Sullivan Lake Water Quality, Ecology and Options for Improvement

Prepared for:

Whakatāne District Council

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Prepared by:

K. D. Hamill (River Lake Ltd)

For information regarding this report please contact:

Keith Hamill

Phone: +64 27 308 7224

Email: keith@riverlake.co.nz

River Lake Ltd

Ground Floor, 13 Louvain Street, Whakatāne, New Zealand

Web: www.riverlake.co.nz

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Cover Photo: Evening at Sullivan Lake, Whakatāne, May 2022.

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Executive Summary

Whakatāne District Council (WDC) has responsibility for managing Sullivan Lake. In order to inform the management of Sullivan Lake, WDC commissioned science investigations to: a) provide robust information on water quality and ecology values, and b) identify key management options for improving water quality and ecological values in Sullivan Lake.

Sullivan Lake condition and values

Morphology

Sullivan Lake is a small, shallow, urban lake located in Whakatāne township. It has an area of 2.7 ha, mean water depth of less than 1m and maximum depth of 2.2m. The lake substrate is predominately soft silts that have an average depth of 0.4m. The lake's catchment area is about 105ha of which about two thirds is in urban landuse and the rest is steep escarpment.

Hydrology

The average hydraulic residence time for Sullivan Lake is 11.4 days; however, because there is limited baseflow, there is little flushing outside of rain events. WDC pumps water from the Whakatane River into Sullivan Lake to improve flushing, and there is potential to increase the flushing during summer dry periods.

Birds

Sullivan Lake provides habitat for a wide range of waterfowl. While birds are an important value of the lake, high densities of birds can reduce water quality. Outbreaks of avian botulism occasionally occur in the lake. There are a complex set of factors associated with avian botulism outbreaks, but one practical action that can help reduce the severity of an outbreak is to collect and dispose of bird carcasses affected by avian botulism.

Fish

Fish present in Sullivan Lake and its catchment include: shortfin eel, common smelt, common bully, inanga, and the introduced fish goldfish and *Gambusia*. However, connection to the Whakatāne River for migratory fish (shortfin eel and inanga) is restricted by the outlet weir and flood gate.

Plants

Mexican water lily commonly covers a large area of Sullivan Lake in the west end near King Street. Apart from the waterlily, aquatic macrophytes have been absent from Sullivan Lake for many years. However, curled pondweed (*Potamageton crispus*) has recently re-established in Sullivan Lake. This is an annual aquatic plant that grows from seeds during late spring, proliferates across the lake during early summer and collapses during mid-summer.

Aquatic plants are a key to maintaining good water quality in natural lakes, by regulating water quality, stabilising sediments, and providing habitat for invertebrates and fish. Studies have found that greater than 30% plant cover is required to maintain a clear-water state. The recent occurrence of *P. crispus* in

Sullivan Lake provides an opportunity to improve the water quality by harvesting. If harvesting is not undertaken prior to collapse of macrophyte beds, then adverse effects on DO could be minimised by increasing the volume of flow augmentation during this time.

Water quality

The water quality of Sullivan Lake is poor; with low water clarity and high concentrations of nitrogen, phosphorus and phytoplankton. Cyanobacteria blooms are very common in Sullivan Lake during summer and autumn, and exceed recreational use guideline (Action mode) about 62% of the time. The dominant cyanobacteria is *Anabaena* sp. The Trophic Level Index (TLI) (6.0) is borderline between supertrophic and hypertrophic.

Phytoplankton growth in Sullivan Lake is more strongly limited by nitrogen than by phosphorus. Occasions with high TN are associated with cyanobacteria blooms – which can fix nitrogen from the atmosphere.

Management interventions that reduce nitrogen loads have good potential to be successful in Sullivan Lake (e.g. wetlands), but management options to remove or bind phosphorus may also be needed to control cyanobacteria blooms in the long term.

Dissolved oxygen and pH regime

The DO regime in Sullivan Lake has high temporal and spatial variability. The lake often has very large diurnal fluctuation in DO due to algae blooms, but the DO regime also appears influenced by heavy rain flushing algae biomass, BOD loads associated with stormwater, the growth of macrophytes curled pondweed moderating phytoplankton biomass, the collapse of curled pondweed exerting an oxygen demand, and aeration from strong winds. At the western end of Sullivan Lake, where waterlily was prevalent, pH and DO concentrations were lower and daily fluctuations smaller than in the main body of the lake. This is likely due to both oxygen demand of organic sediments, and shading by waterlily supressing algae growth.

Interventions to improve water quality

Nine potential management options were identified to address ecological issues associated with Sullivan Lake. The management interventions recommended as highest priority to improve water quality and ecology in Sullivan Lake were:

- Increase the volume of flow augmentation during summer.
- Treatment wetlands to remove nutrients and improve biodiversity.
- Partially dredging the southern end to remove organic sediments and manage water lily extent.
- Bottom-liners to contain the spread of water lily following partial removal.
- Harvesting macrophyte to manage plant cover and remove nutrients – early summer if sufficiently abundant.

The management interventions to improve water quality that could be considered but are less cost-effective or require more investigation are:

- Various measures to reduce catchment sediment and nutrient loads.
- Floating wetlands to remove nutrients and improve biodiversity.
- Sediment phosphorus locking to reduce internal load of P (e.g. applying alum).

The management interventions not recommended for Sullivan Lake at this stage are due to practical difficulties or their likely limited benefits are:

- Spraying macrophytes with herbicide to manage plant cover.
- Grass carp to control aquatic plants.
- Silver carp to control phytoplankton.

There is no single quick fix to improving water quality in lakes, there is no “magic bullet”, but there are effective actions that can shift Sullivan Lake towards a healthier ecosystem. The high priority actions would address multiple issues in a cost-effective way, and with low risk of adverse effects. In particular:

- Increasing the volume of flow augmentation during summer (January to March) is a cost-effective way to increase flushing of phytoplankton and help reduce algae biomass. This would be best undertaken in conjunction with treatment wetlands. Flow augmentation during winter, is not recommended due to elevated nitrate in the Whakatāne River.
- The treatment wetlands will help trap sediment and external nutrients while providing biodiversity benefits.
- Partially dredging the southern end of the lagoon will remove organic sediments which contribute to low dissolved oxygen and internal nutrient loads, while also managing the extent of the water lily cover.
- Placing bottom liners on the lake bed after excavation will slow the expansion of the water lily while retaining core areas of water lily for their ecological and water quality benefits.
- Allowing the growth of curled pondweed during spring and harvesting in early summer is an opportunity to both improve water quality and permanently remove nutrient from the lake system.

Contents

1 Introduction

1.1 Background

Whakatāne District Council (**WDC**) has responsibility for managing Sullivan Lake. In order to inform the management of Sullivan Lake, WDC commissioned science investigations to: a) provide robust information on water quality and ecology values, and b) identify key management options for improving water quality and ecological values in Sullivan Lake.

This work is being undertaken by River Lake Ltd in partnership with NIWA and WSP Ltd. In this report we:

a. Describe the geographical context of Sullivan Lake (including hydrology, morphology).

b. Describe the current state for water quality and ecology.

c. Identify the key issues for Sullivan Lake with respect to water quality and ecology.

d. Describe and prioritise potential management actions to address the key issues.

Pre-feasibility assessments have been prepared for key management options. These assessed the benefits, risks, cost-effectiveness and application to Sullivan Lake, so as to inform prioritisation of action.

1.2 Location and Context

Sullivan Lake is a small (2.7ha), shallow (maximum depth 2.2m), urban lake located in Whakatāne township. It consists of two basins joined by a short channel and has two small vegetated islands (**Figure 1.1)**. The lake has a catchment area of about 105ha of which about two thirds is in urban landuse and one third is the tree covered escarpment east of Valley Road.

The water levels are controlled by a weir at King Street, from which the water flows under King Street into a drain with a gravity discharge to the Whakatāne River (water is pumped during high river flows).

1.2.1 Historical context

Sullivan Lake was originally a naturally oxbow of the Whakatāne River that developed into an oxbow lake wetland system. It was developed from an oxbow wetland into a more formal lake, and set aside as a reserve, when the area was subdivided into residential sections in the 1960s (**Figure 1.2** and **Figure 1.3**).

1.2.2 Management

The Sullivan Lake Reserve Management Plan (2015) provides objectives and policies for the management of Sullivan Lake Reserve. It identifies five goals in managing the reserve:

1. To manage and enhance conservation values.
2. To manage and improve water quality.
3. To actively manage vegetation and open space areas.

4. To provide for a range of passive recreation activities.
5. To plan and manage the effects of utility infrastructure on the reserve.

WDC has undertaken a number of activities to help achieve the goals of enhancing conservation values and water quality. These include:

- Pumping up to about 40 m^3/hour of water from the Whakatāne River into the lake to provide flushing and help improve the water quality.

- Planting fringes of native riparian wetland plants have been established along sections of the lagoons southern side.

- Forming a sediment trap, consisting of a low bund, near the footbridge at the western end of the lake and downstream of the main stormwater inputs. This report includes a discussion of how this feature can be enhanced and improved.

- Removal of fine sediment from eastern end of Sullivan Lake in 2019 using a suction dredge.

Sullivan Lake and reserve are an integral part of the stormwater management network by providing live storage to attenuate peak flows during heavy rain events.

A wastewater pumpstation is located near the foot bridge at the eastern end of the lake. Wastewater over-flows during severe rain-events have occurred in the past, but the risk of this is now low since the pipes and pumping capacity at Douglas Street was upgraded in 2012.

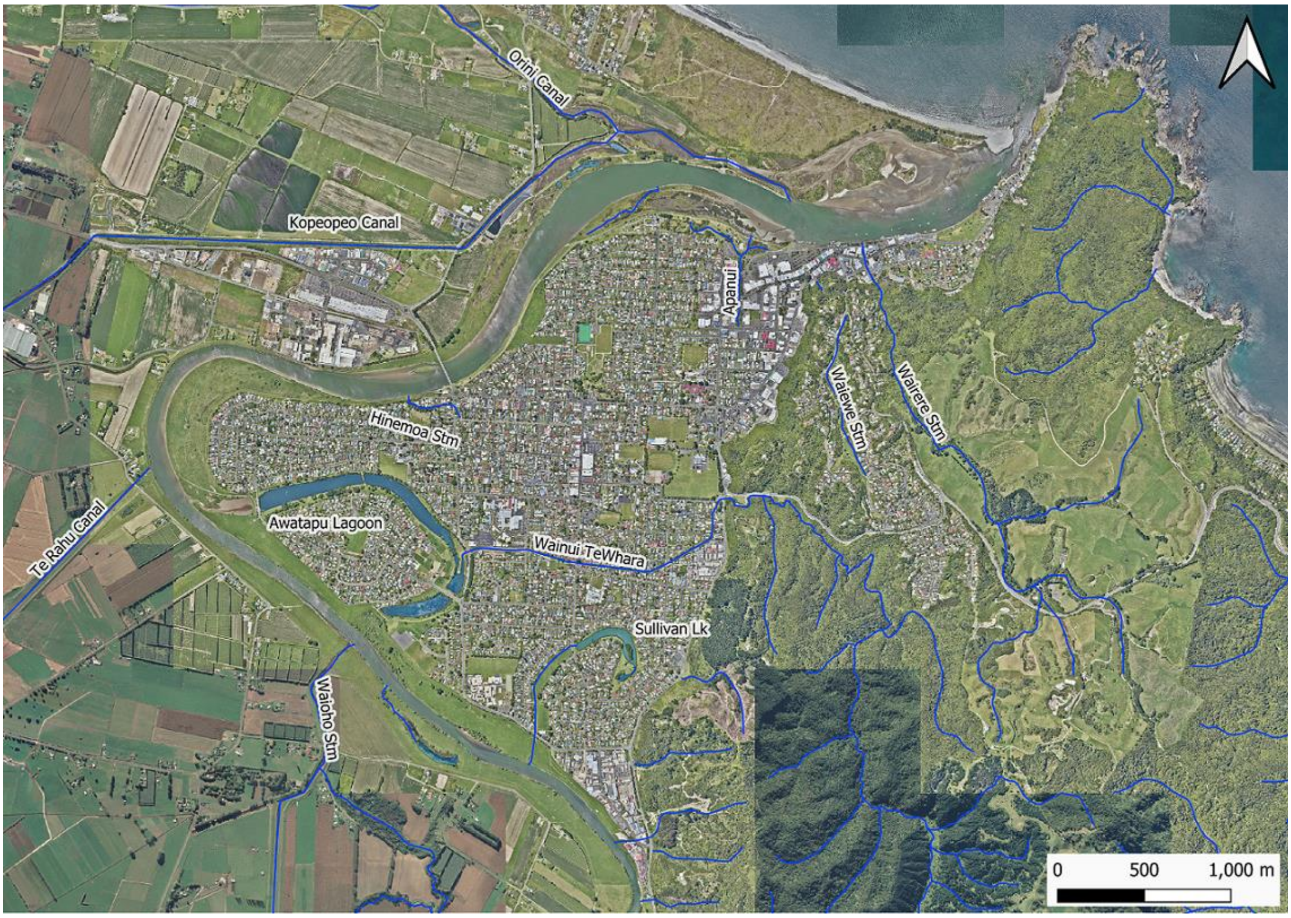


Figure 1.1: Location of Sullivan Lake and stream networks in Whakatāne township.

Figure 1.2: Aerial photos of Sullivan Lake in 1962 (before subdivision while still a wetland system) and 1974 (after subdivision when it was formed into a lake) (Source: Retrolens).

Figure 1.3: Aerial photos of Sullivan Lake in 1982 (top) and 2022 (bottom) (Source: Retrolens).

2 Methods of investigation

The descriptions of Sullivan Lake water quality and ecology used in this report is a synthesis of information from existing reports, analysis of historic datasets and specific investigations and monitoring collected as part of this project.

This project undertook multiple investigations in Sullivan Lake to inform our understanding of the waterbody and key mitigation options, these included:

- Bathymetry survey of the lake using a combination of sonar and manual measurements amongst areas of water lily) which were combined into a Digital Elevation Model (**DEM**).
- Dissolved oxygen, pH and temperature spatial surveys to characterise spatial variability.
- Dissolved oxygen and temperature loggers to characterise diurnal variability.
- Water quality samples of Sullivan Lake surface water during the summer of 2022/23.
- Fish presence using eDNA in Sullivan Lake and inflow stream.

2.1 Bathymetry

A bathymetry survey of the lake was undertaken for Sullivan Lake in June 2022. This used a combination of sonar and manual measurements (n=49) using a 'weighted line' where sonar readings were not practical due the density of water lily. The sonar readings were collected by WDC staff using depth sounder installed on a remote-controlled boat.

A comparison of depth measurements from the lead-weight method and the sonar method found that the lead weight method read was about 0.15m deeper than the sonar method. Manual measurements were reduced by 0.15m to provide consistency with sonar data. The discrepancy may be due to the sonar being mounted below water level, or the sonar bounding off a soft sediment layer that was penetrated by the 'weighted line'.

The sonar data was processed by removing duplicate data (8638 data points remained after cleaning), manually shifting the data to fit within a New Zealand co-ordinate system. The water edge was defined using the Bay of Plenty Regional Council (BOPRC) digital elevation model and assigned a depth of 0.2m to reflect the vertical edging around most of Sullivan Lake. The three sets of depth points (depth sounder, bathymetry and water edge) was used as inputs into a IDW interpolator to create a Digital Elevation Model (**DEM**), that was merged with the BOPRC DEM (**Figure 2.1**). The resulting raster had a water depth per 1 m^2.

Sediment depth was measured at the location of each manual depth reading by measuring the depth that a blunt probe could be pushed into the sediment using a constant pressure.

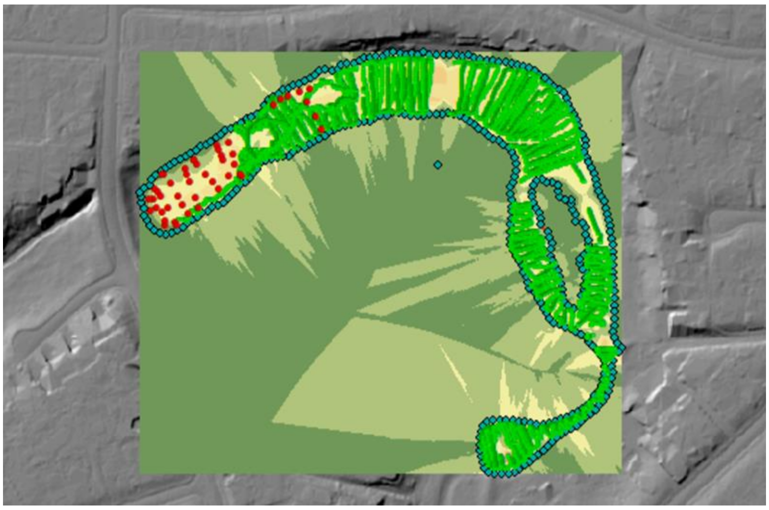


Figure 2.1: The three sets of depth points used to create a Digital Elevation Model for Lake Sullivan. Depth-sounder (green), lead-line (red), and water margin (blue).

2.2 Spatial variability of dissolved oxygen and pH

Synoptic surveys were undertaken in Sullivan Lake to characterise the spatial variability of dissolved oxygen (**DO**), pH and temperature. The surveys were on 5 April 2022 in the early morning (*c.* 6am to 7am) and on 9 April 2022 in mid-afternoon (*c.* 3pm to 4pm). The early morning and afternoon surveys correspond to when diurnal fluctuations of DO and pH are respectively near their minimum and maximum values.

The measurements were collected from a kayak at a sample depth of *c.* 0.2m, using a YSI Pro Plus multi-meter with a polarographic DO sensor. The sample location was recorded using a GPS tracker and linked with each measurement using the date-time stamp. Prior to the survey the time was synchronised between devices, and the multi-meter was calibrated for both DO (at 100% saturation) and pH (three-point calibration).

2.3 Temporal variability of dissolved oxygen

The temporal variability of DO and temperature in Sullivan Lake was characterised by using a Hobo U26 optical dissolved oxygen logger located mid-lake, about 80m west of the Olympic Drive footpath access. The logger was attached to a buoy with the sensor about 0.35m below the water surface. The water depth at this location was 0.95m deep.

The logger was installed on two separate occasions; for 11 weeks in late autumn (from 9 April 2022 and 24 June 2022), and again for eight weeks in early summer (from 14 December 2022 to 8 February 2023).

The DO logger was calibrated before and after deployment using 100% water saturated air. As a further check, separate measurements of dissolved oxygen were made when installing, removing and checking the logger using a calibrated YSI Pro Plus multi-meter with a polarographic DO sensor.

Atmospheric pressure was recorded near the site using a Hobo U20 logger (measuring pressure and temperature). These measurements were used to adjust DO measurements for atmospheric pressure.

Measurements of temperature, DO, pH and electrical conductivity were made at the top (0.1m) and bottom (0.8m) of the water column when the loggers were installed and removed. The top and bottom measurements were similar and there was no thermal stratification – as would be expected in such a shallow lake.

2.4 Water quality sampling

BOPRC undertook regular water quality sampling of Sullivan Lake (at outlet) between September 2001 and June 2008 (including periods of weekly to fortnightly sampling during 2001 to 2003, and in 2007).

BOPRC also collected cyanobacteria samples from Sullivan Lake during summer between February 2013 and February 2020. The frequency of sampling ranged from weekly to monthly. Samples were analysed for species identification, biovolume and potentially toxic biovolume.

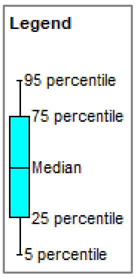
More recent water quality samples were collected as part of this Project and in a co-founded collaboration between River Lake and BOPRC. Water samples were collected from the edge of Sullivan Lake at Olympic Drive footpath access. This occurred on seven occasions between May 2022 and April 2023.

The samples were collected using a sample arm to reach about 2.5m from the water's edge and 0.1m below the surface water. Field measurements (temperature, dissolved oxygen (**DO**), specific electrical conductivity (**EC**), pH) were made from the same location using a YSI Pro Plus multi-meter, and clarity tube. The samples were sent to Bay of Plenty Regional Council laboratory for analysis of: total nitrogen (**TN**), nitrate-nitrite-nitrogen (**NNN**), total ammoniacal nitrogen (**NH4-N**), total phosphorus (**TP**), dissolved reactive phosphorus (**DRP**), turbidity (**TURB**), chlorophyll-a (**Chl-*a***), and *E.coli* bacteria (***E.coli***).

Additional field measurements have been collected on an *ad hoc* basis, including water clarity measurements by the Sullivan Lake Care Group and River Lake Ltd collected between March 2019 and February 2020.

Water clarity was usually measured using a clarity tube. This data was converted to black disc water clarity using the formula provided in Kilroy and Biggs (2002)[1].

Water quality data expressed using box plots show the median, interquartile range, 5 percentile, 95 percentile, minimum, and maximum, as illustrated below.



[1] Clarity tube reading (yCT) < 50cm = black disc (yBD); yCT >50cm adjusted as: yBD = 7.28 x 10^(yCT/62.5).

2.5 eDNA

Waterways contain environmental DNA (eDNA) of organisms present. Analysis of eDNA shed from organism in the water give a qualitative assessment of what fish, aquatic insects, birds and plants may be present (David et al. 2021). Although used as a qualitative tool, the results do indicate the strength of the eDNA signal.

Fish presence in Sullivan Lake and inflow streams was confirmed by collecting eDNA samples from Sullivan Lake on 14 June 2022 and from the main stream entering Sullivan Lake at the foot bridge on 17 November 2022. Samples from the lake were collected from three locations on the northern side. The flow that is normally pumped by WDC from the Whakatane River had been stopped for six days prior to collecting the sample so as to avoid potential inflow of fish eDNA from this source.

Preservative was added to the samples and they were sent to Wilderlab for processing.

2.6 Assessing potential nutrient limitation

In order to accurately assess the extent to which nutrients may limit algal growth in a lake requires detailed investigations and bioassays. However, some indication of potential nutrient limitation can be gained by looking at the absolute concentration of nutrients in the lake and the stoichiometric ratio of N to P and assuming the absence of other factors limiting phytoplankton or macro-algal growth. Nutrient concentrations are balanced when they equate to the Redfield ratio (i.e., 7.2 by mass). In these situations, either or both N or P may limit growth. A TN:TP value less than 7 indicates potential nitrogen limitation, and a TN:TP value greater than 14 indicates potential phosphorus limitation.

Similarly, the ratio of DIN:TP can also be used to indicate potential nutrient limitation. Assuming the absence of other growth limiting factors a DIN:TP of < 1 (by mass) indicates potential N limitation and a DIN:TP > 1 indicates potential P limitation (Schallenberg et al. 2010).

2.7 Lake water quality guidelines

2.7.1 Trophic Level Index (TLI)

Lake water quality is often expressed in terms of trophic state, which refers to the production of algae, epiphytes and macrophytes in a lake. The trophic state of each lake was assessed using the Trophic Level Index (**TLI**) (Burns et al. 2000).

The TLI integrates four key measures of lake trophic state - total nitrogen, total phosphorus, chlorophyll *a* and Secchi depth. The overall TLI score for a lake is the average of individual TLI scores for each variable. The overall score is categorised into seven trophic states indicative of accelerated eutrophication as evidence more nutrients, more algal productivity and reduced water clarity (**Table 2.1**). Regular monitoring over multiple years is usually required to reliably characterise a lake's water quality or TLI.

There were few measurements of Secchi disc from the lake, so the TLI was expressed as **TLI3**, which excludes the Secchi disc measurements.

Table 2.1: Definition of Trophic Levels based on water quality measures (Burns et al. 2000).

Trophic State	TLI Score	Chl *a* (mg/m^3)	Secchi depth (m)	TP (mg/m^3)	TN (mg/m^3)
Ultra-microtrophic	<1	< 0.33	> 25	< 1.8	< 34
Microtrophic	1 - 2	0.33 – 0.82	15 - 25	1.8 – 4.1	34 - 73
Oligotrophic	2 - 3	0.82 - 2.0	15 - 7.0	4.1 – 9.0	73 - 157
Mesotrophic	3 - 4	2.0 - 5.0	7.0 - 2.8	9.0 - 20	157 - 337
Eutrophic	4 - 5	5.0 - 12	2.8 - 1.1	20 – 43	337 - 725
Supertrophic	5 - 6	12-31	1.1 - 0.4	43-96	725 - 1558
Hypertrophic	>6	>31	<0.4	>96	>1558

2.7.2 Cyanobacteria guideline

The NZ guidelines for cyanobacteria in recreational waters (MfE and MoH 2009) that sets an alert level framework for assessing the health risk from planktonic cyanobacteria. The “Action (Red) mode” is triggered when: 1) cyanobacteria biovolume is ≥10 mm^3/L, or 2) potentially toxic cyanobacteria biovolume is ≥1.8 mm^3/L, or 3) cyanobacteria scums are consistently present.

The “Alert (amber mode)” is when cyanobacteria biovolume is 0.5 to <10 mm^3/L, or 2) potentially toxic cyanobacteria biovolume is 0.5 to <1.8 mm^3/L.

The “Surveillance (green) mode” is when the total cyanobacteria biovolume is <0.5 mm^3/L.

2.7.3 National Policy Statement for Freshwater Management (NPS-FM)

The National Policy Statement for Freshwater Management (**NPS-FM 2020**) (MfE 2020) sets out objectives and policies that direct local government to manage water in an integrated and sustainable way. The NPS-FM includes a National Objectives Framework (NOF) which sets compulsory national values for freshwater including: ‘human health for recreation’ and ‘ecosystem health’. Appendix 2 of the NPS-FM sets water quality attributes that contribute to these values, and ranks attributes into bands to help communities make decision on water quality. This includes setting minimum acceptable states called ‘national bottom lines’.

Appendix 2A of the NPS-FM (2020) describes attributes that require limits on resource use, while Appendix 2B of the NPS-FM (2020) describes attributes that require action plans to be developed (**Table 2.2**).

In this report, we discuss water quality state in the context of the NPS-FM bands where possible. For most attributes, insufficient samples have been collected in recent years to accurately define the band for the purpose of the NPS-FM (e.g. *E.coli* bacteria require 60 samples over 5-years), and in these cases the bands only provide a guideline of water quality state. Arguably, Sullivan Lake does not fall within the scope of the NPS-FM because it is an artificial waterbody, nevertheless the attributes bands provide a context for assessing water quality state.

Table 2.2: NPS-FM attributes and values defining different quality bands pertaining to lakes. *E.coli* bacteria and cyanobacteria relate to suitability for contact recreation while the other bands relate to ecosystem health. Bolded values are the national "bottom-lines".

Table 2A - Attributes requiring limits on resource consents

Attribute	Statistic	Units	Band A	Band B	Band C	Band D	Band E
NH4-N	Median	mg/L	≤0.03	**≤0.24**	≤1.3	>1.3	
NH4-N	Maximum	mg/L	≤0.05	**≤0.4**	≤2.2	>2.2	
E.coli bacteria	% samples >260 cfu/100ml	%	≤20%	≤30%	≤34%	≤50%	>50%
E.coli bacteria	% samples >540 cfu/100 ml	%	≤5%	≤10%	≤20%	≤30%	>30%
E.coli bacteria	Median	*E.coli* / 100mL	≤130	≤130	≤130	≤260	>260
E.coli bacteria	95%ile	*E.coli* / 100mL	≤540	≤1000	≤1200	≤1200	>1200
Phytoplankton	Median	mg *chl-a* /m^3	≤2	≤5	**≤12**	>12	
Phytoplankton	Maximum	mg *chl-a* /m^3	≤10	≤25	**≤60**	>60	
TN (polymictic)	Median	mg/m^3	≤300	≤500	**≤800**	>800	
TP	Median	mg/m^3	≤10	≤20	**≤50**	>50	
Cyanobacteria biovolume	80%ile of potentially toxic cyanobacteria	mm^3/L	≤0.5	≤1.0	**≤1.8**	>1.8	

Table 2B - Attributes requiring action plans

Attribute	Statistic	Units	Band A	Band B	Band C	Band D
Submerged Plants Native Condition Index)		%	>75	>50	**≥20**	<20
Submerged Plants Invasive Condition Index)		%	≤1	≤25	**≤90**	>90
Lake-bottom DO	annual minimum	mg/L	≥7.5	≥2	**≥0.5**	<0.5
Mid-hypolimnetic depth	annual minimum	mg/L	≥7.5	≥5	**≥4**	<4
E.coli bacteria Primary Contact sites	95%ile (summer)	*E.coli* / 100mL	≤130	≤260	**≤540**	>540

3 State of Sullivan Lake

3.1 Morphology

Sullivan Lake is small and shallow. The main lagoon has an average depth of about 0.92 m ± 8% [2]. The base is relatively uniform with the deepest point of 2.2m in the southern pond which has been deepened near the culvert from Te Tahi Street. The area of Sullivan Lake is about 2.7 ha and it has a volume of about 22,700 m^3. The water depth under the water lily at the western end of the lake ranged from 0.30m to 1.13m, with an average water depth of 0.89m.

3.1.1 Sediment depth

The soft sediment depth at the western end of the lake ranged from 0m to 0.61m, with an average soft sediment depth of 0.40m.

In 2017 spot measurements of sediment depth found soft sediment depth in the western end of the lake ranged from 0.1 to 0.8m, with an average depth of 0.6m, however the small number of samples collected in 2017 prevents making a reliable comparison with more recent measurements (Hamill 2017).

Figure 3.1: Bathymetry of Sullivan Lake. The darker areas indicate deeper water, with the maximum depth was 2.2m in the southern section and an average depth of 0.84m.

[2] Plus 8% if preferring measurements by 'weighted line' and minus 8% if preferring measurements by the sonar (see method).

3.2 Hydrology

3.2.1 Inflows

Sullivan lake has a catchment area of 105ha, of which about two thirds is in urban landuse and one third is the tree covered escarpment east of Valley Road (**Figure 3.2**). Ten 10 stormwater culverts enter Sullivan Lake. The largest flows enter near the foot bridge at the eastern end of the lake and drain the escapement east of Valley Road. The escarpment catchments respond quickly to rain and can carry large volumes of sediment. There is little room to control the sediment from this catchment (WSP 2021) (**Figure 3.3**).

In addition, WDC pumps up to about 40 m^3/hour (11.1 L/s) into Sullivan Lake from the Whakatāne River to improve flushing (Neil Yeats, WDC pers. comm., May 2022). The water is pumped into a culvert near the Whakatāne water treatment plant and enters Sullivan Lake via a 900mm culvert at the south-eastern end. Pumping occurs nearly continuously throughout the year except during flood events, but often flows rates are closer to 3 L/s.

The water levels are controlled by a weir at King Street, from which the water flows under King Street into a drain with a gravity discharge to the Whakatāne River (water is pumped during high river flows).

There are no measurements from inflow culverts suitable for estimating flow. However, the average catchment inflow to Sullivan Lake is estimated to be 0.0224 m^3/s. This was derived by multiplying the specific mean flow[3] for the catchment (0.02134 $m^3/s/km^2$) by the catchment area (1.05 km^2). Measured inflows from the main culverts (near foot bridge and south lagoon) during baseflow in mid-November 2023 ranged[4] from 6.3 L/s to 7.8 L/s with about 2.4 L/s attributed to flow augmentation via the Te Tahi Street culvert.

3.2.2 Hydraulic residence time

Hydraulic residence time is an important factor in determining the water quality of lakes. In large oligotrophic lakes which act as a sink for nutrients, increasing residence time can be detrimental to water quality, however in shallow eutrophic lakes with high internal nutrient loads, a shorter residence time can improve water quality by better flushing nutrients and phytoplankton biomass (Jørgensen 2002). To be effective residence time should be reduced to less than about 20 days (Hamilton & Dada, 2016; Abell et al 2020).

The average residence time for Sullivan Lake is about[5] 11.7 days. Augmenting the inflow with an additional 40 m^3/hour (11.1 L/s) reduces the average residence time to 7.8 days, but the relative effect will be larger during dry conditions.

A mean residence time of 11 days is short for a natural lake and reflects the shallow depth of Sullivan Lake. However, because Sullivan Lake catchment is highly urbanised, there is limited baseflow and the flushing of the lake is highly skewed towards rain-events. To completely replace the volume of water in

[3] Data modelled by NIWA for New Zealand's Environmental Reporting Series: The Ministry for the Environment and Statistics New Zealand. Flow expressed for each unit of the River Environment Classification (REC).

[4] Both measured on a fine day but the higher flow measured when then had been 20mm of rain on the previous day.

[5] Calculated as: lake volume (22,700 m^3) / (86,400 s/day * catchment inflow (0.0224 m^3/s))

Sullivan Lake in one day would theoretically require runoff from rainfall in the catchment of greater than 21.6mm per day.

One management option to improve water quality in Sullivan Lake may be to increase the volume of flow pumped to Sullivan Lake during summer dry periods when water quality is at its worst and other inflows are low. This may be partially balance by reducing flow augmentation during wet periods.

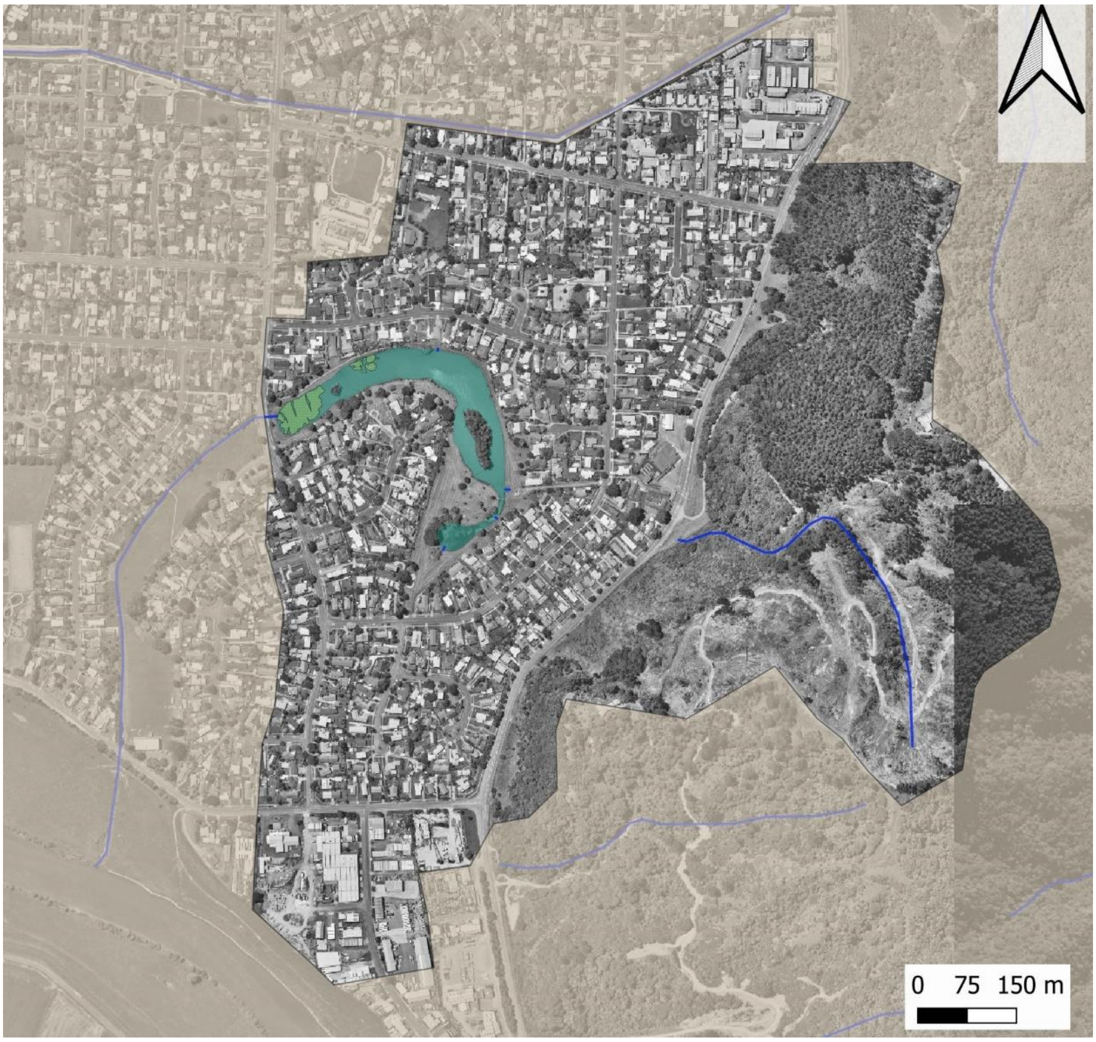


Figure 3.2: Sullivan Lake surface water catchment area (105 ha).

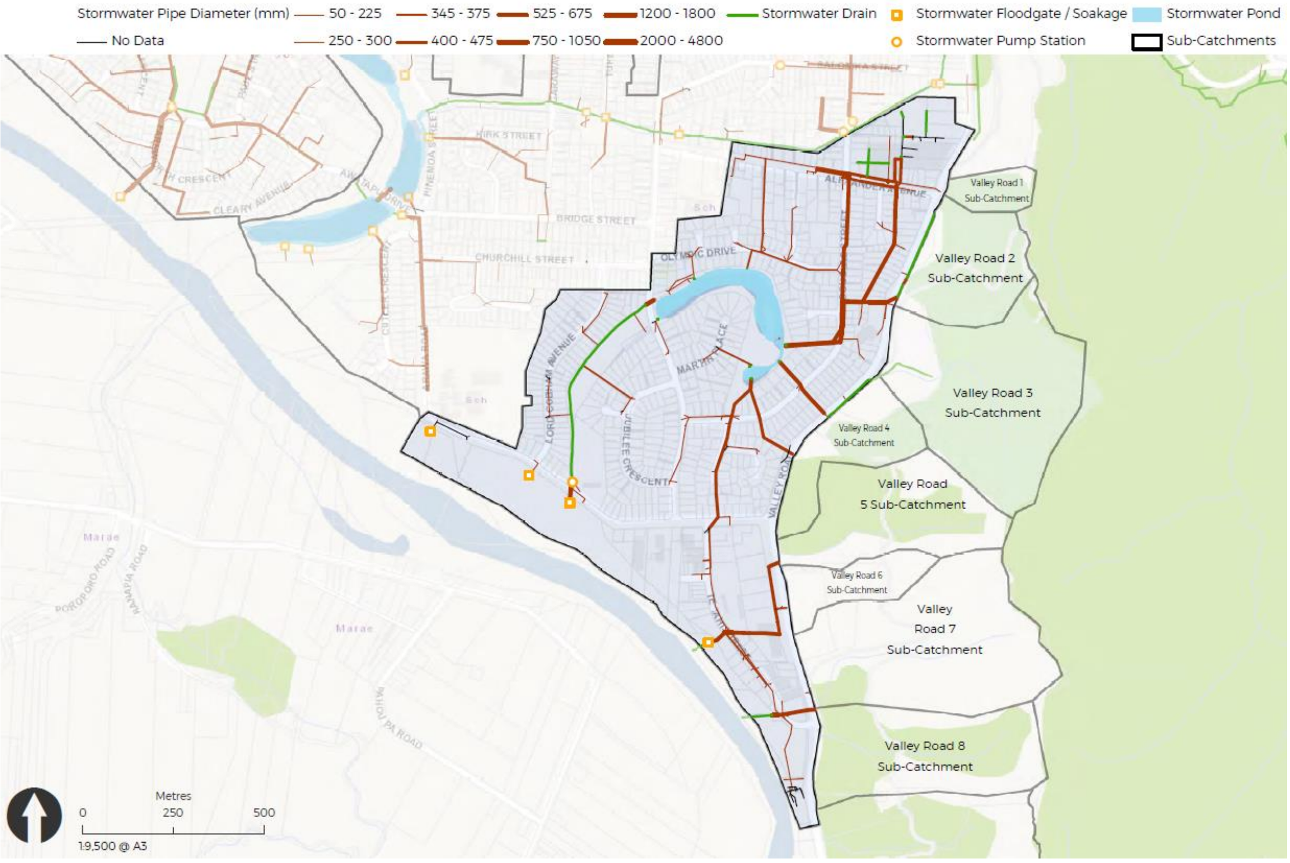


Figure 3.3: Whakatane South Stormwater Catchment entering and leaving Sullivan Lake (WSP 2021).

3.3 Birds

Sullivan Lake provides habitat for a wide range of waterfowl. Bird species commonly using the lake include: pūkeko (*Porphyrio melanotus*); Australian coot (*Fulica atra australis*), little black shag (*Phalacrocorax sulcirostris*), little shag (*Microcarbo melanoleucos*), black shag (*Phalacrocorax carbo*), silver gull (*Chroicocephalus novaehollandiae*), welcome swallow (*Hirundo neoxena*), mallard ducks (*Anas platyrhynchos*), paradise shelduck (*Tadorna variegata*), muscovy ducks (*Cairina moschata*), and mute swan (*Cygnus olor*). Other birds observed on the lake include: Australian shoveler (*Spatula rhynchotis*), spoonbill (*Platalea regia*), and occasionally white heron/ kōtuku (*Egretta alba*).

3.3.1 Avian botulism

Populations of duck can be high and occasionally outbreaks of avian botulism has occurred in wildfowl during extended dry summers. This causes progressive weakness and ascending paralysis and can be a common cause of death in migratory birds. Avian botulism is caused by toxins produced by some strains of the bacteria *Clostridium botulinum*. This is a naturally occurring bacteria found in anaerobic aquatic sediments. Dormant spores are harmless, but the botulism toxin is produced as the cells grow and is released at the end of a growth phase. Common factors in botulism outbreaks are anoxic sediments (devoid of oxygen), warm temperatures (e.g. >20 °C), decomposing organic matter (particularly material high in protein), and water pH in the range of c. 7.5 to 9.0 (Rocke and Samuel 1999).

One feature of avian botulism that perpetuates mass die-off of birds is the 'carcass-maggot cycle'. This is where healthy birds ingest and carry dormant botulinum spores; upon death the spores germinate and grow in the carcass; maggots feeding on the carcass accumulate the toxin, and subsequent consumption of the maggots by other birds causes poisoning and death that perpetuates the botulism outbreak. Collecting and disposing of carcasses can help break this cycle (Rocke and Bolling 2007).

3.3.2 Potential impact of birds on water quality

While birds are an important value of the lake, high densities of birds can reduce water quality. A number of studies have found that water fowl can be a significant source of faecal coliform bacteria to some lagoons and beaches. This is partially because birds tend to defecate directly in the water, and partially because they have relatively high load of nutrients and faecal bacteria relative to their body size (e.g. Flemming and Fraser 2001, Don and Donovan 2001).

3.4 Fish

Fish species confirmed as present in Sullivan Lake are: shortfin eel, common smelt, bully species (probably common bully), goldfish and *Gambusia* (a pest fish). The tributary stream entering from the east supports the fish: shortfin eel, common bully, inanga, and gold fish (**Table 3.1**).

Fish access to the lake from Sullivan Lake to the Whakatāne River is restricted by the flood gate at the outlet near the stop bank, and steep weir at the outlet. However, it appears that at least some individuals of the migratory fish (shortfin eel and inanga) have migrated to Sullivan Lake and it's catchment streams.

Table 3.1: Fish present in Sullivan Lake and the main inflow stream as indicated by eDNA sequences detected.

Common Name	Scientific Name	Sullivan Lake 14/6/2022	Stream entering at Footbridge 17/11/2022
Shortfin eel	*Anguilla australis*	526	4280
Common smelt	*Retropinna retropinna*	21	
Bullies	*Gobiomorphus* sp.	7575	13782
Common bully /Cran bully	*Gobiomorphus cotidianus / basalis*	728	
Inanga	*Galaxias maculatus*		103
Goldfish	*Carassius auratus*	5692	2925
Mosquitofish	*Gambusia affinis*	1198	

3.5 Aquatic Plants

Sullivan Lake is predominantly surrounded by parkland with mown grass and occasional mature tree. A narrow band of native riparian vegetation has been planted along about 240m of the southern edge consisting of flaxes, sedges and rushes. A formal wooden edge delineates the lake from the surrounding land and only a few wetland plants have established in the water near the footbridge (e.g. *Carex secta, Bolboschoenus fluviatilis, Machaerina articulata*).

Most of the year Sullivan Lake is devoid of macrophytes except for Mexican water lily (*Nymphaea* mexicana)[6] which covers a large section of the western part of the lake near King Street (**Figure 3.4**). However, during spring and early summer 2022 the curled pondweed *Potamageton crispus* covered about 90 percent of the open water area in association with epiphytic algae. Patches of floating sweet grass (*Glyceria fluitans*) can occur on the lake margins. Other species that are occasionally present include the floating fern *Azolla* spp., duckweed and pest plants of *Egeria densa*, *Elodea canadensis*, and hornwort (*Ceratophyllum demersum*). These macrophytes remain present in the drain downstream of the Sullivan Lake outlet.

Curled pondweed is an annual aquatic plant, that germinates from seeds during late spring, grows prolifically during early summer and typically collapses in mid-summer to late. During the growth phase, the curled pondweed often improves water quality by taking up nutrients and stabilising sediments. However, pondweed beds typically collapse around mid-summer and subsequent decomposition on the lake bed can reduce dissolved oxygen and cause and release of dissolved nutrients from the substrate that can be used by phytoplankton (Gibbs 2011). Potamogeton has a low tolerance to high temperatures and low alkalinity, but a high tolerance to sediment disturbance and to most aquatic herbicides such as Diquat and Endothall (Gibbs and Hickey 2012).

Macrophyte cover in Sullivan's Lake has been more extensive in the past. Surveys in 1987 found hornwort (*Ceratophyllum demersum*) was by far the most dominant aquatic plant in Sullivans Lake and Awatapu Lagoon (WDC 1990). Aerial photos show water lily present in the early 1980, but its coverage has been variable.

The cover of macrophytes has historically been controlled by use of herbicide sprays. This is a relatively cheap way to control macrophytes, but has can have a number of negative consequences, including decomposing macrophytes causing sediment anoxia, release of nutrients and promotion of algae blooms. Harvesting macrophytes, although more expensive, provide considerably more benefits for improving water quality and ecological values.

Although *P. crispus* is an introduced plant, it is relatively benign compared to hornwort (*Certatophyllum demersum*) and parrots feather (*Myriophyllum aquaticum*) that is dominant in Awatapu lagoon. It is much less dense than hornwort and consequently has less adverse effects on the dissolved oxygen regime during senescence.

[6] Classed in the Bay of Plenty as an "advisory pest plant".

Figure 3.4: Water lilies at Sullivan Lake western end. Top photo: aerial facing west in August 2022; Bottom photo from May 2022.

3.5.1 Role of macrophytes in maintaining good water quality

Macrophyte beds are a key component of healthy lakes. They help improve water quality by stabilising the sediments, absorbing dissolved nutrients, mediating the nutrient release from sediments, and providing habitat for invertebrates that consume phytoplankton (Hilt et al. 2006; Kelly and Jellyman 2007; Schallenberg et al. 2010, Wetzel 1995). Overseas studies have shown that submerged aquatic plant cover needs to be consistently >30% to 60% to ensure a clear-water state (e.g. Jeppesen et al. 1994; Tatrai et al. 2009; Blindow et al. 2002).

It is well documented that shallow, eutrophic lakes can often undergo a regime shift (colloquially called “flipping”) from a clear water, macrophyte-dominated state to a de-vegetated, algae-dominated state with turbid water quality (Scheffer 2004, Tatrai et al. 2009). At least 37 shallow lakes in New Zealand that have undergone a “flip” between clear water and turbid states and/or vice versa.

The risk of a lake flipping to a turbid water quality state increases with increasing nutrient and sediment loads, and typically corresponds to increases in epiphytes, macroalgae, phytoplankton and cyanobacteria (**Figure 3.5**) (De Wit et al. 2001, Scheffer & van Nes 2007). Flipping to a turbid, algae dominated state is more likely when a lake has a high nutrient load, where exotic macrophytes have replaced native macrophytes, and where coarse fish species (e.g. catfish, goldfish, rudd, tench, or koi carp) are present (Schallenberg and Sorrell 2009).

Re-establishing submerged macrophytes is essential for the long-term success when restoring shallow lakes. However, simply establishing macrophyte beds does not always improve water quality even when they improve fish habitat. Ecosystems are complex and often other restoration activity is also needed. Establishing aquatic plants in shallow lakes does not guarantee clear water quality, but without them good water quality is unlikely without other expensive and ongoing interventions (Gulati et al., 2008; Jeppesen et al. 2005).

Native macrophytes are much more preferable than exotic macrophytes because they provide more biodiversity, have less aggressive growth and are less likely to attain high biomass that can adversely affect dissolved oxygen or cause a nuisance for recreation. However, even exotic macrophytes can provide water quality benefits if well managed. Where exotic macrophytes are present, a common challenge for lake management is to retain the benefits of macrophytes in the lake while minimising the problems caused by excessive growth on water quality and recreation.

Sullivan Lake, with the exception of the water lilies at the western end, appears to have had a long period in a turbid, cyanobacteria dominated state. The establishment of curled pondweed in late 2022 may indicate a shift to improved conditions, although epiphytes, macroalgae and cyanobacteria remain abundant. Nevertheless, the establishment of curled pondweed improves water quality during its spring growth as evidenced by relative better clarity and lower nutrients while it was growing in late October 2022[7]. In addition, curled pondweed also provides an opportunity for improved management of water lake water quality and long-term nutrient removal through macrophyte harvesting prior to senescence.

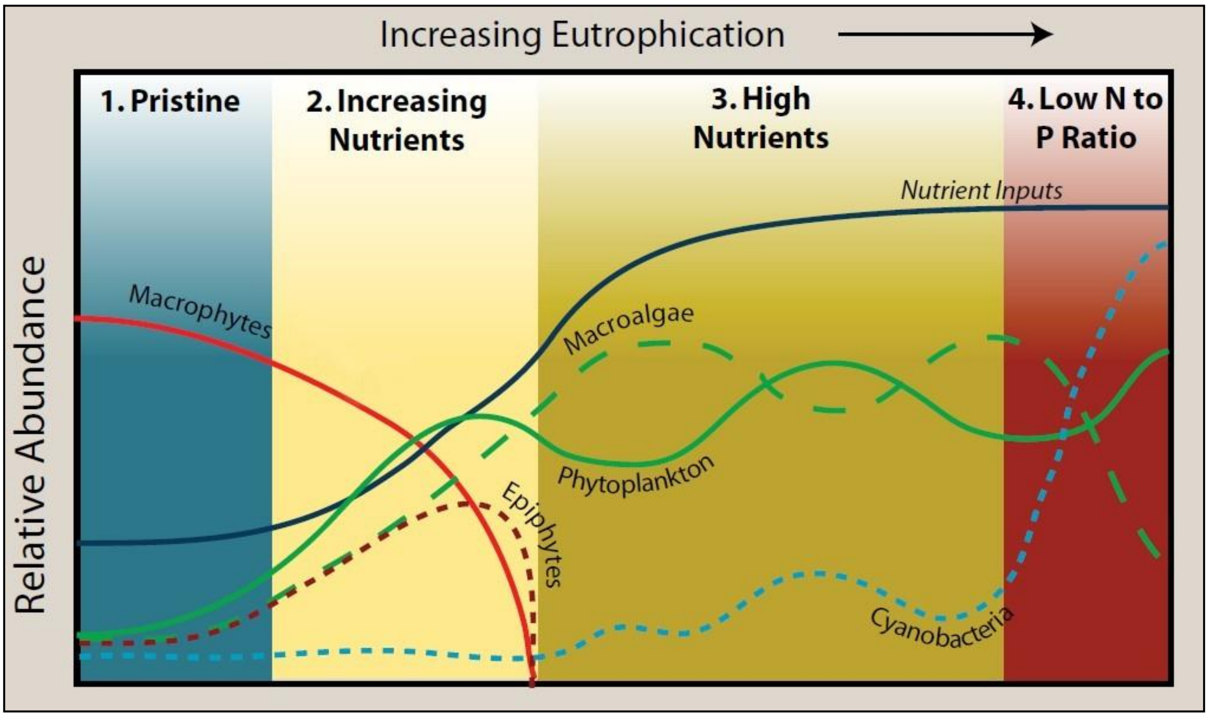


Figure 3.5: Generalised lake response to increasing eutrophication. Sullivan Lake appears to be in Stage 3 to 4 (adapted from De Wit et al. 2001).

[7] 30 October 2022, had TN, TP, Chl-a and black disk clarity of respectively 0.42 mg/L, 0.1 mg/L, 1.6 mg/m^3 and 1.3m.

3.6 Water Quality

3.6.1 Water quality samples

Sullivan Lake has poor water quality characterised by low water clarity (median 0.37m), high concentrations of nitrogen and phosphorus (median 0.48 and 0.143 mg/L respectively), and high concentrations of phytoplankton (median Chl-*a* 19.2 mg/m^3). Algae blooms are common, and are often dominated by potentially toxic cyanobacteria (**Table 3.2, Figure 3.7**). Algae blooms appear least common in November-December (**Appendix 2**).

Recent TLI results are about 5.90, which is borderline between supertrophic and hypertrophic (**Figure 3.6**). This reflects the low clarity and high concentrations of TN, TP, and Chl-*a*. National bottom-line values set for lake attributes in the NPS-FM appear to be exceeded for median TP, median Chl-*a*, maximum Chl-*a*, and cyanobacteria biovolume (see cyanobacteria section below).

Phytoplankton growth in Sullivan Lake appears to be more strongly limited by nitrogen than by phosphorus. The recent TN:TP ratio is about 3.5 (compared to a “balanced” ratio of 7) and the DIN:TP ratio is less than 0.1 (compared to a “balanced” ratio of 1). Absolute values of dissolved organic nitrogen (**DIN**) are often very low (median 0.01 mg/L), while DRP is moderately high (median 0.018 mg/L). High concentrations of TP and chlorophyll-*a* elevates the TLI (i.e. makes it worse) relative to the concentration of TN (expressed as TL-n) (**Figure 3.6**). Occasions with high TN (e.g. May 2022) are associated with cyanobacteria blooms – which can fix nitrogen from the atmosphere.

Water quality in Sullivan Lake considerably improved between 2001 and 2008. This was evident for most water quality variables, but the improvement was particularly strong for chlorophyll-*a* and TN. The TN:TP ratio reduced from about 9 during 2000-2002 to 5 during 2006-2008 (**Figure 3.6**, **Figure 3.7**). The level of improvement would be consistent with recovery from past pollution such as possible sewage overflows.

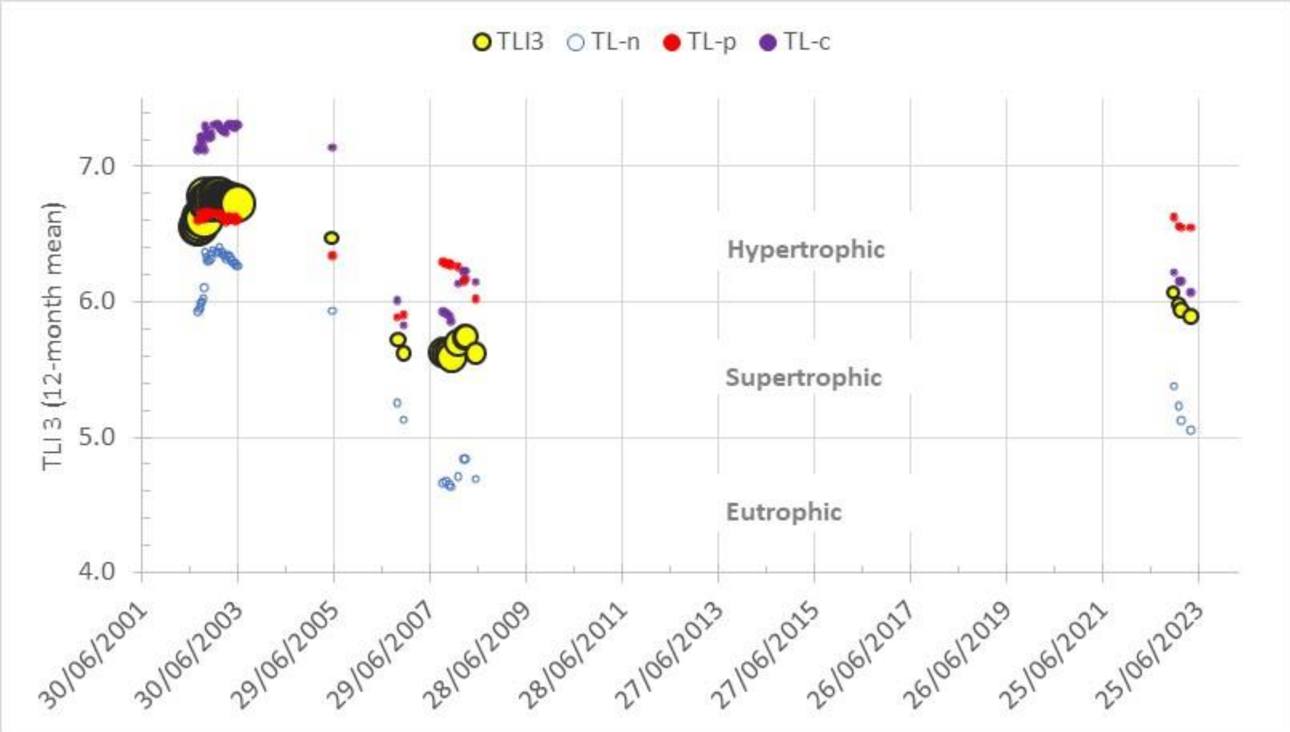


Figure 3.6: Annual Trophic Level Index (TLI3) of Sullivan Lake and its constituents for nitrogen (TL-n), phosphorus (TL-p) and chlorophyll-a (TL-c). For TLI3, the size of the circle indicates the number of samples used to calculate the rolling 12-month average (range 4-26).

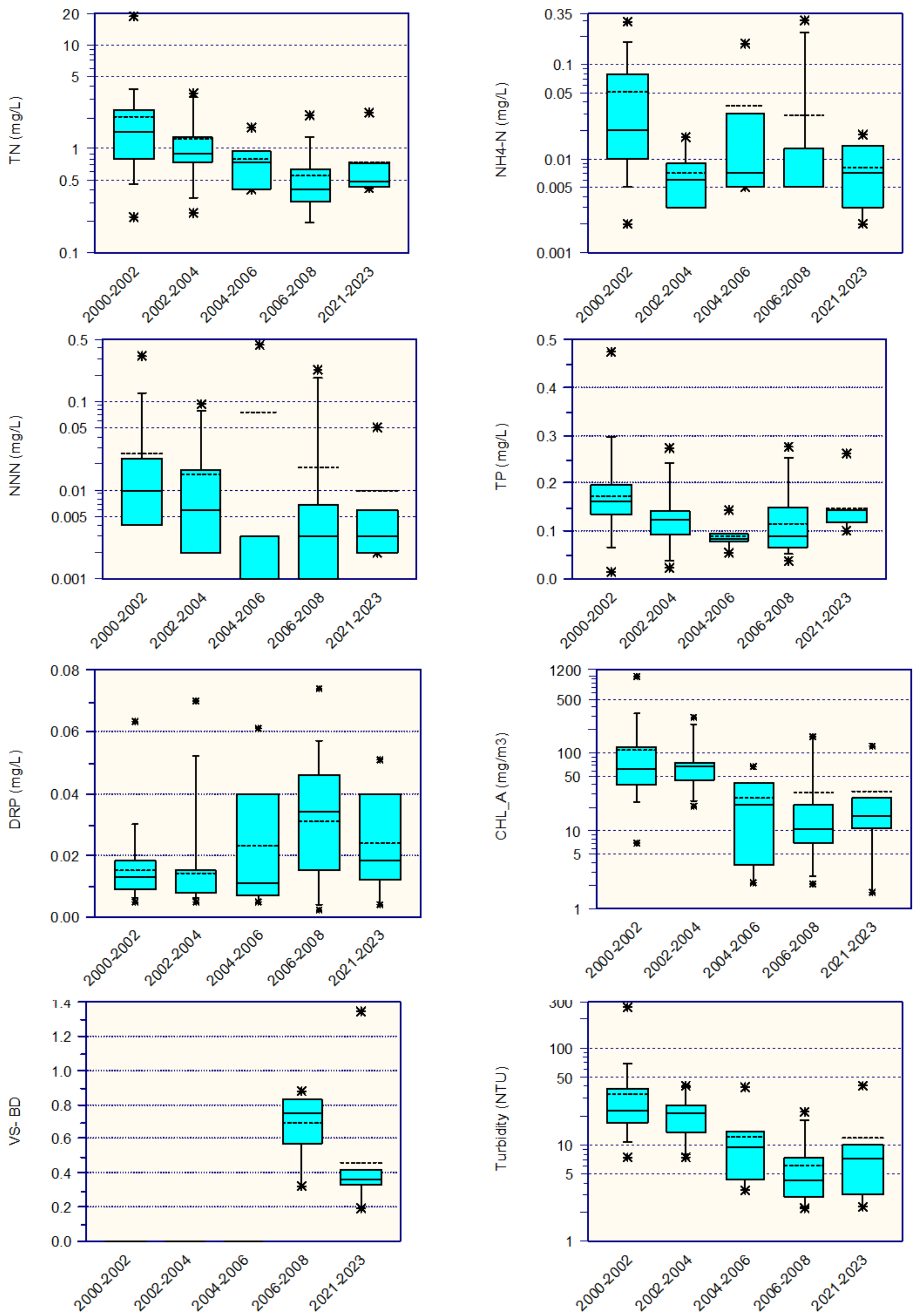


Figure 3.7: Water quality in Sullivan Lake expressed as box plots for two-year (July to June) periods.

Table 3.2: Sullivan Lake water quality summary statistics for two-year (July to June) periods (data from 2004-2006 excluded for clarity of presentation).

Years	Variable	n	Min	Max	Mean	Median	95 %ile	5 %ile
2000-2002	TN (mg/L)	53	0.22	18.95	2.01	1.48	3.76	0.45
2000-2002	NH4-N (mg/L)	53	0.002	0.288	0.051	0.020	0.170	0.005
2000-2002	NNN (mg/L)	42	0.001	0.325	0.026	0.01	0.125	0.001
2000-2002	TP (mg/L)	54	0.013	0.475	0.171	0.161	0.295	0.065
2000-2002	DRP (mg/L)	53	0.005	0.063	0.015	0.013	0.030	0.006
2000-2002	CHL_A (mg/m3)	54	6.9	992	109.4	62.0	326.8	23.6
2000-2002	Turbidity (NTU)	44	7.3	265.0	33.1	22.5	68.8	10.7
2000-2002	TN:TP	53	1.2	95.2	12.0	9.3	25.5	4.0
2000-2002	pH	53	6.9	10.1	8.3	8.3	9.5	7.2
2000-2002	DO (mg/L)	50	5.6	14.4	10.1	10.4	13.3	6.8
2000-2002	EC sp (uS/cm)	43	96	763	150	138	172	104
2002-2004	TN (mg/L)	16	0.25	3.39	1.24	0.91	3.17	0.34
2002-2004	NH4-N (mg/L)	17	0.001	0.017	0.007	0.006	0.017	0.001
2002-2004	NNN (mg/L)	16	0.001	0.093	0.015	0.006	0.078	0.001
2002-2004	TP (mg/L)	16	0.023	0.272	0.124	0.124	0.241	0.038
2002-2004	DRP (mg/L)	17	0.005	0.070	0.014	0.008	0.052	0.006
2002-2004	CHL_A (mg/m3)	17	21	292	76.6	68.0	235.3	24.2
2002-2004	Turbidity (NTU)	16	7.3	40.0	20.8	21.0	38.8	7.8
2002-2004	TN:TP	16	2.8	33	10.9	9.3	28.7	3.5
2002-2004	pH	16	7.2	9.4	8.5	8.7	9.4	7.3
2002-2004	DO (mg/L)	17	8.7	13.6	10.7	10.5	13.4	8.8
2002-2004	EC sp (uS/cm)	16	93	138	109	107	136	93
2006-2008	TN (mg/L)	29	0.05	2.10	0.55	0.40	1.28	0.20
2006-2008	NH4-N (mg/L)	30	0.001	0.300	0.029	0.005	0.221	0.001
2006-2008	NNN (mg/L)	30	0.001	0.225	0.018	0.003	0.181	0.001
2006-2008	TP (mg/L)	30	0.038	0.276	0.115	0.09	0.252	0.051
2006-2008	DRP (mg/L)	30	0.002	0.074	0.031	0.034	0.057	0.004
2006-2008	CHL_A (mg/m3)	28	2.1	167	31.4	10.5	164.3	2.6
2006-2008	VS- BD	10	0.32	0.88	0.69	0.75		0.32
2006-2008	Turbidity (NTU)	27	2.2	22.0	6.1	4.3	17.8	2.3
2006-2008	TN:TP	29	0.6	9.6	4.9	5.4	8.4	1.1
2006-2008	pH	29	6.8	9.5	8.1	8.3	9.4	6.9
2006-2008	DO (mg/L)	25	5.9	18.9	11.1	11.1	15.7	6.5
2006-2008	EC sp (uS/cm)	28	81	3500	256	106	818	87
2021-2023	TN (mg/L)	7	0.42	2.23	0.76	0.48		0.42
2021-2023	NH4-N (mg/L)	7	0.002	0.018	0.008	0.007		0.002
2021-2023	NNN (mg/L)	7	0.002	0.05	0.01	0.003		0.002
2021-2023	TP (mg/L)	7	0.102	0.262	0.147	0.143		0.102
2021-2023	DRP (mg/L)	7	0.004	0.051	0.024	0.018		0.004
2021-2023	CHL_A (mg/m3)	6	1.6	126	32.7	15.7		1.6
2021-2023	VS- BD	10	0.19	1.34	0.46	0.37		0.19
2021-2023	Turbidity (NTU)	6	2.3	40.0	11.6	7.2		2.3
2021-2023	TN:TP	7	3	18.9	5.7	3.5		3.0
2021-2023	E coli (cfu/100ml)	7	30	1600	299	100		30
2021-2023	pH	8	7.0	9.0	7.8	7.6		7.0
2021-2023	DO (mg/L)	8	8.1	15.4	11.0	10.7		8.1
2021-2023	EC sp (uS/cm)	8	132	280	175	157		132

3.6.2 Cyanobacteria

Cyanobacteria are a natural part of the plankton community in lakes but can become a problem when they increase to high concentrations and form 'blooms'. Frequent cyanobacteria blooms are a feature of poor water quality in lakes and are caused by multiple factors including high nutrient concentrations, warm, calm conditions, and wind-driven accumulations of surface scums. High concentrations of cyanobacteria can also pose a potential health risk to recreational users, because they produce a range of different cyanotoxins.

Cyanobacteria blooms are very common in Sullivan Lake during summer and autumn, often exceeding recreational use guidelines (MfE and MOH 2009). Summer monitoring of cyanobacteria in Sullivan Lake from 2013 to 2020 (55 samples) found that the **Alert Mode** (of biovolume 0.5 to 10 mm^3/L) occurred on 45% of occasions, while the **Action Mode** for total biovolume (≥10 mm^3/L) was exceeded on 53% of occasions, and the **Action Mode** trigger for potentially toxic cyanobacteria (biovolume ≥1.8 mm^3/L) was exceeded on 62% of occasions. On 18% of occasions the cyanobacteria biovolume was extremely high, at ten times the Action Mode.

The 80th percentile of potentially toxic cyanobacteria biovolume for the three-year period of 2018-2020 was 23 mm^3/L. This is worse than the NPS-FM national bottom-line (i.e. threshold of 1.8 mm^3/L for "D" Band).

Anabaena spp. is the dominant cyanobacteria present, and it is particularly prevalent during blooms where it can be visible as green flocs suspended in the water (**Figure 3.8**, **Figure 3.9**, **Table 3.3**).

Figure 3.8: Australian coot on Sullivan Lake during a cyanobacteria bloom, January 2022.

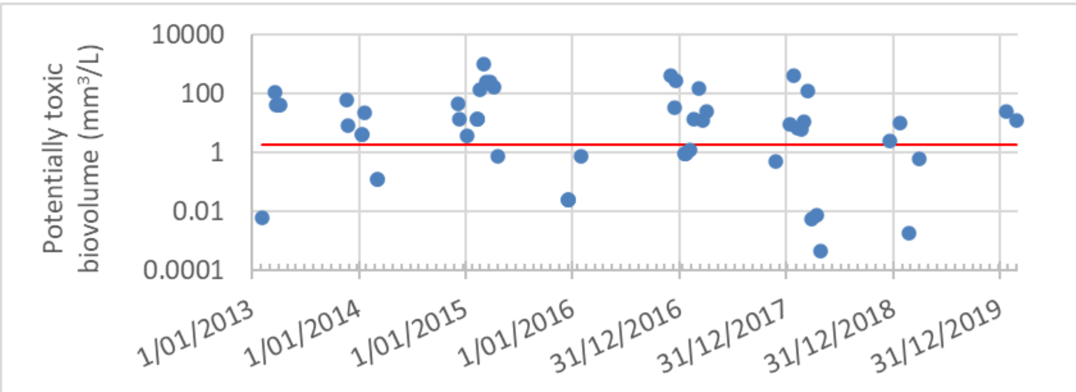


Figure 3.9: Biovolume volume of potentially toxic cyanobacteria in Sullivan Lake during summer ('Action' mode guideline is the red line).

Table 3.3: Occurrence of cyanobacteria species in Sullivan Lake during summer, 2013-2020 (source BOPRC). Shading groups common genesis.

Species	% occurance	% occurance when biovolume ≥10 mm^3/L
Anabaena circinalis	6%	4%
Anabaena lemmermannii	17%	19%
Anabaena spiroides	38%	67%
Aphanocapsa delicatissima	2%	
Aphanocapsa elachista	1%	
Aphanocapsa sp	4%	
Chroococcus dispersus	2%	
Chroococcus limneticus	4%	4%
Merismopedia sp	5%	
Microcystis flos-aquae	2%	
Microcystis sp	1%	
Oscillatoria sp	1%	4%
Phormidium sp	1%	
Planktothrix sp	9%	4%
Pseudanabaena limnetica	1%	
No Cyanobacteria	5%	

3.6.3 Dissolved Oxygen

Dissolved oxygen (DO) is a fundamental for the health of almost all aquatic ecosystems. Reduced concentrations of DO (e.g. <4 mg/L) can impair the growth and reproduction of aquatic organisms, and shift the community composition to more tolerant organisms. As DO further reduces (e.g. 1 to 2 mg/L), death of aquatic organisms becomes increasingly common unless organisms can avoid low DO zones (Davies-Colley et al. 2013). The complete loss of DO (anoxia) from bottom waters of lakes causes changes in geochemistry that facilitates the release of nitrogen (as NH4-N) and phosphorus (as DRP) from the sediment; this can stimulate further eutrophication, which itself contributes to conditions that caused the anoxia.

Algae blooms can cause large daily fluctuations in dissolved oxygen (DO) and pH due to the photosynthesis and respiration of the phytoplankton. Oxygen concentrations will typically increase with photosynthesis during the day, and decrease with respiration at night. Other factors that have an important influence on lake DO, in addition to photosynthesis and respiration, are: wind re-aeration (that moves the DO towards 100% saturation), sediment oxygen demand, and biochemical oxygen demand from the water.

3.6.3.1 Temporal Variation in DO

Dissolved oxygen loggers were installed in Sullivan Lake in autumn 2022 (April to June 2022) and during summer 2023 (December 2022 to February 2023). During autumn 2022 Sullivan Lake had large diurnal fluctuations in DO (commonly changing by 7mg/L) and for a six-week period during April-May the DO was consistently above 100% saturation. These features indicate a high level of primary production which is consistent with observations of an algae bloom occurring at the time. Having super-saturation during the night-time is unusual for a lake and is discussed further below.

A moderate rain event (21mm daily) on 21 April 2022 reduced DO concentrations (to 8 mg/L) - possibly due to suspension of bottom sediment or introduction of a BOD load with the stormwater. A large rain event (40mm daily) on 18 May 2022 resulted in a similar reduction in DO (to 8 mg/L) as well as a reduction in diurnal fluctuations in the subsequent week – suggesting that flushing from the larger rain-event substantially reduced the phytoplankton biomass. Further large rain-events during early June 2022 reduced the DO to below 4 mg/L – indicating a source of BOD from either the catchment stormwater or sediment suspension. DO concentrations below 4 mg/L start to cause chronic effects on sensitive aquatic life. The lake recovered to a healthier, oxygenated DO regime after about two weeks (**Figure 3.10**).

During October 2023 the curled pondweed *P. crispus* germinated and grew to cover about 90% of the Sillivan Lake's open water[8]. During this time chlorophyll-*a* concentrations were low (1.6 mg/m^3). When DO loggers were installed on 14 December 2022, the macrophyte cover had slightly reduced and Chl-a increased to 11.6 mg/m^3; and by 18 January 2023 macrophyte cover was low and an algae bloom was developing (Chl-a of 26.8 mg/m^3). The DO regime was supressed by rain-events on 15 December and 21 December 2022, and recovered in the following week.

Between 27 Dec 2022 to 3 Jan 2023, there were large diurnal fluctuations in DO (c. 6 mg/L), in addition to a steady decline in the daily minimum DO from 9.15 mg/L to 1.46 mg/L (ie. 7.69 mg/L over 7-days). This equates to an oxygen demand of 1.1 mg/L per day, or 1.16 g/m^2/day. DO increased again on 4 January 2023 – probably due to aeration from a strong nor-easterly wind that occurred from 4th to 6th January (**Figure 3.11**). The oxygen demand observed in Sullivan Lake over the new year is likely due to the seasonal collapse of pondweed, causing sediment hypoxia.

During January to February 2023 there were multiple days when the daily minimum DO concentration were not only below guideline values (4 mg/L), but sufficiently low (< 2 mg/L) to cause fish to exhibit avoidance behaviour or potentially acute toxicity. These periods of very low DO occurred in early January (probably due to the pondweed collapse), in mid-late January due to very large diurnal fluctuations associated with an algae bloom, and during 6-7 February, that was associated with a week of heavy rain (and night-time anoxia).

[8] Excluding the area covered by waterlily.

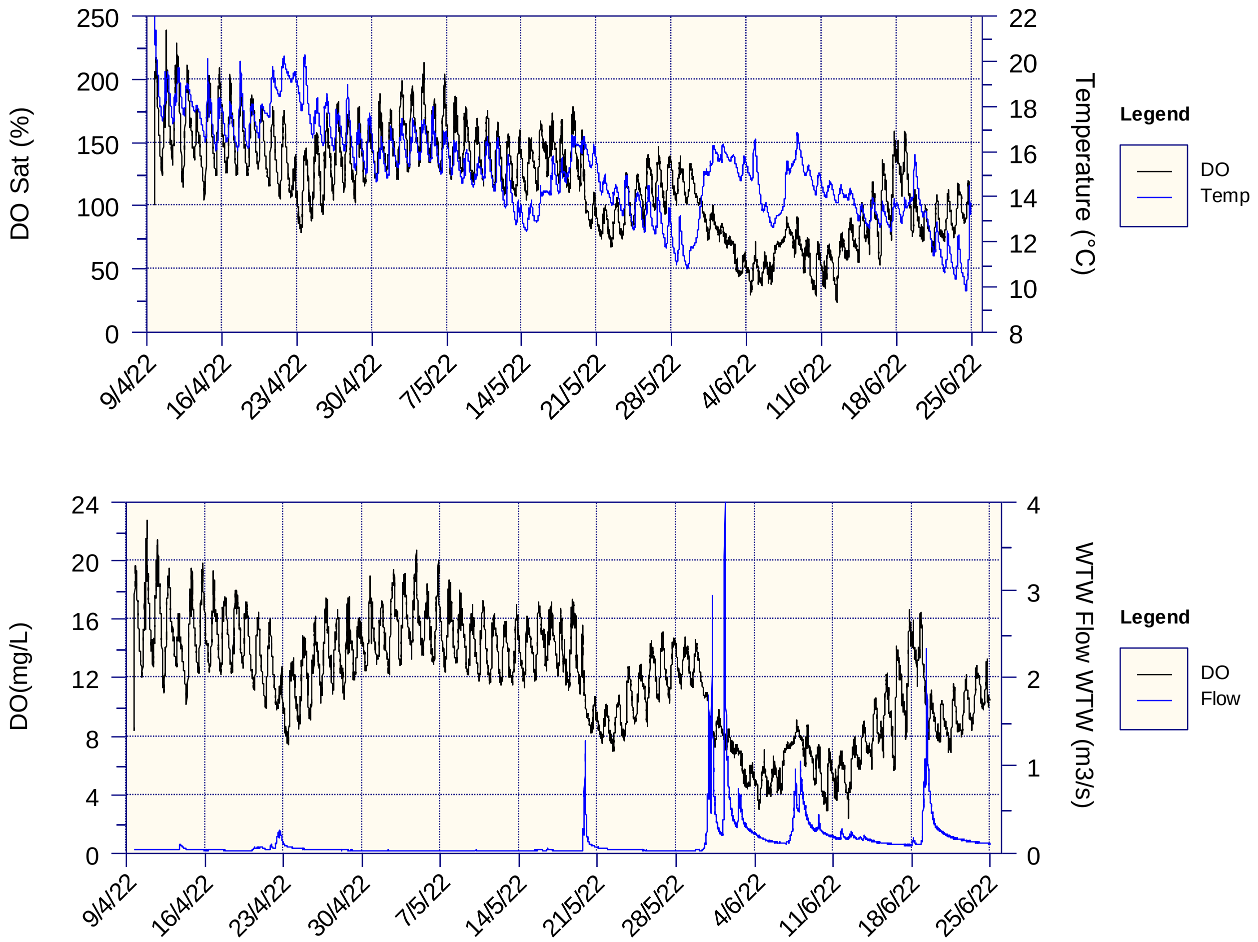


Figure 3.10: Dissolved oxygen in Sullivan Lake during autumn 2022. Expressed as %DO vs. temperature (top graph), and DO vs. flow (bottom graph). Flow in the Wainui Te Whara is used as a proxy for the pattern of inflows to Sullivan Lake.

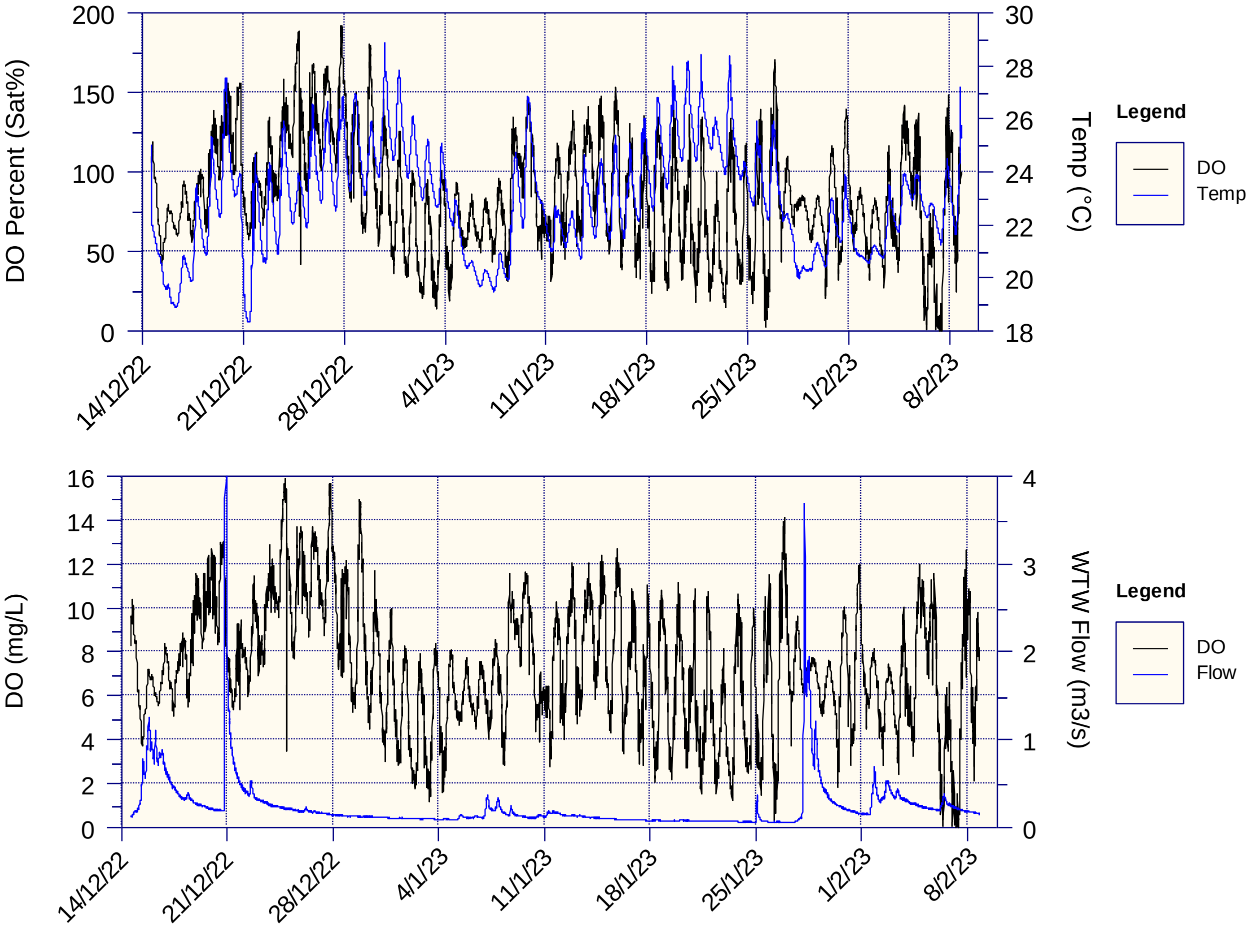


Figure 3.11: Dissolved oxygen in Sullivan Lake during the summer of 2022/23. Expressed as %DO vs. temperature (top graph), and DO vs. flow (bottom graph). Flow in the Wainui Te Whara is used as a proxy for the pattern of inflows to Sullivan Lake.

Night-time supersaturation

Diurnal fluctuations in DO is a common feature of lakes, but it is unusual for a lake to remain super-saturated in DO during the night as occurred throughout April to May 2022 - normally respiration would reduce night-time DO to less than 100% saturation. There are several possible explanations for night-time supersaturation. Accumulation of DO may occur in a waterbody if the import (via photosynthesis and diffusion) exceeds the export (via respiration and diffusion). Algae often produce more oxygen during the day than they consume at night, but this can reverse if photosynthesis is reduced (e.g. by low irradiance), or if respiration increases (e.g. by algae senescence). Super-saturated oxygen stored in the water is slowly lost to the atmosphere by diffusion, and the rate of this loss can increase with mixing, which reduces the DO in the water that can later be used at night.

Burke (1995) found that oxygen produced by benthic cyanobacteria caused supersaturation (up to 370%) in the bottom water of a stratified saline lake. This was because the transport of oxygen across the sediment-water interface was limited by diffusion, and the export of oxygen out of benthic cyanobacteria during the day proceeded faster than the daytime import, thus allowing supersaturation of the bottom waters.

The relative strength of respiration and photosynthesis is influenced by multiple factors. Wieland and Kuhl (2006) found that at low irradiance, oxygen consumption (through respiration) increased more strongly with temperature than production (through photosynthesis), but the opposite occurred at high irradiances.

3.6.3.2 Spatial Variation in DO

Sullivan Lake can have large spatial variations in DO and pH. Synoptic surveys undertaken in the early morning and late afternoon during May 2022 found super-saturation in the main body of the lagoon (consistent with DO logger results) and low DO at the western end where there was dense water lily. Areas of low DO in the western end were independent of whether the samples were collected under the water lily or in open water at the western end (**Figure 3.12** and **Figure 3.13**).

Sullivan Lake also exhibits large spatial variation in water pH. This has the same pattern as DO – high in the main body of the lagoon (associated with algae photosynthesis) and low at the western end (likely) due to respiration/decomposition of organic sediments (**Figure 3.14** and **Figure 3.15**). It is likely that the western section of the Sullivan Lake had organic sediments exerting a high sediment oxygen demand. It is also likely that shading by the waterlily supressed phytoplankton growth, resulting in less extreme diurnal fluctuation in DO and pH.

The pattern observed during the synoptic surveys in April 2022 was consistent with measurements collected through 2022, i.e. lower DO and pH in the western end (near King Street) compared to the main body of the lake (near Olympic Drive walkway) (**Table 3.4**).

Table 3.4: Spatial comparison of temperature, DO and pH in Sullivan Lake at Olympic Drive and the western end at King Street.

Site	Date	Time	Temp. (oC)	%DO	DO (mg/L)	pH
Sullivan Lake Olympic Dv	23/01/22	6:55	24.6	45	3.8	8.3
Sullivan Lake West	23/01/22	6:50	25.0	2.7	0.22	6.8
Sullivan Lake Olympic Dv	09/05/22	17:15	15.3	154	15.4	8.9
Sullivan Lake West	09/05/22	17:24	15.2	133	13.3	8.3
Sullivan Lake Olympic Dv	24/06/22	15:03	11.1	119	13.1	7.1
Sullivan Lake West	24/06/22	14:49	9.9	56	6.3	5.9
Sullivan Lake Olympic Dv	30/10/22	12:00	19.5	119	10.7	9.0
Sullivan Lake West	30/10/22	12:40	18.4	7.7	0.72	6.3

Figure 3.12: Spatial variation of dissolved oxygen in Sullivan Lake during early morning on 5 April 2022. Note super-saturation in the main body of the lagoon and low DO at the western end.

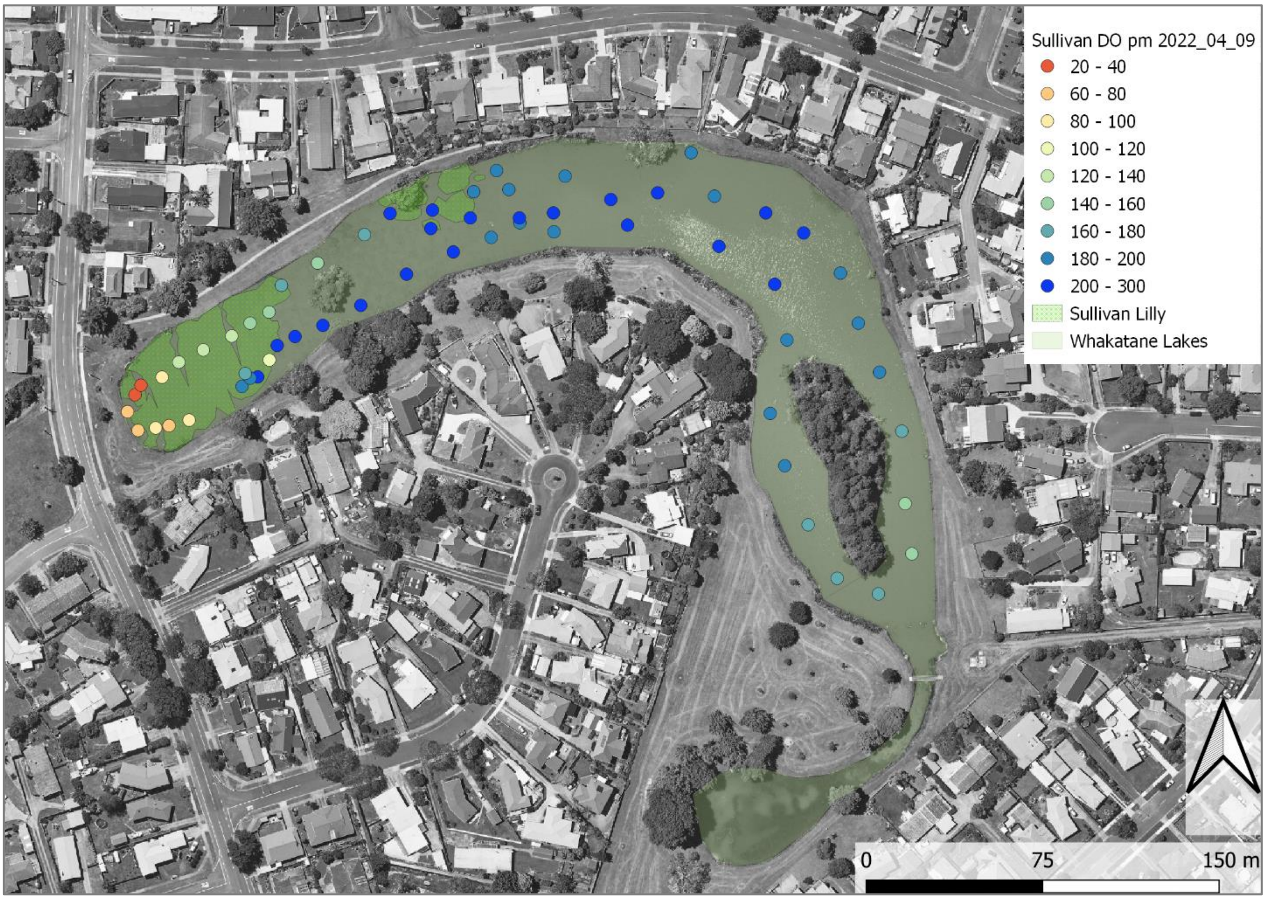


Figure 3.13: Spatial variation of dissolved oxygen in Sullivan Lake during late afternoon on 9 April 2022. Note super-saturation in the main body of the lagoon and persistent low DO at the western end.

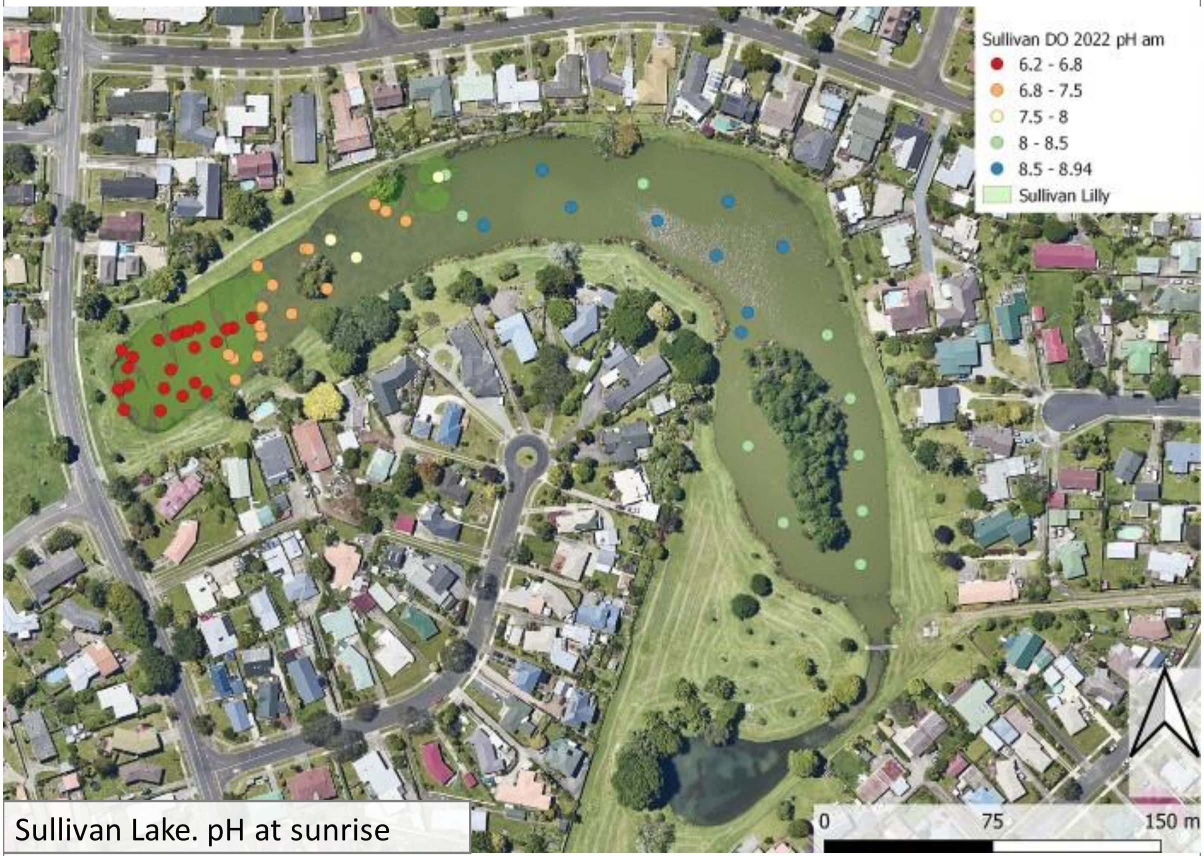


Figure 3.14: Spatial variation of pH in Sullivan Lake during early morning on 5 April 2022. pH is very high in the main body of the lagoon (consistent with photosynthesis) and low at the western end (consistent with respiration/decomposition).

Figure 3.15: Spatial variation of pH in Sullivan Lake during late afternoon on 9 April 2022. pH is very high in the main body of the lagoon (consistent with photosynthesis) and low at the western end (consistent with respiration/decomposition).

3.6.3.3 Summary of DO regime

Overall, the DO regime in Sullivan Lake has high temporal and spatial variability. The lake often has very large diurnal fluctuation in DO due to algae blooms, but the DO regime also appears influenced by heavy rain flushing algae biomass, BOD loads associated with stormwater, the growth of macrophytes curled pondweed moderating phytoplankton biomass, the collapse of curled pondweed exerting an oxygen demand, and aeration from strong winds. At the western end of Sullivan Lake, where waterlily was prevalent, pH and DO concentrations were lower and daily fluctuations smaller than in the main body of the lake. This is likely due to both oxygen demand of organic sediments, and shading by waterlily supressing algae growth.

3.6.4 Metals from stormwater

Stormwater enters Sullivan Lake from the Whakatāne South Stormwater catchment; this includes stormwater from the industrial area around Te Tahi Street. Hamill (2017) found that the lake water itself is well within ANZG default guideline values (**DGV**) for all metals, however the lake sediment was moderately high in zinc (i.e. between the sediment DGV but within the guideline high values). This was likely due to stormwater inputs over many years, including stormwater associated with industrial activities near Te Tahi Street culvert. Other heavy metals (i.e., copper, lead) had sediment concentrations less than the sediment DGV.

3.7 Water quality issues affecting Sullivan Lake

Key ecological and water quality issues identified in Sullivan Lake include:

- Poor water quality: Low clarity, high nutrients (particularly phosphorus), cyanobacteria blooms, and low dissolved oxygen.
- Water quality not suitable for recreational bathing during frequent cyanobacteria blooms and *E.coli* bacteria.
- Extreme fluctuations in dissolved oxygen with daily minimum (night-time) DO commonly below guideline values.
- Lower dissolved oxygen at the western (King Street) end of the lake, probably due to a high sediment oxygen demand.
- Water lily cover expanding at the western (King Street) end to cause issues for aesthetics and boat usage.
- A lack of native aquatic macrophytes to regulate water quality and phytoplankton growth, and providing habitat for invertebrates and fish.
- The recent occurrence of *P. crispus* in Sullivan Lake improves water quality during spring growth, but can reduce DO following collapse and senescence of beds during mid-summer. There is opportunity to improve water quality by harvesting. If harvesting is not undertaken prior to collapse of macrophyte beds, then adverse effects on DO could be minimised by increasing the volume of flow augmentation during this time.
- Lack of emergent macrophytes or wetland margins to improve water quality and provide habitat for invertebrates and fish.
- Siltation from sediment shallowing the lake over the long-term.
- Occasional outbreaks of avian botulism during summer.
- Lack of fish passage for migratory fish (e.g. shortfin eel and inanga).
- Pest plants in drain downstream of Sullivan Lake.

4 Management Actions to improve Sullivan Lake

4.1 Introduction

The Sullivan Lake Reserve Management Plan (**SLRMP**) was updated in 2015 and includes goals to both manage and enhance the conservation values, and to manage and improve the water quality of the lagoon. Although the SLRMP does not identify specific targets, it is clear that further interventions are required to improve the water quality and ecology of the lake.

Approaches to improving lake water quality have been described in several recent reviews for New Zealand lakes (e.g., Abell *et al*. 2020, Hamilton 2019, Abell 2018, Hill 2018, Gibbs and Hickey 2012). Abell et al. (2020) grouped restoration techniques as: a) controlling external loads, b) controlling internal loads, c) biomanipulation and d) hydraulic manipulation. A summary of restoration techniques described in Abell *et al*. (2020) is in **Appendix 2**.

Aquatic macrophytes can be perceived as a nuisance by some lake users. While excessive growth can cause water quality problems, they also have a vital role in maintaining lake water quality and ecology. There are multiple control options for macrophytes that have been discussed in detail in de Winton et. al (2013) and are described on the NIWA website: https://niwa.co.nz/freshwater/our-services/aquaticplants/outreach/weedman .

Sullivan Lake will require an integrated approach that reduces external and internal nutrient loads, and enhances biological processes mediated through aquatic macrophyte and wetland vegetation. Potential intervention measures to address specific water quality and ecological issues in Sullivan Lake are described in **Table 4.1**. A sub-set of these management interventions were selected based on their potential benefits and input from WDC. This section describes these management option, including their benefits, risks and value for money.

The key management interventions assessed for Sullivan Lake are:

- Reducing catchment sediment and nutrient loads (including use of silt traps)
- Increase flow augmentation during summer.
- Treatment wetland to trap sediment, nutrients and improve biodiversity.
- Floating wetlands to remove nutrients and improve biodiversity.
- Sediment phosphorus (P) locking to reduce internal load of P (e.g. using alum).
- Dredging to remove organic sediments near waterlilies.
- Bottom-lining to contain the spread of water lily.
- Harvesting macrophytes to manage plant cover and remove nutrients.
- Grass carp to control / remove macrophytes.
- Silver carp to control phytoplankton.

Table 4.1: Potential management options to address ecological and water quality issues in Sullivan Lake

Issue	Cause	Potential management options
Supertrophic high concentration of nutrients, algae, poor clarity. Cyanobacteria blooms.	External nutrient load.	o Create treatment wetlands at western and southern end of lake. o Floating wetlands for N removal and habitat (risk of root attachemnt in very shallow areas) o Continue to reduce risk of sewage overflows.
	Internal nutrient load via sediment resuspension, anoxia, plant senescence and/or waterfowl.	o Dredging of fine sediment at western end of lagoon. o Harvest curled pondweed in early summer to remove nutrients and avoid collapse. o P-locking
Poor oxygen conditions	Phytoplankton and cyanobacteria blooms	o Manage nutrient loads (as above) o Ensure areas of macrophytes in lake to help suppress excess algae (i.e. maintain some areas of waterlily). o Increase volume of flow augmentation during summer to increase flushing.
	Sediment organic matter exerting a BOD load in eastern end	o Dredging at western end (King St) where more organic muds and lower DO. o Consider harvesting some waterlily prior to winter senescence to reduce BOD load.
Excessive aquatic plant cover affecting recreation and aesthetics	Waterlily cover dominating western end. Provides many WQ benefits but excessive cover adversely affects asethetics, recreation and possibly DO. Very few other macrophytes, but recently curled pondweed growing during spring/summer.	o Manage waterlily extent by dredging soft sediment and lily at western (King St) end (e.g. within 10m from shore). o Contain lily regrowth using mats to cover sediment. o Harvest curled pondweed in early summer prior to collapse. o Harvesting is preferable to spraying as it avoids the risk lof releasing nutrients and oxygen demand. o Grass carp are not recommended for Sullivan Lake as they risks with worsening eutrophication.
Pest plants in drain downstream of Sullivans Lake	Egeria, Elodia	o Direct removal o Targeted herbicide spray
Siltation	Long term siltation from inflowing stormwater. Also biomass accumulation at King Steet end	o Silt traps for main stormwater inflows and eastern end (foot bridge) with possible flocuulation. o Dredging at western end (King St). o Limited opportunity for sediment control devices in stormwater catchment.
Occasional outbreaks of avian botulism during summer	Associated with warm water, anoxic sediments and high density of waterfowl.	o Remove carcasses of dead birds from the lake and margins.
Litter	Rubbish directly and via stormwater	o Operational street sweeping o Litter traps o Regular "pick-ups"
Riparian management restricting development of marginal wetlands		o Plant native riparian vegetation to optimise habitat values. o Create riparian wetlands in water with sloping banks.
Enhance biodiversity		o Create wetlands, consider floating wetlands for invert., bird and fish habitat. o Animal pest control (Halo)

4.2 Reduce external nutrient loads from catchment

4.2.1 General Description

A major driver of lake eutrophication is excess nutrient loading from the catchment, and reducing external nutrient loads is an important strategy for lake restoration. The control of both nitrogen and phosphorus is important in New Zealand lakes where nitrogen limitation of phytoplankton biomass accumulation is common (Abell et al. 2010).

Successful control of external nutrient loads requires knowledge of where, when and how nutrient losses are occurring from the catchment. For many lakes, diffuse pollution from agriculture contributes the majority of nutrients (Gluckman 2017). But in urban catchments, point sources (e.g. sewage or sewage overflows) can be a major source of external nutrient loads and controlling these can provide substantial nutrient load reductions. A summary of key measures to reduce external nutrient loads is provided in **Appendix 3**.

4.2.2 General Application and Constraints

There is typically a lag between reducing external nutrient loads from the catchment and improvements in lake water quality because it takes time to reduce the stores of nitrogen and phosphorus within the lake sediments. Jeppesen et al. (2005) reviewed changes in 35 lakes subject to external nutrient load reductions and found that in-lake TN concentrations typically took <5 years to decline, but in-lake TP typically took 10-15 years. This reflected slower removal of internal phosphorus loads compared to removal of nitrogen by denitrification.

4.2.3 Cost-effectiveness

McDowell and Nash (2012) found that land management strategies (e.g. fertiliser management) were the most cost-effective way of mitigating phosphorus exports. Edge-of-field strategies, which remove P from runoff (i.e., wetlands) or prevent runoff were less cost-effective, but had other benefits including removing other contaminants like nitrogen. Similarly in urban areas, addressing external nutrient loads at source is often the most cost-effective management strategy.

4.2.4 Application to Sullivan Lake

4.2.4.1 Suitability

Potential options to reduce external nutrient loads entering Sullivan Lake from its catchment include:

- Further minimising the (already low) risk of sewage overflows by identifying the potential for stormwater ingress into the sewage system, tracing and addressing any stormwater cross-connections.
- Encourage no or low fertiliser use in the lake's catchment area. Where it must be used, encourage slow-release fertilisers and application when rain is unlikely.
- Use P-sorbents within waterways.
- Sediment traps within culverts.

Lake catchments are particularly sensitive to nutrient inflows. This should be reflected in the priority for reducing the risk of stormwater ingress in the catchments of Sullivan Lake and Awatapu Lagoon. These is also potential for Whakatane District Council to be involved in education of property owners of ways to reduce nutrient loads from urban landuse in the catchment of these lakes.

The practical implementation of P-sorbents within waterways is restricted by the nature of the catchment being either urban or very steep. However, there may be opportunity to use P-sorbents at the outlet of culverts as they enter Sullivan Lake. McDowell et al. (2007) described the use of melter slag contain in a mesh bag (called "P socks") and placed on the bed of the Mangakino Stream (Lake Rerewhakaaitu) to sorb phosphorus. These reduced, on average, the concentration of DRP and TP by 35% and 21% respectively, and reduced loads by 44% and 10% respectively. They were more effective at low-flow.

There is potential to trial melter slag P-socks at the culverts near the Sullivan Lake footbridge, but some additional sampling of inflows is recommended assess their likely effectiveness. Regular monitoring would be needed to assess their effectiveness over time and when they would need to be maintained and replaced.

The implementation of sediment traps in the catchment is restricted by the steep topography. A practical location for sediment traps is at the outlet of culverts as they enter Sullivan Lake. This is discussed further in the context of wetland forebays. There may also be potential for installing proprietary devices to trap sediment within some culvert inlets, for example along Valley Road. Practical locations would need to be identified; they would be most effective on coarse sediment rather than fine sediment[9]. Installing sediment capture devices within the urban stormwater network would require significant capital work and, like all sediment traps, would require regular monitoring and maintenance.

4.2.5 Summary

Reducing external nutrient loads very important for lake restoration and reducing eutrophication. Reducing nutrient loads from within the catchment is often also very cost-effective. For Sullivan Lake there may be potential to:

- Further reduce the (currently low) risk of sewage overflows during heavy rain by prioritising the catchment for reducing storm water ingress.
- Educate land owners about reducing sediment and nutrient discharges to the stormwater network.
- Use P-socks at culvert outlets to bind phosphorus.

[9] E.g. SPEL Stormceptor claims up to 83% removal of suspended solids, 100% removal of sediment >3mm (larger than coarse sand), and 99.9% removal of light liquids (e.g. hydrocarbons) (SPEL Stormceptor product brochure).

- Install sediment traps at culvert outlets (see discussion on wetlands). There may also be opportunities at culvert inlets but practicality and costs need further investigation.

4.3 Increase flushing by flow augmentation

4.3.1 General Description

Manipulating lake inflows to promote flushing can support lake restoration by increasing the rate of phytoplankton algae removal or by diluting poor water quality with higher quality water. Generally, flushing of algae is only effective when it can reduce the hydraulic residence time to less than the time it takes for phytoplankton to double their biomass (c. <20 days) (Jørgensen 2002, Hamilton 2019).

Biological uptake often reduced dissolved nutrients to low levels in lakes. Thus, introducing only a small amount of water, without sufficiently reducing the residence time, can create a risk of introducing additional nutrients in a bioavailable form that promotes additional phytoplankton growth.

The goal of increasing flushing was a major driver for implementing the re-diversion of the Kaituna River to the Maketū Estuary. In this situation, higher flushing by river water has helped reduce the biomass of macroalgae accumulated on the mudflats.

4.3.2 General Application and Constraints

The potential to increase hydraulic flushing is very lake specific and requires a suitable donor water body nearby which can dilute poor quality water with higher quality water, and/or sufficiently increase flushing rates. Consideration also needs to be given to the quality of the water being used for flushing to avoid making water quality issues worse.

Increasing flushing to Sullivan Lake, as proposed below, will not prevent the formation of cyanobacteria blooms, but it will help reduce the intensity of these blooms by removing algae biomass.

4.3.3 Cost-effectiveness

The cost-effectiveness of using flushing to improve lake water quality is very site specific. In the case of Sullivan Lake, increasing the volume of flow augmentation from the Whakatāne River during summer is likely to have moderate cost-effectiveness.

Adding flow to increase flushing will have most benefit during summer/autumn dry periods. It will be considerably less cost-effective during winter when rain-events are common. Augmenting the flow to Sullivan Lake is easy to scale up and adjust according to rain conditions.

4.3.4 Application to Sullivan Lake

Augmenting flow to Sullivan Lake during summer has good potential to improve water quality by increasing flushing via flow augmentation from the Whakatāne River. This should occur summer dry periods when water quality in Sullivan Lake is worse (e.g. cyanobacteria blooms) and water quality in the Whakatāne River is better (e.g. lower dissolved nitrogen).

TN and TP concentrations are typically lower in the Whakatāne River (median 0.19 mg/L and 0.05 mg/L respectively) than Sullivan Lake (median 0.48 mg/L and 0.14 mg/L respectively); but bioavailable DIN and DRP are higher in the Whakatāne River (median 0.12 mg/L and 0.03 mg/L respectively) than in Sullivan Lake (median 0.01 mg/L and 0.018 mg/L respectively)[10]. During summer (December-March), the concentration of dissolved N and P in the Whakatāne River are lower (median DIN and DRP of 0.04 mg/L and 0.023 mg/L respectively), although DRP remains relatively high due to natural geology influences. To minimise the risk of dissolved nutrients in Whakatāne River water stimulating further algae growth, flow augmentation should be limited to January to March (inclusive), with the option of extending this to December to April (inclusive) if proceeded by at several weeks of dry weather and below median river flows.

Flow augmentation should also be applied in conjunction with installing treatment wetlands that will help remove incoming nutrients.

4.3.4.1 Proposed implementation

WDC currently pumps up to about 40 m^3/hour (11.1 L/s) into Sullivan Lake from the Whakatāne River to improve flushing. However, in practice it is often closer to 3 L/s. During summer there is little rain to support natural inflows and lake water quality is typically worse. At this time (January to March inclusive), flow augmentation into Sullivan Lake from the Whakatāne River should be maintained at c. 40 m^3/hour to 50 m^3/hr (11.1 to 14 L/s).

Ensuring the flow pumped into Sullivan Lake of 11 to 14 L/s during summer will ensure a hydraulic residence time of about 20 days even during periods of low flow - which is sufficient to flush phytoplankton.

4.3.4.2 Cost

Infrastructure is already in place. To increase the flow augmentation to Sullivan Lake may require installing and running a larger pump. Rough order costs are[11]: capital expenditure under $5000 and annual operating costs of c. $2000. Infrastructure is mostly already in place. The capital cost is low and it is easy to trial.

4.3.5 Summary

Ensuring flow augmentation from the Whakatāne River can reduce the intensity of algae blooms in Sullivan Lake by increasing flushing for relatively low cost. This would have most benefit and lowest risks if undertaken during summer (January to March) and when implemented in conjunction with the creation of treatment wetlands. Flow augmentation provides little benefit during rain events. If treatment wetlands are constructed, than flow augmentation should be reduced during rain events to ensure wetland treatment of stormwater is optimised.

[10] Hamill et al. 2020

[11] Based on increasing the flow rate to 1000 m^3/day (11.6 L/s) and lifting to 8m head, will require a pump size of about 1.5kW or two pumps of about 1.1kW, plus piping. Operating cost base on kWh for a pump operating 24/7 for 4 month using the pump power calculator in https://www.engineeringtoolbox.com/pumps-power-d_505.html

4.4 Treatment Wetlands

4.4.1 General Description

Wetlands are the 'kidneys of the landscape'. They are a natural interface between land and water that cleans the water. Contaminants are attenuated and removed through processes of denitrification, plant uptake, deposition, adsorption and mineralization. Emergent wetland plants filter the water, enhance denitrification and help remove and immobilise heavy metals from the water (e.g. Kadlec and Wallace 2009, Guigue J et al. 2013).

Constructed treatment wetlands are commonly used to remove sediment, nitrogen (**N**) and phosphorus (**P**) from surface water. Constructed wetlands replicate and optimise the treatment mechanism found in natural wetlands including: denitrification, uptake and storage by plants, precipitation, settling and burial within sediment, and sorption of phosphorus to material.

Numerous guidelines are available to inform the design of treatment wetlands (e.g. Tanner 2020, Farrant et al. 2019). Some key aspects of treatment wetland design are:

- Wetlands should be sized to keep water velocity sufficiently low to avoid scour and to provide sufficient residence time to achieve the required removal rates. contaminant reduction efficacy increases as constructed wetland area increases, but with gradually diminishing returns. Often wetlands are sized to be between 1% and 5% of their contributing catchment (i.e. 100-500 m^2 of wetland per ha) [12].
- Flow must be dispersed across the wetland so that there is minimal short circuiting. This can be achieved by attention to dispersion of inflows, having a length to width ratio of between 5:1 and 10:1,[13] dense planting across the wetland, and banded planting perpendicular to flows.
- Incorporate a sediment forebay/sedimentation pond to settle sediment and assist with regular maintenance. Sedimentation ponds are often sized as 10% of the wetland size or alternative between 40 m^2/ha and 80 m^2/ha of catchment depending on the rainfall intensity.
- Maintain water depths at 0.2-0.4 m to maintain healthy emergent wetland plants and optimise nutrient removal. Deeper water (>1.2m) zones help disperse the flow across the width of the wetland.
- Use soils with low potential for release of P. This might be achieved by mixing with sub-soil or P-retaining material (e.g. allophane, tephra) (Ballantine and Turner 2010).
- Maximise ancillary benefits for biodiversity by using a diverse range of locally sourced wetland plants.

[12] Small wetlands still remove contaminants but have lower percentage removal rates and need more attention to design for bypass flows to avoid being overwhelmed by stormflows.
[13] Not less than 3:1

In-lake wetlands and riparian wetlands work in the same away as treatment wetlands by intercepting and treating groundwater or runoff percolating through the soil. They also provide habitat for zooplankton that predate on phytoplankton and provide a natural control on their biomass.

4.4.2 General Application and Constraints

Treatment wetlands are extensively used to treat stormwater, wastewater and stream inflows to lakes. They are often used to remove sediment, nutrients (N and P), and metal contaminants. The effectiveness of wetlands for nutrient removal depends on a range of factors including: design, hydraulic loading, incoming nutrient concentrations and seasonal temperatures.

Misch et al. (2000) estimated sustainable annual removal rates for non-point source nitrogen and phosphorus of respectively 100 – 400 kg N/ha and 5 – 50 kg P/ha. Hamill et al (2010) used empirical relationships developed by Kadlec and Wallis (2009) to calculate average annual removal rates for constructed wetlands to treat water in the Rotorua catchment of 368 kg N/ha and 11 kg P/ha of wetland. The lower removal rate for P is due to both lower concentrations of P in the incoming water and less efficient removal of dissolved P.

Tanner et al. (2020) calculated the performance of constructed treatment wetlands for pastoral runoff. An appropriately constructed wetland sized at 2% of the catchment area would remove 65%, 36% and 35% of TSS, TN and TP respectively. But this assumes that most P is in particulate form associated with sediment. Wetlands are not very effective at removing P in dissolved form.

Phosphorus removal rates in constructed wetlands can vary widely depending on the design and past land use. If the underlying soil is high in phosphorus, then the wetlands can desorb phosphorus and be a net source of phosphorus. The risk of this occurring can be mitigated, and the ability of wetlands to retain phosphorus enhanced, by augmenting the sediment with phosphorus binding material.

4.4.3 Cost-effectiveness

Wetlands provide multiple benefits to support ecological functions, nutrient removal and biodiversity. Constructed wetlands can be a cost-effective way of removing sediment and nitrogen (estimated as $79 / kg N /yr), but are less cost-effective at removal of phosphorus (estimated at $2550 kg P/yr) (Hamill et al. 2010)[14]. Cost-effectiveness for phosphorus removal is considerably improved if the source of P is predominately associated with particles P sorbing material is used and the sediment forebay is well maintained.

4.4.4 Application to Sullivan Lake

4.4.4.1 Suitability

There is potential to build an in-lake wetland at the southern end of Sullivan Lake near the footbridge to treat the main inflows of and enhance the current sediment trap. These locations allow the wetland to treat water from the main inflows to the lake. This would provide multiple benefits of reducing

[14] Based on long-term sustainable removal rates (excluding sorption to wetland sediments) and using whole-of-life costs (including land acquisition, maintenance and rejuvenation).

sediment accumulation, improved water quality, improved biodiversity and habitat for invertebrates, fish and birds.

A possible layout for a treatment wetland to intercept the main inflows to Sullivan Lake is shown in **Figure 4.1**. This would consist of two treatment wetlands, one in the south capturing Te Tahi Street culvert with an area of c. 1200 m^2 (plus forebay) and one near the foot bridge with an area of 2400 m^2 (plus forebay). The presented wetland layout is smaller than ideal for a treatment wetland[15], nevertheless it would provide cost-effective treatment of inflows in terms of contaminant removal. The forebay at the footbridge for sediment settling currently exists, but would be deepened as part of the work.

The water depth in shallow zones of the wetland should be about 300-400mm, but most emergent wetland plants need to be established in shallower water below the height of the shoots (e.g. about 100mm deep). This can be achieved by either temporarily lowering water levels, or by planting along shallower edges and allowing plants to spread naturally over time. Once established, plants can survive periods of exposure and extend into deeper water. Deep zones (e.g. >1.2m) prevent the vegetative spread of emergent macrophytes.

Riparian wetlands would be low cost and easy to establish using a long reach digger from the lake edge to redistribute sediment.

There are a number of native emergent plants suitable for Sullivan Lake including: *Eleocaris sphacelate, Machaerina articulata*[16]*, Carex secta,* and *Schoenoplectus tabernaemontani*. *Typha orientalis* (raupo) could be considered but would need care to ensure it is contained by surrounding deep zones (F**igure 4.2**)

4.4.4.2 Cost

The cost of establishing areas of wetland filters near stormwater outlets of Sullivan Lake is estimated to cost in the order of $50,000 to $90,000, plus consenting costs. The cost will vary depending on earthwork requirements to shallow some areas and deepen other areas, and the extent of initial planting. Some low P substrate may need to be imported to improve P binding.

The budget will need to allow for control of pest plants during establishment and ongoing removal of sediment from the forebays.

4.4.5 Summary

Treatment wetlands are common and cost-effective way to filter water to remove sediment, nutrients and metals. Wetlands also support ecological functions in lakes and enhance biodiversity. There is good potential to incorporate both treatment wetlands and riparian wetlands into Sullivan Lake to treat inflows and improve biodiversity values.

[15] The catchment contributing to these two wetlands would be about 50 ha, so the total wetland area would need to be about 0.5 ha to be 1% of the catchment.

[16] Formally *Baumea articulata.*

Figure 4.1: Potential layout for a treatment wetland and riparian wetlands in Sullivan Lake to improve water quality and provide biodiversity values.

Figure 4.2: An example of *Baumea sp.* growing along a lake wetland margin (from Tanner et al. 2021).

4.5 Floating Wetlands

4.5.1 General Description

Floating wetlands consist of buoyant mats or platforms that are mass planted with emergent wetland plants, and are anchored on the surface of treatment ponds or nutrient rich lakes. The plant roots grow through the mats and down into the water column forming large, dense mats. Large root systems develop to allow the plants to obtain their nutrient requirements from the water column. Localised anaerobic zones are created beneath/within the floating mats where the process of denitrification is favoured. Biofilms develop over the extensive root surface area and serve to increase organic matter breakdown, nutrient adsorption and trapping of fine particulates (Sukias 2010).

The shade provided by the plant mats reduces algal growth and results in increased settling of suspended solids onto the bottom of the lake.

4.5.2 General Application and Constraints

Floating wetland are widely used around New Zealand for water treatment and ecological enhancement. To be most effective, floating wetlands need to be installed in a location where there is a flow of water passing through them. They are not very effective at removing nutrients if placed in a lake without any current or flow.

Floating wetlands are best used in deeper water (e.g. >1 m) where the plant root systems will not reach the sediment.

The harvesting of plant material is important for long-term sustainable nutrient removal by floating wetlands, and this is particularly important for phosphorus removal (Pavlineri et al. 2017). Some ongoing maintenance is required to control weeds.

The buoyant mats of some floating wetlands can degrade over time and release plastic into the water; however, this can be avoided by using rafts made of HDPE.

4.5.3 Cost-effectiveness

Floating wetlands have similar removal mechanisms to conventional wetlands but are about twice as effective at removing nitrogen and phosphorus as conventional constructed wetlands. Where located where water flows, nitrogen removal rates for floating wetlands are about 584 – 876 kg/ha/yr while phosphorus removal rates are about 7.3 – 18 kg/ha/yr (Tanner et al. 2011).

However floating wetlands are relatively expensive to install, so are best used in situations with high nutrient concentrations to take advantage of their good removal rates, or in situations which utilise their co-benefits in providing for shading the water and providing habitat for birds and fish. Hamill et al (2010) estimated the average cost-effectiveness[17] of floating wetlands as $473 / kg N and $24,000/kg P, however these costs may now be lower with availability of new, cheaper, floating wetland products. Because of their relatively high cost, floating wetlands are better suited to situations that optimise their

[17] Annualised cost spread over 50 years.

treatment ability (i.e., areas with flow and high nutrient concentrations), have space constraints, or where other benefits (e.g., shading, habitat, biomanipulation) are valued.

4.5.4 Application to Sullivan Lake

4.5.4.1 Suitability

Floating wetlands could be installed in Sullivan Lake near the inflows where surface flow treatment wetlands are proposed. They could achieve a similar amount of nutrient removal as surface flow wetlands in about half the area, but they would cost considerably more, and are thus not recommended for mass deployment.

A small number of floating wetlands would be beneficial to enhance settling in sediment forebays, where flows are greatest. Also, their use on the main body of a lake for enhancing habitat remains valuable. Floating wetlands can be used as floating nurseries, with mature plants harvested and planted along the lake's riparian margin.

4.5.4.2 Cost

A rough order cost to install a set four floating wetlands in Sullivan Lake, with an effective surface area of about 16.4 m^2, is $5,000 to $10,000.

4.5.5 Summary

Floating wetlands are a widely used and effective way to remove sediment, nutrients and other contaminants from water. In addition, they provide co-benefits of shading the water and providing habitat for invertebrates, fish and birds. However, they are more expensive than surface flow wetlands, and less cost-effective for most natural waterbodies. Installing floating wetlands in Sullivan Lake would be beneficial, but would be less cost-effective than creating a larger area of conventional wetlands.

4.6 Dredging

4.6.1 Description

Dredging removes lake bed sediments which both deepens the lake and directly removes accumulated nutrients. It is a well-established method to control internal nutrient loads, and is particularly useful when surface sediments are rich in nutrients and anoxic conditions facilitate the release of these nutrients (Abell et al, 2021, Bormans et al, 2016).

Dredging can substantially reduce sediment nutrient releases in small lakes, and can result in considerable improvement in ecological health. Increasing the depth of the shallow lakes can also reduce the wind-driven resuspension of sediment and nutrients even when deepening is limited to localised areas (Penning et al, 2010 in Abell et al, 2021).

Sediment from dredging needs to be safely disposed. A dredging operation is more efficient if the material can be safely disposed of locally. Material could be used to develop area suitable for riparian wetlands but consideration should be given to immobilising nutrients.

4.6.2 Application and Constraints

The effectiveness of dredging depends on the depth of the lake, composition of the sediment, and ability to target organic and nutrient rich sediments. Its benefits will be limited if organic, nutrient rich or contaminated sediments extend deeper than the depth being dredged. Sampling and testing sediments is valuable prior to undertaking dredging.

Dredging is best undertaken in conjunction with reduction in external nutrient loads to reduce the rate at which surface sediments again become enriched.

Dredging operations are intrusive; it disturbs the sediment and can cause short term reductions in water clarity. In some lakes with anoxic sediments, dredging can release sulphide which can be toxic to aquatic life. It also directly removes benthic fauna, which can be a major disadvantage for its application in some lakes where kākahi or koura are present (this is not the case for Sullivan Lake).

4.6.3 Cost-effectiveness

Dredging is expensive and because of the expense, its use is generally restricted to small, iconic lakes. Hamilton et al. (2014) estimated dredging costs of $100,000 per ha for small lakes, but this will be an under-estimate for Sullivan Lake due to its very small size and urban setting.

4.6.4 Application to Sullivan Lake

4.6.4.1 Suitability for lake

The shallow depth of Sullivan Lake makes it suitable for dredging and suction dredging near the footbridge occurred in 2019. Dredging part of the western end of Sullivan Lake has the potential to achieve multiple goals of physically removing a part of the currently extensive water lily cover, removing organic sediments that are likely contributing to depressed dissolved oxygen concentrations, potentially reducing some internal nutrient load, and deepening sections of the lake.

One option is to remove organic sediment and water lily from the King Street end of Sullivan Lake, west of the island, in a *c.* 10m wide band around the lake edge. This would cover about 210m of lake shore. Removal of 0.5m of material over this area would equate to 1000 m^3. Water lily near the centre will be retained. Bottom-liner mats can be laid over the dredged area as a barrier to prevent regrowth of waterlily (discussed below).

The operation could occur with either a long reach digger or suction dredge. The material will need to be dewatered and disposed of and will involve heavy machinery.

Prior to dredging the substrate should be tested for nutrients and organic matter at different depths to inform the potential success of the operation and the depth to which dredging should occur.

4.6.4.2 Cost

A rough order cost of the proposed dredging and disposal is $65,000 to $120,000, excluding the cost of resource consents, contingencies or any bottom-liners (discussed below).

4.6.5 Summary

Dredging is relatively expensive, but dredging part of the western end of Sullivan Lake could provide multiple benefits of reducing water lily cover, improving the oxygen regime, and reducing internal nutrient loads.

4.7 Phosphorus Locking

4.7.1 General Description

Phosphorus locking and flocculation is commonly used for lake restoration around the world. The internal load of phosphorus from lake sediments is reduced and made unavailable for algae use by applying chemicals to bind and inactivate the phosphorus in the water column and as it is mineralised and released from the sediment.

A number of materials can be used to adsorb dissolved phosphorus from lake water or inflows and thus reduce the bioavailability of phosphorus within the lake. These can be applied directly to the lake surface or continually drip dosed into a stream inlet. The materials often also cause the flocculation of suspended sediments from the water column. Many products can be used to bind dissolved phosphorus but the most commonly used and/or effective for lakes are aluminium sulphate ('alum'), Aqual-P (an aluminium zeolite combination product), and Phoslock (bentonite clay modified with lanthanum) (Douglas 2016, Wagner 2017, Abell et al. 2021).

Flocculation can be enhanced by adding a separate flocculant; commonly used flocculants include polyaluminium chloride (PAC) and polyacrylamide (PAM). PAM is promising as a flocculant in turbid freshwater systems because they are very efficient and can have low eco-toxicity when formulated in the anionic form (Gibbs and Hickey 2017). Products such alum, Phoslock and Aqual-P perform a dual function of adsorbing dissolved phosphorus and physically capping the sediment.

The alum causes aggregation of particulate matter and causes it to sink to the lake bed and this has potential to removed cyanobacteria /algae within the water column. On the sediment surface, alum forms a thin layer a few millimetres thick, and this layer of alum can sequester DRP as it released from the sediment.

Phosphorus locking methods are widely used in lake restoration including their successful use in Lake Okaro and Lake Rotorua (McBride et al. 2018, Hamilton 2019, Abell et al. 2021). However, the effectiveness of phosphorus locking for lake restoration is lake specific depending on water chemistry, hydraulics, timing and the presence of macrophyte beds. For example, it has been highly effective in inflows to Lake Rotorua but has had very limited effect in inflows to Lake Rotoehu – likely due to interference by ions in geothermal waters and flocculation with hornwort beds (Eger 2018).

4.7.2 General Application and Constraints

It is important to consider site-specific constraints when identifying appropriate products and application strategies. pH is an important consideration; pH >8.5 results in the release of phosphorus bound to aluminium or iron, making products like alum and Aqual-P ineffective. For alum applications in low-alkalinity lakes, it is necessary to use with a buffer (e.g. sodium carbonate or bicarbonate) to maintain pH >6.5 and avoid the formation of toxic Al^{3+} ions (Hickey and Gibbs 2009).

Consideration should be given to the potential ecotoxicological effects of materials being used to avoid acute or chronic effects on lake ecology, but assessment of these risks is well documented (Tempero 2015, 2018, McBride et al. 2018). Consideration also needs to be given to cultural concerns regarding the application of material to lakes.

Geoengineering using phosphorus locking needs to be tailored for a specific lake. It is advisable to evaluate efficacy based on jar tests, laboratory experiments and small-scale field trials. Consideration needs to be given to the costs, ecotoxicity and risk of smothering benthic biota (Hickey and Gibbs 2009).

Table 4.2 provides a summary of geoengineering materials and their applications. The materials likely to be most applicable to Sullivan Lake or Awatapu Lagoon are alum, Phoslock and Aqual-P. PAC is a flocculant and can be used in combination with alum of Phoslock which absorb the phosphorus. For the immediate management of cyanobacteria blooms, algaecide (e.g. hydrogen peroxide) can used in before phosphorus locking to reduce the risk them later floating from sediments to re-emerge as blooms. Alum and Aqual-P have reduced P binding at high pH.

The longevity of phosphorus locking will depend on incoming nutrient loads, rates of burial and resuspension. Sediment locking is typically less effective in shallow lakes because of higher rates for burial and wind resuspension. One study found alum treatment was typically effective for 15 years in deep lakes compared to five years in shallow lakes (Huser et al. 2016 in Abell 2018). In Lake Ōkaro, alum treatment has been undertaken twice a year for most years since 2013 to control algae.

Scholes (2018) test two water treatment products in mesocosm trials in Sullivan Lake, these were PAP-5 Melter slag (a by-product of the iron making process) and Pond Treat PT-450 (anon-pathogenic microbial enzyme treatment). Both were effective at reducing algae biomass, reducing nutrients and improving water clarity, but he noted that the use of these products should be in conjunction with controlling external inputs.

4.7.3 Cost-effectiveness

The use of phosphorus locking material to control eutrophication can be effective, reliable and cost-effective. One study of four urban lakes found in-lake alum treatment was *c*. 50 times more cost-effective than catchment-based measures to reduce storm water nutrient loads (Huser et al 2016). However, they are not suitable for all lakes.

4.7.4 Application to Sullivan Lake

4.7.4.1 Suitability

Sullivan Lake is nitrogen limited but reducing phosphorus concentrations is important for controlling cyanobacteria. The source of phosphorus in Sullivan Lake is not well understood, there may be a legacy of phosphorus rich sediment from historical sewage spills. Anoxic conditions that would cause P release appear relatively uncommon in the main water column, but there may localised anoxia at waters in the western end of the lake or at the sediment surface. DRP may also be released with wind-induced mixing mobilising porewater with bottom sediments.

Phosphorus locking (probably with alum) may be a useful remediation for Sullivan Lake, but more information is required before determining its likely success, including the extent to which phosphorus is being released from the sediments.

The longevity of applying alum (or another product) is unknown, but may be short-lived if bottom sediment is resuspended or if there is settling of P-rich sediments. Any application of P-locking should occur after creating of treatment wetlands to reduce nutrient and sediment inflows.

To better assess the potential for successful P-locking in Sullivan Lake will require collecting water and sediment samples from around the lake and incubating sediment cores to determine DRP release rates. This information is also required to calculate application rates. Application rates can be calculated using the areal load of TP in the top 4 cm of sediment plus the areal load of DRP in the overlying water. The amount of buffer required is normally about twice the amount of alum but needs to be checked using lake water.

4.7.5 Summary

The use of phosphorus locking material to control eutrophication can be effective, reliable and cost-effective when appropriately tailored for a lake. Phosphorus locking may be a useful restoration tool to control cyanobacteria blooms in Sullivan Lake, but additional investigations are required to determine its likely success, and any implementing of P-locking should occur after creating treatment wetlands to reduce nutrient and sediment inflows.

Table 4.2: Lake geoengineering materials used for phosphorus inactivation and flocculation (reproduced from Table 3 in Hamilton 2019).

Flocculant	Active com-pound	Carrier	Requirements	Cost (approx.)	Side effects/toxicity	Application difficulty
Alum, PAC	Al^{3+}	None.	Mostly used with a buffer; careful checks required to avoid acidification	Low	Free (uncomplexed) Al ion toxicity to biota (primarily a gill toxicant), highly pH dependent (Gensemer and Playle 1999)	Low-medium
Phoslock®	La^{3+}	Bentonite	-	Medium-high	Low-alkalinity waters could lead to greater susceptibility of biota to side effects from La^{3+}	Medium
Chitosan (Zou et al. 2006)		Has been used in association with flocculants for sinking (ballast) purposes	Check for contaminants released by flocculant (if used)	High	Benign: toxicity to higher organisms highly unlikely but appears to act as an algaecide to cyanobacteria	High
Oxygen nanobubble modified natural particles (Zhang et al. 2018)	-	A local mined soil is often used, to which oxygen nanobubbles are impregnated	Has not been scaled up; still experimental (laboratory-scale)	Likely to be high	Benign unless the modified soil releases contaminants	Medium
Aqual-P	Al^{3+}	Zeolite	-	Medium	Evidence to date indicates Aqual-P is relatively benign	Medium

4.8 Macrophyte harvesting to manage aquatic plants and reduce nutrients

4.8.1 General Description

Aquatic macrophytes ('lake weeds') are an important part of lake ecosystems, and moderate water quality by stabilising sediment and cycling nutrients from the sediments and water column. However, excessive growth of (usually) exotic invasive macrophytes can cause a nuisance or contribute to water quality problems. Harvesting and removal of macrophyte biomass can control excessive cover and remove carbon and nutrients from the lake system. This prevents the nutrients being cycled back into the lake water column during periods of pant senescence or die-off.

Macrophyte harvesting in lakes is usually done by a custom-made boat-operated harvester that cuts the plants below the water surface and collects the mown sections (**Figure 4.1**). Larger harvesters can cut plants up to about 2m below the water surface. Harvested material is transported to the lake shore where it is dewatered and removed for disposal (e.g. to compost). To maximise nutrient removal from a lake, the harvested material should be removed from the catchment or treated in a way so as to prevent nutrient leaching back to the lake.

The harvester mows off the top of surface reaching weed beds, it does not pull up the roots, and the macrophytes grow back over time. This regrow is itself be beneficial for water quality as macrophytes reduce the amount of dissolved nutrient available for algae growth.

Macrophyte harvesting can be undertaken using a long reach digger with a modified cutting head, but this method is limited to the reach of the digger, so is more suited to drains.

Figure 4.1: A lake macrophyte harvester in operation on Lake Rotoehu (source: www.lakeweed.co.nz).

4.8.2 General Application and Constraints

Macrophyte harvesting is commonly used to control macrophytes in both small ponds and large lakes. Macrophyte harvesting is commonly used in New Zealand to reduce nutrient loads (e.g., hornwort harvested from Lake Rotoehu by Bay of Plenty Regional Council (**BOPRC**) (Horne 2020)), reduce nuisance macrophyte cover in drains, hydro lakes (e.g. Genesis) and stormwater ponds (Auckland Council).

Its suitability as a method depends on goals for lake management, site constraints and the biomass of plants present. It is effective at managing dense macrophyte beds, however because macrophyte beds help maintain a clear water state in lakes, harvesting operations should be done in a way to ensure weed beds to not collapse without any replacement native communities to replace them.

Harvesting is not a suitable method to eradicate weeds or control new incursions, as plant fragments caused by harvesting can act as propagules.

Harvesting operations need to follow good biosecurity practices to avoid the spread of pest plants and animals. This requires cleaning all equipment before transporting between waterbodies by following the Check, Clean and Dry procedures from Ministry for Primary Industries (**MPI**)[18].

Permission is required from Ministry of Primary Industries (**MPI**) if transporting, outside a catchment, pest plants classified as 'unwanted organisms' under the Biosecurity Act (1993). This would apply to hornwort (*Certatophyllum demersum*) and parrots feather (*Myriophyllum aquaticum*) both found in Awatapu Lagoon, but not to the waterlily or curled pondweed (*Potamogeton crispus*) that occurs in Sullivan Lake.

4.8.3 Cost-effectiveness

Macrophyte harvesting has multiple benefits of controlling excessive macrophyte biomass to maintain recreational and water quality values, maintaining some macrophyte cover to support biodiversity and water quality benefits, and removing a load of carbon, nitrogen (N) and phosphorus (P) from the lake system. An alternative practice of herbicide spraying is cheaper to achieve the single purpose of controlling macrophyte cover, but does not achieve any of the co-benefits for water quality.

Lake weed harvesting of hornwort from Lake Rotoehu (Bay of Plenty) removes about 1.2 kg N and 0.16 kg P per tonne of wet weed (Gibbs 2015). The harvesting from Lake Rotoehu is estimated to cost about $53,000 per year and remove about 320 kg P/yr and 2,400 kg N /yr (Hamilton and Dada 2016), i.e. a cost-effectiveness of $166 /kg P and $22 / kg N. However, the cost of small-scale operations is considerably more. Weed harvesting from Awatapu Lagoon South, Whakatāne in 2019 cost c. $35,000 for c. 200 tonnes of weed which would have removed about 240 kg of N and 32kg of P with a cost-effectiveness of $146 /kg N and 1095 / kg P. This cost included consenting, establishment, harvesting, dewatering and disposal. It may be higher if the weed harvester has to be transported further.

4.8.4 Application to Sullivan Lake

4.8.4.1 Suitability

Macrophyte harvesting could effectively manage the cover or waterlily and/or curled pondweed in Sullivan Lake, but would be most cost-effective on occasions when curled pondweed is growing across the lake. To achieve benefits of managing curled pondweed, the harvesting operation will need to occur after it has become surface reaching and before it collapses, i.e. between about late October and early January. The cost-effectiveness of macrophyte harvesting from Sullivan Lake will be considerably less in years where curled pondweed is not prolific.

Sullivan Lake would require a small harvester, i.e. a smaller boat-based harvester or amphibious machine[19]. Harvested material needs to be dewatered for one to two days and the site chosen for this needs to be accessible to a truck and digger.

Using herbicide to control aquatic plants is cheaper than mechanical harvesting but does not provide any water quality benefit because it does not remove any organic matter or nutrients. Some

[18] https://www.mpi.govt.nz/outdoor-activities/boating-and-watersports-tips-to-prevent-spread-of-pests/check-clean-dry/#CCDmethod

[19] Enviroland's uses an amphibious machine that uses a cutting bar to cut vegetation below the water and once cut, the now floating vegetation is collected and pile on the bank for removal (https://www.envirolands.co.nz/services/).

communities have concerns about the potential health and environmental risks from frequent herbicide use.

4.8.4.2 Cost

The cost of macrophyte harvesting can vary widely depending on the scale of the operation. The cost of macrophyte harvesting in Sullivan Lake is expected to be in the range of $20,000 to $35,000 per harvest, but will depend on the amount to be harvested. The cost is likely to reduce as operations become more efficient, e.g. for composting.

4.8.5 Summary

Macrophyte harvesting is a widely used method for controlling excessive macrophyte biomass, improving water quality, and contributing to long-term removal of nutrients from the lake system. A small harvester could be used in Sullivan Lake between late spring to early summer to reduce the cover of curled pondweed and waterlily. It is more expensive than herbicide spray, but provides water quality benefits not achieved by herbicide spraying.

4.9 Bottom-liner to contain plant growth

4.9.1 General Description

Lining the bottom of waterbodies can be used to eradicate aquatic weeds and reduce regrowth by excluding light and preventing root access to substrate. A range of different material can be used for bottom-liners; plastic or sheets polyethylene woven mats are most common, but liners can also be made from woven cloth, jute or flax. The liners are usually held in place using gravel, sand bags or stakes. They are best installed when plants height is low (e.g early spring) (De Winton et al. 2013).

Bottom-liners have been successfully used to reduce the extent of a *lagarosiphon* cover in Rosie Bay, Lake Waikaremoana, and to manage water lily in Lake Ōkāreka (de Winton et al. 2013).

Harakeke flax mats (called Uwhi) have been trailed in Lake Rotorua (Hamurana springs) to smother invasive aquatic weeds. These have been found to last longer than hessian mats and the Uwhi were also used as a refuge for kōura.[20]

4.9.2 General Application and Constraints

Bottom-liners work better on flat slopes rather than on steep slopes. The build-up of gas from organic sediment can cause bubbles under liners and cause them to dislodge, but this can be avoided by perforating the liners or using woven mats that are permeable to gas. Consideration needs to be given to possible decrease in oxygen at the sediment interface below the liners, and the impact of this on benthic biota and geochemistry (de Winton et al. 2019).

[20] Rotorua Te Arawa Lakes Strategy Group agenda, 23 September 2023. https://nuwao.org.nz/uwhi-harakeke-weed-mats/

Bottom-liners are not effective against floating plants (e.g. *Azolla* sp) and are less effective for non-rooted plants like hornwort.

The effectiveness of bottom liners at controlling regrowth can last for years, depending on the materials used and the lake conditions. The life-span of control will be reduced by build up of sediments over the mats, and maintenance may be required if sediment depth over mats increases to more than 4cm (de Winton et al. 2013).

Hessian/coconut fibre mats have been successful in controlling lagarosiphon, egeria and hornwort, but disintegrate within about 10 months. The use of natural materials that decompose are an advantage if trying to reestablish native plants.

4.9.3 Cost-effectiveness

Most cost occurs with installation but regular checks are advised to ensure bottom-lining remain in-place. Build-up of sediment over the mats may require removal. De Winton et al. (2013) estimated the cost of bottom-liners to be $30,000 per ha, excluding any sediment removal. The expense makes them more suited to small scale applications.

4.9.4 Application to Sullivan Lake

4.9.4.1 Suitability

In Sullivan Lake, placing bottom-liners following dredging of water lily could be an effective way to control water lily regrowth. The shallow water depth and flat bottom of Sullivan Lake makes bottom-liner relatively easy to install. Permeable woven mats should be used to allow gas exchange and avoid the build up of gases. A minimum liner width of 3m should be used to avoid water lily rhizomes.

4.9.4.2 Cost

The rough order cost of bottom-lining to contain water lily following dredging is about $5000 to $9,000. Assuming bottom-lining would occur in a 3m wide strip around 200m perimeter of water lily (600m^2). This excludes any consenting or maintenance costs.

4.9.5 Summary

Bottom-lining could be an effective method to control the spread of water lily in Sullivan Lake following its removal by dredging. There is potential to trial in a small area to assess its effectiveness.

4.10 Grass Carp to Control Aquatic Plants

4.10.1 General Description

Grass carp (*Ctenopharyngodon idella*) are introduced herbaceous fish that are bred in New Zealand for aquatic vegetation control. They are non-selective grazers and if stocked in sufficient numbers grass carp can completely eradicate submerged aquatic vegetation. Even when all submerged vegetation is gone, a few fish can often survive by consuming fallen leaves, riparian grasses and epiphytic algae (de Winton et al. 2013).

4.10.2 General Application and Constraints

Grass carp are only suitable in lakes requiring the near complete eradication of aquatic plants. They will eradicate both submerged plants and emergent wetland plants if the water is depth is sufficiently deep (e.g. 0.5m), although they don't graze some short growing turf plants or floating plants (e.g. *Azolla*) (Rowe and Schipper 1985 in de Winton 2013). This makes grass carp unsuitable for lakes where retaining aquatic plants or wetlands are required for water quality or habitat purposes.

The loss of aquatic plants caused by grass carp can, in some lakes, contribute to algae blooms. Grass carp introduced in 2010 to Lake Heather and Lake Swan, Northland, were very effective at removing pest macrophytes. In Swan Lake they removed most of the *Egeria* and about 40% of the hornwort in 12 months. The next summer Lake Swan developed algae blooms (Gibbs and Hickey 2012).

The loss of aquatic vegetation caused by grass carp may affect the habitat or spawning of other fish and of invertebrate (e.g. zooplankton). This can lead to more predation, changes in fish species composition and changes in the diversity and abundance of zooplankton composition (Rowe 1984, de Winton et al. 2013).

Grass carp are not suitable in waterbodies where they might escape and regulatory approval from DOC and MPI is required before transferring grass carp (Hofstra 2011).

The retrieval and eradication of grass carp following plant eradication can be challenging and potentially costly.

4.10.3 Cost-effectiveness

Grass carp can be a cost-effective way to remove aquatic vegetation, but eradication once vegetation is eradicated can be challenging. Having grass carp in a lake is incompatible with goals to improve water quality and biodiversity values. The loss of vegetation can sustain or worsen poor water quality and algae blooms.

4.10.4 Application to Sullivan Lake

4.10.4.1 Suitability

Grass carp are not very suitable for Sullivan Lake because submerged vegetation is largely absent from the lake, with the exception of water lily and occasion spring growth of *P. crispus*. These macrophytes provide water quality benefits and are relatively easy to manage by other methods.

Stocking grass carp in Sullivan Lake would prevent the use of treatment wetlands to help improve water quality. They would also require additional barriers to stop fish escaping, and this would be inconsistent with aspirations to improve fish passage.

4.10.5 Summary

Grass carp can be an effective way to completely eradicate submerged aquatic vegetation and wetland plants. They are unsuitable for lakes where retaining aquatic plants or wetlands are required for water quality or habitat purposes. They are not well suited for Sullivan Lake because submerged vegetation is largely absent with the exception of water lily and occasion spring growth of *P. crispus*. Stocking grass carp would prevent the use of other actions to improve water quality (e.g. wetlands).

4.11 Silver carp to control phytoplankton

4.11.1 General Description

Silver carp (*Hypophthalmichthys molitrix*) are an introduced planktivorous fish that are bred in New Zealand for control of phytoplankton. They do not breed in small lakes and must be stocked at high density to provide control (de Winton et al. 2013). They have a habit of jumping when disturbed, which can be a hazard in some waterways.

Silver carp are opportunistic filter feeders that will consume phytoplankton, cyanobacteria, zooplankton and detritus. Thus, they may have potential to control cyanobacteria in small eutrophic lakes (Rowe 2010, Ma et al. 2012). However, they selectively graze larger zooplankton (e.g. *Daphnia* sp.) and phytoplankton which can shift the species composition towards smaller species.

Sometimes introduction of silver carp causes more phytoplankton growth. Grazing of silver carp reduces the abundance of zooplankton, which in turn reduces zooplankton grazing of phytoplankton. Often the silver carp grazing of phytoplankton cannot compensate for the reduction in zooplankton grazing, resulting in an increase in phytoplankton biomass and lower clarity (Shen et al. 2021, Zhao et al in de Winton 2013).

4.11.2 General Application and Constraints

Silver carp have been used in the USA to reduce cyanobacteria blooms in reservoirs. However, their success in improving water quality in hyper-eutrophic ponds and lakes is variable. They have been stocked in several small NZ lakes (e.g. Lake Orakai, Lake Omapere), but there was insufficient monitoring to assess their success or effects. There is limited information available in New Zealand to assess benefits and risks of silver carp (Rowe 2010).

Silver carp produce substantial floating faecal matter, this can affect the aesthetics of the water which may affect their application (de Winton 2013).

Silver carp are not suitable in waterbodies where they might escape and regulatory approval from DOC and MPI is required before transferring grass carp (de Winton et al. 2013).

4.11.3 Cost-effectiveness

Costs of establishing silver carp are likely to be similar to grass carp, but removal could be more challenging and costly if they are found to be unsuccessful. They may be effective at controlling cyanobacteria blooms but their success is uncertain and there is a possibility that worse outcomes occur. Any introduction of silver carp should occur with lake monitoring, which may increase its cost.

4.11.4 Application to Sullivan Lake

4.11.4.1 Suitability

The benefit of silver carp as a lake restoration tool is controversial. Silver carp might be effective at controlling cyanobacteria blooms in Sullivan Lake but their success is uncertain, and there is a risk of unintended consequences. If silver carp are unsuccessful, then removing them would be difficult and costly, making the use of silver carp a high-risk technique.

Silver carp would require additional barriers to stop fish escaping, which would further restrict any native fish passage.

4.11.5 Summary

Silver carp are not recommended of Sullivan Lake because of their uncertainty, risk and difficulty to later remove. Although they may be effective at controlling cyanobacteria blooms, their success is uncertain and they may influence the ecosystem in unexpected ways. The need to contain silver carp is incompatible with improving any fish passage to Sullivan Lake.

4.12 Summary: Actions to improve water quality and ecology

Intervention options to improve water quality in Sullivan Lake are summarised in **Table 4.2**. The high priority actions were chosen that would address multiple issues in a cost-effective way, and with low risk of adverse effects.

There is no single quick fix to improving water quality in lakes. There is no “magic bullet”. A danger of seeking a quick fix to a particular water quality issue is that it aggravates other issues; this is because biological systems are interconnected. The path towards sustainable improvement in lake water quality requires reducing both external and internal nutrient loads, and improving the functioning and diversity of aquatic habitat.

4.12.1 Priority interventions

The management interventions recommended as highest priority to improve water quality and ecology in Sullivan Lake are:

- Increase flow augmentation during summer (January to March)
- Treatment wetlands to remove nutrients and improve biodiversity.
- Dredging to remove organic, nutrient rich sediments and partially remove water lily.
- Placing bottom-liners in to contain the spread of water lily following partial removal.
- Harvesting macrophyte to manage plant cover and remove nutrients – early summer when required.

The management interventions to improve water quality that could be considered but are either less cost-effective or require additional investigation are:

- Floating wetlands to remove nutrients and improve biodiversity. A small number of floating wetlands would be beneficial near sediment forebays of the inflows.
- Sediment phosphorus locking to reduce internal load of P (e.g., applying alum).
- Various measures to reduce catchment sediment and nutrient loads.

The management interventions not recommended for Sullivan Lake at this stage are due to practical difficulties or their uncertain or limited benefits are:

- Grass carp to control aquatic plants (due to impact on other restoration actions and difficulty removing).
- Silver carp to control phytoplankton (due to uncertainty of outcome and difficulty removing).

In addition to water quality interventions:

- Fish passage to Sullivan Lake could be improved with instillation of a fish friendly flap gate at the Whakatāne River outlet and a retrofitting a ramp and /or spat rope over the outlet weir.
- The risk of avian botulism can be reduced by collecting and disposing of carcasses during an outbreak.

Table 4.2: Summary of intervention options to address ecological and water quality issues in Sullivan Lake

Priority	Intervention Option	Description	Effectiveness in Sullivan Lake	Limitations
High	Increase flow augmentation	Increase flow augmentation to 1000 m3/day during summer dry periods better flush algae and nutrients.	WQ - Moderate.	Possible flow restrictions on water take during summer
High	Dredge sediments western end	Remove organic sediment and water lily from the western end of Sullivan Lake to control cover and to reduce internal load of nutrients.	WQ - Moderate to high. Weeds - High	Suited to small lakes due to high cost. Risk of poor WQ during operation. Disposal can be costly.
High	Bottom-lining to contain the spread of water lily	Bottom-line to restrict regrowth of water lily following removal.	Weeds - High	Costly at large scale Must first reduced weed biomass. Possible risk to benthic fauna
High	Treatment wetlands and riparian wetlands	Treatment wetlands and sediment traps to remove nutrients and create habitat.	WQ - High Habitat - High	Requires a large area. Moderate to high capital cost. Good design critical to ensure P removal.
High	Harvest curled pondweed	Harvest pondweed in early summer to remove nutrients and reduce plant cover.	WQ - Moderate / High Weeds - High (but short term). Habitat - Moderate	Requires ongoing effort. Access to equipment limited at peak times. Limited value if low density or weed. Not suited for eradication.
Moderate	Floating wetlands	Floating wetlands to remove nutrients and provide habitat.	WQ - High Habitat - High	Costly compared to wetlands. Best suited to near inflows with high nutrient concentrations.
Moderate	Phosphorus locking / flocculation	P-inactivation to reduce internal P load.	WQ - High (but may require repeated application)	Reduced efficacy in shallow lakes with sediment resuspension. pH conditions are critical. Can be culturally sensitive.
Moderate	Measures to reduce catchment sediment and nutrient loads	Reduce external nutrient loads including: - Investigate stormwater ingress to further reduce risk of sewage overflows. - Educate property lowers to reduce nutrient in stormwater. - P-socks at culvert outlets to bind P.	WQ - High (address root causes)	Investigations required to find sources. A social challenge to achieve changes in landuse or land management. May be limited opportunity in Sullivan Lake.
Low; selective use	Herbicide	Herbicide spray to reduce plant cover when required.	Plants - High (and can be selective)	Is effective, cheap and easy. But risk of WQ issues and cultural sensitivity.
Not advised	Silver carp	Silver carp to reduce phytoplankton / cyanobacteria.	WQ - Moderate but uncertain and controversial.	Risk of worsening WQ. Changes ecosystem structure. Fish must be contained. Difficult to remove fish.
Not advised	Grass carp	Grass carp to eradicate aquatic plants.	Plants - High (total eradication) Habitat - negative for Sullivan due to total loss of macrophytes	Total eradication of plants. Not compatible with wetlands. Risk of worsening WQ. Fish must be contained. Difficult to remove fish.

5 Conclusions and Recommendations

5.1 Conclusion

The water quality in Sullivan Lake is poor with low water clarity, high nutrient concentrations and high phytoplankton growth indicative of supertrophic to hypertrophic conditions. The lake is likely to have internal loading of nutrients from the sediment either by sediment suspension and via occasional bottom water/sediment anoxia. There are also internal nitrogen loading via cyanobacteria.

This report has identified priority management interventions that are cost-effective and have a track record of working in small lakes. There is no single quick fix to improving water quality in lakes. Improvement of water quality in Sullivan Lake over the long term will require efforts to reduce nutrient loads (internal and external) and enhance natural processes that attenuate nutrients. Establishing wetlands and aquatic plants are important for maintaining reasonable water quality in small natural lakes, but ongoing management of plant cover may be required in lakes dominated by exotic plants to avoid excessive biomass causing further water quality problems.

5.2 Future monitoring and investigations

Water quality monitoring of Sullivan Lake has been limited in recent years. While we have been able to draw useful information about the current state and issues affecting Sullivan Lake, additional monitoring would provide greater understanding and certainty. Monitoring is also an important part of management remediation options by measuring success in achieving specific outcomes and identifying where different management interventions may need to be implemented. This type of outcome monitoring focuses on specific aspects of the lake ecology or water quality. For example, more intensive monitoring may be required over spring and summer to better understand the dynamics of curled pondweed cover, phytoplankton biomass, nutrients and dissolved oxygen. Similarly, stormwater inflows might be monitored during rain-events to better understand the catchment inputs on the lake.

In the context of limited budgets, a balance needs to be found between monitoring and implementing actions. In our view, initiating actions to improve the lakes water quality should not be delayed by monitoring; monitoring should be used to support and inform action rather than delay action through lack of resources.

General monitoring that would assist in managing Sullivan Lake and understanding the success of any mitigation should include:

- Monitoring water quality of main stormwater inflows to Sullivan Lake during rainfall to characterise the quality and contribution of stormwater entering the lake (including an estimate of flow from the culvert).
- Monitor the development of pondweed cover during spring and early summer to inform potential management actions such as harvesting or increasing the volume of flow augmentation.

- Monitoring water depth of any sediment trap to inform maintenance actions.
- Investigating the potential for using P-locking products. Including sampling of surface sediment (for TP and Al) and overlying water (for DRP and hardness), incubation of sediments to assess P release.
- Investigating the oxygen demand from sediment obtained from different locations in the lake would improve understanding of spatial variation in DO with Sullivan Lake, and better inform locations to focus any dredging.
- Following interventions to remove sediments at the western end of the lake, then repeat synoptic surveys of DO and pH.
- Recording incidences of avian botulism to inform potential management actions of removing and safely disposing of any carcases.
- Water quality monitoring of the lake surface water with a minimum frequency of two monthly and analysing at least the variable of: Temperature, specific EC, DO, %DO, water clarity, pH, TN, TP, Chl-*a*, and *E.coli* bacteria. Field observations of macrophyte cover. More frequent monitoring may be required to assess the effectiveness of some management actions.
- Dissolved oxygen logger during summer to assess DO fluctuations and success in reducing periods of low DO.
- Counts of waterfowl using the lake would allow estimates of the potential contribution of waterfowl to nutrient and bacteria loads.

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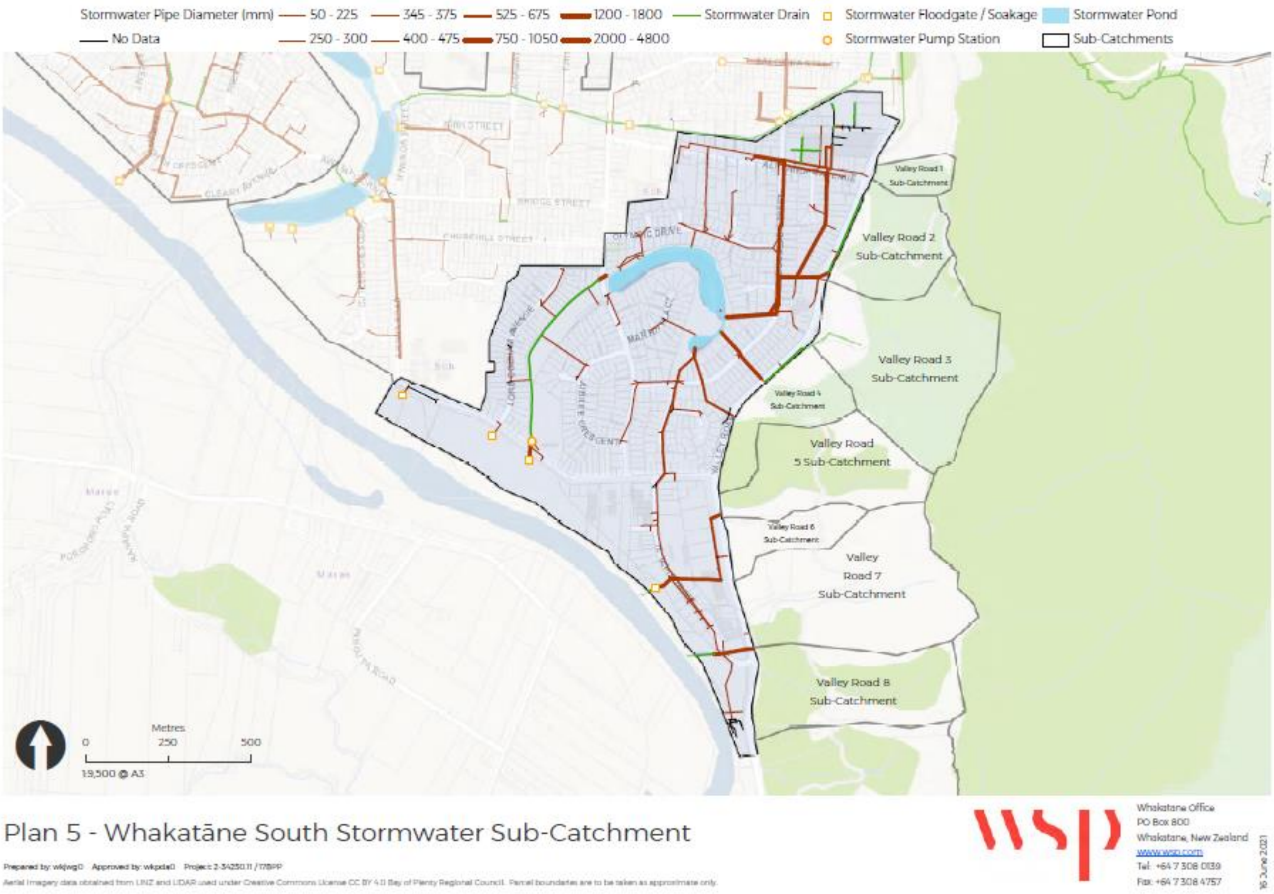
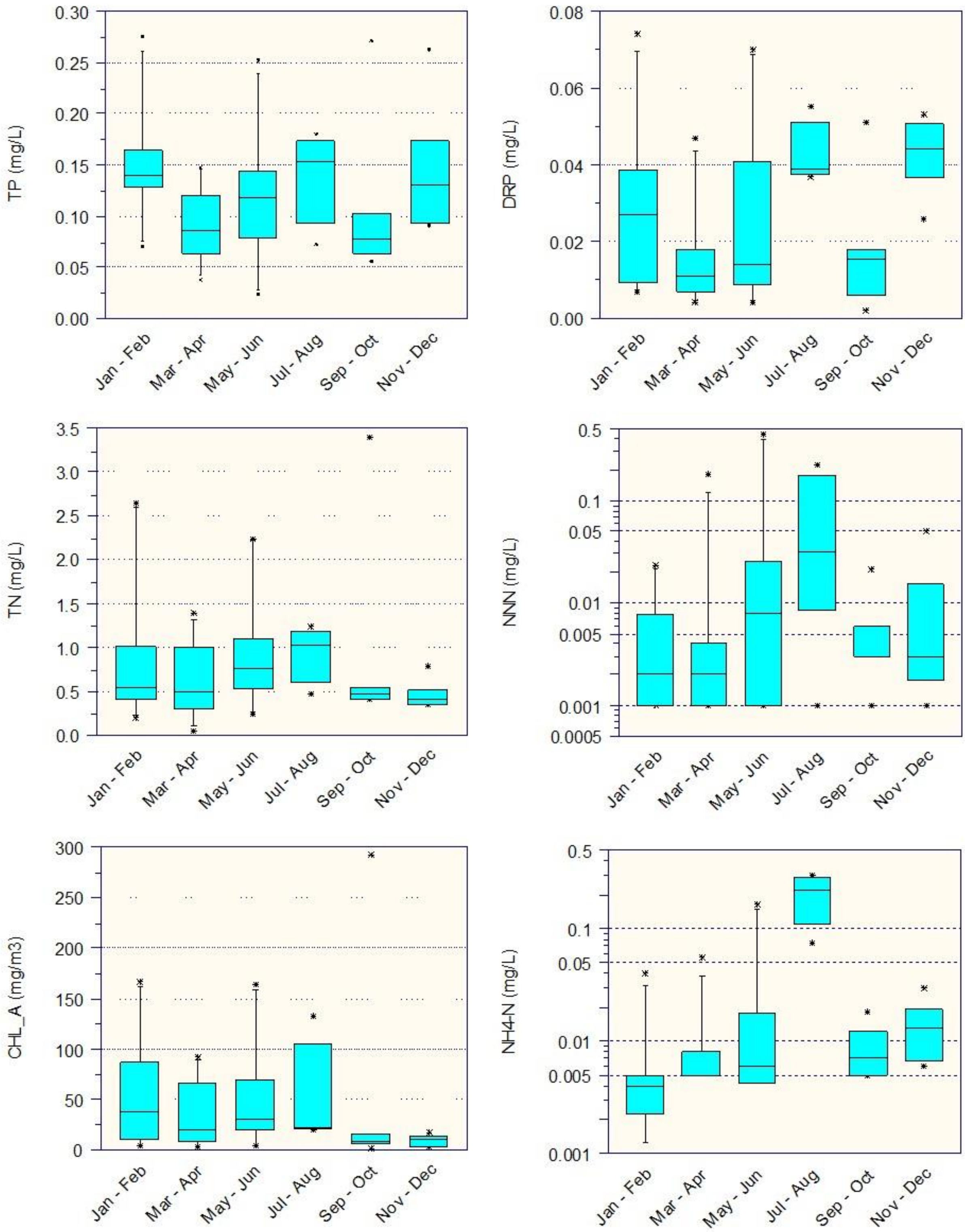
Appendix 1: Sullivan Lake stormwater

Whakatāne South Stormwater catchment connected with Sullivan Lake (WSP 2021)



Appendix 2: Seasonal water quality Sullivan Lake

Seasonal water quality in Sullivan Lake since 1 January 2023. Box plots without 95%ile bars have less than 12 data points.



Appendix 3: Restoration techniques to address eutrophication in shallow lakes.

Restoration techniques to address eutrophication in shallow lakes. Reproduced from Table 1 in Abell et al. (2020).

Group	Restoration method	Purpose	Application	Examples	Advantages	Disadvantages	References
Reduce external nutrient loads	Diffuse and point source control	Minimize nutrient loading	Essential component of a sustainable lake restoration strategy to control eutrophication	• Lake Müggelsee, Germany • Lake Peipsi, Estonia/Russia • Loch Leven, Scotland • City Park Lake, Louisiana, USA	• Addresses the root cause	• Sufficient reductions typically require major economic costs, for example, to fund land-use change or improved wastewater treatment	Ruley and Rusch (2002); Jeppesen et al. (2005)
Reduce internal nutrient loads (physical)	Dredging	Reduce internal loading by removing nutrient-enriched sediments	Best suited to small lakes and/or iconic lakes due to the high costs	• City Park Lake, Louisiana, USA • Lake Kraenepoel, Belgium	• Directly removes nutrients • Increases depth	• Expensive • Disposal of dredgeate can be difficult	Peterson (1979, 1981); Van Wichelen et al. (2007)
	Sediment capping (passive)	Reduce internal load by creating a physical barrier between benthic sediments and the water column	Generally suited to smaller lakes with high internal loads	• Taihu Lake, CN (one embayment)	• Maybe opportunities to use inexpensive local soil/sand	• Adverse effects to benthic biota such as mussels	Xu et al. (2012)
Reduce internal nutrient loads (chemical)	Phosphorus inactivation/ flocculation	Reduce concentrations of dissolved nutrients (primarily P) by adsorption. May be combined with flocculant use to remove organic material	Generally suited to smaller lakes with high internal loads	• Minneapolis Chain of Lakes, USA • Lake Rotorua, New Zealand	• Potentially rapid improvements • Cost-effective (internal) load reductions • Well-established	• Reduced efficacy in shallow lakes due to sediment resuspension • Adding chemicals to waterbodies can be culturally/ socially sensitive • Metal toxicity needs to be considered • Not a sustainable solution alone	Welch et al. (1988); Huser et al. (2016); Smith et al. (2016); Wang and Jiang (2016); Vargas and Qi (2019)
Bio-manipulation	Fish removal (zooplanktivorous)	Increase cladoceran zooplankton biomass → reduce phytoplankton biomass	Applicable to lakes with abundant zooplanktivores, for example, juvenile *Perca fluviatilis*	• Lake Vaeng, Denmark	• Established method in western European lakes with abundant zooplanktivores	• High, ongoing effort required to maintain low biomass • Results are inconsistent • Only suitable for lakes with abundant zooplanktivorous fish	Meijer et al. (1999); Søndergaard et al. (2008)
	Fish removal (benthivorous)	Reduce bioturbation and nutrient excretion	Applicable to lakes with high biomass of benthivorous fish such as *Cyprinus carpio*	• Wolderwijd, The Netherlands • Lake Susan, Minnesota, USA • Lake Ohinewai, New Zealand	• Can support biodiversity objectives if fish are invasive	• High, ongoing effort required to maintain low biomass • Results are inconsistent	Meijer et al. (1999); Søndergaard et al. (2008); Bajer and Sorensen (2015); Tempero et al. (2019)

Promote bivalves	Increase filtration rates and phytoplankton grazing	Untrialled as a deliberate method, although potentially suitable for lakes that are very shallow (relatively low volume) and oligo-mesotrophic (more suitable physicochemical habitat conditions)	◦ Lake Faarup, Denmark (following an undesired invasion by zebra mussels)	◦ Could promote biodiversity if native species are used	◦ Requires suitable host fish for larval development ◦Habitat conditions may be unsuitable in lakes that are the greatest priorities for restoration	Jeppesen et al. (2012); Burns et al. (2014)
Macrophyte harvesting	Remove nutrients present in plant tissues	Very shallow (low volume) lakes with high abundance of invasive macrophytes	◦ Lake Wingra, Wisconsin, USA ◦ Lake Rotoehu, New Zealand	◦ Removing invasive plants can promote native plant biodiversity ◦ Plants could provide a resource (e.g., feedstock), pending research and development	◦ High, ongoing effort required to maintain low biomass ◦ Nutrient removal expected to be minor compared with external loads	Carpenter and Adams (1978); Quilliam et al. (2015)
Floating wetlands	Uptake dissolved nutrients. Potentially also increase denitrification and settling.	Small lakes, embayments, and drains where high coverage is feasible	◦ Lake Rodó, Uruguay	◦ May provide additional habitat values ◦ Can provide a visual focus for lake restoration efforts	◦ Field trials that demonstrate successful application to manage eutrophication are lacking ◦ Not applicable to restore medium-large lakes ◦ Plant harvesting necessary for optimum performance	Rodríguez-Gallego et al. (2004); Pavlineri et al. (2017); Bi et al. (2019)
Algicides	Directly reduce phytoplankton biomass	May be suitable as an emergency measure	◦ Cazenovia Lake, New York, USA	◦ Effective at causing rapid short-term declines in phytoplankton biomass with sufficiently high doses	◦ Toxic effects on other biota ◦ Sediment contamination ◦ Culturally/socially controversial ◦ Not generally recommended as a lake restoration method	Effler et al. (1980); Fan et al. (2013)

(continued)

Table 1. Continued.

Group	Restoration method	Purpose	Application	Examples	Advantages	Disadvantages	References
	Macrophyte reestablishment	Promote reestablishment of macrophytes by planting founder colonies and/or protecting plants with exclosures and wave buffers	Suitable for lakes that have experienced improved clarity but macrophyte reestablishment is hindered by lack of viable seeds/propagules or grazing	◦ Delta Marsh, Manitoba, Canada	◦ Can yield improved macrophyte growth in some areas	◦ Only suitable for lakes that have already been partially restored and have suitable light conditions and substrate	Evelsizer and Turner (2006); Hilt et al. (2018)
Hydrologic alterations	Inflow diversion	Reduce external loads	Applicable to lakes for which external loads are dominated by a single surface inflow, and there is a suitable receiving waterbody nearby	◦ Lake Rotoiti, New Zealand	◦ Step-change reductions in external loads	◦ Potential ecological impacts to receiving waterbody ◦ High capital costs ◦ Feasibility depends on local hydrology and not possible for most lakes	Hamilton and Dada (2016)
	Increase dilution and/or flushing	Dilute poor-quality lake water with higher quality water	Applicable to lakes for which there is a suitable donor waterbody nearby	◦ Moses and Green lakes (USA) ◦ Lake Veluwe (The Netherlands) ◦ West Lake, CN	◦ Major improvement in water quality possible	◦ Potential ecological impacts to donor waterbody ◦ High capital costs ◦ Feasibility depends on local hydrology and not possible for most lakes	Welch (1981); Ibelings et al. (2007); Jin et al. (2015)
	Water-level management	◦ Increasing depth can reduce sediment resuspension ◦ May restore riparian vegetation, depending on the hydrologic regime	Very shallow lakes or lakes where the riparian vegetation communities are impaired due to the existing hydrologic regime	◦ Volkerak–Zoommeer lake system, The Netherlands	◦ Can improve habitat for plants and wildfowl	◦ Can only improve water quality indirectly ◦ Land tenure and surrounding topography can be a constraint to increasing lake level ◦ Not a primary method to reduce trophic status	Gulati and van Donk (2002)

Summary of efficacy and cost of phosphorus mitigation strategies for farms (reproduced from Table 2 of McDowell and Nash 2013).

Strategy	Main targeted P form(s)	Effectiveness (% total P decrease)	Cost, range ($ per kg P conserved)†	Cost, Waikakahi ($ per kg P conserved)†
Management				
Optimum soil test P	dissolved and particulate	5–20	highly cost-effective‡	(15)
Low solubility P fertilizer	dissolved and particulate	0–20	0–20	0
Stream fencing	dissolved and particulate	10–30	2–45	14
Restricted grazing of cropland	particulate	30–50	30–200	na
Greater effluent pond storage/application area	dissolved and particulate	10–30	2–30	13
Flood irrigation management§	dissolved and particulate	40–60	2–200	4
Low rate effluent application to land	dissolved and particulate	10–30	5–35	27
Amendment				
Tile drain amendments	dissolved and particulate	50	20–75	na
Red mud (bauxite residue)	dissolved	20–98	75–150	na
Alum to pasture	dissolved	5–30	110 to >400	na
Alum to grazed cropland	dissolved	30	120–220	na
Edge of field				
Grass buffer strips	dissolved	0–20	20 to >200	30
Sorbents in and near streams	dissolved and particulate	20	275	na
Sediment traps	particulate	10–20	>400	>400
Dams and water recycling	dissolved and particulate	50–95	(200) to 400¶	200
Constructed wetlands	particulate	–426 to 77	100 to >400#	300
Natural seepage wetlands	particulate	<10	100 to >400#	na

† Numbers in parentheses represent net benefit, not cost. Data taken as midpoint for average farm in Monaghan et al. (2009a).

‡ Depends on existing soil test P concentration.

§ Includes adjusting clock timings to decrease outwash <10% of inflow, installation of bunds to prevent outwash, and releveling of old borders.

¶ Upper bound only applicable to retention dams combined with water recycling.

Potential for wetlands to act as a source of P renders upper estimates for cost infinite.