GOLDER

REPORT

Cyanobacteria Blooms in Pegasus Lake

Assessment of potential management options

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Distribution List

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Executive Summary

Pegasus Lake is a 5.3 m deep, 10.1 ha artificial lake situated in Pegasus Town, northeast of Woodend, North Canterbury. The main inflow (>90 %) into the lake is groundwater with a surface outflow to the wetland on the eastern side of the site known as the Eastern Conservation Management Area (ECMA).

In February 2017, Todd Property Pegasus Town Limited (TPPTL) was issued with a non-compliance notice from the Canterbury Regional Council, with respect to Condition 8 (c) and (d) of Consent No: CRC135321, relating to cyanobacterial blooms and seasonal stratification.

Following a report evaluating a wide range of potential cyanobacterial bloom management options, a workshop was held in November 2017 and attended by TPPTL, Golder Associates (NZ) Limited (Golder), Waimakariri District Council and Canterbury Regional Council.

The purpose of the workshop was to facilitate discussion and interchange of ideas in respect of potential management strategies for Pegasus Lake. Several questions were raised during the workshop, while five potential cyanobacterial bloom management options were selected for further investigation.

Further water quality monitoring was undertaken over the 2017-2018 summer period to help address the questions raised, while further scientific evaluation of the potential bloom management options was undertaken to better understand the potential for these options to prevent blooms and stratification in Pegasus Lake.

Workshop questions and answers

1. *Once stratification becomes established is it persistent or intermittent?*

Dissolved oxygen (DO) loggers installed near the top and the bottom of the water column from December 2017 to April 2018 confirmed that stratification was persistent in Pegasus Lake from at least early summer. Nutrient samples collected during December indicated that stratification, and associated anoxia, had been established for some time prior to the installation of the DO loggers. These results are consistent with the results from monthly water quality monitoring that has been undertaken from July 2014 to present.

2. *Are the two SolarBee® mixers having any effect on the stratification?*

DO, temperature and pH measurements taken immediately alongside the two SolarBee® mixers and at increasing distances from the mixers showed that there was no measurable effect on stratification. Results indicated that the SolarBee® mixers did not provide any localised mixing of the hypolimnion.

3. *What is the hydraulic residence time of the lake?*

Based on the volumetric calculation of the lake basin and flow measurements made during the summer period, the hydraulic retention time in the lake was 204 days.

4. *What is the current nutrient status of the lake and are nutrients being released during anoxia?*

Due to excessive nutrients and high phytoplankton biomass, the lake is currently classified as being hypertrophic. Measured high concentrations of NH_4-N in the bottom waters confirmed that dissolved nutrients are being released, via microbial breakdown of organic matter, under anoxic conditions during stratification. The high organic nutrients and phytoplankton biomass coupled with seasonal anoxia and high NH_4-N concentrations in the bottom waters during anoxia, suggests that internal nutrient loading is high in Pegasus Lake.

Cyanobacterial bloom management options

The five potential cyanobacterial bloom management options are summarised in Table 1.

In summary, it is concluded that:

- No single option provides a solution to the prevent both persistent stratification and cyanobacterial blooms.
- No combination of options is guaranteed to prevent persistent stratification and / or cyanobacterial blooms.
- There is no evidence to suggest that stratification of the water column was the cause of cyanobacterial bloom development in Pegasus Lake.

Table 1: Summary of the five potential management options for Pegasus Lake.

Option	Overcome anoxia	Prevent blooms	Indicative Capital cost ($)	Indicative annual operational cost ($)
Active sediment capping	No*	Potential	$350,000	None but cap replaced every 4 years.
Algal turf scrubber	No*	Potential	$850,000	$30,000
Artificial mixing or aeration	Potential	No	$1,050,000	$20,000
Flushing	No	No	$5,000,000 +	$20,000
Adaptive Management Regime	No	No	Not estimated	Not estimated

Note: * = these options could reduce the biological mediated DO consumption thereby reducing the overall DO stratification, but they are unlikely to prevent DO stratification driven by the inflow groundwater.

Artificial mixing / aeration provides a potential solution to reduce both in-lake biological and groundwater inflow mediated DO stratification. While artificial mixing / aeration provides a potential, but not guaranteed, solution to achieving compliance with condition 8(c) there is no evidence to suggest this will prevent cyanobacterial blooms.

Several options (active sediment capping and algal turf scrubber) provide potential nutrient management solutions, which in turn, may reduce the amount of algal / cyanobacterial biomass in the lake and the potential for blooms to develop. Reducing algal / cyanobacterial biomass would lead to an overall reduction in oxygen consumption by algae / cyanobacteria and heterotrophic bacteria. However, any groundwater inflow mediated DO stratification would still persist.

While there isn't a single option, or combination of options, that can guarantee to overcome both cyanobacterial blooms and DO stratification in Pegasus Lake, several options could be combined in an attempt to provide more effective management of the lake, rather than achieving compliance with both conditions 8 (c) and (d).

The artificial mixing / aeration option could be combined with either flushing option (surface or full lake) or the Algal Turf Scrubber, but not active sediment capping, as a capped layer would need to remain undisturbed.

Assessing the level of effectiveness of the management options, with the exception of adaptive management, would require pilot scale trials carried out in Pegasus Lake. The scale of the trials may trigger a requirement for resource consent to undertake the trials. Assessing the likelihood of costs and time to obtain resource consents is not part of this report.

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1.0 INTRODUCTION

Pegasus Lake is a 5.3 m deep, 10.1 ha artificial lake situated in Pegasus Town, northeast of Woodend, North Canterbury. The main inflow (>90 %) into the lake is groundwater with a surface outflow to the wetland on the eastern side of the site known as the Eastern Conservation Management Area (ECMA). Todd Property Pegasus Town Limited (TPPTL) has resource consents that authorise the taking and using of groundwater, the taking and diversion of surface water, the damming of water, and the discharge of storm water.

In February 2017, TPPTL was issued with a non-compliance notice from the Canterbury Regional Council (CRC), with respect to Condition 8 (c) and (d) of Consent No: CRC135321.

It is noted that compliance with Conditions 8 (a) and (b) is accepted as being achieved. Condition 8 states:

Notwithstanding Condition (7) the quality of the water in the lake shall, in general, meet the following criteria:

a. It is suitable for the activities and uses for which the lake and its water are proposed in the Lake Management Plan to be used for; and
b. It is generally suitable for secondary contact recreation; and
c. It does not result in persistent seasonal stratification leading to oxygen depletion in the lake; and
d. It does not result in toxic or nuisance algal blooms.

In order to address CRC's concerns regarding compliance for Condition 8 (c) and (d), TPPTL commissioned a literature review on potential cyanobacterial bloom management strategies for Pegasus Lake (see Golder 2017). Following the release of the report a workshop was held in November 2017 and attended by TPPTL, Golder Associates (NZ) Limited (Golder), Waimakariri District Council and Environment Canterbury.

The purpose of the workshop was to facilitate discussion and interchange of ideas in respect of potential management strategies for Pegasus Lake as well as the applicability of each of the potential management strategies identified in Golder (2017) for bloom management in Pegasus Lake.

Several questions were raised during the workshop, including:

1. Once stratification becomes established is it persistent or intermittent?
2. Are the two SolarBee® mixers having any effect on the stratification?
3. What is the hydraulic residence time of the lake?
4. What is the current nutrient status of the lake and are nutrients being released during anoxia?

In addition to the questions raised, five potential cyanobacterial bloom management options were selected for further scientific evaluation from the list of 16 contained in Golder (2017). These options were:

- Active sediment capping;
- Algal turf scrubber;
- Artificial mixing or aeration;
- Flushing; and
- Adaptive Management Regime.

The purpose of this report[1] is to provide scientific information to address these questions raised during the workshop and to provide further scientific evaluation of the potential bloom management options for Pegasus Lake to help facilitate informed discussions. The report is not intended to provide final recommendations with respect to any management option for the lake.

[1] This report is subject to the limitations in Appendix A.

2.0 STRATIFICATION

Stratification is a phenomenon where the water column separates into three distinct thermal layers. Cooler, denser water settles to the bottom of the lake, forming the hypolimnion, whereas the warmer, less dense water floats on the surface, forming the epilimnion. A thin layer of water (called the thermocline) separates the epilimnion from the hypolimnion. The greater the temperature difference between the epilimnion and the hypolimnion, the stronger the resistance to wind-induced mixing of the layers. Typically, this is the result of the sun rapidly warming both the surface inflows and the surface water in the lake, creating density differences through the water column, leading to stratification. Thermal stratification can lead to chemical stratification, where changes in the water chemistry result from differing biological processes in the epilimnion and the hypolimnion.

One of the concerns raised during the workshop was the limited understanding on the dynamics of stratification in Pegasus Lake. In South Island shallow lakes (< 10 m depth), stratification typically occurs in mid to late summer, although Maitai Reservoir, Nelson, stratifies from October to April each year (Kelly 2015). In Pegasus Lake, stratification has been reported as occurring as early as October or November, and persisting until late February or March, as indicated by monthly water quality profiles measured down the water column (e.g. Golder 2018). The workshop participants agreed that installation of dissolved oxygen (DO) loggers in the lake would better clarify if stratification was permanent during the summer period or broke down and re-established between the monthly monitoring occasions. In mid-December 2017, a monitoring station was installed at the Mid-Lake site. The station comprised of one DO logger installed at 0.5 m below the water surface and one at 5.0 m (just above the bottom of the lake) on a floating buoy. The two loggers were set to record temperature and DO concentrations every 15 minutes and the loggers were downloaded and cleaned at least monthly until they were removed from the lake in mid-April 2018, following turnover of the lake water column.

The results from the DO loggers showed that the lake was strongly stratified at the time of installation on 19 December 2017, with stratification not breaking down until 6 April 2018 when the lake turned over, the water column remained almost completely mixed, up to the time when the loggers were removed on 19 April 2018 (Figure 1). Monthly DO water column profiles measured since July 2014 showed a distinctive pattern of mid to late spring establishment of strong stratification that persists until mid to late autumn (Figure 2). However, it should be noted that weak stratification events (as indicated by slight decreases in the DO concentration and/or temperature in the bottom waters) have also been recorded on all monthly monitoring occasions during the winter / early spring period (May - September) since July 2014. These results confirm that strong stratification is well established by early summer, which is considered to be earlier than other shallow Canterbury lakes. Nutrient sampling undertaken in December 2017 further supports early onset of lake stratification (see Section 4.2).

One plausible explanation for the early onset of both thermal and chemical (DO) stratification in Pegasus Lake is that over 90 % of the inflows are derived from groundwater. While the exact source of groundwater supplying Pegasus Lake is unknown, groundwater quality monitoring bores within the immediate vicinity of Pegasus township show that during the spring / early summer period, groundwater temperature ranged from 12.1 to 13.0 °C, while dissolved oxygen (DO) percentage saturation ranged from 12.2 to 55 % for the period 2007 - 2017 (source https://gis.ecan.govt.nz). Surface water temperatures in Pegasus Lake at this time of the year have ranged from 14 - 20 °C. As groundwater enters the lake, it would initially underflow into the lake water column where it will either become an interflow (that is, where the inflow reaches neutral buoyancy, detaches from the lake side and insets into the water column at some intermediate depth), or remain an underflow to the bottom of the lake. Whether the groundwater inserts into the water column as an underflow or an interflow is dependent on the temperature differences between the groundwater and lake water. As water column temperature increases, the likelihood of the groundwater becoming an underflow increases.

Once the lake thermally stratifies, oxygen replenishment of the bottom waters can only occur through slow diffusion of gas from the epilimnion. High bacterial respiration in the bottom waters consumes oxygen faster than it can be replenished resulting in strong DO stratification through the water column, leading to anoxia, over time. If the groundwater source to Pegasus Lake has low DO concentrations, this would further exacerbate DO stratification and hypolimnetic depletion when the groundwater underflows.

In addition to the potential role of groundwater in establishing stratification in Pegasus Lake, both the lake's size and depth determine its stratification potential. The mixing characteristics, or the potential to stratify, of Pegasus Lake can be approximated using only the morphometric data of the lake and the equation of Davies-Colley (1988):

$$\hat{E} = 7.69f^{0.463}$$

where $\hat{E}$ is the depth of the epilimnion (m) and f is the fetch (km) defined as the square root of the lake surface area. For Pegasus Lake, this equated to an epilimnion depth of 4.5 m, 0.8 m above the maximum depth of the lake. The thermal characteristics of a lake can be further classified using the ratio of $\hat{E}/z_m$ (where z_m is the maximum depth of the lake) according to the criteria of Patalas (1984). With a $\hat{E}/z_m$ of 0.8, Pegasus Lake would be classified as being *0.5< $\hat{E}/z_m$<1.0 '2-layer lake' (epilimnion and metalimnion) - lake may be turned over by strong wind events* (Patalas 1984). This suggests that Pegasus Lake, even without any groundwater influences, may be at least weakly stratified for much of the year. This confirms the modelled prediction that Pegasus Lake would at least weakly stratify below 4.5 m water depth for much of the year. However, it should be noted that the slight decreases in both the DO and temperature during weak stratification were unlikely to have any significant ecological effect, with DO remaining at, or above, 80 % saturation in the bottom waters, and temperature differences <2°C.

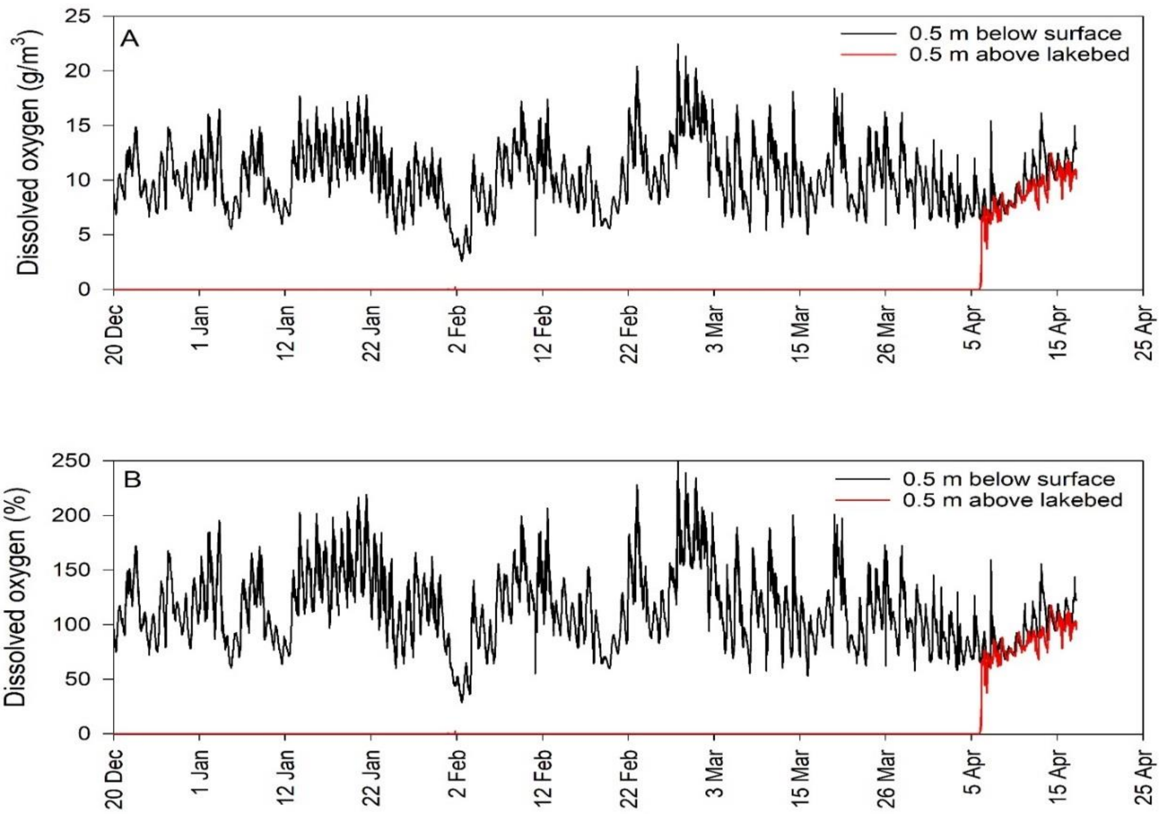

Figure 1: Dissolved oxygen in A) g/m^3 and B) percentage concentration, measured in Pegasus Lake from 19 December 2017 to 19 April 2018.

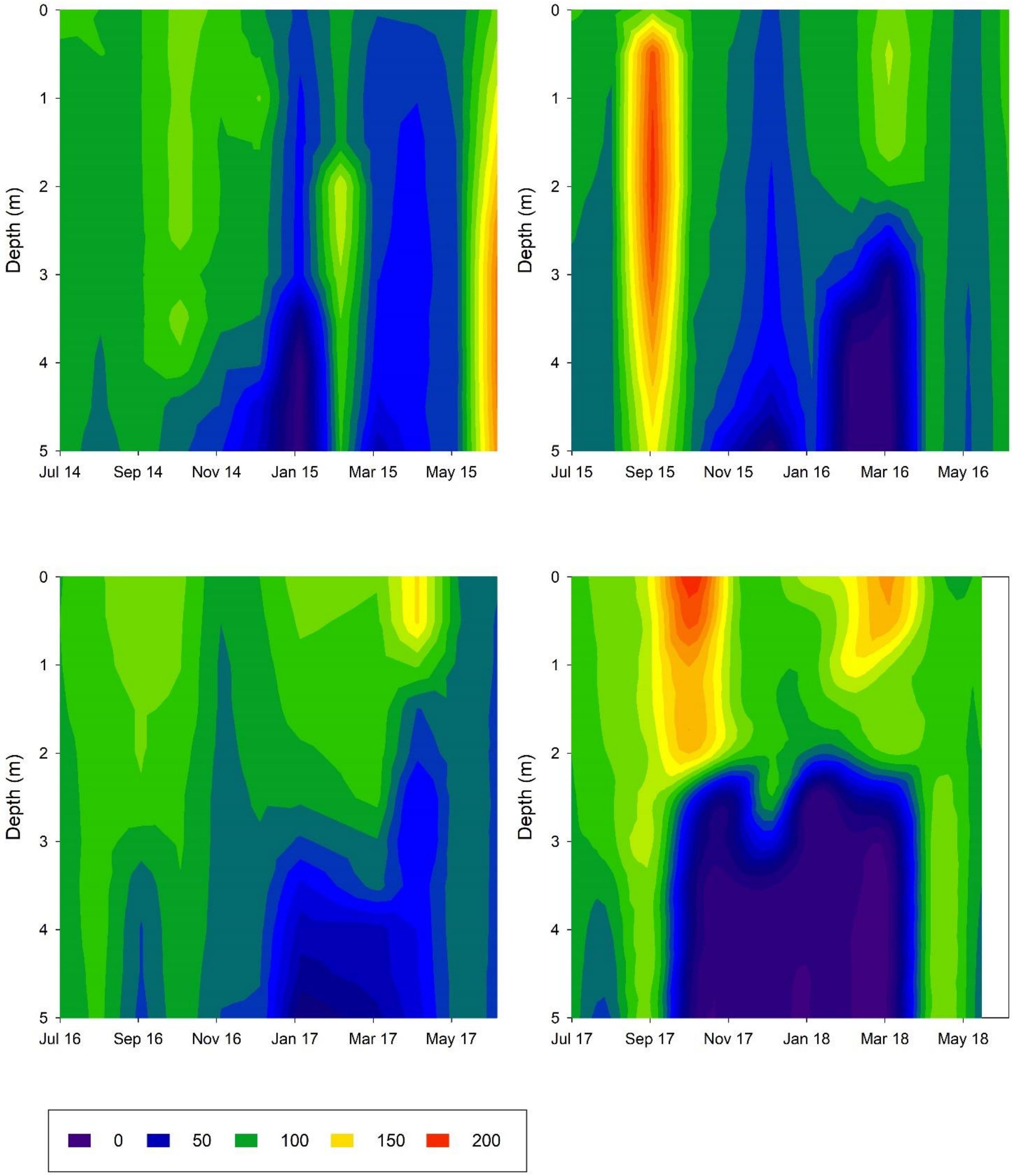

Figure 2: Monthly dissolved oxygen percentage saturation measured down the depth of a mid-lake water column profile in Pegasus Lake from July 2014 to May 2018.

2.1 SolarBee® Efficiency

Two SolarBee® mixers were installed in Pegasus Lake for the purpose of mixing the water column and preventing stratification (Figure 3). To date, the two SolarBee® mixers installed in Pegasus Lake have not been successful in preventing stratification across the whole lake. One of the concerns raised during the workshop was whether this was due to insufficient mixers across the whole lake, or if they were having any effect at all. Visser et al. (2016) noted that solar-powered water mixers, such as the SolarBee®, typically only mix the epilimnetic water, meaning that they have a limited zone of vertical mixing, do not mix into the hypolimnion, and, at times have been shown to strengthen stratification. Similarly, in a review of management strategies for cyanobacteria and algal blooms Lürling et al. (2016) noted that solar mixers in shallow systems were not successful in bloom control, with 50 % of trials using mixers resulting in enhanced blooms.

In order to determine both the vertical and horizontal mixing zone of the two SolarBee® mixers in Pegasus Lake, DO, temperature and pH profiles were conducted down the entire vertical extent of the water column from each mixer and at increasing horizontal distances from each during February 2018. The results showed that the mixers had no measurable effect on the DO, temperature or pH of the water column and did not provide any localised breakdown of stratification (Figure 4). Given that stratification was well established in the lake at the time of these measurements, it was not possible to ascertain whether the SolarBee® mixers were enhancing stratification, or not. Based on these results, it is unlikely that the installation of more SolarBee® mixers would help to fully mix the water column and prevent stratification in Pegasus Lake.

Figure 3: One of the SolarBee® mixers installed in Pegasus Lake.

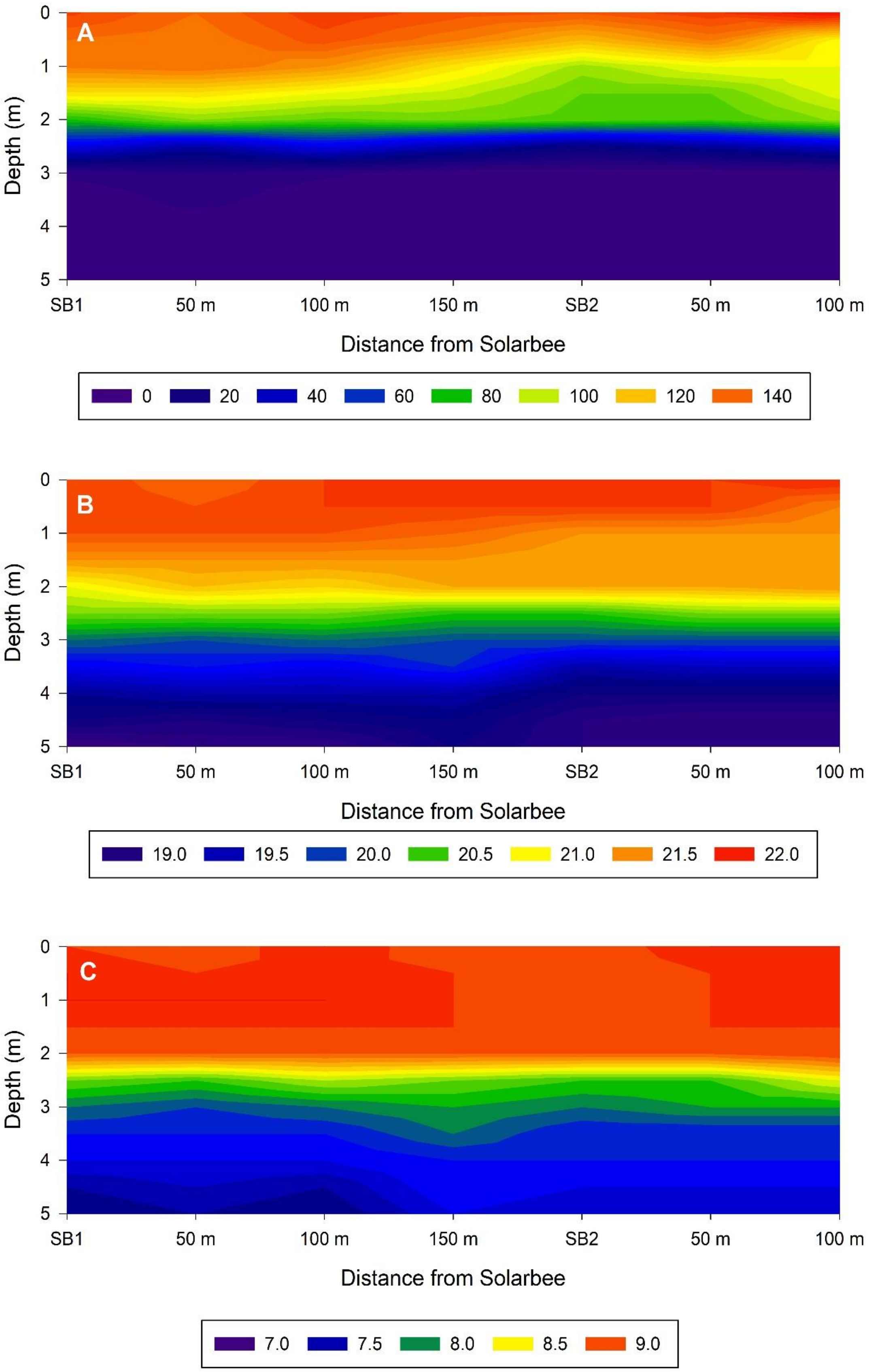

Figure 4: Water column profiles of A) dissolved oxygen percentage saturation, B) temperature (ºC) and C) pH at increasing distances from the two SolarBee® mixers (SB1 and SB2) in Pegasus Lake during February 2018.

3.0 LAKE VOLUME AND HYDRAULIC RESIDENCE TIME

The outflow rate was originally estimated to be 12 l/s, giving a reported hydraulic residence time (HRT), or how long a parcel of water remains in the lake, in Pegasus Lake of >1 year (Beca 2007). HRT plays a role in the flushing potential of both nutrients and phytoplankton from the lake, with long HRT resulting in retention of these both, which can enhance internal nutrient recycling. One of the questions raised during the workshop was what is the current HRT of the lake?

In order to calculate the HRT, the lake volume needed to be determined. Lake volume was calculated using the bathymetric data map supplied by Woods Consultants. The areas within the depth counters were calculated using imageJ software and the lake volumes below each of the depth counters calculated using the prismoidal formula:

$$V = \frac{h}{3}[A + (AA)^{0.5} + A]$$

where V = volume between two parallel areas (A) at a distance (h) apart (Green 1975). The depth-area-volume for each contour is presented in Table 2. The total volume of the lake was 463,038 m^3, based on an area of 10.14 ha.

Outflow from Pegasus Lake was calculated using the weir head - discharge relationship for v-notch weirs using the formula of Gouvernement du Québec (2008). The median flow over the two v-notch weirs during summer was 0.02 m^3/s^{-1}. Based on the total volume calculated above, and adjusting for precipitation and evaporation, this equates to a median HRT of 204 days. However, it should be noted that monitoring did not represent a full hydrological year and temporal variability in both groundwater and surface water inputs may affect estimated HRT.

Table 2: Depth-area-volume calculated for Pegasus Lake based on the prismoidal formula of Green (1975).

Depth (m)	Area of contour (m^2)	Volume between depths (m^3)	Cumulative volume (m^3)	% total volume
0	101,378	98,917	463,038	100
1	96,478	92,423	364,120	79
2	88,428	83,862	271,697	59
3	79,378	70,394	187,835	41
4	61,777	60,442	117,441	25
5	59,116	55,121	56,999	12
5.2	51,221	1,878	1,878	0.4
5.3	0	0	0	0

4.0 LAKE NUTRIENTS AND ALGAL BIOMASS

During the workshop one of the questions raised concerned the current nutrient status of the lake and if any nutrients are released during periods of anoxia in the bottom waters. Details of the nutrient sampling and results are presented below.

4.1 Nutrient Sampling

Nutrient sampling was undertaken in the lake on 18 December 2017, 13 February 2018 and 19 April 2018. On each occasion, water samples were collected from the following discrete depths, 0, 1, 2, 3, 4, and 5 m below the water surface, at the Mid-Lake Site, using a horizontal Van Dorn water sampler. At the same time, the environmental variables pH, conductivity, temperature and DO were recorded at 0.5 m increments down the water column. Nutrient samples were kept cool, in the dark and transported to the NIWA Hamilton Water Quality Laboratory where sub-samples were then filtered for subsequent analysis of nutrients, total suspended solids (TSS), organic matter and chlorophyll-*a* (see Section 4.3).

Nitrate-nitrogen (NO_3-N), ammoniacal-nitrogen (NH_4-N), dissolved reactive phosphorus (DRP), total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) were determined on a Technicon 2 flow injection analyser, after UV irradiations in respectively alkaline or acidic conditions (Downes 2001). Dissolved inorganic nitrogen was derived from NH_4-N + NO_3-N, dissolved organic nitrogen (DON) and phosphorus (DOP) were determined as TDN minus DIN and TDP minus DRP, respectively. Particulate nitrogen (PN), particulate phosphorus (PP), total nitrogen (TN) and total phosphorous (TP) were determined on the flow injection analyser following digestion.

4.2 Nutrient Results

During the December 2017 and February 2018 sampling occasions, the percentage of TN comprising of PN decreased with increasing depth, with over 60 % of TN comprised of PN in the surface waters, decreasing to <34 % at 5 m depth (Table 3). However, by the time of the April sampling, >65 % of TN was comprised of PN throughout the entire water column (Table 3). The percentage of TP comprised of PP was less variable between sampling occasions and with water depth, ranging from 61 - 95 %, with no obvious relationship with depth (Table 3). TN, TP and chlorophyll *a* concentrations (see below) were consistent with a trophic level index (TLI) of ***hypertrophic***, meaning that nutrients in the lake were supersaturating, phytoplankton growth was excessive and water quality poor.

On all sampling occasions, DIN was predominantly comprised of NH_4-N, regardless of the sampling depth (Table 3). During the December and February sampling, NH_4-N concentrations in the 5 m water sample was 539 and 149 g/m^3, respectively, while during April NH_4-N concentrations had declined to 2 g/m^3 (Table 3). High NH_4-N concentrations in the hypolimnion is consistent with bacterial breakdown of organic matter and release of nutrients under anoxic conditions. NH_4-N was not converted to NO_3-N as nitrifying bacteria are aerobic, meaning that they require oxygen to grow. DRP concentrations were higher in December compared to February and April but showed little variability with depth (Table 3).

DON and DOP are the nitrogen (N) and phosphorus (P) containing components of dissolved organic matter, respectively, and are part of the biologically reactive nutrient pool in a lake. During the stratified period, DON concentrations were highest in the surface water and lowest at 5 m depth, on both sampling occasions but were relatively uniform throughout the water column during the April sampling, when the lake was no longer stratified (Table 3). Low DON in the bottom water during stratification is consistent with the breakdown and conversion of DON to NH_4-N. DOP concentrations throughout the water column appeared to be unaffected by stratification (Table 3).

An alternative hypothesis is that released P is rapidly taken up by the phytoplankton cells. Many algal species are capable of luxury uptake of P, with increased uptake typically occurring under low N:P ratios (Liu & Vyverman 2015). N and P are essential macronutrients for microalgal growth and the ratio of these nutrients is considered important for optimal growth in most aquatic ecosystems (Sutherland et al. 2014). N:P ratios can be used to predict nutrient limitation status, although concentrations, especially of P, can be difficult to interpret as far as what is actually available to the microalgae (Dodds, 2003). For freshwater microalgae, N and P potentially co-limit production over a wide range of N:P ratios (~10 to 30), while ratios above 30 suggest P limitation and below 10 suggest N limitation (Dodds, 2003). On the three sampling occasions in Pegasus Lake TN:TP ratios ranged from 3 to 9, while DIN:DRP ratios ranged from 0.5 - 7 (excluding the 5 m depth during stratification) suggesting potential N limitation of algal growth in the lake. This is indicated by the rapid uptake of the NH_4-N into the particulate fraction following lake turnover in April.

While the PN and PP fractions were not analysed for their respective proportions of inorganic and organic N and P, given the high percentage of organic matter comprising the TSS (see Section 4.3 below), it is plausible to suggest that the majority of particulate nutrients were in the organic form and therefore available for nutrient recycling. High organic nutrients and phytoplankton biomass coupled with seasonal anoxia and the measured high NH_4-N concentrations in the bottom waters during anoxia, suggests that internal nutrient loading is high in Pegasus Lake. It was not possible to sample the diffuse groundwater inflows to determine inflow nutrient concentrations. However, based on the assumption that nutrient concentrations in the lake immediately following filling were indicative of the groundwater inflow nutrient concentration, and on the assumption that groundwater quality has not declined between 2010 and 2018, we can estimate the proportion of internal to external nutrient loading in Pegasus Lake. Based on median nutrient concentrations across all water depths for all three sampling occasions, and the 2010 nutrient concentrations as a proxy to inflows, internal nutrient loading contributed 70 % of the TN and 74 % of the TP to the lake.

Modelled volumetric concentrations of the dissolved, particulate and total nitrogen, as well as the chlorophyll-*a* concentration (see Sections 4.3 & 4.4), was used to track the fate of the NH_4-N from the hypolimnion when stratification broke down and the lake fully mixed (Table 4). Both the increase in PN coupled with the doubling of chlorophyll-*a* in April relative to December suggests that NH_4-N was taken up by the algae into its biomass, following full mixing of the lake in early April.

Table 3: Nitrogen and phosphorus species measured at discrete depths down the water column at the Mid-Lake site in Pegasus Lake on three separate occasions. All concentrations are in mg/m^3.

Water depth (m)	NH_4-N	NO_3-N	DIN	DON	TDN	PN	TN	DRP	DOP	TDP	PP	TP
18 December 2017												
0	4	<1	4.5	554.5	559	931	1490	5	29	34	187	221
1	2	<1	2.5	522.5	525	945	1470	4	30	34	193	227
2	7	<1	7.5	551.5	559	951	1510	5	32	37	188	225
3	3	<1	3.5	486.5	490	1080	1570	5	28	32	212	244
4	3	<1	3.5	449.5	453	452	905	6	25	31	133	164
5	539	<1	539.5	384	923	462	1385	3	19	22	439	461
13 February 2018												
0	27	<1	27.5	700.5	728	1172	1900	2	64	66	185	251
1	3	<1	3.5	554.5	558	842	1400	2	31	33	148	181
2	3	<1	3.5	492.5	496	174	670	2	31	33	52	85
3	4	<1	4.5	496.5	501	284	785	2	41	43	73	116
4	3	<1	3.5	434.5	438	179	617	2	29	31	66	97
5	149	<1	149.5	464.5	614	260	874	2	53	55	195	250
19 April 2018												
0	1	<1	1.5	396.5	398	1052	1450	2	22	24	142	166

Water depth (m)	NH_4-N	NO_3-N	DIN	DON	TDN	PN	TN	DRP	DOP	TDP	PP	TP
1	3	<1	3.0	416.0	419	1111	1530	1	20	21	148	169
2	2	<1	2.6	424.4	427	843	1270	1	21	22	121	143
3	<1	<1	1.0	411.0	412	968	1380	1	19	20	130	150
4	7	<1	7.6	414.4	422	838	1260	1	20	21	114	135
5	2	<1	3.0	427.0	430	830	1260	2	21	23	381	404

Note: NH_4-N = ammoniacal nitrogen, NO_3-N = nitrate nitrogen, DIN = dissolved inorganic nitrogen, DON = dissolved organic nitrogen, TDN = total dissolved nitrogen, PN = particulate nitrogen, TN = total nitrogen, DRP = dissolved reactive phosphorus, DOP = dissolved organic phosphorus, TDP = total dissolved phosphorus, PP = particulate phosphorus, TP = total phosphorus.

Table 4: Modelled volumetric concentration of nitrogen, phosphorus and chlorophyll-*a* in Pegasus Lake. Concentration in tonnes/lake.

Date	NH_4-N	DRP	PN	PP	TN	TP	Chl-a
18 December 2017	0.032	0.002	0.389	0.100	0.654	0.115	0.054
13 February 2018	0.012	0.001	0.254	0.057	0.516	0.077	0.043
19 April 2018	0.001	0.001	0.444	0.076	0.636	0.086	0.100

Note: NH_4-N = ammoniacal nitrogen, DRP = dissolved reactive phosphorus, PN = particulate nitrogen, PP = particulate phosphorus, TN = total nitrogen, TP = total phosphorus, Chl-*a* = chlorophyll *a*.

4.3 Phytoplankton Community Sampling and Analysis

In order to better understand the phytoplankton community within Pegasus Lake, samples were collected for biomass and community composition as described below.

For phytoplankton biomass, samples were collected as part of the nutrient sampling (See Section 4.1). TSS concentration was measured by filtering a known volume through a pre-weighed Whatman GF/F filter, dried then re-weighed to determine mass. Filters were then placed in a furnace at 450 °C for four hours, re-weighed and the volatile fraction (organic matter) calculated as the mass lost on ignition. Chlorophyll-*a*, a proxy for phytoplankton biomass, was extracted in 90 % acetone at 4°C, in the dark, for four hours. Samples were then centrifuged at 3000 rpm for 10 minutes and the absorbance of the supernatant read on a fluorometer before and after acidification.

For phytoplankton species determination, weekly surface water samples were collected from four sites around the lake and preserved in Lugols solution on site. These sites were Jetty 1, Beach 3, Beach 4 and Southern Outlet (see Golder 2018 for site location details). The environmental variables pH, DO, temperature and conductivity were recorded at each site using a calibrated handheld YSI meter. At each site, water colour was determined in the field using the Munsell colour standards (as described in Davies-Colley et al. 1993). Preserved phytoplankton samples were returned to Algal Services Laboratory, NIWA Hamilton, for taxonomic identification and species counts.

For phytoplankton community composition, data from all four sites was averaged, to avoid any bias in the sampling caused by wind-driven events across the lake. A cluster analysis with an *a posteriori* test for significance was undertaken using the SIMPROF procedure in the University of Plymouth's programme PRIMER (Clarke & Warwick, 2001). SIMPROF was considered appropriate for initially exploring the dataset to identify significant changes in the communities as there was no prior structuring of samples into groups and individual abundance was expressed in relative terms as a percentage of the total abundance for that species over a given sampling occasion (Clarke & Warwick, 2001). Dendrogram matrices were constructed to represent the statistically significant communities between sampling occasions.

LINKTREE procedures were used to explore whether significant differences in phytoplankton communities between sampling occasions could be explained by environmental variables. The LINKTREE procedure uses a constrained clustering technique to construct a hierarchical tree to identify the best subset of environmental variable thresholds that explain significant patterns based on assemblage data (Clarke & Warwick, 2001). The samples are first treated as one group and subdivided according to the variables that maximised the between cluster variance. The degree of separation is indicated by A %, the higher the A %, the higher the variance between the separations and significant (5 % level) thresholds of environmental variables are identified (Clarke & Warwick, 2001). The environmental variables used in the analysis were mean daily surface DO, temperature, irradiance and wind speed for the four days prior to sampling, as well as pH and conductivity. SIMPROF, Dendrogram plots, Bray Curtis similarity matrices and LINKTREE analyses were carried out in PRIMER, v. 7.

4.4 Phytoplankton Community Results

In the upper 4 m of the water column, on all sampling occasions, TSS were comprised predominantly of organic matter, varying between 65 and 94 % of the total mass. However, this proportion decreased to between 25 and 47 % at 5 m depth (Table 5). The autotrophic index is a measure of the ratio of organic matter to chlorophyll-*a*. The index is a useful indicator on the proportion, or influence, of heterotrophic organisms on the planktonic community. The higher the autotrophic index, the greater the proportion of heterotrophic to autotrophic organisms, with values exceeding 400 indicating a system heavily impacted

(Collins & Weber 1978). During stratification, the autotrophic index increased with increasing depth, with a maximum value of 582 measured at 5 m depth during the February sampling, almost six times the value measured at the surface (Table 5). This is consistent with the almost zero light and zero oxygen conditions in the bottom waters during this time of the year. At the time of the April sampling, following lake turnover, the autotrophic index at 5 m was lower than all other depths sampled, indicating a reduction in the concentration of heterotrophic organisms in the bottom waters (Table 5).

Over the January to early April period, the phytoplankton community was highly unstable, with rapid (<1 week) turnover of the dominant species on many occasions (Table 6 & Figure 5). LINKTREE analysis identified seven occasions where statistically significant changes in the microalgal community occurred (Figure 6). For four of the occasions, mean daily DO concentrations were identified as the main environmental driver of community change (Figure 6). However, it is likely that DO concentrations were actually a proxy for other processes occurring in the lake, including increased heterotrophic respiration caused by grazer, bacteria or virus blooms that may negatively affect the phytoplankton community. Zooplankton grazer blooms have been identified as a major contributor to rapid phytoplankton community turnover in a hypertrophic artificial pond (Sutherland et al. 2017). pH, conductivity and temperature were identified as significant ($p < 0.01$) environmental variables that explained the onset of the cyanobacterial bloom, while conductivity and temperature were the environmental variables that explained the remaining significant ($p < 0.01$) community changes (Figure 6).

High daytime pH is indicative of dissolved inorganic carbon (DIC) limitation in the lake. DIC limitation occurs when photosynthetic-mediated carbon removal greatly exceeds the rate of respiration-mediated replenishment in the water column. At high pH most DIC is in the form of HCO_3 or CO_3^{2-} and phytoplankton require metabolically expensive carbon concentrating mechanisms (CCM) to assist with carbon uptake (Sutherland et al. 2015). Cyanobacteria have been shown to have superior CCM systems and are able to outcompete other algae under these conditions (Low-Décarie et al. 2014).

It is plausible to suggest that the competitive advantage of cyanobacteria during a period of high pH and temperature was the main reason for bloom development in Pegasus Lake.

There is no evidence to suggest that stratification of the water column was the cause of cyanobacterial bloom development in Pegasus Lake.

Table 5: Total suspended solids, organic matter, chlorophyll-*a* and autotrophic index sampled at discrete depths in the water column at the Mid-Lake site in Pegasus Lake.

Water depth (m)	Total suspends solids (g/m^3)	Organic matter (g/m^3)	Chlorophyll-*a* (mg/m^3)	Autotrophic index
18 December 2017				
0	28.5	25.8	132	195
1	30.1	27.8	143	194
2	31.3	28.8	153	188
3	30.6	27.3	136	201
4	13.8	10.8	48	226
5	50.6	12.8	43	298
13 February 2018				
0	32.2	24.5	236	104
1	22.8	20.3	161	126
2	4.2	3.2	22.0	146
3	5.5	4.0	20.8	193
4	3.8	2.5	9.0	275
5	13.6	6.3	10.9	582
19 April 2018				
0	26.3	24.6	231	231
1	25.1	23.6	266	266
2	20.5	19.2	197	197
3	23.1	21.3	217	217
4	19.7	18.4	216	216
5	61.5	23.8	140	140

Table 6: Numerically dominant phytoplankton species in Pegasus Lake between the periods 8 January to 9 April 2018.

Date	Phyllum	Taxa	Cells per ml
8/01/2018	Desmids (Zygnemophyceae)	*Closterium aciculare*	9822 ± 1091
15/01/2018	Desmids (Zygnemophyceae)	*Closterium aciculare*	5559 ± 1521
23/01/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	4160 ± 3123
29/01/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	2821 ± 1676
7/02/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	2022 ± 1389
7/02/2018	Green algae (Chlorophyta)	*Closteriopsis* sp.	1851 ± 544
13/02/2018	Blue greens (Cyanobacteria)	*Dolichospermum circinalis*	48415 ± 51661
19/02/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	2745 ± 1500
26/02/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	2049 ± 1058
5/03/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	5508 ± 2326
12/03/2018	Blue greens (Cyanobacteria)	*Dolichospermum circinalis*	270065 ± 518960
19/03/2018	Blue greens (Cyanobacteria)	*Dolichospermum circinalis*	8103 ± 15676
26/03/2018	Blue greens (Cyanobacteria)	*Dolichospermum circinalis*	1800 ± 3245
26/03/2018	Flagellates/Unicells	Flagellates/Unicells	1706 ± 470
3/04/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	4353 ± 4602
9/04/2018	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*	1725 ± 543

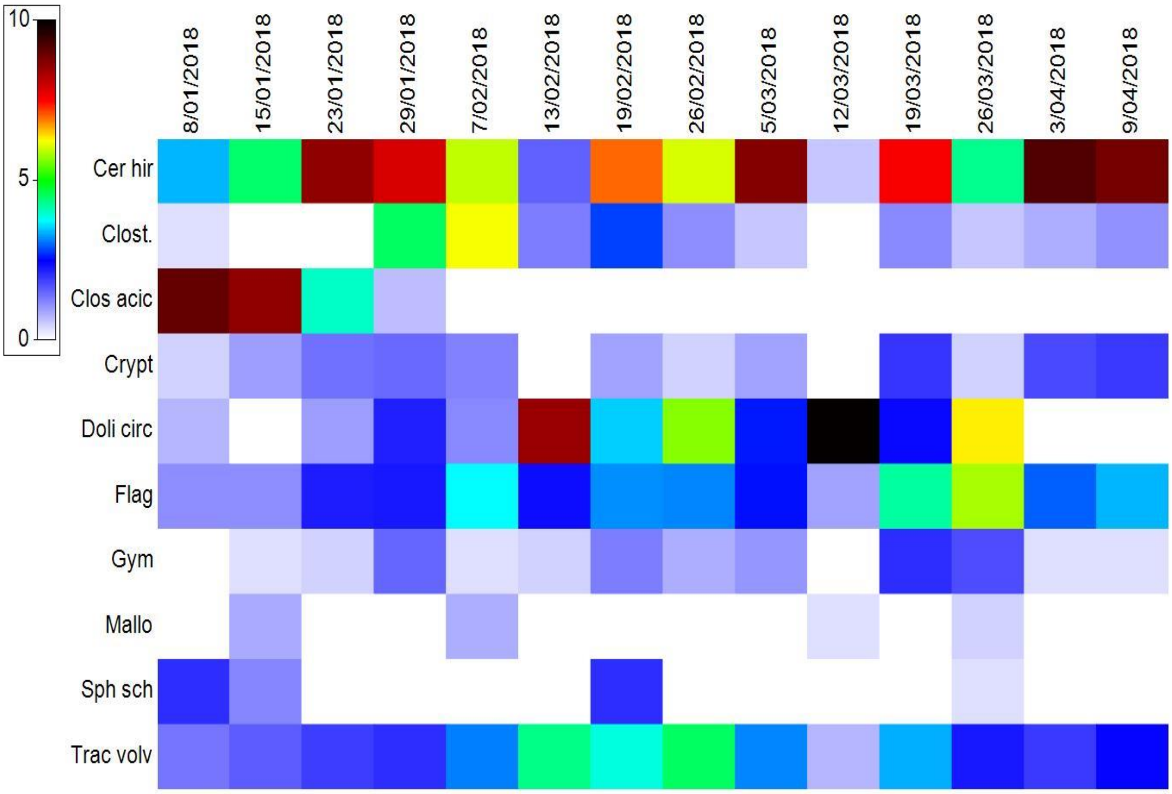

Figure 5: Dendrogram showing change in community composition between weekly sampling occasions in Pegasus Lake between January and April 2018. Species names are listed in Appendix B.

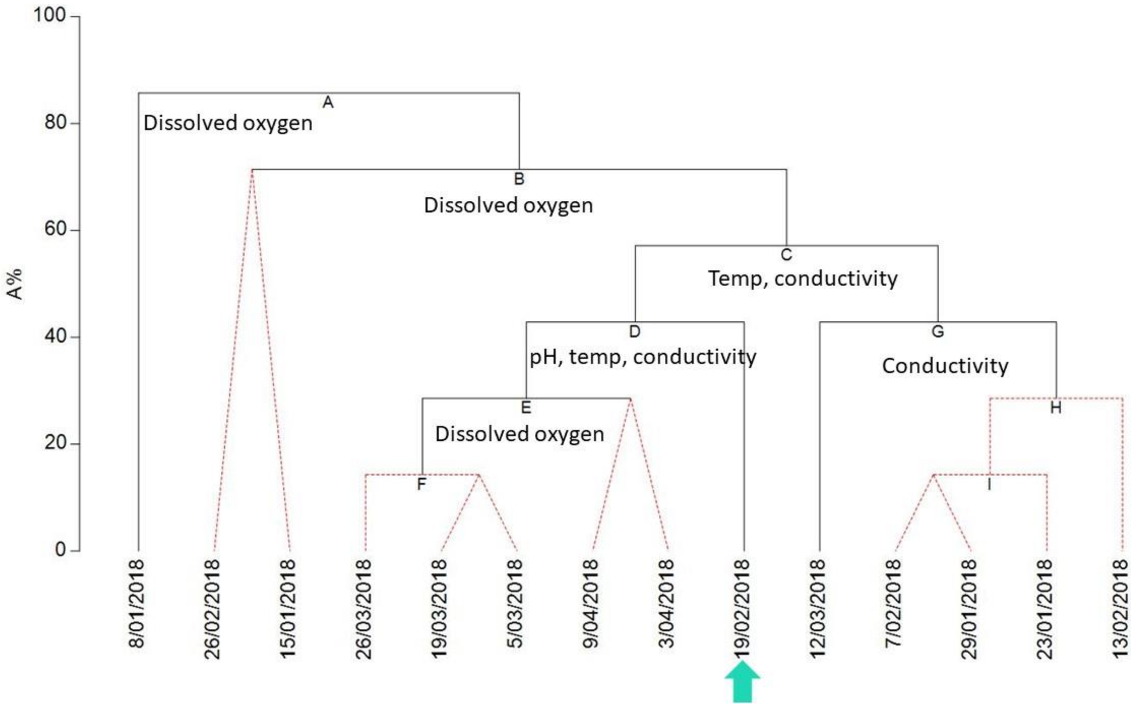

Figure 6: LINKTREE showing significant (p <0.01) changes in the phytoplankton community composition in Pegasus Lake (as indicated by black branch) and the environmental variables that best explain the differences. Onset of cyanobacterial bloom in Pegasus Lake indicated by arrow.

5.0 MANAGEMENT OPTIONS

The five potential cyanobacterial bloom management options shortlisted for further scientific evaluation during the workshop are detailed below. Both capital and operational costings estimated for each option are indicative only as most options are purpose built and would require bespoke design. All cost estimates exclude any provision for trials, stakeholder engagement, consulting and council fees, as well as resource consent applications. The likelihood of obtaining resource consents that may be required, the process and information and cost and time required to obtain any required consents is not part of this report.

5.1 Active Sediment Capping

Active sediment capping (ASC) is the process whereby a P-inactivation agent is used to bind and block the release of P from the lake sediment. When applied correctly and in sufficient quantities, ASC has been shown to substantially reduce the internal nutrient load of P. The reduction of P loading can often be an effective approach to phytoplankton biomass control in highly enriched lakes where there is a high internal P loading (Gibbs et al. 2011), such as is the case in Pegasus Lake. P-inactivation agents are either applied as a flocculation agent to strip dissolved P from the water column, or as a capping agent over the surface of the sediment (Cooke et al. 2005). These agents are typically modified clays enriched with aluminium (Gibbs et al. 2011), iron (Zamparas et al. 2012) or lanthanum (Haghseresht et al. 2009). Further details on ASC can be found in Golder (2017).

Gibbs et al. (2011) examined the efficiency of four P-inactivation agents for P binding efficacy and the minimum dose rates required in soft waters. The four agents were Allophane, Phoslock®, Z2G1 and Alum. Gibbs et al. (2011) demonstrated that all four products did block the release of P from the sediment but the amount of product required to completely block all releasable P from the top 4 cm of sediment varied as a function of both the P-binding capacity of the product and its interaction with the biogeochemistry of the lake water and sediments (Table 7). The authors also noted a number of non-target effects of all four products which are summarised in Table 7.

Both Alum and Phoslock® have been applied to lakes in New Zealand. Alum was used successfully to reduce P concentrations in Lake Rotorua, by dosing into an inflowing stream (Hamilton et al. 2016), while direct dosing of zeolite into Lake Okaro also successfully prevented cyanobacterial bloom formation (Gibbs & Champion 2013). Phoslock® was applied to Lake Okareka on three occasions between the period August 2005 and March 2007 (McIntosh 2007). Following application, phosphorus leaching from the sediment decreased for at least six months, while water column physio-chemical properties improved, with shorter and later periods of anoxia compared to pre-application (McIntosh 2007). Alum has a peak binding capacity at pH 7, but loses its binding capacity at pH 8.5, whereas Phoslock® works best between pH 8 -11. Pegasus Lake has a median mid-morning pH 8.5, meaning that of the two agents, Phoslock® would be the most suitable for the lake (Max Gibbs personal communications).

Gibbs et al. (2011) noted several factors that may negatively affect the efficacy of the ASC in lakes including the depth of the lake (with shallow lakes being more susceptible to sediment resuspension and transport), depth-induced pressure effects, spatial variability in the amount of bioavailable P in the sediment and disturbance of the ASC layer by benthic biota through bioturbation or groundwater inflows. Of these factors, sediment resuspension and sediment disturbance are the two main concerns for Pegasus Lake. ASC resuspension may occur during high wind events sufficient enough to disturb the lakebed, particularly if the ASC re-suspends at a higher rate than the sediment, as is the case for Alum. Sediment and ASC disturbance may occur if groundwater percolates through the bottom of the lakebed in Pegasus Lake.

The longevity of the P-inactivation agent, regardless of which one is chosen, is finite, with continued input of DRP into the lake eventually exhausting the residual P-binding capacity of the ASC agent, while subsequent deposition of detritus will resupply P to the sediment and eventually bury the ASC layer (Welch & Cooke 1999, Gibbs et al. 2011). Another key factor in determining the longevity of the ASC agent is the mineralisation rate of the organic P in the sediment into DRP and its subsequent diffusion out and binding with the ASC agent (Gibbs et al. 2011). Gibbs et al. (2011) estimated that the ASC agent would last at least four years in soft water lakes, based on their mesocosm experiment, but predicted greater longevity with inflow nutrient management and reduction in algal blooms.

Table 7: P-inactivation agents, their phosphorus binding and removal capacity (at a phosphorus sediment concentrations of 3.17 g P m^{-2}) and measured negative effects. Data source Gibbs et al. (2011).

Product	P-removal efficacy (g m^{-2})	P-binding capacity (g m^{-2})	Negative effects
Allophane	220	200	Suppression of coupled nitrification / denitrification under aerobic conditions. Enhanced NH_4-N release (66 % increase).
Phoslock®	280	260	Suppression of coupled nitrification / denitrification under aerobic conditions. Enhanced NH_4-N release (457 % increase). Leaching of Lanthanum into the water column.
Z2G1	190	150	Suppression of coupled nitrification / denitrification under aerobic conditions. Enhanced NH_4-N release but rapidly bound to the agent.
Alum	80	70	Suppression of coupled nitrification / denitrification under aerobic conditions. Enhanced NH_4-N release (70 % increase). Easily disturbed and resuspended.

It was not possible to sample the sediment in the bottom of Pegasus Lake, with Ekman grabs being unsuccessful in collecting a sample. This suggests that the amount of sediment and associated P available for mineralisation in the bottom of the lake is in limited quantity.

Low sediment volume, coupled with PP comprising between 61 - 95 % of the total P pool, suggests that the use of ASC agents, alone, to remove DRP from the water column and bind up the DRP released out of the sediment is unlikely to be sufficient to mitigate the current phytoplankton growth and subsequent bloom issues.

Phytoplankton precipitation, using a combination of a flocculent (Flock) and P-inactivation agent (Lock) technique, has been shown to be an effective method for reducing the TP concentrations in the water column (Lürling & van Oosterhout 2013, Noyma et al. 2017).

Lake Rauwbraken is a 4 ha, 15 m deep groundwater fed lake in the Netherlands. Cyanobacterial blooms have been an issue in the lake from the mid-1990s, resulting in a 4-month swimming ban due to blooms of *Planktothrix rubescens*, *Microcystis aeruginosa*, *Aphanizomenon flos-aquae*, *Anabaena* spp. and *Woronichinia naegeliana* (Lürling & van Oosterhout 2013). A trial experiment using the Flock & Lock method, resulted in a 92 % reduction of TP, an 80 % reduction of phytoplankton biomass and no cyanobacterial blooms in the five years following a single treatment (Lürling & van Oosterhout 2013). For this trial, Lürling & van Oosterhout (2013) applied the Flock & Lock treatment over 3 days, using the following protocol:

- Day 1 application of 2 tons of Phoslock® to remove dissolved phosphorus from the water column;
- Day 2 application of 2 tons of Polyaluminium chloride (PAC), pH buffered with 75 kg of $Ca(OH)_2$ to flocculate the phytoplankton;
- Day 3 application of 16 tons of Phoslock® to act as both a ballast to sink the phytoplankton and a sediment cap to prevent P-efflux (Lock).

All compounds were applied at the water surface from a barge, through a spray manifold after mixing with lake water, thus the Phoslock® was applied as a slurry, and the flocculent was strongly diluted (Lürling & van Oosterhout 2013.

The Flock & Lock method could be applied to Pegasus Lake as both Phoslock® and PAC are commercially available in New Zealand. PAC is currently used in the water treatment industry for the flocculation of organic and metal colloids prior to sedimentation and/or filtration (WaterNZ 2013). Based on estimates of Gibbs et al. (2011) and Lürling & van Oosterhout (2013) the cost of ASC in Pegasus Lake is estimated to be **$350,000**[2] (excluding GST).

However, as there are uncertainties around exact quantities to cause an effect in Pegasus Lake, large scale mesocosm trials using Pegasus Lake water would be required to confirm both dosing rates and efficacy.

ASC typically fails when incorrectly applied, including at quantities too low to form a thick uniform layer on the lake bed (Gibbs et al. 2011). Another issue to consider for Pegasus Lake is the rip-rap bottom lining of the lake that may interfere with the distribution of ASC across the lake floor.

As the Flock & Lock method has not been trialled in New Zealand it is difficult to predict with any certainty the likelihood of success in Pegasus Lake.

An indication of the pros and cons of ASC, using Flock & Lock, in Pegasus Lake is listed below:

Pros:

- Partially proven to work at full-scale in NZ lakes (Phoslock and PAC used in separate applications)
- Relatively definable capital and operational costs once dosing rates are known
- Results apparent within several weeks
- Potential to provide effectiveness for up to 4 years if successful

Cons:

- Limited sediment in Pegasus Lake
- Capping agent readily disturbed with percolating groundwater
- May increase NH_4-N release at the expense of nitrification under aerobic conditions
- Requires successful flocking of algal biomass before sedimentation
- Partially proven to work at full-scale in NZ lakes (Phoslock and PAC used in separate applications)

[2] Lürling & van Oosterhout (2013) do not explain their rational for using 2 tons of PAC for treating the lake. The quantity estimated for Pegasus Lake has been based on the assumption that Lürling & van Oosterhout (2013) used chlorophyll-*a* concentration as the basis for their calculation. The actual amount required may affect cost estimates.

5.2 Algal Turf Scrubber

Algal turf scrubbers (ATS) are a biological based active treatment system used for the treatment of diffuse pollution in a range of water types, from wastewater, storm water, lakes and rivers (Sutherland & Craggs 2017). ATS are essentially an artificial flow-way that promotes periphytic algal growth and nutrient assimilation. By integrating flowing water with high light and frequent harvest, periphyton have been shown to achieve high levels of nutrient removal and productivity across a broad range of aquatic habitat types and nutrient pollution levels.

ATS are a structure comprised of a solid bottom, such as compacted earth, an impermeable membrane to prevent groundwater intrusion and seepages from the ATS, periphyton attachment structure, such as a 2-D or 3-D mesh, a gentle slope and a flow control mechanism. Further design details can be found in Sutherland & Craggs (2017).

With respect to the treatment of Pegasus Lake water and its subsequent return to the lake, a pump would be required to deliver the water to the ATS and a second pump required to return the water to the lake. In addition to the pumps, pipework would be required to ensure that water returned to the lake was sufficiently far enough away from the outflow to the ATS or the existing outflows, to prevent a short circuit of treated water. In addition to this, a rock filter would be required at the head of the ATS, to remove phytoplankton biomass and associated particulate nutrients from the lake water, and one at the end of the ATS to remove any sloughed off periphyton before the water is returned to the lake.

ATS also require on-going harvesting of the periphyton biomass. During summer, harvesting typically occurs weekly, decreasing to monthly during winter. The harvested biomass requires either disposal or can be reused as an organic fertiliser. An example of an ATS used to treat polluted waters is shown in Figure 7.

Phytoplankton biomass retained in the rock filter would not be able to be easily removed and any retained biomass will eventually decay, which could lead to localised odour issues.

Figure 7: Algal turf scrubber (right) used to treat polluted water in Chesapeake Bay, USA and harvester removing periphyton from the flow-way (top left). Photo source Sandia National Laboratories.

Compared to the other options, ATS require regular active management in terms of harvesting of algal biomass. ATS are also susceptible to gazer infestation or heavy rain wash-off, which would both reduce the performance of the system for short periods of time (week/s).

ATS provide similar natural ecosystem amenity to that of a shallow stream, and wading birds are attracted to the ATS surface where they eat the macroinvertebrates that colonise the algae (Sutherland & Craggs 2017).

However, while landscaping can be done around the ATS, such a structure growing filamentous algae may not be regarded as being aesthetically pleasing.

ATS have not been trialled at pilot-scale or full-scale in New Zealand, to date. However, overseas, both pilot-scale and full-scale ATS have been built (several at hectare scale) and operated for treatment of secondary wastewater, dairy farm effluent, finfish aquaculture effluent and agricultural runoff (Sutherland & Craggs 2017). Similar algal-based systems for the treatment of diffuse pollution in waterways have also been operated overseas (Sutherland & Craggs 2017). Pilot-scale trials using Pegasus Lake water would be required to determine the likelihood of success at full-scale.

For treatment of Pegasus Lake water, a 40-day HRT is likely to be required. This HRT would allow for sufficient nutrient removal without compromising the treatment process, nor allowing sufficient internal nutrient recycling to occur in the lake.

For a 40-day HRT, a 1.25 ha ATS would be required. Given the limitations of available space around Pegasus Lake, the ATS could be divided into several smaller units or be installed as an island in the middle of the lake. Using the ATS would lower the lake by approximately 2 - 5 cm, which should not be noticeable to the general public.

The cost estimate for the ATS for Pegasus Lake is approximately **$850,000**[3] (excluding GST), while annual operating costs (including disposal of algae costs) are anticipated to be **$30,000**. Separation of the ATS into several smaller units may result in increased construction and capital costs. A SWOT analysis for using ATS to remove nutrients and phytoplankton biomass from Pegasus Lake water is presented in **Error! Reference source not found.**.

An indication of the pros and cons of ATS to remove nutrients and phytoplankton biomass from Pegasus Lake water is listed below:

Pros:

- Proven technology at full scale
- Natural treatment with no chemicals added to the lake
- Can recover nutrients for re-use as fertiliser

Cons:

- High capital and operational costs
- Requires substantial land area immediately next to, or in the lake
- Aesthetically intrusive with potential for odour issues
- Vulnerable to grazers and other disturbance

[3] As the high capital costs for the ATS at Pegasus Lake is related to the pumps and pipework required, further refinement of cost estimate would require specific location of ATS site and return pipes.

5.3 Artificial Mixing or Aeration

There are two main approaches used to reverse the effects of lake stratification. The first approach is the use of artificial (mechanical) mixing to overcome the physical stratification of the water column, allowing the surface and bottom waters to interchange. The second approach is hypolimnetic aeration, whereby air is introduced into the hypolimnion at a rate sufficient to overcome the rate of oxygen consumption. Mixing devices have been utilised successfully in a number of lakes and reservoirs in New Zealand to prevent, or overcome, anoxia, including in a number of drinking water reservoirs in the Auckland region. However, reported success in managing / or preventing cyanobacteria blooms has been limited.

As noted in Golder (2017) its efficiency as a bloom management strategy is limited to lakes with a mean depth within the range of 16 - 30 m.

To date, there are no reported case studies of hypolimnetic aeration devices being used in New Zealand.

The use of artificial mixing or aeration of the hypolimnion as a mechanism to breakdown stratification, and potentially control cyanobacteria, in Pegasus Lake is discussed below. However, there is no evidence to suggest that stratification of the water column was the cause of cyanobacterial bloom development in Pegasus Lake.

5.3.1 Artificial (mechanical) mixing

The purpose of mechanical mixing is to either prevent the onset of stratification or to break down existing stratification in the water column. This is achieved by supplying sufficient energy to create upwelling water currents that disrupt the physical stratification, resulting in the mixing of surface aerated waters with the bottom waters.

Typically, mixing devices are considered to be more energy efficient that hypolimnetic aeration devices as they can be turned on and off as required, thus reducing their overall operating costs. However, this may not necessarily be the case for Pegasus Lake due to the potential of continual, or near continual seepage of low DO groundwater into the lake, which may necessitate continual mixing during the critical months (November - March).

The amount of mixing energy input required is determined by the strength of the thermocline, which can be quantified by calculating the potential energy anomaly (PEA). PEA is defined as the amount of mechanical energy (per m^3) required to instantaneously homogenise the water column with a given density stratification (Burchard & Hofmeister 2008). PEA was calculated based on the monthly water column profiles in Pegasus Lake from July 2014 to May 2018, using the equation of Simpson (1981):

$$\emptyset = \frac{1}{D}\int_{-h}^{n} gz\,(\bar{\rho} - \rho)dz;\ \bar{\rho} = \frac{1}{h}\int_{-h}^{0} \rho\, dz$$

where ρ is the water density at stratification, $\bar{\rho}$ is the mean water density of the entire water column, D is the water column height $(n + h)$ and g is the gravitational acceleration.

In Pegasus Lake, PEA ranged from 0 (when the lake was fully mixed) to 35.5 J/m^3, when the lake was at its strongest measured stratification in January 2018 (Figure 8).

However, it should be noted that as these values are calculated on the monthly water column profiles, they do not necessary represent the strongest stratification event for any given month or any given year. For example, for the January 2018 PEA estimate, surface water temperatures were 5 °C lower than the maximum recorded for that month (sustained for a period of eight days). Assuming a similar temperature attenuation over the mixed layer of the water column during the peak water temperature period in January, PEA, and therefore energy input required to breakdown stratification, would increase by 150 % to 53.6 J/m^3.

Further refinement of PEA estimates in Pegasus Lake would be required to determine the maximum energy inputs required for mixing, to determine the size and or numbers of the mechanical mixers required.

There are a number of disadvantages associated with mechanical mixing including:

1) Insufficient power inputs would result in failure to break down stratification;

2) Warming of the bottom waters may increase the DO consumption rates;

3) Complete mixing may result in sub-optimal DO concentrations throughout the water column.

DO concentrations in the water column following full mixing were estimated from the monthly water column profiles from July 2014 to May 2018 for Pegasus Lake (Figure 9). Fully mixed DO concentrations were below 5 g/m^3 on one occasion and below 7 g/m^3 on 10 monthly occasions (Figure 9). 7 g/m^3 is the minimum recommended 7-day mean DO protective level for adult fish in New Zealand waters (Franklin 2014).

It should be noted that these DO estimates are conservative as no allowances have been made for any increase in DO consumption as a result of warming of the bottom waters or decrease in photosynthetic rates in response to increased mixing depth. Sutherland et al. (2015) demonstrated that, in highly productive systems, photosynthesis, as measured by oxygen evolution, decreased with increasing mixing depth, while respiration increased.

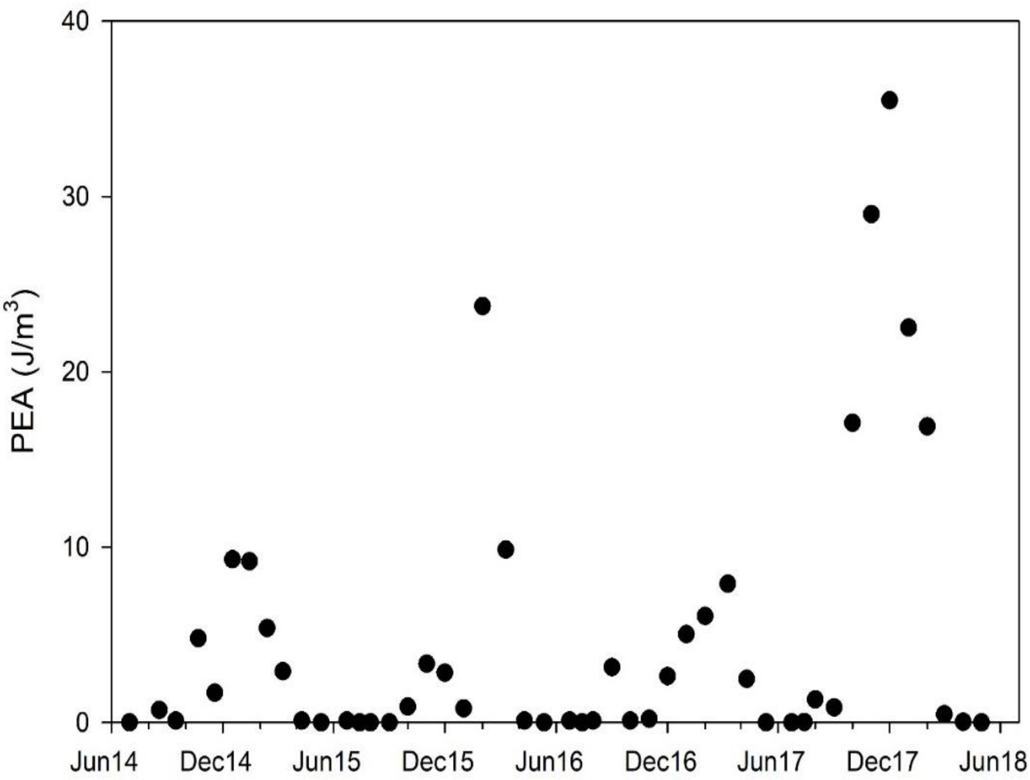

Figure 8: Potential energy anomaly (PEA) in Pegasus Lake calculated from monthly water column profiles from July 2014 to May 2018.

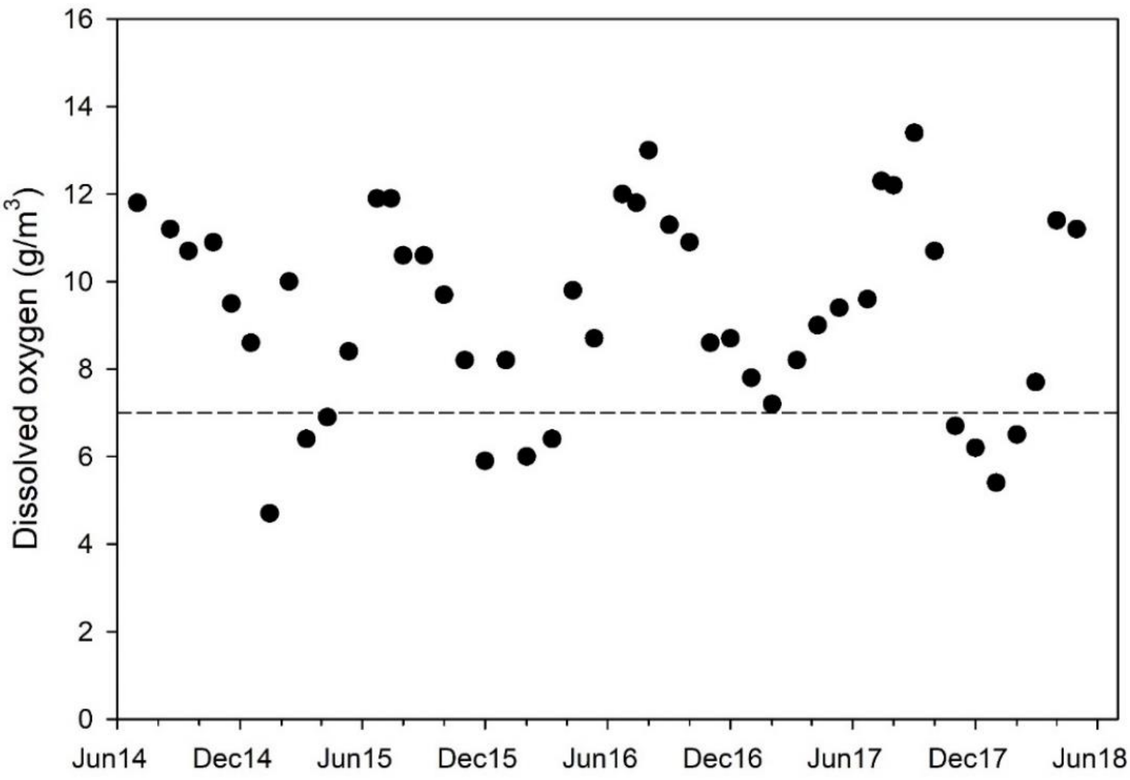

Figure 9: Estimated dissolved oxygen concentrations in the water column, following mechanical mixing, based on monthly water column profiles from July 2014 to May 2018. Black dashed line indicates the minimum recommended 7-day mean DO protective level for adult fish in New Zealand waters.

The estimated peak summer PEA for Pegasus Lake was similar to the maximum PEA (50 J/m^3) reported for the Maitai Reservoir in Kelly (2015) and it is likely that capital and operational costs would be similar. Kelly (2015) estimated capital cost for the design and construction of a mechanical mixing system for Maitai Reservoir to be **$1,050,000**, with annual operations costs within the order of **$20,000** (power and maintenance).

An indication of the pros and cons for using mechanical mixing to improve water quality in Pegasus Lake is listed below:

Pros:

- Reduce incidents of NH_4-N release under anoxic conditions
- No addition of chemicals to the lake
- If applied at sufficient scale for sufficient periods of time could break down stratification and prevent anoxia in the lake

Cons:

- High capital and operational costs
- Warming of bottom layers may increase oxygen demand
- Unlikely to overcome cyanobacterial bloom (there is no evidence to suggest that stratification of the water column was the cause of cyanobacterial bloom development)

5.3.2 Hypolimnetic aeration

Hypolimnetic aeration systems work by pumping compressed air to the bottom of the water column through a series of diffuser ports. Fine bubbles of compressed air exit the diffuser and entrain the hypolimnetic water upwards as the bubbles rise to the surface, expanding in the process (Zic & Stefan 1990). This can result in mixing of the entire vertical water column, preventing, or reducing, stratification by providing an even (isothermal) distribution of temperature and a uniform distribution of DO down the water column. Several designs of hypolimnetic aerators have been used in lakes and reservoirs, with the most commonly employed designs including full (or partial) lift aerators, bubble plume diffusers and Speece cones (Kelly 2015).

The efficiency of hypolimnetic aerators, including the system design, diffusers and manifolds, varies according to the hypolimnetic depth, temperature gradients and oxygen depletion rates (Ashley & Hall 1990).

For these reasons, relatively small changes in both the design and operation can result in significant variation in the system's performance and detailed engineering design works are required to make predictions of the system requirements for a given lake or reservoir (Kelly 2015).

Aeration rates required for Pegasus Lake were estimated from the mean total oxygen demand (TOD) measured at 3, 4 and 5 m depth on two occasions in October and November 2017. While TOD is anticipated to vary from month to month in the lake, most likely increasing with increased water temperature and microbial community, this estimate gives an indication of the potential amount of aeration required for Pegasus Lake. Estimated aeration rates were calculated at 60 % and 90 % oxygen transfer efficiency, as reported from previous aeration studies (e.g. McQueen & Lean 1989, Burris & Little 1998, McGinnis 2000). Transfer efficiency is the efficiency at which gas bubbles dissolve into the surrounding solution and is dependent on the partial pressure coefficients of the gases in the water column, the diameter of the gas bubble, the depth of the aeration device and the height of the water column. Estimated aeration rates for Pegasus Lake hypolimnion are presented in Table 8**Error! Reference source not found.**.

Table 8: Estimated airflow rates required to offset the rate of oxygen depletion in the hypolimnion in Pegasus Lake.

Parameter	Pegasus Lake
Depth of thermocline (m)	2.5
Hypolimnetic volume (m^3)	229,766
Total hypolimnetic DO consumption range (kg/day)	37 - 63
Airflow rate range (60 % efficiency) (l/s)	3.0 - 5.1
Airflow rate range (90 % efficiency) (l/s)	2.0 - 3.4

As mentioned above, capital and operational costs cannot be estimated without a detailed engineering design works specifically for Pegasus Lake. However, with a number of common elements between the mechanical mixing and aeration options (such as a compressor building, compressors, electricity installation and supply, construction and overhead costs), it is plausible to suggest that costs the aeration system would be similar to the mechanical mixing system.

Aeration of the lake could lead to a change in the buoyancy of the water, resulting in a lower density than non-aerated water. The lake is currently used for both swimming and watercraft activities and a decrease in water density would mean that a person in the water would sink lower and need to exert far more energy to keep from sinking and the risk of drowning.

Given the shape of the lake basin, this is likely to affect a substantial area of the lake, although the very shallow regions of the beaches may not be affected

An indication of the pros and cons for using aeration to improve water quality in Pegasus Lake water is listed below:

Pros:

- Reduce incidents of NH_4-N release under anoxic conditions
- No addition of chemicals to the lake
- If applied correctly, at sufficient scale, can break down stratification

Cons:

- High capital and operational costs
- Shape of lake requires numerous aerators
- Reduction in buoyancy and associated health and safety risks
- Aeration efficiency can be highly variable, with lowest efficiency likely in summer
- Unlikely to overcome cyanobacterial bloom as the dominant bloom causing species is capable of overcoming mixing induced turbulence

5.4 Flushing

Flushing or dilution of the lake can help to improve water quality in the lake by both the dilution of nutrient concentrations in the lake and removal (flushing) of algae and particulate nutrients from the lake. Successful flushing requires flushing water to be of considerably lower nutrient concentration than the in-lake concentrations and to be of sufficient volume to flush the lake in a short (1-2 months or less) period of time.

5.4.1 Surface flushing

Surface flushing of the lake can be used as a control measure to reduce the duration of the bloom once it has established in the lake. This would involve raising the lake level in the weeks prior to the bloom onset, followed by rapid release of the water to create a surface flush. Raising the lake level can be easily achieved in Pegasus Lake by raising the adjustable weir control structure early in December. This would create a longer retention of both rainwater and groundwater inflows in the lake and would raise the lake to an acceptable limit until bloom onset (in early February). Once the bloom has become established, rapid release of this additional water would create a surface draw that would flush a large proportion of the bloom out of the lake within a short period of time. However, surface flushing would have no effect on stratification.

Given that the cyanobacteria bloom onset in Pegasus Lake typically occurs during a drier period of weather, surface flushing could only be used once during the bloom event. However, the length of bloom duration in Pegasus Lake has typically been short and a single flush may be sufficient to reduce its duration. If there was sufficient supplementary water available to provide a short duration flow augmentation, a second flush within a bloom cycle could be achieved. The secondary flush would be achieved by raising the lake level over several days (e.g. seven days) followed by a subsequent rapid release of that water downstream. To raise the lake by 0.3 m (the available freeboard between the weir plate and the top of the weir control structure) over a seven-day period, an additional 50 l/s would be required.

An indication of the pros and cons for using surface flushing as a means to manage cyanobacterial blooms in Pegasus Lake water is listed below:

Pros:

- No addition of chemicals to the lake
- No capital and minimal operational costs

Cons:

- One-off opportunity during a given bloom cycle
- Shape of lake may result in pockets of cyanobacteria retained in the lake

5.4.2 Full lake flushing

Full lake flushing is achieved by augmenting the inflows with supplementary water. If managed successfully, flow augmentation could beneficially improve the water quality in Pegasus Lake by a) diluting water nutrients and b) removing phytoplankton biomass and associated particulate nutrients. Full augmentation of the lake, which is the complete turnover of the lake water within a relatively short period of time, would help to 'reset' the system to at least pre-bloom conditions, if not better. Immediate improvements in the visual appearance of the lake would be achieved as the colour and clarity in the lake is driven by the phytoplankton community. Improvements in the TLI, a measure of the health of a lake, would also be immediately realised in the lake.

Based on the estimated lake volume in Table 2, the flow rate required to turn the lake over in five days is 1.07 m^3/s and in 10 days is 0.54 m^3/s.

High outflows from the lake may negatively affect the receiving environment, resulting in negative consequences for the downstream receiving environment.

At this stage, it is unknown what potential water sources, if any, are available to flush the lake. However, in order to estimate the possible outcomes in the lake following a flush, water quality from the nearby Ashley River was used to predict TLI scores. As the response of the phytoplankton is unclear, due to a) possible flushing avoidance strategies by motile species, b) increased growth response due to reduction in light limitations and c) potential 'short-circuiting' of flushing water out of the lake, a conservative approach using varying percentage removal rates of phytoplankton / particulate nutrients was used in estimating the TLI following lake turnover. Estimated TLI outcomes are presented in Table 9. Currently, Pegasus Lake is categorised as being Hypertrophic, the highest (poorest quality) category in the TLI score system. This means that, based on the predicted outcomes in Table 9, flushing of the lake would result in a decrease in TLI by one to two categories to either Supertrophic or Eutrophic, depending on the percentage removal of phytoplankton achieved with the flush.

Capital and operational costs for flushing would be dependent on the source of water. If new bores are required to be drilled, associated pump, pipework and electrical work, as well as management or mitigation of downstream flooding, then the capital cost estimate for full flushing is likely to exceed **$5,000,000**. Cost estimate excludes any downstream flood mitigation measures and costs associated with obtaining resource consents. A full engineers design would need to be undertaken before a more accurate costing can be determined.

Table 9: Predicted trophic level index (TLI) outcomes following an augmented flush of Pegasus Lake with varying removal rates of phytoplankton.

Removal of phytoplankton (%)	Total nitrogen	Total phosphorus	Chlorophyll-*a*	TLI
50	391	83.5	71.0	5.6 Supertrophic
60	313	66.8	56.8	5.4 Supertrophic
70	235	50.1	42.6	5.0 Supertrophic
80	157	33.4	28.4	4.5 Eutrophic

An indication of the pros and cons for using full lake flushing as a means to manage water quality and cyanobacterial blooms in Pegasus Lake water is listed below:

Pros:

- Improved water quality within a relatively short timeframe
- No addition of chemicals to the lake
- No potentially adverse effects to the lake biota
- Flushing rates can be modified to suit water quality objectives

Cons:

- Flooding issues for downstream receiving environment
- Substantial supplementary water supply is required, and availability and source of water is unknown
- Drought years may prevent application
- Dilution effect lessened with degradation of catchment water
- May not be sufficient dilution to achieve desired outcome

5.5 Adaptive Management Regime

Under the adaptive management approach, monitoring for cyanobacterial bloom development, and the management of lake closures, would still be undertaken in accordance with the Ministry for the Environment Guidelines (2009). For Pegasus Lake, this presently involves monitoring for the presence of cyanobacteria during the summer months, placement of warning placards around the lake during the bloom event and management of the timing of events involving the use of the lake to ensure that they occur outside of the potential bloom periods.

Since the start of blooms in Pegasus Lake in the summer of 2015, bloom onset has occurred at similar times each summer, with three of the four blooms starting in the 7th week of the calendar year (Figure 10). If the predictability of bloom onset continues, both monitoring and management response can be targeted around the early February period.

In addition to the current monitoring and lake-closure management, additional measures could be implemented under the adaptive management option to help lessen the length of time the lake is in bloom. These measures are discussed below.

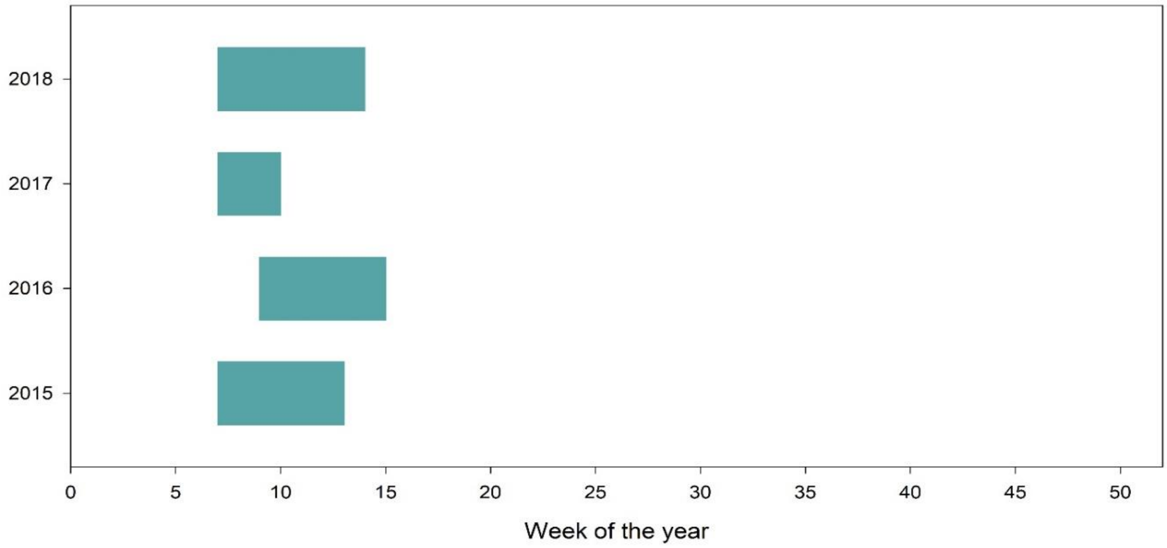

Figure 10: Time of bloom onset and duration for the summers 2015 to 2018.

5.5.1 Weir management

Water leaving the lake is channelised through corrugated iron culverts (three for the northern outlet and one for the southern outlet), underneath the road bridges, towards the weir. Immediately downstream of the road bridges, the area in front of the weir widens, which acts to slow the water down, particularly at the margins. Cyanobacterial biomass accumulates at the margins, preventing it from being flushed from the lake. In addition to this, the open area provides habitat for macrophytes to grow, which, when they reach the surface, act as a filter that retains cyanobacteria biomass upstream of the weir (Figure 11). This accumulated biomass acts as a re-inoculum source that is blown back into the lake under easterly winds. Modification of the weir to prevent biomass accumulating at the margins and prevention of surface reaching weed stands would help to flush the bloom from the lake and prevent re-inoculum.

Figure 11: Looking downstream at the Southern Outlet, showing surface reaching macrophytes retaining cyanobacteria scum upstream of the weir structure.

5.5.2 Beach management

Benthic algae occur along the shallow margins of the lake and periodically lift off from the lake floor, either as a result of trapped gas in the mat or from wave action and is washed ashore (Figure 12). Cyanobacteria trapped along the shoreline presents a potential hazard to children and dogs playing in the area, as well as providing a re-inoculum source for the lake during rain or storm events. Periodic removal of the shoreline accumulated algae helps to reduce both risk and re-inoculum.

5.5.3 Weed and grass management

The dominant macrophyte in Pegasus Lake is the introduced curly pondweed, *Potamogeton crispus*. In Pegasus Lake, *P. crispus* is currently managed with the regular application of the aquatic herbicide, Diquat. However, due to its low susceptibility to diquat (Wells & Sutherland 2002), biomass cover is only temporarily reduced in the lake and regrowth occurs rapidly (author's own observations). *P. crispus* is vulnerable to high water temperatures, with die-off occurring when water temperatures reach 25 °C (Sutherland 2005). Surface water temperature regularly exceeds 25 °C during mid to late January in Pegasus Lake, resulting in *P. crispus* die-off. The decaying biomass either washes up on shore, trapping cyanobacteria or acts as a nutrient pulse to stimulate algal and cyanobacterial growth in the lake (Figure 13).

Mechanical harvesting, such as raking or dragging of chains, would better manage the *P. crispus* biomass in Pegasus Lake than current chemical control. Mechanical harvesting would remove biomass from the lake, minimise shoreline accumulation (and associated trapping of cyanobacteria) and reduce nutrient recycling.

Grass that is mown around the perimeter of the lake provides an additional nutrient input as clippings are blown into the lake. A cut and collect mowing technique for the grass on the lake side of the perimeter pathway minimises this source of nutrient. Catching all of the grass clipping and removal offsite would eliminate this source of nutrients into the lake.

Figure 12: Benthic algae washed onto the beach, trapping cyanobacteria biomass on the surface (as indicated by white arrows).

Figure 13: Decaying *Potamogeton crispus* washed up on shore at Pegasus Lake.

An indication of the pros and cons for adaptive management of Pegasus Lake is listed below:

Pros:

- Achieves the objectives of the Lake Management Plan
- Proven protocols and procedures for notifying and managing cyanobacterial bloom durations
- Low capital and operational costs

Cons:

- No reduction in cyanobacterial bloom duration
- No ability to improve water quality

6.0 VALIDATION OF MANAGEMENT OPTIONS

The likelihood of success for each option is dependent on a number of factors. The design, operation and management of all five options would need to be specific to Pegasus Lake, taking into account the morphology and physico-chemical environment of the lake.

With the exception of the adaptive management option, a pilot-scale trial would need to be carried out to validate the system within the Pegasus environment. This would require the trial to be either established in the lake (such as mesocosms for 'Flock & Lock') or alongside the lake (such as an ATS).

The purpose of any trial would be to validate the efficiency of the option, any interactions between the option and the surrounding environment and / or public, as well as any potential ecological effects.

It should be noted that success at pilot-scale does not always guarantee success at full-scale as many factors that may negatively affect performance at full-scale can be better managed or eliminated at pilot-scale.

Due to the likely scale of any trial, resource consents may be required to undertake any trials.

6.1 Active Sediment Cap (Flock & Lock) Trial

An ASC, or flock & lock, trial would involve the establishment of six mesocosms in the lake, similar to those shown in Figure 14. Each mesocosm would measure approximately 1.5 m in diameter and extend down to 5 m below the water surface. The mesocosms would be attached to floats and anchor points to ensure they maintain their position in the water column (Figure 14). Installation of the mesocosms would require divers and a boat with a winch system.

Flock & lock would be added to three mesocosms, while three would serve as the control. The water quality parameters dissolved oxygen, temperature, pH, conductivity, turbidity, nutrients (dissolved, particulate and total), and chlorophyll-*a*, along with both phytoplankton and zooplankton community dynamics, would be measured in each mesocosm at times zero, one week, one month and two months post application. The trial would be undertaken during the summertime period, when the ecosystem is at its most stressed and algal biomass at its maximum.

As this trial involves the application of chemicals to water, a resource consent would be required. Part of the consent conditions may include undertaking an assessment of effects on fish (in this case bullies, eels, inanga, and potentially trout) by enclosing them inside cages in the mesocosms during, at least part of, the trial. Regardless, of whether it is included in the trial consent conditions, it is recommended that a fish assessment is carried out as part of the trial to provide supporting information for consent application for full lake treatment. A resource consent may also be required for the temporary installation of the mesocosm setup in the lake.

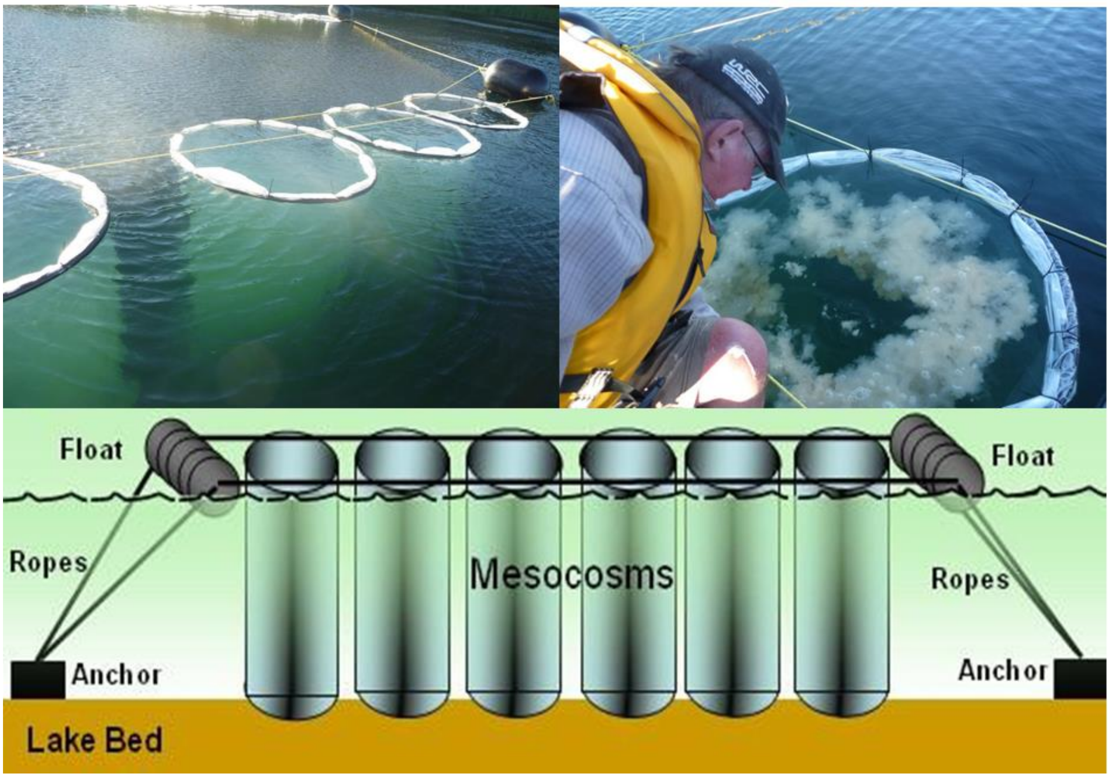

Figure 14: Example of lake-based mesocosms used for sediment capping experiments in Lake Rotorua. Photo source NIWA.

6.2 Algal Turf Scrubber Trial

An ATS trial would involve the installation of a 3-channel flow-way approximately 1 m wide by 50 – 100 m long (available space dependant), with a sand/rock filter at the end. An example of a two-channel ATS flow-way is shown in Figure 15. The flow-way would need to be set up either alongside, or downstream of, the lake. Water would be pumped from the lake into a header tank, which will, in turn, gravity feed onto each of the 3-channels, through flow-control valves. Each channel will have a different flow rate, resulting in different hydraulic retention time and hydraulic loading rates on each channel. The purpose of this is to determine the maximum flow rate for the smallest size of ATS for a full-scale system. Local filamentous algae would be seeded onto the flow-way to initiate periphyton development.

An ATS trial would run for 10 weeks, with sampling undertaken twice per week. The water quality parameters dissolved oxygen, temperature, pH, conductivity, turbidity, nutrients (dissolved, particulate and total), and chlorophyll-*a*, will be measured before the water enters the flow-way and as it exists the flow-way. In addition to this, periphyton biomass (ash free dry weight and chlorophyll-a) will be measured twice weekly.

An ATS trial is less time critical than the other trial options, however, trials undertaken during the summertime period, would provide estimation of maximum removal rates by the periphyton.

Figure 15: Example of a two-channel trial-based algal turf scrubber for nutrient removal. Photo source University of Maryland.

6.3 Artificial Mixing / Aeration Trial

An artificial mixing and aeration trial would require a similar mesocosm set up as that described in Section 6.1 Active Sediment Cap (Flock & Lock) Trial and Figure 14. A total of nine mesocosms would be established in the lake, with three mesocosms assigned to mixing, three mesocosms assigned to aeration and three mesocosms would serve as the control. For the mixing trial, small submersible pumps would be used to mix the water column within the mesocosm. For the aeration trial, diffusers, similar to those used in fish aquaria, would be installed in the bottom of the mesocosm to replicate hypolimnetic aeration (Figure 16).

The water quality parameters dissolved oxygen, temperature, pH, conductivity, turbidity, nutrients (dissolved, particulate and total), and chlorophyll-*a*, along with the phytoplankton community dynamics, will be measured near the surface and near the bottom of each mesocosm at times zero, one week, one month, two months and three months post mixing /aeration commencement. The trial would be undertaken during the summertime period, when the energy requirements to overcome stratification are at the maximum. A resource consent may be required for the temporary installation of the mesocosm setup in the lake.

Figure 16: Example of air diffusers used in fish aquaria

6.4 Lake Flushing Trial

The lake flushing trial could be carried out in two stages. The first stage would assess the effectiveness of a surface flush during a cyanobacterial bloom while the second stage would assess the effectiveness of a full lake flush.

6.4.1 Surface flushing trial

A trial of surface flushing would require raising the lake level by 0.3 m (the maximum height between the weir plate and the top of the weir control structure (see Figure 17)) leading into summer. This can be achieved by raising the weir plates at both the northern and southern outlets, allowing the lake level to rise another 0.3 m as groundwater and rainfall enter the lake. Once the bloom has become established, the weir plates would be immediately lowered to cause a rapid drop in the lake level. This would result in approximately 30,000 m^3 of surface water leaving the lake, via the two outlets. It is estimated that, based on the geometry of the weir plates, it would take one day to flush this volume of water from the lake.

Success of the trial would be determined by the biovolume counts of cyanobacteria present at the monitoring sites, which forms part of the current summertime bloom monitoring programme.

Figure 17: Northern outlet weir control structure.

6.4.2 Full lake flushing trial

A full lake flushing trial would require a very similar mesocosm set up as that described in Section 6.1 Active Sediment Cap (Flock & Lock) Trial and Figure 14. Three mesocosms would be assigned a 5-day flush, three mesocosms assigned a 10-day flush and three mesocosms would serve as the control. For the 5-day flush, 1/5 of the mesocosm water volume would be replaced each day for five days, while for the 10-day flush 1/10 of the mesocosm water volume would be replaced each day for 10 days.

The water quality parameters dissolved oxygen, temperature, pH, conductivity, turbidity, nutrients (dissolved, particulate and total), and chlorophyll-*a*, along with the phytoplankton community dynamics, would be measured near the surface and near the bottom of each mesocosm according to the schedule in Table 10. The trial would be undertaken during the summer time period, when the ecosystem is at its most stressed and algal biomass potential at its highest. A resource consent may be required for the temporary installation of the mesocosm setup in the lake.

Table 10: Sampling schedule for full lake flushing mescocosm trial.

Sampling schedule	5-day flush	10-day flush	Control
Time zero	✓	✓	✓
Time 5-day	✓	×	✓
Time 10-day	✓	✓	✓
Time 20-day	✓	✓	✓
Time 40-day	✓	✓	✓

7.0 PATHWAY TO FULL-SCALE IMPLEMENTATION

All treatment options will require stakeholder engagement prior to both the trial and the implementation of the full-scale system. At full-scale, all treatment options will require resource consents, while the active sediment cap (flock & lock) trial will also require resource consents for the trial. Apart from stakeholder engagement, which is anticipated to be on-going during the process, the pathway to full -scale implementation for each of the five treatment options is outlined below.

7.1 Active Sediment Cap (Flock & Lock)

The implementation pathway for treating Pegasus Lake with active sediment capping (flock & lock) is outlined in Figure 18. The first stage is to apply for a resource consent for the application of chemicals to the lake. Once the consent has been obtained, a trial to determine the efficiency of flock & lock in Pegasus Lake would be undertaken. The results from the trial will help inform the dose rate required for full-scale treatment to inform cost estimates. A second resource consent for applying chemicals to the entire lake would be required before dosing can begin. An example of full-scale dosing of a lake with Phoslock® is shown in Figure 19. Once the lake has been treated, the next stage is to carry out monitoring of the success of the treatment as well as any associated consent monitoring requirements. For the flock & lock treatment, the only on-going maintenance requirement is repeat treatment of the entire lake approximately every four years.

Figure 18: Implementation pathway for active sediment capping (flock & lock) in Pegasus Lake.

Figure 19: Example of full-scale Phoslock® application in a lake.

7.2 Algal Turf Scrubber

The implementation pathway for employing algal turf scrubbers to remove nutrients and phytoplankton from Pegasus Lake water is outlined in Figure 20. The first stage is to establish the nutrient/phytoplankton removal rates as well as the hydraulic loading rate (or how much water and at what flow rate) for the ATS, using Pegasus Lake water. This will determine the size of the full-scale system required to treat the lake water to achieve sufficient nutrient removal. The estimated ATS size of 1.25 ha was based on treatment trials conducted overseas as well as Pegasus Lake's volume and the proposed lake turn-over rate of 40 days.

Once the size of the full-scale system has been determined, the next stage is to decide on the location of the system in, or around Pegasus Lake.

Based on the assumption that an ATS of 1.25 ha would be required, it is unlikely that a single full-scale system could be built alongside the lake, based on the available land surrounding the lake.

Alternative options would be to either build a floating ATS in-lake, or create several smaller units located around the lake, on the lake reserve ground. The space requirements for either a floating in-lake ATS, or several smaller ATS around the lake edge are visually represented in Figure 21.

Once the final design and location have been agreed upon, the costs for the construction and any associated resource consents can be estimated.

The next stage is the resource consent application process prior to the construction and implementation of the full-scale ATS. Following implementation of the ATS, monitoring of the performance of the system will be required initially, as well as any associated consent monitoring requirements. On-going maintenance of the system, including harvesting and disposal, or re-use, of the periphyton, will be required.

Figure 20: Implementation pathway for algal turf scrubber at Pegasus Lake.

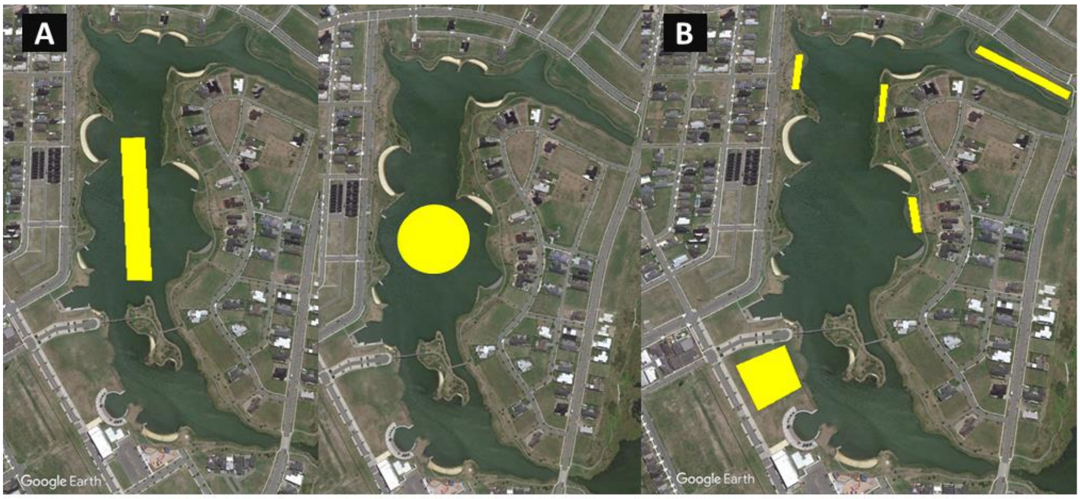

Figure 21: Examples of area required in or around the lake for full-scale algal turf scrubber either A) floating in-lake system or B) several smaller scale systems located around the lake reserve.

7.3 Artificial mixing or Aeration

The implementation pathway for the use of artificial mixing or aeration to overcome anoxia and manage cyanobacterial blooms in Pegasus Lake is outlined in Figure 22. The first stage is to determine the effectiveness of either artificial mixing or aeration as a means to prevent anoxia in the lake and / or manage cyanobacterial blooms. The results of the trial will help inform the design of the full-scale system. The second stage is to design a full-scale mixing or aeration system for Pegasus Lake and produce a cost estimate, including the associated pump house.

Obtaining resource consents for the construction and operation of the system is the next step in the pathway, followed by the actual construction and implementation. Once implemented, monitoring of the system's performance will be required initially, as well as any associated consent monitoring requirements. On-going maintenance will be required over the lifetime of the system.

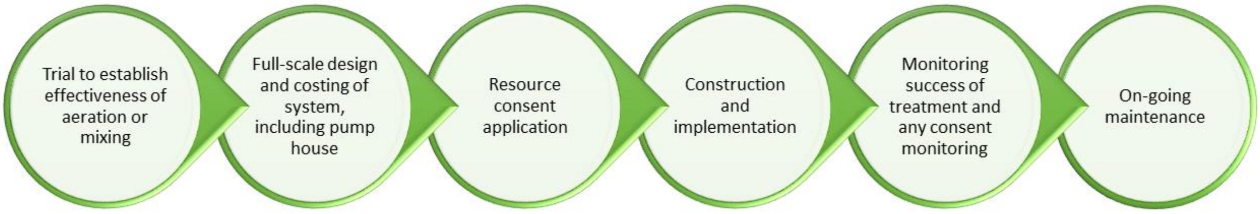

Figure 22: Implementation pathway for artificial mixing or aeration in Pegasus Lake.

7.4 Flushing

7.4.1 Surface flushing

Compared to the other treatment options, the implementation pathway is relatively short (Figure 23). If surface flushing trials demonstrate that this is a successful management technique, the next stage is to update the lake management plan detailing the procedures for operating the surface flush during the summer period. Once updated, surface flushing can be used as a management tool for cyanobacterial blooms when sufficient water has been retained in the lake leading up to the bloom.

Figure 23: Implementation pathway for surface flushing of Pegasus Lake.

7.4.2 Full flushing

The implementation pathway for full flushing of Pegasus Lake as a means of managing cyanobacterial blooms through dilution-mediated nutrient reduction is outlined in Figure 24. The first step in this pathway is to undertake trials (as described above) to establish its effectiveness.

Should the trials prove effective, sufficient water supply to provide the full lake flush needs to be sourced and cost of translocating this water to the lake estimated.

The next stage in the pathway is obtaining resource consents for the take, use and discharge of the flushing water. Should the necessary resource consents be granted, the next stage involves the construction of infrastructure associated with the water translocation. Following construction, the flushing regime can be implemented based on the results of the trials. Monitoring of the success of full lake flushing and any associated consent condition monitoring will need to be a carried out initially. On-going maintenance for the system will include upkeep of the infrastructure, adjustments of HRT should catchment water quality decline and maintenance of any erosion control measures put in place downstream of the lake discharge.

Figure 24: Implementation pathway for full flushing of Pegasus Lake.

7.5 Adaptive Management

While the adaptive management option provides no active intervention with respect to cyanobacterial bloom management, there are a number of steps that can be implemented to help better manage the lake. In addition to the current summer-time bloom monitoring and reporting programme, several options for better lake management were discussed in Section 5.5. The implementation pathway for those options is outlined in Figure 25. The first stage in this pathway is to update the lake management plan to incorporate a change in the aquatic weed management practices from chemical to mechanical, as well as a beach clearance regime to remove any weed/periphyton washup on the beach. The next stage is to investigate improving both the design and management of the two lake outlets to prevent retention of cyanobacterial biomass in the lake. Implementing outlet design and management changes is the final stage in the adaptive management implementation. On-going maintenance of both the outlets and the beaches will be required as part of the adaptive management plan.

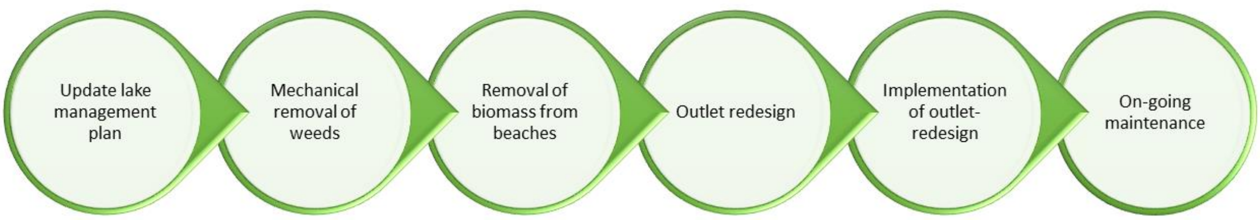

Figure 25: Implementation pathway for adaptive management of Pegasus Lake.

8.0 SUMMARY

Several questions raised during the workshop with respect to Pegasus Lake and associated cyanobacterial blooms were addressed in this report. These questions were:

1. *Once stratification becomes established is it persistent or intermittent?*

DO loggers installed near the top and the bottom of the water column from December 2017 to April 2018 confirmed that stratification was persistent in Pegasus Lake from at least early summer. Nutrient samples collected during December indicated that stratification, and associated anoxia, had been established for some time prior to the installation of the DO loggers. These results are consistent with the results from monthly water quality monitoring that has been undertaken from July 2014 to present.

2. *Are the two SolarBee® mixers having any effect on the stratification?*

DO, temperature and pH measurements taken immediately alongside the two SolarBee® mixers and at increasing distances from the mixers showed that there was no measurable effect on stratification. Results indicated that the SolarBee® mixers did not provide any localised mixing of the hypolimnion.

3. *What is the hydraulic residence time of the lake?*

Based on the volumetric calculation of the lake basin and flow measurements made during the summer period, the hydraulic retention time in the lake was 204 days.

4. *What is the current nutrient status of the lake and are nutrients being released during anoxia?*

Due to excessive nutrients and high phytoplankton biomass, the lake is currently classified as being hypertrophic. Measured high concentrations of NH_4-N in the bottom waters confirmed that dissolved nutrients are being released, via microbial breakdown of organic matter, under anoxic conditions during stratification.

The five potential cyanobacterial bloom management options are summarised in Table 11.

No single option provides a solution to both DO stratification and cyanobacterial blooms in Pegasus Lake.

Several options (active sediment capping and algal turf scrubber) provide potential nutrient management solutions, which, in turn, would reduce the amount of algal/cyanobacterial biomass in the lake and the potential for blooms to develop.

Reducing algal/cyanobacterial biomass would lead to an overall reduction in oxygen consumption by algae/cyanobacteria and heterotrophic bacteria. However, any groundwater inflow mediated DO stratification would still persist.

Artificial mixing/aeration provides a potential solution to reduce both in-lake biological and groundwater inflow mediated DO stratification. However, it is noted that there is no evidence to suggest that stratification of the water column was the cause of cyanobacterial bloom development in Pegasus Lake.

None of the solutions proposed can guarantee prevention of cyanobacterial blooms in the lake.

While there isn't a single option, or combination of options, that can guarantee to overcome both cyanobacterial blooms and DO stratification in Pegasus Lake, several options could be combined in an attempt to provide more effective management of the lake. The artificial mixing/aeration option could be combined with either flushing (either surface, or full lake) or Algal Turf Scrubber but not Active Sediment Capping, as the capped layer needs to remain undisturbed.

Validation of any management option, with the exception of adaptive management, would need to be carried out in the Pegasus environment using pilot-scale trials to determine the likelihood of success for each option.

Table 11: Summary of the five potential management options for Pegasus Lake.

Option	Overcome dissolved oxygen stratification	Prevent blooms	Indicative Capital cost ($)	Indicative annual operational cost ($)
Active sediment capping	No*	Potential	$350,000	None but cap replaced every 4 years.
Algal turf scrubber	No*	Potential	$850,000	$30,000
Artificial mixing or aeration	Potential	No	$1,050,000	$20,000
Flushing	No	No	$5,000,000 +	$20,000
Adaptive Management Regime	No	No	Not estimated	Not estimated

Note: * = these options could reduce the biological mediated DO consumption thereby reducing the overall DO stratification, but they are unlikely to prevent DO stratification driven by the inflow groundwater.

9.0 REFERENCES

Ashley KI, Hall KJ 1990. Factors influencing oxygen transfer in hypolimnetic aeration systems. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie 24: 179-183.

Beca 2007. Pegasus Lake Design Report. Appendix 3 Ground water modelling report. Prepared for Pegasus Town Limited.

Burchard, H., Hofmeister R 2008. A dynamic equation for the potential energy anomaly for analysing mixing and stratification in estuaries and coastal seas. Estuarine Coastal and Shelf Science. 77: 679-687.

Burris VL, Little JC 1998. Bubble dynamics and oxygen transfer in a hypolimnetic aerator. Water Science and Technology 37: 293-300.

Clarke KR, Warwick RM 2001. Changes in Marine Communities. An Approach to Statistical Analysis and Interpretation, second ed. Plymouth Marine Laboratory, UK.

Collins GB, Weber CT 1978. Phycoperiphyton (algae) as indicators of water quality. Transactions of the American Microscopical Society 97: 36-43.

Cooke GD, Welch EB, Peterson SA, Nichols SA 2005. Restoration and management of lakes and reservoirs. Boca Raton, CRC Press, 616p.

Davies-Colley RJ 1988. Mixing depths in New Zealand lakes. New Zealand Journal of Marine and Freshwater Research 22(4): 517-528.

Davies-Colley RJ, Vant WN, Smith DG 1993. Colour and clarity of natural waters: science and management of optical water quality. Ellis Horwood Limited. 310 pp.

Dodds WK 2003. Misuse of inorganic N and soluble reactive P concentrations to indicate nutrient status of surface waters. Journal of the North American Benthological Society 22(2):171-81.

Franklin PD 2014. Dissolved oxygen criteria for freshwater fish in New Zealand: a revised approach. New Zealand Journal of Marine and Freshwater Research 48(1): 112-126.

Gibbs M, Champion P 2013. Opportunities to address water quality issues in Lkaes Wiritoa and Pauri. NIWA Envirolink Report 1196. HAM2013-053.

Gibbs MM, Hickey CW, Özkundakci D 2011. Sustainability assessment and comparison of efficacy of four P-inactivation agents for managing internal phosphorus loads in lakes: sediment incubations. Hydrobiologia, 658(1):253-75.

Golder 2017. Cyanobacterial bloom management strategies for Pegasus Lake. Report prepared by Golder Associates (NZ) Limited for Todd Property Pegasus Town Limited dated November 2017.

Golder 2018. Annual Report for Water Quality in Pegasus Lake and Eastern Conservation Management Area 2015-2016. Report prepared by Golder Associates (NZ) Limited for Todd Property Pegasus Town Limited dated April 2018.

Green JD 1975. Physico-chemical features of Lake Ototoa, a sand-dune lake in northern New Zealand. New Zealand Journal of Marine and Freshwater Research 9 (2): 199-222.

Gouvernement du Québec 2008. Guide d'échantillonnage à des fins d'analyses environnementales : Cahier 7 – Méthodes de mesure du débit en conduit ouvert, Centre d'expertise en analyse environnementale du Québec, 248 p

Haghseresht F, Wang S, Do DD 2009. A novel lanthanum-modified bentonite, Phoslock, for phosphate removal from wastewaters. Applied Clay Science, 46(4):369-75.

Hamilton DP, Salmaso N, Paeral HW 2016. Mitigating harmful cyanobacterial bloos: strategies for control of nitrogen and phosphorus loads. Aquatic Ecology 50: 351-366.

Kelly, D 2015. Feasibility assessment for destratification mixing and hypolimnethic aeration in the Maitai Reservoir. Cawthron Report No. 2720, prepared for Nelson City Council.

Liu, J. Vyverman, W 2015. Differences in nutrient uptake capacity of the benthic filamentous algae Chadophora sp., Klebsormidium sp., and Psedanabaena sp. Under varying N/P conditions. Bioresource Technology 179: 234-242.

Low-Décarie E, Fussmann GF, Bell G 2014. Aquatic primary production in a high-CO_2 world. Trends in Ecology & Evolution, 29(4):223-32.

Lürling M, van Oosterhout F 2013. Controlling eutrophication by combined bloom precipitation and sediment phosphorus inactivation. Water research 47(17): 6527-6537.

Lürling M, Waajen G, de Senerpont Domis LN 2016. Evaluation of several end-of-pipe measures proposed to control cyanobacteria. Aquatic Ecology 50(3):499-519.

McGinnis D 2000. Predicting oxygen transfer in hypolimnetic oxygenation devices. MSC thesis, Virginia Polytechnic Institute and State University.

McIntosh J 2007. Phsolock application – Lake Okareka Final Report. Report by Environment Bay of Plenty 2007/23.

McQueen DJ, Lean DRS (1989). Hypolimnetic aeration: An overview. Water Pollution Research 21: 205-217.

Ministry for the Environment 2009. New Zealand Guidelines for Cyanobacteria in Recreational Fresh Waters– Interim Guidelines. Wellington: Ministry for the Environment and Ministry of Health.

Noyma NP, De Magalhães L, Miranda M, Mucci M, van Oosterhout F, Huszar VL, Marinho MM, Lima ER, Lürling M 2017. Coagulant plus ballast technique provides a rapid mitigation of cyanobacterial nuisance. PloS one, 12(6):e0178976.

Simpson JH 1981. The shelf-sea fronts: Implications of their existence and behaviour. Philosophical Transactions of the Royal Society of London. Series A, Mathematical and Physical Sciences 302 (1472):531-546.

Sutherland, DL 2005. Potamogeton crispus literature review. NIWA internal report 52A.

Sutherland DL, Craggs RJ 2017. Utilising periphytic algae as nutrient removal systems for the treatment of diffuse nutrient pollution in waterways. Algal Research, 25 (496-506).

Sutherland DL, Turnbull MH, Broady PA, Craggs RJ 2014. Effects of two different nutrient loads on microalgal production, nutrient removal and photosynthetic efficiency in pilot-scale wastewater high rate algal ponds. Water Research, 66:53-62.

Sutherland DL, Montemezzani V, Howard-Williams C, Turnbull MH, Broady PA, Craggs RJ 2015. Modifying the high rate algal pond light environment and its effects on light absorption and photosynthesis. Water Research, 70:86-96.

Sutherland DL, Turnbull MH, Craggs RJ 2017. Environmental drivers that influence microalgal species in fullscale wastewater treatment high rate algal ponds. Water Research, 124:504-12.

Visser PM, Ibelings Bw, Bormans M, Huisman J 2016. Artificial mixing to control cyanobacterial blooms: a review. Aquatic Ecology 50: 423-441.

WaterNZ 2013. Standard for the supply of polyaluminium chloride for use in water treatment. Second Edition. 14pp.

Welch EB, Cooke GD 1999. Effectiveness and longevity of phosphorus inactivation with alum. Lake and Reservoir Management, 15(1):5-27.

Wells RD, Sutherland DL 2002. Evaluating the efficiency of diquat on the submerged macrophytes in the Avon River, Christchurch. NIWA Client Report, CHC2001.

Zamparas M, Gianni A, Stathi P, Deligiannakis Y, Zacharias I 2012. Removal of phosphate from natural waters using innovative modified bentonites. Applied Clay Science, 62:101-106.

Zic K, Stefan HG 1990. Lake / reservoir destratification induced by bubble plumes. Model description and user manual for use with Westex and CE-Qual- R1 dynamic Lakes/reservoir models. US Army Corp of Engineers. St Anthony Falls Hydraulic Laboratory, Project Report No. 313. 41 p.

APPENDIX A

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APPENDIX B

Dominant phytoplankton species

Table 12: Frequently occurring phytoplankton species in Pegasus Lake during the period January to April 2018.

Code	Phyla	species
Cer hir	Dinoflagellates (Dinoflagellata)	*Ceratium hirundinella*
Clost	Green algae (Chlorophyta)	*Closteriopsis* sp.
Clos acic	Desmids (Zygnemophyceae)	*Closterium aciculare*
Crypt	Golden-brown algae (Cryptophyceae)	*Cryptomonas* sp.
Doli circ	Cyanobacteria	*Dolichospermum circinalis*
Flag	Flagellates/Unicells	Flagellates/Unicells
Gym	Dinoflagellates (Dinoflagellata)	*Gymnodinium* sp.
Mallo	Golden-brown algae (Cryptophyceae)	*Mallomonas* sp.
Sph sch	Green algae (Chlorophyta)	*Sphaerocystis schroeteri*
Trac volv	Euglenoids (Euglenoidea)	*Trachelomonas volvocina*

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