Data analysis and literature review to inform Lake Hood water quality management

Prepared for Ashburton District Council and Ashburton Aquatic Park Charitable Trust

April 2024

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NIWA CLIENT REPORT No: 2024068CH
Report date: April 2024
NIWA Project: AAP24502

Revision	Description	Date
Version 1.0	Report sent to client	27 March 2024
Version 1.1	Revised report sent to client	17 April 2024

Quality Assurance Statement		
	Reviewed by:	David Plew
	Formatting checked by:	Rachel Wright
	Approved for release by:	Scott Larned

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Executive summary

The Lake Hood Water Quality Task Force (hereafter the Task Force) approached NIWA to provide information regarding possible control options to reduce the risk of future cyanobacteria blooms in Lake Hood. The lake is man-made, owned by Ashburton District Council (ADC), and fed by and drains to the Ashburton River. The lake first opened in 2002 and experienced its first cyanobacteria bloom in early 2023 and its second bloom in January 2024.

The Task Force requires analysis and interpretation of existing lake and inflow water quality data and literature reviews to summarise important characteristics of the cyanobacteria species identified in the lake, determine published relationships between water column phosphorus concentrations and cyanobacteria growth, and review options for the control of cyanobacteria. The Task Force intends to use this information to aid in decision making regarding actions to reduce the risk of cyanobacteria blooms in Lake Hood.

In Lake Hood, DRP has been low at the outlet (< 0.005 mg/L) but much higher in the intake and Carters Creek (used to be as high as ~0.040 mg/L but more recently usually < 0.030 mg/L). The data and literature suggest that the bioavailable P concentration must be at least less than ~0.020 mg/L to reduce growth, but this may not eliminate *Dolichospermum* blooms in Lake Hood. Targeted investigations and experiments would be needed to confirm P limitation and refine the estimated threshold of ~0.020 mg/L P to restrict growth of the *Dolichospermum* species in Lake Hood.

The primary objective was to aim to identify the drivers contributing to cyanobacteria blooms, based on the available data.

Based on the supplied data and some additional data that we obtained, we found the following.

- While the annual and January mean air temperatures have increased since 2010, similar changes were not observed in lake water temperature. However, temperatures as high as 24.3°C (in the lake outlet) and regularly reaching or approaching 20°C indicate conditions favouring cyanobacteria growth. Generally, differences between surface and bottom temperatures were < 0.5°C, indicating vertically mixed conditions, but a 1.4°C difference was observed at a location near the lake intake on 16 January 2024, indicating that stratification occurs at least periodically.

- No conclusions can be drawn from wind speed observations. Assuming that the Lake Hood canals are wind-shaded by structures and trees, changes in wind speed are unlikely to have a great effect on vertical mixing in the canals.

- Dissolved oxygen concentrations were generally good (> 5 mg/L), except on two occasions (29 January 2019: 2.93–3.65 mg/L in Carters Creek, the lake intake, and the outlet; 30 March 2021: 2.35 mg/L in the intake and 2.97 mg/L in the outlet). A dissolved oxygen profile obtained on 16 January 2024 indicates hypoxia (< 2 mg/L dissolved oxygen) during stratification. Hypoxia can lead to phosphorus release from the lake sediments. Results from aeration trials using an aerator setup in a test canal indicate that the aeration setup did not prevent water column stratification and occurrence of hypoxia.

- pH values appear to trend upward over the 2014–2023 record. Values > 10 or approaching 10 enable sediment release of phosphorus, regardless of oxygen levels. Maximum field values > 11 were observed in October 2021 at the lake intake and well 2, and pH was 10 in October 2023 at the lake outlet. Laboratory measurements were lower (always < 10) than field measurements, indicating some discrepancy between the two measurements, perhaps due to sample handling and holding times or other differences in the pH measurement methods.

- Low turbidity measurements and the fact that the lake is shallow suggest that light usually reaches the lake bottom and does not limit algal growth.
- Sediment phosphorus content values at the Trial Canal and lake intake sites were moderate (710 and 820 mg/kg dry mass, respectively) and could contribute to algal blooms.
- Groundwater total nitrogen has increased in well 2 from 2015 to 2023.
- Total phosphorus values were highest in the lake intake and Carters Creek inflows.
- The highest total biochemical oxygen demand values were found in the lake outlet.
- The highest chlorophyll *a* values (> 0.04 mg/L) were recorded in the lake outlet, while values were generally < 0.02 mg/L at the lake intake and in Carters Creek.
- *Dolichospermum* is the dominant cyanobacteria genus identified in samples collected in 2023 and 2024.

Based on a literature review on cyanobacteria characteristics, we note the following key cyanobacteria characteristics for managing blooms:

- *Dolichospermum* blooms have been associated with toxins.
- *Dolichospermum* can regulate its buoyancy and thus move vertically in the water column to optimise its access to light and nutrients. This can lead to shading of non-nuisance phytoplankton species.
- *Dolichospermum* can fix nitrogen from the atmosphere, giving it another advantage over other phytoplankton.
- *Dolichospermum* produces specialist cells called akinetes, which are spores that can propagate quickly. "Seed banks" made up of akinetes allow *Dolichospermum* to survive harsh conditions and cause blooms when conditions (e.g., light, nutrients, temperature) are right.

Based on a literature focusing on lake management options specifically intended to limit cyanobacteria blooms in Lake Hood, we suggest consideration of flushing, nutrient controls in inflows (esp. Carters Creek and ideally also in the intake culvert), sediment capping, and potentially sonication. The options we considered are summarised in Table 0-1.

Table 0-1: Summary of options, their purpose, and associated risks for controlling cyanobacteria in Lake Hood. Green options are considered potentially feasible for Lake Hood. Red options are not recommended for Lake Hood. Uncoloured options require further investigation before recommendations regarding their application to Lake Hood can be made.

Intervention	Target/purpose	Risks
Physical controls		
Hydraulic flushing, inflow diversion	Increase flow through the lake and thus decrease the hydraulic residence time, which would flush blooms out of the lake faster. Records show that there were only eight days in a 92-day period on which water was taken. More frequent water intake times could reduce the residence time in the lake but likely not (or only minimally) in the canals. A secondary outlet could be useful in rapidly releasing water from the main lake and enhancing circulation. Divert inflowing water from Carters Creek to better flushed parts of the lake.	While the main lake may profit from this, the canals may still not have enough water movement and keep blooms trapped in areas with longer water residence times. Surface scum formation may be prevented but cyanobacteria blooms are unlikely to be prevented just by creating another outlet. A nutrient-rich diverted inflow could negatively affect the new receiving environment.
Phytoplankton harvesting	Water filtration Withdraw water from near the surface rather than the lake bottom to remove cyanobacteria concentrated at the surface.	Water filtration may not be cost-effective and will not prevent bloom formation. The lake is quite shallow and easily mixed by wind, which means that cyanobacteria likely only form surface scums during calm conditions. This will not prevent bloom formation.
Artificial destratification	Install aerators to increase vertical mixing and prevent stratification and deoxygenation of bottom waters and thus P release from the sediments.	This could be promising but would only affect the sediment P load to the lake. If blooms are primarily fuelled by external nutrient sources (e.g., Carters Creek or groundwater), then this method is unlikely to be effective. In addition, if this is not timed well, then nutrients released from the sediments during stratification would be mixed to the surface and actually fuel blooms.
Aeration/oxidation	Increase dissolved oxygen to avoid hypoxia and reduce phosphorus release from the sediments. Phosphorus fuels algal growth.	This is considerably more expensive than destratification and Lake Hood is likely too shallow for this to work well.
Nanobubbles	Produce very small oxygen bubbles (nanobubbles), to oxygenate the water and sterilise and/or kill cyanobacteria and deactivate toxins.	The process has not been well defined and this is currently an expensive option.

Intervention	Target/purpose	Risks
Drawdown	Reduce the lake water volume or even entirely empty the lake to expose sediment and dry it out. This would eliminate seed populations of cyanobacteria.	This may be cost-effective but not aesthetically pleasing and would disrupt recreational use of the lake for weeks to months at a time. It would be harmful to fish and desirable macrophyte communities.
Dredging	Remove lake sediments that can release P during thermal stratification.	This does not affect external nutrient loading from Carters Creek and groundwater that may fuel blooms.
Benthic barriers	Apply clay, silt, sand, and gravel from external sources to bury the surface nutrient-enriched sediment.	This does not affect external nutrient loading from Carters Creek and groundwater that may fuel blooms.
Sonication	Break cyanobacteria cells using sonic pressure waves to rupture the gas vacuoles in the cells. This could be very effective in the summer. Reseeding of populations will be prevented.	This does not remove the driver of the blooms. This would require repeated treatments.
Chemical controls		
Hydrogen peroxide	Liquid application of this sterilising agent to lyse cyanobacteria cells.	Cyanobacteria blooms could return after a few weeks or months and repeated application may be required.
Flocculation and sediment capping	Apply a metal salt (alum or Phoslock®) to prevent sediment P release. Alum is also a flocking agent that can remove a cyanobacteria bloom in hours by flocculation and settling it to the sediment, clearing the water column. Alum floc on the sediment surface can strongly suppress germination of the algae spores (“seed banks”) in the sediments.	Multiple or regular application may be required.
Phosflow	Apply a bead form of Phoslock® in pouches that can be placed in the inflows (Carters Creek and Ashburton River intake culvert) to reduce inflow phosphorus concentrations.	The number of pouches required and disposal and pouch replacement costs need to be calculated. This may become a long-term commitment unless upstream phosphorus controls are put in place.
Algicides	Apply chemical compounds (usually copper-based) to kill cyanobacteria.	This can result in unwanted ecotoxicological effects or secondary pollution.
Biological controls		

Intervention	Target/purpose	Risks
Weed harvesting	Remove weeds and thus nutrients bound up in those weeds from the lake. This could also prevent anoxia in bottom waters.	If a hired weed harvester that is used elsewhere were used periodically, there is a risk of introducing fragments of other weeds to Lake Hood. Nutrient loads from inflows and the sediments (in anoxic conditions) could still fuel algal blooms.
Biomanipulation	Introduce new bacteria species to either outcompete cyanobacteria for nutrients or directly kill cyanobacteria.	With any new species introduction, there is risk of extreme perturbation of the lake ecosystem (e.g., one introduced species becomes dominant and outcompetes desirable native species) and unwanted consequences. Controlled laboratory and field trials are required before any such approach is considered for Lake Hood.
Floating wetlands	Construct floating wetlands that remove contaminants, especially nitrogen from the lake.	This could be an aesthetically pleasing option but restrict some recreational use of the lake. It may be difficult to quantify the effectiveness of nutrient removal by floating wetlands.

Before implementation of any large-scale control efforts, we recommend the following next steps:

1. Modelling
    a. A **hydrodynamic model** would enable estimation of residence times in different parts of the lake, i.e., the canals and the main lake, which will differ. A key preliminary step is to calculate the lake **water balance** from inflow, outflow, rainfall, and water level data spanning at least one year but ideally several years.
    b. A linked catchment and lake **water quality model** would quantify the nutrient loads from each source and estimate sediment nutrient fluxes over time, allowing for determination of the main nutrient source driving cyanobacteria blooms.
    c. **Scenario modelling** using the linked models would allow for testing of different management options (e.g., reduced nutrient loads, sediment capping) before implementation.
2. Data collection for model development
    a. Regular (at least monthly) **water column P and N sampling in the canals** where blooms are most likely to form.
    b. **Field measurements of fluxes of P from the sediment** for comparison with inflow nutrient loads.
    c. Controlled laboratory or mesocosm **experiments** using cyanobacteria from Lake Hood to describe the growth rate as a function of N and P concentration.
3. Ongoing water quality monitoring is required to determine the effectiveness of any mitigation measures.

1 Introduction

1.1 Background

Lake Hood is a man-made lake owned by Ashburton District Council (ADC). The lake is fed by the Ashburton River when flows exceed the minimum required for the river. The lake was commissioned in 2002 and experienced its first cyanobacteria bloom in early 2023. It was subsequently closed to contact recreation from 16 March to 15 May 2023 (McCracken et al. 2023). Another bloom was first detected in early January 2024 and the lake is currently subject to controls under the Interim 2009 *New Zealand Guidelines for Cyanobacteria in Recreational Fresh Waters* (Wood et al. 2009).

Following the first cyanobacteria blooms, the Lake Hood Water Quality Task Force (hereafter the Task Force) was assembled to determine and implement possible solutions that would reduce the risk of future cyanobacteria blooms. The Task Force was formed in May 2023 and members represent the following:

- the Joint Venture Committee, which is developing the lake extension and associated subdivisions and currently managing the lake and lake surrounds,
- residents,
- the Ashburton Aquatic Park Charitable Trust, which developed the original lake and is a Joint Venture partner, and
- the Ashburton District Council, which owns the lake, lake park surrounds, and future lake development land.

Lake Hood was constructed on riverbed gravels and first opened for recreational use in April 2002. The original lake area, including canals, is approximately 85 hectares (McCracken et al. 2023). From 2010, the lake was expanded to the north, adding approximately 26 hectares, including canals, to the original lake. Two thirds of the original lake bottom was sealed (covered) with clay excavated from a nearby site and the remaining lake bottom receives groundwater inflows.

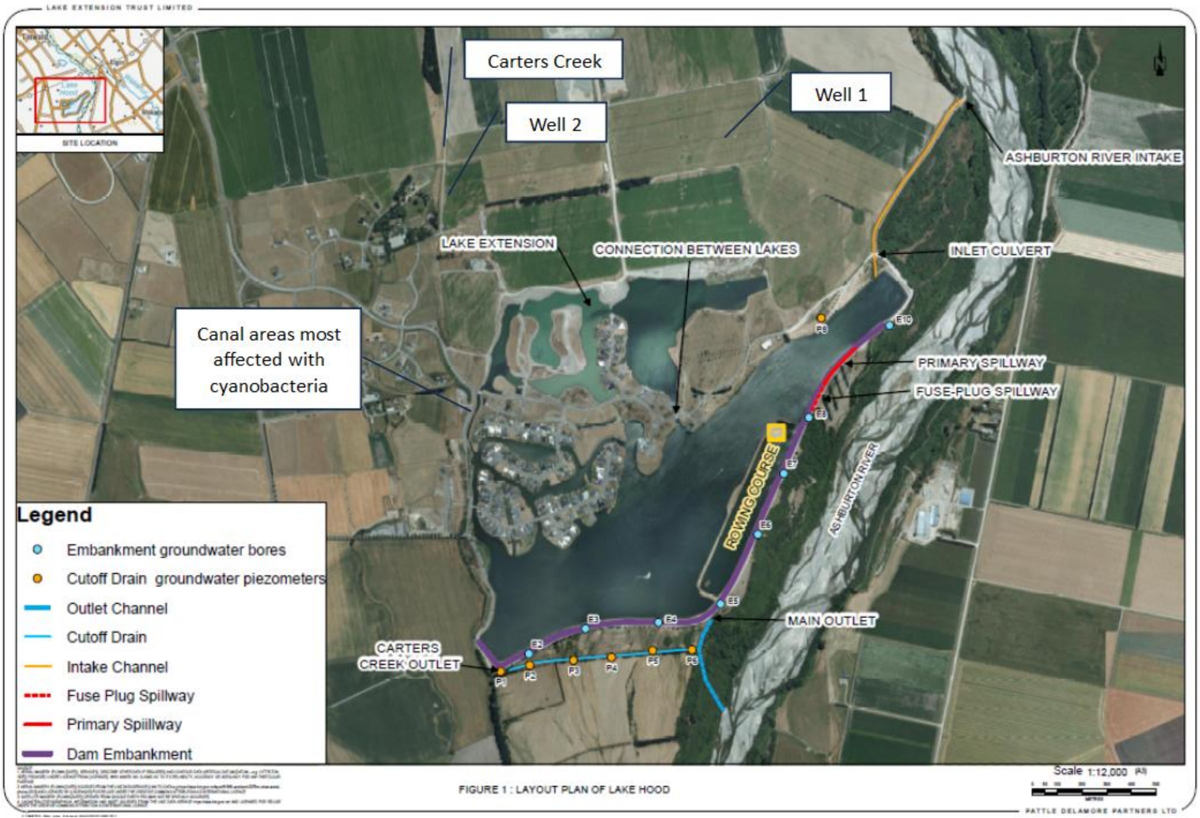

Figure 1-1: Map of Lake Hood and its Ashburton River intake and main outlet, as well as sampling sites and other features of interest. Map provided by the Task Force. The inlet sampling location is at the intake weir. We assume that the Carters Creek sampling location is in the creek before it enters the northwest lake extension and the lake outlet sampling location is in the lake itself near the outlet channel.

1.2 Objectives

The Task Force requires:

1. analysis and interpretation of existing lake and inflow water quality data, and
2. literature reviews to:
   - summarise important characteristics of the cyanobacteria species identified in the lake,
   - determine published relationships between water column phosphorus concentrations and cyanobacteria growth, and
   - review options for the control of cyanobacteria.

The Task Force intends to use this information to aid in decision making regarding actions to reduce the risk of cyanobacteria blooms in Lake Hood.

The primary objective is to identify the drivers contributing to cyanobacteria blooms, based on the available data.

1.3 Scope

The scope of the present work was to analyse and interpret existing lake and inflow water quality data, and to conduct the literature reviews described above to summarise important characteristics of the cyanobacteria species identified in the lake, relationships between water column phosphorus concentrations and cyanobacteria growth, and options for the control of cyanobacteria. The scope excludes formal trend analyses of time-series data, comprehensive cost/benefit analysis, and specific recommendations regarding cyanobacteria control options for Lake Hood.

2 Methods

The following tasks were completed in this study.

1. Examine the Lake Hood water quality database containing monthly consent compliance data collected since the lake was commissioned in 2002, and weather data from the Ashburton Airport weather station, to determine what changes in the lake and surrounding environment may have led to the blooms experienced in 2023 and 2024 to assist the Task Force in prioritising possible solutions to control and reduce the risk of future blooms.

2. Research the characteristics of the cyanobacteria species identified in the laboratory reports, with the purpose of informing the Task Force on which mitigation measures should be investigated to reduce the risk of future blooms. This includes information on the relationship between phosphorus levels in the lake and the potential for cyanobacteria blooms, with the aim of providing advice on the level of phosphorus removal needed to reduce the risk of future blooms.

3. Carry out a literature survey on options for the control of cyanobacteria and summarise these options. This will include chemical controls and consumption of the current toxin-producing cyanobacteria by other, beneficial species of bacteria, which are solutions being proposed by various companies wishing to assist the Task Force.

To complete task 1, we used data provided by the Task Force and plotted time series showing potential temporal and spatial variation in temperature, dissolved oxygen, nutrient, and cyanobacteria concentrations. While formal trend analyses of these time series were beyond scope, we did both visually inspect data and fit linear regressions to see if any trends were apparent. In the plots that follow in Section 3, we only show fitted linear regressions where the p value for the slope is < 0.05 (the p value indicates the probability that the slope is not significantly different from zero, $p < 0.05$ indicates the probability that there is a non-zero slope is at least 95%). We also assessed weather conditions using data from the Ashburton Aero weather station. NIWA did not collect any additional data as part of this work.

The following files were provided to NIWA.

1. A spreadsheet containing monthly consent compliance data since the lake was commissioned.

2. Test reports on cyanobacteria samples collected during the 2023 and 2024 blooms.

3. Test reports on lake sediment samples collected in September 2023.

4. Test reports on lake water quality at selected points that commenced in 2023 and are ongoing.

5. A spreadsheet containing weekly monitoring results from an aeration trial under way in one of the canals.

6. T&T reports on the expansion of the original lake.

7. NIWA reports on lake weed control.

8. Environment Canterbury reports on Carters Creek water quality.

The water quality database contains information from ~20 sites, including piezometer data, field measurements and water quality data including pH, water temperature, conductivity, water level, dissolved oxygen, tube clarity, turbidity, total suspended solids, total nitrogen, total ammoniacal nitrogen, nitrite nitrogen, nitrate nitrogen, total Kjeldahl nitrogen, dissolved reactive phosphorus, total phosphorus, total biochemical oxygen demand, chlorophyll *a*, faecal coliforms, *E. coli*, total dissolved nitrogen, total dissolved phosphorus, and volatile suspended solids (2015–2023 with data gaps).

Cyanobacteria test reports are four Cawthron laboratory reports (pdf files) from 2023. In addition, we directly had access to one NIWA test report from 2024.

We received spreadsheets for the "4 Point Water Tests" for three dates in 2023 and one date in 2024. These contain field measurements, climatic conditions, sediment and surface water monitoring results for four sites in the lake.

Aeration test data were provided in the form of 21 Excel spreadsheets that include canal aeration trial data (temperature, dissolved oxygen).

The following reports were provided to NIWA:

- Howard-Williams (short communication, 2017) Review of assessment of environmental effects
- McCracken et al. (2023) Briefing paper on cyanobacteria bloom risk
- Environment Canterbury (2019) Sources of nitrate in groundwater in the Tinwald, Ashburton area
- Sutherland et al. (2014) Lake Hood Aquatic Weed Survey 2014
- Tonkin & Taylor Ltd (2014) Lake Hood review of water management
- Tonkin & Taylor Ltd (2008) Implications for Water Quality – Assessment of the Extension to Lake Hood
- Tonkin & Taylor Ltd (2008) Lake Hood water quality monitoring and ecology
- NIWA short report (Clayton 2012) on lake inspection in January 2012

We reviewed the existing reports to better understand the history of Lake Hood and what has been monitored and recommended to date. Next, we reviewed the data listed above and produced graphics to qualitatively identify trends in the time series.

3 Water quality observations

Time series of water quality data are required to determine natural conditions and the effects of management interventions. This section serves to identify background conditions and changes in the measured variables that may co-occur with or contribute to cyanobacteria blooms in Lake Hood.

Lake Hood operates under consent CRC162113. Key requirements of the consent are as follows.

- The rate at which water is discharged from Lake Hood to the Ashburton River via the Lake Hood Outlet Drain, Bayliss Stream and Carters Creek shall not exceed a combined rate of 6.5 m^3/s for the purposes of maintaining water levels and water quality management.
- A representative sample of water shall be collected during the last five working days of each month and tested for total phosphorus (TP), nitrate-nitrogen (NO_3-N), suspended solids (SS), biochemical oxygen demand (BOD), pH, *Escherichia coli* (*E. coli*), total nitrogen (TN), dissolved reactive phosphorus (DRP), and temperature at the following locations:
  - the outlet of the existing outlet structure, being representative of any surface discharge from the lake
  - Carters Creek upstream of Lake Hood
  - Ashburton River intake canal immediately downstream of the Ashburton River intake structure
  - Groundwater up-gradient of Lake Hood in one of three bores (sampled quarterly only for TN, NO_3-N, and DRP).
- The discharge from Lake Hood (representing lake surface water) shall meet the following water quality standards:
  - SS < 50 mg/L
  - BOD < 5 mg/L
  - pH 6.5–8.5
  - *E. coli* rolling 4-monthly median < 126/mL
  - TN ≤ 110% of the groundwater or surface water inflow concentration or 3 mg/L (whichever is greater, based on the average monitoring result of the preceding 12-month period)
  - DRP < 0.3 mg/L
  - temperature ≤ 3°C above or below the temperature of the receiving water at the time of discharge.

3.1 Water level

Time series of water level at Carters Creek, lake intake and outlet, and in wells 1 and 2 (Figure 3-1a,b) do not show any significant trend from 2015 to 2023. Electrical conductivity has significantly decreased since 2015 only at the lake intake and well 1 ($p < 0.05$, Figure 3-1c).

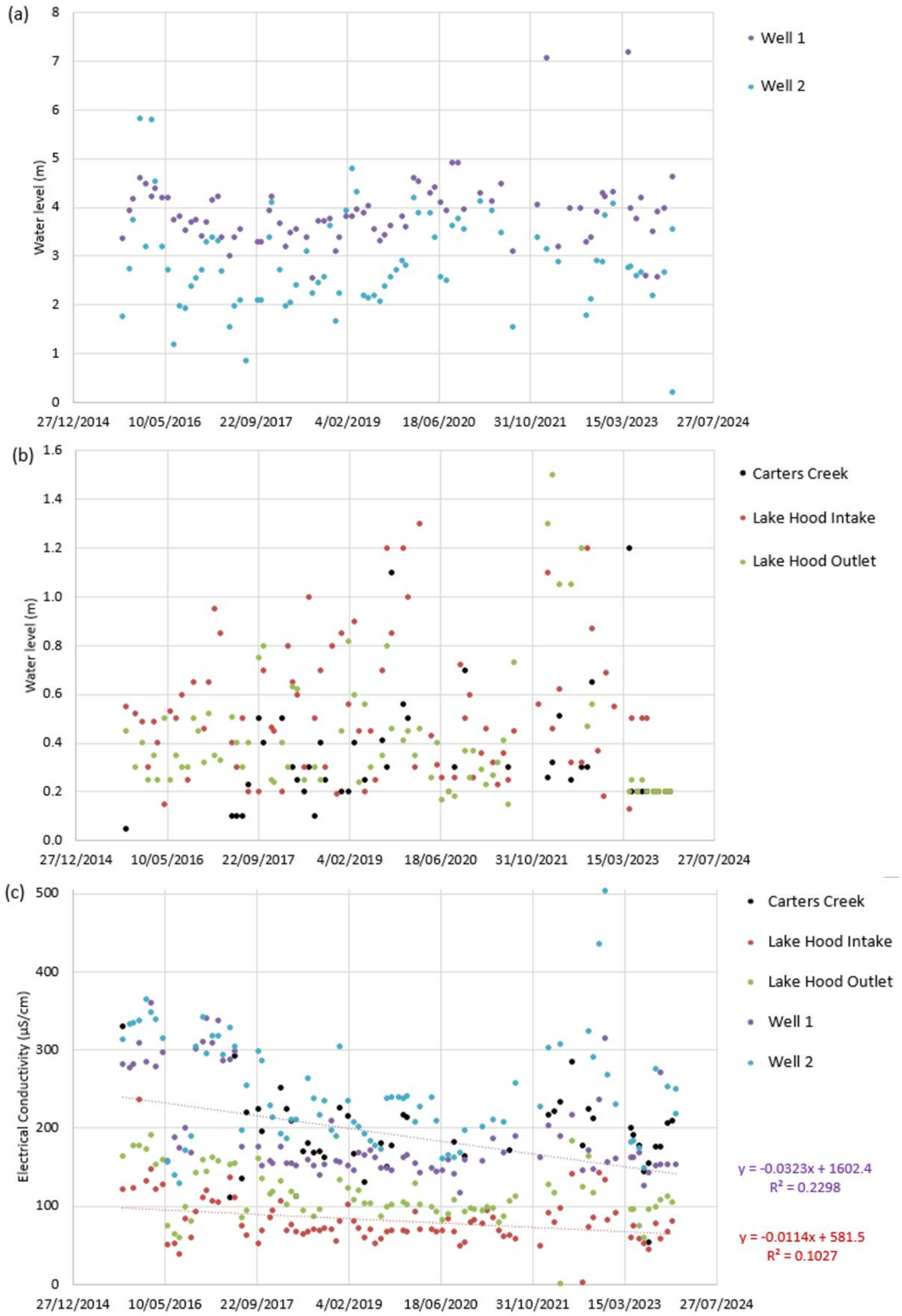

Figure 3-1: Time series of water level at (a) well 1 and 2, (b) Carters Creek, lake intake and outlet, and (c) electrical conductivity at five locations. Dashed lines are linear regressions, only shown for significant trends ($p < 0.05$). Equations for each location are marked by colour. Note that two outliers are not shown: 5.5 m at Lake Hood intake on 25 October 2016 and 3.0 m at Carters Creek on 27 February 2018.

3.2 Meteorological conditions

High temperatures can drive cyanobacterial growth, and thus air temperature can have a strong impact on cyanobacteria population dynamics. We obtained available air temperature data for Ashburton airport (2007–2023) at an elevation of 88 m above mean sea level (masl) from the CliFlo database (cliflo-niwa.niwa.co.nz) and considered time series of air temperature and wind speed.

While time series of annual mean air temperature increased significantly (Figure 3-2a), mean air temperature in January (as a representative summertime period) does not show any significant trend (although it increased). The mean air temperature in June (as a representative wintertime period) also increased significantly.

High wind speed can be an indicator of mixing in the lake and can reduce the risk of cyanobacteria bloom formation in the lake. The annual mean and summer wind speeds decreased from 2007 to 2023, but their trends were not significant (Figure 3-2b).

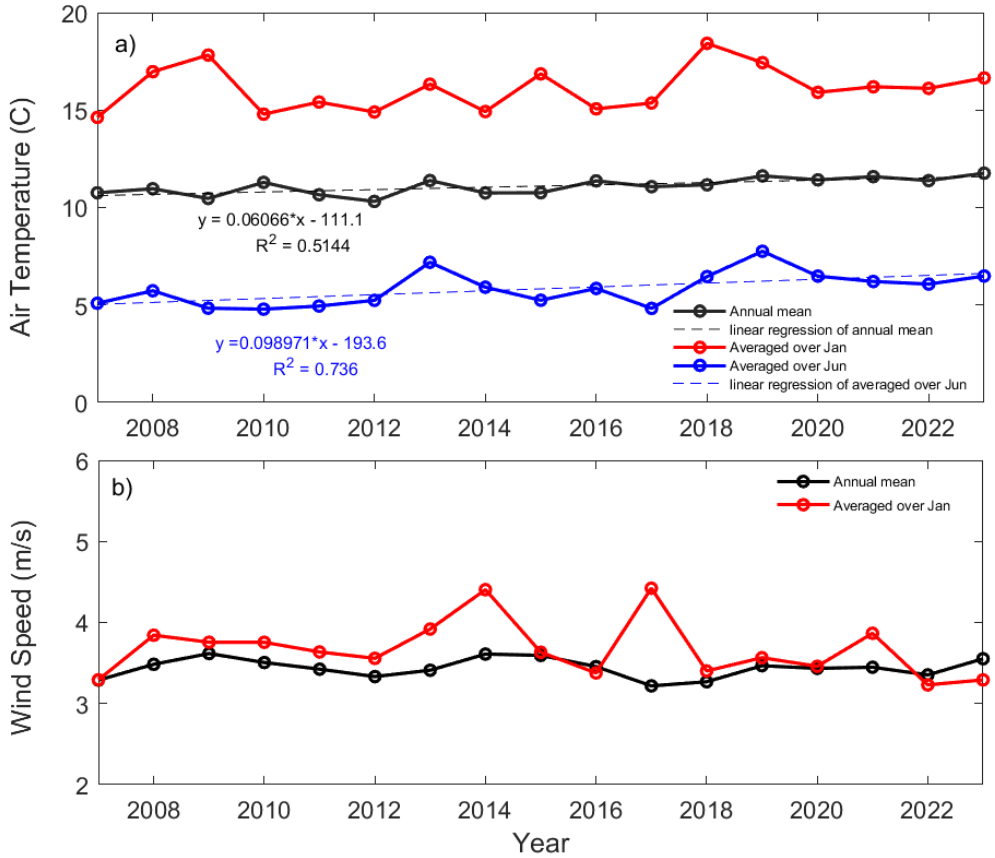

Figure 3-2: Mean annual (black) and averaged over January (red) air temperature (a) and wind speed (b) near Lake Hood. Dashed lines are linear regressions starting in 2001. Dash-dotted lines show linear regressions ($p < 0.05$) starting in 2010. Data from Ashburton airport (2007–2023) at 88 masl. Obtained from cliflo-niwa.niwa.co.nz.

3.3 Water temperature and dissolved oxygen

Figure 3-3 shows time series of surface water temperature measurements in 2015–2023 at five sites: Carters Creek, lake intake, lake outlet, well 1, and well 2. While the water temperature varied seasonally between 3.8 and 24.3°C, there was no significant trend. The highest recorded temperature (24.3°C) was at the outlet on 30 January 2018. A slight increase in the coldest winter water temperatures (i.e., minimum values each year) since 2017 corresponds with the warming winter air temperature (Figure 3-2a). The warmer winter temperature in recent years may have contributed to cyanobacteria blooms in the summer by enabling spores (or "seed banks") to survive or thrive.

Measured profiles of water temperature at the outlet (max 3 m depth), intake (max 1.9 m depth), New Extension Near Model Boat Club Dock (max 1.5 m depth), and Test Canal Mid-point (max 1.5 m depth) on 16 January 2024 (when the wind was moderate) and on 13 December 2023 (moderate wind condition) show a < 0.5°C difference between the surface and bottom water temperature (not shown here). The only exception was at the lake intake on 16 January 2024, when a difference in the water temperature of 1.4°C (Figure 3-4a) was observed. These measurements indicate that periods of stratification occur in the lake; however, this may be an effect of warmer incoming water sitting above cooler lake water (or sometimes vice versa, if incoming water is cooler).

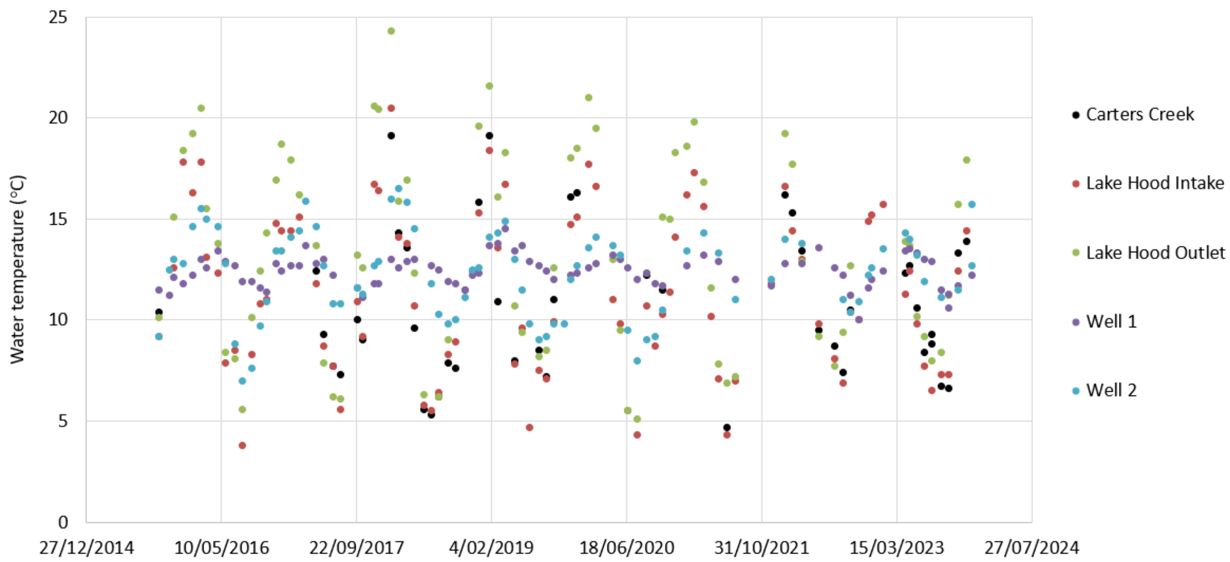

Figure 3-3: Time series of the measured water temperature at two inflows, two groundwater wells, and the outlet of Lake Hood.

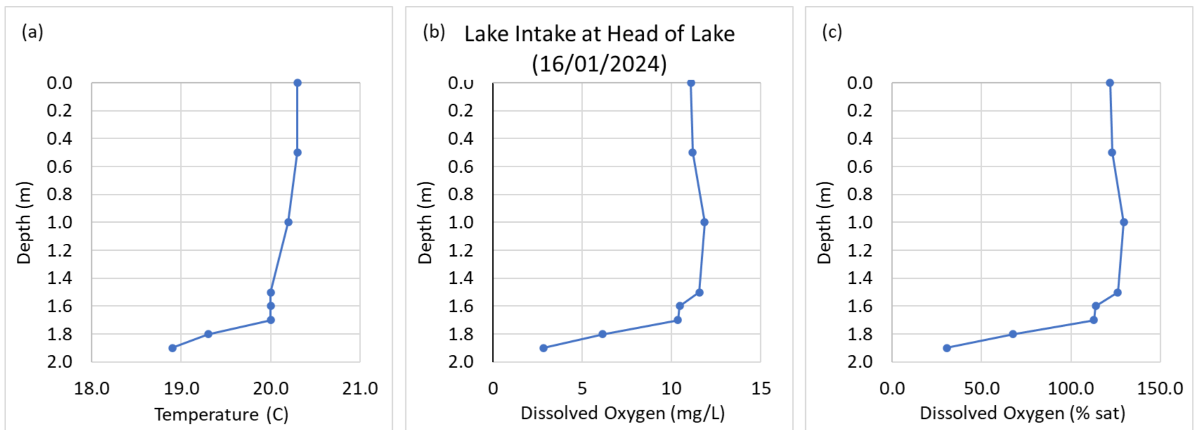

Figure 3-4: Profiles of water temperature (a) and dissolved oxygen concentration and saturation (b and c, respectively) at lake intake at head of lake on 16/01/2024.

3.4 Dissolved oxyen

Figure 3-5 shows time series of field measurements of dissolved oxygen (DO) in 2015-2023 in Carters Creek, the lake intake, and the outlet. The measured oxygen concentration was more than 5 mg/L most of the time, except on 29 January 2019, when DO was 2.93–3.65 mg/L across these locations, and 30 March 2021, when DO was 2.35 and 2.97 mg/L at the intake and outlet, respectively (Figure 3-5a). Time series of near-surface DO do not show any significant trends and concentrations are generally high (> 5 mg/L).

Profile measurements of dissolved oxygen at the outlet, intake, New Extension Near Model Boat Club Dock, and Test Canal Mid-point on 13 December 2023 and 16 January 2024 show < 0.5 mg/L difference in the dissolved oxygen between surface and bottom (not shown here), except at the lake intake on 16 January 2024, when a difference in DO of 8.23 mg/L was recorded (Figure 3-4). The bottom DO at this time was 2.85 mg/L, which is close to hypoxia (DO < 2 mg/L). Therefore, low bottom water oxygen concentrations may occur during stratified periods in Lake Hood.

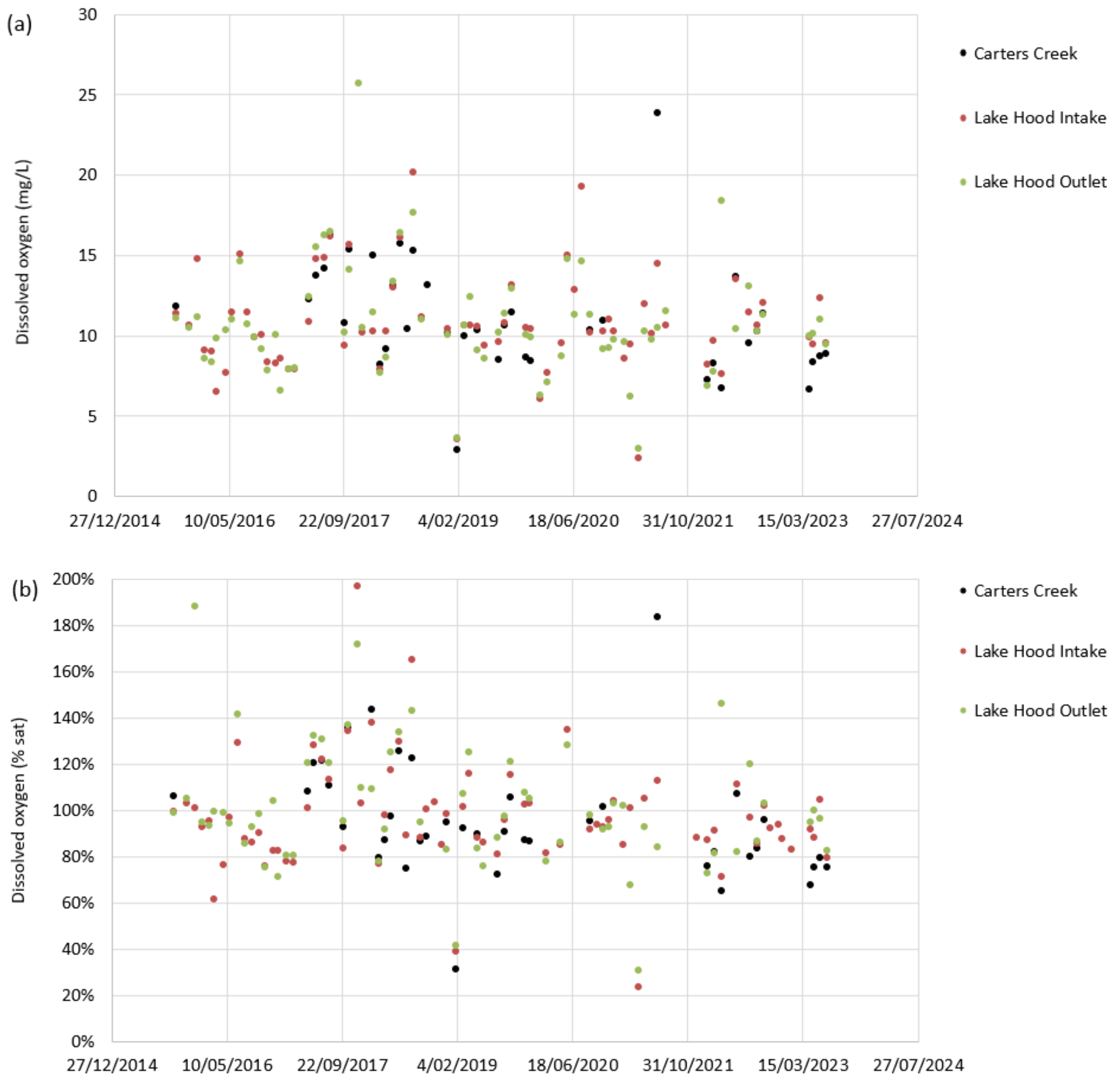

Figure 3-5: Time series of the measured dissolved oxygen concentration and saturation (a and b, respectively) in two inflows and at the lake outlet.

Time series of the surface and bottom water temperature at the canal east end, west end, and mid-end (Figure 3-6) show that the aerator that was turned on 30 October 2023 did not change the stratification pattern, i.e., diurnal stratification with a 1–2°C temperature difference between the surface and the bottom continued to occur in the water column. More importantly, profiles of dissolved oxygen at the west end near diffuser 6 on 8 January 2024 (Figure 3-7) show near-bed hypoxia (DO = 0.25 mg/L). These results indicate that the aeration setup could not prevent water column stratification and occurrence of hypoxia.

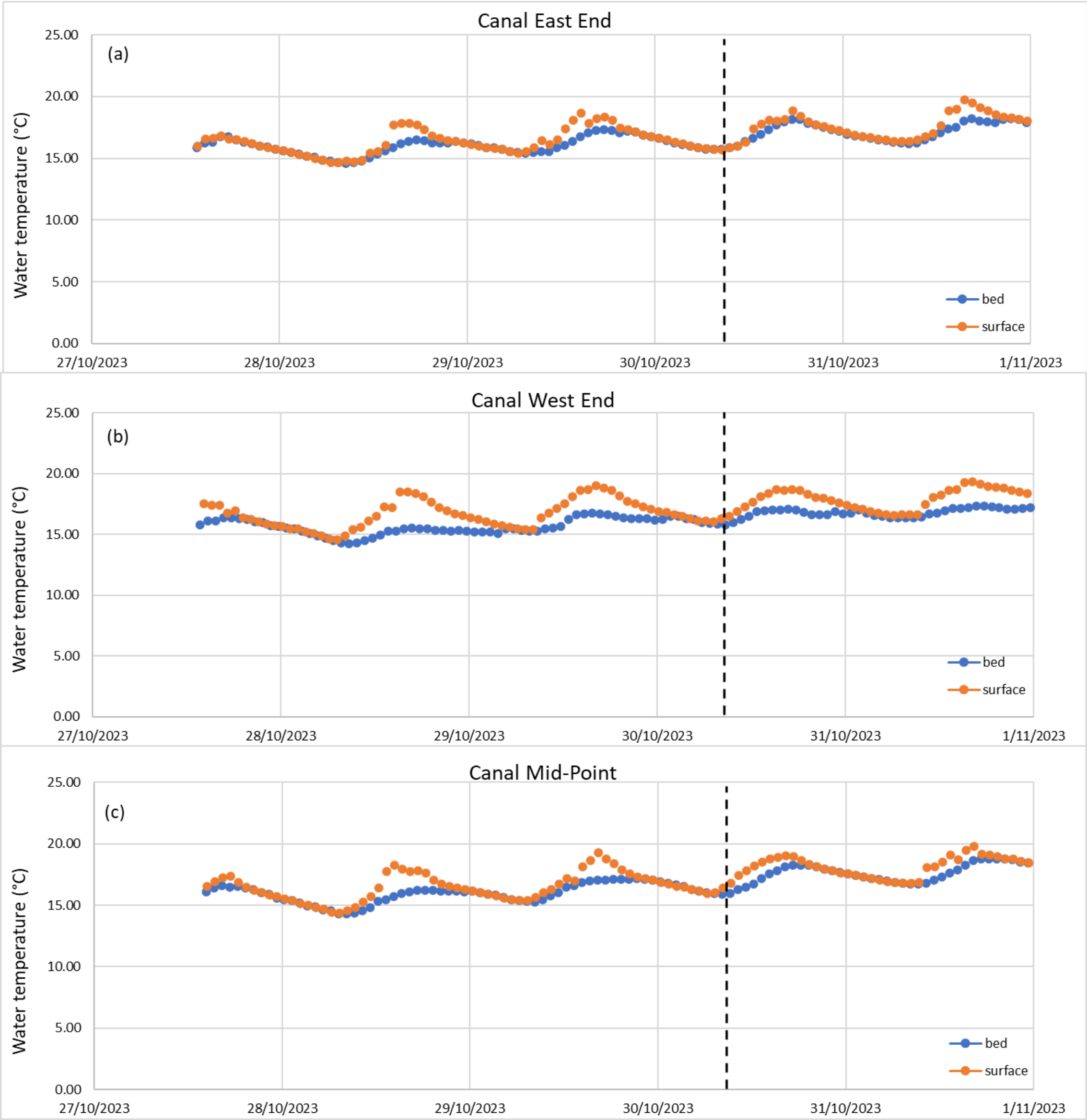

Figure 3-6: Time series of the surface and bottom water temperature at (a) the canal east end, (b) west end, and (c) mid-point. The dashed line shows when the aerator was turned on (9:00 am on 30 October 2023).

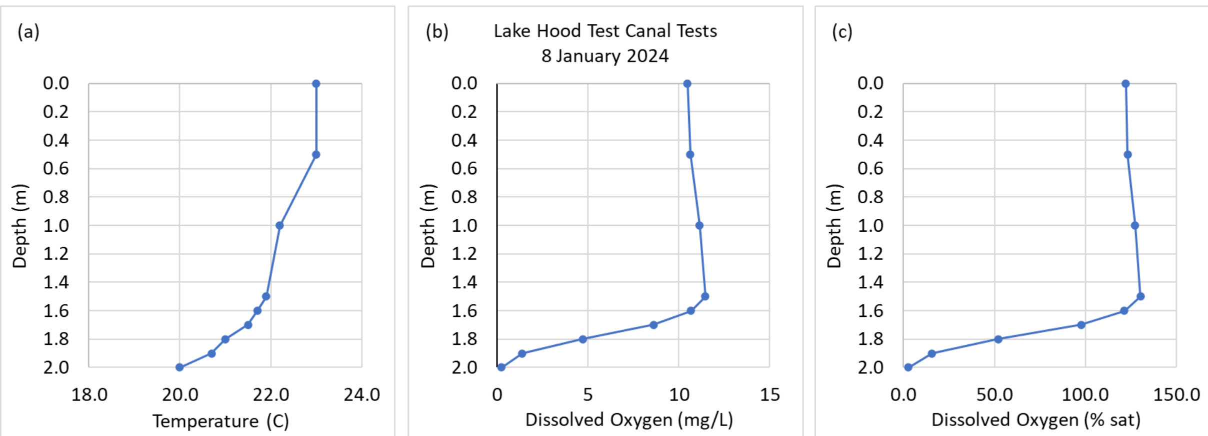

Figure 3-7: Profiles of (a) water temperature, (b) dissolved oxygen concentration, and (c) dissolved oxygen saturation at Lake Hood Test Canal on 8 January.

3.5 pH

High pH (>10) can result in phosphorus release from the sediments, regardless of oxygen levels (Gibbs et al. 2022). Time series of pH measurements in 2015–2023 in well 2, the lake intake, and in the lake outlet (Figure 3-8 a) show that pH values are usually within the consented range (6.5–8.5), with the highest values in the lake outlet. There are statistically significant increasing trends in pH. Maximum field values of > 11 were observed in October 2021 at the lake intake and well 2. At the lake outlet, the pH increased from 7.34 in May 2023 to 10.01 in October 2023 and 9.23 in November 2023. We note that laboratory pH measurements were lower than field measurements (Figure 3-8); this may be due to inconsistent field probe calibration, sample handling and holding times, or other differences in the pH measurement methods. A statistically significant ($p < 0.05$) trend was only observed in the lab measurements of the lake intake samples.

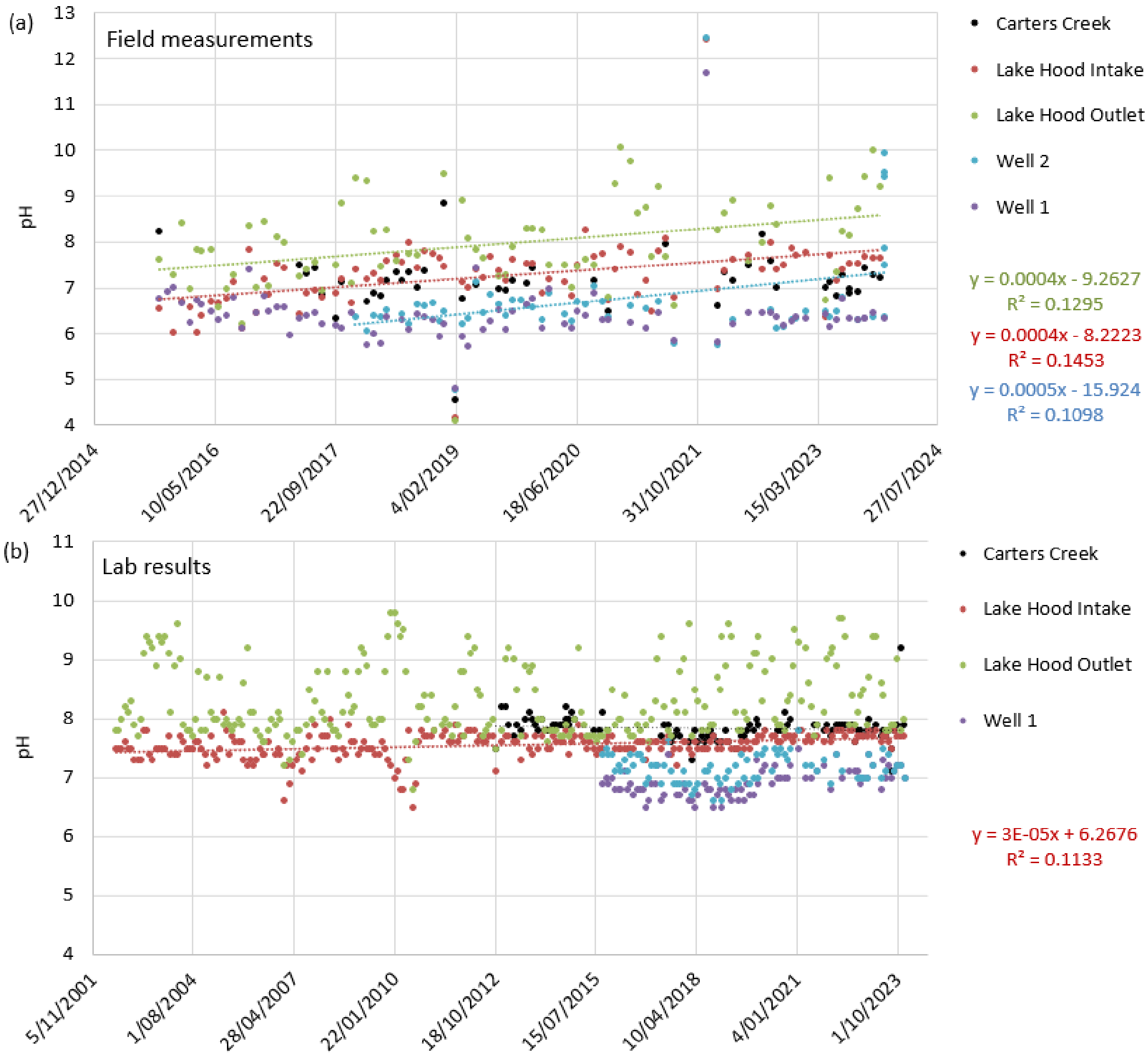

Figure 3-8: Time series of pH from measurements (a) and lab results (b) with linear regressions for significant trends (dashed lines). Equations are marked by colour for each location. Note the difference in the x-axis limits in the two panels.

3.6 Total suspended solids, turbidity, and water clarity

Total suspended solids (TSS) is a measure of particles suspended in the water column. This can include sediments, debris, and live or decaying organic matter including phytoplankton. Turbidity is a measure of the cloudiness or haziness caused by particles in water and thus affects water clarity. Water clarity is a measure of how far or deep light penetrates through the water by a visual clarity measure. Water clarity can be an indicator for algal or macrophyte growth because light conditions can limit growth.

Since 2001, total suspended solids are usually below the consent limit (50 mg/L) but have increased slightly in the Lake Hood intake, with maximum values of >500 mg/L in February 2018, December 2021, and August 2023 ($p < 0.05$, Figure 3-9a). There are no significant trends in TSS in Carters Creek or in the lake outlet (Figure 3-9b). A maximum value of 67 mg/L at Carters Creek was recorded in August 2023 (Figure 3-9b).

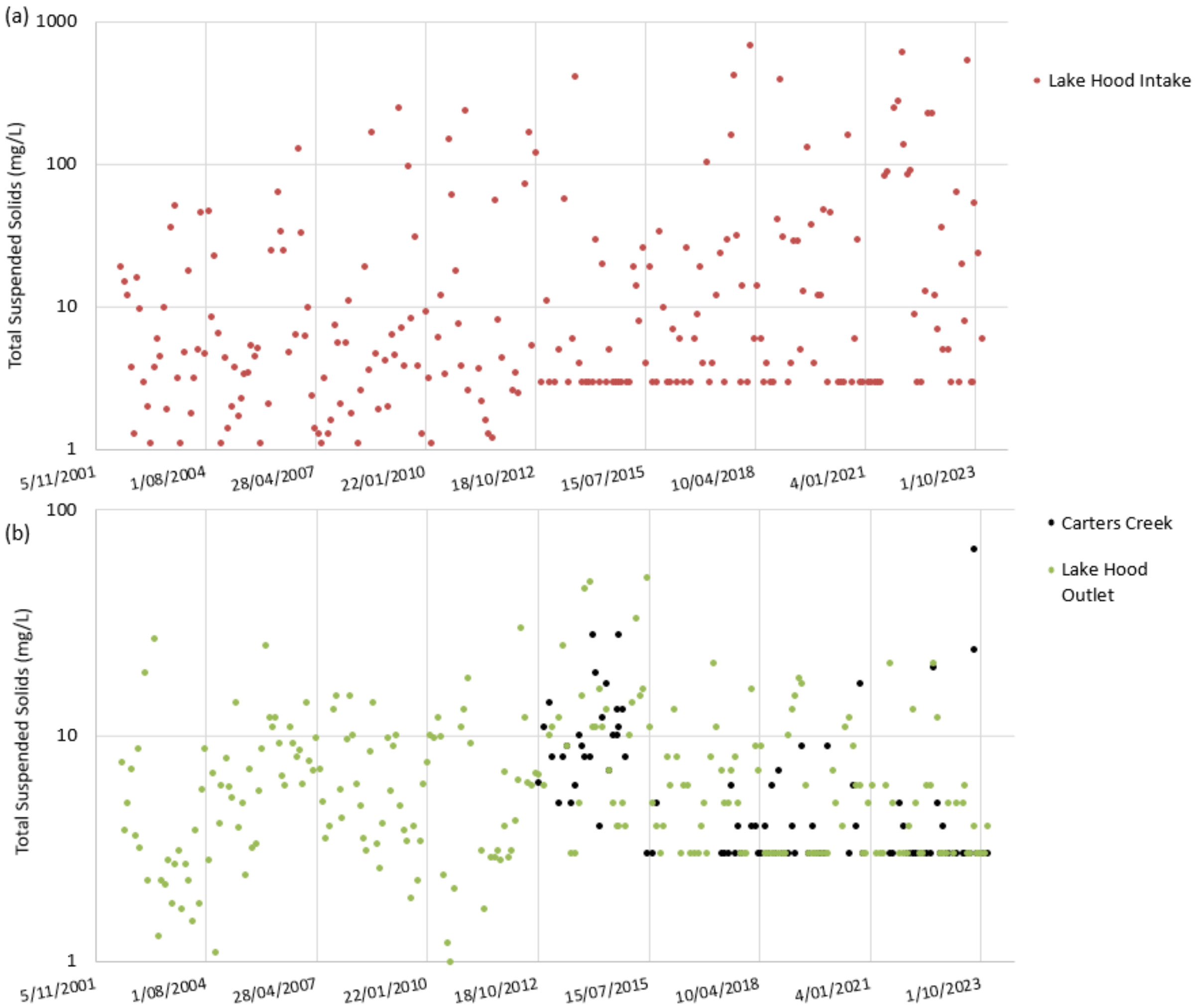

Figure 3-9: Time series of total suspended solids in (a) the Lake Hood intake and (b) Carters Creek and the Lake Hood outlet. Note that the y-axes are on a logarithmic scale and that the detection limit appears to be 3 mg/L since 12/12/2012.

In Lake Hood, water clarity was measured using a clarity tube, which is appropriate for measuring water clarity in relatively turbid water, where visibility through the water is less than 1 m. There are no significant trends in the clarity tube measurements or in turbidity at Carters Creek, the lake intake and outlet in 2015–2023 (Figure 3-10). Clarity tube values cannot exceed 1.0 m (the length of the tube used in the measurements), but we note that several measurements are close to or even equal to 1.0 m. Secchi depth is a more common measure of visual water clarity or transparency through the lake water column and is obtained by lowering a Secchi disk into the lake until it is no longer visible and measuring the length of the submerged line. Given that clarity tube measurements are high in Lake Hood, visual water clarity measured by a Secchi disk might be a more informative measure and an indicator of light penetration through the water column to a certain depth. Then, the photic threshold, or the depth beyond which little to no photosynthesis occurs, could be defined.

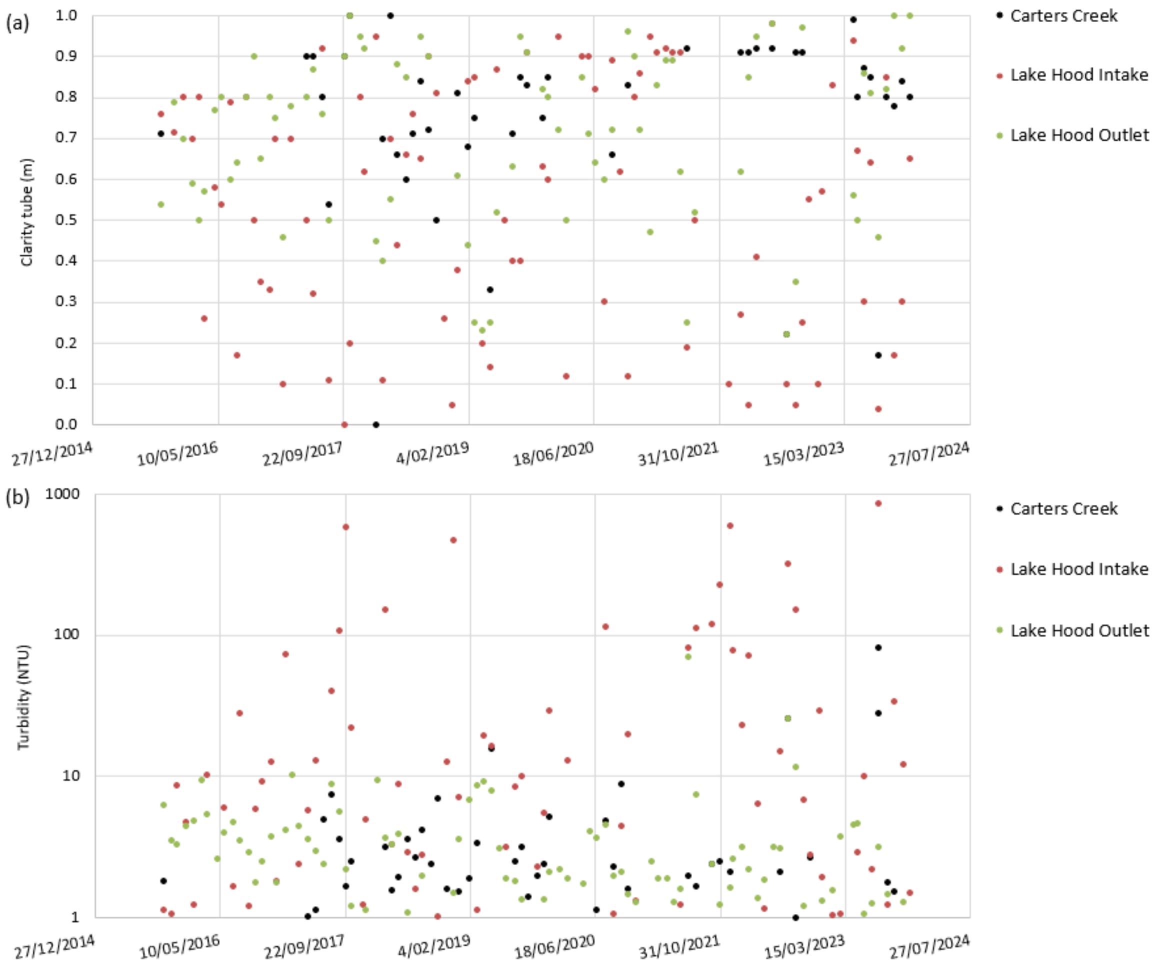

Figure 3-10: Time series of total suspended solids in (a) water clarity (clarity tube) and (b) turbidity at Carters Creek and the Lake Hood intake and outlet. Clarity tube values are always between 0 and 1 m (i.e., 1 m is the maximum possible value). Turbidity is plotted on a logarithmic scale.

Poor water clarity can indicate light limitation for phytoplankton and there are several measures of water clarity and influencing factors. Often, a relationship between TSS and turbidity can be determined, but this can vary greatly between different water bodies or different locations within a water body (e.g., 1:2 to 2:1 turbidity to TSS ratio in a US reservoir, Brown 1984). Measures of water clarity can be also related to turbidity.

Based on the water clarity tube measurements and the fact that Secchi depth is likely greater (as in Litton et al. 2004 and based on 2007–2008 Secchi depth measurements in Lake Hood, Tonkin & Taylor 2008), we expect light to penetrate through the water column, usually to the lake bottom. Low turbidity measurements and the fact that the lake is shallow (mean 2.39 m, max 4.25 m; Tonkin & Taylor 2008) also suggest that light likely reaches the lake bottom and does not limit algal growth. Even if light conditions are suboptimal beyond a certain depth, light is unlikely to limit the cyanobacteria observed in Lake Hood, because they have the ability to control their buoyancy and thus adjust their vertical location in the water column and seek optimum light conditions for growth (see section 4.1.1).

We could measure light conditions in the lake over time and then calculate the light extinction coefficient and estimate the photic depth with respect to changes in TSS or turbidity, but – again – it seems unlikely that light is a limiting factor for cyanobacteria growth in Lake Hood.

3.7 Sediment nutrient content

Sediment nutrient content can be an indicator for potential sediment nutrient loading to Lake Hood. Phosphorus is especially of interest, as it is the limiting nutrient for cyanobacteria growth (see section 4). Lake Hood sediments were sampled once at four locations on 5 September 2023 and analysed for P, N, and C (Table 3-1).

Sediment total phosphorus content in Lake Hood ranged between 440–820 mg/kg. Sediment total phosphorus content values in the range 200–500 mg/kg can be considered low, 500–1000 mg/kg can be considered moderate, and >1000 mg/kg can be considered high (Pettersson et al. 1988). For example, sandy, coastal sediment P content can be 10 mg/kg or less, while iron- and carbonate-rich gyttja (sediment rich in organic matter in eutrophic lakes) P content can be as high as 10,000 mg/kg (Holtan et al. 1988). The sediment P content in different lakes considered by Petterson and Jansson (1988) ranged 550–649 mg/kg dry mass. In Lake Taihu (a large, shallow, eutrophic lake in China with frequent cyanobacteria blooms), sediment P content measurements ranged 330–1030 mg/kg dry mass (Trolle et al. 2009). Based on the available data for Lake Hood, the sediment total P content appears to be moderate.

Sediment total nitrogen content ranging 1,000–5,000 mg/kg can be considered low, 5,000–10,000 mg/kg can be considered moderate, and >10,000 mg/kg can be considered high. In Lake Taihu, sediment N content measurements ranged 380–2,370 mg/kg (Trolle et al. 2009). Based on the available data for Lake Hood (700–4,400 mg/kg), the sediment total nitrogen content appears to be low.

Sediment total organic carbon content ranging 1–5 g/100 g can be considered low, 5–10 g/100 g can be considered moderate, and >10 g/100 g can be considered high. For example, in Lake Taihu, sediment organic C content measurements ranged 1.68–9.38 g/100 g (Trolle et al. 2009). Based on the available data for Lake Hood (0.44–3.7 g/100 g), the sediment organic C content appears to be low.

Table 3-1: Total phosphorus, nitrogen, and carbon content in lake sediment samples collected from four sites on Lake Hood on 5 September 2023.

Site	Total P content (mg/kg dry mass)	Total N content (mg/kg dry mass)	Total organic C content (g/100 g dry mass)
Trial Canal	710	4,400	3.7
Lake intake	820	1,500	1.12
Lake outlet	440	900	0.71
New extension	550	700	0.44

The limited sediment sampling results suggest that the phosphorus nutrient content could contribute to the nutrient load that fuels cyanobacteria blooms.

3.8 Nutrients and chlorophyll *a*

In an earlier communication (2017) by Clive Howard-Williams to the Task Force, it was stated that nutrient and chlorophyll concentrations in the lake were high compared to concentrations in the Ashburton River. The river was deemed strongly phosphate limited – although no evidence was provided to show this (AEE section 8.1.9, as cited by Clive Howard-Williams), and the bioavailable nutrients dissolved inorganic nitrogen (DIN = NH_4-N + NO_3-N + NO_2-N, the sum of ammoniacal nitrogen, nitrate nitrogen, and nitrite nitrogen) and dissolved reactive phosphorus (DRP) were lower in the lake than in the river.

McCracken et al. (2023) also indicated that phosphorus is an important driver of cyanobacteria growth in Lake Hood:

> *The Carters Creek sample taken after a storm flow shows total nitrogen at nearly three times the 2023 lake level and total phosphorus at over 40 times the lake level. During the July 2023 flood, over 65% of the total phosphorus was dissolved reactive phosphorus, the most readily available form for algal and cyanobacterial growth. The Carters Creek water quality is important in that the creek discharges to the west-most canal where water flows are minimal and exchange between the canal and the main body of the lake is thought to be low (except during Carters flood flows), and where the 2023 Cyanobacteria bloom was first detected*.

Based on the available data, total nitrogen (TN) increased in well 2 in 2015–2023, with values of > 6 mg/L since August 2023 (Figure 3-11a). The maximum TN value of 10.3 mg/L was recorded in September 2023 at this location. TN does not show any obvious trends or large changes at other locations but has slightly decreased in well 1. Nitrate-nitrite (NO_3 and NO_2) in wells 1 and 2 were generally > 2 mg/L and increased significantly in well 1 in 2015–2023. Lake Hood inlet and outlet nitrate-nitrite concentrations were generally < 2 mg/L.

Total phosphorus (TP) values were highest in the lake intake and Carters Creek (Figure 3-12b; records at the wells are limited to 2015–2016). While peaks of TP > 0.2 mg/L were recorded in the lake intake over several years, the maximum value of 0.57 mg/L in the lake intake was in August 2023, which corresponds with the highest recorded dissolved reactive phosphorous (DRP) value of 0.38 mg/L at this location (Figure 3-12c). DRP values in other locations were generally < 0.15 mg/L.

Figure 3-11: Time series of (a) total nitrogen, (b) nitrate-nitrogen, (c) total phosphorus, and (d) dissolved reactive phosphorus. Dashed lines are linear regressions for significant trends ($p < 0.05$). Equations for each location are marked by colour. Not shown: DRP was > 0.1 on 28/7/2022, 25/7/2023, and 24/7/2023.

While the highest values of total biochemical oxygen demand (TBOD) were found in the lake outlet, it was generally < 2 mg/L in the lake intake and Carters Creek (Figure 3-12a), well below the consent condition of 5 mg/L.

Chlorophyll *a* (chl *a*) is a measure of phytoplankton biomass and showed slight decreases in Carters Creek since 2015 (section 3.8); however, there are no apparent chl *a* trends in the lake outlet or intake. Corresponding with phosphorus records, the highest values of chlorophyll *a* (> 0.04 mg/L) were recorded in the lake outlet, and chlorophyll *a* was generally < 0.02 mg/L at the lake intake and Carters Creek (Figure 3-12b). While chlorophyll *a* decreased slightly in Carters Creek since 2015, there are no apparent trend directions for chlorophyll *a* in the lake outlet or intake.

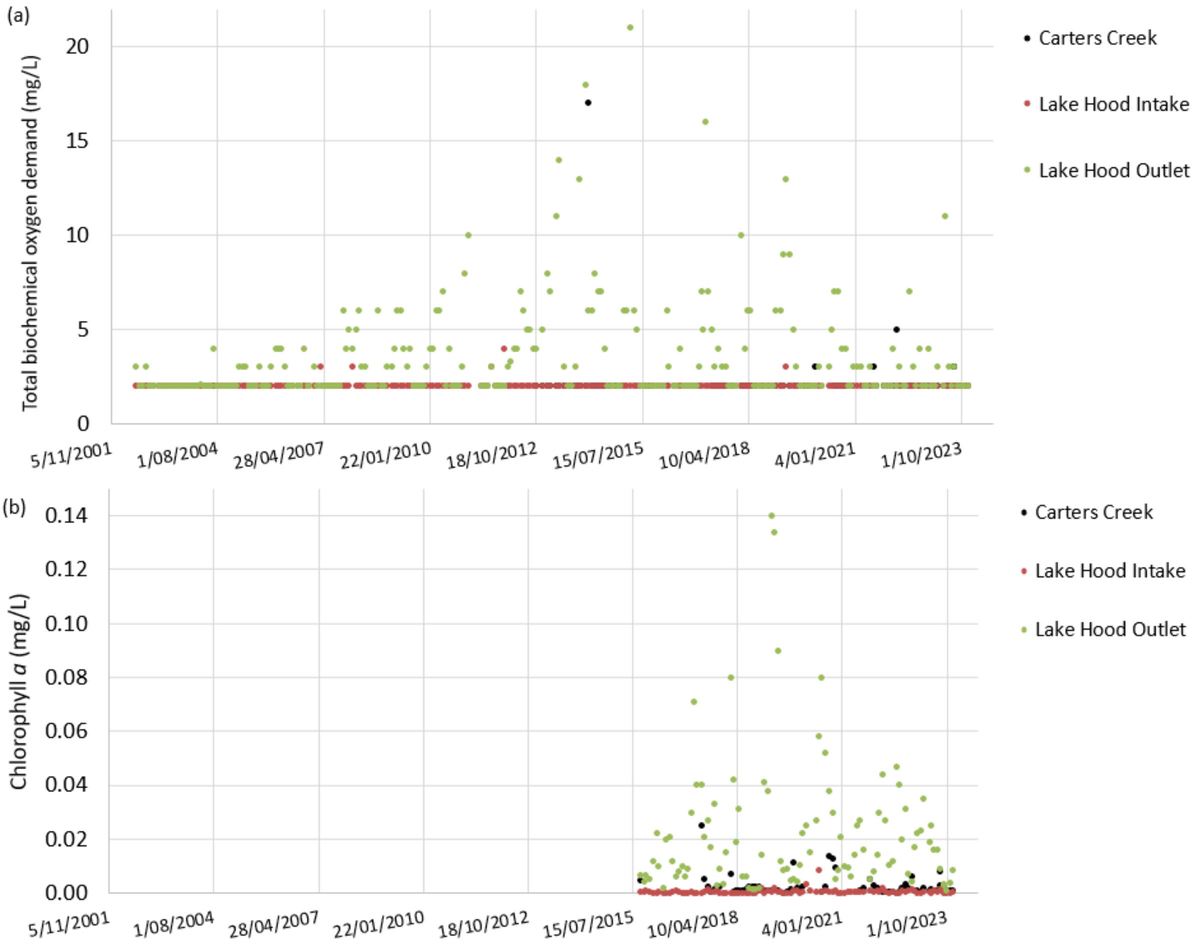

Figure 3-12: Time series of (a) total biochemical oxygen demand and (b) chlorophyll *a*. Not shown: BOD was 39 mg/L on 11/12/2013.

3.9 Trophic state

The Trophic Level Index (TLI, Burns et al. 1999) is widely used in New Zealand to track and report on the level of eutrophication in lakes. TLI is calculated from TN, TP and chlorophyll *a* concentrations (and Secchi depth can also be included, where available). In 2023, mean chlorophyll *a* was 13.5 mg/m^3, TN was 1,065 mg/m^3, and TP was 15.3 mg/m^3 in the Lake Hood outlet, representative of lake conditions. Based on these mean annual values, the trophic level index (TLI, Burns et al. 1999) is 4.76, indicating that Lake Hood is eutrophic, which is consistent with the assessment by Tonkin and Taylor (2008). In addition, the N:P ratio is high (69), indicating low P relative to N and thus P limitation for algal growth.

There are, unfortunately, no long-term data available on the phytoplankton population composition in Lake Hood (chlorophyll *a* is a measure of overall phytoplankton biomass but does not describe what species are present). Counts were obtained for cyanobacteria in the summer of 2023 (9 February 2023), and elevated levels of cyanobacteria were reported in subsequent counts from 23 March 2023, 29 March 2023, and 24 April 2023 by Cawthon and in full counts by NIWA for a sample from 4 January 2024 (see detailed discussion in section 4). The only non-cyanobacterial species mentioned in the Cawthron reports for 2023 was the large dinoflagellate *Ceratium* sp., which peaked in January. *Ceratium* sp*.* (motile) is likely to compete with *Dolichospermum* as they are both advantaged by being able to move up and down through the water column (buoyancy control) and are favoured under stratified and/or stable low wind/low mixing conditions (see section 4.1.1). This species was again prominent in the bloom seen recently, in March 2024. All other main phytoplankton species present in the March 2024 sample were identified. The other species in the March 2024 sample were dominated again by motile species of *Cryptomonas* sp. (Cryptophyceae) followed by several other motile species from a range of phyla including *Chroomonas* sp., *Eudorina* sp., *Trachelomonas volvocina,* and *Haematococcus* sp. Most of the remaining non-motile species were green algae, with only one diatom observed in the sample. Motile genera are likely advantaged by being able to maintain themselves at optimal levels for light in the water column if mixing was low. Domination of motile species (or species with buoyance control) indicates that motility is advantageous in Lake Hood either to gain nutrients or to maintain time in the best light environment for growth. This could also lead to the occurrence of shading of other non-motile phytoplankton groups if low mixing allows phytoplankton to concentrate at an optimal light depth. The occurrence of these species is expected in a eutrophic environment with low P compared to N, as diatoms are poor competitors at low phosphate concentrations (Egge 1998). Cyanobacteria are discussed in detail in the following section.

4 Cyanobacteria

Cyanobacteria are a group of phytoplankton also commonly called blue-green algae that can be of concern because some types can produce toxins that can compromise human, animal, and ecosystem health. Blooms observed in Lake Hood in 2023 and 2024 were sampled on at several sites on five occasions (Figure 4-1).

Figure 4-1: Approximate phytoplankton sampling locations in Lake Hood. Map provided by the Task Force (Les McCracken, March 2024).

Several different cyanobacteria were present in water samples collected from Lake Hood in 2023 and 2024. The sampled blooms were dominated by the genus *Dolichospermum* (previously found within the genus *Anabaena*, but separated from this now benthic genus in 2009, Wacklin 2009). In 2023, the following *Dolichospermum* species were identified: *Dolichospermum spiroides* (dominant), *Dolichospermum circinale* (subdominant, Figure 4-2), and *Dolichospermum cf. planctonicum. Dolichospermum circinale* dominated in March 2023, peaking at 15,000 cells per ml and at ~3 mm^3/L in biovolume (Ski Lane first Jetty, Figure 4-3). *Dolichospermum spiroides* dominated the cyanobacteria bloom in January 2024 at 61,000 cells/ml and ~9 mm^3/L in biovolume (Lake Hood Huntingdon Ave Canal, Figure 4-2). In water samples collected during another bloom in late February 2024, *Dolichospermum circinale* was dominant (~34,000 cells per ml) and *Dolichospermum spiroides* was subdominant (2,203 cells per ml), with *Dolichospermum planctonicum* also present (33 cells per ml) at Lake Hood Bayliss Beach. Similar *Dolichospermum circinale* cell counts (~30,000 cells per ml) were found at Halston Close-Carters Creek Canal. The canal areas were the most affected by cyanobacteria.

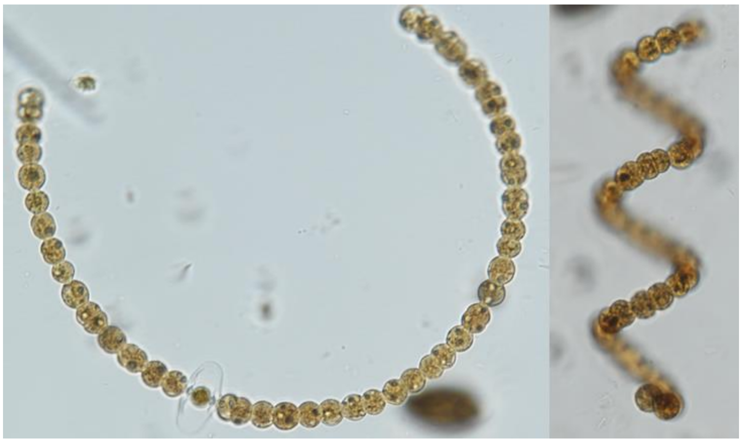

Figure 4-2: Images of *Dolichospermum circinale* (left) and *Dolichospermum spiroides* (right) taken of water samples collected for analysis from Lake Hood in March 2024. Source: Karl Safi, NIWA.

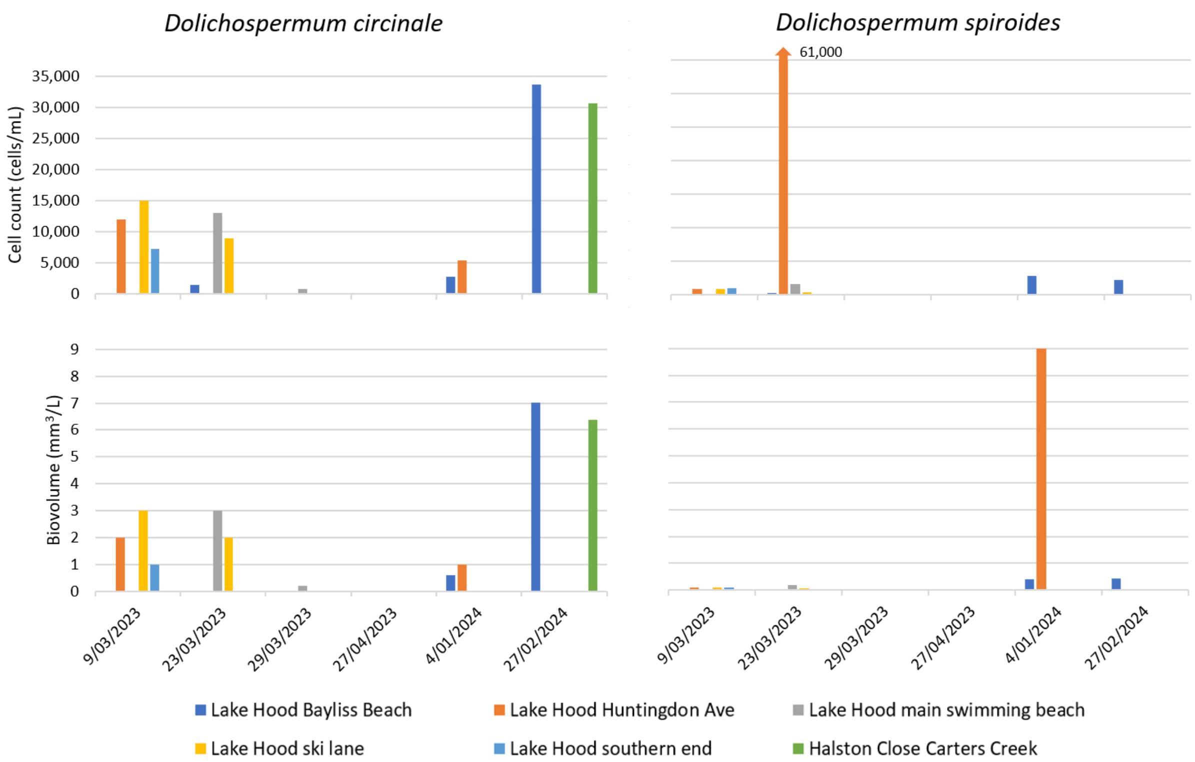

Figure 4-3: Cyanobacteria laboratory results from five sampling sites and dates for the two most dominant species, *Dolichospermum circinale* (left) and *Dolichospermum spiroides* (right) in Lake Hood. Biomass is presented as cell counts (top) and biovolume (bottom).

All other cyanobacteria were reported at much lower concentrations and were either smaller in size (and therefore biovolume), non-toxin producers, or benthic. These are unlikely to be of significant concern at present. *Aphanocapsa* sp. and *Pseudanabaena* sp. cells are generally < 3 µm in diameter, smaller than *Dolichospermum* cells (> 5 µm in diameter). *Limnococcus limneticus*, although larger in diameter, occurred in very low numbers and is not known to be toxic. *Phormidium* sp. (7.0–7.9 µm), although a recognised toxin producer in New Zealand, is benthic (bottom-dwelling) and was observed in very low numbers. *Phormidium* sp. do not propagate in the larger lake as they require surfaces to grow on; however, this cyanobacterium could be coming from river inflows (detached benthic material) or could propagate on the lake fringes/edge, in which case its numbers may be underestimated in water column samples. *Aphanizomenon* was detected in extremely low numbers. This species has similar characteristics to *Dolichospermum* and could be problematic if found in large numbers. In 2024, toxin producing *Microcystis* sp. were also detected at low concentrations of 24 cells/ml and could also become problematic if found at much higher concentrations.

4.1 *Dolichospermum* characteristics

Dolichospermum blooms can be a threat to the environment and human health due to toxin production. This genus can produce a variety of highly potent toxins including the neurotoxins anatoxin and saxitoxin and the liver toxins microcystin and cylindrospermopsin (O'Neil et al. 2012, Li et al. 2016, Capelli et al. 2017, Österholm et al. 2020). This genus also produces lipopolysaccharides which can cause skin or lung irritations and gastroenteritis. In New Zealand, *Dolichospermum lemmermannii* has been found to produce anatoxin-a and microcystin (pers. comm. K. Thompson, unpublished data). Other *Dolichospermum* sp. are commonly seen in New Zealand as part of mixed cyanobacteria blooms associated with toxins, but in in those cases toxin production cannot be clearly attributed to this genus. Overseas, the genus has been associated with the production of anatoxin-a, anatoxin-a(S), cylindrospermopsins, and microcystins, so these toxins may also be produced in New Zealand (Guidelines for DWM 2020).

Dolichospermum has some special characteristics that may have led to its dominance in Lake Hood. These features are described in detail in the following sections.

4.1.1 Buoyancy control and temperature preference

Dolichospermum cells have gas vesicles that allow for buoyancy regulation in the water column. Compared to other phytoplankton without this capability, *Dolichospermum* may have a competitive advantage (Walsby et al., 1995). *Dolichospermum* can use its gas vesicles to move to optimal depths in the water column, to place itself in optimal light conditions to maximise growth during the day and descend to deeper water to obtain nutrients at night. This occurs most efficiently in stratified (not well mixed) deeper waters but can also occur in mixed shallow waters.

Thermal stratification divides the water column into a warmer top and a cooler bottom layer, often leading to lower nutrient concentrations in the top following phytoplankton growth (Figure 4-4). This provides an advantage for *Dolichospermum* because it can move down to the bottom layer to obtain more nutrients. Stratification may induce anoxic conditions (< 0.5 mg/L DO) in the bottom waters of the hypolimnion that can promote the release of nutrients, especially phosphorus, iron, and manganese from the sediments. This could increase *Dolichospermum's* advantage by increasing nutrient availability (Ismail et al. 2002, Dengg et al. 2023, Figure 4-4).

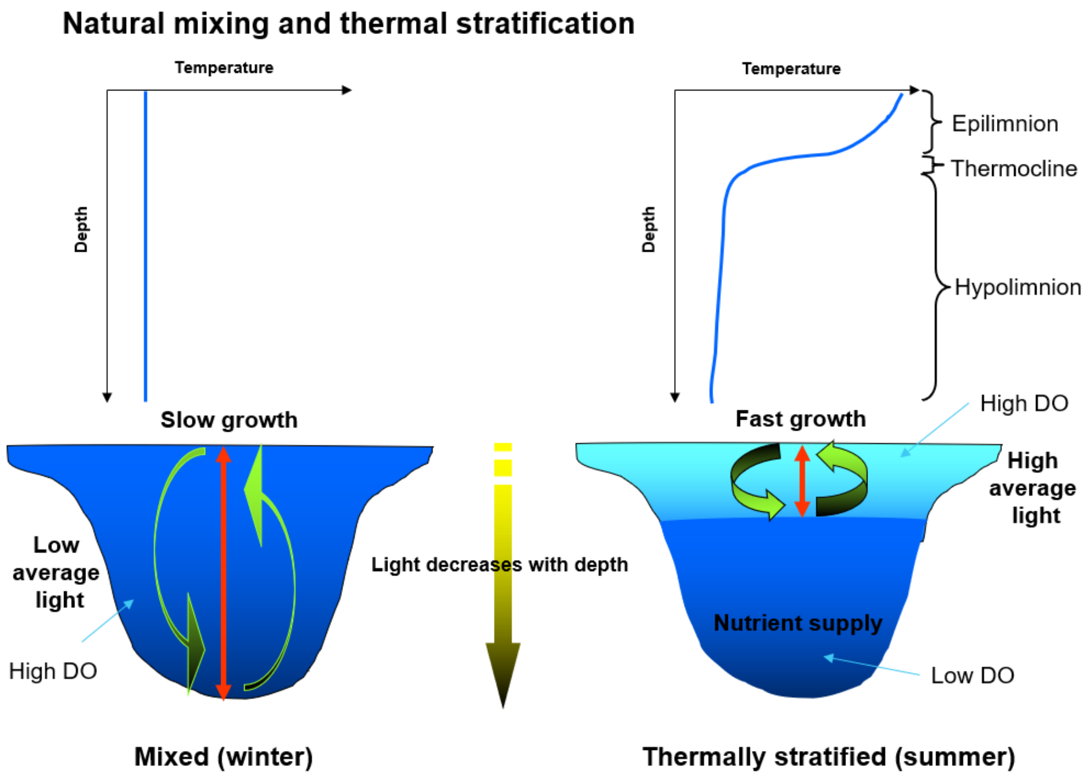

Figure 4-4: Diagrams showing how thermal stratification can affect the conditions for the growth of phytoplankton populations in the water columns of lakes. Source: Max Gibbs, NIWA.

Increasing temperatures due to climate change are predicted to favour cyanobacteria growth over other groups of freshwater phytoplankton (Paerl and Huisman 2008, Paerl and Huisman 2009, Paerl and Paul 2012). Cyanobacteria growth rates are higher at temperatures exceeding 25°C (Robart and Zohary 1987, Butterwick et al. 2005), while other algae have growth optima at temperatures below 25°C (Peeters et al. 2007).

Warming is also expected to enhance nutrient loading by inducing thermal stratification and phosphorus release from lake sediments (Jeppesen et al. 2009) and mineralisation in catchment soils (Moss et al. 2011). For example, Westwood and Ganf (2004) found that *D. circinale* populations grew faster when exposed to longer periods (multiple days) of stratification than short periods of stratification or mixed conditions. Experiments showed that *Dolichospermum* growth decreased with artificial mixing preventing thermal stratification (Reynolds et al. 1983, Nakano et al. 2001).

4.1.2 Nitrogen fixation

Dolichospermum can turn simple cells into heterocysts (Komárek and Anagnostidis 1989, Komárek 2010). Heterocysts are specialized nitrogen-fixing cells, which form during low nitrogen conditions (Stewart 1973, Figure 4-5). The primary function of these specialised cells is to fix nitrogen from the atmosphere which can then be used for biomass production (Schindler et al. 2008).

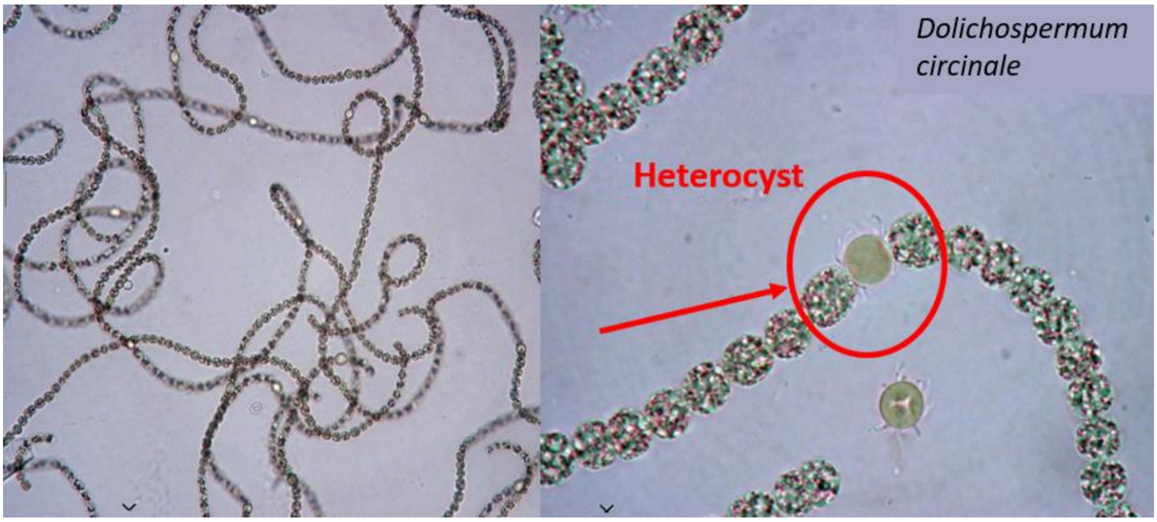

Figure 4-5: Images showing *Dolichospermum circinale*, showing a nitrogen fixing heterocyst within the filaments. Source: Karl Safi, NIWA.

4.1.3 Akinete production

Dolichospermum can also turn simple cells into akinetes, which are resistant spores that can quickly grow into new filaments or form seed banks (Komárek and Anagnostidis 1989, Komárek 2010). Akinetes can sustain long-term dormancy if conditions do not favour their germination, enabling *Dolichospermum* populations to survive under harsh conditions (such as overwintering, dry seasons, flooding; Stüken 2006). Once a bloom has occurred, the establishment of seed banks makes reoccurring blooms during optimum environmental conditions likely.

4.1.4 Allelochemical interactions

Cyanobacteria may dominate phytoplankton blooms due to production of allelochemicals that inhibit the growth of other phytoplankton (Schlegel et al. 1999, Fujii et al. 2002, Suikkanen et al. 2004, Suikkanen et al. 2005, Gorokhova and Engström-Öst 2009, Engström-Öst et al. 2011). For example, allelochemicals produced by *D. lemmermannii* decreased cell numbers of two phytoplankton taxa, *Rhodomonas* sp. and *Thalassiosira weissflogii* (Suikkanen et al. 2004).

4.1.5 Higher pH preference

Alkalinity and pH determine the chemical speciation of inorganic carbon, such as carbonate, bicarbonate, and carbon dioxide. Low carbon dioxide concentrations favour the growth of several cyanobacterial species. Hence, low alkalinity and hardness and high pH (a result of photosynthesis) give cyanobacteria a competitive advantage (Health Canada 2000, edited 2002). Elevated and apparently increasing pH in Lake Hood may not only promote P release from the sediments (see section 5) but also contribute directly drive cyanobacteria blooms.

4.2 Likely drivers of cyanobacteria blooms in Lake Hood

Considering water quality conditions based on the available data (section 3) and cyanobacteria observations, there are likely several drivers of blooms in Lake Hood:

- high temperature (regularly ~20°C and as high as 24.3°C in the lake outlet; advantageous for cyanobacteria, Robarts and Zohary 1987)
- high pH

- *Dolichospermum* mobility through buoyancy control
- *Dolichospermum* in Lake Hood fixes nitrogen (evidence of heterocysts suggests this)
- likely P limitation

Phosphorus is commonly considered to be the most limiting nutrient in freshwater ecosystems (O'Neil et al. 2012). High P concentrations often co-occur with severe cyanobacterial blooms in many regions of the world, such as in large lakes in North America and in China (Huang et al. 2016). In a recent study, which analysed data from 464 lakes covering a 14,000 km north-south gradient in the Americas and three lake depth categories, Bonilla et al. (2023) found that phosphorus was the primary resource explaining cyanobacterial biomass. Nitrogen was less significant and largely associated with shallow lakes (< 3 m depth). In this study, water temperature was not significantly related to cyanobacteria biomass. In a recent study of eight NZ lakes, Guildford et al. (2022) concluded that evidence of P limitation was stronger than for N limitation and more effort is needed to reduce P inputs to protect and remediate NZ lakes.

Ratios of TN:TP are sometimes used to determine N or P limitation for algal growth. Ratios greater than 15:1 can be indicative of potential P-limitation, and ratios less than 7:1 can be indicative of potential N-limitation, while ratios between 15:1 and 7:1 can be indicative of potential N- and P-colimitation (White 1983, Vant 1987, Havens et al. 2003, McDowell et al 2009, MfE 2007, Abell et al. 2010). A high concentration of P and a low N:P ratio can favour the development of cyanobacterial blooms (Smith 1983, Ekholm 2008), as some cyanobacteria can make up for nitrogen deficits by fixing nitrogen from the atmosphere. In a study on Lake Vombsjön (Li et al. 2018), TP had a stronger positive correlation with cyanobacteria biomass than dissolved inorganic phosphorus (DIP), and DIN had a stronger negative correlation with cyanobacteria biomass than TN. In this case, DIN:TP predicted cyanobacteria biomass development better than TN:TP. Hence, a low N:P ratio alone does not necessarily prevent the development of cyanobacterial blooms. These studies and other recent evidence suggest that different cyanobacteria have distinct ways of responding to P availability and that the control of cyanobacterial blooms by targeted nutrient reduction largely depends on the dominant species.

Lake Hood has a ratio of TN:TP greater than 15:1 (69:1 for mean TN and TP in the lake outlet in 2023) which is indicative of potential P-limitation. However, this could be different in the canals, where P concentrations are presumably higher (based on P concentrations measured in Carters Creek). Because *Dolichospermum* sp. is the dominant cyanobacterium in Lake Hood, P reduction may be effective in restricting cyanobacteria blooms, as P reduction is more effective in controlling nitrogen fixing *Dolichospermum* than non-nitrogen fixing cyanobacteria like *Microcystis* sp. (Wan et al. 2019). This and other studies suggest that P reduction may be the best approach for reducing cyanobacterial bloom formation in Lake Hood. However, N load reductions may also be required, as one study indicates that highly P-limited, eutrophic conditions can be "vulnerable to more intense and toxic (due to increased biomass) blooms of *Dolichospermum*" (Kramer et al. 2022).

Target N and P levels for management efforts are difficult to define, but some studies provide at least indicative ranges for N and P concentrations that may limit *Dolichospermum* growth in Lake Hood. Wang et al. (2022) incorporated parameter variability into cyanobacteria growth models and reported half saturation constant ranges for N and P based on experimental work by others. The half saturation constant in the Monod growth model (frequently used) represents the nutrient concentration at which the cyanobacteria growth rate is half of its maximum growth rate. For *Dolichospermum flos-aquae*, the nitrate (NO_3) half saturation constant range was 0.0014–0.5295 mg/L (Donald et al. 2013, Baldia et al. 2007) and the phosphate (PO_4, indicative of DRP) half saturation constant range was 0.0006–0.0208 mg/L (Willis et al. 2017, De Nobel et al. 1997). In a study using long-term data for > 2000 Finnish lakes, the authors concluded that TP thresholds of 0.010–0.061 mg/L may be appropriate for limiting toxic cyanobacteria growth (Vuorio et al. 2020).

In Lake Hood, DRP has been low at the outlet (< 0.005 mg/L) but much higher in the intake and Carters Creek (used to be as high as ~0.040 mg/L but more recently usually < 0.030 mg/L, Figure 3-11). The available information suggests that the bioavailable P concentration must be at least less than ~0.020 mg/L to reduce growth, but this may not eliminate *Dolichospermum* blooms in Lake Hood. If bioavailable N concentrations were also reduced to less than ~0.530 mg/L in the lake (noting that nitrate-nitrite concentrations have been as high as 6 mg/L in Carters Creek and Wells 1 and 2, Figure 3-11), the risk oftoxic blooms may be further reduced. Targeted mesocosm (e.g., benthic chamber) or laboratory experiments would be needed to confirm P limitation and better define P thresholds to restrict *Dolichospermum* growth in Lake Hood.

5 Cyanobacteria control options

There are many ways to control cyanobacteria and other algae in small lakes; some are cutting edge, and others have been shown to be effective. Based on the data analysis and literature review in previous sections, we recommend focusing on limiting nutrient availability (specifically P) to control cyanobacteria in Lake Hood. Nutrient concentrations in the lake are determined by nutrient loads to the lake. External loads are nutrients entering the lake via river/stream and groundwater inflows or sometimes as atmospheric deposition (usually negligible). The main external loads to Lake Hood are the Ashburton River intake, Carters Creek, and groundwater. The internal load refers to P stored in lake sediments that is released to the lake water column during low oxygen (or high pH) conditions. Identification of the main P load to Lake Hood requires a detailed water balance and nutrient loading calculations, which we were outside the scope of this work but are recommended.

Nutrient loads can be managed at the source upstream in the catchment – not discussed here – or by means of in-lake interventions (Paerl 2014). There are many traditional and novel in-lake methods for preventing nuisance or harmful algal blooms, as described by Hamilton and Patil (2022) with respect to Lake Rotorua. In-lake options for controlling cyanobacteria were categorised into physical, chemical, and biological control methods in the following subsections.

5.1 Physical controls

5.1.1 Hydraulic flushing and inflow diversion

Physical changes to the lake itself could assist with hydraulic flushing or moving water through the lake faster and thus reducing its residence time. This could be achieved by increasing taking more water from the Ashburton River and releasing more water back to the river, to increase flows through the lake. From 8 January 2024 to 8 April 2024, the total water intake volume was 71,743 m^3 with water takes on only eight days in that period. This means that times in between takes were as long as 16 days until mid-February and there were no takes after that (to the end of the record). This lack of flow-through may well have contributed to lake warming, the formation of anoxia, and ultimately cyanobacteria blooms.

A caveat with this option is that while the main lake may benefit from this, the canals may still not have enough water movement and keep blooms trapped in areas with longer water residence times. Increasing inflow and outflow may have a dilution effect and decrease the water residence time in the main lake basin but is unlikely to improve circulation in the canals, given the distance between the intake and the canals. Hydrodynamic modelling would enable quantification of local water residence times, i.e., how long water remains in a canal before it is flushed to the main lake basin.

Diverting inflows from Carters Creek to better flushed parts of the lake or bypassing the lake altogether could remove a large fraction of the overall nutrient load to the lake (e.g., see Spigel and Ogilvie 1985). However, a nutrient-rich diverted inflow could negatively affect the new receiving environment – be that another part of the lake (if this part is not flushed well enough), the Ashburton River, or somewhere else.

Creation of a second outlet to enhance circulation is another option that cannot be well assessed without hydrodynamic modelling of the lake. This may work to enhance circulation and prevent surface scum formation, but this is unlikely to prevent cyanobacteria blooms.

5.1.2 Phytoplankton harvesting by filtration

Water filtration may remove phytoplankton and cyanobacteria, but this may not be cost-effective and will not prevent bloom formation.

Withdrawing water from near the surface rather than the bottom could remove cyanobacteria concentrated at the surface, but the lake is quite shallow and easily mixed by wind, which means that cyanobacteria likely only form surface scums during calm conditions. Withdrawing water from near the surface will not prevent bloom formation but can remove cyanobacteria if water is drawn from the surface in the canals where the blooms are concentrated and removed.

5.1.3 Artificial destratification

Aerators could be installed in the lake to increase vertical mixing and prevent deoxygenation of bottom water. Artificial destratification by aeration has been trialled in Lake Hood and in theory could control cyanobacteria by (a) reducing the P supply by oxygenating the whole lake to allow iron precipitation of P and (b) the critical depth effect, where the cells are circulated deep into the water column and experience light limitation. In Lake Hood, P reduction is likely to have a larger effect than critical depth (Sverdrup 1953) unless there is sufficient algal biomass present to restrict light penetration below 4 m.

However, based on the data provided from the aeration trials in a test canal, it does not appear that the system was able to mix (destratify) the water column (see section 3.3). System modifications (e.g., the type of diffusers and the way they are installed) may lead to improvements but these would have to be tested.

This could be a promising approach but would only affect the P load released from sediments to the lake (i.e., the internal P load). If blooms are primarily fuelled by external nutrient sources (e.g., Carters Creek or groundwater), then this method is unlikely to be effective.

5.1.4 Aeration/oxidation

Effective aeration or oxidation of bottom water (hypolimnion) would provide enough DO to available iron (Fe) to bind PO_4 (the most bioavailable form of phosphorus). This would lock P to the sediments and prevent it from fuelling algal growth.

Unlike circulation and destratification, this technique does not break the thermocline (the boundary between warmer surface and cooler bottom water masses). This is considerably more expensive than destratification.

Oxygenation can be achieved by injecting pure oxygen from a lakeside oxygen plant into the hypolimnion in the form of very fine bubbles, e.g., the Speece cone (Speece et al. 1973). It is a technique used in deeper lakes in the northern hemisphere, where the cooler hypolimnion is important as a habitat for fish and a supply of cold water is required for end users. Lake Hood is probably too shallow to allow this technique to work properly. The running cost could be a major issue as the amount of oxygen required has to match or be greater than the water chemical oxygen demand (COD) and the sediment oxygen demand (SOD). Mixing using aeration only requires enough air to bring the bottom water to the surface where oxygenation occurs as diffusive exchange from the atmosphere.

5.1.5 Nanobubble technology

Nanobubble technology produces very small oxygen bubbles in the water, much smaller than those from the Speece cone. These nanobubbles are claimed to oxygenate and sterilize the water and kill cyanobacteria. It is also claimed to destroy toxins. The exact process has not been well defined but anecdotal evidence from shallow lake trials and a recent experiment by NOAA (https://coastalscience.noaa.gov/news/nccos-validates-nanobubble-technology-for-remediation-of-harmful-freshwater-algal-blooms/) suggests that it works but is still very much in the development stage. Early versions of the nanobubble devices were relatively expensive with multiple devices required to treat medium size lakes and reservoirs.

The technology is expensive at present, but costs should decrease once the product is widely accepted. To date, there is no peer-reviewed literature available to enable the product to be scientifically evaluated.

5.1.6 Drawdown

Drawdown or a drastic reduction of the lake water volume or even entirely emptying the lake would allow shallow exposed sediment to dry out and thus eliminate seed populations of cyanobacteria that cause harmful algal blooms.

The drying time of exposed lake sediments will depend on sediment depth, weather conditions, and soil composition. Deeper sediments are expected to take longer to dry out than shallow sediments. Dry, warm weather conditions allow sediments to dry quicker than cool, wet weather conditions allow. Fine sediments (e.g., of high clay content) may retain moisture longer than coarser sediments (e.g., sands).

Whether algae seed banks survive drying periods depends on the species' resilience, burial depth, temperature, and nutrient availability. For example, *Anabaena* seed banks were not able to germinate after three days of desiccation at 25°C (Tsujimura 2004). Some species can survive desiccation longer than others (Ellegaard and Ribeiro 2018). Seed banks that are buried deeper within the sediments may be better protected from desiccation than those in surface sediments. Large temperature fluctuations may reduce seed bank survival, while nutrient-rich sediments may prolong survival.

Negative sides of lake draining must also be considered. This approach may be feasible and potentially cost-effective but not aesthetically pleasing and would disrupt recreational use of the lake for weeks to months at a time. This would negatively affect fish, macroinvertebrates, and desirable macrophytes in lake. In addition, there may be a nutrient release pulse not long after refilling the lake, as nutrients released from biomass including phytoplankton, cyanobacteria seed banks, and weeds could in turn fuel new algal blooms (Carmignani and Roy 2017).

5.1.7 Dredging and benthic barriers

Dredging is an option to remove lake sediments that can release P during thermal stratification. There two forms to consider: (a) dry dredging of exposed sediment when the reservoir is low and (b) wet dredging, hydraulic or pneumatic dredging when the reservoir is full. This does not affect external nutrient loading from Carters Creek and groundwater that may fuel blooms.

Benthic barriers are applications of clay, silt, sand, and gravel from external sources to bury the surface nutrient-enriched sediment. This technique has been used for many years in the United States. While these barriers block the release of P from the sediment, they can also cause the sediment beneath the barrier to become strongly anoxic, thereby liberating toxic sulphide ions (e.g., hydrogen sulphide, H_2S), which can sterilize the phytoplankton seed banks in the sediment. Any H_2S released from the sediment is rapidly oxidised to sulphate and therefore is not a significant issue.

5.1.8 Sonication

Sonication breaks algae cells and may thus be used to treat existing blooms, but this does not remove the cause or driver of blooms. New blooms can therefore develop soon after sonication due to the release of nutrients from the decaying phytoplankton cells. This method uses a sonic pressure wave to rupture the gas vacuoles in the cyanobacterial cells, causing them to sink into light limiting conditions. *Dolichospermum* has gas vacuoles, which sonication could rupture and thereby cause the cyanobacteria to sink and potentially clear the local water column. In some studies, sonication has been shown to be highly selective, removing only those species with gas vacuoles.

This treatment has been used successfully in the Te Tahi drinking water reservoir in the Waipa district in Waikato, where sonicators have been used to remove a *Dolichospermum* sp. in bloom and have prevented its re-occurrence. A similar, small study using sonication on *Woronichinia naegeliana* (Bober and Bialczyk 2017) was conducted in Lake Rotoroa (Hamilton Lake). This study showed that sonication had the potential to remove or at least reduce the cyanobacteria population. However, sonication was not as effective on this species as hoped; the sonicators were unable to remove most of a surface scum population in a controlled laboratory environment after four days, although disruption to cells was apparent (Thompson 2011). This technique may be useful in curbing a nuisance bloom but requires a follow-up treatment to control the germination of the seed population being released from the sediments. Collateral damage may include the loss of some beneficial zooplankton grazers, which may also be killed by sonication. A potential downside to sonication is that because it ruptures the cells, it may also release algal toxins and is likely to release the taste and odour compounds otherwise held within the cyanobacteria cells. *Dolichospermum* is known to produce geosmin or 2-methylisoborneol, which are such taste and odour compounds (Bowmer et al. 1992, Blevins et al. 1995). The cost of sonication may be high as the zone of influence is relatively small and would require repeat treatment if seed banks in the sediments promote new blooms of the cyanobacteria fuelled from the sediments.

5.2 Chemical controls

5.2.1 Hydrogen peroxide

Hydrogen peroxide is an effective sterilising agent and would work much the same way as the nanobubble technique, but by applying the treatment as a liquid. This technique has potential as a targeted treatment. A whole lake treatment using hydrogen peroxide at a concentration of 2 mg L^{-1} in Lake Koetshuis, a small (12 ha), shallow (depth 2 m) lake in the Netherlands, was undertaken to control cyanobacteria (Matthijs et al., 2012). Laboratory experiments determined the rate of application as the minimum concentration required to remove cyanobacteria, while not harming zooplankton. After application, the levels of hydrogen peroxide dropped from 2 mg L^{-1} to 0.7 mg L^{-1} after 24 h and were below detection level after two days. There was an immediate decrease in cyanobacteria, with an 18–30% reduction within 3 hours and a 99% reduction within 10 days. Cyanobacterial biomass remained low for 7 weeks after treatment, only increasing after 7 weeks due to an input of water from another lake with the same species of cyanobacteria. Green algae and zooplankton species were not significantly affected by hydrogen peroxide addition.

5.2.2 Flocculation or sediment capping

This technique controls phosphorus release from the sediments during stratification (low oxygen concentrations in the hypolimnion) by means of flocculation and/or sediment capping. For this technique, a metal salt, either aluminium sulphate (alum) or lanthanum chloride (Phoslock®) is used on a bentonite carrier, also known as lanthanum modified bentonite (LMB). Both products have been used successfully and extensively overseas for limiting sediment P release and thus breaking the cyanobacteria growth cycle. Alum has an additional advantage over LMB in that it can remove a cyanobacteria bloom in a matter of hours by flocking and settling the bloom to the sediment, thereby clearing the water column. Flocculation occurs when a flocculant (or flocking agent, such as alum) causes suspended particles to aggregate and form a floc, which settles out more readily than individual smaller particles. According to a brochure distributed by PET Water Solutions, Phoslock® is effective between pH 5 and 9, effective under anoxic conditions, does not affect water pH and conductivity, increases sediment stability, and its binding capacity does not decrease with time; it thus could be appropriate for use in Lake Hood.

A recent innovation has been the development of the so-called flock-and-lock technique, where P and cyanobacteria are removed from the water column with a flocculant such as polyaluminium chloride and then locked in the sediment with Phoslock® (Lürling and van Oosterhout 2013). Another advantage of alum over Phoslock® that could benefit Lake Hood is that the alum floc on the sediment surface can strongly suppress or prevent germination of the algae seed bank in the sediments. Phoslock® has a weaker effect than alum (Kelly 2007).

This control option does not remove phosphorus, but it locks it in place. Multiple or regular applications may be required.

5.2.3 Phosflow

Phosflow is a pellet or bead form of Phoslock® that can be obtained in 1.8 kg or 3 kg bags from Aquatic Technologies. It is said to reduce phosphorus concentrations by up to 0.2 mg/L treating 450,000 L of water with one 1.8 kg pouch or treating 750,000 L of water with one 3 kg pouch. The pouches would be placed in the inflow and could last up to 2 months. After use, the phosphorus-saturated beads “can be repurposed as slow-release fertilisers or for composting, minimising environmental impact” (aquatictechnologies.com.au).

This could be an option for phosphorus removal in Carters Creek, potentially in the inflow culvert (taking water from the Ashburton River) and in the canals of Lake Hood, but the placement, number of pouches, and the requirements for disposal and replacement must be determined.

5.2.4 Algicides

Algicides are chemical compounds intended to kill unwanted algae. They are usually copper-based and not suitable for a drinking water supply reservoir. There can be unwanted or unintended ecotoxicological effects or secondary pollution (Hamilton and Patil 2022).

5.3 Biological controls

5.3.1 Weed harvesting

Removing invasive weeds can prevent anoxia of bottom waters and release of bioavailable nutrients after weed beds collapse and decay. Native turf communities could then become established, and these are often associated with better water quality. Weed removal can be achieved by using a mechanical weed harvester. Weeds removed from the lake should be appropriately disposed of (e.g., composted).

In 2004, two thousand grass carp were introduced to control weeds, but this control is now considered insignificant (McCracken et al. 2023). A mechanical harvester was hired from the Christchurch City Council for two seasons, but this became untenable as *Egeria* developed in

Christchurch waterways and this was not present in Lake Hood and its introduction to the lake was to be avoided. Thus, weeds in the canals were subsequently sprayed with Diquat starting in 2010, but costs are high and decomposing weeds result in nutrient release and subsequent recycling by aquatic plants and algae. Extensive weed growth in the canals inhibits water movement.

Weed harvesting and removal from the lake has been suggested by Donna Sutherland. This would not only remove the flow-inhibiting weeds in the canals but also prevent the recycling/uptake of nutrient that would be released from decomposing weeds if they were left in the canals. There remains a risk that nutrient loads from inflows and sediments (in anoxic conditions) could fuel algal blooms.

5.3.2 Biomanipulation

High Tech SEBS Bacterial Culture by Bio Control Solutions has been proposed to the Task Force. According to advertising material, the bacterial culture contains more than 5 billion organisms per mL (Figure 5-1). Some of these bacteria not directly target cyanobacteria but could indirectly affect them by altering nutrient dynamics in the lake. Some strains may produce compounds with algicide properties which could inhibit cyanobacteria growth. Introducing these bacteria could potentially alter the nutrient dynamics of the lake, affecting the bioavailability of nitrogen, phosphorus, and other nutrients to algae. These bacteria may compete with each other and with cyanobacteria for nutrient sources and some may consume cyanobacteria or produce compounds that inhibit cyanobacteria growth. The effectiveness of bacterial treatments depends on environmental conditions (pH, temperature, dissolved oxygen levels, nutrient concentrations).

As with any non-native species introductions, the introduction of non-native bacteria to Lake Hood carries risk. Non-native species may become dominant, disrupting native microbial communities, or produce harmful metabolites. Before a decision is made to introduce any bacteria species, we recommend a detailed literature review on the potential unintended consequences, controlled laboratory and field trials, and consultation with regulatory agencies.

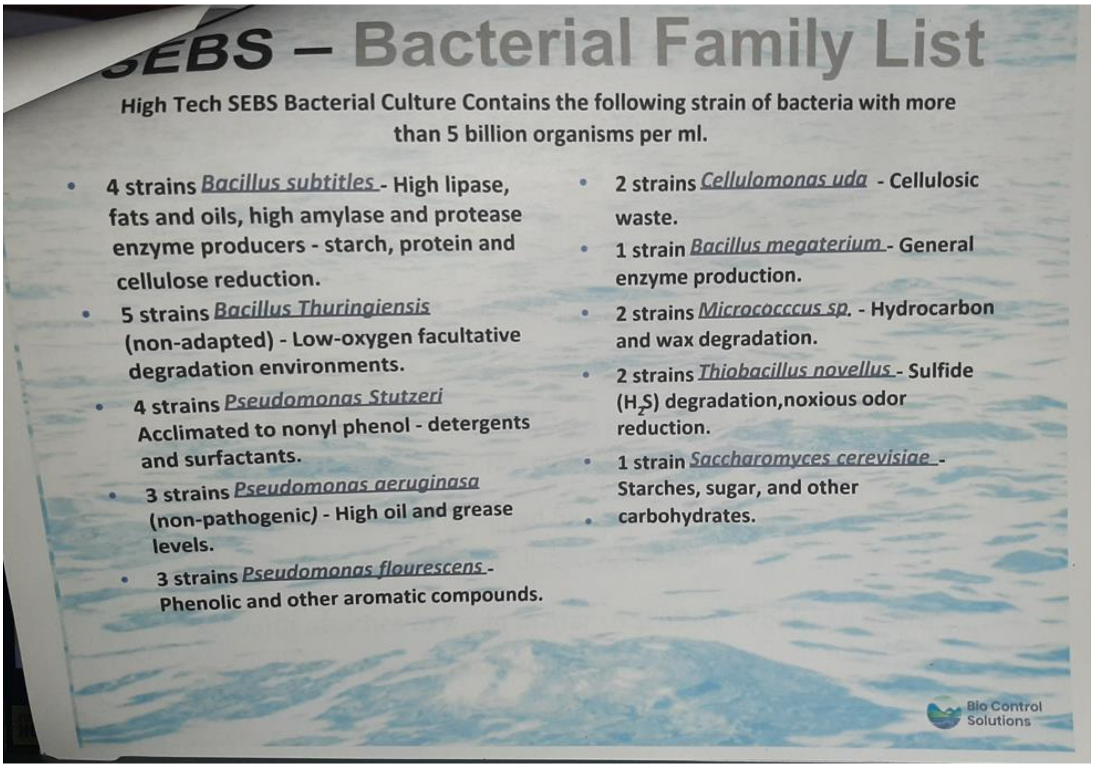

Figure 5-1: Advertising material for a bacterial solution by Bio Control Solutions. Image provided by Les McCracken.

Specific enzyme bacterial solution (SEBS) products to control toxic algae have been developed from natural bacteria which can inactivate cyanobacteria and digest the final, inert biomass. We are aware that two such products have been tested in laboratory trials using lake water from a residential, man-made lake that is affected by cyanobacteria blooms. Two treatments were tested: addition of SEBS AFS630 NZ (a surface biomass treatment) and AIO405 NZ (a water column specific microbial blend). One treatment alone was not as effective as applying the two treatments simultaneously. Sampling on days 7 and 34 after treatment, compared to initial conditions, showed large increases in the number of cyanobacteria species and overall biomass (cells/volume and biovolume) present in the control (untreated) sample, compared to much smaller increases in cyanobacteria in the treated samples. The single SEBS treatment was not as effective as the combination of two SEBS. We note that the results of these trials have not been published.

Bay of Plenty Regional Council (BOPRC) has used MuckBiotics bags (13.6 kg) containing < 3.5 x 10^{13} bacteria cells and other products from Parklink Ltd (Parklink Ltd - Water and Wastewater Treatment - New Zealand), as long as their application met consent conditions for the amount of bacteria applied (2.1 x 1014 cells per month for six months per year). The types of bacteria were limited to *Bacillus*, *Nitrosomonas*, *Nitrobacter*, and *Pseudomonas* as per the council's resource consent, first issued in December 2015 (pers. comm., Justine Randell). The biotreatment was applied in Ōtautū Bay, Lake Rotoehu, and initiated by the local community for the purpose of improving water quality in the bay. The consent allows for the introduction of bacteria to accelerate natural degradation and thus reduce sludge building up on the sediments and enabling contact recreational use of the bay area. Monthly sediment monitoring indicates that the biotreatment did not affect sediment biota (specifically chironomids). Sediment samples are analysed for N, P, and C content. Records spanning 2010-2023 show significantly lower sediment N, P, and C content since 2016 (Randell, Compliance Summary Report 2022/2023 Resource Consent 68488, Condition 8): TN content decreased from > 1.6 g/100 g dry mass to < 0.2 g/100 g dry mass, TP content decreased from > 0.12 g/100 g dry mass to ~

0.02 g/100 g dry mass, and TC content decreased from > 10 g/100 g dry mass to < 2 g /100 g dry mass. Weekly lake cyanobacteria monitoring in 2020-223 indicated that there were several days when potentially toxic biovolumes were high (> 1.8 mm^3/L, red alert level). The cyanobacteria alert levels are for Lake Rotoehu overall and not specific to Ōtautū Bay. The Water Quality Technical Advisory Group, to whom monitoring results from the biotreatment programme were presented, recommended that the biotreatments are ceased, because there was not enough evidence to suggest that these treatments in a small area of the lake were useful. No control area was monitored for sediment nutrient content, and there are concerns that while the bacteria may accelerate biomass decomposition, this might release nutrients back into the water column where they are available for uptake by cyanobacteria blooms washed into the bay from the open lake. Instead of localised biotreatments, the application of alum to the whole lake is now being considered.

5.3.3 Floating wetlands

Floating wetlands are raft structures that can float on a lake surface and support plant growth which removes nutrients from the lake water. Specifically, nitrogen is removed by denitrification. Well-designed constructed floating wetlands can effectively remove contaminants and enhance biodiversity (Bi et al. 2019). Floating wetlands tend to be bespoke for a given water body and quantification of nutrient removal can be complicated or not feasible (Bi et al. 2019, Hamilton and Patil 2022).

5.4 Other options

Natural lake mixing by wind might be enhanced by removing structures or trees around the lake, enabling wind to vertically mix the lake more frequently. Given that Lake Hood was constructed for residential lakeside properties, this is likely not an option.

Booms or other skimming structures could be constructed in the canals to remove surface scum. This may not be aesthetically pleasing but could be immediately effective in reducing or mostly removing an algal bloom.

If Carters Creek were diverted through a constructed wetland, then natural nutrient removal by wetland organisms could occur before that water enters Lake Hood. This would require significant construction and the effectiveness of nutrient removal could be difficult to quantify. Long hydraulic residence times and potentially a large land area might be required for such a wetland to be effective. This option would require a detailed investigation before construction.

5.5 Application to Lake Hood

Of these techniques, P-inactivation using alum likely has the most promise because it would remove the existing cyanobacteria bloom, inactivate the P being released from the sediment during stratification and cap the seed bank in the sediment. This would allow the implementation of an aeration system without the risk of stimulating an algal bloom when the aerator is turned on.

While a treatment with alum would reset the lake in the very short term, it may not be appropriate as a long-term solution due to cost and the public perception that a chemical is being added to the lake.

If used as described and applied with the correct amount of buffer, alum is harmless but very effective against cyanobacteria and for managing the internal P load of a reservoir. It is the same chemical used in almost every town and city water treatment plant for flocking the suspended solids out of the domestic water supply.

6 Conclusions and recommendations

Cyanobacteria have been observed in Lake Hood in 2023 and 2024. Blooms started in poorly flushed canals on the west side of the lake and were later found in the ski lane on the eastern lake shore (pers. comm., David West and Les McCracken, 19 January 2024). We considered time series of changes in meteorological conditions (i.e., air temperature and wind speed), water temperature and dissolved oxygen profiles and time series, water level, pH, total suspended solids, and nutrient and chlorophyll *a* concentrations at several monitoring locations in and around the lake (surface water, groundwater, sediments, inflows). Our key findings based on the available data are:

- The highest TN concentrations were at Carters Creek and well 2 and the highest TP concentrations were at the lake intake and Carters Creek. In Lake Hood, DRP has been low at the outlet (< 0.005 mg/L) but much higher in the intake and Carters Creek. Sediment samples collected in September 2023 show that total phosphorus content was moderate (highest in lake intake), and total nitrogen content and total organic carbon was low. We did not identify the main contributing nutrient load to the lake, but it appears that the lake intake, Carters Creek, groundwater, and sediments are all relevant nutrient sources. A detailed water balance and assessment of nutrient loads (flows × nutrient concentrations) would be required to determine the dominant nutrient source to the whole lake and/or to specific subbasins like the western canals.
- Profiles of temperature and oxygen show that stratification occurs at least occasionally for short periods in the lake and the dissolved oxygen concentration at the bottom can be close to hypoxic (DO < 2 mg/L). The trial aeration setup did not appear to prevent water column stratification and occurrence of hypoxia.
- A time series of pH shows that pH has increased since 2015, and it was > 10 on several occasions, including at the lake outlet in 2023. The dissolved oxygen concentration has also decreased in the same period. pH >10 or low oxygen concentrations can result in increased phosphorus release from the sediments.
- Mean air temperature has increased significantly since 2007, while there was no significant trend in the mean wind speed, which likely does not affect mixing in the water at sheltered locations. High air temperature can have a strong impact on cyanobacteria population dynamics.
- *Dolichospermum* is the dominant cyanobacteria genus identified in samples collected in 2023 and 2024. These cyanobacteria have been associated with toxins and can regulate their buoyancy to optimise their position in the water column with respect to light and nutrient availability. They fix nitrogen from the atmosphere, which gives them another advantage over other phytoplankton. They also produce seed banks which can survive harsh environmental conditions and seed new blooms.
- The data and literature suggest that the bioavailable P concentration must be at least less than ~0.020 mg/L to reduce growth, but this may not eliminate *Dolichospermum* blooms in Lake Hood. Targeted investigations and experiments would be needed to confirm P limitation and refine the estimated threshold of ~0.020 mg/L P to restrict growth of the *Dolichospermum* species in Lake Hood.

We described several potential mitigation and control options and suggest consideration of the following:

- **Flushing:** Increased water takes from the Ashburton River may reduce the hydraulic residence time and thus flush unwanted algae out of the lake and improve in-lake circulation, but reducing nutrient concentrations in the main water source of the lake (i.e., the Ashburton River) will likely have the greatest positive effect in reducing algal blooms throughout the lake over time, provided that legacy phosphorus stores in the sediments are capped.
- **Reducing inflow nutrient loads:** As previously recommended by Tonkin and Taylor (2008), "Controlling nutrient input loads is likely to represent the best and most sustainable long-term option for maintaining or improving water quality in the extended lake". We agree that increasing flow through the lake and reducing inflow nutrient concentrations or diverting nutrient-rich ground and surface water inflows should help improve the trophic state of the lake, but we also recommend capping the lake sediments to prevent P release during high pH and/or low DO conditions. **Trialling Phosflow in Carters Creek** with control measurements (sampling upstream and downstream of treatment) would allow for an assessment of the effectiveness of the approach.
- **Sediment capping**: Application of a metal salt (alum or Phoslock®) can prevent P release from the sediments. Alum is also a flocking agent that can remove a cyanobacteria bloom in hours by flocculation and settling flocs to the sediment. Alum floc on the sediment surface can strongly suppress germination of algae spores (the "seed bank") in the sediments.
- **Sonication**: Sonicators can be used to break cyanobacteria cells using sonic pressure waves to rupture the gas vacuoles in the cells. This could be very effective in the summer. Reseeding of populations can also be prevented this way.

Based on the currently available data, the main source of phosphorus cannot be identified (i.e., inflow loads vs sediment P flux). Therefore, if the inflow loads are dominant phosphorus capping may not be effective.

Models are important tools that could be developed and used to determine the effects of climate change (e.g., changes in air temperatures and wind speed), changes in nutrient loading, and management options affecting cyanobacteria blooms in Lake Hood. Knowing which of the nutrient loads to the lake is the main contributor to lake water column nutrient concentrations, and in turn the main driver of cyanobacteria blooms, would allow for targeted nutrient management. Predicted outcomes for possible future scenarios allow for effective lake water quality management. Monitoring data are required to calibrate and validate models.

Before implementation of any large-scale control efforts, we recommend the following next steps:

1. Modelling

   a. A **hydrodynamic model** would enable estimation of residence times in different parts of the lake, i.e., the canals and the main lake, which will differ. A key preliminary step is to calculate the lake **water balance** from inflow, outflow, rainfall, and water level data spanning at least one year but ideally several years.

   b. A linked catchment and lake **water quality model** would quantify the nutrient loads from each source and estimate sediment nutrient fluxes over time, allowing for determination of the main nutrient source driving cyanobacteria blooms. A catchment model would estimate daily nutrient loads from external sources (Ashburton River inflow, Carters Creek). A groundwater model or a simpler analysis (groundwater inflow × nutrient concentration) would estimate the groundwater nutrient load to the lake. A lake water quality model with a sediment module would estimate nutrient fluxes from the sediments over time.

   c. **Scenario modelling** using the linked models would allow for testing of different management options (e.g., reduced nutrient loads, sediment capping) before implementation.

2. Data collection for model development

   a. Regular (at least monthly) **water column P and N sampling in the canals** where blooms are most likely to form.

   b. **Field measurements of fluxes of P from the sediment** for comparison with inflow nutrient loads.

   c. Controlled laboratory or mesocosm **experiments** using cyanobacteria from Lake Hood to describe the growth rate as a function of N and P concentration. This information would also help refine suggested lake nutrient concentration targets.

3. Ongoing water quality monitoring is required to determine the effectiveness of any mitigation measures. Current monitoring is unlikely to be adequate for assessing mitigation performance. Any choice of measures must include consideration of regulations, social and cultural values, available funding, long-term feasibility, and desired outcomes.

7 Acknowledgements

The Lake Hood Task Force agreed the scope of work and Ashburton District Council and Ashburton Aquatic Park Charitable Trust jointly funded this work. Les McCracken provided all the data analysed in this report. We thank Les McCracken and David West from the Task Force for their prompt responses to questions regarding this work. We thank David Plew for his reviews and helpful suggestions that greatly improved this report.

8 Glossary of abbreviations and terms

ADC	Ashburton District Council
BOD	Biochemical oxygen demand
Chl *a*	Chlorophyll *a*
DO	Dissolved oxygen
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
DRP	Dissolved reactive phosphorus
E. coli	*Escherichia coli*
Fe	Iron
MfE	Ministry for the Environment
N	Nitrogen
N_2	Atmospheric dinitrogen
NH_3	Ammonia
NO_3-N	Nitrate nitrogen
NO_2-N	Nitrite nitrogen
P	Phosphorus
PO_4	Phosphate, a readily bioavailable form of phosphorus
SS	Suspended solids
TLI	Trophic level index
TN	Total nitrogen
TP	Total phosphorus

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