Lake Hayes Water Quality Remediation Options

Prepared for Otago Regional Council

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Cover Photo: Reflections of the southern hills from Lake Hayes [Photo montage by Max Gibbs].

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Executive summary

To progress the development of a Long-Term Plan (LTP) for the management and restoration of Lake Hayes, Otago Regional Council (ORC) commissioned the National Institute of Water and Atmospheric Research Ltd (NIWA) to provide expert advice for detailed scoping on the options for remediation of Lake Hayes water quality. NIWA was asked to review a recent report, prepared for the Friend of Lake Hayes Society Inc (Schallenberg and Schallenberg 2017), which provides a restoration and monitoring plan for Lake Hayes and covers a range of options for improving the water quality of Lake Hayes. In addition, NIWA was asked to expand the detail on three of these options in order that they could be costed. Within the scope of the report, NIWA was requested to present information on other options that might be useful in the restoration of Lake Hayes and to include information and comments on a new nanobubble technique. Comments on catchment management were to be minimal as this is being covered in a separate report.

Report review

The Schallenberg and Schallenberg (2017) report is well written and provides a wealth of information on the history of the lake and the probable causes of the lake changing from a presumably pristine alpine lake to a highly degraded, supertrophic lake. A key message in the report is that the lake appears to be at a 'tipping point' for recovery. However, although the report was written in 2017, it contains few data beyond 2015, without presenting more recent evidence to support this contention. Neither is there an indication of what level of improvement might be expected. The report explains much of the variability in the data by conventional limnological processes and also includes a novel '*Ceratium* pump' theory to explain recent increases in total phosphorus (TP) and total nitrogen (TN) in the upper water column during the period of thermal stratification.

The sections on monitoring (at different time scales) and the need for a nutrient budget are comprehensive and the report reinforces the need for long-term datasets for trend analysis and understanding how the lake might be restored. The report presents a restoration strategy for Lake Hayes and includes a feasible restoration plan and timeline. Some of the actions in this restoration plan are explained in more detail in the appendices. Specifically: 1) the use of food web biomanipulation to reduce the magnitude of the summer bloom of *Ceratium hirundinella*; 2) the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water; 3) alum dosing for phosphorus immobilisation – together with a rough estimate of the cost of alum dosing, and 4) catchment management to restore and protect Lake Hayes.

While these actions are well presented, there is an error in the section on augmentation of Mill Creek with Arrow River irrigation water that needs to be corrected to give a better estimate of the likely efficacy of this option.

Three defined options for water quality improvement

To aid interpretation of some of the data and provide information on the hydrodynamics of Lake Hayes, additional NIWA and University of Otago unpublished data was included in this report. Additional information on the growth habit of *Ceratium hirundinella* was compiled from the literature and included in this report.

Defined option 1: *Flow augmentation from the Arrow River irrigation scheme*. Although this option was considered to of minor importance in the Schallenberg and Schallenberg (2017) report, when the entrainment factor of surface lake water into the density current from Mill Creek is included, there is the potential for this option to prevent the bottom water (hypolimnion) becoming anoxic during the stratified period, thereby eliminating the internal phosphorus (P) load which is required by *Ceratium* for growth. There are issues with this option in that it is 'fragile' and relies on the ambient temperature to cool the Mill Creek water sufficiently to cause the density current to plunge to the hypolimnion. In a warm year this might not happen and the internal P load could return along with a substantial algal bloom.

Defined option 2: *Destratification*. The use of a bubble plume air curtain across the middle of the lake would generate circulation currents that would prevent the lake from becoming thermally stratified and mix the lake water column, keeping the lake well oxygenated. These conditions would eliminate the internal P load and reduce the proliferation of algal blooms. The mixing regime would not eliminate algal growth in the lake, rather it would change the algal assemblage from harmful cyanobacteria and *Ceratium* species to more benign species. Eventually these would diminish dramatically and the lake water quality would improve rapidly.

Defined option 3: *Sediment capping*. The range of available sediment capping and P elimination agents has been discussed. It was concluded that the 'best' option would be to treat the lake with alum using a low dose drip feed into the Mill Creek inflow. This approach would deliver the alum to the hypolimnion in the density current without impact on the lake surface waters. Using real-time monitoring data, adaptive management strategies could control the application to when it was required and was being delivered into the hypolimnion for the most efficient and cost-effective management of the internal P load.

Other options 1: *Nanobubble technology*. This was determined to be un-proven technology and the information available was entirely based on company brochures with no peer-reviewed scientific papers to back up the claims made. While this situation frequently arises when new techniques are developed, it is common to put out a methods paper for the scientific community to review. This has yet to be done. The technique appears to work but has the limitation that it requires the nanobubbles to provide all of the oxygen to aerate the hypolimnion. In Lake Hayes, this represents about 1.6 t O_2 d^{-1}. Putting this into perspective, the underflow from Mill Creek with augmentation can provide up to 1.8 t O_2 d^{-1}. The other main drawback for nanobubble technology is the cost at about $4.7 million for the 7 units required for Lake Hayes.

Other options 2: *Hypolimnetic withdrawal*. A switchable outflow control structure is described that allows the current outflow regime to operate from autumn to spring and then change to draw hypolimnetic water in summer. During summer the hypolimnetic siphon would reduce the DRP in the lake and discharge it into the Kawarau River. This approach has been used successfully overseas and on one lake in New Zealand, and requires further investigation by an engineer to assess feasibility for Lake Hayes, if it is considered as a practical mitigation measure.

Other options 3: *Biomanipulation*. Biomanipulation is a technique which relies on a set of conditions prevailing to enable a specified end result. This approach is covered in detail in the Schallenberg and Schallenberg (2017) report. However, while biomanipulation has been successfully used in many shallow lakes in Europe and the USA as well as in New Zealand, the success stories have all been in shallow lakes and ponds and there is no compelling literature base that suggests the technique would work in a lake as deep as Lake Hayes.

Other options 4: *Catchment management*. This strategy is assessed as being fundamental to the long-term restoration of Lake Hayes. A key element of catchment management would be the reduction of fine sediment loads to Lake Hayes, as this is the primary vector for transporting P into the lake. Catchment management will take time to become apparent but, in the long-term, it will improve water quality in Lake Hayes.

It is the considered opinion of the author of this report that the most effective short-term mitigation method in Lake Hayes would be destratification.

1 Introduction

Otago Regional Council (ORC) are developing a Long-Term Plan (LTP) for the management and restoration of Lake Hayes. To help facilitate this, ORC commissioned the National Institute of Water and Atmospheric Research Ltd (NIWA) to provide expert advice for detailed scoping on the options for remediation of Lake Hayes water quality. A report prepared for the Friend of Lake Hayes Society Inc (Schallenberg and Schallenberg 2017) provides a restoration and monitoring plan for Lake Hayes that covers a range of options for improving the water quality of Lake Hayes that require further consideration.

The specific tasks requested and covered in this report are:

1. Review and comment on the Schallenberg and Schallenberg (2017) report advising on mitigation options for improving Lake Hayes water quality.
2. Look at, but not be limited to, three defined options for water quality improvement, being:
   a. Flow augmentation from the Arrow River irrigation scheme - The best strategic timing of water releases, the potential efficacy based on flow rates and volumes available. Seasonality of releases.
   b. Destratification – Provide advice on what the challenges are; if it is possible given the shape and stratification dynamics of the lake; what the best options would be; Provide sufficient advice to allow for design and costings to be carried out. Required duration of de-stratification?
   c. Sediment capping – what the best medium is to use, amounts required and how best to apply it. How long will sediment capping be effective for?
3. Comment on the likely effect of each option on water quality in Lake Hayes, including algal growth.
4. Assess the appropriateness and infrastructure/cost (if available) requirements to employ nanobubble technology to aerate the Lake Hayes hypolimnion.
5. Provide comment on the importance of long-term catchment management given current nutrient and sediment loads entering Lake Hayes from Mill Creek.
6. Provide sufficient detail to quantify what is required to install and manage each of the options so that detailed cost estimates can be made to support our Long-Term Plan (LTP) development. How much, where, when etc., and if required, discuss and advise service providers on technical details of the options. Describe the secondary or amenity effects of each option (noise, water clarity, odour, visual impact, etc.).

To facilitate this work, ORC will provide:

- All available relevant data on water quality and stratification dynamics on Lake Hayes.
- Technical details and constraints in relation to the flow augmentation option, including maximum flow rates, seasonality of water availability and augmentation water quality.
- Comments on the completed draft within 5 working days of receipt of the draft report.

The output from this work will be a comprehensive report, peer reviewed to NIWA standards, providing expert advice for detailed scoping on the options for remediation of Lake Hayes water quality.

1.1 Background information

This section presents the current state of knowledge about Lake Hayes in terms of its biogeochemistry in order to understand the factors that are driving the lake water quality and whether/how the water quality is changing over the period of the available monitoring data. This section also includes aspects of the physics, hydrodynamics and biology in order to adequately review the Schallenberg and Schallenberg (2017) report. It may also help understand how this lake works and to identify any critical points in the various biogeochemical cycles where an intervention at a specific time may have a major effect on the water quality.

The water quality of Lake Hayes has been classified as supertrophic with a Trophic Level Index (TLI) >5. This classification is addressed in the Schallenberg and Schallenberg (2017) report. Conversely, the National Policy Statement on Freshwater (Freshwater NPS), National Objective Framework (NOF) attributes for phytoplankton abundance (measured as chlorophyll *a*), total nitrogen (TN) and total phosphorus (TP) (MfE 2014), which are the legislated management requirements, is not addressed and are outside the scope of this report and are not discussed.

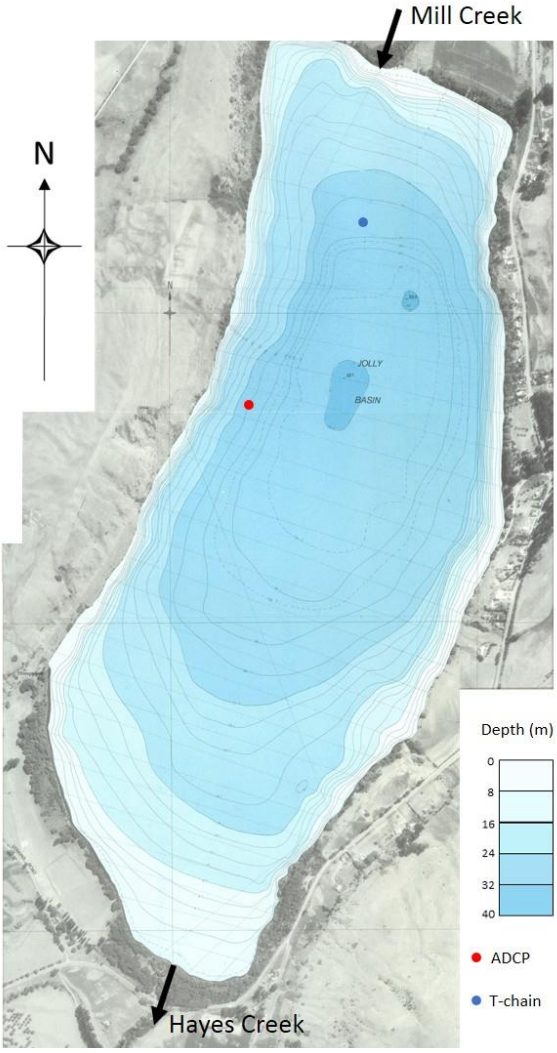
Lake Hayes is a small glacial lake formed about 10,000 y BP (Lowe and Green 1987). Prior to 1740, the landscape was likely to have been Kahikatea forest with large wetlands in the western Mill Creek area. Deforestation of the catchment began around 1740 when the Kahikatea forest was largely destroyed by fire and probably continued through the 1800's as miners and settlers harvested trees for shelter and firewood (Robertson 1988). The catchment became native tussock grassland. At present, it is largely agricultural land, which was developed by drainage of wetlands and the application of superphosphate to grow grass for the dairy industry (Schallenberg and Schallenberg 2017).

Accelerated eutrophication of the lake began when superphosphate was introduced in 1950, spread by aerial top-dressing across the Mill Creek catchment. A top-dressing plane crashed into Lake Hayes in August 1952. A cheese factory north of the lake discharged effluent whey, with a phosphorus (P) load of about 1000 kg y^{-1}, into Mill Creek from 1912 to 1955 (Robertson 1988). Septic tank effluent from lake shore residences enter the lake through the groundwater inflow (Selvarajah 2015).

Investigation of possible nutrient limitation in Lake Hayes in 2006 (Bayer et al. 2008), when *Ceratium* was the dominant phytoplankton species, indicated that algal growth was stimulated by additions of N and the trace element boron and zinc. P additions had no effect. Bayer et al. also comment that P was often in surplus in the lake in relation to nutrient demands of phytoplankton i.e., low N:P ratios[1]. Schallenberg and Schallenberg (2017) comment that reducing P levels in the lake to the point where they can restrict phytoplankton blooms is important. They also comment that "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."

[1] N:P ratios <10 are considered to indicate N-limitation and alga growth is likely to be stimulated by the addition of N. N:P ratios >17 are considered to indicate P-limitation and algal growth is likely to be stimulated by the addition of P. Between these values, addition of either N or P, or both N and P, is likely to stimulate algal growth.

Fine sediment, washed into Mill Creek from the catchment (Figure 1-1), is accumulating in the bottom of Lake Hayes. This sediment typically has high P concentrations bound to the iron and manganese oxides in the soil particles. When the lake water is well oxygenated, the P remains attached to the metal oxides and is in a particulate form, which is not available for plant growth i.e., it cannot be taken up by phytoplankton (free floating algae). Conversely, when the lake has low dissolved oxygen (DO) concentrations, the iron and manganese dissolve and the P is released into the water column as phosphate, commonly called dissolved reactive P (DRP). This transformation is reversible with DRP being bound to the iron and manganese oxides when the lake becomes oxygenated again (Schallenberg and Schallenberg 2017). DRP is readily available to sustain phytoplankton growth. When DRP is released from the sediment stored in the bottom of the lake it is referred to as internal cycling.



Lake Hayes morphological characteristics:

Area	2.76 km^2
Length	3.1 km
Mean depth	18.0 m
Maximum depth	33.0 m
Volume	55.1 x 10^6 m^3
Volume of hypolimnion	28.9 x 10^6 m^3
Catchment land area	44.0 km^2
Surface elevation	315 m
Theoretical mean residence time[2]	3.82 y
Residence time (added inflow)[3]	2.98 y.
Inflow via Mill Creek (mean 1984-97)	0.43 m^3 s^{-1}

Figure 1-1: Lake Hayes bathymetry. Red dot indicates position of an acoustic doppler current meter (ADCP) deployment. Blue dot is position of thermistor chain deployment. (Chart redrawn from Hurley 1981).

[2] Lake volume divided by total annual inflow (data from Caruso 2000a, b)
[3] See section 3.2 calculations

Apart from monitoring data from Otago Regional Council and University of Otago, there have been several short-term investigations by other institutes, including NIWA, that provide additional information on in-lake processes in Lake Hayes. The data may aide interpretation of the monitoring data.

For example, when the lake is thermally stratified in summer, wind blowing across the lake establishes a circulation pattern in the lake with surface water moving with the wind flow but the upper and lower water columns moving in the opposite direction (Figure 1-2). Water movement around the thermocline was in the same direction as the wind with evidence of turbulence (Figure 1-2, C). These data were recorded at the ADCP site (Figure 1-1) on a bottom-mounted ADCP in 24 m water depth recording in 1-minute bursts every 5 minutes with 1-m depth intervals.

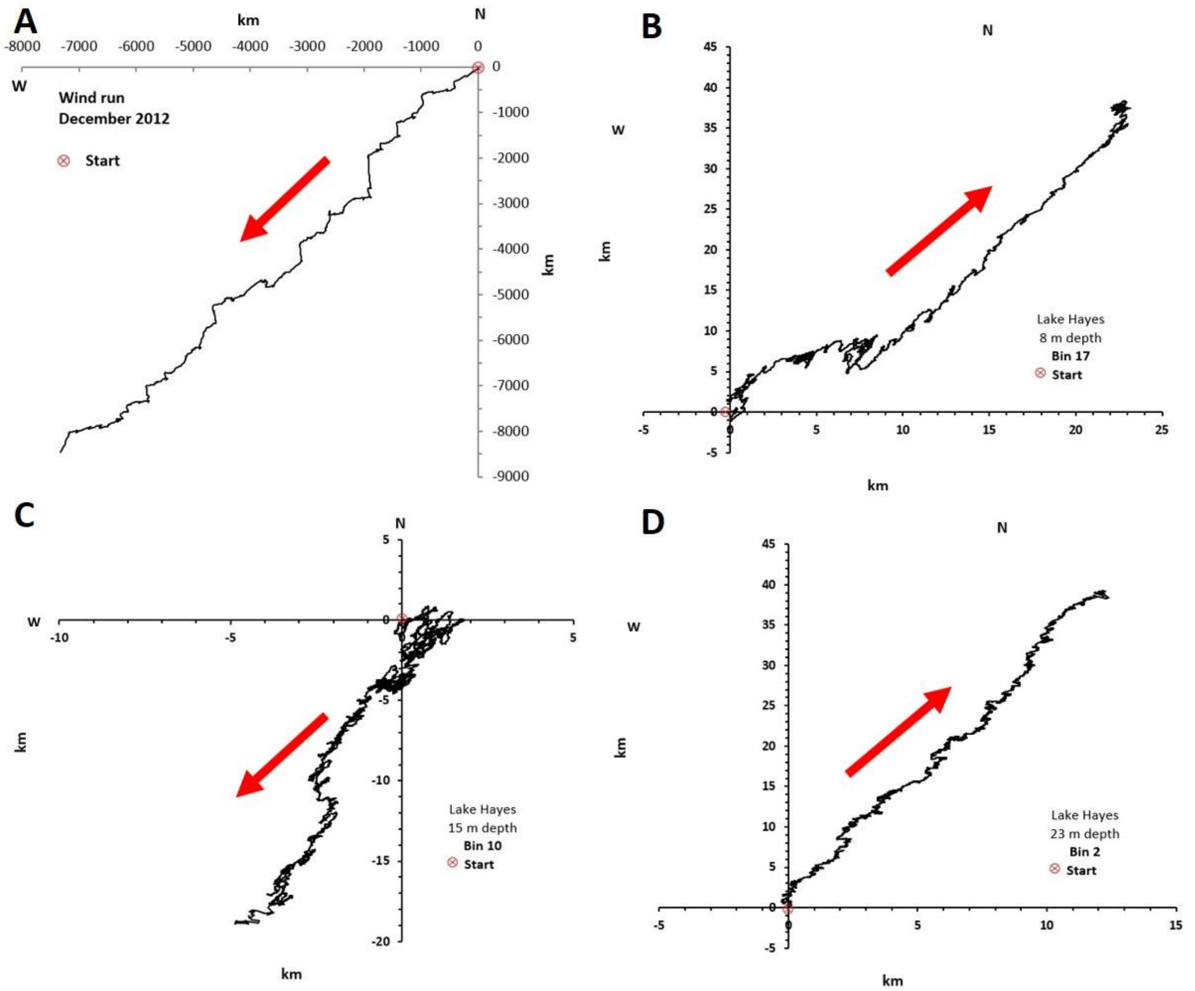


Figure 1-2: Wind run and acoustic doppler current meter (ADCP) progressive vector plots at selected depths. Flow directions indicated by red arrows: A) Persistent wind run from the north; B) Epilimnion current flow at 8 m depth was to the north; C) Thermocline current flow at 15 m depth was to the south; D) Hypolimnion current flow at 23 m depth was to the north. (NIWA unpublished data).

The full suite of ADCP profile data show the currents reverse at different depth ranges (Figure 1-3). This limited flow data indicates that persistent wind flow across the surface of the lake in one direction can induce a subsurface return flow which has resulted in a pair of vertical circulation cells, one in the epilimnion and the other in the hypolimnion. The southwards return flow of the epilimnion cell coincides with the thermocline, which is a zone of turbulence and can result in upwards mixing of nutrient enriched bottom water into the epilimnion and downwards mixing of DO from the epilimnion into the hypolimnion.

A similar current feature has been recorded in Lake Rotorua (Gibbs et al. 2016). There is insufficient ADCP data from Lake Hayes to define the horizontal currents as found in the Gibbs et al. (2016) study. These are highly likely and would account for spatial distribution of a *Ceratium hirundinella* bloom reported in Schallenberg and Schallenberg (2017).

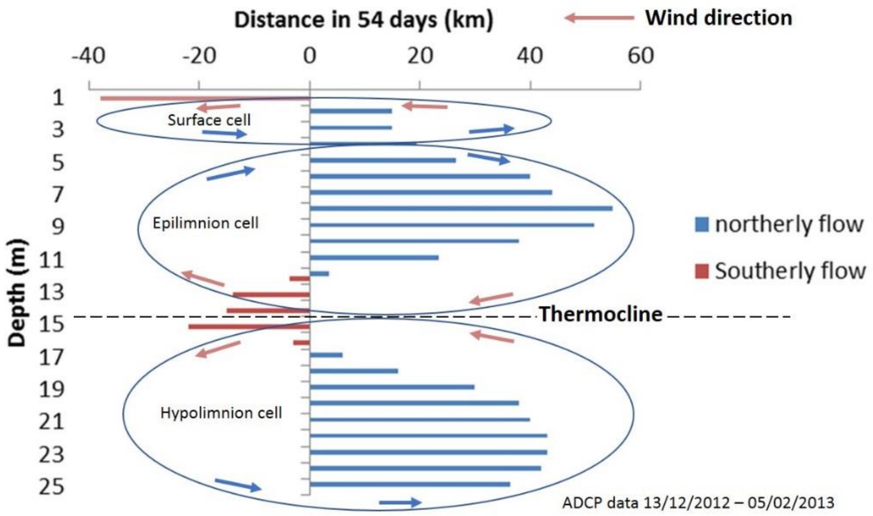


Figure 1-3: Current flow direction in Lake Hayes in response to wind forcing from the north. Stylised circulation cells are consistent with data shown. There is insufficient data available to show lateral circulation patterns. Current data recorded at the ADCP location (Figure 1-1) between 13/12/2012 and 05/02/2013. (NIWA unpublished data).

Temperature and dissolved oxygen data, from a thermistor chain moored in 25 m water depth (Figure 1-1)and recording at 15-minute intervals, provide insights into possible mixing processes in Lake Hayes (Figure 1-4).

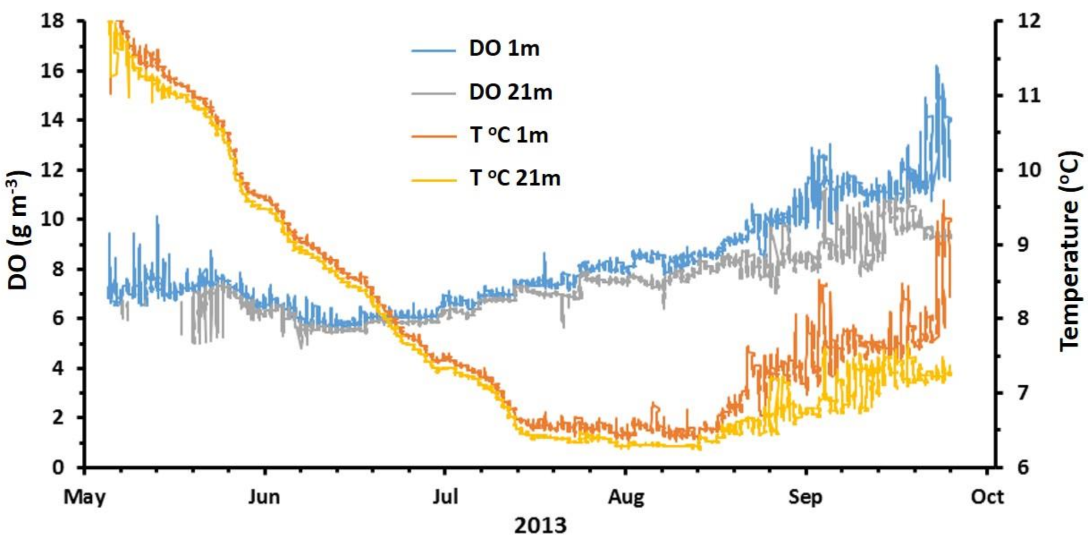


Figure 1-4: Surface and bottom water temperature and dissolved oxygen data from Lake Hayes. Data from a thermistor chain deployed in 25m water depth (Figure 1-1) between May and September 2013 showing the beginning of thermal stratification. (NIWA unpublished data).

These data, recorded during the winter mixed period, show the expected diurnal temperature cycle of heating and cooling in the surface waters as well as diurnal changes in the DO concentrations apparently associated with phytoplankton photosynthesis during the daylight hours in spring (Figure 1-4). These diurnal cycles can be detected down to 8 m depth, synchronous with the surface temperature and DO. The unexpected diurnal cycles in the 21 m deep data (Figure 1-4) are lagging about 3 hours behind the daylight cycles (Figure 1-5) and can only have been caused by temperature and DO changes in the Mill Creek inflow water, which would have been plunging as a density current at that time and intruding through the lake at a depth of equal density or along the lake bed. The lag will be due to the travel time from the Mill Creek mouth to the thermistor chain, a distance of about 500 m and represent an average flow velocity of about 5 cm s^{-1} for the intrusion current.

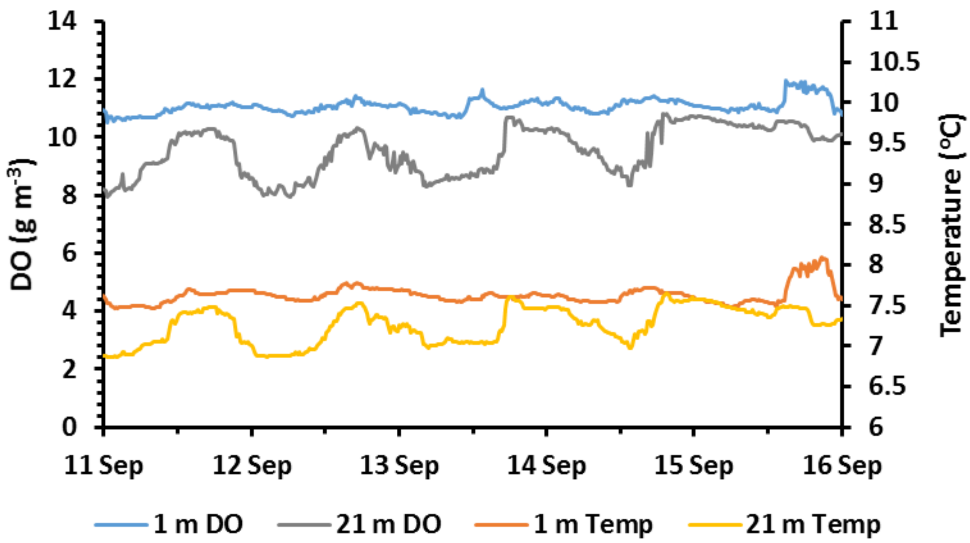


Figure 1-5: Surface and 21 m depth temperature and dissolved oxygen from a thermistor chain in Lake Hayes in winter 2013. While the surface lake water data show little variation over the diurnal cycle, the temperature and DO in the 21-m data show substantial variation over the diurnal cycle, which appears lag the midday high by about 3 hours. This is evidence that the Mill Creek inflow forms a density current. (NIWA unpublished data).

The mean annual flow in Mill Creek is estimated to be 0.43 $m^3 s^{-1}$ (Caruso 2000a, b). If this inflow is >0.5°C cooler than the lake surface water temperature, the inflowing water will plunge as a density current and sink to a depth of equal density before penetrating into the lake as an intrusion layer (Gibbs and Hickey 2012). In the plunge process the inflow water entrains surface lake water into the density current and this can increase the volume of the intrusion layer by about 5-fold, depending on the temperature difference between the inflow and the lake. It also provides a mechanism by which nutrients and phytoplankton cells in the lake surface water can be rapidly dispersed throughout the lake (Vincent et al. 1991).

This dispersal process is potentially very important during storm events when Mill Creek can transport relatively large amounts of fine sediment from the catchment into the lake. Fine sediment typically has high concentrations of P bound to the iron and manganese particles in the sediment and is the main vector for P transport into lakes. This is recognised in the Schallenberg and Schallenberg (2017) report as a major source of the legacy[4] P accumulating in the lake sediments.

An example of a flood event hydrograph relative to the total P (TP) transported in Mill Creek (Caruso 2000b) suggests that the majority of the TP may enter the lake in just a few hours during a storm event (Figure 1-6). This is consistent with a study on the Ngongotaha Stream at Rotorua, which found that 42% of the annual sediment load was carried on just three days and that 25% of the annual load was carried in a period of 16 hours at the peak of the flood event (Hoare 1982).

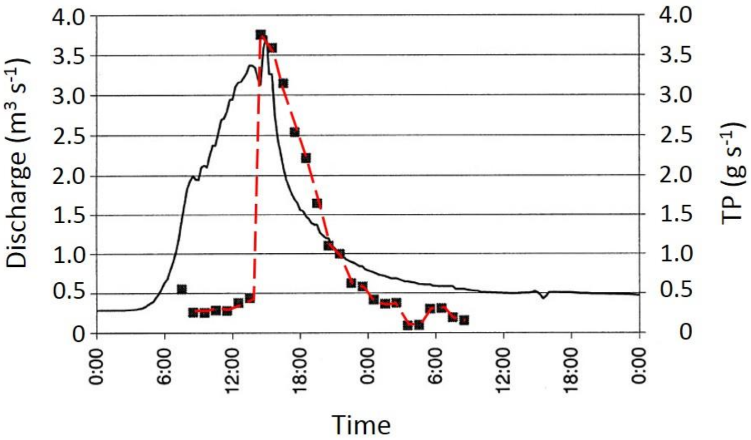


Figure 1-6: Correlation between a flood hydrograph and Total P (TP) load in Mill Creek. Red dashed line indicates the TP load. (Redrawn from Caruso 2000b).

These additional data and information augment the data and information presented in the Schallenberg and Schallenberg (2017) report.

[4] Legacy P: P transported into a lake in suspended sediment settles to the bottom and accumulates rather than being flushed out. Accumulation occurs over many years and remains as a legacy from previous historical activities in the catchment.

2 Review of the Schallenberg and Schallenberg (2017) report

2.1 Overview

The Schallenberg and Schallenberg (2017) report provides a comprehensive general review and analysis of the information on Lake Hayes, and includes recommendations on restoration options for this lake. It brings together the history of Lake Hayes and the changes and events that have transformed a pristine, and presumably oligotrophic, alpine lake into a highly degraded, supertrophic lake, which has, in recent times, experience substantial algal blooms. The report summarises the physical, chemical and biological aspects of the Lake Hayes ecosystem and gives a good background into some of the factors that have resulted in the accelerated eutrophication of this lake. It also offers some interesting ideas, such as the "*Ceratium* pump theory", to explain some of the temporal changes in in-lake nutrient concentrations. The recommendations are aligned with the evidence available on how the lake is operating and contemporary understanding of lake restoration science.

However, although the report was written in 2017, there is little contemporary data beyond 2015 and the appendices are essentially reproductions of earlier work without major updates. There is a serious flaw in the discussion of the use of irrigation water from the Arrow River to augment the flow in Mill Creek for enhanced flushing as a restoration strategy. Parts of that section of the report should be rewritten to provide correct information.

Most of the terminology is explained but not coherently in one place. For example, an important concept is the trophic level index (TLI). This is introduced on Page 5 without an explanation of what it is and how it is determined, and no reference as to where that information can be obtained. The trophic level classification table from Burns et al. (2000), which contains the TLI scale and parameter ranges appears in an appendix on Page 45, should be cross-referenced from the text.

2.2 Detailed comments

Since the TLI was developed in the late 1990's (Burns et al. 2000), the TLI assessment of water quality has been widely used in New Zealand to compare the water quality of all lakes. Although intended as an index for comparison of water quality between different lakes, the TLI has also been used as a tool for management of lake water quality. How this is achieved using the linkage between the four TLI components – water clarity, chlorophyll *a*, total nitrogen (TN) and total phosphorus (TP) – is not mentioned but should be to provide improved clarity.

The report identifies that Lake Hayes has poor water quality and is classified as supertrophic based on the TLI of >5, but suggests that the lake may be approaching a tipping point of recovery. This conclusion is based on changes in water quality and biological indicators. For example, the report authors suggest that there has been an apparent decrease in TP concentrations in the hypolimnion since about 2000 indicating a reduction in internal P cycling in the lake, and the report claims there have been several recent years with very low phytoplankton biomass and high water clarity in summer: 2009/10, 2012/13 and 2016/17. A clear water phase also occurred in 2017/18. Notwithstanding this, the ORC time-series chlorophyll profile data do not support that 2012/13 was a clear water year. The ORC data show that, in 2012/13, mean maximum chlorophyll concentrations averaged 72.2 mg m^{-3} and the peak concentration was 137 mg m^{-3}. In 2009/10, the mean maximum chlorophyll concentrations averaged 6.1 mg m^{-3} and the peak concentration only reached 13.5 mg m^{-3}.

There are other indicators of water quality improvement and lake recovery which have not been discussed. These are found in the more recent time-series data that were previously used to show the progressive water quality degradation over time.

Evidence of lake water quality degradation is shown in the report are a progressive increase in TLI from about 3.6 (mesotrophic) in 2004 to about 5.2 (supertrophic) in 2015 and a reducing mean water clarity, measured as Secchi disk depth, from about 6 m in 1950-53 to <2.5 m in 2015. The loss of bottom water (hypolimnion) dissolved oxygen is mentioned but not analysed. A consequence of hypolimnetic anoxia is the release and recycling of P, which has been brought into the lake with the elevated sediment loads during storm events (Figure 1-6). This is discussed in the report.

Assessment of oxygen loss from the hypolimnion can be standardised and expressed as a hypolimnetic oxygen depletion (HOD) rate for interannually comparison. This has not been done in the Schallenberg and Schallenberg (2017) report. As part of this review a simple evaluation of the changes in HOD was carried out using available data from the summer stratified period (Nov-April inclusive). This information indicates a long-term degradation of lake water quality, which was already degraded in the 1950s due to land drainage, farming and the discharge of effluent whey from a cheese factory into the lake (Schallenberg and Schallenberg 2017). Literature data show a progressive loss of bottom water oxygen which was at a minimum of 13% saturation in 1953-54 (Jolly 1968) to complete anoxia (0% saturation) for a period of about 4 months each year in 1969-71 (Burns and Mitchell 1974). This period of anoxia gradually increased to about 5 months in 2008 (ORC data). The rate of change in the HOD rate, calculated from September to February each summer, has been relatively slow with a mean HOD rate of 97.5 mg m^{-3} d^{-1} from 1992 to 2008 (minimum 81 mg m^{-3} d^{-1}, maximum of 115 mg m^{-3} d^{-1}) (Figure 2-1).

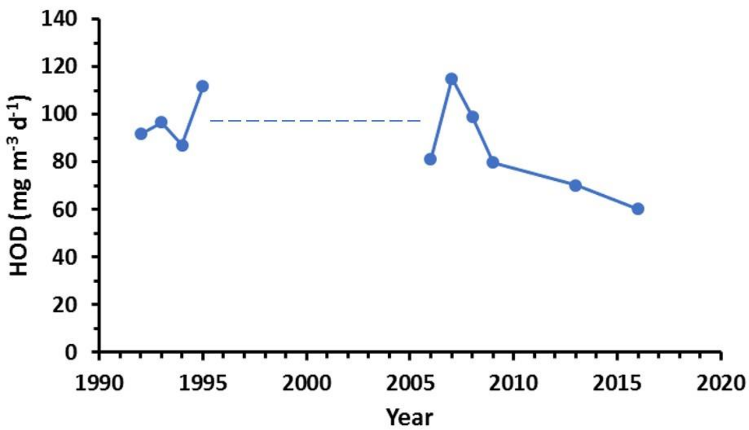


Figure 2-1: Time-series hypolimnetic oxygen depletion (HOD) rates for Lake Hayes. Mean HOD rate from 1992 to 2008 is 97.5 mg m^{-3} d^{-1}. From 2008 to 2016 the HOD rate appears to have been decreasing indicating a reduction in sediment oxygen demand, implying an improvement in water quality (Otago Regional Council data).

Recent data from two of these measures of degradation also support the suggestion in the report that Lake Hayes might be at or approaching a recovery tipping point. Firstly, the HOD rate has decreased from 99 mg m^{-3} d^{-1} in 2008 to 60 mg m^{-3} d^{-1} in 2016 (Figure 2-1). This change can be seen in the time-series DO profile data (Figure 2-2) with the slower reduction in DO concentration in spring 2016 and anoxia reached in March 2017 instead of January as in the 2007-8 plot.

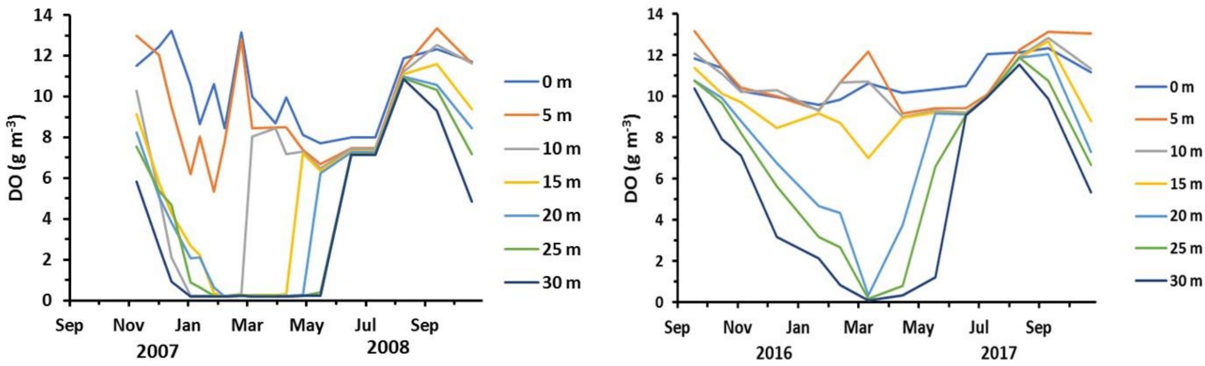


Figure 2-2: Time-series dissolved oxygen profile data from 2007-8 and 2016-17. In 2007-8 the period of bottom water anoxia was 5.5 months but in 2016-17 it was about one month.

Secondly, in 2007-8, the period of bottom water anoxia was 5.5 months but in 2016-17 it was only about one month. The major differences between these two time-series plots are that 1) in 2007-8 the anoxic zone extended up to 8 m below the surface while in 2016-17 the anoxic zone was below 20 m, and 2) re-oxygenation of the bottom water began in March 2017 while it began in May in 2008. These recent DO concentrations indicate that the period when P is recycled from the sediments is greatly reduced, consistent with the author's suggestion of a reduction in internal P cycling in the lake. The DO time-series plots suggest this change has been more recent i.e., since 2008 and not 2000.

The correlation between the three clear water years with low *Ceratium hirundinella* biomass and the super abundance of the water flea, *Daphnia pulex*, is important as it indicates that there is a food web mechanism that might be used to reduce or manage the recent *Ceratium* blooms. Two possible scenarios are that either the large *Daphnia* are grazing on the *Ceratium* cells directly or they are grazing on the bacteria used by *Ceratium* as a food source to augment their nutrient requirements.

Coupled with the other indicators of water quality improvement (Schallenberg and Schallenberg 2017 report section 2.3), the report correctly interprets the marked fluctuations in water clarity as indications that the lake is close to a tipping point for recovery from eutrophication but to what degree is not specified. An expectation of the degree of recovery should be given. The final key point raised in the report (section 2.3) summaries the current state of knowledge as well as a way forward:

"*The current situation suggests that appropriate restoration measures could result in stable 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.*"

2.3 Restoration strategies

Restoration strategies suggested in section 4 of the Schallenberg and Schallenberg (2017) report fall into five main categories: 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 (listed in the report, Table 1). The discussion of these options in this section of the report provides a comprehensive basis for the development of a restoration plan for Lake Hayes.

One factual issue with this section of the report is the statement that *"Lake Hayes has a water residence time of around 1.8 years (Caruso 2000)"*, is incorrect. The value of 1.8 years in the Caruso (2000b) paper is attributed to a paper by Jolly (1968) but that residence time value is not apparent in the Jolly (1968) paper. As part of this review the residence time[5] for Lake Hayes was calculated to be 3.8 years, based on the volume of the lake and the average total annual inflow volume from 1984 to 1997 via Mill Creek, as published in the Caruso (2000b) paper. This is more in keeping with expectations for a mean annual inflow of 0.43 $m^3 s^{-1}$.

A second issue with this section of the report is that, while the formation of a temperature driven density current from Mill Creek inflow is recognised, the calculations used do not take into account the entrainment[6] of surface lake water into that inflow when considering oxygen and nutrient transport.

The entrainment effect can be understood by looking at the situation in other lakes. For example, the Tongariro River inflow to Lake Taupo, entrains surface water into the plunging inflow and has increased the volume of the inflow by a factor of up to 4. The Ohau Channel inflow to Lake Rotoiti was measured at 3.5 (Vincent et al. 1991) before the diversion wall was installed. A more detailed evaluation of the oxygen transport into Lake Rotoiti via underflowing density current used an entrainment factor of 5 (Gibbs 1986). The amount of water entrained into the inflow is a function of the Froude number, the temperature difference between the inflow water and the surface water where the inflow enters the lake, the velocity of the inflow and the depth and width of the inflow channel (Spigel et al. 2005). As the temperature difference and/or the velocity increases, the entrainment factor decreases to around 2, but as the temperature difference decreases the entrainment factor increases to the point where a density current is not formed and the inflow water disperses across the lake surface as a buoyant plume (Gibbs and Hickey 2012).

In the heat of summer, the effect of entraining surface water is to raise the temperature of the resultant density current so that it is less likely to plunge below the thermocline and is more likely to insert as an intrusion layer in the epilimnion or around the depth of the thermocline in summer.

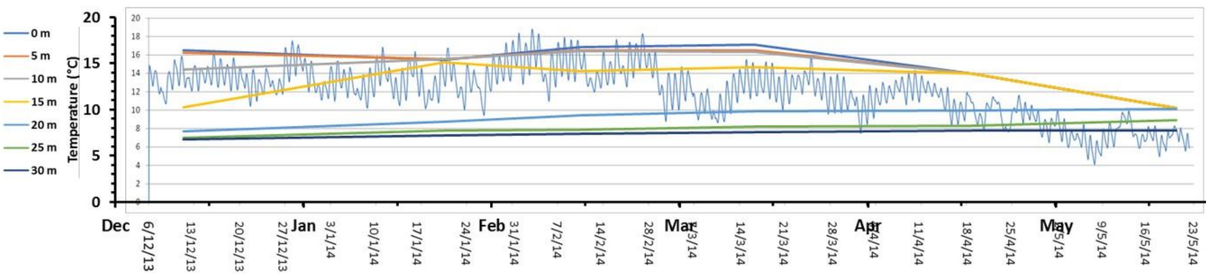


Figure 2-3: Example of time-series temperature data from Mill Creek (blue wavy line) overlaid with 2016/17 Lake Hayes temperature profile data (colour depth scale on left axis). Mill Creek data (from 2012 to 2013) show a strong diurnal cycle that was mostly warmer than the thermocline at 15 m depth in spring and summer before cooling to metalimnetic temperatures in autumn (March). Entrainment of warmer surface lake water into the Mill Creek inflow would result in the inflow water inserting at a depth higher in the water column than predicted by the absolute temperatures shown. Temperature data from ORC.

[5] Residence time is calculated as lake volume divided by the total inflow per year.
[6] An inflow that plunges as it enters a lake draws surface lake water into the inflow and forms a temperature-induced density current with a temperature somewhere between the inflow temperature and the temperature of the lake surface water. The density current will plunge to a depth of neutral density and then flow as an intrusion layer into the lake at that depth. The amount of water entrained is the entrainment factor. An entrainment factor of 4 means the density current has a volume 4 times larger than the inflow volume.

In Lake Hayes, this could result in the transport of fine sediment and nutrients from the catchment via Mill Creek and particulate matter, including phytoplankton, from the littoral zone of the lake, into the epilimnion or metalimnion (thermocline) in summer. As climate warming progresses, Mill Creek may become warmer in summer with the diurnal temperature cycle favouring more inflow into the epilimnion than the hypolimnion (Figure 2-3). The result of this process is likely to be seen as an increase in TP and TN in the epilimnion and a decrease in these nutrient components in the hypolimnion in summer. This effect could explain the recent increase in TP and TN concentrations in the epilimnion, as shown in Figure 6 of the Schallenberg and Schallenberg (2017) report, and which are the basis for the *Ceratium* nutrient pump theory.

In contrast with hot summers, as the lake cools in autumn and develops cold edge water at night because of frosts, the low night time temperatures in the Mill Creek water will be cooled further by entraining the cold edge water, causing the density current to plunge as an underflow to a greater depth than predicted by the absolute temperatures shown (Figure 2-3). The cold underflow is likely to flow along the lake bed and, as the water in that flow would be 100% saturated with oxygen, it would begin to re-oxygenate the hypolimnion. This is consistent with re-oxygenation of the hypolimnion from March onward, as observed in the 2016-17 DO time-series data (Figure 2-2). This hydrodynamic regime may account for the gradual reduction in the HOD rate since 2008 (Figure 2-1).

With this hydrodynamic regime distributing inflow water and associated nutrients to continuously differing depths in the lake over the diurnal temperature cycle, the restoration strategy of reducing the external loads to the lake will be fundamental to the long-term restoration of the lake.

Because catchment rehabilitation actions are typically slow to show improvement in lake water quality, the implementation of additional restoration strategies using food web manipulations, augmented hydraulic flushing and other in-lake actions, such as P sequestration, are likely to accelerate the restoration of the lake. Discussion of these additional restoration strategies is well considered and forms a coherent overall strategy for the restoration of Lake Hayes. The report includes tables showing the proposed restoration plan for the lake, a suggested time table and the goals/targets that should achieve a successful restoration of Lake Hayes.

2.4 Lake water quality and health monitoring

Although Lake Hayes is one of the most thoroughly monitored and studied lakes in New Zealand, there are gaps in the cumulative data that can lead to uncertainty in identifying trends. This section of the report (section 5) is excellent and reinforces the need for a monitoring program that has been designed for the lake, and includes monitoring factors beyond simple water quality variables. To this end the report includes a well thought out monitoring program for the lake and has co-ordinated this into a table that prioritises the type of monitoring, its frequency and the technology required.

2.5 Appendices

Additional details to the brief discussion of the four main restoration proposals in the body of the report are contained in a set of four appendices – 1) Food web biomanipulation, 2) Augmentation of inflows, 3) In-lake actions (alum dosing) and 4) Catchment management.

Appendix 1: **Food web biomanipulation as a restoration tool for Lake Hayes** is well set out and the practical considerations for this option are discussed. Because it is not practical to breed enough *Daphnia* to stock the lake or to remove the juvenile perch, which are zooplankters (zooplankton grazers), by netting, the key to the success of this option may be the ability to stock the lake with piscivorous trout and eels or large piscivorous sterile perch to consume the juvenile perch. There are other possibilities discussed.

Overall, the biomanipulation option as a concept appears feasible. Inclusion of examples where this approach has succeeded would be useful.

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 involves some complicated calculations around the addition of Arrow River water to Mill Creek in order to increase the flushing rate and reduce the residence time. The Arrow River water has lower DRP concentrations than Mill Creek and would not add substantial amounts of P to the lake. This section sets out the relationship between dissolved oxygen and the release of P from the legacy in the sediments and revisits an earlier report on this option (Schallenberg 2015). It sets out to answer four questions that could affect lake water quality:

1. *Could the augmented inflow flush displace substantial amounts of water from the lake?* Although the incorrect residence time of 1.8 y instead of 3.8 y is given, this does not affect the calculation of the additional proportion of water displaced from the lake. At just 7% increase it would take a long time (probably 10s of years) for the lake water quality to improve to eutrophic or mesotrophic classification.

2. *Would the augmentation flow displace bottom water?* The calculations for this question do not include the effects of entrainment of surface lake water into the density current generated by the Mill Creek water inflow. Consequently, the conclusions derived from the analyses done are incomplete and do not consider the changes in the depth of intrusion for the density current over the diurnal temperature cycle and over the summer – autumn temperature cycle.

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?* Again, the calculations for this question do not include the effects of entrainment of surface lake water into the density current generated by the Mill Creek water inflow. As the entrainment factor can substantially increase the volume of the density current, the mass of oxygen transported will be increased by a similar ratio over the amount estimated for the Mill Creek inflow by itself. While the question asks about the augmented flow, the answer given focuses only on the augmentation water rather than the combined flow. While the conclusion that "injecting the Arrow River augmentation flow directly into the bottom waters would not overcome deoxygenation in this lake" is correct, the option of mixing the Arrow River water into Mill Creek before it enters the lake, and thereby increasing the amount of surface water entrained in the underflowing density current, may result in a very different conclusion.

4. *How much P and chlorophyll a could the augmented flow flush from the lake and what effect would this have on trophic state*? These calculations are essentially correct except that the question has been answered on the basis of the augmentation water rather than the augmented flow i.e., Mill Creek plus Arrow River irrigation water. The augmentation water would from the Arrow River would displace about 13% of the epilimnion above 12 m depth while the augmented flow would displace about 64%. In

terms of the displacement of chlorophyll *a* (algal biomass) the augmented flow could effectively reduce the average chlorophyll *a* concentration from 30 mg m^{-3} to around 12 mg m^{-3}, improving the water quality from supertrophic (TLI >5) to eutrophic (TLI between 4 and 5). However, algal growth would continue to increase the concentration of chlorophyll *a* and there would most likely be no change in the lake trophic status. The report comments around a gradual improvement in lake water quality over time with enhanced flushing probably are correct.

The caveats included in this section are valid and should be considered before employing augmentation as a tool for the restoration of Lake Hayes.

Because of the various errors and omissions associated with the entrainment of surface water into the inflow and the consideration of the augmentation water rather than the combined augmented inflow, the information and conclusions drawn in this appendix need to be revised after new calculations.

Appendix 3: A rough Lake Hayes alum dosing estimate. This appendix is a 'cut and paste' of calculations done for Lake Hayes by John Quinn, Max Gibbs and Chris Hickey (NIWA) in 2012.

Appendix 4: Catchment management to restore and protect Lake Hayes should be the primary focus of any restoration package on Lake Hayes. Catchment management strategies will have long-term benefits and can be augmented by short-term strategies that address specific issues designed to improve the water quality of the lake.

The identification of which part of the catchment the phosphorus is coming from (e.g., Caruso 2000b) should allow targeting of management strategies for best use of limited resources.

2.6 Summary

Within the context of the original request by the Friend of Lake Hayes Society Ltd, the Schallenberg and Schallenberg (2017) report presents a comprehensive description of the current state of the Lake Hayes and summarises the major issues affecting lake water quality. It presents recommendations and discusses realistic restoration options together with a restoration strategy with timelines. The report also recommends monitoring strategies for assessing lake status and recovery to a stable water quality state.

Data used within the report is mostly pre-2015 and the more recent data may have changed or reinforced some of the conclusions. The omission of the entrainment factor requires a re-assessment of the conclusions about the efficacy of the use of irrigation water to augment the Mill Creek inflow as a strategy for accelerating the restoration of Lake Hayes water quality.

2.7 Recommendations

- The report needs to be revised to correct or qualify the identified errors and to revisit the effects of entrainment on the oxygen transport via the Mill Creek density current into the hypolimnion.
- The report needs to manage the reader's expectation of what is meant by 'recovery' and 'restoration' in terms of the likely level of improvement and the timeframes for that improvement.

3 A look at three defined options for water quality improvement

The three defined options for water quality improvements – flow augmentation from the Arrow River irrigation scheme, destratification and sediment capping – are in-lake remedial actions that could potentially have immediate impacts on lake water quality. Flow augmentation from the Arrow River irrigation scheme has largely been covered in the Schallenberg and Schallenberg (2017) report review but will also be addressed in this section in more detail with an inclusion of the entrainment factor associated with the Mill Creek inflow.

To assess the three defined options for water quality improvement, all available and relevant data on Lake Hayes were compiled to provide time series databases of nutrients and chlorophyll-*a* (as an indicator of algal biomass), as well as temperature, DO, pH, suspended solids and alkalinity. This information includes depth-correlated temperature, DO, chlorophyll-*a* and specific conductivity profiles through the full depth of the lake water column, and data from occasional research studies.

These occasional research studies include periods of high frequency temperature and DO data from a thermistor chain with multiple sensors at fixed depths and fitted with near surface and near bottom DO loggers, recording at 15-minute intervals in summer (Figure 1-4, Figure 1-5). Lake circulation patterns were inferred from an acoustic Doppler current profiler (ADCP) mounted on the bottom of the lake on the western side and recording in burst mode for 1 minute at 5-minute intervals (Figure 1-2, Figure 1-3). Key findings from these ad hoc research projects have been presented in Section 1.1 of this report.

An outcome from the Schallenberg and Schallenberg (2017) report review is the inverse relationship between water clarity and biomass of *Ceratium hirundinella*. If the water clarity in Lake Hayes is to improve, *Ceratium* biomass needs to be reduced and therefore it is important to understand the factors that favour *Ceratium* growth and dominance in the lake phytoplankton community. Consequently, information on the physiology of this nuisance bloom-forming dinoflagellate has been compiled from literature, including information from the Schallenberg and Schallenberg (2017) report. A summary of key factors are presented here.

3.1 *Ceratium hirundinella*

Ceratium hirundinella is a motile, horned dinoflagellate, with a cell length of 150 µm to 200 µm. It does not have a silica sheath as found on diatoms. Rather it has an armoured thecae comprising cellulose plates and giving the cells a high carbon to volume ratio. Because it is motile, it has the ability to regulate its depth for optimal light for photosynthesis during the day, and can move down to the thermocline to find nutrient N and P from the hypolimnion at night. It usually confines itself to a well-defined depth layer during the day (Figure 3-1) but may disperse more widely at night. It has essentially the same physical requirements for growth as for cyanobacteria (i.e., calm conditions and thermal stratification) and its growth is favoured by low N:P ratios. (Hart and Wragg 2009). They found that *Ceratium* became dominant when N:P ratios were around 5 but disappeared when the N:P ratio was >10. Nakano et al. (1999) also found that *Ceratium* blooms in Lake Biwa were related to increases in P in the water column but found that growth may be nitrogen limited. i.e., increases in both N and P with an N:P ratio <10 are conditions that favour the development of *Ceratium* blooms.

Unlike cyanobacteria, *Ceratium* is mixotrophic, i.e., it can gain energy both through photosynthesis as plants do, and by feeding on bacteria as some protozoans and zooplankton do (Gerdeaux and Perga 2006). Consequently, while *Daphnia* may be physically large enough to consume *Ceratium* cells, their ability to outcompete *Ceratium* for available bacteria may be the main cause of the reduction in *Ceratium* biomass in some years., i.e., the *Ceratium* may become carbon limited.

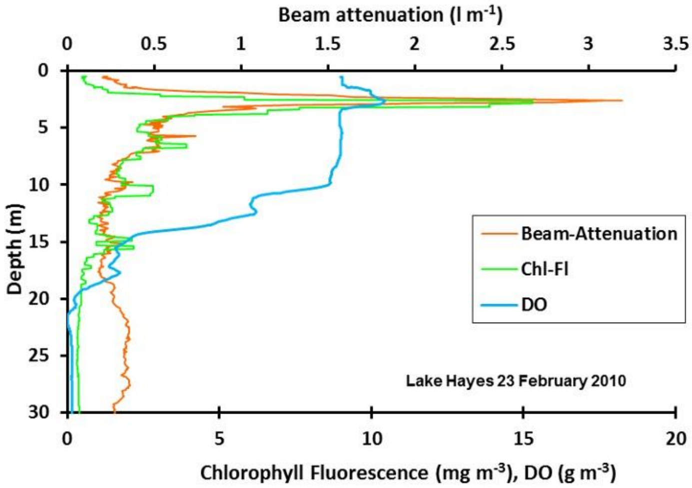


Figure 3-1: Particulates (beam attenuation), chlorophyll fluorescence and dissolved oxygen profiles in Lake Hayes on 23/02/2010, about 10 am. The beam attenuation shows that most particles were concentrated in a thin layer around 3 m depth. The chlorophyll fluorescence peak at the depth of the beam attenuation peak shows that the particles were phytoplankton (identified as *Ceratium hirundinella*). The DO increase at the depth of the chlorophyll peak shows that the *Ceratium* were photosynthesising and therefore they were alive. The thermocline depth was 10 m. (NIWA unpublished data).

Ceratium can tolerate a broad range of temperature with a development range between 4°C and 23°C (Heaney et al. 1983) and an optimal growth range between 12°C and 23°C (Hutchinson 1967, Heaney et al. 1988). *Ceratium* over-winters by producing cysts that fall to the sediment in autumn and germinate in spring (Heaney et al. 1983).

During the summer growth period, *Ceratium* cells divide at night (between 9 pm and 9 am with the majority of divisions occurring at about 3 am in the morning (Heller 1977, Heaney and Talling 1980). At this time, the *Ceratium* cells would be on the thermocline with access to high P concentration from the hypolimnion. It is possible that they rest on the thermocline to facilitate cell division.

Low light and temperature are unfavourable conditions for *Ceratium* in temperate zone lakes (Heaney et al. 1988) whereas high temperatures are unfavourable conditions in subtropical zone lakes in summer (Pollingher and Hickel 1991).

Growth is potentially regulated by ecological factors such as the presence of an anoxic hypolimnion and water column stability (Pérez-Martínez and Sânchez-Castillo 2002). In this case the anoxic hypolimnion provides the excess P required for growth.

Based on the above information, managing *Ceratium* blooms in Lake Hayes will require the reduction of both external and internal P inputs to the lake water column. Destratification of the lake could remove the thermocline, which appears to be a resting zone for *Ceratium* at night as well a place for cell division and a 'refuelling' site for gathering P.

Destratification would also oxygenate the bottom water and reduce or prevent the release of the DRP flux from the sediment – a key part of the internal cycling of P. Augmenting the Mill Creek inflow with excess irrigation water from the Arrow River would increase the transport of oxygen into the lake. This could reduce the DRP flux from the sediment by precipitating it with the ambient iron and manganese oxides. Sediment capping with alum would also sequester any DRP from the water column and would cause the precipitation of the particulate P and *Ceratium* cells in the resultant floc.

3.2 Flow augmentation from the Arrow River irrigation scheme

While it would be a relatively simple procedure to introduce irrigation scheme water from the Arrow River into Mill Creek, estimating the effect of that augmentation on the restoration prospects for Lake Hayes is more complex. If it is assumed that Arrow River and the Mill Creek waters are 100% saturated with oxygen, say 10 g m^{-3}, the mass of oxygen in the combined flow can be calculated as the total flow times the oxygen concentration. Using the irrigation water volumes provided in the Schallenberg and Schallenberg (2017) report and the mean monthly flow data from the Caruso (2000b) report, the estimated mean daily mass transport of oxygen from the Mill Creek inflow in January would be about 0.45 t d^{-1}. This is much less than the oxygen demand estimated in the HOD rate of about 1.57 t d^{-1} for January 2017. However, if an entrainment factor of 4 is included in the calculation, the mass of oxygen in the density current formed by the Mill Creek inflow would be about 1.8 t d^{-1}, more than enough to compensate for the oxygen demand in the hypolimnion.

The complexity of this process is apparent when considering that Mill Creek is already flowing into the lake carrying about 1.6 t d^{-1} during January and the augmentation water would only add another 0.2 t d^{-1}. Despite this, the hypolimnion is still losing oxygen so that it became anoxic in March 2017. This implies that other factors are influencing the mass transport of oxygen into the hypolimnion of Lake Hayes and that the HOD rate is the net loss of oxygen from the hypolimnion.

The main factor affecting oxygen transport into the hypolimnion will be the temperature difference between inflow water from Mill Creek and the surface water in Lake Hayes (Figure 3-2).

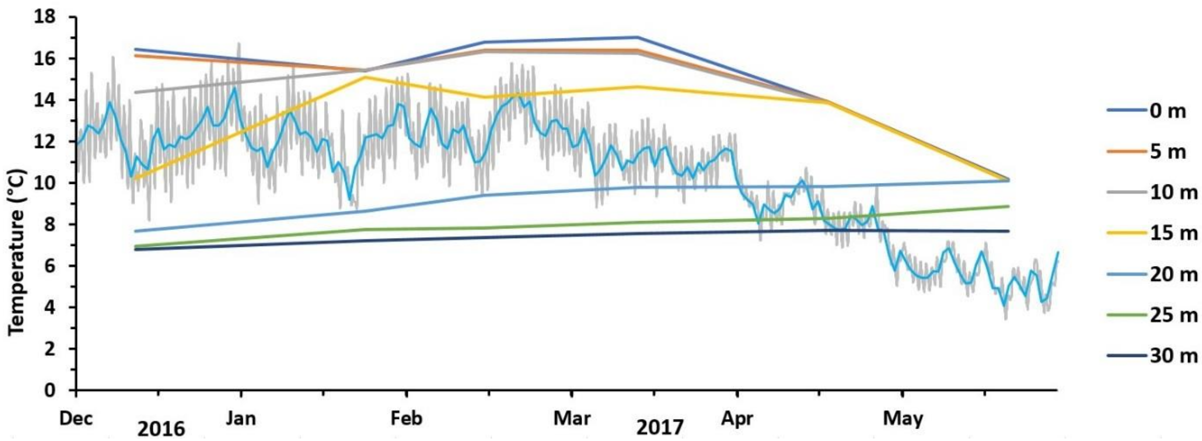


Figure 3-2: Comparison of Mill Creek water temperatures (wavy blue [daily] and grey [hourly] line) with lake temperatures at different depth (coloured lines - depth scale right hand axis). Mill Creek temperature data from 2016 to2017 show the diurnal cycle of heating and cooling (grey hourly line) and have been manually aligned with periodic Lake Hayes temperature profile data. (Temperature data from ORC).

Mill Creek data show a strong diurnal temperature cycle which has a range of about 4°C between midday and midnight. In Lake Rotoiti, that level of temperature difference was enough to direct water from the Ohau Channel from Lake Rotorua into a surface buoyant plume during the day or cause it to plunge as an underflowing density current along the bed of Lake Rotoiti at night (Vincent et al. 1991). There is no question that a similar effect will be happening in Lake Hayes, although the magnitude will be smaller. The intrusion layer from Mill Creek inflow is apparent in the 21-m depth temperature and DO data from the thermistor chain (Figure 1-5). The other difference will be how long the underflowing density current feature is operating each day and whether the density current flows into the hypolimnion (condition A, Figure 3-3) or finds a depth of neutral density higher in the lake and lifts off the lake bed to become an intrusion flow (condition B, Figure 3-3).

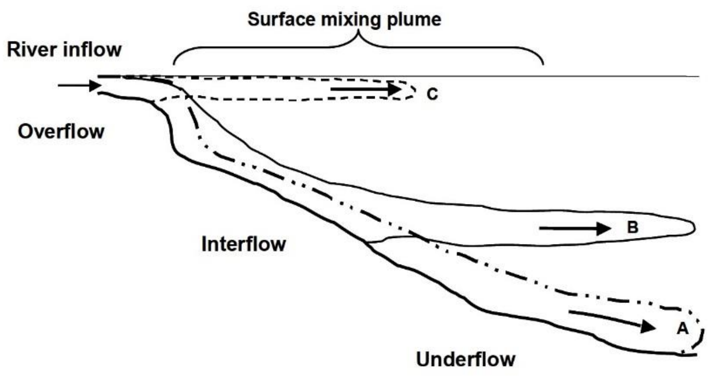


Figure 3-3: Density currents. A) Very cold water will become an underflow along the lake bed, **B**) medium temperature water will plunge to a depth of equal density where it becomes an interflow, **C**) warmer water enters the lake surface as a buoyant overflow which may form a visible plume on the surface. Both interflow and underflow entrain surface water with the flow. (Figure 3-7 from Gibbs and Hickey 2012).

Physical factors that influences the formation of a density current is the shape (width and depth) of the stream mouth and the bed shape in the lake at the mouth. In Lake Rotoiti, the Ohau Channel flowed over a sill with a 5-m drop-off giving a sharp transition from inflow to lake and a geometry that encourages entrainment of the lake surface water into the flow (entrainment factor of about 4-5). In Lake Hayes, Mill Creek flows into 8 m deep water and this is likely to result in a similar inflow/entrainment geometry. The entrainment factor can be calculated but requires more data than is currently available. It would be reasonable to expect an entrainment factor of around 4.

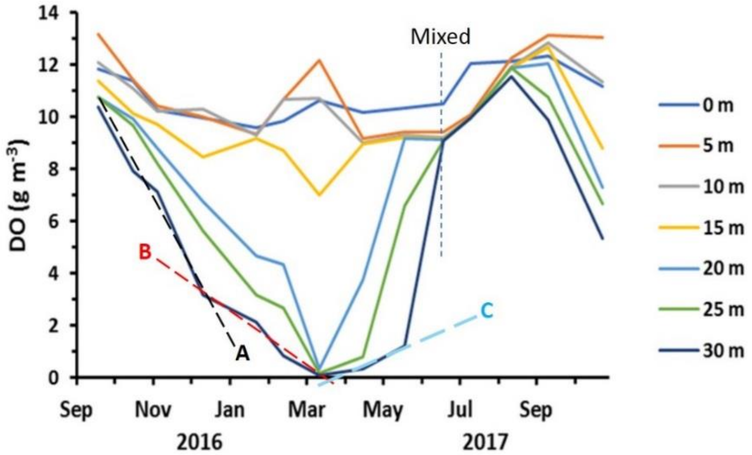


Figure 3-4: Time-series dissolved oxygen (DO) profiles in Lake Hayes 2016/17. Broken lines are regression lines which indicate the rate of hypolimnetic DO depletion at different times across the stratified period: A) spring (November-December: [HOD rate 85 mg m^{-3} d^{-1}], B) summer (January-March): [HOD rate 35 mg m^{-3} d^{-1}], C) Autumn (March-May): [HOD rate -16 mg m^{-3} d^{-1}]. The lake was fully mixed in June.

Comparing the Mill Creek temperature data with the 15-m depth temperature (Figure 3-2), it is apparent that through December 2016 the inflow would most likely form an interflow into the epilimnion or metalimnion (thermocline). Without a large proportion of underflow, the oxygen depletion rate is relatively rapid at 85 mg m^{-3} d^{-1} (line A, Figure 3-4). In January 2017, the inflow is likely to be switching between interflow and underflow with an increasing proportion of underflow contributing oxygen the hypolimnion resulting in a reduction in the oxygen depletion rate to around 35 mg m^{-3} d^{-1} (line B, Figure 3-4).

Around March, the Mill Creek water temperature decreased markedly (Figure 3-2) and it is likely that the density current would be mostly as an underflow. If the transport of oxygen in that underflow was greater than the oxygen demand in the hypolimnion, the hypolimnion would begin to re-oxygenate. The apparent reoxygenation starting in March 2017 at a rate of around 16 mg m^{-3} d^{-1} (line C, Figure 3-4) is consistent with an oxygen input load greater than the oxygen demand in the hypolimnion. The oxygen recharge between the May and June sampling dates shows an average reoxygenation rate of around 300 mg m^{-3} d^{-1}. However, this reoxygenation phase is normally associated with lake turn over that occurs over a period of about a day, which would represent a rate of around 8000 mg m^{-3} d^{-1} (Figure 3-5).

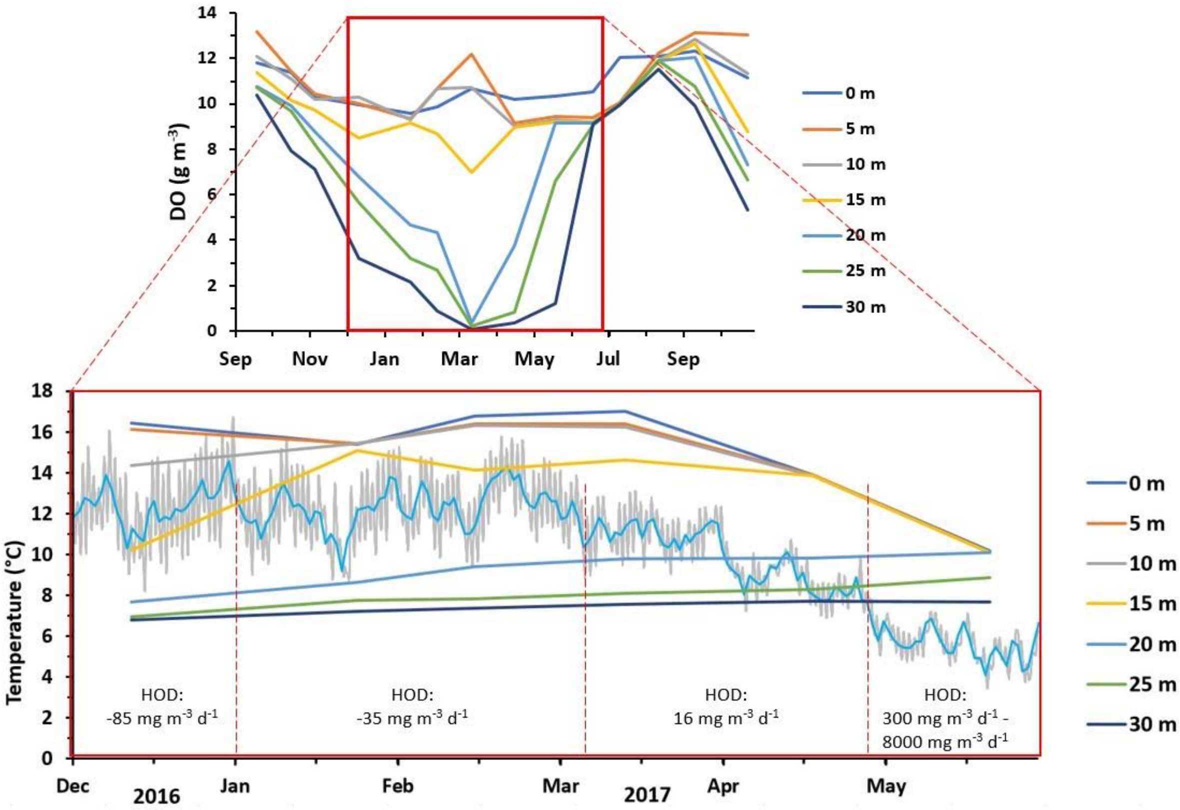


Figure 3-5: Alignment of dissolved oxygen data with the temperature data from Figure 3-4. Negative HOD values indicate oxygen loss.

These data indicate that, assuming the water temperatures in Mill Creek follow a similar pattern each year, the present inflow from Mill Creek with entrainment is sufficient to slow the oxygen depletion in summer and reoxygenate the hypolimnion in autumn. Augmentation of the Mill Creek inflow with irrigation water from the Arrow River would increase the mass of oxygen transported by the density current. If the temperature of the water from the Arrow River is colder than Mill Creek (large rivers are often colder than small streams) then the augmented flow would be likely to enhance underflow instead of interflow through November to January. This would greatly reduce the degree of oxygen depletion in the hypolimnion.

The brief period of anoxia in March 2017 suggests that the lake is close to a tipping point of no anoxia and therefore a greatly reduced internal loading of DRP, which is likely to be driving the growth of the *Ceratium* blooms. Enhanced underflow in spring and early summer could be sufficient to prevent that anoxia occurring.

Caution: These scenarios are based on the temperature regimes in the lake and in Mill Creek in 2016-17. There is no guarantee these conditions are the new norm and will not change to a different regime that reduces the underflow as climate variability causes larger swings in the temperature cycle.

For example, temperature data from Mill Creek in 2018 reached day time maxima of 23.4°C (29 January 2018), which was 5°C warmer than the same period in 2017. There is no lake profile water quality data available for early 2018 at the time of writing this report to determine what affect this had on hypolimnetic oxygen depletion. There was, however, an increase in algal biomass suggesting a possible release of DRP from the sediment. A chlorophyll fluorescence profile on 20 January 2018 indicates thermal stratification and the dominant algal species was the diatom. cf. *Cyclotella* sp.

When the Mill Creek inflow temperatures are aligned with the depth of the thermocline i.e., about 15 m depth, from January to March (Figure 3-2), there is another factor that that may determine the depth of insertion of the density current and its ultimate mixing in Lake Hayes, i.e., wind speed and direction.

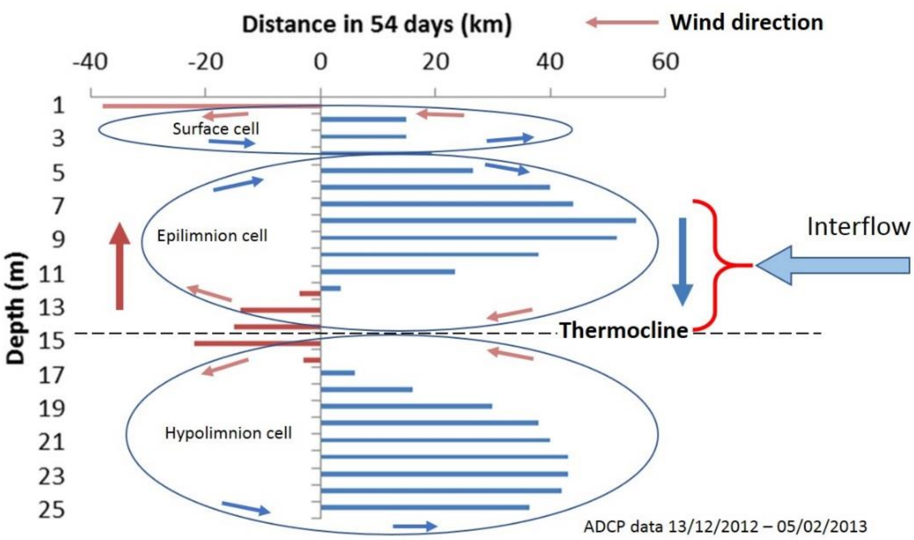


Figure 3-6: Lake circulation currents are likely to affect the depth of insertion of the interflow. Figure shows the effect of a persistent northerly wind flow from December 2012 to February 2013. (NIWA unpublished data).

The short-term deployment of an ADCP in Lake Hayes in December 2012 through to February 2013 found that the persistent northerly wind flow along the axis of the lake set up internal lake currents that formed contra-rotating vertical cells in the water column. While this is not unusual in many lakes, in Lake Hayes, the cells meet at the thermocline depth with downwelling at the upwind end and upwelling at the downwind end. This means that water from Mill Creek will be drawn down to the thermocline if it forms an interflow that inserts below about 10 m depth (best guess) (Figure 3-6). Consequently, the oxygenated water in the density current is likely to be entering the hypolimnion for a longer period each day than indicated by temperature difference alone. When the inflow forms an underflowing density, this pattern of lake currents will not affect the depth of insertion below the thermocline but will rapidly disperse the oxygenated water throughout the hypolimnion. When the wind direction changes to a persistent southerly, the downwind end is against the Mill Creek inflow and this would most likely prevent all but an underflowing density current from reaching the hypolimnion.

Further evidence of reducing oxygen depletion is shown in DRP concentration data from the hypolimnion (Figure 3-7) and the maximum chlorophyll data in the epilimnion (Figure 3-8).

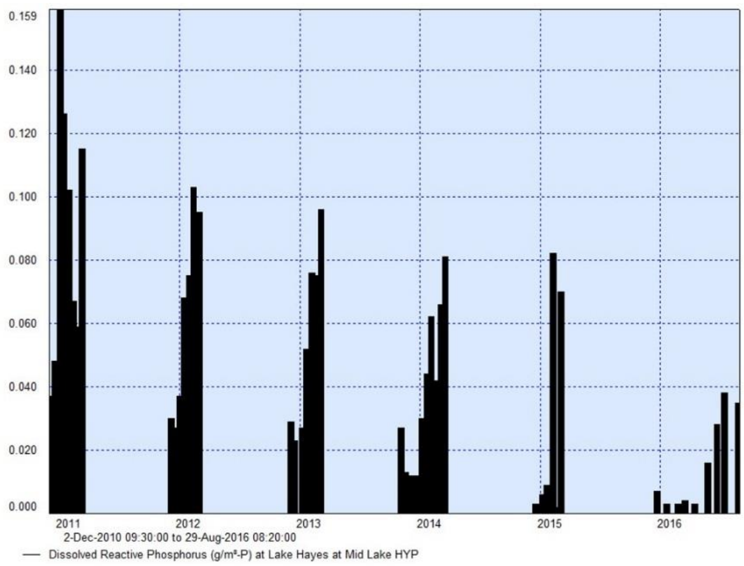


Figure 3-7: Time-series dissolved reactive phosphorus (DRP) concentrations in the hypolimnion from 2011 to 2016. Between 2011 and 2015 (inclusive) data was only collected between December and March (inclusive). (Data from ORC).

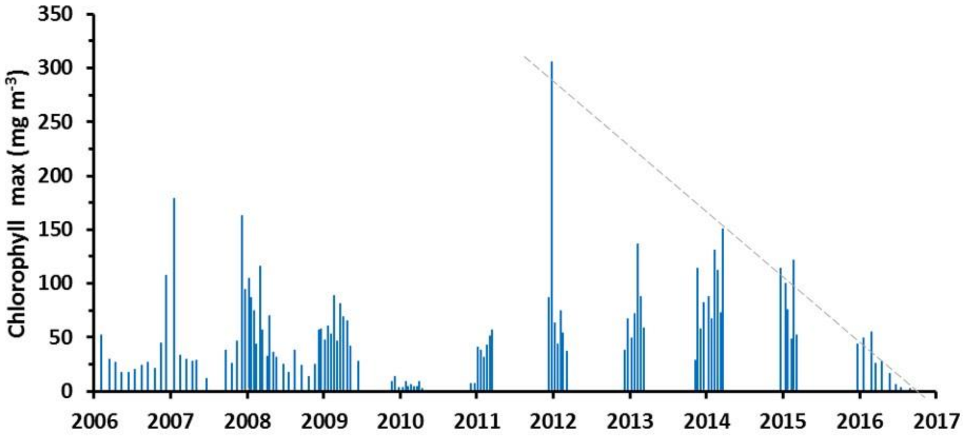


Figure 3-8: Maximum chlorophyll concentrations in summer from time-series profiles in Lake Hayes. Profiles were collected at the mid lake site as part of the ORC monitoring of the lake. From 2011 to 2015, the data were only collected between December and March (inclusive) and do not show the winter concentrations. Dashed line provides a visual indication of trends based on the maximum chlorophyll concentrations.

Phosphorus is released from the sediment under anoxic conditions and, between 2011 and 2016, the DRP concentrations have gradually reduced with the release time moving towards March, consistent with a reduction in the intensity of anoxia. The very low DRP release in 2016-17 is consistent with the very brief period of anoxia that summer (Figure 3-4) and may be the cause of the low algal biomass that summer (Figure 3-8) rather than just grazing pressure from zooplankton.

The apparent decrease in maximum chlorophyll concentrations from the extreme high in 2012 (Figure 3-8) appears to be consistent with the reduction in DRP in the hypolimnion (Figure 3-7), although there may be other plausible explanations for this change in the time-series data. Note that summer 2009-10 was a clear water year (Schallenberg and Schallenberg 2017) with the appearance of *Daphnia pulex* in high abundance. This occurrence disrupted NIWA and University of Otago experiments looking at flocculation and sediment capping (personal observations by the author).

The 2016-17 chlorophyll maximum concentrations were not as low as in 2009-10 but this was also classified as a clear water year, with the occurrence of high numbers of *Daphnia pulex* (Schallenberg and Schallenberg 2017) although there are zooplankton data to support this statement.

A prediction in December 2017 that summer 2017-18 was also set to be a clear water year (Adam Uytendaal, ORC, pers. comm.) did not allow for climate variability and the warmest summer in many years. This presumably resulted in release of DRP from the sediment, which a proliferation of algae and a reduction in water clarity.

3.2.1 Conclusions

There are indications that Lake Hayes is moving from a supertrophic condition towards a eutrophic condition with slowly improving water quality. Whether this is a tipping point for recovery, as suggested in the Schallenberg and Schallenberg (2017) report, is uncertain. Indications of improving water quality include:

- Hypolimnetic oxygen depletion rates are decreasing and are presently less than half the maximum rate recorded in 2007.

- The period of bottom hypolimnion anoxia has reduced from >5 months in 2007-8 to <1 month in 2016-17.
- Reoxygenation of the hypolimnion from the Mill Creek inflow is implied in the data from January in the summer of 2016-17. There was no indication of reoxygenation in summer 2007-8 until March when the water column began to reoxygenate from the 10-m isobath downwards rather than from the bottom (Figure 2-2).
- The release of DRP from the sediment has been slowly decreasing since 2011.
- Algal biomass, as indicated by chlorophyll concentrations, appears to have been decreasing since 2011.

These changes are all consistent with a change in the amount and duration of the underflow from the Mill Creek inflow. Other factors could also explain these changes. For example, a gradual reduction in maximum algal biomass in the epilimnion would result in a lower organic carbon load in the sediment and therefore a reduced sediment oxygen demand. This would be consistent with the reducing HOD. This mechanism appears to be occurring in Lake Rotorua where the water quality has improved from eutrophic to mesotrophic after reducing the DRP input to the lake.

Acoustic doppler current profiler data indicates that wind flow along the axis of the lake may be enhancing the mixing of DO into the lake when there is a northerly wind and could be suppressing that mixing during a southerly wind. A schematic diagram (Figure 3-9) shows stylized flow paths of oxygen transport from several different sources using the different mechanisms. This is based on the minimal amount of data available and, although consistent with other studies, is speculative for Lake Hayes and requires further investigation.

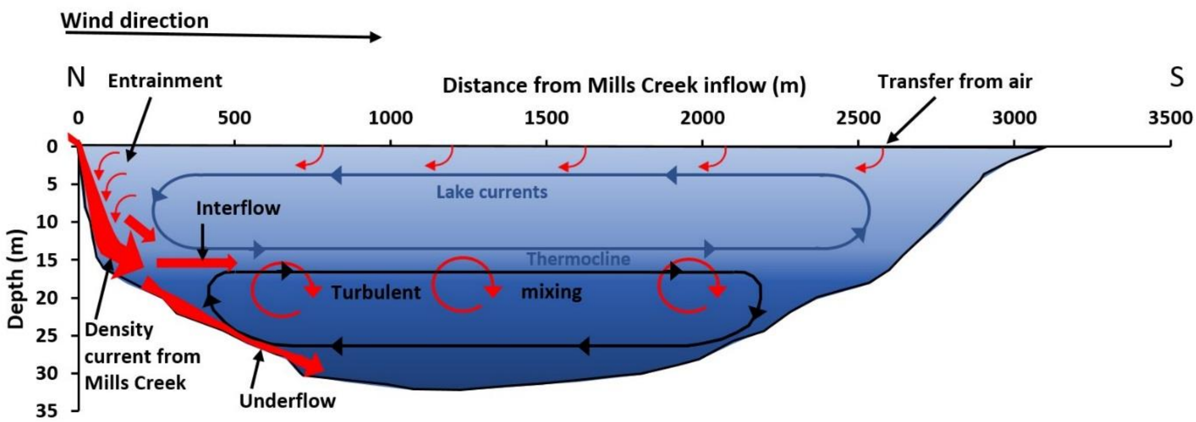


Figure 3-9: Schematic diagram showing a vertical cross-section along the north-south axis of Lake Hayes overlaid with oxygen flow pathways and transport mechanisms. Stylised oxygen input pathways (red) show where entrainment occurs. Internal lake currents – blue (epilimnion), Black (hypolimnion) – interact with the oxygen pathways and mix oxygen into the hypolimnion due to turbulent mixing at the thermocline (Schematic by Max Gibbs).

If the addition of irrigation water from the Arrow River causes the temperature to fall, the period of underflow will increase and the hypolimnion is likely to remain aerobic throughout summer and the recovery of the lake will be accelerated.

However, if the addition of the irrigation water causes the temperature in Mill Creek to rise, the period of underflow most likely will be reduced and the apparent natural recovery of the lake could be reversed.

Caution is required. This is a very fragile situation that is likely to change if climate variability causes Mill Creek to warm as is indicated in the 2018 data, i.e., the reoxygenation of the hypolimnion by the oxygen transfer from the Mill Creek inflow cannot be relied on as the long-term solution for restoration of Lake Hayes.

3.2.2 Recommendations

A lake monitoring buoy should be installed on Lake Hayes to obtain high frequency (15-minute interval) temperature and dissolved oxygen concentrations at selected depths in the lake water column in order to understand how this lake works.

As temperature during the stratified period appears to be critical to the recovery of Lake Hayes using the augmentation of Mill Creek, it is critical to measure the temperatures of Mill Creek and the irrigation water before any augmentation is implemented. Temperature monitoring should continue for a full year to enable comparison with temperature measurements from the lake monitoring buoy and determine when it is appropriate to add irrigation water and when it is not.

In the interim, the long-temperature record from Mill Creek and the periodic temperature and DO profiles from Lake Hayes may provide sufficient data to give an indication of what temperature regime causes underflow and reoxygenation of the lake hypolimnion, and how often those conditions occurred over the data record. These data could be modelled to determine when and where in Lake Hayes the Mill Creek water inserts and mixes each year.

In-lake currents induced by different wind directions should be measured in order to understand how these currents affect the dispersion of DO into Lake Hayes. A pair of ADCPs should be used, one on each side of the lake, in order to determine the lateral circulation patterns as well as the vertical currents. These data should be correlated with wind speed and direction and rainfall records for the Lake Hayes area to further develop the mechanism for reoxygenation suggested.

3.2.3 Secondary or amenity effects

Because the addition of augmentation water to Mill Creek will only increase the volume by a small amount (14%, Schallenberg and Schallenberg 2017) there should be no visual impact on the stream and any change in water clarity would most likely be imperceptible to the eye. Provided the irrigation water added to Mill Creek is odourless, there should be no perceptible change in odour and only a small change in the sound of the stream as flows over the gravel beds.

3.3 Destratification

3.3.1 Why destratify the lake?

Destratification removes the thermocline, which is the single greatest barrier to the transfer of dissolved oxygen down to the bottom of the lake. The loss of DO due to sediment decomposition processes is the main cause of bottom water anoxia. This causes P release from the sediment as an internal load that stimulates the development of summer algal blooms, most recently of the dinoflagellate, *Ceratium hirundinella*. As discussed in section 3.1, *Ceratium* moves up through the water column to the well-lit photic zone during the day and then settles down to the thermocline at night. It is this night phase where the thermocline is important as the *Ceratium* cells use it as a resting platform so they can take up DRP and ammonium from the hypolimnion. They may also use it as a resting place during cell division, which also occurs at night. Removing the thermocline would interrupt feeding and cell division, thereby attacking the key growth steps in the *Ceratium* life-cycle. Studies also indicate that surged operation of the aeration system may be used manage some algal species (Lilndenschmidt 1999).

A paper by Kirke (2000) provides an overview of destratification and mixing using aeration and mechanical mixers. There are several options for lake mixing (e.g., Singleton and Little 2006a, b) and destratification. Most of these either use air-lift techniques or propellers to induce vertical mixing currents to the surface, or water jets to carry surface water down into the lake. The simplest aerator is the bubble-plume system where compressed air is blown through pipes to one or more diffusers on the lake bed. This diffuser system is referred to as a sparge pipe. The design of the sparge pipe produces a rising column of air bubbles, which entrains bottom lake water into the plume and induces a vertical current in the water column (Schladow 1993) (Figure 3-10). The depth of the air outlet is important - the greater the depth, the more efficient the mixing (Cooke et al. 1993). This is because bubbles produced from the sparge pipe expand as they rise to the surface. When they leave the sparge pipe jets they double in size and their volume increases by a factor of 4 for every 10 m they rise (Figure 3-10 A). This means that, in Lake Hayes at 30 m maximum depth, the bubbles will expand by a factor of 64 as they reach the surface.

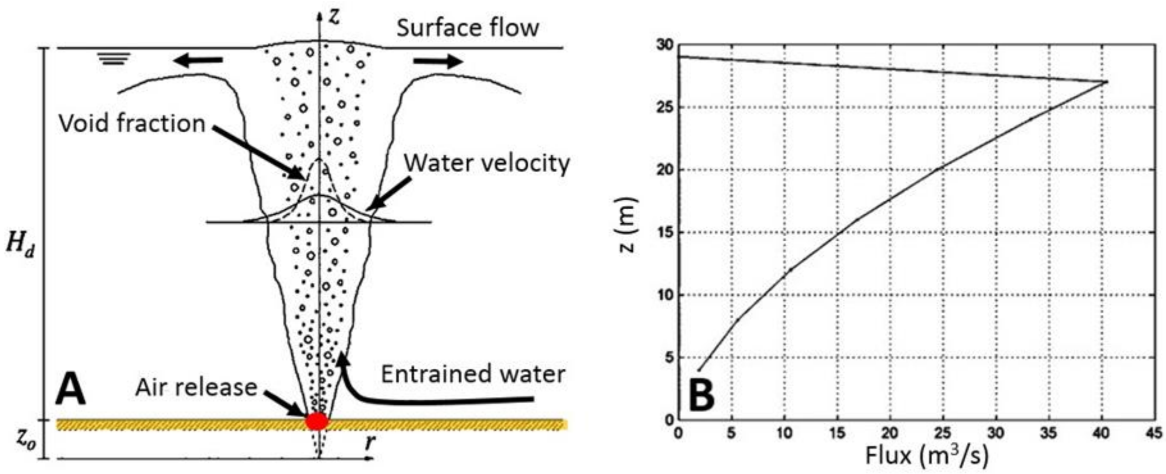


Figure 3-10: Bubble plume representation **A**) Near isothermal water column, **B**) estimation of vertical water movement (flux) in m^3 s^{-1} (Stylised images redrawn from internet images. Flux values would depend on the volume of air and the length of the diffuser(s)).

Because the air bubbles occupy a finite volume in the water column but have almost no mass, the water in the bubble plume is less dense (more buoyant) than the adjacent lake water and the water inside the bubble plume rises to the surface as a buoyant plume. As the bubble plume rises, it entrains bottom water and ambient water from outside the plume so that the total flux of water moving upwards in the plume increases (Figure 3-10,B). The amount of air pumped into the sparge line will influence the efficiency of the rising plume. Too little and the plume may not form. Too much and the bubbles may disrupt the integrity of the plume reducing its efficiency.

The air flow required to de-stratify a lake is a function of the shape of the lake and the surface area. USEPA estimate that a compressed air flow rate of 9.2 m^3 min^{-1} km^{-2} lake area should be sufficient to achieve adequate surface re-aeration in most lakes (Lorenzen and Fast 1977). This is equivalent to 9.2 x 35.31 = 325 cubic feet per minute (cfm).

At the surface, the entrained water is dispersed laterally away from the bubble field as a surface current (Figure 3-11), which will propagate across the lake until it reaches an obstruction or the shore. There the hydraulic inertia of the water in the surface current causes it to plunge to the lake bed where it will replace the bottom water being entrained into the bubble plume. Note that, although the air bubbles in the plume make the water less dense, the density difference is small and insufficient to pose a hazard to boats or swimmers.

Figure 3-11: View of the surface bubble field along the axis from a bubble plume aeration system. This bubble plume aerator was installed in Lake Waikapiro, Hawke's Bay. This photo shows the current lines moving away to the left and right from the line of the bubble plume. [Photo by Andy Hicks, Hawke's Bay Regional Council].

The bubble plume aeration/mixing mechanism is more efficient than any other method of de-stratifying a lake, or preventing it from becoming thermally stratified. The air used to generate the bubble plume has very little effect on the oxygenation of the bottom water entrained to the surface in the plume. Oxygen adsorption from the atmosphere re-oxygenates the oxygen depleted water as it flows across the surface of the lake.

3.3.2 Bubble plume aerator design and positioning

Because Lake Hayes was formed from in a glacial valley, the lake morphometry is simple with no ridges or rock features on the bottom (i.e., there is nothing to obstruct the circulation currents that will be generated by the destratification mechanism). The lake is elongated – 3.1 km long by about 800 m wide and is essentially bath-shaped. The main inflow, Mill Creek, is at the northern end and the outflow, Hayes Creek, is at the southern end. The deepest part of the lake is near the middle at about 1.2 km from the northern end. This would be the ideal position for the installation of a bubble plume aeration system.

The design of the bubble plume aerator is very simple. It comprises a sparge pipe diffuser attached by tethers to a heavy cable (ballast rope) that is laid across the lake bed (or reservoir) and is anchored at each end with heavy concrete blocks (Figure 3-12). The sparge pipe diffuser has a series of small (1–1.5 mm) holes drilled through the upper side at 20–30 cm intervals along its length. Compressed air pumped into the tube from a shore mounted compressor, escapes through these holes as tiny bubbles, which rise to the surface as a bubble plume. The heavy ballast rope keeps the sparge pipe diffuser at the bottom of the lake when that pipe is full of air. A buoyancy tube is also attached to the ballast rope (Figure 3-12). During normal operation, the buoyancy tube is full of water and is held at the bottom of the lake by the weight of the ballast rope. When the water is blown out of the buoyancy tube and it is full of air, its positive buoyancy exceeds the negative buoyancy of the ballast rope and the aerator system floats to the surface. This enables the whole system to be installed or raised for cleaning without requiring the use of divers.

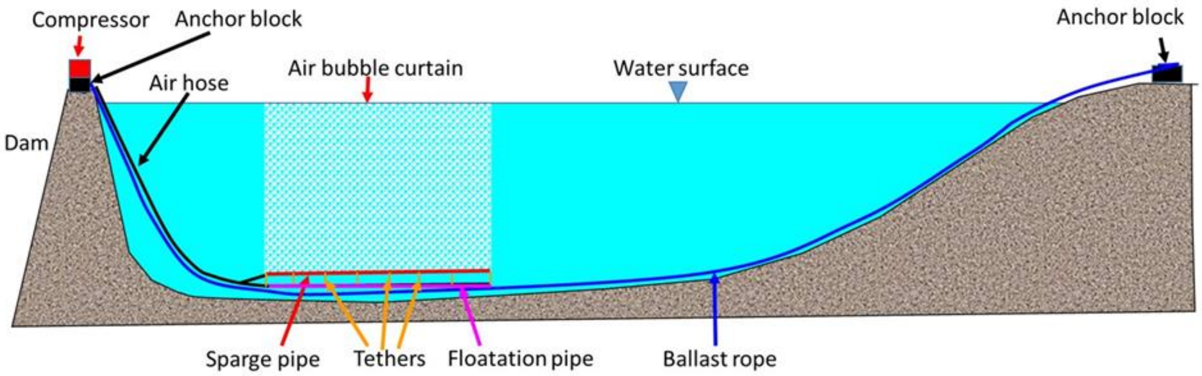


Figure 3-12: Schematic diagram of a bottom-mounted bubble curtain aerator system in a reservoir. The sparge pipe for small reservoirs is typically 50–100 m but can be much longer for larger reservoirs and lakes. The ballast rope is anchored at both ends and orientated along the deep axis of the reservoir or across the deepest part of a lake, so that it lies on top of the bed. The ballast rope can be 32 mm trawler rope or a similar sized plastic sheathed multi-core steel cable.

Orientation of the sparge pipe and its length depends on how the system is to be used. In operation, the bubble plume forms a curtain along the length of the sparge pipe. This results in a continuous strip of rising water that generates a laminar flow current on each side of the bubble curtain. In Lake Hayes, the sparge pipe should be placed across the deepest and narrowest part of the lake so that the laminar current flow is focused along the length of the lake (Figure 3-13).

The circulation currents will flow to each end of the lake at the surface adsorbing oxygen from the atmosphere before plunging to the bottom and flowing along the lake bed to the aerator sparge line. This orientation generates a circulation flow path (Figure 3-14) that carries oxygen to the bottom of the lake from the atmosphere and enhances the underflow from the Mill Creek inflow.

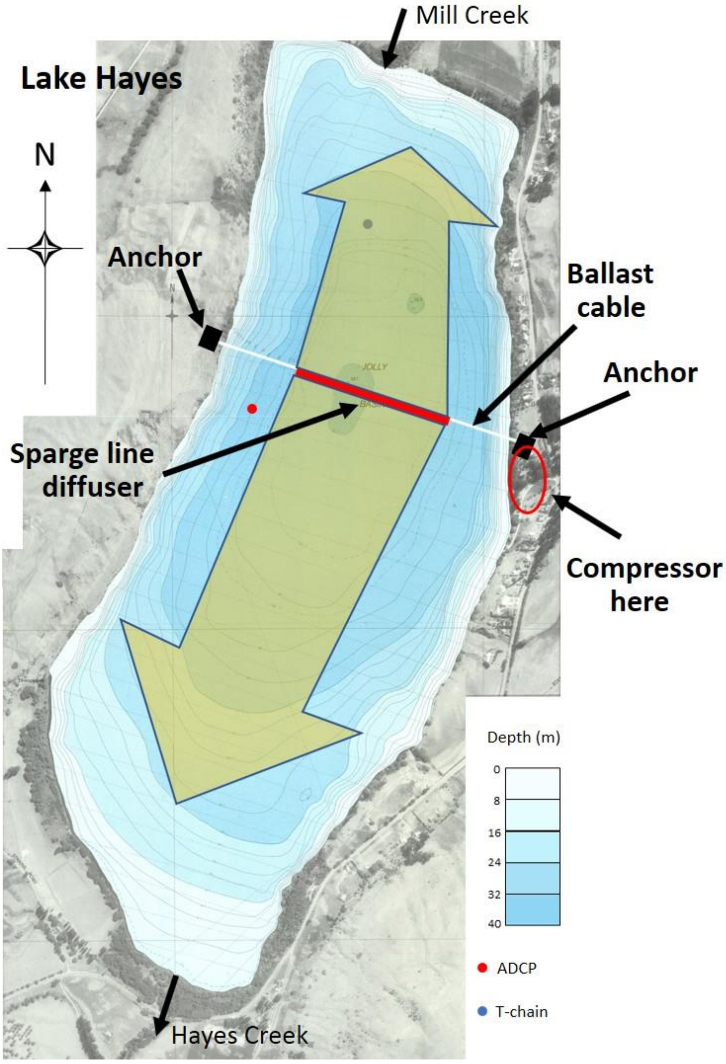


Figure 3-13: Optimal positioning of bubble curtain aeration sparge line diffuser in Lake Hayes. The expected surface flow patterns are indicated by the orange shaded arrows. The return paths are along the lake bed beneath.

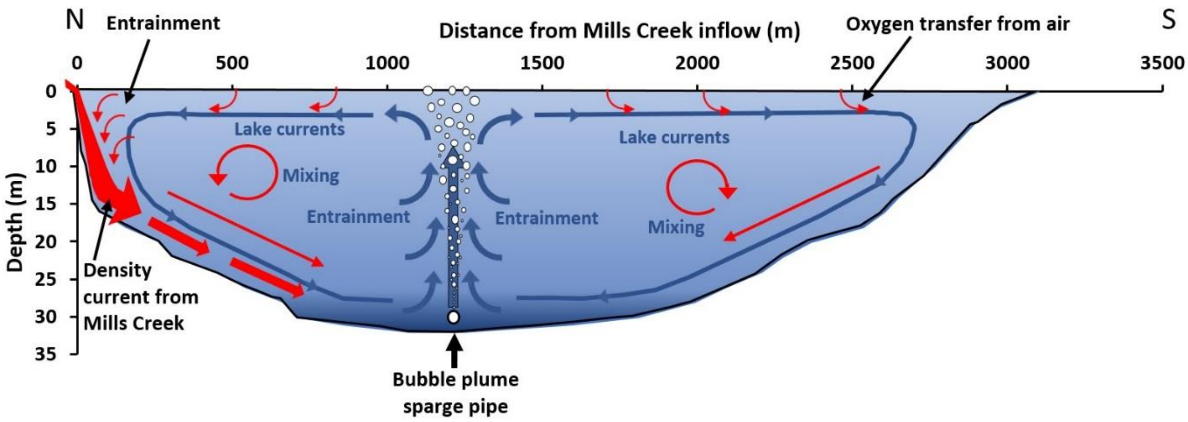


Figure 3-14: Schematic diagram of the likely circulation flow patterns in Lake Hayes with the bubble plume aerator operating. Red flow paths indicate oxygenated water. Blue lines are expected lake current flow paths. [Schematic drawn by Max Gibbs].

3.3.3 Compressor Cost estimate

Destratification of Lake Hayes is feasible and has the potential to prevent the annual cycle of bottom water anoxia and P release from the sediment as an internal load thereby suppressing the development of summer algal blooms. With an area 2.76 km^2 and a depth of 33 m, Lake Hayes is well within the capability of a single stage aeration system using a bubble plume aerator. An example of a large reservoir (72 m deep, surface area 6.22 km^2 and capacity 139 million m^3) that has successfully been mixed using aeration is El Capitan, a water supply reservoir for San Diego city in California (Fast 1968).

The bubble plume aeration for Lake Hayes (33 m deep = 3.3 atmospheres at 14 psi per at atmosphere) would require an air compressor with a minimum pressure of 3.3 x 14 psi = 46 psi or 315 kpa). With a surface lake surface area of 2.8 km^2, the compressor would need a minimum air flow of 2.8 x 9.2 = 25.8 m^3 min^{-1} or 910 cfm. For continuous operation, these specifications can be met using an 11kW screw compressor. Servicing would be once per year in winter when the aerator would be off. Estimated capital cost of the compressor is about $12K plus running costs.

The cost of the sparge line diffuser system would depend on the length of the sparge pipe (about 500 m) and diameter of the various pipes used (sparge pipe at 32 mm ID and floatation tube at 40 mm ID plus the compressed air line ~200 m), the heavy rope ballast weight (800 m) and the size of the anchor blocks required to secure the system in the lake. The system also requires a manifold to connect the compressor to the air line and connect the other pipes together.

As an example of likely costs, a similar size system installed in Lake Opuha near Timaru is estimated to have cost about $250 ± 50k installed. The compressor used is 75 kW in size, delivering 400 cfm at 80 psi. Costs associated with electricity supply are extra.

A more accurate price estimate of capital equipment, as well as running and maintenance costs for a system for Lake Hayes could be made once the design has been finalised.

3.3.4 Operational considerations

The **timing for starting the aeration system is critical**:

- It takes less energy to mix the lake when the temperature gradient across the thermocline is small, i.e., commencement of aeration should not be delayed until the thermocline is well-established.
- If it is delayed until thermal stratification has become established and the hypolimnion has become anoxic, the rising plume will entrain nutrient rich bottom water into the epilimnion and stimulate a phytoplankton bloom.

The turn on time must be determined for each year from in-lake measurements rather than by calendar date. Based on data from two periods in the temperature – DO records from Lake Hayes (Figure 3-15) an aeration system, if installed, should have been turned on in September and turned off in May the following year. Ideally, the turn on should be earlier than later and the aerator should always be turned on if the DO concentration falls below 7 g m^{-3}. Turn off should be at the end of April or early May but is not as critical.

The caveat to this turn on regime is that, if the optimum start time is missed, a late turn on may still be possible if the water column nutrient concentrations remain low and there is no algal proliferation in the lake. The risk is greatly increased that any nutrients accumulating in the hypolimnion will be mixed up into the water column and will stimulate an algal bloom. In that situation, it may be prudent not to use the aeration system for that year.

For example, a late turn on resulted in a major algal bloom in Lake Waikopiro, Hawke's Bay in summer 2017-18. When the bloom suddenly collapsed, the oxygen demand from the decomposing algal cells caused the entire lake to become hypoxic (<2 g m^{-3} DO) and resulted in an extensive fish kill.

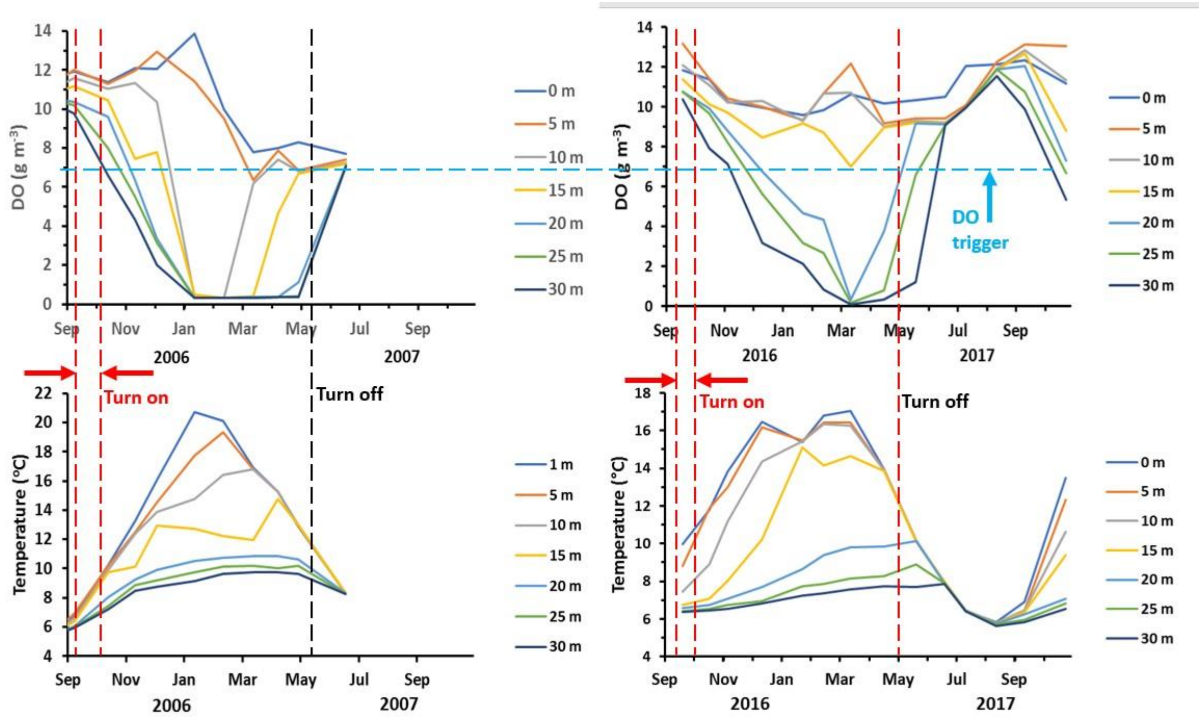


Figure 3-15: Examples from 2006-07 and 2016-17 of how to determine the time for turning the aeration system on and off. The two red vertical dashed lines between horizontal red arrows indicate the range tolerance for turn on based on the degree of thermal stratification and the level of oxygen depletion. The single red vertical dashed line indicates when it is safe to turn the system off. The horizontal blue dashed line is the trigger level for dissolved oxygen. If the DO concentration falls below this line the aeration system must be turned on as soon as possible.

3.3.5 Conclusions

Destratification will mix the lake through its full depth, providing several lake water quality benefits:

- An aeration system using a sparge diffuser across the narrowest part of Lake Hayes will cause the lake to become de-stratified in summer and will improve the water quality of the lake.
- Development of bottom water anoxia will be eliminated, minimising release of DRP from the sediments.
- Ammonium released from the sediments will be nitrified into nitrate, some of which may be denitrified into nitrogen gas and lost to the atmosphere.
- The reduction in DRP concentration without a substantial reduction in DIN produces a high N:P ratio which does not favour the growth of cyanobacteria or *Ceratium*

hirundinella, so these species are likely to become a minor component of the phytoplankton assemblage in the lake.

- The supply of oxygen to the sediment-water interface will facilitate decomposition of organic carbon in the sediment and water column, converting it to carbon dioxide.
- Organic carbon removed as carbon dioxide will reduce the sediment oxygen demand, allowing the lake to begin the recovery process.
- The release of carbon dioxide will generally reduce the lake pH, minimising the possibility of toxic ammonia formation.
- The full depth mixing and associated turbulence will not favour the development of cyanobacteria or *Ceratium hirundinella*, which require calm conditions, and they will be carried below their critical depth where they will die due to light limitation. However, other phytoplankton species that require turbulence to suspend them in the water column, such as diatoms and chlorophytes, are likely to become dominant and the lake algal assemblage will change from toxic blue greens and nuisance dinoflagellate species to non-toxic greens and diatoms. These species have critical depth limitations associated with light limitation so their abundance will also decrease due to full lake depth mixing. Consequently, phytoplankton biomass will decrease generally, leading to other lake water quality improvements:
  - The water column will clear (have lower mass of suspended particulate material), and the critical depth will increase because light can penetrate deeper into the lake.
  - The depth of the euphotic zone will also increase and the native aquatic macrophytes will increase in their depth range, providing safe refugia for small fish (bullies) and zooplankton from predatory trout and perch.
  - Higher oxygen levels at increasing depths should also benefit filter-feeding freshwater benthic organisms (e.g., freshwater mussels) as well as fish, which will have an increased habitat range encompassing the full depth of the lake.

The main disadvantage of destratifying the lake is that the lake temperature will rise. In Lake Hayes the temperature of the bottom water will increase, although it is unlikely to reach the mean temperature of 17-18°C at 2 m depth in summer, and the surface temperature is likely to fall as solar heating is dispersed through a greater volume of water and mixes with the cooler bottom water. This degree of lake warming is well below temperature that would be harmful to fish (maximum for trout is ~25°C) and therefore should not cause a problem.

The aeration could be run in conjunction with the augmentation of Mill Creek inflow providing a more efficient use of the aeration system in cooler years.

Aeration is not normally a long-term solution when restoring lakes and the destratification system would need to be run each year to manage the internal P load and the proliferation of algae. However, over a period of 5 to 10 years, with reduced inputs of organic matter and particulate P from the catchment, it is likely that sediment oxygen demand in the lake will reduce to the point where it is insufficient to cause hypolimnetic anoxia if the destratification system was turned off. Once this level of recovery has been reached, it would be a negative feed-back loop where the algal biomass in the lake reduces each year causing a lower sediment oxygen demand, which does not cause hypolimnetic anoxia and therefore does not release P to stimulate algal growth.

3.3.6 Recommendations

If an aeration system is to be considered for Lake Hayes, it should be a bubble curtain type using a sparge line diffuser across the bottom of the lake at the location and orientation indicated in Figure 3-13.

Dissolved oxygen and temperature profiles should be measured in the lake at weekly intervals from aeration turn on to March and thereafter at two weekly intervals until turn off, either manually of from a monitoring buoy. This will enable adaptive management of the amount of aeration required as the lake mixes and to maintain that mixing.

3.3.7 Secondary or amenity effects

Because the air compressor will be running continuously (24/7), there will be a potential for noise that will need to be blocked by the design of the compressor shed and the mounting block for the compressor. The bubble curtain produced by the aeration system will disturb the lake surface immediately above the sparge line. This will be very local to the aeration system and is not a health and safety issue for recreational users of the lake. However, it may disrupt the perfect reflections from the lake surface in the morning and evening under calm conditions. To accommodate tourists and photographers in general, the bubble curtain could be switched off for an hour at those times on calm days. Operated as the designed and with correct timing of turn on each year, the aeration system should have no adverse effect on water clarity and is likely to improve the clarity by preventing the development of algal proliferations.

3.4 Sediment capping

P removal using sediment capping, attacks the problem of the internal P load from the sediment driving the algal blooms, by sequestering the P into a non-bioavailable form. A description of the internal P load cycle is provided in the literature (James 2016; Spears et al. 2007) along with management strategies, which are discussed in detail in the lake sediment phosphorus release management—decision support and risk assessment framework (Hickey and Gibbs 2009).

The internal P load cycle is regulated by the dissolved oxygen concentration in the water. As the oxygen concentration reduces, the reduction-oxidation (REDOX) potentials of iron (Fe) and manganese (Mn) minerals in the sediment fall and, when the REDOX potential reaches a value around zero, these minerals transform from an insoluble oxidised ferric (Fe^{3+})/manganic (Mn^{3+}) form to a soluble reduced ferrous (Fe^{2+})/manganous (Mn^{2+}) form. Under well oxygenated conditions, ferric and manganic oxides sequester DRP from the water column and bind it as non-bioavailable P. Under anoxic conditions, the ferrous and manganous ions are in solution and there is no solid mineral matrix to hold the DRP, which is freely available for plant growth in the water column. The solubility of the metals changes with the REDOX potential and the sequestration/release process is reversible.

Another factor affecting the P binding to the iron oxides is pH. Under high pH (>9.2) the iron oxides form oxy-hydroxides which are soluble and the DRP bound to the insoluble iron oxides is released into the water column, even in well oxygenated water. High pH values are produced during photosynthesis by bicarbonate-adapted plants such as exotic aquatic macrophytes (pond weeds) and cyanobacteria. The ability of cyanobacteria to raise the pH and release DRP allows it to ‘mine’ P from the inshore sediments where it can grow to bloom proportions (Seitzinger 1991; Gao et al. 2012) when the epilimnion is otherwise depleted in P. It is not known whether *Ceratium hirundinella* can do the same.

Because the iron and manganese processes that bind DRP are reversible, these two metals are unsuitable as sediment capping agents. In contrast, aluminium (Al) compounds can irreversibly bind DRP from aerobic waters at normal pH ranges, i.e., the bound P is not released when the water column becomes anoxic and, if applied to anoxic water, it can sequester all of the DRP from that water. The rare earth metal, lanthanum (La), has similar P-binding characteristics as aluminium. Calcium can also sequester DRP in some conditions (high pH), but generally has a low affinity for P. All of these metals form the basis of commercially available sediment capping agents but behave differently in waters of different pH (Figure 3-16).

The P-binding efficacy of Al, La and calcium are strongly influenced by pH with the operating pH range for each metal increasing from Al < La < Ca (Figure 3-16).

3.4.1 Alum

The most common sediment capping agent is aluminium sulphate, which is supplied as the octadecahydrate ($Al_2(SO_4)_3.18H_2O$) in a 47% solution that has a pH of 3. The pH curve (Figure 3-16, Al) shows that alum is most efficient at binding DRP in the pH range of 4 to 6.5. Above that the P binding efficacy decreases, becoming <15% at pH 9. The major concern with using alum is that, below pH 5, it is in the trivalent ionic form Al^{3+}, which is highly toxic to aquatic biota. If there is insufficient DRP to bind to the aluminium applied, there will be free Al^{3+} ions in the water. To overcome this problem in soft water lakes, alum is applied with a buffer.

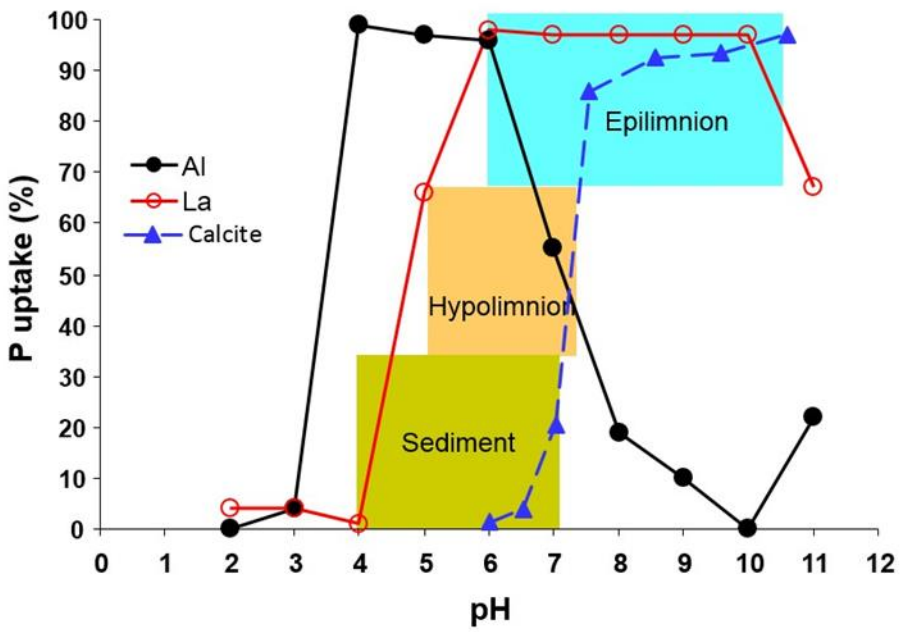


Figure 3-16: pH effects on DRP precipitation with Al and La salts, and adsorption by calcite. (Al and La data redrawn from Peterson et al. 1976, calcite data extracted from Olila & Reddy 1995). Coloured blocks indicate the typical pH ranges found in different parts of a lake. (Redrawn from Gibbs and Hickey 2012, Gibbs and Hickey 2017)).

Because it is a solution, alum can be sprayed on the lake surface or injected at the required depth into the lake water column. It can also be added to an inflow stream which could carry the alum floc in the density current formed when the stream enters the lake.

While the highest P-binding efficiency for alum is around pH 4 (Figure 3-16), the alum floc forms best at around pH 6.5 to 7. At lower pH the floc may not form leaving toxic trivalent Al^{3+} ions in the water if there is insufficient DRP to bind with. In soft water lakes the alum solution may require buffering to around pH 6.5 with sodium aluminate or sodium bicarbonate, to allow the alum floc to form.

Soft water lakes generally have low alkalinity of around 20 gEq (gram equivalents measured as g $CaCO_3$ m^{-3}). The nominal alkalinity value above which little or no additional buffer is required is 80 gEq. Lake Hayes has an alkalinity of around 150 gEq (Reid et al. 1999) and an average pH of around 7—7.5, and therefore should not require additional buffering over the natural buffering capacity of the lake water. When tested in Lake Hayes in 2010, the floc formed rapidly (Figure 3-17) and slowly settled though the water column over a period of about a day. The alum floc adsorbs DRP and aggregates particulate material, including zooplankton and algal cells, as it settles to the lake bed.

Figure 3-17: Example of alum floc formation in Lake Hayes in a trial mesocosm. [Photo: Ciska Overbeek, 22/02/2010].

3.4.2 Lanthanum

The use of La in the sediment capping material works differently to alum. This formulation, marketed as Phoslock®, requires an inert carrier for the La salt, Lanthanum chloride. To achieve this, during manufacture, it is physically blended with a fine bentonite clay, which is subsequently dried into fine granules. This granular material is typically applied as a surface treatment (Spears et al. 2013) that disperses through the water leaving a suspension of clay particles in the water column for up to 20 days, as the fine clay particles are slow to sink.

The highest P-binding efficiency for lanthanum is between pH 6 and pH 10 (Figure 3-15). Below and above this pH range, the P-binding efficiency is greatly reduced. Lanthanum binds rapidly with DRP in the water to form rhabdophane, a hydrous phosphate of La ($La(PO_4).H_2O$), which is insoluble and non-toxic. However, if there is insufficient DRP in the water column to bind all of the La, the residual La can form toxic trivalent ions (La^{3+}) ions in the water column below pH 6. This is comparable with the trivalent ions formed by aluminium at pH below 5.

3.4.3 Calcite

Calcium carbonate (calcite, $CaCO_3$) or calcium hydroxide (lime, $Ca(OH)_2$) can be added to lakes as phosphorus precipitants (Dittrich et al. 1997; - 2011). Calcite sorbs DRP, especially when the pH exceeds 9.0, and results in significant P removal from the water column. Phosphate adsorbs at the calcite surface, or binds inside a crystal during the formation of $CaCO_3$ precipitate when calcium hydroxide is applied (Kleiner, 1988; House, 1990). Various calcite forms have also been reported for their potential use as active barriers for sediment capping to reduce phosphorus release from sediments (Hart et al. 2003).

Unlike alum and lanthanum, where P-binding occurs at circum-neutral to moderately high pH, calcium binding of P only occurs at high pH. At high pH, and concentrations of Ca^{2+} and soluble P, hydroxyapatite is formed.

$$10\ CaCO_3 + 6\ HPO_4^{2-} + 2\ H_2O = Ca_{10}(PO_4)_6(OH)_2 + 10\ HCO_3^-$$

Hydroxyapatite has its lowest solubility at pH >9.5 and binds phosphorus strongly at high pH (Cooke et al. 2005). The major drawback for using calcite is the amount required. As an active barrier, it needs to be applied in a layer about 5 cm thick whereas alum and lanthanum capping layers are typically <1-2 mm thick.

3.4.4 Other capping agents

Other products that could be used for P-sequestration if applied as a sediment cap include:

- **Aqual P™**, which is a modified zeolite used to carry alum or poly aluminium chloride (PAC) to the bed of a lake. Because the active ingredient is aluminium, the properties are similar to alum except that it doesn't release DRP under anoxic conditions (Gibbs et al. 2011). The main advantages are that it is in a granular form for lake treatment and the zeolite used in its manufacture adsorbs ammonium from the water column so the product reduces both P and N in the one treatment. The disadvantages are that it is more expensive than alum and it has little capacity as a flocculent so does not clear the water column to the same degree as alum, which is a flocculent.

  However, because it settles rapidly, it can be targeted to specific areas such as the hypolimnion with little drift into the epilimnion or littoral zones. It is designed to block the release of DRP from the sediment under anoxic conditions. For this reason, Aqual-P™ is applied at the end of the mixed period when all DRP has been sequestered into the sediment and before thermal stratification and reduced DO concentrations begin to release DRP into the water column.

  Where cyanobacteria may be a nuisance inshore, an application of Aqual-P™ in the littoral zone may prevent the bloom developing by blocking the high pH release from the cyanobacteria. [To start the pH driven release the cyanobacteria must grow using the natural diffusion of DRP across the sediment-water interface. As this increases it becomes a positive feedback loop with the pH rising thereby releasing more DRP and allowing more cyanobacteria growth (Seitzinger 1991)].

- **Allophane**, is a natural volcanic rock, rich in both iron and aluminium. It has similar P-binding characteristics as alum and doesn't release DRP under anoxic conditions (Gibbs et al. 2011). The major problem for the use of allophane is finding a source which is not already contaminated with P and has a useful amount of P binding capacity left.

Both of these capping agents work well when applied as a thin layer 2 mm to 5 mm thick of < 1 mm grainsize material. The disadvantage is the amount required to put a 5-mm layer across the whole lake. For Lake Hayes, with a surface area of 2.76 km^2, the required alum application would be equivalent to a 0.4 mm layer across the lake bed.

3.4.5 What is the best medium to use?

Alum is the most cost effective permitted sediment capping agent available in New Zealand. This is because it is: 1) easy to transport (in 220 litre barrels or tanker trucks); 2) readily available around New Zealand for drinking water supply treatment; and 3) easy to apply, either by surface spray (Figure 3-18), subsurface injection, or drip feed into streams. The other capping agents cost more by up to a factor of 5 and, as solid material, application costs are more than for the liquid.

The amount of alum required to treat Lake Hayes as a one-off dose was estimated in a 2012 report prepared by John Quinn, Max Gibbs and Chris Hickey (NIWA) and reproduced in the Schallenberg and Schallenberg (2017) report. In 2012, the DRP concentration in Lake Hayes hypolimnion was around 300 mg m^{-3} (M. Schallenberg, unpublished data) and it was estimated that it would require 535,343 kg of alum to bind all the P released from the lake sediments. Alum costs about $1000 per tonne, giving a capping agent cost of NZ$535,343.

Since 2012, the mean DRP concentration in the hypolimnion of Lake Hayes during the stratified period has reduced from 300 mg m^{-3} to about 50 mg m^{-3} in 2016 (ORC monitoring data). Without more detailed data, a best estimate of the amount of alum required in 2016 is from a pro rata scaling from the 2012 estimate which gives the total amount as about 90,000 kg at a cost of NZ$90,000. These numbers need to be checked using the latest nutrient data from Lake Hayes and the current price of alum from Ixom (formerly Orica Chemicals).

3.4.6 Application method

Alum has been applied to lakes overseas using sun-surface injectors. While this technique has been used to apply a slurry of Aqual-P™, it has not been used for alum. Alum has been applied by surface spraying from an air boat (Figure 3-18) (Paul et al. 2008).

Figure 3-18: An air boat was used to apply alum to Lake Okaro near Lake Rotorua.. [Photo by Max Gibbs].

Lake Okaro with a surface area of 0.31 km^2 is much smaller than Lake Hayes with a surface area of 2.76 km^2 and took two days to complete the alum application. On a pro rata basis, it is likely to take about 18 days to spray Lake Hayes. Because the alum is applied at the surface and must pass through the complex circulation currents in the lake water column, it is not certain where the alum would settle in the lake over the 18 day period. Using the estimated 5 cm s^{-1} flow rate (section 1.1), the alum would travel about 15.5 km from where it was applied, i.e., about 3 times around the lake.

In lake Rotorua, alum has been drip fed at a rate of about 1 g m^{-3} into two inflow streams, which have naturally high DRP concentrations, to reduce the external P-load on that lake. A similar approach could be used in Lake Hayes by drip feeding the alum into the Mill Creek inflow and allowing the density current and lake circulation flows (Figure 3-9) to disperse the alum into the hypolimnion. This would require a storage tank and a metering pump to be installed beside Mill Creek but eliminates the dosing of the lake surface where the alum could adversely affect zooplankton populations.

The advantage of this technique is that the dose used can be adaptively managed by adjusting the drip rate up or down for optimum DRP sequestration as the DRP concentration changes in response to DO concentration changes in the hypolimnion. The adaptive management process could be automated by linking it with the temperature data from the lake and Mill Creek, so that alum dosing only occurred when the density current was plunging to the lower water column. This application process would minimise the amount of alum applied to the lake and would allow the dosing cost to be spread over time rather than have a single lump-sum cost.

This method of application will still cap the sediment and the capping effect will last as long as it takes for fresh sediment inputs from the catchment to bury the alum layer in the lake sediments. This may range from 5 to 10 years, given that the DRP concentrations are naturally declining, implying a reduction in sediment from the catchment.

3.4.7 Summary

Sediment capping is a method of preventing the release of P from the lake sediments. The most cost-effective product is alum either as a liquid from Ixom (previously Orica Chemicals) or bound to a zeolite carrier as a fine granular product available as 'Aqual-P™' from Blue Pacific Minerals, Tokoroa.

- Applied as single dose of the granular product in winter, it forms a pre-emptive sediment cap that will intercept the DRP diffusing out of the sediments as the DO concentrations reduce during the onset of thermal stratification in spring.
- Applied as a single dose of the liquid product via a surface spray or subsurface injection at the end of summer when the lake is strongly thermally stratified and all of the DRP has been released into the hypolimnion but before the thermocline begins to sink, it will sequester all of the DRP and irreversibly bind the P, which will settle onto the sediment surface.
- Applied as a liquid via a drip feed into the Mill Creek inflow, with automated adaptive management, it will be transported to the hypolimnion in the natural density current only when it will be able to sequester the DRP, thereby reducing the amount of product applied and focusing the product where it is needed. This should reduce wastage and therefore cost.
- Alum can be applied to Lake Hayes water without the need for an additional buffer.
- Once the DRP is bound to the alum and has settled to the sediment surface, it cannot be released by the hypolimnion becoming anoxic or the sediment being anoxic when it is subsequently buried.
- The longevity of the treatment is determined by the amount of new P-enriched sediment carried into the lake from the catchment or sedimenting algal biomass that grew in the lake. This will bury the original alum-bound P, which will remain in the sediment as a thin layer of non-bioavailable P. This layer will not be affected by diagenesis but may continue to sequester DRP released by diagenetic processes deeper in the sediment.
- Whichever method dosing is used, alum treatment is compatible with the augmentation of Mill Creek with irrigation water from the Arrow River, which has low DRP concentrations.

3.4.8 Recommendations

An Otago Regional Council representative should visit Bay of Plenty Regional Council to talk about the dosing systems they have put in place and see the dosing stations.

If cyanobacteria are beginning to grow in the edge water from accumulation as wind-drifts, a targeted dose of either Aqual-P™ or Phoslock® granules in the littoral zone is likely to prevent these wind-drift accumulations becoming substantial blooms by blocking their pH driving mechanism for the release of bioavailable P and causing local P-limitation.

3.4.9 Secondary or amenity effects

If alum is used for P removal and sediment capping and is applied to Mill Creek by drip feed, there will need to be a stream-bank installation similar to those used by Bay of Plenty Regional Council on the Utahina and Puarenga streams at Rotorua or on Soda Spring stream at Lake Rotoehu. The latter installation was housed in a small shed, behind a wooden fence and has minimal visual impact. The periodic delivery of bulk alum to the storage tank would be required.

3.5 Other mitigation techniques

There are several other mitigation strategies that could improve Lake Hayes water quality.

3.5.1 Nanobubble technology

This very recent technology which is said to take the hypolimnetic oxygenation process using Speece cones (Speece et al. 1973) to the next level. At present there have been a number of product documents and videos presentations but no peer reviewed publications, which give the information required to assess the efficacy of this product. The product brochures say the nanobubbles are produced through the walls of a special ceramic cone over which water flows. This water carries the nanobubbles into the lake where they sink the bottom. This concept may work in rivers and shallow lakes but may not work where the internal lake currents can be substantial, as in Lake Hayes. If it did work as described and the nanobubbles were entrained into the density current, this could be a useful tool.

The main issue with the nanobubble technology is that it relies on providing all of the oxygen to sequester the DRP in the hypolimnion. Based on the calculations of HOD in December 2016 (average 60 mg m^{-3} d^{-1}) it would need to provide >1.57 t O_2 d^{-1} (Section 3.2) just to hold the DO concentration at the level it was when the nanobubble technology was switched on. That oxygen input would need to be much larger if the nanobubble technology was going to re-oxygenate the lake. The other issue is that the special ceramic cone used to generate the nanobubbles is produced in only one facility in Japan. I believe that no one else has been able to reproduce this cone, without which the nanobubble machine does not work. This is a high risk situation if this vital part fails.

The supply agents advise that, if the power fails, water can slowly seep into the ceramic cone causing a blockage and the nanobubble machine will not work when the power is restored. This situation can be recovered by removing the cone from the water and allowing it dry for 24-48 hours.

A quotation for using the nanobubble technology in Lake Hayes was obtained (Appendix A). The design supplied included seven nanobubble 'engines' spaced around the lake at an installation cost of $4.7 million.

3.5.2 Hypolimnetic withdrawal

Hypolimnetic withdrawal is an in-lake restoration technique based on the selective discharge of bottom water to enhance the removal of nutrients and dissolved metals that accumulate when the hypolimnion becomes anoxic. Comparison of water quality variables before and during treatment in about 40 European and 8 North American lakes indicates that hypolimnetic withdrawal is an efficient restoration technique in stratified lakes (Nürnberg 2007). This technique was successfully used in New Zealand in the Upper Huia Reservoir for Watercare Services Ltd before aeration systems were installed (Spigel and Ogilvie 1985; Gibbs and Howard-Williams 2017).

Nürnberg (2007) reported that water quality improvement was apparent in decreased summer average epilimnetic P and chlorophyll-*a* concentrations, increased water clarity (Secchi disk depth), decreased hypolimnetic P concentration and anoxia. In particular, summer average P decreases were significantly correlated with annual water volumes and P masses withdrawn per lake area, indicating the importance of hydrology and timing of the treatment. Lake size is normally not a problem and hypolimnetic withdrawal has been used on lakes from 1.5 to. 1500 ha, spanning a 1000-fold area and a 2300-fold volume.

While almost all hypolimnetic withdrawal applications are conducted in lakes or reservoirs with outlets, the technique has also been applied to a relatively large seepage lake with no outlet by using a siphon (Lathrop et al. 2005). The hypolimnetic siphon approach is passive and can be designed for almost any natural lake where there is no bottom water offtake valve as found in reservoirs. In such lakes, the hypolimnetic siphon would draw the outflow water from the bottom of the lake rather than the surface. The disadvantage of this is that the bottom water would be rich in DRP, dissolved metals (primarily iron and manganese) and potentially ecotoxic high ammonia and sulphide concentrations. To overcome this problem, the hypolimnetic water would need to pass through an oxygenation process, such as a gravel cascade to agitate the water, followed by a settling pond/wetland to remove the particulates.

The design of the hypolimnetic siphon is conceptually simple, comprising a weir through which the draw pipe from the bottom of lake passes and a downstream wet land or aeration station through which the water flows before reaching the Kawarau River.

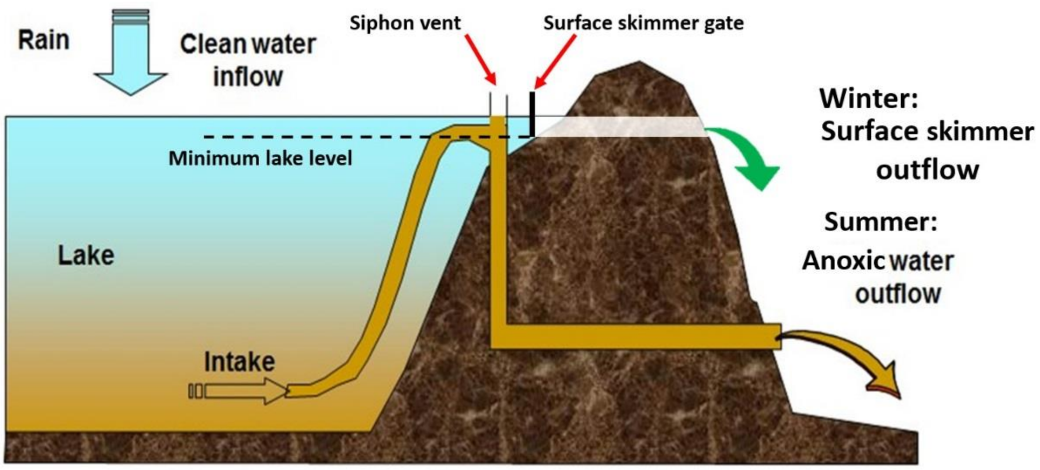


Figure 3-19: Stylized hypolimnetic siphon design through a weir, with a change-over gate to allow surface skimming if a surface algal bloom develops. (Schematic diagram drawn by Max Gibbs).

In summer, anoxic water drawn from the bottom of the lake flows out under the hydraulic pressure of the overlying lake water. The vent tube could be set at the height of the maximum required lake level so that the lake is always within a specified water level range. Note that in summer, the outflow is less than in winter. Because the lake naturally drains into the Kawarau River, the anoxic water would rapidly oxygenate and any DRP would be sequestered by the iron in that river sediment.

In winter when the lake is mixed, or during an algal bloom, the bottom water draw tube to the siphon would be closed and the skimmer gate opened to allow algal proliferations to be skimmed off the lake surface. This is the natural flow path and the siphon would be left in this configuration to accommodate flood events in winter. The surface skimmed material input to the Kawarau River would be unchanged from what it currently is as that is the natural connection from the lake.

Apart from the initial construction cost, this mitigation measure would have very low operational costs and would gradually mine the lake of DRP. If considered practical, a feasibility study would be required.

3.5.3 Biomanipulation

Biomanipulation is a technique which relies on a set of conditions prevailing to enable a specified end result. This approach is covered in detail in the Schallenberg and Schallenberg (2017) report.

Biomanipulation has been tried over many years (Shapiro and Wright 1984; Shapiro 1990) and successfully used in many shallow lakes in Europe and the USA (e.g., Carpenter et al. 1987; Dawidowicz 1990; Irvine et al. 1990; Jeppesen et al. 1990; Moss 1990; Timms and Moss 1984). In New Zealand, Howard-Williams (1981) found that the native aquatic macrophyte, *Potamogeton pectinatus*, was able to remove dissolved nitrogen and phosphorus compounds from lake water.

Biomanipulation can be a successful alternative to physical and chemical treatments to accelerate lake recovery often delayed because of persistently high rates of internal loading. Therefore, a combination of load reduction and in-lake restoration measures including biomanipulation is likely to improve water quality greatly (Kasprzak et al. 2002).

However, while there are success stories, there are also occasions when this technique does not work and cladocera cannot control algal blooms (Gliwicz 1990). Biomanipulation tends to work well in shallow lakes since organisms such as zooplankton are not spatially separated by depth. In deeper lakes, anoxia in the hypolimnion reduces the zooplankton population and the algal cells can fall out of the habitable water column resulting in further reductions in their population due to food limitation.

No compelling examples could be found of successful restoration of lakes as deep as Lake Hayes but there are numerous successful restorations of small shallow lakes and ponds.

3.5.4 Catchment management

Catchment management is undoubtedly the ultimate solution to the long-term restoration of Lake Hayes. With the development of appropriate farm management plans and restrictions placed on the application of superphosphate fertiliser in the catchment, the external nutrient loads, both N and P, on the lake will reduce to the point where the lake no longer develops an anoxic hypolimnion and the internal P load is reduced to near zero.

Fine soil has high concentrations of P and therefore, surface runoff will carry the P from diffuse sources in different sub catchments (Caruso 2000b). This information can be used to target the sub catchments with the highest P output. Mitigation measures could include the installation of detention bunds and wetlands to trap the fine sediment before it gets into the surface waters of a stream.

Catchment management strategies will take time (many years) to become apparent but the long-term effects will be a stable recovery of the lake not reliant on a cold year for enhanced underflow (augmentation), continuous aeration each summer or the addition of a sediment capping agent to reduce the DRP from the internal load. The internal load will no longer be replenished from the catchment and will gradually become buried or mined from the lake by natural in-lake processes.

Notwithstanding the importance of long-term catchment management given current nutrient and sediment loads entering Lake Hayes from Mill Creek, it is appropriate to implement short-term strategies that will reduce the magnitude of the water quality problems while the catchment management strategies become established. These can be phased out when the catchment management strategies are established.

4 Summary

This report has reviewed the Schallenberg and Schallenberg (2017) report and found a section that requires further work in the form of calculating the likely effects of using the excess irrigation water from the Arrow River to augment the flow in the Mill Creek inflow to Lake Hayes. Otherwise the Schallenberg and Schallenberg (2017) report provides a comprehensive summary of the issues and develops a workable set of management strategies with timelines for implementation and expected responses, that should be considered further.

In this report three defined options for water quality improvement have been elaborated in detail making use of the most recent data from ORC and the previously unpublished data from NIWA and University of Otago one-off studies to aide interpretation. Other potential mitigation strategies have also been considered and together with the importance of long-term catchment management given current nutrient and sediment loads entering Lake Hayes from Mill Creek.

The following table lists some of the overview assessments of these mitigation measures:

Table 4-1: Assessment of mitigation strategies considered for Lake Hayes in this report. Based on assessments in Section 3 in that order.

Strategy	Pros/Cons	Issues	Risk	Compatibility	Cost
Augmentation (Section 3.2). (Medium to long-term strategy).	**Pro**: Can potentially transfer sufficient oxygen to hypolimnion to prevent deoxygenation when it works but fragile. **Con**: Dependent on climate patterns.	Relies on cold temperatures in spring (Mill Creek and Arrow River irrigation water) to enhance underflow.	A warm dry spring could result in negligible oxygen transfer and strong bottom water anoxia (with subsequent P-release from sediments).	Augmentation of a natural process. Will complement other remediation processes.	Unknown.
Destratification (Section 3.3). (Short-term strategy – run on annual basis).	**Pro**: Will eliminate the internal P load. Disrupts *Ceratium* growth pattern. International precedents as effective remediation approach. **Con**: Will raise the temperature of the bottom water. Need for regular or continuous monitoring for operational management. Will probably need to be maintained for 5–10 years	Timing of turn-on is critical – must be before thermal stratification develops in spring.	Turn on while there is stratification and hypolimnetic deoxygenation and algal proliferation in the lake could trigger a major algal bloom; Failure of the compressor or power supply.	Will enhance augmentation density currents; Not compatible with sediment capping.	\$250K to \$350K capital cost plus running cost.

Strategy	Pros/Cons	Issues	Risk	Compatibility	Cost
Sediment capping (Section 3.4). (Medium to long-term strategy as a single dose. Short-term strategy run each year, if run as a drip feed.).	**Pro**: Alum or Aqual-P can lock P in the sediments and reset the internal P load to zero. International precedence as effective remediation approach for P-limited lakes. **Con**: Can release toxic trivalent Al^{3+} if the pH is <5. Other products were not recommended **Lanthanum**; **Calcite.**	Addition of a non-natural product to a lake may be seen as being not culturally acceptable. This could prevent the use of this technology to restore the lake.	Little risk in Lake Hayes. Could be applied as a drip feed into Mill Creek to keep the alum additions to a minimum.	Compatible with augmentation; Not compatible with destratification.	$90K to $550K as single dose. longevity 5 to 10 years. Drip feed option costs lower but ongoing.
Nanobubbles (Section 3.5.1) (Short-term strategy – run on annual basis).	**Pro:** Theoretically should work well to oxygenate hypolimnion without de-stratifying lake. **Con**: No documentation to support claims of efficacy. No international precedence. Very high capital cost.	Relies on several proprietary devices to oxygenate the hypolimnion of the lake.	Only one supplier of key ceramic cone element.	Could be used with sediment capping Not compatible with destratification.	$4.7 million capital plus running costs.
Hypolimnetic siphon (Section 3.5.2). (Medium to long-term strategy).	**Pro:** Can be used to reduce internal P load and skim surface algae. **Con:** Need for oxygenation and settling of discharge. Possible downstream effects in Kawarau River. Not readily automated.	Anoxic bottom water will be discharged into Kawarau River.	Design must include human exclusion from the siphon vent.	Not compatible with destratification.	Unknown probably <$100K.
Biomanipulation (Section 3.5.3). (Medium to long-term strategy).	**Pro:** Attacks the *Ceratium* bloom. Many successful studies in shallow lakes. **Con:** No international precedents for use in deep lakes. Likely high risk and low	No effect on internal P load to stop the bloom continuing.	Requires removal of planktievorous fish by enhancing piscivorous fish populations.	Not compatible with sediment capping as a single dose.	Unknown.

Strategy	Pros/Cons	Issues	Risk	Compatibility	Cost
	effectiveness as single management strategy.				
Catchment management (Section 3.5.4). (Long-term management strategy).	**Pro:** Will work given time (>10 years) **Con**: May have high costs for farmers. Probably requires other remediation measures to manage internal lake loads in the short-term.	Slow to see results but when they are achieved it will be a permanent fix.	Cyclonic storms that could destroy the structure put in place to reduce sediment loads etc.	Compatible with all short-term measures.	Unknown – but will cost lots for some options, and less for others.

5 Recommendations

To achieve rapid improvement in the water quality of Lake Hayes requires both short-term and long-term mitigation strategies.

A critical evaluation and multi-criteria scoring evaluation (including weighting of criteria) should be undertaken on the range of potential remediation technologies/approaches which are suitable for Lake Hayes. This will assist in ranking of the suitability of the management options relative to the short- and long-term goals for remediation (e.g., oligotrophic or mesotrophic state) – together with other community asperations. It should be noted that the implementation of multiple approaches is likely to achieve the best management outcomes.

Additional monitoring information may be beneficial in contributing to the technical information to undertake this assessment. The following list is in the order of least intervention in the lake and includes:

- **Catchment management:** Develop and implement catchment strategies. These will take time to become effective, but they will succeed.
- **Augmentation:** Measure the temperatures in the irrigation water and Mill Creek at 15-minute intervals, continuously.
  - Install a lake monitoring buoy in the lake with telemetered output of temperature and DO concentrations at multiple depths for use with automated management systems.
  - Deploy a pair of ADCP current meters on the lake bed, on opposite sides of the lake, to assess both vertical and horizontal in-lake currents. These are important for determining how the lake mixes and where the Mill Creek water inserts in the lake when it forms a density current.
  - After checking the irrigation water and Mill Creek temperature differentials, implement the irrigation water augmentation of Mill Creek. This connection should be managed to only augment when the irrigation water is colder than Mill Creek.
- **Sediment capping** (using P-binding)**:** If permitted, implement drip feed dosing of Mill Creek with alum, using automated feedback from the DO and temperature sensors on the lake buoy and the Mill Creek temperature data for adaptive management of the metering system. Dosing should only be used when the DO data indicates the P is being released from the sediment.
- **Destratification:** If alum dosing is not permitted, install a destratification system across the middle of the lake using the recommended turn-on protocols.
- **General monitoring:** Maintain the current SOE and inflow stream monitoring program with full profiles at two-weekly intervals (summer- January, February, March and April, inclusive) and at monthly intervals (winter- May to December, inclusive) until a telemetered lake buoy can be installed to provide these data at higher frequency.

The monitoring data should be used to drive the development of a DYRESM-CAEDYM model of the lake, which can be used to test these and other restoration measures.

6 Considered opinion

In my opinion, the most effective short-term mitigation method in Lake Hayes would be destratification. This is a proven technique that can be designed to suit the lake and can be adaptively managed as the lake condition improves. It adds nothing to the lake except air to keep the lake mixed. Mixing will prevent cyanobacteria and possibly *Ceratium* blooms developing.

Destratification will not necessarily eliminate algal blooms immediately but it can over time. The initial response will be a shift in the algal species assemblage to non-toxic species and species which will be more susceptible to light limitation from deep mixing below critical depth[7]. At 30 m deep, Lake Hayes has sufficient depth for the critical depth to be effective. Critical depth would likely be around 15 m. If the average algal assemblage spends more time below the critical depth than above, the algal biomass in the lake will decrease and the load of organic carbon reaching the lake bed will decrease. This can become a negative feed-back loop for algal growth and, over time, the biomass produced each year will diminish as the amount of organic matter driving bottom water oxygen depletion reduces annually.

The other effect of destratification is the supply of oxygen to the bottom of the lake and the sediments. The entrained oxygen from the surface in the return flow water will reduce the sediment oxygen demand (SOD) and the decomposing carbon will be released as CO_2 gas. As the SOD decreases, the HOD rate will decrease and there should become a time when, without the destratifier operating, thermal stratification will not result in the development of an anoxic hypolimnion with the concomitant release of DRP from the sediment.

This latter condition is reliant on the catchment management strategies reducing the external carbon, nutrient and suspended solids loads to the lake.

7 Acknowledgements

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[7] 'Critical Depth' is defined as a hypothesized surface mixing depth at which phytoplankton growth is precisely matched by losses of phytoplankton biomass within this depth interval. If phytoplankton spend more time below the critical depth than above, the biomass will decrease.

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Appendix A Nanobubble installation quotation for Lake Hayes

PACIFIC DRIVELINE LTD
MARINE & INDUSTRIAL DRIVELINE SPECIALISTS

Quotation No: 241

PACIFIC DRIVELINE LIMITED
PO BOX 97869
MANUKAU CITY
MANUKAU 2241, AUCKLAND
Tel: (09) 262-3241
Fax: (09) 262-3240
Email: pacdrive@henleygroup.co.nz
GST: 79-834-009

Attention: Max Gibbs **Company #** P66 **Quote Date** 22.01.2018

To Account:
NIWA
LAKE HAYES

Delivery:
NIWA
LAKE HAYES

Description	Quote Qty	Unit Price	Line Total
NBG-150A2GR NANOBUBBLE GENERATOR SYSTEM: GRID SUPPLY *PDL Scope of supply: Turn key operation. Grid Power Supply.* *1 x Custom Camo 40ft container system* *2 x 150A nanobubble generators,* *1 x Centrifugal Pump System,* *1 x Oxygen Generator System,* *80m suction and delivery pipes, floats, weights, screens, solar light.*	5.00	$537,842.00	$2,961,760.00
NBG-150A2GE NANOBUBBLE GENERATOR SYSTEM: GEN SUPPLY *PDL Scope of supply: Turn key operation. Generator Power Supply.* *1 x Custom Camo 40ft container system* *2 x 150A nanobubble generators,* *1 x Centrifugal Pump System,* *1 x Oxygen Generator System,* *80m suction and delivery pipes, floats, weights, screens, solar light.*	2.00	$592,352.00	$1,184,704.00

WARRANTY / NOTES

Nanobubble Generator Systems come as a complete wired and plumbed integrated package, including; security alarms & cameras, timers, monitored 24/7, installation of container and associated piping on site.

Customer Scope of supply: Road access to agreed system location. Grid power to container as required. Fuel Supply for generator powered systems (2).

Delivery: 24 weeks from receipt of 50% deposit, balance due 30 days following install of each unit model.

Warranty: 18 months from install sign-off and system operation.

Periodic Servicing / Maintenance of system and associated equipment: Price TBA.

Sub Total	$4,146,464.00
G.S.T	$621,969.60
Total	$4,768,433.60

Terms and Conditions: Page 1 of 1
Bank - ANZ Albany, 010277 0047338 00
INDENTED ITEMS ARE NOT RETURNABLE
Customer is liable for collection costs for outstanding accounts.
All above subject to our terms and conditions, a copy of which is available on our website.

NAME:__________ **SIGNATURE:**__________ **DATE:**__________

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