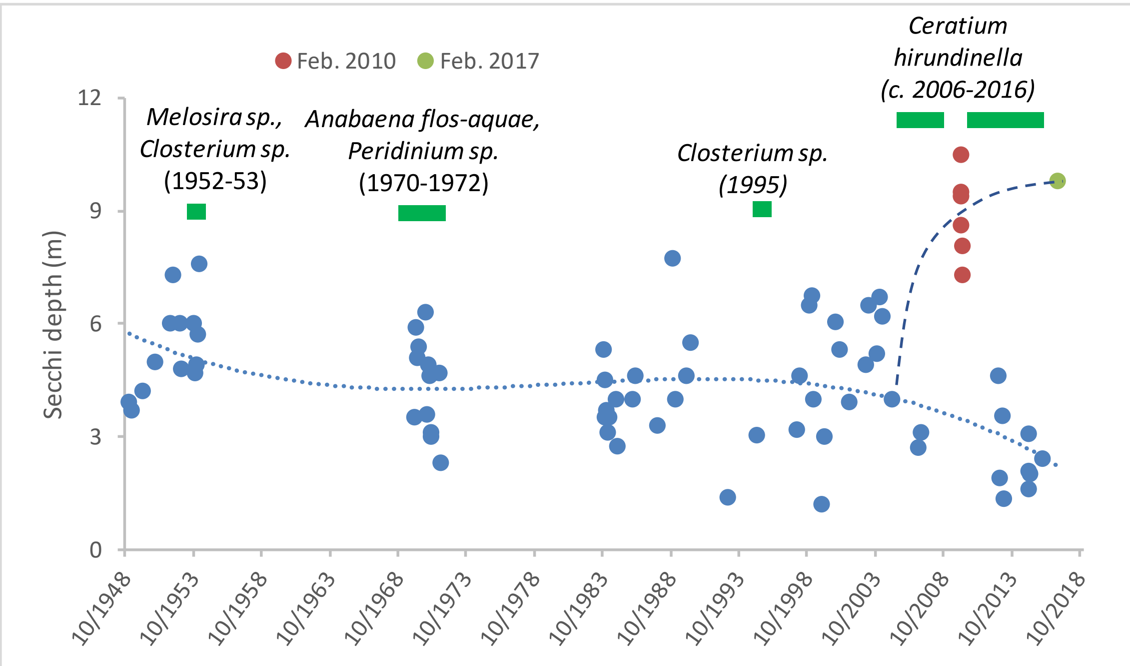

Figure 5. Historical summer (November to April) water clarity (Secchi disk depth) measurements in the open waters of Lake Hayes. The dominant phytoplankton species causing summer blooms are shown, if reported. Phytoplankton information and data from 1952/53 are from H. Jolly (1959) and Burns & Mitchell (1974), from 1970-72 are from Burns & Mitchell (1974), and from 1995 are from C.W. Burns, unpublished data. Blue dot Secchi data from 1984-2015 are from unpublished Otago Regional Council data, M. Schallenberg unpublished data, and Caruso (2001). Red dots are from M. Schallenberg, unpublished data. Green dot is a datum from the Otago Regional Council.

The average water clarity of the lake was rather stable from 1970 to 2006 (Fig. 5), until the brown-coloured dinoflagellate alga, *Ceratium hirundinella*, began to form dense blooms in the lake (Bayer et al. 2008). These blooms were more severe than most previous blooms, further reducing summer water clarity (Fig. 5). *Ceratium* blooms were also associated with fish kills and decreased angler interest in the lake, as discussed in Section 1.3, and began to cause skin and mucous-membrane irritation in at least one long-term local resident who regularly swam in the lake (M. Schallenberg, pers. comm.). Curiously, the severe *Ceratium* blooms were not associated with increased external nutrient loading from Mill Creek or the springs (Bayer & Schallenberg 2009; LAWA website) or with increases in internal nutrient recycling during the summer anoxic period. In fact, phosphorus concentrations in the anoxic deep (25m) summer waters appeared to have been decreasing from around the year 2000 and ammoniacal N concentrations were also low during the *Ceratium* bloom period compared to in the 1980s and 1990s (Fig. 6).

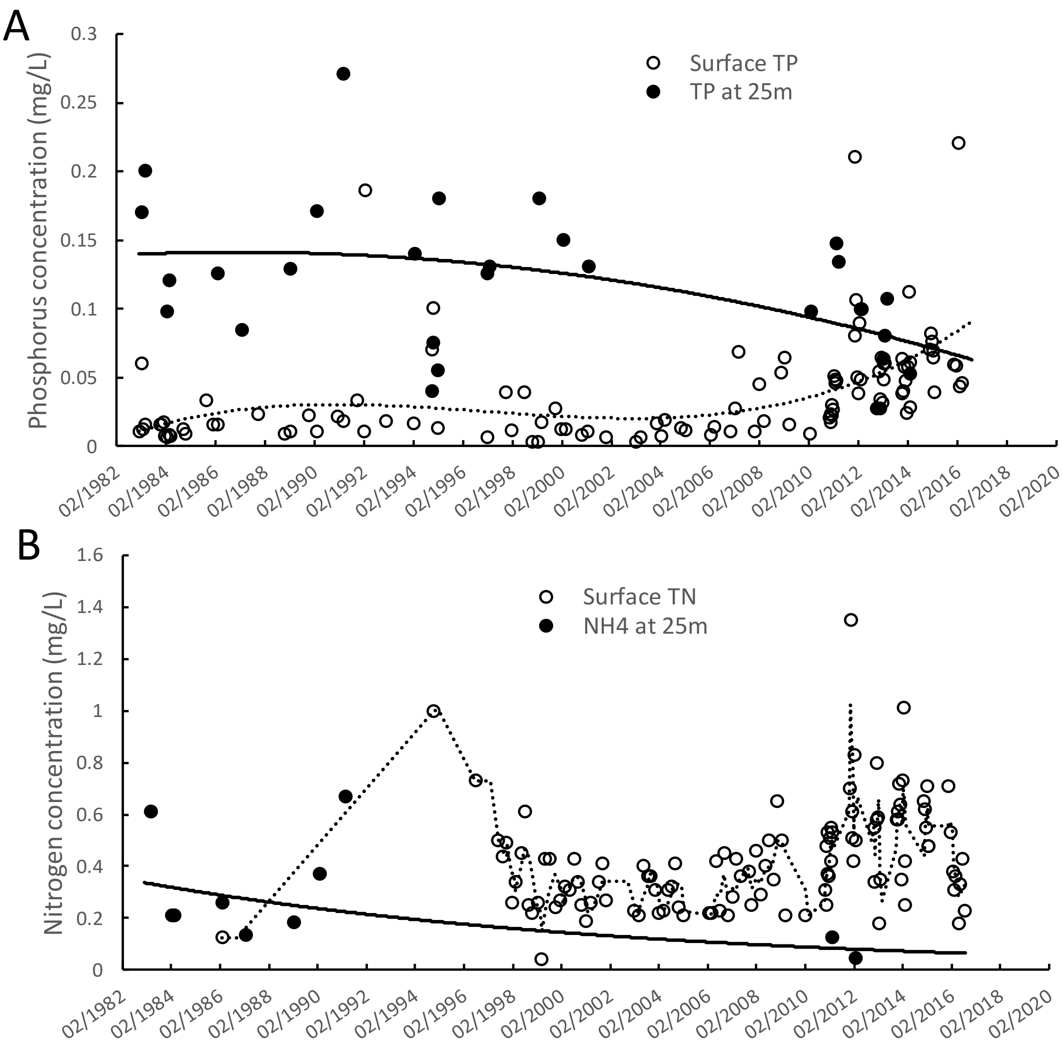

Figure 6. Trends in the concentrations of phosphorus and nitrogen concentrations in Lake Hayes during the stratified period (Nov-April inclusive) from 1983 to 2016. A. Total phosphorus at the lake surface and at 25 m depth (anoxic bottom waters). B. Total nitrogen at the lake surface and ammoniacal nitrogen at 25 m depth (anoxic bottom waters). Data are from the Otago Regional Council.

This intriguing situation of worsening algal blooms while internal and external nutrient loads had not measurably increased was frustrating for locals and recreational users of the lake. However, this apparent enigma can be explained by ecological peculiarities of the *Ceratium* alga, shown in Figure 7. This dinoflagellate is a large, spikey, motile (swimming) alga that is mixotrophic, meaning that it can gain energy both via photosynthesis (as plants do) and also by feeding on bacteria (as some protozoans and zooplankers do).

Figure 7. *Ceratium hirundinella* showing one of the two flagella used for locomotion/swimming. Total length of the cell is typically 150 µm to 200 µm. Photo: https://why.gr.

Mixotrophic dinoflagellates including *Ceratium* have been reported to become abundant and dominant in the phytoplankton communities of lakes during periods of lake recovery from eutrophication (Jeppesen et al. 2003; Gerdeaux & Perga 2006, Mehner et al 2008), partly due to their ability to supplement their nutrient requirements by feeding on bacteria (Gerdeaux & Perga 2006). *Ceratium* is also able to migrate vertically in the water column on a daily basis, enabling it to access recycled nutrients in the deep waters of lakes at night while also enabling high rates of photosynthesis in the upper water column during the day (James et al. 1992). In Lake Hayes, as in other lakes, these strategies probably enabled *Ceratium* to become highly competitive by nocturnally migrating and accessing nutrient-rich bottom waters at night and by feeding on bacteria when nitrate, ammonium and phosphate concentrations in the surface waters of the lake are scarce (i.e. during summer).

2.2 A *Ceratium* nutrient pump hypothesis

If *Ceratium* in Lake Hayes undertakes a day-night migration to harvest recycled N and P from the bottom waters during summer, then it is expected that *Ceratium* would transfer significant amounts of phosphorus from the bottom waters into the mixed layer during daytime, when it migrates to the surface layer to photosynthesise. The accumulation of recycled phosphorus in the bottom waters of Lake Hayes has been a major component of the P budget of the lake; however, Figure 6A shows that the recycled P contribution from summer bottom waters has decreased while the summer surface water phosphorus concentration has increased since the time that *Ceratium* began to bloom, in 2006. This raises the possibility that *Ceratium* may translocate P from the bottom waters to the surface waters of the lake during summer (Fig. 6A). We see a similar pattern over time for N (Fig. 6B), but, unlike for P, recycled ammoniacal N is only a small part of the N budget of the surface waters of the lake.

Because *Ceratium* blooms have been associated with increases in surface water P concentrations in summer (Fig. 6A), the apparent *Ceratium*-mediated transfer of P from the bottom waters to the surface waters in summer is expected to enhance the flushing of P out of the lake via Hayes Creek. This water flowing out of the lake is surface water and, therefore, the transfer of P from bottom waters to surface waters increases the flushing of P out of the lake. This hypothesis is consistent with the increasing concentrations of total phosphorus at the Hayes Creek outflow between 1993 and 2008 reported by Bayer & Schallenberg (2009). This enhanced P flushing should accelerate the recovery of Lake Hayes by eventually breaking the summer anoxia-phosphorus-algae feedback cycle that had been delaying recovery of the lake.

As the flushing of P from the lake progresses, it is expected that *Ceratium* blooms will eventually become self-limiting due to this apparent translocation and enhanced flushing of P from the lake. Currently, the declining levels of P available in the summer deep waters of the lake may already be reducing the competitive advantage that *Ceratium* has over other algae in the lake. A further reduction in *Ceratium's* competitive advantage could occur if the density of bacteria in the lake, which may supplement *Ceratium's* energy requirements, were also to decline.

2.3 Is Lake Hayes approaching a recovery tipping point?

We have shown that the bottom water nutrient concentrations which reflect internal recycling of legacy nutrients, have been decreasing in recent years. The apparent *Ceratium*-mediated transfer of P to the surface layers has probably increased the flushing of P out of the lake via Hayes Creek by increasing surface water nutrient concentrations in summer. Since 2006, *Ceratium* blooms have plagued the lake, where *Ceratium* has outcompeted other algae and cyanobacteria probably by harvesting phosphorus from deeper waters in summer and by grazing on bacteria to supplement its nutrition. While the lake has suffered severe *Ceratium* blooms in most summers since 2006, the developments described above suggest that the lake is on a trajectory toward recovery from historically high nutrient loads.

Further evidence of this is the fact that in the summer of 2009/10 and 2016/17, the lake experienced unprecedented water clarity (Fig. 5) and very low *Ceratium* biomass. The *Ceratium* hiatus in 2009/10 lasted only one summer, but the reduction in algal biomass in the surface waters (Fig. 8) and the increase in water clarity (Fig. 5) was striking. While the reduced internal P recycling probably contributed to these clear water summers, another interesting feature of these summers was the persistence of the water flea, *Daphnia pulex*, in the lake over the summer period. *D. pulex* is an intense grazer of algae (Burns 2013) and our sampling of zooplankton in the summers of 2009/10, 2012/13 and 2015/16 indicate that during summers when *Ceratium* bloomed, *D. pulex* was absent from the lake. Thus, we believe that food web interactions related to summer *Daphnia* presence in the lake also contributed to the sudden shift of Lake Hayes from a eutrophic condition with severe summer *Ceratium* blooms to summers with very low *Ceratium* biomass (Figure 9).

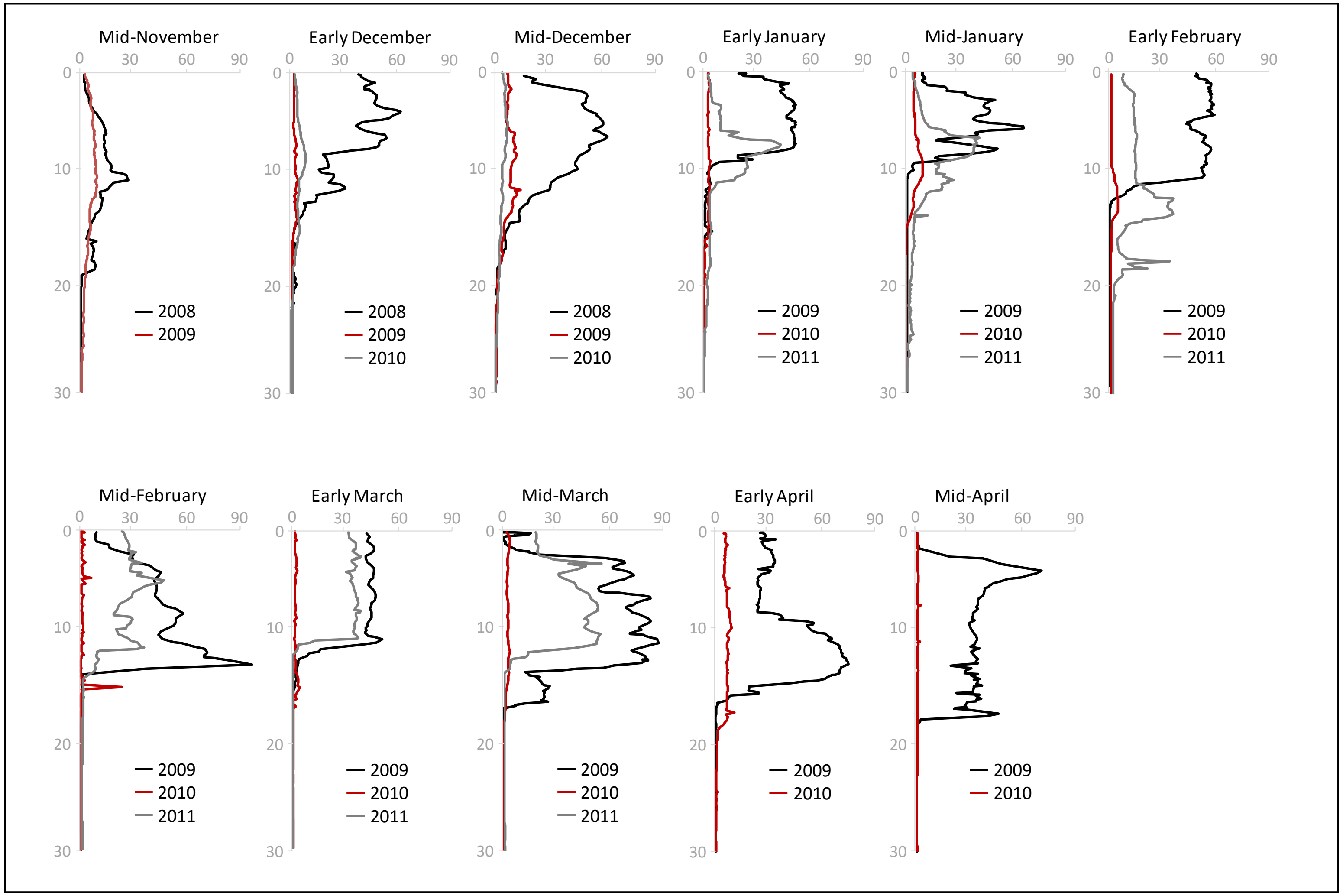

Figure 8. Vertical profiles of chlorophyll *a* in the summers of 2008/09, 2009/10 and 2010/11. Profiles were measured by the Otago Regional Council. Chlorophyll *a* data are *in vivo* fluorescence measurements from an uncalibrated datasonde and are, therefore, approximate concentrations in μg/L.

Rapid changes in trophic state are common in shallow lakes, which can fluctuate markedly in water clarity from year-to-year (Mitchell 1988; Scheffer 2004; Schallenberg & Sorrell 2010), when nutrient loading approaches a tipping point. However, such behaviour is not as common in deep, seasonally stratifying lakes, but has been reported in relation to species invasions (e.g., Lakes Erie and Ontario due to zebra mussel invasion) and to the dynamics of algal pathogens (e.g., pathogenic fungi controlling *Ceratium* spp.; Heaney et al. 1988). Circumstantial evidence described here suggests that Lake Hayes has entered a phase of recovery whereby nutrient availability is approaching a recovery tipping point and that food web interactions in some years may have tipped the lake into a temporary recovery from eutrophication (Fig. 9). These food web interactions are discussed in more detail in Appendix 1.

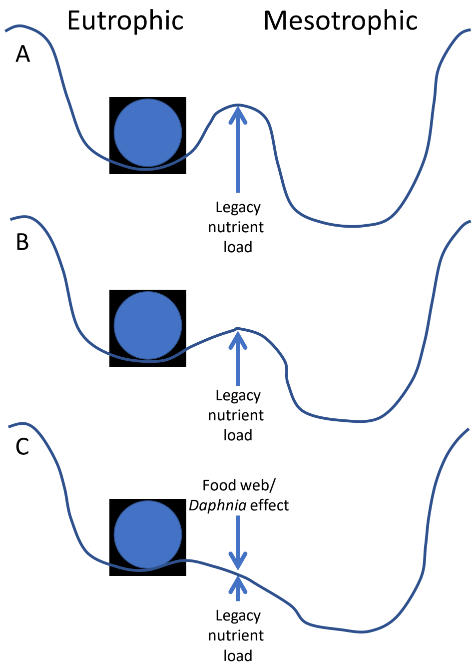

Figure 9. A conceptual model showing alternative stable states for Lake Hayes. The lake is represented by the ball, which is held in the eutrophic state by intensely recycled phosphorus (A). Recent reductions in P recycling have weakened the resistance to recovery (B). In the summers of 2009/10 and possibly 2016/17 (to be confirmed), the combination of reduced P recycling and food web effects reduced resistance to recovery further, allowing a temporary shift to a clear water state (C). In summer 2010/11, the *Ceratium* bloom returned and the lake shifted back to a eutrophic state (Fig. 8).

Experiments done on the Lake Hayes phytoplankton community in 2006 (Bayer et al. 2008) indicated that the phytoplankton community in the lake (dominated by *Ceratium* at the time) was stimulated by additions of N and the trace elements boron and zinc. In the four experiments conducted, phosphorus additions did not stimulate phytoplankton production. This result supported an analysis of N:P ratios in the lake, which also suggested that P was often in surplus in the lake water relative to N (in relation to the nutrient demands of phytoplankton) (Bayer et al. 2008). It would be interesting to now re-examine the nutrient supplies in the lake to test whether a decade of *Ceratium* dominance in the system has reduced phosphorus levels in the lake to the point where they can again begin to restrict phytoplankton blooms. The re-establishment of P-limitation of phytoplankton growth would have the added benefit of removing the competitive advantage of N-fixation, which historically dominant bloom-forming phytoplankters such as *Anabaena sp.* are capable of.

Our analysis of water quality data has yielded some insights into the drivers of phytoplankton blooms in Lake Hayes and it highlights the importance of having a detailed understanding of the nutrient budgets of lakes affected by nutrient enrichment. The combination of Otago Regional Council State of the Environment monitoring data and the University of Otago's occasional research projects on the lake provides a useful perspective on the factors driving large changes in water quality of the lake over time. Although the available data are patchy and many knowledge gaps would need to be filled to confirm the hypothesis presented here, the combined use of lake data, information on overseas lakes and expert experience and deduction provide a compelling hypothesis concerning the recent condition of the lake and what could be done to speed its recovery.

2.3 Key points on water quality analysis

The key points from our analysis of water quality data are as follows:

- The internal recycling of phosphorus during summer, which has been a large source of P to the lake at winter turnover, has been decreasing in recent years such that it is now so reduced so as to have little effect on the surface water concentrations of P.
- *Ceratium* blooms began in response to reduced internal nutrient recycling and external nutrient loading and its proliferation has probably been due to the fact that it can harness nutrient resources unavailable to most other algae.
- The *Ceratium* blooms have probably transferred recycled P from bottom waters to the surface waters, enhancing flushing of legacy P from the lake.
- Since *Ceratium* can't add P or N to the lake, it's apparent translocation of P (and maybe N) to the surface waters probably increases the flushing of these nutrients out of the lake and will eventually reduce nutrient availability to the point where Ceratium may become limited by low nutrient availability.
- Lake Hayes appears to be approaching a recovery tipping point, where nutrient availability and food web factors prevent the development of significant algal biomass in occasional summers.
- At this point, the food web factors assisting the temporary recovery seem to involve *Daphnia* persistence over the summer months.
- The current situation suggests that appropriate restoration measures could result stable in improvements in summer water clarity, reductions in *Ceratium* summer biomass, and the re-oxygenation of the bottom waters of the lake. These factors appear to be facilitated by maintaining a low nutrient availability and a high summer *Daphnia* density.

3 Lake Hayes historical timeline of events

The information presented in Sections 1 and 2 can be summarised in a timeline describing the trajectory of Lake Hayes and its catchment, as related to the health of the lake. The significant historical events in Figure 10 show how complex the Lake Hayes system is and how a combination of historical factors (e.g., fish introductions, fertiliser overuse, dairying, wetland drainage, etc.) and current factors (e.g., *Ceratium* blooms, *Daphnia* dynamics, decreasing nutrient loads, etc.) affect the current conditions and trajectory of the lake. The complexity of the Lake Hayes system highlights that management of the lake must consider a comprehensive range of factors that operate on a range of time scales and that often interact with each other to affect lake health.

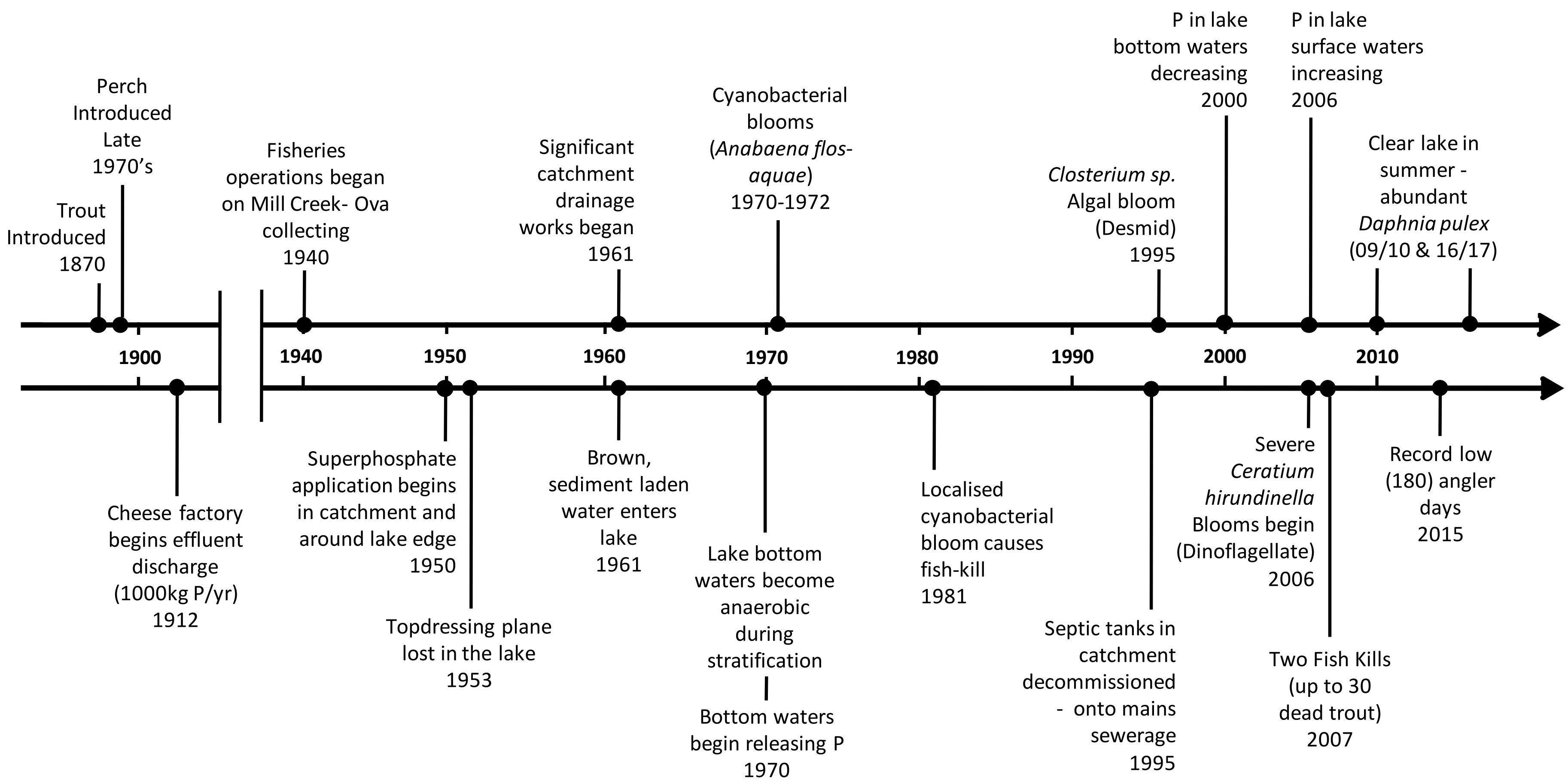

Figure 10. Historical timeline of major events impacting Lake Hayes and its catchment.

4 A restoration strategy for Lake Hayes

Restoration options for Lake Hayes have been discussed since the early 1970s, soon after the lake experienced its first severe algal blooms (Mitchell & Burns 1972) and have been revisited numerous times since then (Robertson 1988; Bayer & Schallenberg 2009; Ozanne 2014). Numerous strategies have been considered in terms of effectiveness and cost-effectiveness. The strategies fall into five main types: 1. catchment rehabilitation to reduce external nutrient loads to the lake, 2. reduction of internal nutrient loads/recycling, 3. food web manipulation, 4. flushing of water through the lake, and 5. other in-lake actions (Table 1).

Mitchell & Burns (1972) discussed and costed options including diversion of inflows away from the lake, flushing the lake with irrigation water and oxygenation of the bottom waters. Robertson (1988) placed much emphasis on catchment mitigation strategies, while subsequent reports have focused more on controlling in-lake P loading/recycling from the hypolimnion. The focus on internal P loading was sensible when the lake was releasing large legacy amounts of P into the water from the anoxic bottom waters on an annual basis and when the growth of the dominant phytoplankers was limited by P availability. However, our analysis of water quality data show that nowadays, the P concentration in the deep bottom waters (25 m) is similar to the P concentration in the surface waters during summer. In addition, the concentration of P in the Hayes Creek outflow had been rising at least up to 2009 (Bayer & Schallenberg 2009), indicating that while external P loads had reduced (Bayer & Schallenberg 2009) more P was being flushed from the lake. Our analysis suggests that *Ceratium* blooms that have plagued the lake since the mid-2000s have transferred bottom water P to the surface of the lake, enhancing the flushing of legacy P out of the lake.

Lake Hayes has a water residence time of around 1.8 years (Caruso 2000). Flushing of the lake could potentially be enhanced by augmenting Mill Creek with cleaner water from the Arrow River Irrigation Scheme, as was suggested by Mitchell & Burns (1972). When surplus water from the scheme is available (November to June – see Appendix 2), it could be used to augment the flushing of the lake. Augmented flushing of the lake could enhance the flushing of algae and P out of the surface waters of the lake. It could also add dissolved oxygen if the augmented flow were to plunge into the bottom waters during the stratified period. Total P concentrations are currently almost uniform in the surface and bottom waters in summer (Fig. 6A) and algae are concentrated in the surface layer in summer (Fig. 8). According to the surplus irrigation water that would be available, the augmentation flow available from the Arrow River would flush an additional 7% of the lake volume if the full surplus were used from September to June (Appendix 2). The relative temperatures (and therefore densities) of Mill Creek and lake water during this period suggests that Mill Creek (including any augmentation flow), is not likely to plunge into the bottom waters of the lake (Appendix 2). Therefore, while flow augmentation would enhance flushing of the lake by 7% per annum, it is unlikely to significantly alter the oxygen concentration or dynamics in the bottom waters of the lake. Thus, according to the analysis, the main benefits of flow augmentation to the recovery of the lake would not occur immediately, but would accrue over time, as long as the nutrient concentrations in Mill Creek don't increase.

We have no data showing that *Ceratium* in Lake Hayes migrates vertically in the water column, drawing recycled nutrients from the bottom waters to the surface waters. However, this nutritional strategy, along with the ability to feed on bacteria, potentially allows Ceratium to persist and outcompete other algae when dissolved, inorganic plant nutrients (i.e., nitrate, phosphate, ammonium) are scares. Thus, studies have shown that dinoflagellates such a *Ceratium* may be indicators of reducing inorganic nutrient availability in lakes (Jeppesen et al. 2003; Gerdeaux & Perga 2006; Mehner et al. 2008).

If our hypothesis is correct and internal nutrient recycling is playing a diminishing role in fuelling algal blooms in the lake, then restoration actions aimed at reducing internal P loading/cycling will provide diminishing benefits into the future. Previous reports have recommended strategies to reduce internal P loading/recycling in the lake. However, in light of the information provided here, the restoration benefits of such actions into the future should be carefully scrutinised in terms of their costs and potential benefits. An analysis of the cost of alum treatment for Lake Hayes was carried out by John Quinn and Max Gibbs of NIWA in 2015 (Appendix 3), based on a maximum accumulation of dissolved reactive phosphorus in the bottom waters of 300 mg/m^3 of P, which was estimated based on previous measurements of P made in 1994/95 and 2012/13 (M. Schallenberg, unpublished data). However, more recent data and the analysis presented in Section 2 suggest that this may be an overestimate of the current maximum accumulated P by a factor of two. So, their estimated cost for an alum treatment of $535,000 (Appendix 3) may also be an overestimate by a factor of around two. Otago Regional Council samples being collected this summer will confirm whether or not the cost estimate in Appendix 3 can be substantially reduced.

While gains have been made in reducing nutrient loads to the lake from septic tanks, and nutrients in Mill Creek and the springs (Bayer & Schallenberg 2009), the condition of Mill Creek has stabilised since around 2005 (LAWA website; Appendix 4) and there are indications that summer nitrogen concentrations in Mill Creek may be increasing, although not yet statistically significant (LAWA website; Appendix 4). In comparison to other upland streams, Mill Creek is higher than average in nitrogen and *E. coli* (faecal bacteria) concentrations and in turbidity, while it is lower than average in phosphorus concentrations (LAWA website). Because Lake Hayes is an important and sensitive receiving environment, we suggest that the concentrations of contaminants flowing into Lake Hayes from all sources should be better than average. We, therefore, recommend that attention be refocused on land practices and nutrient, *E. coli* and sediment losses from the Lake Hayes catchment in order to further improve water quality in the lake (Appendix 4). The rapid rate of land development and the diversity of land uses in the catchment both highlight the importance of the use of best management practices and suggest that land/nutrient management mechanisms might be appropriate to protect the lake from future degradation. These should focus on phosphorus and nitrogen and should also account for fluxes during flood flows.

In the summers of 2009/10 and 2016/17 the lake did not experience algal blooms. This suggests that the lake's condition is destabilising and has begun to flip between algal blooms and clear water summer conditions. This kind of behaviour is a characteristic of complex systems as they approach pressure-response tipping points or thresholds. We hypothesise that the recovery from external and internal nutrient loading is creating instability in the lake and that the clear water summers are evidence that a stable recovery is attainable.

Table 1. Lake Hayes restoration actions discussed and recommended in four reports. Actions recommended in each report are shaded green.

Mitchell & Burns (1972)	Robertson (1988)	Bayer & Schallenberg (2009)	Ozanne (2014)	This report
Reducing external nutrient loading				
	• Wetland re-establishment • Reduce fertilizer application and runoff • Establish streambank buffer • Control channel clearance and drainage operations in catchment • Manage future development P load			**Collaborative catchment management plan** • Wetland re-establishment • Reduce fertilizer application and runoff • Establish streambank buffer • Control channel clearance and drainage operations in catchment • Manage future development P load
Change in catchment land use	Change in catchment land use			
Divert high P water out of the lake	Divert high P water out of lake		Divert high P water out of lake	
	Reduce runoff from animal stocking			
Reducing internal nutrient loading				
	Chemical P precipitation/inactivation	Chemical P precipitation/inactivation	Chemical P precipitation/inactivation	Chemical P precipitation/inactivation
	Hypolimnetic withdrawal	Hypolimnetic withdrawal	Hypolimnetic withdrawal	
Hypolimnetic aeration	Hypolimnetic aeration	Hypolimnetic aeration	Hypolimnetic aeration	
		Dredging	Dredging	
			Sediment oxidation	
			De-stratification	
Food web biomanipulation				
		Daphnia enhancement	*Daphnia* enhancement	*Daphnia* enhancement
Flushing				
Enhance flushing	Enhance flushing, remove flushing restriction at the outlet	Enhance flushing	Enhance flushing	Enhance flushing
Other in-lake strategies				
			Use of floating wetlands, algicides, pathogenic bacteria and ultrasound	

Why did the water clarity of the lake suddenly improve in these summers of 2009/10 and 2016/17? Data and observations suggest that the summer persistence of *Daphnia* in the lake was associated with the clear water phases. Current studies by the University of Otago Zoology Department and collaborators are examining the potential of perch recruitment to regulate *Daphnia* density via predation by juvenile perch on *Daphnia.* Preliminary evidence suggests that the perch life cycle and spawning time in the lake may be associated with crashes in *Daphnia* densities during the summers when algal blooms were severe (Appendix 1). The data in Appendix 1 outline compelling evidence that food web biomanipulation to reduce juvenile perch numbers and increase summer *Daphnia* densities could push the lake into a more stable clearwater phase. While research is ongoing, a key component of the research will be completed by the end of 2017 and we have begun to consider various approaches that could be used to reduce the numbers of young perch in the lake (see Appendix 1 for examples).

4.1 Restoration plan and timeline

Table 2 and Figure 11 outline a proposed restoration plan and timeline for accelerating the recovery of Lake Hayes to a stable desirable state and for controlling catchment development to prevent potential future increases in nutrient and sediment loading from the catchment to the lake. Table 3 sets out potential restoration targets for the recovery. These are suggested targets only because interested members of the community, iwi and stakeholders together with the Otago Regional Council would need to vet any final targets. We present the draft targets in Table 3 to help initiate a collaborative community project to set final restoration targets.

Table 2. Proposed restoration plan for Lake Hayes.

Action - in order of priority	Reasons/benefits	Time	Likely effectiveness	Cost	Cost effectiveness
Catchment management plan and actions	• High population growth rate • Diverse land uses • Valuable and sensitive receiving environment • Community education, collaboration and buy-in • Holistic improvement	• Start the process immediately • In 2017/18 undertake a management plan feasibility study • Continuing development and refinement over time	• Large number of diffuse and long term benefits including education, stakeholder involvement, and improvement in water quality • Effectiveness in improving water quality will depend on feasibility to be determined in catchment management plan	• Unknown, but some costs can be covered under normal Regional and District Council business	• Likely to be highly cost-effective in the long term
Biomanipulation	• Potentially restore eels to the lake • Potentially increase native bully and koaro densities in the lake • Reduce the density of non-native perch	• Immediately begin looking into options • In early 2018, undertake a feasibility study in light of information in Helen Trotter's MSc thesis (end of year) • Carry out biomanipulations in 2018	• Likely to be effective, but to what extent depends on feasibility study • Some techniques may require follow up or ongoing biomanipulations	• Unknown. Depends on techniques used • Unlikely to be expensive	• Potentially highly cost-effective • Some techniques may require ongoing biomanipulations to keep perch recruitment down

Table 2. Proposed restoration plan for Lake Hayes, continued.

Action - in order of priority	**Reasons/benefits**	**Time**	**Likely effectiveness**	**Cost**	**Cost effectiveness**
Flushing	• Increase flushing rate but requires upfront connection cost and continual water purchase	• Can be carried out immediately if cost-benefit is favourable	• Potentially helpful in the long-term	• $22,000 hook-up cost plus $35,000 p.a. for 200 L/s capacity	• Moderate long-term cost-effectiveness
Alum dosing	• Expensive, but likely to reduce internal P loading further	• Reassess costs in light of declining P concentrations in bottom water • If sufficiently cost-effective, could be carried out in summer 2017/18	• Effective immediately upon use • May require follow up treatments after 5 years	• Between $250,000 and $500,000, depending on reassessment using 2017 lake phosphorus data	• Low to moderate immediate cost-effectiveness

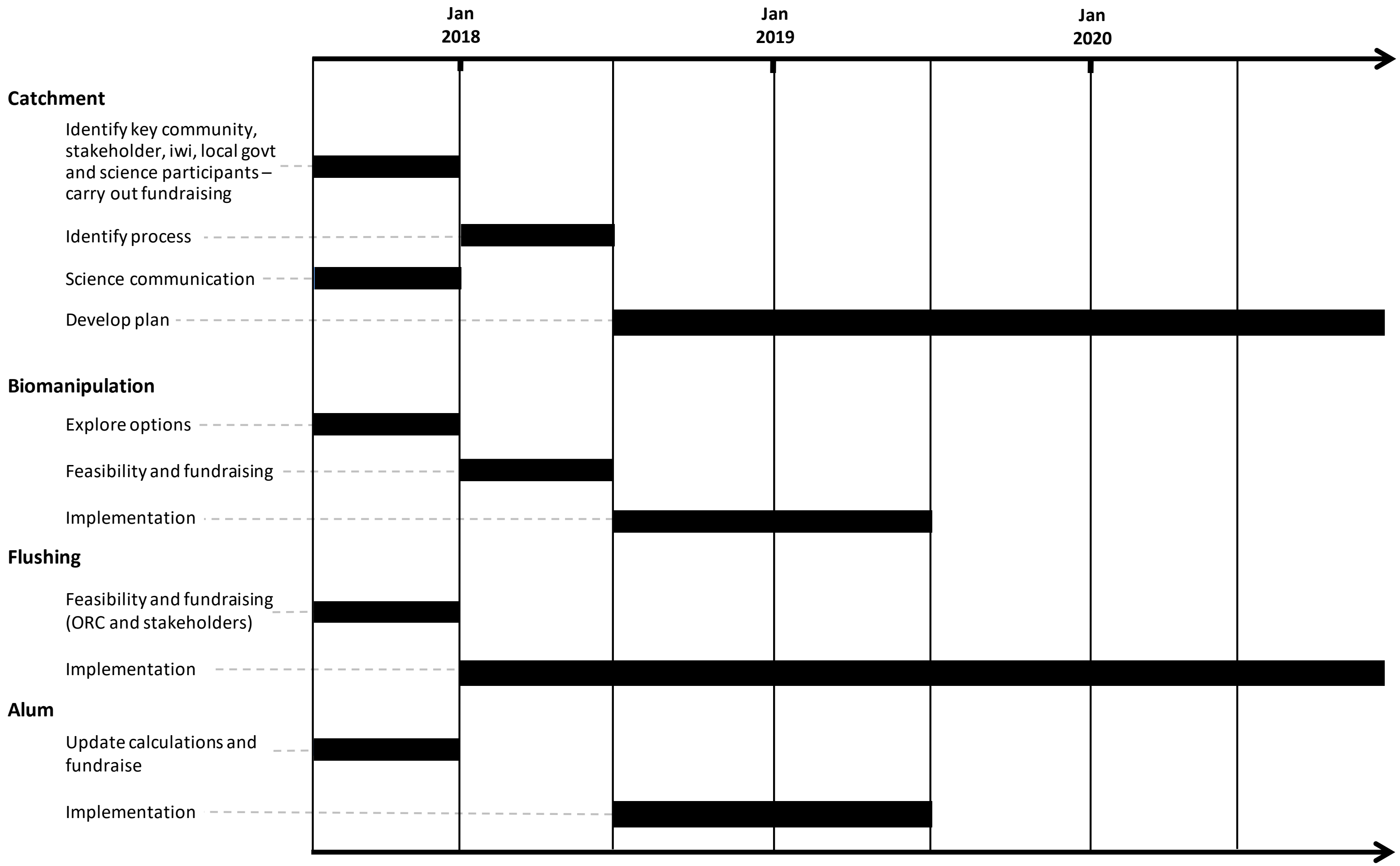

Figure 11. Proposed timeline for planning and implementation of lake restoration strategies.

Table 3. Suggested restoration targets for Lake Hayes. A successful restoration should achieve these targets consistently, from year to year.

Lake condition desired	Measurable targets to meet condition	Parties to help develop final targets
Lack of summer algal blooms and clear water which encourages recreational activities including swimming/bathing	• Trophic lake index during summer months < 4 • Secchi disk depth in summer months > 4 m	Otago Regional Council, locals, Fish & Game, rowing clubs, etc.
Inhibition of internal nutrient loading	• Improving oxygen content of the summer bottom waters to eventually achieve a condition where oxygen is never fully depleted • Nutrient budget shows a net loss of P from the lake on an annual basis, eventually stabilising at an ecologically sustainable level of P accumulation (permanent burial)	Otago Regional Council, University of Otago
Improved trout fishery	• Increase angler hours to 1995 levels • Reduce turbidity and temperature in Mill Creek • Maintain or increase Mill Creek flow rates	Fish & Game
Reduce nutrient loading to the lake from Mill Creek and the springs	• Reduce nitrate concentrations in Mill Creek • Reduce suspended sediment (turbidity) and total phosphorus concentrations in Mill Creek during high flow events • Develop a catchment management plan to ensure improved land management practices are adopted throughout the catchment	Otago Regional Council, Fish & Game and catchment land owners
Restore longfin eels to the lake	Achieve an eel trap and transfer target	Iwi, Contact Energy, DoC, Fish and Game
Maintain or enhance native biodiversity	Develop a strategy to prevent the incursion of invasive non-native species	Ministry of Primary Industries, DoC, Otago Regional Council

5 Water quality and lake health monitoring for Lake Hayes

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 from which to interpret and understand lake conditions and trends in the country. Despite the scientific interest it has attracted and the relatively good lake monitoring dataset that exists, the cumulative data assembled on the lake are barely enough to allow for a good understanding of the lake's changes over time.

5.1 The importance of regular and consistent monitoring

Lake Hayes and its catchment constitute a complex, linked terrestrial and aquatic ecosystem, which is highly valued by the local community. The water quality monitoring of the lake by the Otago Catchment Board/Otago Regional Council has been sporadic. Therefore, the dataset for the lake has many gaps and changes in sites and lake depths sampled, making it difficult to extract robust long-term data and to derive clear interpretations of how the lake has responded. The pitfalls of intermittent sampling can be seen in the dataset, where Secchi disk and trophic level indicators were not sampled in the summer of 2009/10, when the lake experienced a highly unusual and dramatic shift to what was likely an oligotrophic condition. This illustrates the potential for the lake to undergo rapid changes from year-to-year. Such dynamics are important to understanding the condition of the lake and its trajectory over time. Variability over time is an important indicator of lake condition, and trend and variability can best be assessed based on regular, long-term sampling of the lake's state.

5.2 The importance of nutrient budgets

Lake Hayes has undergone major changes in water quality as a result of major changes in nutrient loads over time. It's resistance to improvement from the 1980s to 2000s was in part due to the legacy of phosphorus loading that occurred in the 1960s and 1970s, which had been held in the lake bed sediments and recycled on an annual basis for decades. This illustrates the effect of legacies and time lags, which can affect lake condition. Consequently, the lake has probably rarely been in a nutrient equilibrium or steady state, where nutrient loads are balanced with nutrient concentrations and nutrient losses (via sedimentation and outflow).

Nutrient budgets involve the simultaneous intensive sampling of nutrient inputs (surface water and groundwater), nutrient concentrations (vertically resolved in a stratified lake) and nutrient losses. In the absence of such a nutrient budget (calculated at least on an annual basis), understanding of whether the lake is becoming cleaner or more polluted is only inferable via careful inference based on incomplete water quality data.

Nutrient budgets are a standard approach to understanding the nutrient sink/source dynamics of a lake in relation to nutrient loads from the catchment. To fully understand the condition and trajectory of a lake, it helps to understand if the lake is absorbing or shedding nutrients in relation to its nutrient inputs. Robertson (1988) calculated a P budget for the lake over a two-year period, which showed that the lake was retaining P. The current condition of *Ceratium* blooms is probably helping speed the shedding of legacy phosphorus from the lake bed to the outflow of the lake. Therefore, although the lake appears to be degrading, it is probably improving (flushing P), although only a current or recent nutrient budget can confirm this.

5.3 The importance of long term datasets

As described above, Lake Hayes and its catchment have probably rarely been in nutrient equilibrium over the past 50 years. The Secchi disk depth data, reaching back to the late 1940s show how water clarity has degraded, has been highly variable with a stable average for over 30 years, has further declined and has recently begun to swing sharply and intermittently from very low to extremely high clarity. This long-term information is very helpful in understanding many important characteristics of the lake such as its pre-degradation condition, how far the lake has departed from that condition, the time scales of change (which can indicate resistance to change), where the lake's tipping points might be, how close it might be to recovery, and so on.

Furthermore, long term data provide the best chance of identifying small and slow but important changes in the lake such as climate change impacts or depletion of the recycled phosphorus pool. The longer the dataset, the easier it is to distinguish ecological signal from stochastic noise in the dataset.

5.4 The importance of monitoring factors beyond simple water quality variables

Water quality (e.g., nutrients, water clarity, algal biomass) relates directly to the appeal of the lake for recreational activities like boating, fishing and swimming. So, it is understandable that statutory obligations for monitoring lakes focus mainly on water quality. However, to understand how and why changes to water quality occur in lakes, it is important to have information on supporting factors of the lake, which either drive or help explain changes in water quality. These could be related to climate change (temperature, mixing, etc.), which councils routinely monitor. However, key secondary factors often relate to components of the food web, such as the dominant phytoplankton species, zooplankton density, aquatic plant distributions, the presence of invasive species, etc. Early warning indicators of change in lakes are most likely to be changes in community composition of biological communities such as algae or zooplankton (Schindler 1987). For example, our interpretation of the data from Lake Hayes suggests that the development of *Ceratium* blooms in the mid-2000s both indicated an improvement in nutrient conditions and facilitated further improvement in nutrient status. One could not come to this conclusion based on water quality information alone.

5.5 The importance of monitoring change on different time scales

Regular monthly sampling of a lake allows for the analysis of lake changes on three important time scales: monthly, annual, and inter-annual. However, sometimes monthly sampling is too coarse a time scale to understand key drivers and processes. For example, climate warming should increase the period of time that a lake is thermally stratified. Changes in the timing and period of stratification can affect the ecology of the lake ecosystem and water quality (e.g., Winder & Schindler 2004). Unfortunately, monthly sampling will not reveal the exact timing of lake turnover or when the first day of seasonal thermal stratification occurs. So, monthly temperature profiles don’t help identify some key climate change-related effects that could affect the state of the lake. Other important factors, such as: i) when the lake’s temperature reaches a threshold for perch spawning or ii) when temperature becomes stressful to brown trout or iii) how rapidly dissolved oxygen in the bottom waters is depleted, can best be determined from high frequency measurements using *in situ* lake monitoring sensors.

5.6 The importance of monitoring at different places in the lake

In a lake with a simple basin shape and bathymetry, like that of Lake Hayes, it is tempting to think that monitoring only at the deepest site will be adequate. While one site can provide very useful data, water quality factors can vary substantially across Lake Hayes and with depth in the lake. For example, *Ceratium*, a motile alga, is often observed in distinct brown patches from the surface of the lake (Fig. 12). If the distribution of the dominant alga is patchy in the lake, sampling at a single site won’t give an accurate estimate of the biomass of algae in the lake.

Figure 12. Brown-coloured, patchy *Ceratium* bloom in Lake Hayes, summer 2006/07. Photo: Otago Fish & Game.

Jolly (1952) indicated that there was high spatial variation in the densities of zooplankton due to the effect of winds and currents. We have also observed high spatial variability in zooplankton distributions in the lake.

Ceratium has the ability to migrate vertically in the water column and is often seen to intensify in biomass from early morning to midday while zooplankton species also migrate vertically in the lake from day to night (Jolly 1952; James et al. 1992). So, monitoring of the lake should take into account such vertical movements of algae and zooplankton so that better estimates of biomass of these can be monitored.

5.7 Suggested monitoring for Lake Hayes

While the historical Otago Catchment Board/Otago Regional Council data has been valuable for explaining changes in water quality and nutrient dynamics, lake monitoring could be improved along the lines discussed above. Comprehensive and good quality lake monitoring data are especially important in the context of lake restoration because the effectiveness and cost-effectiveness of restoration actions can only be ascertained by careful monitoring of changes in the lake. Furthermore, year-to-year variation in factors such as temperature, mixing depth, timing of stratification, zooplankton dynamics, fungal pathogen dynamics and the effect of floods can impact on expected recovery and such dynamic behaviour needs to factored into assessments of restoration success.

Below, we present some options for improving the monitoring of the water quality and health of Lake Hayes to support restoration actions.

Table 3. Suggested lake monitoring approaches in order of priority.

Priority	**Type of monitoring**	**Frequency and technology**
1a.	Sampling by boat at 2 deep water sites (31m and c. 26m) 1. CTD datasonde casts (Temp, DO, Chl *a*, phycocyanin) 2. Samples at 5m, 10m, 15m, 20m 25m, 30m for: • Total, dissolved inorganic N and P • Chlorophyll *a* and pH (only at 5m) 3. Samples at 5m, 10m and 15m for phytoplankton species 4. Vertical zooplankton hauls for species and density of Daphnia 5. Secchi depth	Monthly; various standard methods
1b.	P budget Measure total P and flow rate (where relevant) in: • Mill Creek (plus flow) • Spring (plus flow) • 6 depths in the lake at 31m site (1a.) • Hayes Creek outflow (plus flow)	Monthly; standard wet chemistry methods
2.	Profiling lake monitoring buoy at 31m site • Temp • DO • Chl *a* • Phycocyanin (cyanobacteria)	Hourly; Limnotrack monitoring buoy
3.	Survey aquatic plants using divers (e.g., LakeSPI) At 4 fixed transects record: • Maximum depth of plants • Native species distributions and % cover • Presence and cover of non-native species • Health of plants	Every 5 years; Scuba divers (e.g. LakeSPI methodology)

Priority 1a. monitoring covers the basic water quality parameters at 2 sites and 6 depths on a monthly basis. This protocol accounts for spatial variation in the lake by sampling 2 sites and for vertical variation by sampling 6 constant depths and obtaining some more detailed information using a profiling CTD datasonde. It recommends sampling the phytoplankton and zooplankton communities, which provides valuable information on subtle changes in biological communities that can be related to subtle environmental changes. Currently the Otago Regional Council is using a monitoring programme similar to this, but at a single site. The Otago Regional Council monitoring effort has not been consistent on an annual basis, instead involving a cyclical monitoring rotation in which the lake is monitored for three years and then not monitored for a number of years. A regular, annual and long-term monitoring commitment for lakes yields better datasets and leads to a better understanding of lake behaviour and management requirements.

Adding priority 1b. measurements of total P in the inflows and outflows of the lake would provide the data required to calculate a phosphorus budget for the lake. This would indicate the internal and external P loads to the water column and the amounts of P lost via the outflow and via permanent burial in the sediments. With this information, it would be possible calculate a P budget (e.g., Robertson 1988) to determine whether the lake is a sink or source of P in a particular year, enabling the tracking of how the lake is responding to historical and current P loads.

Priority 2 monitoring provides high frequency vertical profiles of selected variables. This would allow the timing of various important events such as stratification/destratification to be identified. In addition, the timing and position in the water column where temperature and DO thresholds are exceeded can also be identified and compared across years. This would provide valuable background information for interpreting various stressors and drivers of changes in the lake.

Priority 3 monitoring focuses on the important littoral (submerged plant) zone of the lake, which is the primary habitat for many important aquatic organisms. It would allow for the determination of the health and biodiversity of the aquatic plant community, the distribution of plants in relation to water depth (indicating whether the plant community is shallowing, stable or deepening in the lake), as well as the status of non-native aquatic plants in the lake. The Otago Regional Council has had such aquatic plant surveys carried out in 1992 and 2001, but to our knowledge, these have not been carried out since. As a result, we have little understanding of how *Ceratium* blooms have affected plant distributions and associated fish habitat in the lake.

The comprehensive monitoring programme described here would provide information on lake health at different spatial and time scales, enabling the careful monitoring of changes in algal blooms, dissolved oxygen, trophic state (nutrient status), climate-related effects, phytoplankton, zooplankton, aquatic plant communities and invasive species within these communities. It would also provide data required to calculate nutrient budgets for the lake at monthly, annual and longer time scales. Together these monitoring strategies would be capable of sensitively tracking changes in the lake and would also provide valuable background information for targeted research on lake ecology. Finally, long term data are extremely valuable in understanding lake functioning and trends and. Therefore, there should be a commitment to continue regular monitoring over the long term, without hiatuses.

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Appendix 1: Food web biomanipulation as a restoration tool for Lake Hayes

Food web cascades in lakes

Studies overseas and in New Zealand lakes have found that lakes can undergo rapid changes in water quality if nutrient loading is pushed beyond certain tipping points and when invasions by non-native fish and aquatic plant species cause major changes in lake food webs and nutrient cycles (Scheffer 2004; Schallenberg & Sorrell 2010). Once in a degraded, eutrophic state, lakes can exhibit inertia and resistance to restoration by nutrient reduction alone (Fig. 1).

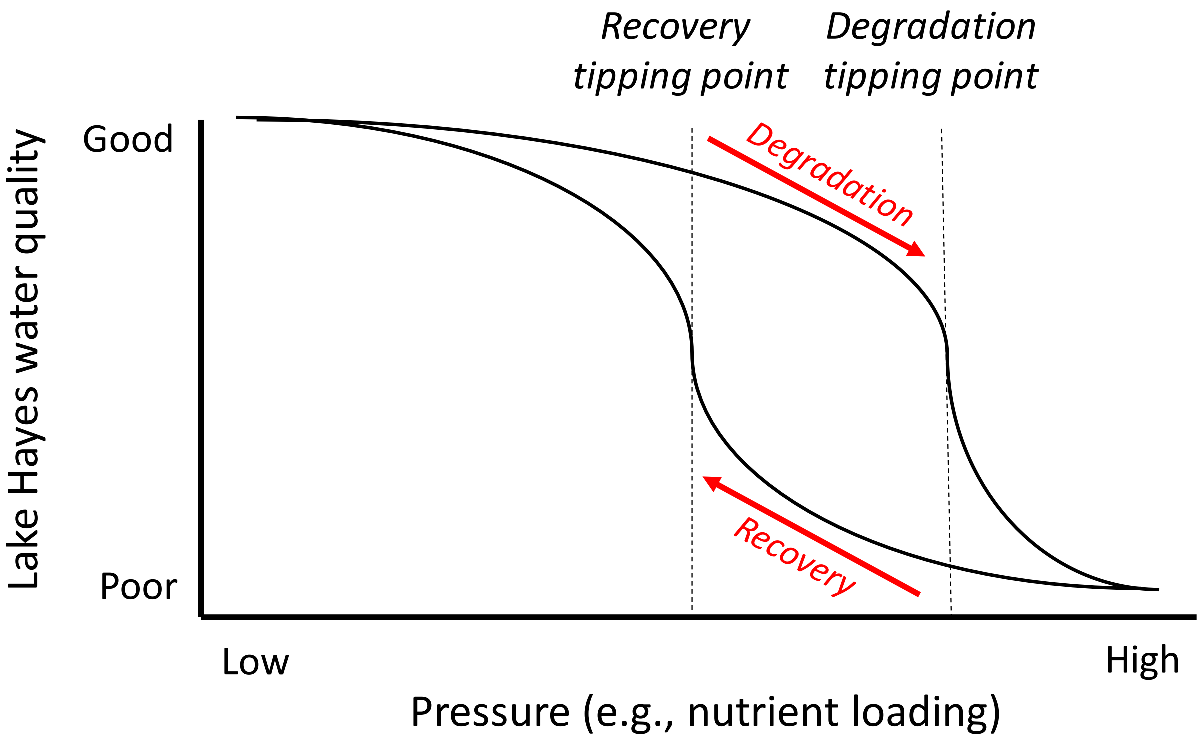

Figure 1. Idealised, non-linear relationship between water quality and environmental pressures in lakes, showing tipping points and inertia to degradation and to recovery that are typical of many shallow lakes.

However, in concert with nutrient load reduction, food web manipulations (fish stocking/removal) have been shown to successfully assist a return to a clear water state, usually by releasing zooplankton (which graze on algae) from predation pressure by fish (Søndergaard et al. 2007). Such food web biomanipulation aims to initiate, or strengthen, a top-down trophic (food chain) cascade by reducing zooplanktivory, resulting in an increase in the abundance and size of zooplankton (Shapiro 1980; Burns et al. 2014). If the zooplankton is dominated by large species (e.g. *Daphnia* sp.), increased grazing pressure on phytoplankton and ultimately increased water clarity result from a successful biomanipulation of the lake food web (Reynolds 1994) (Fig. 2). A cascade to reduce algae biomass may focus on increasing piscivorous fish density and/or decreasing zooplanktivorous fish density.

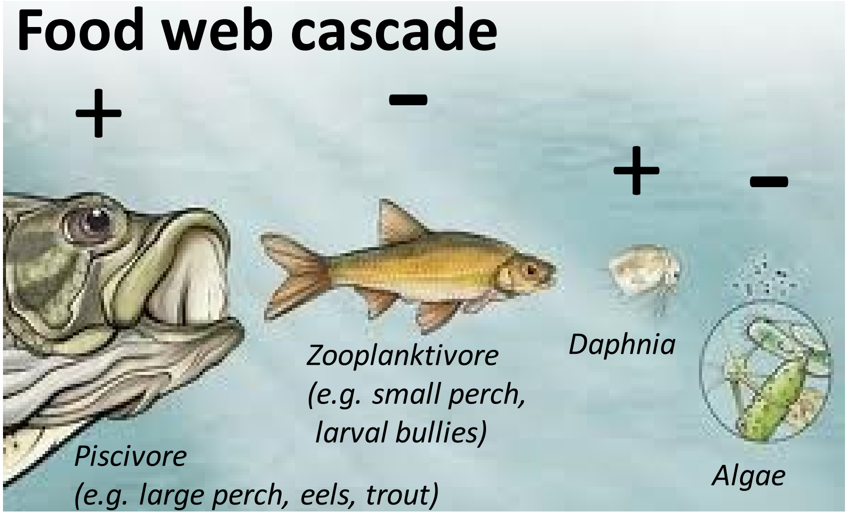

Figure 2. Example of a food web cascade showing how increasing the density of piscivorous fish can cascade (left to right) to reduce densities of algae.

Evidence for a link between *Daphnia* persistence in summer and the inhibition of *Ceratium* blooms in Lake Hayes

In the summers of 2009/10 and 2016/17, Lake Hayes exhibited unexpected, rapid improvements in water clarity and quality. Studies by the University of Otago Zoology Department revealed that the invasive zooplankter, *Daphnia 'pulex'* (Fig. 3), had colonised the lake, probably in the mid-2000s, and was attaining higher densities than had been previously recorded when the native *Daphnia thomsoni* was the sole *Daphnia* species in the lake (Fig. 3). Work by Professor Carolyn Burns of the Zoology Department has shown that *Daphnia 'pulex'* has higher temperature preferences than *Daphnia thomsoni* (called *D. carinata* in her study) and reaches higher densities than the native species (Burns 2013).

Figure 3. *Daphnia 'pulex'* from Lake Hayes, February 2010. Photo: Ciska Overbeek.

High *Daphnia* densities in 2010 occurred at the time when Lake Hayes attained and sustained unusually high water clarity throughout the summer (Fig. 4; Main Report - Fig. 5), suggesting a link between *Daphnia* summer density and summer algal biomass in the lake. In support of this apparent link between high *Daphnia* density and clear water in summer, we have observed that in the summers of 2012/13 and 2015/16, when *Ceratium* blooms occurred, *Daphnia* were absent in the lake during summer. In addition, zooplankton samples collected during the summer of 2016/17 (another summer with unusually high water clarity in the lake, Main Report – Fig 5), *Daphnia* continued (densities not yet determined).

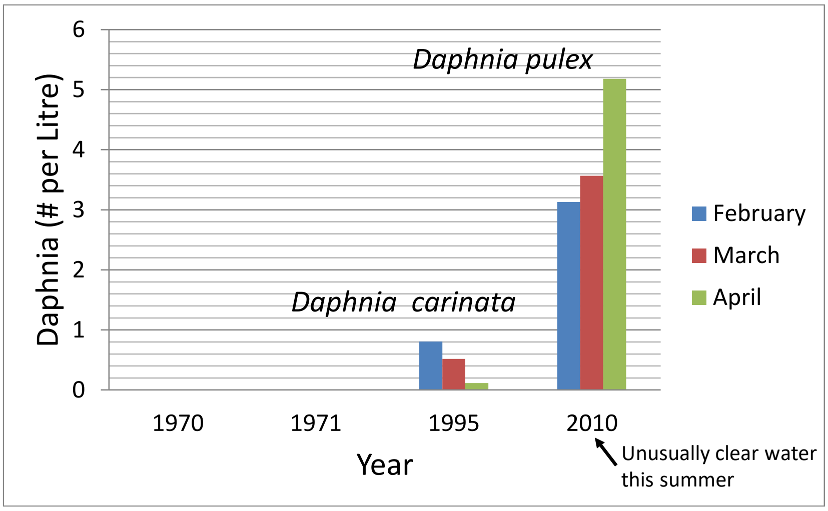

Figure 4. *Daphnia* densities in summer months in Lake Hayes in 1970 and 1971 (no *Daphnia* present in summer), 1995 (low densities present in summer), and 2010 (high densities present in summer).

The potential role of small perch in determining water quality

The clear water summer of 2009/10 did not persist into the summer of 2010/11. Observations of high juvenile perch numbers in 2010/11, suggest that perch recruitment increased in response to the plentiful *Daphnia*, and it is hypothesised that *Daphnia 'pulex'* density (and grazing on phytoplankton) was suppressed by predation on *Daphnia* by juvenile perch. These findings have prompted further investigations into the potential of biomanipulation as an approach to improve the water quality of Lake Hayes.

To test this hypothesis and to confirm that biomanipulation of *Daphnia* densities could facilitate a switch to a clear water state, studies by the University of Otago Zoology Department and collaborators from other research institutes and universities are underway to determine the strength of food web interactions between fish, zooplankton and phytoplankton in Lake Hayes. The relative importance of nutrients versus the food web structure in controlling phytoplankton abundance is being examined in order to evaluate the potential for food web manipulation to facilitate a shift back to a stable clear water state in Lake Hayes.

While declining water quality in New Zealand has attracted many millions of dollars of clean up funds for lake restoration, biomanipulation of the pelagic food web has only been attempted in one other New Zealand lake, the Lower Karori Reservoir, Wellington (Smith & Lester 2007). In this very small reservoir, the removal of perch resulted in enhanced zooplankton densities, reduction in algal biomass, and improved water quality (Burns et al. 2014), suggesting that reducing predation of small perch on *Daphnia* in Lake Hayes may help improve water quality in the lake.

Ideally, the lake food web models produced from this research will provide confidence in applying a biomanipulation approach in the lake. Key information needed to develop a successful biomanipulation approach includes:

1. Which fish species in the lake are responsible for reducing summer *Daphnia* densities, and at what size and life stage does predation on *Daphnia* occur?
2. How might *Daphnia* reduce *Ceratium* biomass? The large size of *Ceratium* in relation to *Daphnia* food size preference suggests that the interaction between *Ceratium* and *Daphnia* is not a direct grazing effect of the zooplankter on the algae. Rather *Daphnia* grazing on bacteria may reduce bacterial prey available for *Ceratium*. Alternatively, strong *Daphnia* grazing on other algae may increase light penetration and oxygen production in the deeper waters of the lake, restricting the upward flux of phosphorus toward the thermocline. There are other possible interactions.
3. What are cost-effective approaches for reducing zooplanktivorous fish densities (i.e., young perch) in spring and early summer?

Helen Trotter is currently doing an MSc thesis study to answer some of these questions. Her preliminary data from 2015/16, show that the recruitment of perch in early summer coincided with a sharp decline and eventual disappearance of *Daphnia* from the lake by January (Fig. 5). When her study is completed (end of 2017), sufficient information will be available to help produce a feasibility study for a biomanipulation intervention to cause a favourable food web cascade in Lake Hayes.

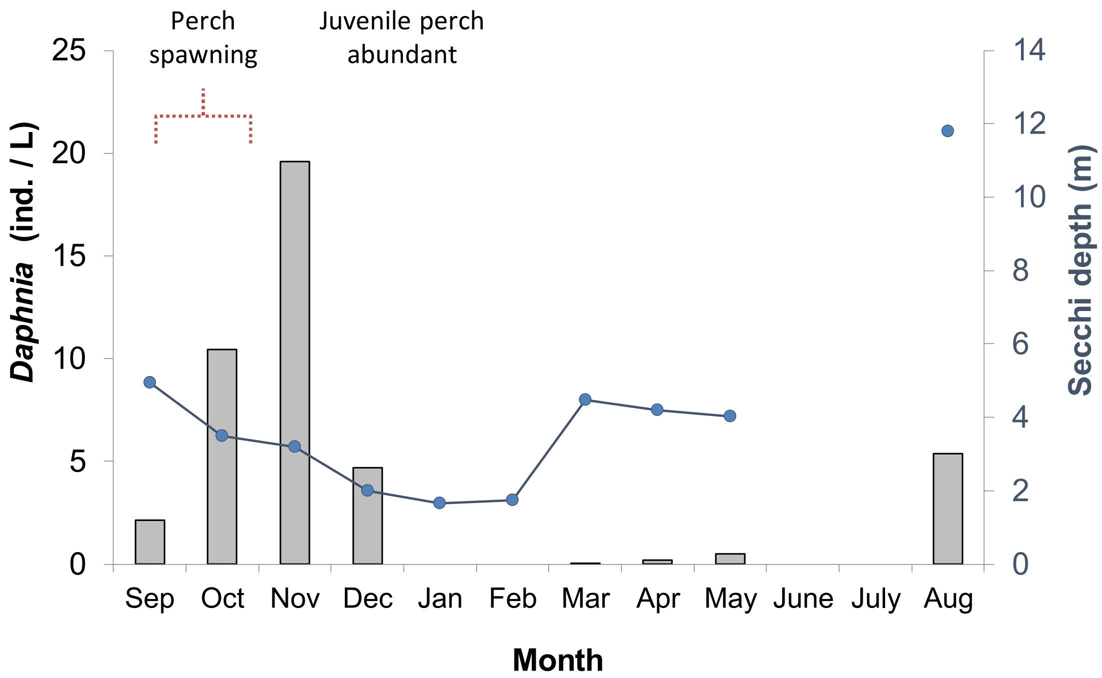

Figure 5. *Daphnia* density (bars) and Secchi disk depth (line) during the period September 2015 to August 2016. Note that when juvenile perch become abundant in December and January, the *Daphnia* decline and disappear from the lake, resulting in a decrease in water clarity to around 2 m. Data from Helen Trotter, University of Otago.

Potential ways to stimulate an effective food web cascade

Below we briefly introduce some potential approaches that could be used to induce a favourable food web cascade in Lake Hayes. These approaches could be further tested with respect to cost-effectiveness once Helen Trotter's study is complete.

1. **Adding *Daphnia* to the lake**. This is not seen as a practical option because of the large numbers of *Daphnia* that would be needed and the size of the facility that would be required to breed up such high numbers of *Daphnia*. *Daphnia* are intense grazers and require large amounts of food, further challenging the ability to produce the numbers that would be needed to affect algae in the lake. Furthermore, unless controls were also placed on planktivorous fish in the lake, the addition of large numbers of *Daphnia* would simply provide more food for planktivorous fish.
2. **Removal of juvenile perch**. This is not seen as practical because of the large size of the lake and because of the presence of aquatic plants in the shallow zones of the lake, which would make large-scale netting ineffective.
3. **Stocking of piscivorous trout and eels**. This may be a cost-effective and practical option. Fish to be stocked should be large enough to no longer feed on *Daphnia*, but instead feed on small fish (zooplanktivores). Fish & Game Otago may be able to raise hatchery brown trout to contribute to such a project. Longfin eels would have recruited into Lake Hayes prior to the construction of the Roxburgh Dam (1957). Subsequent to the dam, recruitment of eels, and eel biomass will have been severely or even completely reduced. Working with Contact Energy to trap and transfer returning eels from the Roxburgh Dam to Lake Hayes may have some merit because eels are effective piscivores capable of preying on young perch.
4. **Rearing and stocking of large, piscivorous sterile perch**. Perch longer than c. 150mm are known to be piscivorous. In some fish species (and possibly also in perch), it is possible to induce sterility by managing temperature of the developing ova (eggs). It may be possible to rear sterile perch to a large enough size so that they shift from zooplanktivory to piscivory. Stocking sterile piscivorous perch into Lake Hayes could reduce juvenile perch numbers through cannibalism, which is common in perch.
5. **Removing perch ova (eggs) from the lake**. Artificial perch spawning substrates could be built and deployed in the lake to attract perch spawn. Once spawning time is over but before egg hatching, the substrates could be removed, reducing perch recruitment. This strategy could complement strategy 4, by providing perch eggs for rearing into piscivorous perch.
6. **Encouraging fishing of perch**. Encouraging the catching of perch could possibly reduce perch numbers in the lake. If spawning perch could be targeted, this could potentially reduce perch recruitment. However, removing large perch from the lake could induce stunting in the population, which could increase predation pressure on *Daphnia* (see section 1.3.2. of the main report).

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Appendix 2: A preliminary assessment of the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water to speed the recovery of the lake

This Appendix is updated from: Schallenberg M. (2015) A preliminary assessment of the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water to speed the recovery of the lake. University of Otago Limnology Report No. 18, prepared for the Friends of Lake Hayes. Oct. 30, 2015.

Background

Lake Hayes usually undergoes thermal stratification from September to May or June. During this period, the warmer surface water is separated from the denser, colder water at the bottom of the lake. Due to the breakdown of algal material which settles to the bottom, the oxygen content of the bottom water declines during the stratified period, with the lake bed beginning to become anoxic in December to January (Fig. 1).

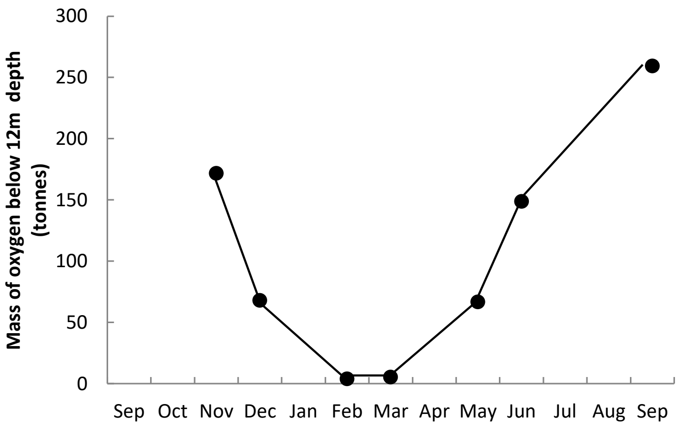

Figure 1. Mass of oxygen in the bottom waters (below 12m) of Lake Hayes, summer 2012/13.

As this occurs, phosphorus, which is bound to the sediments when oxygen is present, becomes liberated from the sediment and diffuses into the bottom waters and accumulating there until the end of the stratified period (Fig. 2).

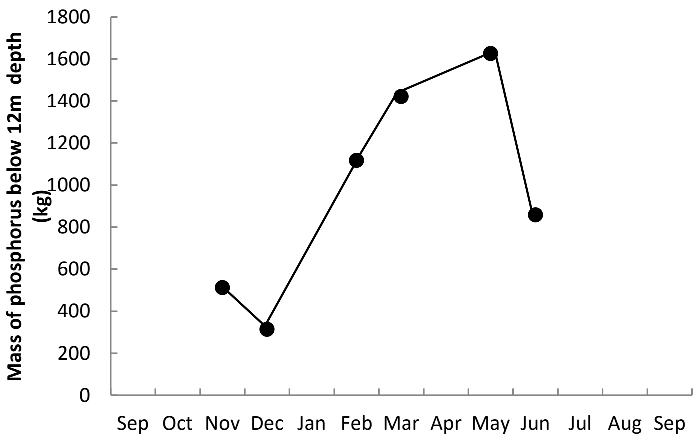

Figure 2. Mass of phosphorus in the bottom waters (below 12m) of Lake Hayes, summer 2012/13.

Stratification usually breaks down in June when the lake again mixes from top to bottom and phosphorus is diluted and also re-bound to particles in the water column.

The Friends of Lake Hayes have been examining potential methods for restoring Lake Hayes. A proposal has been put forward to help speed the recovery of Lake Hayes by augmenting the inflow to the lake at Mill Creek with water from the Arrow River Irrigation Scheme, sourced from the Arrow River near Macetown (Fig. 3).

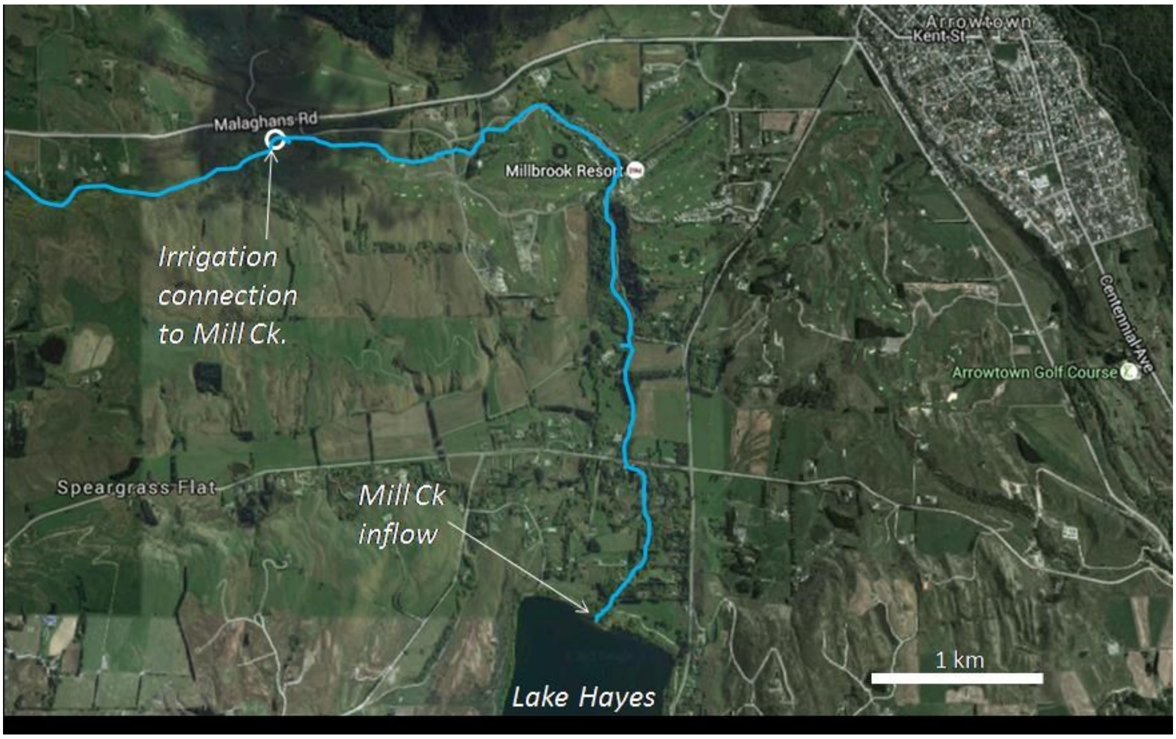

Figure 3. Map of the Lake Hayes area, showing Mill Creek and the potential connection point of the Arrow River Irrigation Scheme.

In this preliminary report, I use available data to try to answer four key questions regarding this potential restoration idea: 1. Could the augmented inflow flush displace substantial amounts of water from the lake? 2. Would the augmentation flow displace bottom water? 3. Could the augmented inflow supply enough dissolved oxygen to the bottom water to prevent its deoxygenation and, thereby, prevent P release from the sediments? 4. How much P and chlorophyll *a* could the augmented flow flush from the lake and what effect would this have on trophic state?

1. Could the augmented inflow displace substantial amounts of water from the lake?

This proposal would increase the flushing of the lake, which currently replaces its water roughly every 1.8 years (Caruso 2000). If the Arrow River water is more dilute than the lake water (with respect to phosphorus), then the flushing effect could remove some of the recycled phosphorus from the lake by displacement. The magnitude of the enhanced flushing effect would be proportional to: 1. the difference in nutrient concentrations between the Arrow River and the lake water that it displaces and 2. the amount of water available for flushing.

Recent ORC data show little difference in TP in surface and bottom waters (see main report, Fig. 8). Below, I have examined how beneficial the augmented flow could be for flushing phosphorus from the lake.

For these calculations, I have used the following information:

1. Available Arrow River flows: 200 litres per second for September, October, April, May and June. 100 litres per second for November to March (inclusive) (Table 1)

2. Arrow river phosphorus concentrations (Otago Regional Council data; Table 2)

3. Lake temperature profiles (University of Otago; Fig. 4)

4. Lake phosphorus concentrations from summer 2012/13 (University of Otago)

Table 1: Available water from the Arrow River Irrigation Scheme (info provided by Rob Hay).

Month	Cubic m per day	Cubic m per month	Cumulative irrigation inflow
Sept	18000	540000	540000
Oct	18000	540000	1080000
Nov	9000	270000	1620000
Dec	9000	270000	1890000
Jan	9000	270000	2160000
Feb	9000	270000	2430000
March	9000	270000	2700000
Apr	18000	540000	2970000
May	18000	540000	3510000
June	18000	540000	4050000

Table 2. Typical phosphorus concentrations of the waters of Lake Hayes (University of Otago) and the Arrow River (Otago Regional Council data from site at Morven Ferry Rd.).

Month	Lake Hayes surface water TP (µg/L)	Arrow River TP (µg/L; ORC data*)
Nov	27	14
Dec	52	9
Feb	47	7
March	116	8
May	43	9
June	69	5

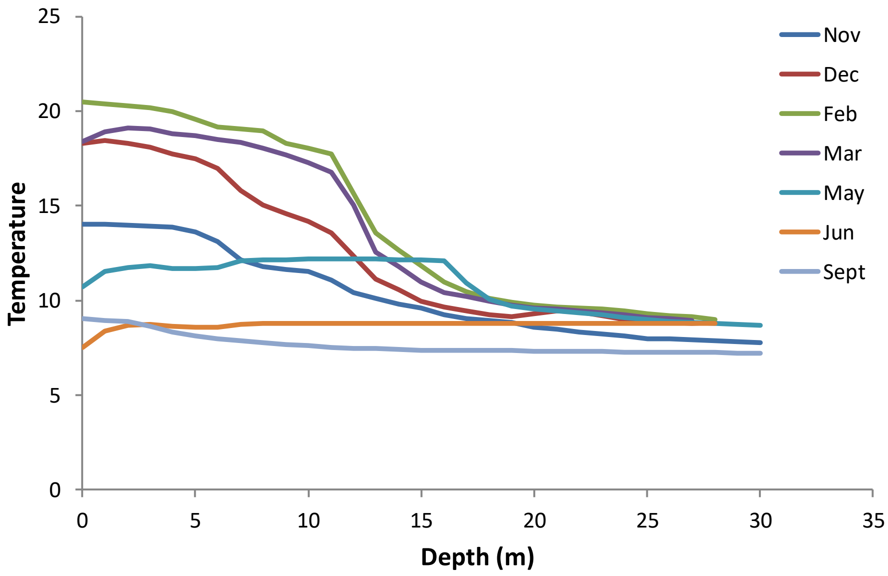

Figure 4. Lake temperature profiles from the summer of 2012/13.

Table 3 shows that the P concentration in the Arrow River is much lower than the lake P concentration of the lake, indicating that the Arrow River water would be suitable for the dilution and displacement of P-rich lake water.

Using the above information, I calculated the cumulative input of Arrow River water from September to June and compared that with the lake volume. I calculated this cumulative flushing volume as a percent of the whole lake volume.

The calculations show that the flushing effect of the Arrow River augmented inflow would displace a small percentage of the lake volume – only approximately 7% of the whole lake volume by the end of the stratified period (Fig. 5).

While these flushing effects are not substantial, they are not insignificant and could, over many years help reduce lake P concentrations and recycling. However, the addition of Arrow River water to Mill Creek could increase the loading rate of dissolved inorganic nutrients to Lake Hayes during the summer months (DSIR 1973), when these are often in very low concentrations in the surface water of the lake. Therefore, even though the concentrations of added nutrients from the proposed augmentation might be small, and the net P and N balance might be negative, it is possible that the addition of small amounts of available N and P could have a somewhat stimulatory effect on the lake's phytoplankton during summer. Therefore, before this restoration method is employed, further thought should be given to this potential stimulatory effect.

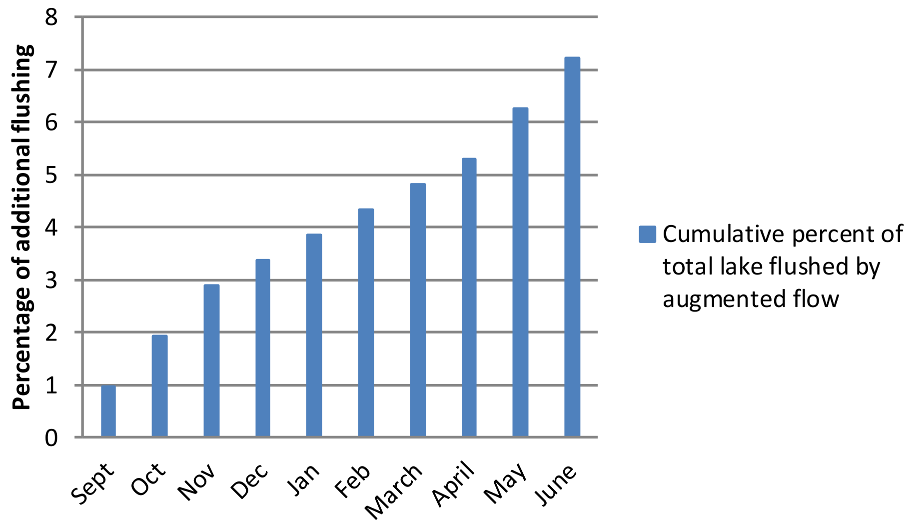

Figure 5. Cumulative proportion of the total lake water that could be flushed by Arrow River water, using the maximum amount of augmentation water available (200 L/s in shoulder seasons and 100 L/s in summer).

2. Would the augmentation water displace bottom water?

The colder the water, the denser it is (this is true down to 4°C). So, to displace the colder bottom water of Lake Hayes, the combined Mill Creek/Arrow River inflow would have to be colder than the surface layer of the lake and, ideally, it should be as cold/dense as the bottom water of the lake.

For these calculations, I have used the following information:

1. Available Arrow River flows: 200 litres per second for September, October, April, May and June. 100 litres per second for November to March (inclusive) (Table 1)

2. Lake temperature profiles (University of Otago; Fig. 4)

3. Mill Creek temperatures (Otago Regional Council data; Fig. 6).

I have assumed the following for these calculations:

1. The combined Mill Creek/Arrow River inflow would be the same temperature as the current Mill Creek inflow.

To test whether the inflow would be likely to plunge to the bottom layer of Lake Hayes, I compared the temperatures of Mill Creek with the temperatures of the lake, over the stratified period (Fig. 6). The data show that only toward the very end of the stratified period (in May), does the temperature of Mill Creek. approach that of the bottom water of the lake. Prior to that time, the inflow would either flow into the warm surface water or would flow between the layers (but not enter the bottom water layer).

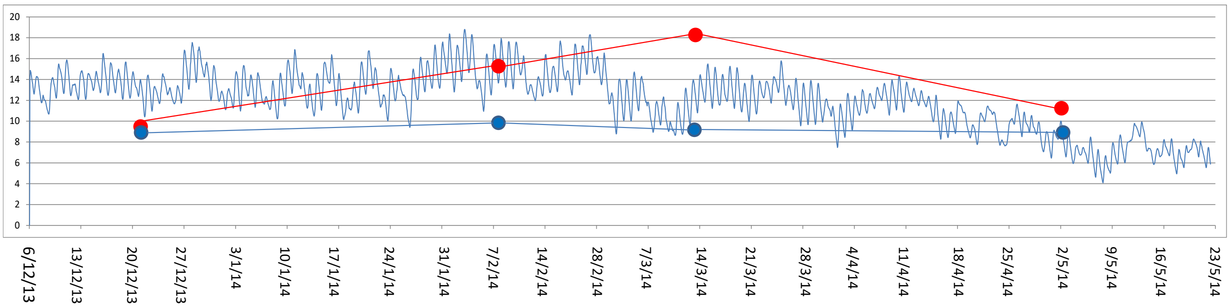

Figure 6. Temperature data for Mill Creek (blue line; 2013/14) and Lake Hayes (blue and red lines with dots; 2012/13). The blue dots show the lake bottom water temperatures and the red dots show the lake surface water temperatures. Mill Creek data were supplied by the Otago Regional Council.

Addressing the above assumption, is it possible that the temperature of the Arrow River augmented flow might lower the temperature of Mill Creek enough to allow both volumes of water to plunge into the bottom of Lake Hayes? Unfortunately, we don't have temperature data for the Arrow River at the offtake site or at the site where the irrigation water would connect to Mill Creek. This connection site is 4 km upstream from where Mill Creek enters Lake Hayes (Fig. 3), so even if the Arrow River water were substantially colder than Mill Creek, by the time it was transported from near Macetown to the Mill Creek connection site, diluted by Mill Creek and then transported 4 km downstream, any temperature benefit from the Arrow River is likely to have been lost. However, I have not been able to confirm this with data or modelling.

3. Could the augmented inflow supply enough dissolved oxygen to the bottom water of Lake Hayes to prevent its deoxygenation?

Another potential benefit of the injection of Arrow River water into the bottom waters of Lake Hayes is that the addition of oxygenated Arrow River water to the bottom waters of the lake might prevent deoxygenation of the bottom waters, maintaining P binding in the sediment of the lake.

For these calculations, I have used the following information:

1. Available Arrow River flows: 200 litres per second for September, October, April, May and June. 100 litres per second for November to March (inclusive) (Table 1)

2. Lake temperature profiles (University of Otago; Fig. 4)

3. Estimates of the volume of 1 m-thick slices of Lake Hayes (calculated from the NZ Oceanographic Institute bathymetric chart)

I have assumed the following for these calculations:

1. The combined Mill Creek/Arrow River inflow would discharge into the bottom waters of Lake Hayes
2. That the combined Mill Creek/Arrow River inflow would have an oxygen content approximating 100% air saturation (i.e., equilibration with the atmosphere).

For these calculations, I cumulatively added the mass of oxygen that would exist in the Arrow River augmented flow over the period for which water would be available. This mass of oxygen was then compared to the mass of oxygen in the bottom waters of Lake Hayes during the same period (the stratified period). Figure 7 shows that the cumulative input of oxygen is only relatively minor compared to the oxygen holding capacity of the bottom waters of Lake Hayes (indicated by the September value, when the bottom waters were mostly oxygenated). The rate of oxygen supply to the bottom waters (the slope of the line = 0.0888 tonnes of oxygen supplied per day) is also small compared with the rate of oxygen loss from the bottom waters in spring and summer (from November-February; 1.93 tonnes of oxygen consumed per day). Thus, the rate of oxygen consumption in the bottom water is 22 times greater than the rate of oxygen supply which could be contributed to the Arrow River augmentation, if it were injected directly into the bottom waters. This indicates that injecting the Arrow River augmentation flow directly into the bottom waters would not overcome deoxygenation in this lake.

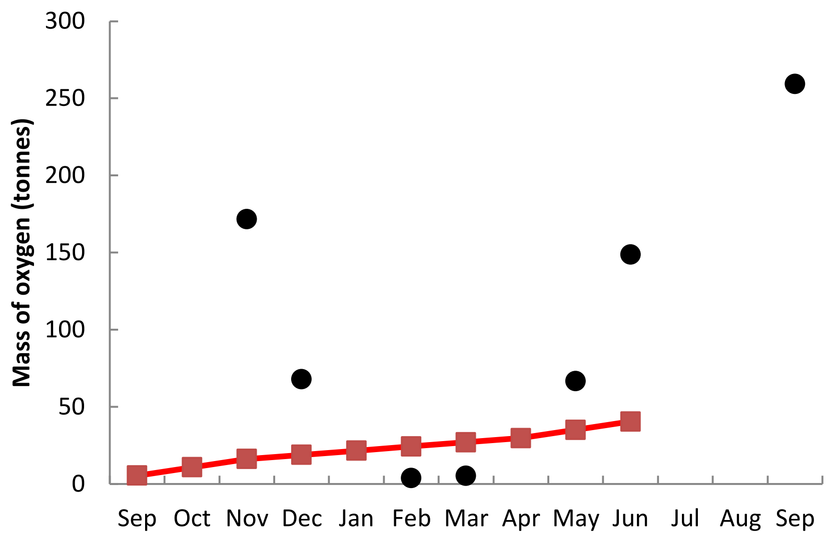

Figure 7. The mass of oxygen in the bottom water of Lake Hayes (2012/13; black dots) and the cumulative mass of oxygen estimated to be in the proposed augmented Arrow River inflow (red squares).

3. How much P and chlorophyll a could the augmented flow flush from the lake and what effect would this have on trophic state?

Displacement of surface water:

It appears from the above analysis in Section 2 that the augmented flow from the Arrow River would largely flow into the upper surface water layer of Lake Hayes. I calculated the amount of lake surface water that would be displaced by the cumulative input of Arrow River water from September to June. The volume of the surface water layer (to 12 m depth) is 31.03 million cubic metres, and the cumulative inflow from the Arrow River is 4.05 million cubic metres by the end of June. Thus, the Arrow River would displace around 13% of the lake's surface water over the stratified period.

Displacement of total phosphorus:

The average total phosphorus concentration in the surface water of Lake Hayes from September to June is 59 mg/m^3, while that in the Arrow River (at Morven Ferry) is 9 mg/m^3 (Table 2). The difference in concentration is 50 mg/m^3. When multiplied by the volume of the lake's surface layer and by the cumulative inflow from the Arrow River, respectively, the phosphorus in the lake displaced by the augmented flow would equal approximately 11% of the phosphorus content of the surface layer of the lake. This would bring the average phosphorus concentration in the surface water down from 59 mg/m^3 to around 52.5 mg/m^3, by the end of the augmentation period in June. The lake's trophic state would remain high as the boundary between mesotrophic (moderately productive) and eutrophic (productive) is 20 mg P/m^3. By these estimates of the average augmented lake phosphorus concentration, the lake would remain in the supertrophic category (48 – 96 mg P/m^3) (see Appendix 2.1). However, persistent flushing of this sort over a number of years could contribute to an improvement of the lake's trophic state.

Displacement of chlorophyll a (algal biomass):

The average chlorophyll *a* content of the surface water of Lake Hayes from September to June is estimated to be around 30 mg/m^3 (Bayer & Schallenberg 2009). We have no chlorophyll *a* data for the Arrow River, but this is expected to be quite low during moderate to low flow periods (probably not more than 2 mg/m^3 of chlorophyll *a* during the augmentation period). Again, multiplying by the volume of the lake's surface layer and by the cumulative inflow from the Arrow River, respectively, the chlorophyll *a* in the lake displaced by the augmented flow would equal approximately 12% of the chlorophyll *a* content of the surface layer of the lake. This would bring the average chlorophyll *a* concentration in the surface water down from 30 mg/m^3 to around 26.7 mg/m^3, by the end of the augmentation period in June. The lake's trophic state would remain high as the boundary between mesotrophic (moderately productive) and eutrophic (productive) is 5 mg $Chla/m^3$. By these estimates of the average augmented lake chlorophyll *a* concentration, the lake would remain in the supertrophic category (12 – 31 mg $Chla/m^3$) (see Appendix 1). However, persistent flushing of this sort over a number of years could contribute to an improvement of the lake's trophic state.

Caveats

There are a number of caveats that should be considered before employing augmentation flow from the Arrow River to help flush and, thereby, restore Lake Hayes. For example, the increased flow discharge and velocity of Mill Creek could increase stream bed erosion and reduce nutrient attenuation by stream periphyton due to the more rapid descent of water downstream to the lake. This would have the effect of increasing sediment and nutrient loads to the lake. In addition, some of the costly augmented flow could be lost to aquifer recharge in the catchment, in effect reducing the desired flushing effect. A groundwater hydrologist could advise on the likelihood of this occurring. Furthermore, as mentioned above, the augmentation flow, although low in nutrient concentrations relative to the lake, would likely add dissolved inorganic N and P to the lake during the summer months, when these nutrients are in short supply. This could stimulate phytoplankton production.

Summary

In Table 3, I summarise the information presented in this report and I show some issues to consider regarding the findings of the report. The above caveats should also be carefully considered before augmentation flow is employed for lake flushing.

Table 3. Summary of findings assessing the potential for Arrow River augmentation to speed the recovery of Lake Hayes.

Augmentation questions	Answer	Things to consider
1. Would it flush a substantial amount of phosphorus from the lake?	• Up to 11% per annum	• If internal load increases again, this could be useful if it could displace bottom water.
2. Would it naturally plunge into the bottom waters or would it flow into the surface waters of the lake?	• Naturally, the inflow is likely to be less dense than the cold bottom water, meaning it will flow over top of the bottom water, displacing and flushing surface water only.	• This conclusion assumes that the combined Mill Creek/Arrow River inflow would not be colder/denser than the current Mill Creek inflow. Temperature data are lacking to test this assumption.
3. If it were injected into the bottom waters, could it supply enough oxygen to prevent the bottom water from losing all of its oxygen during the stratified period?	• No, the oxygen augmentation effect is small compared to the oxygen demand of the bottom waters of the lake.	• In the calculations, I didn't include the oxygen that could also be supplied by the Mill Creek inflow. • Assuming that the Mill Creek discharge is around the same as the Arrow River augmented flow, and assuming that Mill Creek flows could also be harnessed and injected into the bottom waters of the lake, then the oxygen supply rate that I calculated would be doubled. • Injecting both these inflows into the bottom waters would still be insufficient to prevent deoxygenation of the bottom waters because the oxygen demand is around 10 times greater than the combined supply rate would be.
4. Could the augmented flow displace substantial amounts of phosphorus and chlorophyll *a* from the lake?	• The augmented flow would reduce the average surface water phosphorus concentration in the period from September to June by 11% and the chlorophyll *a* concentration by 12%. Neither of these reductions would reduce the trophic status of the lake from its current supertrophic condition.	• Persistent flushing of around 11% of the phosphorus and 12% of the phytoplankton from the lake per year could contribute to a speeding of the lake's recovery if maintained for a number of years.

Acknowledgements

I thank Rob Hay (Friends of Lake Hayes) for providing information on the available flows from the Arrow River irrigation scheme and for other background information about the proposed augmented inflow. I thank Dean Olsen from the Otago Regional Council for providing data on Mill Creek and

Arrow River temperatures and water quality. The New Zealand Ministry of Business, Innovation and Employment have funded this work via a subcontract with NIWA.

Reference

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DSIR (1973) The prospects for restoring Lakes Hayes and Johnson. Report prepared by the Freshwater Section of the Ecology Division of the DSIR. 6 p.

Appendix 2.1

Attribute ranges for different lake trophic levels. From Burns, N, Bryers, G, & Bowman, E (2000).<u>Protocols for monitoring trophic levels of New Zealand lakes and reservoirs</u>.Available from www.mfe.govt.nz.

Lake type	Trophic level	Chla ($mg\ m^{-3}$)	Secchi depth (m)	TP ($mg\ m^{-3}$)	TN ($mg\ m^{-3}$)
Ultra–microtrophic	0.0–1.0	0.13–0.33	31-23.5	0.84–1.8	16–34
Microtrophic	1.0–2.0	0.33–0.82	23.5-14.8	1.8–4.1	34–73
Oligotrophic	2.0–3.0	0.82–2.0	14.8-7.8	4.1–9.0	73–157
Mesotrophic	3.0–4.0	2.0–5.0	7.8-3.6	9.0–20	157–337
Eutrophic	4.0–5.0	5.0–12	3.6-0.7	20–43	337–725
Supertrophic	5.0–6.0	12–31	0.7-0.3	43–96	725–1558
Hypertrophic	6.0–7.0	>31	<0.3	>96	>1558

Appendix 3: A rough Lake Hayes alum dosing estimate

This Appendix is based on a report that was prepared by John Quinn, Max Gibbs and Chris Hickey (NIWA) in 2012.

The most common chemical method for capping phosphorus in lake bed sediment is to distribute alum (aluminium sulphate) solution into the lake, which flocculates and settles to the lake bed where it binds free phosphorus in the sediments, even under anoxic conditions. During the process of flocculation, alum also collects algae and suspended solids, clarifying lake water. Under conditions of pH > 6.5, alum can bind sufficient phosphorus to create conditions where restricted P availability limits algal growth. Alum applications have been successfully used in Lakes Okaro (Paul et al. 2008) and Rotorua (Hamilton et al. 2015) in the Bay of Plenty to reduce phosphorus concentrations in lake water.

Because of alum's flocculating capability, it is best used when the bottom waters are anoxic and phosphorus released from the sediments has accumulated to its maximum level (toward the end of the stratified period in Lake Hayes). As a sediment capping agent, the effect of a single appropriately-dosed treatment can last for 5 years and sometimes up to 20 years (Welch & Cooke 1999). Studies on toxicity of aluminium derived from alum to sediment-dwelling fauna and to fish indicate that as long as the lake pH buffering (i.e. alkalinity) is sufficient to preclude acidification, then toxic effects are minimal (Tempero 2015).

The least expensive way to deliver alum to lakes is to add it to inflowing tributaries. This will be most effective if the tributaries are colder than the lake water and carry the alum directly to the bottom waters where dissolved reactive phosphorus concentrations are highest and where phosphorus cycling is strongest. This is the approach used to deliver alum to Lake Rotorua (Hamilton et al. 2015).

Below is a rough calculation of the estimated cost of an alum treatment for Lake Hayes. The amount of P to be sequestered in this calculation is based on the maximum dissolved reactive phosphorus concentration at end of stratification in recent years (c. 300 ppb; M. Schallenberg, unpublished data) multiplied by the hypolimnetic (pertaining to the deep water layer) volume (28,937,495 m^3). This calculates a total hypolimnetic dissolved reactive phosphorus amount to be 8,681 kg, equating to 4.15 g $P/m^2$of hypolimnetic lake-bed area. When the top of the hypolimnion (deep water layer) is at 10m depth, around 78% of the lake bed is within the hypolimnion.

At a pH of 7 the aluminium-phosphorus binding is < 50%. If we conservatively assume 20% binding efficiency, then we will need an Al:P ratio of 5. Alum comes as 666.42 g/mol (octadecahydrate) containing 2mol of Al, which is 54g/mol, so the amount of alum needed is 535,343 kg. Alum comes in an aqueous solution of 47% alum, so the volume needed for the treatment of Lake Hayes would be 856,412 L- equivalent to ca. 40 standard (22,000L) water tanks full or ca. 20 tanker-trailer loads.

At a cost of alum solution at $1000/tonne, the cost of the alum for a Lake Hayes treatment under the above assumptions would be $535,343, which would provide an alum dose to the hypolimnetic sediments of 200 g alum per m^2 of hypolimnetic lake bed.

Dosing would occur during the stratified period (mid-October to May) by addition to the Mill Creek inflow stream which would ideally carry the alum into the bottom waters. Lake currents would provide further mixing of the alum floc around the lake. Studies carried out by NIWA in 2011/12 confirmed the existence of sufficient hypolimnetic currents to distribute the alum throughout the hypolimnion.

The lake is well buffered, so buffering the alum additions to avoid pH drops below 5 (which would cause concern regarding aluminium toxicity) would not be necessary.

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Appendix 4: Catchment management to restore and protect Lake Hayes

Importance of the Mill Creek catchment

Prior to major catchment drainage operations in 1961, there had been no reports of discoloured waters flowing into Lake Hayes and the bottom waters were aerobic during summer with little internal P release. After the wetland drainage and channelization works in the early 1960's, growing pressures from further catchment alterations and land use conversions continued and lake health further declined into the 70's and 80's. This prompted investigations into catchment nutrient loading and ways to mitigate the nutrient increases observed. The vast majority of nutrients enter the lake through the Mill Creek catchment (ORC 1995), with around 80% of the P load bound to soil particles which were historically mobilised through channel cutting and removal of the catchment wetlands which acted as sediment sinks. The high historical catchment P load had settled in the lake to become the internal P load which greatly contributed to the decline and maintenance of poor lake health seen over recent decades. Previous publications have outlined the importance of reducing nutrient concentrations in Mill Creek and noted this as a requirement for the successful restoration of Lake Hayes (Robertson 1988; ORC 1995).

Bayer & Schallenberg (2009) found nutrient levels in Mill Creek had decreased from the 1980's to 2009 however, more recent data (2005-present) available on the LAWA website, indicates the improving water quality trends in Mill Creek have since stabilised and may be, in some cases reversing. Fore example, summer nitrate and ammonia concentrations in Mill Creek appear to have been increasing since 2005 along with *E. coli* counts (Fig. 1). Compared with other upland rivers in New Zealand, levels of *E. coli*, turbidity, TN and nitrate are worse than the average upland river (LAWA 2017). The sensitivity and importance of this lake, together with the current and projected population growth rate and associated land use changes in the area, should recommend better than average water quality in the Mill Creek catchment.

Previous suggestions for catchment management

Robertson (1988) looked at the Lake Hayes catchment in detail, describing historical land use change and outlining potential mitigation measures that would lead to reduced inputs of P into the lake. Most of these recommendations involved controlling catchment landuse practices such as reducing fertiliser runoff, controlling channelling operations and managing future development in the catchment to ensure a reduced external P load to the lake. In addition, Robertson (1988) suggested many of what we now call on-farm best management practices (BMPs) such as reducing fertiliser use and establishing stream bank buffers. Improvement of land use practices in the Mill Creek catchment was recommended by the author to be highly important for the restoration of Lake Hayes.

In 1995 the Otago Regional Council (ORC) developed a Lake Hayes Management Strategy with the overall goal being to 'to improve the water quality of Lake Hayes, to achieve a standard suitable for contact recreation year round and to prevent further algal blooms' (ORC 1995). The strategy highlighted the major catchment issues affecting water quality and put in place relevant policies as well as outlined ambitious actions the ORC would take to reduce the P load in the catchment. Examples of these actions included negotiating with landowners around Mill Creek to establish riparian zones, advocating and assisting with the protection and re-establishment of wetlands and encouraging sustainable land use in the catchment, among many others. Some successes have since been documented, an example being the decommissioning of septic tanks in the catchment as encouraged in the strategy, however it is unknown how many actions have been carried out and how many policies have been implemented. Section 1.9 of the management strategy stated that reviews of the strategy will be undertaken at 5 year intervals, including an assessment of any changes in the catchment P load, however no notes on these can be found. It would be useful to document which policies and actions outlined in the strategy have been implemented and which are still required, to assess both the success of the management strategy to date and what remains to be done in the future, particularly in light of the rate of development in the catchment.

Wetland restoration/re-establishment

The two major catchment management directions actions recommended for Lake Hayes are wetland restoration/re-establishment, and on-farm BMP's (Robertson 1988; ORC 1995; Bayer 2009; Ozanne 2014). Wetland re-establishment was discussed by Robertson (1988) who noted the draining and channelization of numerous wetlands in the catchment during the 1960's led to a decrease in the sediment retention and buffering capacities of the Mill Creek catchment. The option for wetland restoration in the catchment was looked into to recover some of these lost ecosystem services and the cost was estimated at around $50,000 (Robertson 1988).

However, the Lake Hayes Management Strategy (ORC 1995) mentioned a report commissioned by the ORC looking into the viability of wetland re-establishment in the catchment which found it to be unfeasible. The main reasons given were the long calculated retention time required to settle out sediment bound P and the large areas required in order to reduce the catchment P load by a significant amount (ORC 1995). The study reportedly stated that the largest site available for wetland re-establishment was 93ha of land which was deemed non advantageous due to its position in the upper catchment. A review of the methodology of the report, particularly in identifying P loss hotspots in the catchment, and therefore how much can be removed by targeting different subcatchment areas, would be useful. Caruso (2001) filled some of these knowledge gaps by measuring P loads at multiple points in different subareas of Mill Creek to determine P hotspots, even identifying these down to the individual property level. Interestingly the author found that the O'Connell Creek catchment was a hotspot of P release (Figure 2) and it was suggested that the results of the investigation could inform more targeted mitigation actions in the catchment (Caruso 2001).

Best Management Practices (BMPs)

Best management farming practices include actions such as managing land for erosion and leaching, managing to minimise losses of sediment and nutrient to waterways, and stock exclusion from waterways. Ensuring on-farm BMPs are employed in the catchment is an obvious requirement for successful lake restoration. Using the results from Caruso (2001), a strategy for targeting BMPs, particularly in subareas or even on properties which are contributing high P loads would be highly beneficial. BMPs for lifestyle blocks, golf courses and activities that disturb ground in the catchment should also be communicated to relevant land owners and developers.

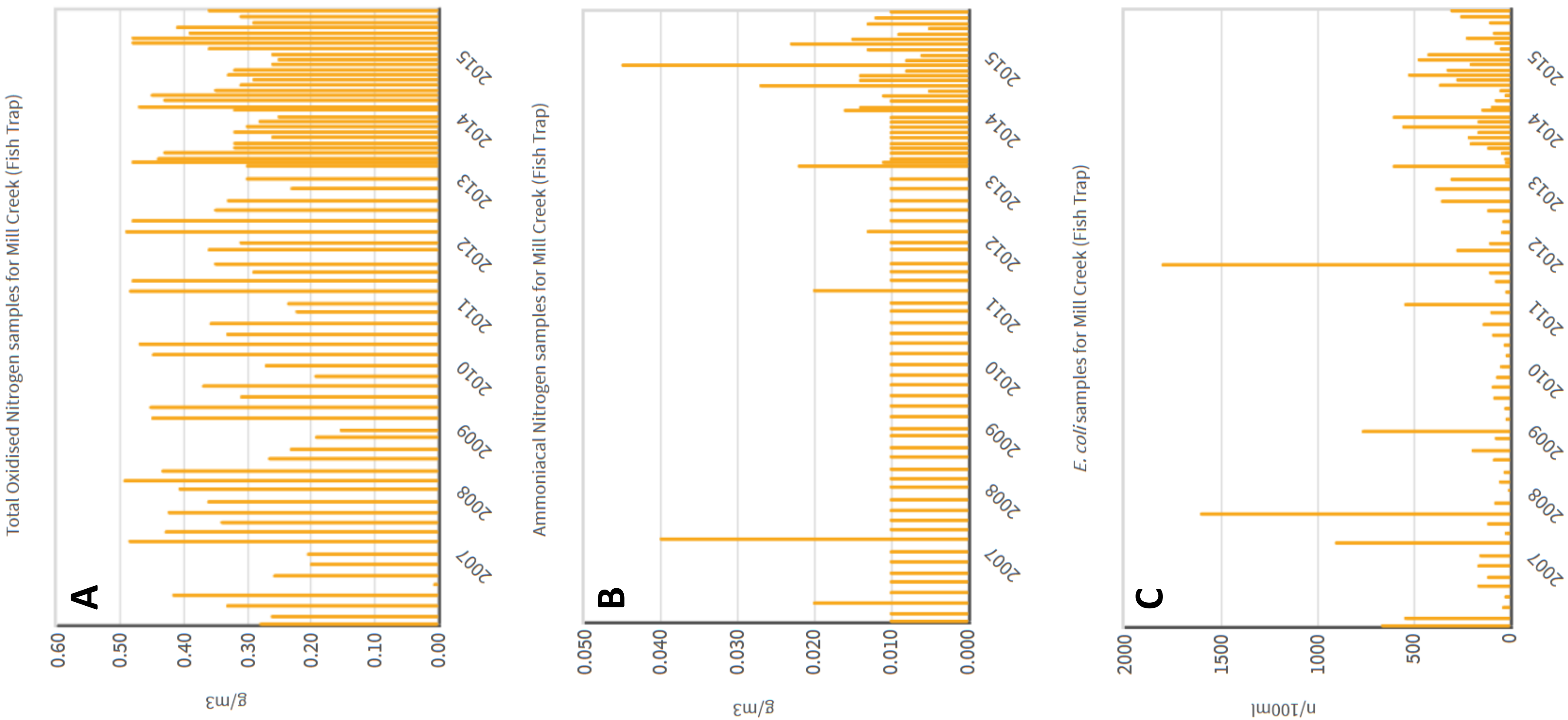

Figure 1. Nitrate (A), ammonium (B) and *E. coli* concentrations (C) in Mill Creek from 2006-present. Summer nitrate concentrations (the regular periods of lowest nitrate concentrations) appear to have been increasing over the sampling period (A). In addition, recent ammoniacal N (B) and *E. coli* concentrations (C) may also be increasing. From LAWA website (https://www.lawa.org.nz/explore-data/otago-region/river-quality/clutha-river/mill-creek-(fish-trap)/)

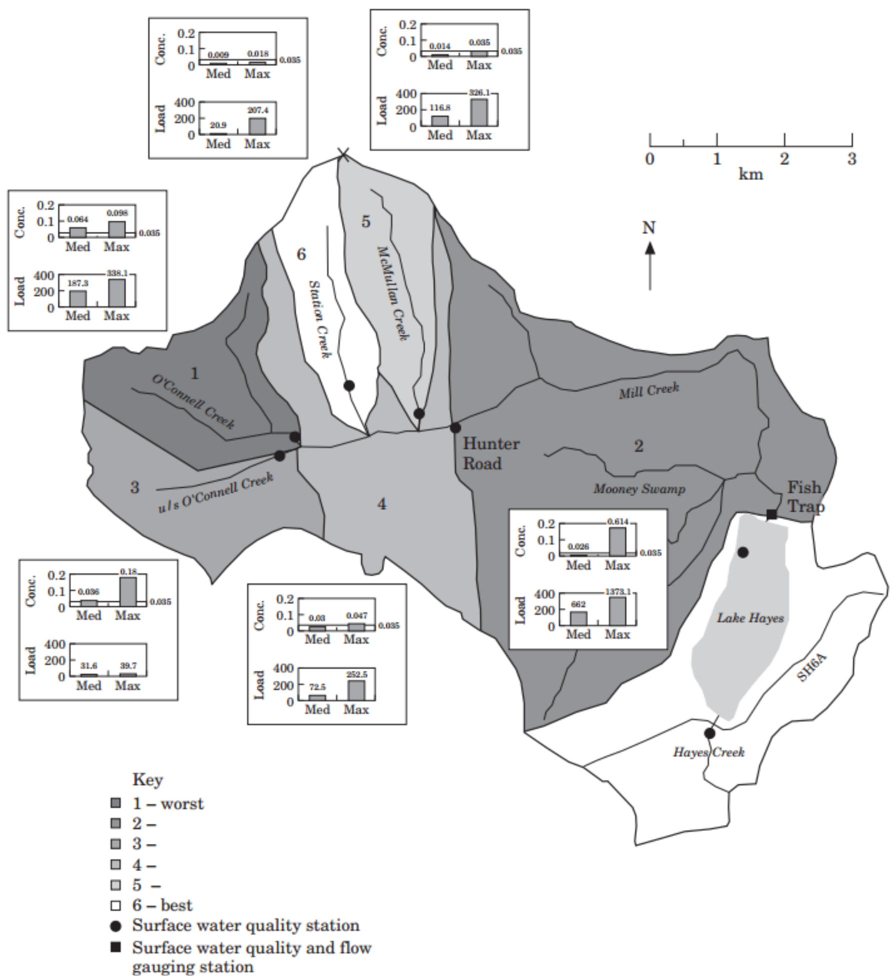

Figure 2. Results from Mill Creek catchment subarea phosphorus targeting in Caruso (2001), showing catchments ranked from highest to lowest P contribution. Taken from Caruso (2001).

A lake catchment management plan to ensure continuing reductions in nutrient and sediment losses from the catchment.

In order to ensure that landowners in the catchment minimise nutrient and sediment concentrations in streams and springs draining into Lake Hayes, a collaborative, community-driven lake and catchment management plan could be developed and implemented by undertaking the following:

i. Identify iwi, stakeholders, industries, scientists and other interested parties.
ii. Review the ORC (1995) Lake Hayes Management Strategy and its implementation.
iii. Undertake a catchment-wide N, P, sediment and *E. coli* survey based on the design of Caruso (2001) to identify current hotspots of contaminant contributions to the lake.
iv. Determine the feasibility of setting nutrient caps on the catchment.
v. Collaboratively develop a lake/catchment management plan with community participation at all stages.

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