Awatapu Lagoon Water Quality, Ecology and Options for Improvement

Prepared for:

Whakatāne District Council

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Cover Photo: Floating wetlands in Awatapu Lagoon Central, Whakatāne, April 2022.

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Executive Summary

Whakatāne District Council (**WDC**) has responsibility for managing Awatapu Lagoon. In order to inform the management of Awatapu Lagoon, 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.

Awatapu Lagoon condition and values

Morphology

Awatapu Lagoon is a 12.9 ha oxbow lake created when the Whakatāne River was straightened in 1970. The lagoon is divided into three sections. The outlet is in the western lagoon (4.62 ha) which is connected to the Whakatāne River via fish-friendly flap gates, the central lagoon (ca. 5.56 ha), and the southern lagoon (2.75 ha).

The water depth in the lagoon is typically about 1.7m, with the deepest section about 4.3m deep. The lagoon is shallower (about 0.5m) near the confluence of the Wainui Te Whara Stream where stream sediment has been deposited.

Hydrology

Awatapu Lagoon has a total catchment area of 720 ha. The main flow into Awatapu is from the Wainui Te Whara Stream (median flow of 0.061 m^3/s), which enters the central lagoon. The southern lagoon has no baseflow and only a small stormwater catchment. It is connected to the main lagoon via a 2m culvert under Bridge Street. Water flows out of Awatapu Lagoon at its western end via twin 1.15 m diameter culverts under Awatapu Drive, to an arm of the Whakatāne River. These have fish-friendly flap gates (**FFG**) to allow water exchange. Tidal water flows into the lagoon via the FFGs and cause small tidal fluctuations within the lagoon. The tidal range in the Whakatāne River near the Awatapu Lagoon outlet is between about 1m and 1.65m respectively for a neep and spring tide. When the FFGs are operating, the tidal range in western Lagoon west is about 0.05m to 0.08m.

The tidal water entering Awatapu during spring high tides is brackish when river flows are low; this causes the bottom water of the western lagoon and the deepest part of the central lagoon to periodically be brackish during summer and autumn.

The median hydraulic residence time for combined Awatapu Central and Awatapu West is about 28 days. The average residence time for Awatapu South is about 61 days – but will be much longer during summer.

Birds

Awatapu Lagoon provides breeding and feeding habitat for a large number of water birds. Pūkeko, mallard and Australian coot (Nationally uncommon) are the most abundant waterfowl on the lagoon. Threatened species are also present, including: Royal spoonbill (Nationally uncommon), New Zealand dabchick (Nationally Increasing), bittern (kotuku) (Nationally Critical).

The bird life on Awatapu Lagoon has benefited from trapping of predators, and protection provided by floating wetlands.

Fish

Fish access to Awatapu Lagoon and the Wainui Te Whara Stream catchment is facilitated by fish-friendly flap gates on the two outlet culverts. A wide variety of fish have been recorded in Awatapu Lagoon and in the wider Wainui Te Whara catchment, including: shortfin eel (*Anguilla australis*), longfin eel (*Anguilla dieffenbachia*), speckled longfin eel (*Anguilla reinhardtii*), inanga (*Galaxias maculatus*), banded kōkopu (*Galaxias fasciatus*), shortjaw kōkopu (*Galaxias postvectis*), giant kōkopu (*Galaxias argenteus*), common bully (*Gobiomorphus cotidianus*), giant bully (*Gobiomorphus gobioides*), redfin bully (*Gobiomorphus huttoni*), common smelt (*Retropinna retrpinna*), goldfish, brown trout (*Salmo trutta*), and gambusia (*Gambusia affinis*). Many of these fish are classified as threatened. Goldfish and common bully were most abundant in the lagoon, while shortfin eel and longfin eel are most abundant in the Wainui Te Whara Stream.

Plants

A narrow band of raupō (*Typha orientalis*) extends along much of the lagoon margin, interspersed in places with reed sweetgrass (*Glyceria maxima*), and patches of *Machaerina articulata*.

Aquatic vegetation in Awatapu Lagoon is dominated by hornwort (*Ceratophyllum demersum*) and parrot's feather (*Myriophyllum aquaticum*) – both are exotic pest plants. These can be rooted but in Awatapu more commonly occur as floating mats (often c. 0.5m thick) that cover large areas of the lagoon. The southern lagoon has almost 100% coverage of hornwort and parrot's feather, but floating mats of hornwort commonly form large floating mats in the central and western lagoons. The growth of hornwort in the western lagoon can be limited by the brackish water. The floating fern *Azolla filiculoides* can also cover large areas of Awatapu during summer, and is easily recognised by its purple colour.

Aquatic plants are key to maintaining good water quality in natural lakes, by regulating water quality, stabilising sediments, and providing habitat for invertebrates and fish. However, the extensive cover of hornwort and parrot's feather in Awatapu causes major water quality problems with low dissolved oxygen below the thick floating mats of hornwort.

Water quality

Awatapu Lagoon has poor water quality. Its trophic Level Index (**TLI**) score (5.1) is borderline between eutrophic and supertrophic. It has low water clarity (median 0.62m), and high concentrations of total nitrogen (median 0.34 mg/L), total phosphorus (0.069 mg/L), and phytoplankton (median Chl-*a* 10.4 mg/m^3). Algae blooms are common during summer and are often dominated by potentially toxic cyanobacteria.

National bottom-line values set for lake attributes in the National Policy Statement for Freshwater Management (**NPS-FM)** are not met for median total phosphorus (**TP**), median chlorophyll-*a,* maximum chlorophyll-*a*, and cyanobacteria biovolume, or dissolved oxygen in the bottom waters. However, the microbial water quality of Awatapu Lagoon west appears to be in the NPS-FM attribute Band B, which is suitable for Human Contact recreation.

The Western and Southern lagoons have broadly similar water quality, but Awatapu Lagoon South has considerably worse water quality compared to the rest of the lagoon. TP concentrations are considerably higher in the lagoon than in the Wainui Te Whara Stream.

A distinctive feature of Awatapu Lagoon is that the Central Lagoon and Western Lagoon experience both thermal stratification and salinity stratification. These develop with the inflow of brackish water during low river flows, but can persist for months, resulting in anoxic bottom waters and internal nutrient release.

Management interventions to improve water quality are likely to require targeting loads for both nitrogen and phosphorus.

Dissolved oxygen and pH regime

When Awatapu Lagoon West and Lagoon Central are stratified the surface water is usually replete in oxygen, while the bottom water has very low oxygen. Past monitoring found that anoxic bottom waters occur on 89% of occasions in Awatapu Central and 75% of occasions in Awatapu West. During periods with high macrophyte cover, the southern lagoon can have very low dissolved oxygen concentrations. When the stratification breaks down, the bottom water becomes aerated but the mixing of the bottom water reduces the dissolved oxygen (**DO**) in the surface water to about 30% to 70% saturation.

There can be strong spatial variability in surface water DO associated with the distribution of floating mats of hornwort. Thick mats block the air from aerating the surface water and partial decomposition within the mats creates an oxygen demand. In 2021, DO was measured at less than 5% saturation directly below an expanse of thick macrophyte mats floating over deep water. Furthermore, the sinking of hornwort mats to the lake bed and their subsequent decomposition will be a major contribution to accumulated organic matter that drives the rapid decline in DO in bottom water following stratification.

Interventions to improve water quality

There is no single quick fix to improving water quality in lakes, and no “magic bullet”; but there are effective actions that can shift Awatapu Lagoon towards being a healthier ecosystem. Highest priority should be given to actions that would address multiple issues in a cost-effective way, and with low risk of adverse effects.

- Harvesting and control of aquatic pest macrophytes is a priority to improve the DO regime, reduce organic matter load to lake sediments and reduce nutrients.

- Constructing wetlands around the lagoon would provide multiple benefits in removing nutrients, providing habitat for aquatic life and increasing biodiversity values. The most cost-effective place to locate constructed wetlands is near the shallow delta of the Wainui Te Whara Stream. Floating wetlands may be a cost-effective alternative if used in locations where a large amount of earthwork would be required to form a conventional wetland.

- Phosphorus locking of sediments has considerable potential to reduce the internal load of phosphorus from anoxic bottom waters. However, this needs to be undertaken in conjunction with actions to reduce the organic matter load from hornwort.

- There is potential to harness tidal fluctuations to drive a flow of water from the Whakatāne River to Awatapu South via a culvert with flap gates or via a syphon system. This would improve flushing but would likely be insufficient to stop cyanobacteria blooms in the absence of other actions. It would do little to reduce excessive macrophyte growth and associated problems.

- There is potential to reduce catchment sediment and nutrient loads to Awatapu Lagoon. One option for further investigation is to use detainment bunds within the upper Wainui Te Whara catchment.

Contents

1 Introduction

1.1 Background

Whakatāne District Council (**WDC**) has responsibility for managing Awatapu Lagoon. In order to inform the management of Awatapu Lagoon, 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.

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

a. Describe the geographical context of Awatapu Lagoon (including hydrology, and morphology).

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

c. Identify the key issues for Awatapu Lagoon 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 Awatapu Lagoon, so as to inform prioritisation of actions to improve water quality and ecological values.

1.2 Location and Context

Awatapu Lagoon is a 12.9 ha oxbow lake created when the Whakatāne River was straightened in 1970. The lagoon is divided into three sections. The outlet is in the western lagoon (4.62 ha) which is connected to the Whakatāne River via fish-friendly flap gates; the central lagoon (ca. 5.56 ha) between Bridge Street and the foot bridge; and the southern lagoon (2.75 ha) which was the original entrance from the Whakatāne River (**Figure 1.1**). The southern lagoon is connected to the central lagoon via a 2m diameter culvert under Bridge Street.

The Lagoon has a total catchment area of 720 ha, predominantly from the Wainui Te Whara Stream which enters the central lagoon. Tidal water from the Whakatāne River also flows into the lagoon via the fish-friendly flap gates. The tidal water entering Awatapu during high tides is brackish when river flows are low; this causes the bottom water of the western lagoon and the deepest part of the central lagoon to be periodically brackish.

The water depth in the lagoon is typically about 1.7m with the deepest areas of 4.3m. There has been considerable sedimentation at the confluence of the Wainui Te Whara Stream, reducing the water depth to less than 0.5m in places. In this area WDC has formed a delta and settling area to manage sediment inputs.

Awatapu Lagoon is an integral part of the stormwater management network by providing live storage to attenuate peak flows during heavy rain events.

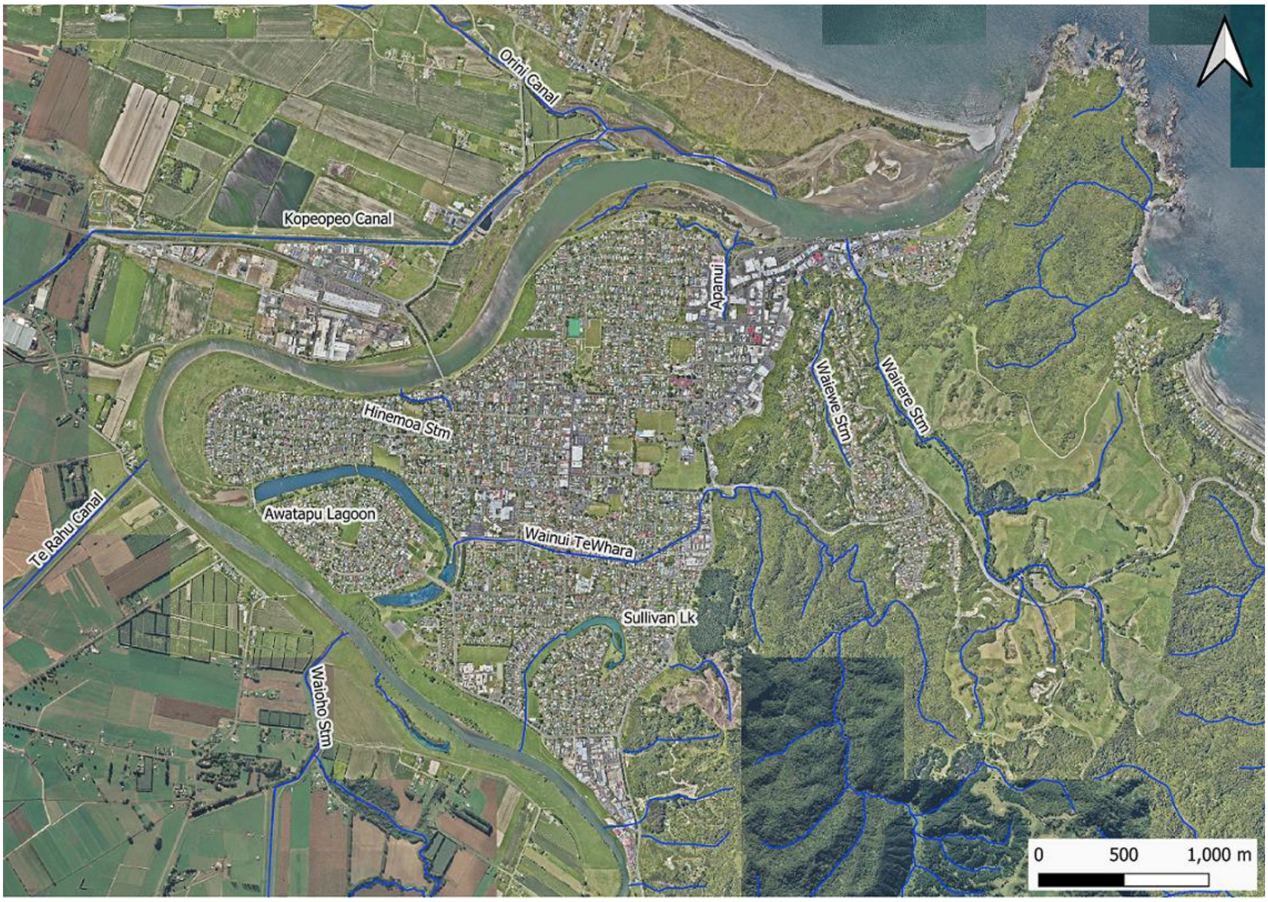


Figure 1.1: Location of Awatapu Lagoon and stream networks in Whakatāne township.

1.2.1 Historical context

Awatapu Lagoon was originally a meander of the Whakatāne River, but was formed into an oxbow lake when the Whakatāne River was straightened in 1970 as part of flood protection works. Around the same time, the Wainui Te Whara Stream was diverted to flow directly into Awatapu Lagoon instead of following its original channel that entered the Whakatāne River upstream of Landing Road Bridge via what is now known as Hinemoa drain. Historical aerial images show that sediment from the Wainui Te Whara Stream quickly formed a delta in the new oxbow lake (**Figure 1.2** and **Figure 1.3**).

The lagoon and associated parkland is surrounded by a secondary stop-bank system, designed to contain flood waters from the Wainui Te Whara Stream and direct urban stormwater.

1.2.2 Management Plans

Awatapu Lagoon Management Plan 1990

The Awatapu Lagoon Management Plan (WDC 1990) identifies values and uses of Awatapu Lagoon and set pragmatic management objectives including:

> "*1. The primary function of the reserve shall be to preserve and manage the natural qualities of the reserve in perpetuity and to provide for the usage and enjoyment of the public for recreation,*

and to ensure that this management does not compromise the area's integrity as a flood control area."

The Awatapu Lagoon Management Plan recognised the value of the lagoon for flood control, recreation and wildlife. It also identified some key issues for the lagoon including sedimentation, nutrient enrichment, excessive growth of pest aquatic plants and rubbish in the lagoon. WDC has implemented one of the policies to address sedimentation by forming a delta at the inlet of the Wainui Te Whara Stream to reduce sediment deposition in the main body of Awatapu Lagoon.

South Awatapu Working Group

During 2018 and 2019 WDC convened Awatapu Working Group to consider remediation work on the steep bank below Cleary Ave on the southern lagoon; however, concern about the state of Awatapu Lagoon led to the scope expanding to also consider options for enhancing Awatapu Southern Lagoon. The Working Group identified the following preferred options:

- Creating wetlands in the lagoon. The preferred for the southern lagoon was having up to 30% wetland cover and ensuring the retention of open water.
- Provide for the flow of water through the southern lagoon, by either a connection to the Whakatāne River, or via pumping.
- Harvesting and removal of aquatic macrophytes. This was undertaken on the southern lagoon once in late summer 2019.
- Replacing the existing flap gates with light weight flap gates to increase tidal inflows. This was implemented in 2022.
- Excavate under foot bridge of Awatapu Lagoon to improve water circulation.

Awatapu Ōtamakaokao Community Plan (2022)

More recently, the Ōtamakaokao Kaitiaki Trust led the development of the Awatapu Ōtamakaokao Community Plan (2022). The Community Plan seeks to express vision and aspirations of the community based around four pou: Pou Taiao (environment), Pou Tikanga (culture), Pou Tangata (people), and Pou Tuahu (economy). The aspirations for the Awatapu Lagoon and surrounding Ōtamakaokao area under Pou Taiao are:

- The lagoon is connected to Ōhinemataroa (Whakatāne River).
- The awa is clear and safe to swim.
- Biodiversity is present and abundant.
- The awa is a food source.
- Native plants flourish.
- Rubbish is absent.

Recent work in the Ōtamakaokao area to support these aspirations for the lagoon include:

- Riparian planting along the edge of Awatapu Lagoon near the Wainui Te Whara delta by Ōtamakaokao Kaitiaki Trust and Halo with the support of WDC and Bay of Plenty Regional Council (**BOPRC**) (2019-2021).
- Pest animal control by Halo.
- Installation of numerous floating wetlands by Whakatāne Intermediate School and by Ōtamakaokao Kaitiaki Trust.
- Rubbish collection by Whakatāne Intermediate School and other groups.
- Installation of rubbish bins by WDC.
- Harvesting and removal of aquatic macrophytes on the southern lagoon in late summer 2019.
- Replacing the existing flap gates with light weight flap gates to increase tidal inflows in 2022.

Figure 1.2: Aerial photos of Awatapu Lagoon area in October 1962 before diversion of the Whakatāne River and of the Wainui Te Whara Stream (top); and in April 1974 after the diversions and creation of the Awatapu oxbow (bottom) (Source: Retrolens).

Figure 1.3: Aerial photos of Awatapu Lagoon in December 1982 showing the development of a large delta at the entrance of the Wainui Te Whara Stream (Source: Retrolens).

2 Methods of investigation

The descriptions of Awatapu Lagoon 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 Awatapu Lagoon to inform our understanding of the waterbody and key mitigation options, which included:

- Water quality samples of Awatapu Lagoon surface water during the summer of 2022/23.
- Dissolved oxygen, pH and temperature spatial surveys to characterise spatial variability.
- Dissolved oxygen and temperature loggers to characterise diurnal variability.
- Tidal water level for estimating potential tidal river inflows.
- Fish and waterfowl presence using eDNA in Awatapu Lagoon and Wainui Te Whara Stream.

Sites in Awatapu Lagoon used for regular water quality sampling and for logging of dissolved oxygen, salinity and water level are shown in **Figure 2.1**.

Figure 2.1: Location of water level and water quality monitoring sites in Awatapu Lagoon (aerial image: Open Commons, LINZ 2022).

2.1 Water quality sampling

BOPRC monitored water quality of Awatapu Lagoon at the Awatapu Drive site monthly from January 2015 to December 2020 (n=59). Samples were analysed for: temperature (**Temp**.), pH, dissolved oxygen (**DO**), specific electrical conductivity (**EC**), total nitrogen (**TN**), nitrate-nitrite-nitrogen (**NNN**), total ammoniacal nitrogen (**NH4-N**), total phosphorus (**TP**), dissolved reactive phosphorus (**DRP**), turbidity (**TURB**), total suspended solids (**TSS**), *E.coli* bacteria (***E.coli***) and visual clarity using a clarity tube. In addition, chlorophyll-*a* (**Chl-*a***) was measured from March 2019 to December 2020.

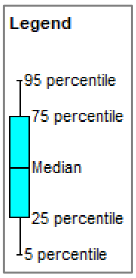
From February 2021 to April 2023 River Lake Ltd, with sample analysis support of BOPRC, collected summer / autumn samples from the following sites: Awatapu Lagoon South at Bridge Street (n=8), Awatapu Lagoon West at Foot Bridge (n=5), Awatapu Lagoon West mid-lake (top and bottom) (n=10), Awatapu Lagoon Central mid-lake (top and bottom) (n=10), Wainui Te Whara at Hinemoa Street (n=12) and Wanui Te Whara at Valley Road (n=7) (**Figure 2.1**). These samples were analysed for the same suite of variables as previously measured by BOPRC.

The mid-lake samples from Awatapu Lagoon West and Awatapu Lagoon Central included samples from both the top water (0.2m depth) and the bottom water hypolimnion/halocline (c. 0.6m above lake bottom, 2.2 to 2.8m depth). Depth profiles of temperature, DO, and sp. EC were measured at the same time as these samples were collected. In addition, depth profiles had been measured during summer/autumn by River Lake Ltd prior to 2021; in total 19 depth profiles were measured from Awatapu Central and Awatapu West between February 2017 and April 2023. Field measurements were made using a YSI Pro Plus multi-meter.

BOPRC also collected cyanobacteria samples from Awatapu Lagoon west during summer between February 2015 and February 2021. The frequency of sampling ranged from weekly to monthly, with a total of 30 samples collected over this time. Samples were analysed for species identification, biovolume and potentially toxic biovolume.

Water clarity was typically measured using a clarity tube. This data was converted to black disc water clarity using the formula provided in Kilroy and Biggs (2002)[1]. For the purpose of estimating the TLI, black disc water clarity was converted to TLI by multiplying by 1.2, following the approach in the Ministry for the Environment (**MfE**) water quality guidelines (1994).

The data analysis used an aggregated dataset for Awatapu Lagoon West consisting of "top" water samples from sites: Awatapu Lagoon at the Riverside Drive, Awatapu Lagoon West mid-lake and Awatapu Lagoon West at Foot Bridge.



Water quality data was expressed using box plots showing the median, interquartile range, 5th-percentile, 95th-percentile, minimum, and maximum, as illustrated here.

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

2.2 Temporal variability of lake dissolved oxygen and salinity

The temporal variability of dissolved oxygen and temperature was characterised using dataloggers. In Awatapu Central, a Hobo U26 optical DO logger was installed from 23/2/2017 to 11/3/2017. The logger was suspended from a buoy at the mid-lake central lagoon site, with the sensor at a depth of 1.3m for the first four days, after which the sensor was raised to 0.5m depth.

In Awatapu West, DO and electrical conductivity loggers were installed on a buoy at the mid-lake west lagoon site from 9/4/2022 to 25/6/2022 (although an increase in water level and movement of macrophyte mats prevented the loggers from being retrieved from this site until November 2022). DO and EC measurements were made at two fixed depths of 0.25m below the water surface (at time of installation) and 0.6m above the lake bottom using Hobo U26 optical DO loggers and Hobo U24 EC loggers.

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 and bottom of the water column when the loggers were installed and removed.

2.3 Spatial variability of dissolved oxygen and pH

Synoptic surveys were undertaken in Awatapu Lagoon to characterise the spatial variability of dissolved oxygen (**DO**), pH and temperature. The surveys took place on 11 April 2022 in the late afternoon (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).

2.4 Tidal water level for estimating potential tidal river inflows

The relative water level, tidal dynamics and salinity influence from the Whakatāne River were investigated by installing loggers in the Whakatāne River and in Awatapu Lagoon. The loggers were installed for eight weeks during December 2022 to February 2023, and for c. six weeks during April and May 2023 (**Table 2.1**).

Floods were frequent during this period (e.g. 17 December 2022, 22 December 2022, 29 January 2023, 4 February 2023, 4 May 2023, 11 May 2023), but the logging also included baseflow conditions. On 26 January 2023 the daily mean flow in the Whakatāne River was 26.7 m^3/s (about 75% of median flow), and the Wainui Te Whara Stream had a median flow of 0.066 m^3/s (about median flow).

Logging during April and May 2023 was a period of receding river flow; at the end of the monitoring period. On 30 April 2023, the flow in the Whakatāne River and the Wainui Te Whara Stream were respectively 45 m^3/s and 0.065 m^3/s (just over median flow); and on 27 May 2023 the flow in the Whakatāne River and the Wainui Te Whara Stream were respectively 83 m^3/s and 0.13 m^3/s – which is about twice the median flows for these rivers.

All water level loggers were surveyed into a reference point by WSP, and the level data was expressed as metres RL to Moturiki datum. Spot measurements of water level were undertaken at the same time as the datum survey.

Table 2.1: Deployment of water level loggers, location, period and variables measured (water level, electrical conductivity, temperature).

Site	Latitude, Longitude (NZTM)	Logger variable	Period deployed
Awatapu West Pump Station	1948019, 5790694	Water level	Permanent
Whakatāne Rv at Awatapu Outlet	1947977, 5790697	Water level, EC, temperature	11/12/2022 - 13/2/2023
		Water level, temperature	15/4/2023 - 27/5/2023
Whakatāne Rv at Awatapu "Inlet"	1948589, 5789986	Water level, temperature	15/4/2023 - 27/5/2023
Awatapu at Bridge St Culvert	1949077, 5790185	Water level, temperature	1/5/2023 - 27/5/2023

2.5 eDNA

Waterways contain environmental DNA (eDNA) of organisms present. Analysis of eDNA shed from organisms in the water gives 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 indicate the strength of the eDNA signal.

Samples of eDNA were collected to supplement existing information on the presence of fish and birds in Awatapu Lagoon and the Wainui Te Whara Stream. Following collection, the samples were preserved and sent to Wilderlab for processing. The eDNA sample dates and sites were:

- 17 November 2022 from Awatapu Lagoon west, lagoon central, lagoon south, and Wainui Te Whara at Hinemoa Street.
- 30 September 2022 from Wainui Te Whara at Hinemoa Street and Wainui Te Whara at Valley Road.
- 28 October 2021 – a composite sample from Awatapu Lagoon central collected by Halo.

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 of more nutrients, more algal productivity and reduced water clarity (**Table 2.2**). Regular monitoring over multiple years is usually required to reliably characterise a lake's water quality or TLI.

TLI variables have not been consistently sampled. TLI estimates prior to March 2019 are calculated using only TN and TP (TLI 2). Secchi disc depth has not been directly samples in Awatapu; instead water clarity was measured using clarity tube. Results were adjusted to be expressed as black disc and these black disc was multiplied by 1.25 to provide an estimate of Secchi depth.

Table 2.2: 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) sets an alert level framework for assessing the health risk from planktonic cyanobacteria. The “Action (Red) mode” is triggered when either: 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 decisions 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 2.3**).

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, Awatapu Lagoon may not fall within the scope of the NPS-FM because it is an artificially created waterbody; nevertheless the attribute bands provide a context for assessing water quality state.

Table 2.3: 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 Awatapu Lagoon

3.1 Morphology

Awatapu Lagoon is a 12.9 ha oxbow lake. The lagoon is divided into three sections. The outlet is in the western lagoon (4.62 ha) which is connected to the Whakatāne River via fish-friendly flap gate, the central lagoon (ca. 5.56 ha), and the southern lagoon (2.75 ha).

Comprehensive bathymetry data is not available for Awatapu Lagoon, but spot measurements over about 20 transects across the lagoon estimate an average depth of 1.8m, 1.5m and 1.5m respectively in the western, central and southern sections. The maximum depth is about 4.3m, located in the north-western end of the central lagoon – an area that would have originally been the outside bend of the river. The lagoon is shallowest (<0.5m) near the confluence of the Wainui Te Whara Stream where stream sediment has deposited.

The volume of Awatapu Lagoon, under baseflow conditions, is approximately 209,200 m^3, consisting of 84,000 m^3, 82,800 m^3 and 42,400 m^3 respectively for Awatapu west, Awatapu central and Awatapu south.

3.1.1 Sediment depth

Most of Awatapu Lagoon has substrate organic muds over the original river gravels. The Wainui Te Whara has deposited sandy substrate near the delta area, which is overlayed with organic muds. Measurements of soft sediment depth in Awatapu south (April 2022) found a median depth of 0.2m and a range of 0.03m to 0.4m. Decomposition of macrophytes is likely a major contributor to organic mud substrate in the lagoon.

3.2 Hydrology

3.2.1 Inflows

The main flow into Awatapu is from the Wainui Te Whara Stream, which enters the central lagoon and flows west towards the outlet. The southern lagoon has no baseflow and only a small stormwater catchment (ca. 43 ha); it is connected to the main lagoon via a 2m culvert under Bridge Street.

Awatapu Lagoon has a total catchment area of 720 ha. About 83% (598ha) of the catchment contributes to the Wainui Te Whara Stream upstream of Valley Road; *c*. 44ha contributes to the lower Wainui Te Whara Stream below Valley Road, *c*. 40 ha of urban catchment contributes directly to the southern lagoon, and *c*. 37.8 ha of urban catchment contributes directly to the western and central lagoons (**Figure 3.2**).

The upper catchment of the Wainui Te Whara (above Valley Road) consists of steep hillside predominantly covered by exotic forest, native forest (64%) and farmland (35%). It cascades steeply down Mokorua gorge and at the base of the hill (downstream of Valley Road) the gradient flattens and the stream flows through urban area.

The median flow from the Wainui Te Whara Stream at Valley Road gauging station is 0.061 m^3/s; the exceedance flow of 25% and 10% are respectively 0.113 m^3/s and 0.204 m^3/s (BOPRC flow duration calculations, period 2006-2023). Accounting for the additional catchment area below Valley Road, the median flow of the Wainui Te Whara at the entrance to Awatapu Lagoon would be about 0.071 m^3/s. For the purpose of comparison, the Whakatāne River - which formed the Awatapu Lagoon basin - has a median flow of 36.2 m^3/s and mean flow of 57.3 m^3/s (EBOP 2007).

The southern section of Awatapu Lagoon only receives stormwater inflows during rain events. The average catchment inflow to Awatapu lagoon south is estimated[2] to be 0.008 m^3/s.

Water flows out of Awatapu Lagoon at its western end via twin culverts under Awatapu Drive (each 1.15 m diameter), to an arm of the Whakatāne River. During flood events the water is pumped. Fish-friendly flap gates (**FFG**) are installed on the outlet culverts which allow the inflow of water from the Whakatāne River during high tides. These gates close near the upper end of the tidal range and prevent floodwater entering during floods.

Tidal water from the Whakatāne River flows into the lagoon via the FFGs. The tidal water entering Awatapu during high tides is brackish when river flows are low; this causes the bottom water of the western lagoon and the deepest part of the central lagoon to be periodically brackish.

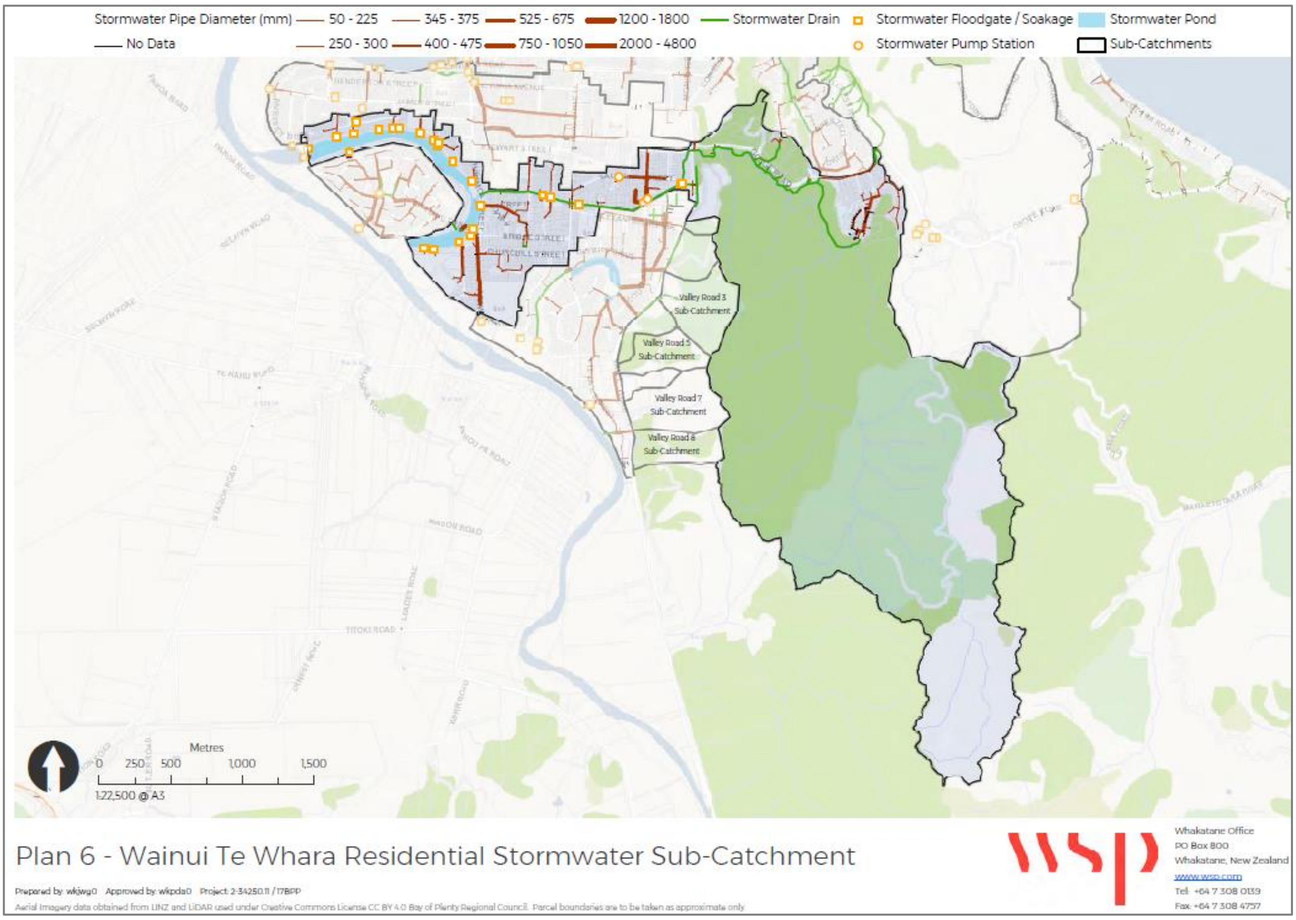


Figure 3.2: Wainui Te Whara Stormwater catchment including Awatapu Lagoon (WSP 2021).

[2] This was derived by multiplying the specific mean flow for the catchment (0.019 $m^3/s/km^2$) by the catchment area (0.43 km^2). Data modelled by NIWA River Environment Classification (REC).

3.2.2 Tidal water level fluctuations and salinity

The variation in water level tidal range in the Awatapu lagoon and the Whakatāne River was investigated to inform an analysis of how effective a new opening to the Whakatāne River from Awatapu south might be in providing additional flow through Awatapu Lagoon. The water level results are presented in **Figures 3.3** to **Figure 3.6**.

Key features of the monitoring are:

- During baseflow conditions, the Whakatāne River near Awatapu inlet has a spring tide range of about 1.65m (-0.45m to 1.2m RL) and a neep tide range of about 1.03m (-0.19m to 0.84 m RL) (**Figure 3.3**).
- The minimum water level in the river embayment downstream of the Awatapu outlet culvert appears to be constrained by a shelf between the embayment and the main river channel. This was seen in the water level at low tide not dropping below about -0.19m RL during spring tides in April 2023 (when river flows were 45 m^3/s) (**Figure 3.4**), and not dropping below -0.25m RL during spring tides on 27 January 2023 (when river flows were 26.7 m^3/s) (**Figure 3.5**).
- During January 2023 spring tides, Awatapu Lagoon at the outlet culvert had a tidal range of 0.039m (0.234m to 0.274m RL) with about a 2-hour delay for outgoing and a 3-hour delay for incoming tides. During May 2023 spring tides, the tidal range was 0.05 to 0.08m (0.330m to 0.409m RL) (**Figure 3.5**).
- The water level[3] in Awatapu at Bridge Street culvert was about 210mm higher than Awatapu lagoon at the outlet culvert and 125mm higher than the median water level in the Whakatāne River at the outlet. This water height differential will be less at lower flows. The tidal range in the lagoon is smaller (0.04m) in at the Bridge Street culvert than near the outlet (**Figure 3.6**).
- Brackish water (e.g. spec. EC >1000 uS/cm) only reached Awatapu Lagoon outlet during a spring high tide (24th to 27th Jan 2023) and when the river flow had dropped to 26.7 m^3/s (three quarters of median flow) (**Figure 3.5**).

Figure 3.7 illustrates the measured culvert invert heights and water level tidal ranges when river flows are about median. At lower river flows, the river and lagoon water levels are lower, the tidal ranges higher and the water level gradient on the lagoon between Bridge Street and the outlet culvert is expected to be less.

[3] Measured during late May 2023 when the Wainui Te Whara flow was about 0.13 m^3/s.

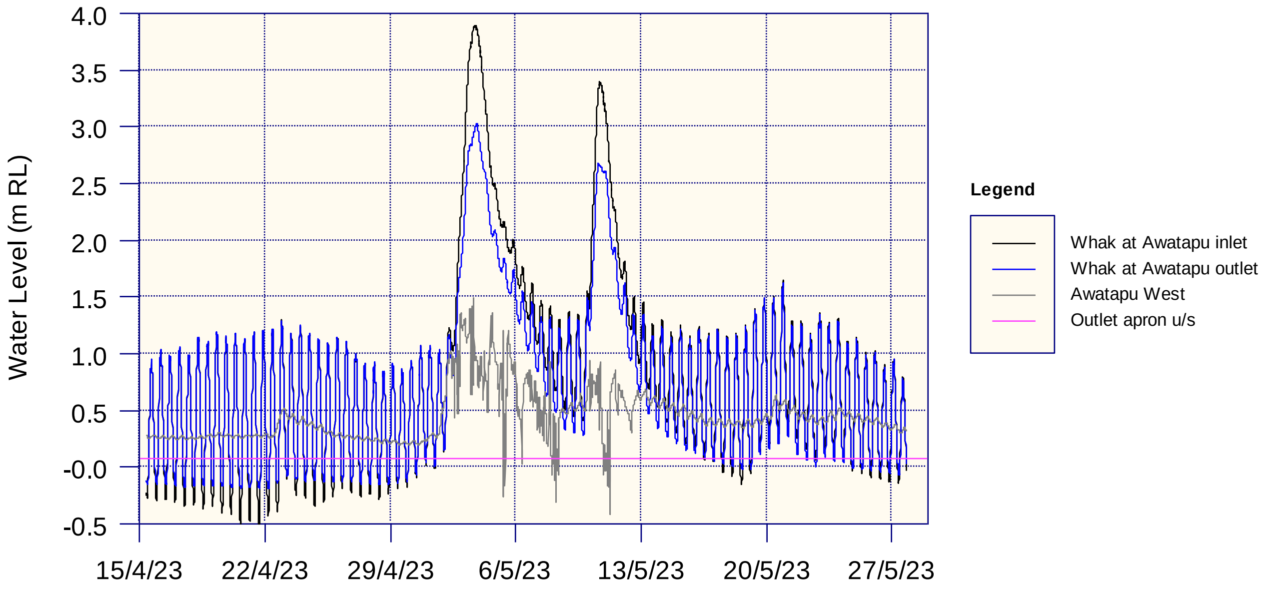


Figure 3.3: Water level in Awatapu Lagoon outlet culvert (west), in the Whakatāne River downstream of the outlet, and in the Whakatāne River 1.2km upstream at the original "inlet".

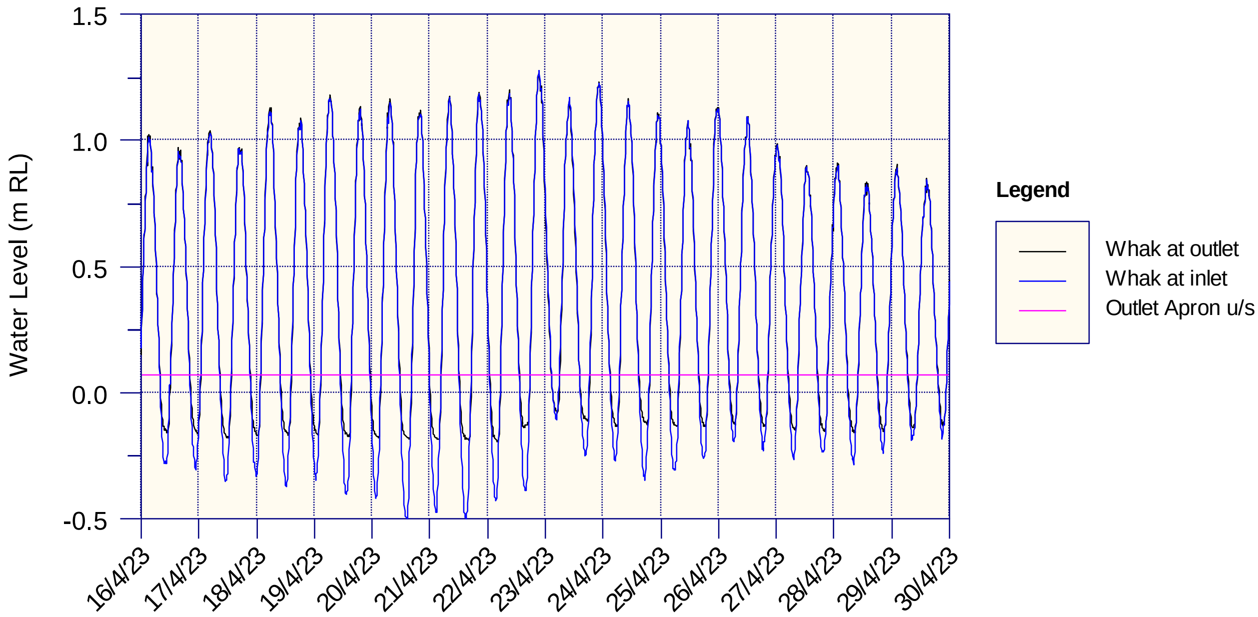


Figure 3.4: Water level in the Whakatāne River downstream of the outlet and 1.2km upstream at the original "inlet". At spring low tides, the minimum water level in the river embayment downstream of the Awatapu outlet culvert appears to be constrained by a shelf between the embayment and the main river channel.

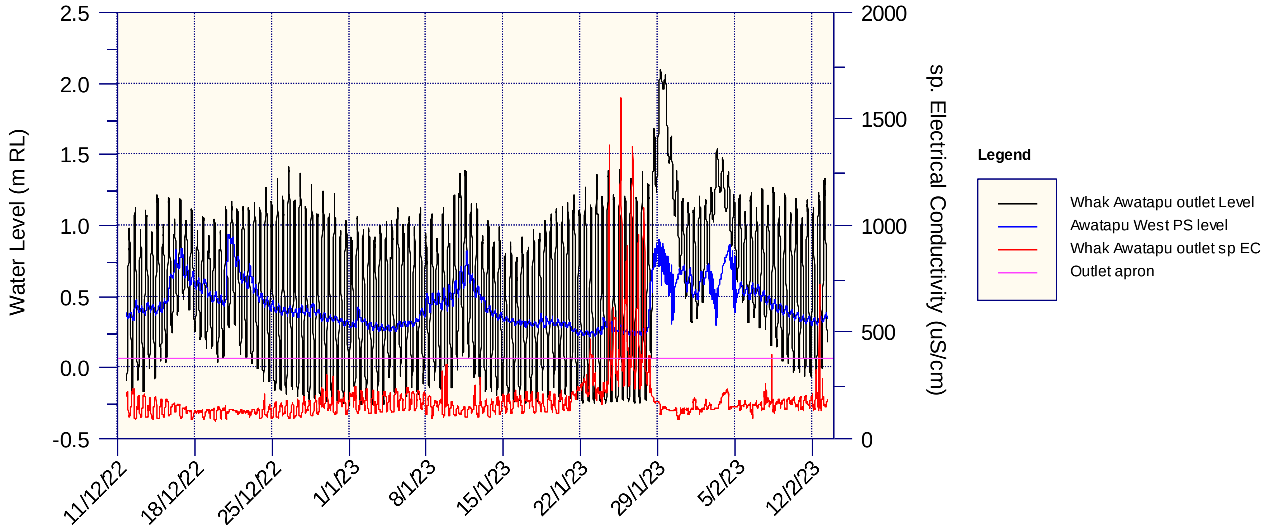


Figure 3.5: Water level and specific electrical conductivity (EC) in the Whakatāne River downstream of the outlet, overlaid with water level at Awatapu Lagoon outlet culvert (west). Higher salinity water only reached Awatapu lagoon during spring high tides and low flows.

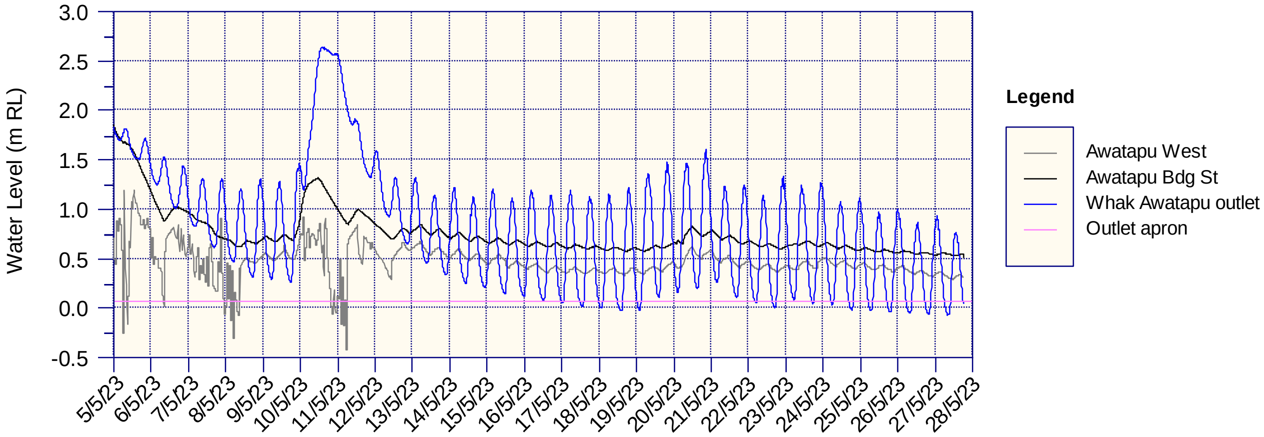


Figure 3.6: Water level in Awatapu Lagoon at Bridge Street culvert, outlet culvert (west) and in the Whakatāne River downstream of the outlet.

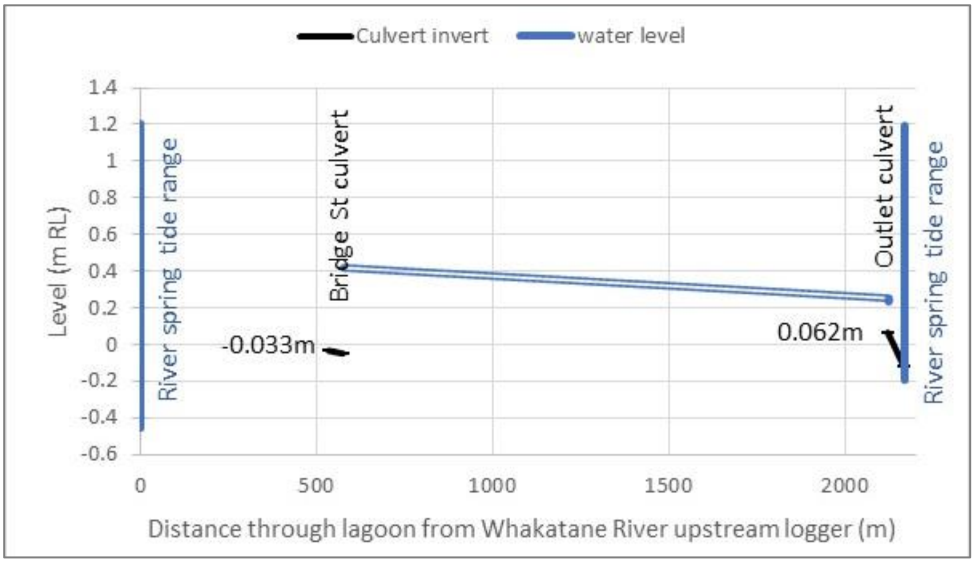


Figure 3.7: Schematic of culvert invert heights and water level tidal ranges when flows are about median. At spring low tides, the minimum water level in the river embayment downstream of the Awatapu outlet culvert appears to be constrained by a shelf between the embayment and the main river channel.

3.2.3 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 median residence time for combined Awatapu Central and West Lagoon is about[4] 28 days. The average residence time for Awatapu South is about 61 days. The residence time will be much longer during summer low flows.

To completely replace the volume of water in Awatapu Central and West would require a flow in the Wainui Te Whara Stream of about 1.93 m^3/s in a single day, or an average of 0.0965 m^3/s over 20 days. A flow of 0.096 m^3/s has an exceedance duration of 31%, and is common during winter and spring. To fully replace the water in Awatapu Lagoon South would require an inflow of about 0.491 m^3/s in a single day, or an average of 0.024 m^3/s over 20 days [5].

One management option to improve water quality in Awatapu may be to increase the volume of water flowing into it. Options to achieve this are discussed in the subsequent management sections of this report and include: directing flow from the Wainui Te Whara Stream to enter Awatapu South before flowing through the rest of the lagoon, and/or taking water from the Whakatāne River to the Awatapu lagoon south (via a new culvert, or enhanced syphon, or pump system over the stop bank).

[4] Calculated as: volume of Awatapu west and central (166,800 m^3) / (catchment inflow (0.07 m^3/s) x 86,400 s/day)
[5] Equivalent to a flow exceedance duration in the Wainui Te Whara Stream of 84%

3.3 Birds

Awatapu Lagoon provides breeding and feeding habitat for a large number of water birds including: Australian coot (*Fulica atra*), pūkeko (*Porphyrio melanotus*), mallard ducks (*Anas platyrhynchos*), paradise shelduck (*Tadorna variegata*), Grey duck hybrids, Australian shoveler (*Spatula rhynchotis*), NZ kingfisher/ kōtare (*Todiramphus sanctus*), Black shag (*Phalacrocorax carbo*), black swan (*Cygnus atratus*), Royal spoonbill / kōtuku (*Platalea regia*), silver gull (*Chroicocephalus novaehollandiae*), New Zealand dabchick (weweia, *Poliocephalus rufopectus*), and occasional bittern (kotuku), and whitefaced heron (*Egretta alba)*. The presence of dabchick is notable because they are rare (about 2000 individuals in NZ[6], with a Conservation Status of “recovering”) and they have successfully bred and raised young on Awatapu Lagoon in recent years. Pūkeko, mallard and Australian coot are the most abundant waterfowl on the lagoon. Threatened species are present include: Australasian bittern (kotuku) (Nationally Critical), Australian coot (Nationally uncommon), Royal spoonbill (Nationally uncommon), New Zealand dabchick (Nationally Increasing) (Robertson et al. 2021).

Bird life on Awatapu Lagoon has likely benefited from intensive trapping of rats and stoats that has been undertaken around the lagoon by conservation groups such as Halo. The floating wetlands on Awatapu provide protected nesting habitats for many of the birds.

3.3.1 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 recorded in Awatapu Lagoon include: shortfin eel (*Anguilla australis*), longfin eel (*Anguilla dieffenbachia*), speckled longfin eel (*Anguilla reinhardtii*), inanga (*Galaxias maculatus*), common bully (*Gobiomorphus cotidianus*), giant bully (*Gobiomorphus gobioides*), redfin bully (*Gobiomorphus huttoni*), common smelt (*Retropinna retrpinna*), goldfish, brown trout (*Salmo trutta*), and gambusia (*Gambusia affinis*) (Hicks et al. 2015, eDNA records in **Table 3.1**).

Fish recorded in the Wainui Te Whara Stream include: Longfin eel (*Anguilla diefffenbachii*), shortfin eel (*Anguilla australis*), redfin bully (*Gobiomorphus huttoni*), giant bully (*Gobiomorphus gobioides*), banded kōkopu (*Galaxias fasciatus*), shortjaw kōkopu (*Galaxias postvectis*), giant kōkopu (*Galaxias argenteus*), common smelt (*Retropinna retrpinna*), and brown trout (*Salmo trutta*). Many of these are diadromous fish that need to migrate to the sea (via Awatapu Lagoon) as part of their life-cycle (Hamill 2015, **Table 3.1**).

Hick et al. (2015) fished Awatapu using an electric fishing boat in 2005 and 2014. They caught goldfish, shortfin eel, common bully, giant bully, inanga, brown trout, smelt and gambusia. Goldfish and

[6] http://nzbirdsonline.org.nz/species/new-zealand-dabchick

common bully were most abundant. The fishing confirmed that the lagoon had large goldfish but no koi carp. Sampling of the Wainui Te Whara by Hamill (2015) found shortfin and longfin eel to be most abundant – with a large abundance as elva.

Many of the fish species present in Awatapu and the Wainui Te Whara Stream have a threat classification, i.e. longfin eel, giant kōkopu, inanga are classed "At-Risk – Declining", shortjaw kōkopu is classed "Nationally Vulnerable", giant bully is classed "At Risk Nationally uncommon". Gambusia is a national pest "unwanted organism" (Dunn et al. 2018).

Fish access into Awatapu Lagoon and the Wainui Te Whara Stream is facilitated by the use of fish-friendly flap gates (**FFG**), which remain open for a wide tidal range. The southern lagoon appears to have less diversity of fish (**Table 3.1**), which may reflect the poor water quality conditions.

Table 3.1: Fish present in Awatapu Lagoon and the Wainui Te Whara Stream (WTW) as identified in eDNA samples. Numbers are the eDNA sequences detected.

Scientific Name	Common Name	28/10/21 Awatapu central	17/11/22 Awatapu central	17/11/22 Awatapu west	17/11/22 Awatapu south	17/11/22 WTW Hinemoa	30/09/22 WTW Valley Rd
Anguilla australis	Shortfin eel	✓ 181	✓ 478	✓ 260	✓ 1387	✓ 2989	✓ 466
Anguilla dieffenbachii	Longfin eel		✓ 13			✓ 1061	✓ 219
Anguilla reinhardtii	Speckled longfin eel	✓ 10		✓ 48	✓ 19		
Gobiomorphus cotidianus	Common bully	✓ 70	✓ 617	✓ 249			
Gobiomorphus huttoni	Redfin bully	✓ 16				✓ 3452	✓ 1305
Gobiomorphus sp.	bully (unknown species)		✓ 1220	✓ 8896	✓ 38		
Galaxias maculatus	Inanga	✓ 211	✓ 8	✓ 158		✓ 122	✓ 876
Galaxias fasciatus	Banded kokopu					✓ 101	✓ 47
Galaxias postvectis	Shortjaw kokopu					✓ 130	
Retropinna retropinna	Common smelt	✓ 61	✓ 52			✓ 618	✓ 141
Salmo trutta	Brown trout					✓ 340	✓ 225
Carassius auratus	Goldfish	✓ 3162	✓ 9738	✓ 12297	✓ 9357		
Gambusia affinis	Mosquitofish	✓ 197	✓ 1103	✓ 416	✓ 804	✓ 10	✓ 15

3.5 Riparian and Aquatic Plants

3.5.1 Riparian and Emergent plants

Awatapu Lagoon is surrounded by urban parkland consisting of mature trees and mown grass. Mature native trees are established along the northern edge of the western lagoon and parts of the central lagoon. Swamp mire (*Syzgium maire*) and willow are common near the water's edge near the southern shore. A fringe of flax (*Phormium tenex*) is also common. Where riparian restoration planting has occurred, a more diverse range of native vegetation occurs, including: mānuka (*Leptospermum scoparium*), kahikatea (*Dacrycarpus dacrydioides*), *Coprosma tenuicaulis*, flaxes (*Phormium cookianum*), toetoe (*Austroderia fulvida*), rushes including *Machaerina articulata, Juncus egariae, J. pallidus, Leptocarpus similis*), and sedges including *Carex secta*, *C. lambertiana, Cyperus ustulatus,* and *Bolboschoenus fluviatilis*.

A narrow band of raupō (*Typha orientalis*) extends along much of the lagoon margin, interspersed in places with reed sweetgrass (*Glyceria maxima*), and patches of *Machaerina articulata*. Water pepper

(*Persicaria hydropiper*), swamp willow weed (*Persicaria decipiens*) and creeping bent (*Agrostis* sp) are common on the bank edge and extending to shallow margins. *Glyceria maxima* is invasive and can form dense monocultures that provide poor habitat for wetland birds, it is considered a pest in many regions. It should be controlled to prevent further spread amongst raupo in Awatapu.

3.5.2 Aquatic plants

In 1979, soon after the lagoon was formed, extensive beds of aquatic macrophytes *Lagarosiphon major* and *Potamogeton sp*. covered much of Awatapu Lagoon (BioReasearches 1979). By 1987, hornwort (*Ceratophyllum demersum*) was by far the most dominant aquatic plant in Awatapu Lagoon and parrot's feather (*Myriophyllum aquaticum*) was spreading, having established in the mid-1980s (WDC 1990).

Currently, the aquatic vegetation in Awatapu Lagoon is dominated by hornwort (*Ceratophyllum demersum*) and parrot's feather (*Myriophyllum aquaticum*)[7] – both are exotic pest plants. These can be rooted, but in Awatapu they more commonly occur as floating mats (often *c.* 0.5m thick) that cover large areas of the lagoon. The southern lagoon consistently has almost 100% coverage of hornwort and parrot's feather during summer. Floating mats of hornwort are also common in the central and western lagoons. The growth of hornwort in the western lagoon can be limited by the brackish water. The floating fern *Azolla pinnata* can also cover large areas of Awatapu during summer, and is easily recognised by its purple colour. *Azolla* is often found in association with duckweed (*Lemna minor*) **(Figure 3.8**).

The growth of hornwort, parrot's feather and Azolla is seasonal, but partially decaying floating mats of can persist in the lagoon through winter until they are eventually washed out during a flood, or sink to the bottom. Epiphytic algae often grow on the macrophytes, and can be seen on the surface as a light green slime.

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. However, the extensive cover of hornwort and parrot's feather in Awatapu causes major water quality problems with low dissolved oxygen below the thick floating mats of hornwort.

Extensive cover of the floating *Azolla* sp. often corresponds to clearer water as the shading and nutrient uptake reduces phytoplankton growth. However, *Azolla* can affect the aesthetics. *Azolla* also has the potential to fix atmospheric nitrogen due to a symbiotic relationship with a cyanobacteria within the plant, although this mechanism is weak when background nitrogen concentrations are sufficiently high for the plant.

The cover of hornwort and parrot's feather has historically been controlled by use of herbicide sprays. This is a relatively cheap way to control macrophytes, but 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, provides considerably more benefits for improving water quality and ecological values. WDC harvested macrophytes from

[7] First observed in about 1984.

south Awatapu lagoon in 2019, but a lack of any subsequent control allowed re-establishment of a dense cover of hornwort/parrot's feather within two years.

Figure 3.8: Top Photo: Awatapu Lagoon west with patches of floating hornwort in (August 2022). Bottom Photo: Southern Awatapu Lagoon with 100% cover of hornwort, parrot's feather and interspersed with Azolla sp. (March 2019).

3.5.3 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 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 3.9**) (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 long-term success when restoring shallow lakes. However, simply establishing macrophyte beds does not always improve water quality even when they improve habitat for fish and birds. Ecosystems are complex and often other types of 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.

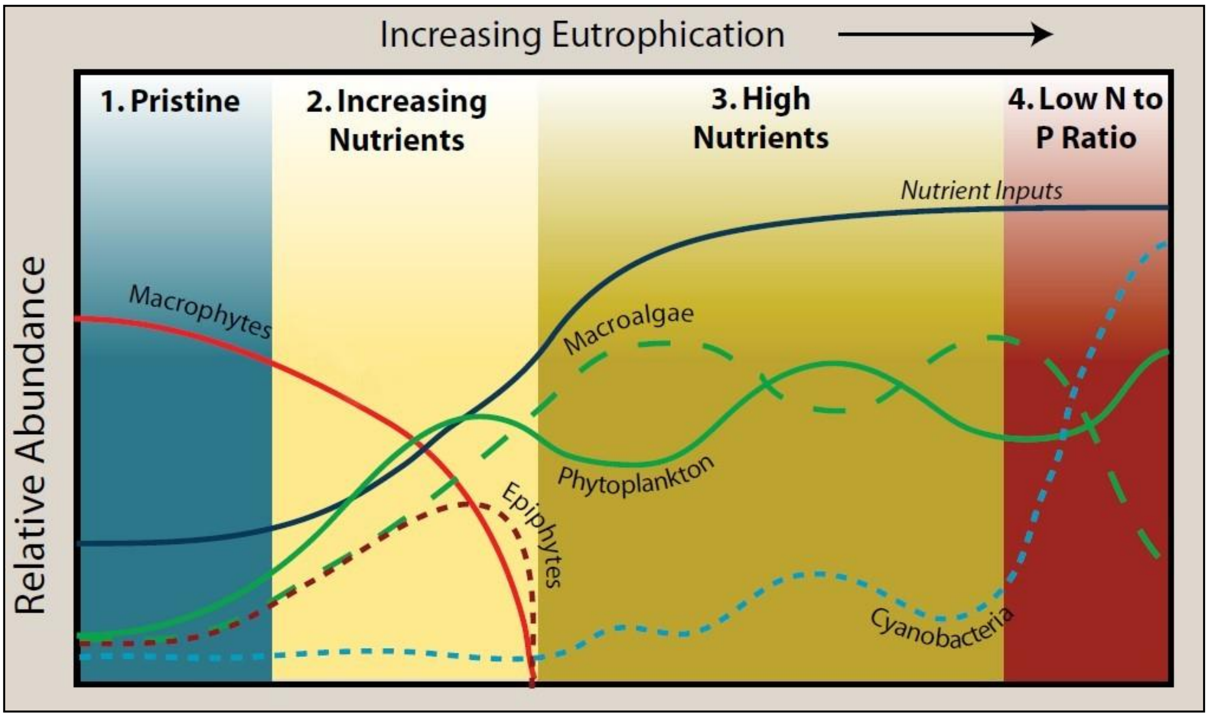


Figure 3.9: Generalised lake response to increasing eutrophication. Awatapu Lagoon appears to be in Stage 2 to 3 (adapted from De Wit et al. 2001).

3.6 Water Quality

3.6.1 Water quality

Awatapu Lagoon has poor water quality. Its TLI score is about 5.1, which is borderline between eutrophic and supertrophic. It has low water clarity (median 0.62m), high concentrations of total nitrogen (median 0.34 mg/L), total phosphorus (0.069 mg/L), and phytoplankton (median Chl-*a* 10.4 mg/m^3). Algae blooms are common during summer, and are often dominated by potentially toxic cyanobacteria (**Figure 3.10, Table 3.2, Figure 3.11**).

National bottom-line values set for lake attributes in the NPS-FM are not met for median TP, median Chl-a, maximum Chl-a, and cyanobacteria biovolume, or dissolved oxygen in the bottom waters. However, the microbial water quality of Awatapu Lagoon west appears to be in NOF Band B, which is suitable for Human Contact recreation.

Phytoplankton growth in Awatapu Lagoon appears to be more strongly limited by nitrogen than by phosphorus. The recent TN:TP ratio is about 4.8 (compared to a “balanced” ratio of 7) and the DIN:TP ratio is between about 0.16 and 0.4 (compared to a “balanced” ratio of 1). Absolute values of dissolved organic nitrogen (**DIN**) are often very low (median 0.02 mg/L), while DRP is moderate (median 0.012 mg/L). The components of the TLI in **Figure 3.10** show that TP is high relative to TN or Chl-*a*.

There is some indication that TP and DRP concentrations in Awatapu West may have reduced in the last 10 years, but the sampling has not been sufficiently consistent to have confidence in a trend. An apparent decline in TLI is mostly due to including Chl-a in more recent calculations rather than any real improvement (**Figure 3.11**).

Cyanobacteria blooms during summer can be associated with large diurnal fluctuations in dissolved oxygen and pH spikes in TN, due to the fixing of atmospheric nitrogen by cyanobacteria.

The microbial water quality of Awatapu Lagoon west is likely to be in NOF Band B for Human Contact. The median *E. coli* bacteria concentration is 43 cfu/100mL and the 95 percentile 295 cfu/100mL. Microbial water quality is good during stable conditions but elevated *E.coli* can occur during high flows into the lagoon.

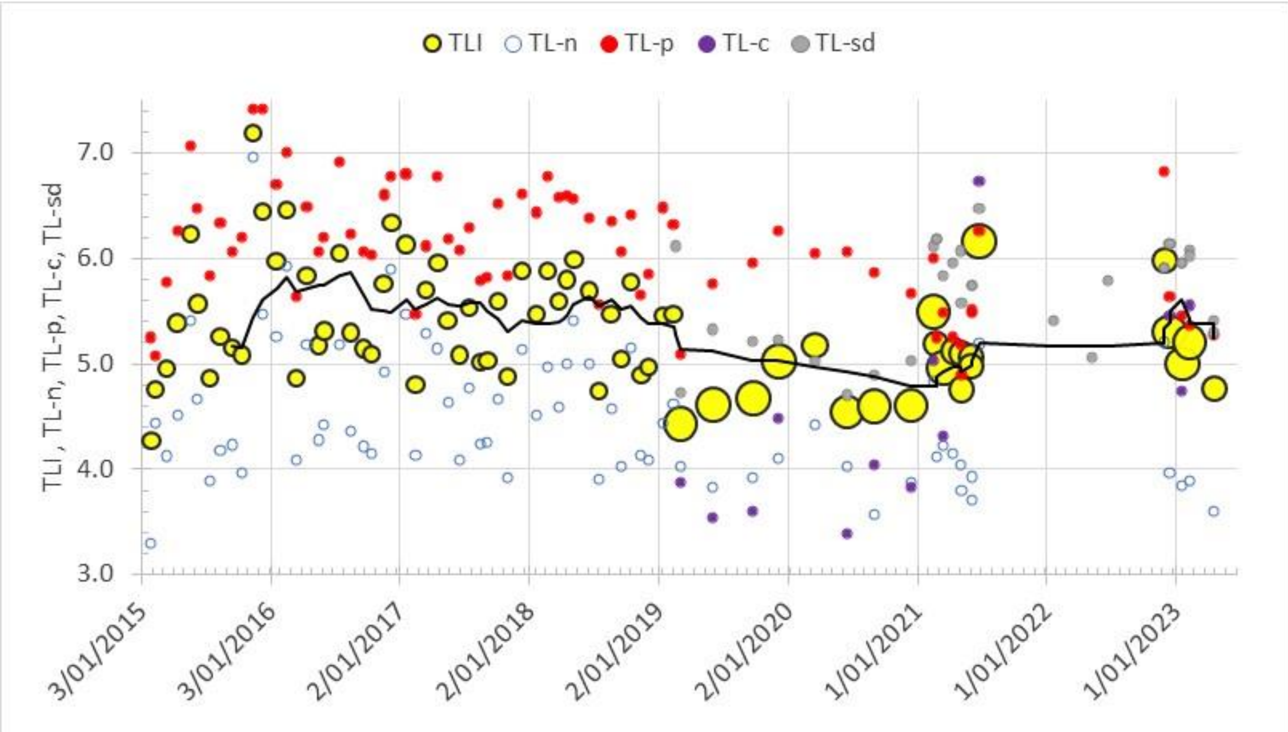


Figure 3.10: Awatapu Lagoon West Trophic Level Index (TLI) and its constituents for nitrogen (TL-n), phosphorus (TL-p), chlorophyll-*a* (TL-c), and Secchi depth (TL-sd). The size of the TLI circle indicates the number of components used for its calculation (range 2 to 4); and the line shows the TLI 12-point average.

Table 3.2: Water quality summary statistics for Awatapu Lagoon West, grouped by hydrological year (July to June). Samples prior to January 2021 were sampled from Awatapu Lagoon West at Awatapu Drive, TLI prior to March 2019 was estimated using only TN and TP.

Years	Variable	n	Min	Max	Mean	Median	95 %ile	5 %ile	75 %ile
2013-2019	TN (mg/L)	52	0.20	3.27	0.64	0.50	1.40	0.31	0.81
2013-2019	NH4-N (mg/L)	52	0	0.42	0.066	0.01	0.319	0.002	0.085
2013-2019	NNN (mg/L)	51	0	0.32	0.051	0.007	0.237	0	0.037
2013-2019	TP (mg/L)	52	0.05	0.29	0.127	0.119	0.221	0.054	0.153
2013-2019	DRP (mg/L)	52	0.01	0.13	0.027	0.023	0.058	0.011	0.031
2013-2019	CHL_A (mg/m3)	2	3.3	4.5	3.9	3.9		3.3	
2013-2019	VS- BD	3	0.48	1.59	1.01	0.95		0.48	1.43
2013-2019	Turbidity (NTU)	51	1.2	72.0	14.0	11.3	30.9	2.6	18.2
2013-2019	TLI *	52	4.3	7.2	5.4	5.4	6.4	4.6	5.8
2013-2019	Temp (degC)	52	9.5	27.4	18.1	18.3	26.0	10.2	22.3
2013-2019	pH	53	6.5	10.0	7.5	7.3	8.9	6.7	7.6
2013-2019	EC sp (uS/cm)	53	110	4510	595	238	2408	134	666
2013-2019	DO (mg/L)	51	0.2	16.8	8.9	8.7	14.4	3.1	10.7
2013-2019	E coli (cfu/100ml)	52	7	520	123	109	382	12	162
2013-2019	TN:TP	52	2.9	11.2	4.9	4.5	8.6	3.4	5.3
2019-2023	TN (mg/L)	20	0.24	0.85	0.40	0.34	0.85	0.25	0.39
2019-2023	NH4-N (mg/L)	20	0	0.07	0.013	0.006	0.058	0.003	0.013
2019-2023	NNN (mg/L)	20	0	0.18	0.022	0.005	0.128	0.001	0.018
2019-2023	TP (mg/L)	20	0.04	0.18	0.08	0.069	0.151	0.045	0.098
2019-2023	DRP (mg/L)	20	0	0.06	0.019	0.012	0.052	0.005	0.03
2019-2023	CHL_A (mg/m3)	14	2.9	169	24.6	10.4	147.2	3.0	18.8
2019-2023	VS- BD	29	0.35	1.62	0.75	0.62	1.39	0.45	1.00
2019-2023	Turbidity (NTU)	14	2.5	10.0	4.9	4.2	9.5	2.5	6.6
2019-2023	TLI *	20	4.6	6.2	5.1	5.0	6.1	4.6	5.2
2019-2023	Temp (degC)	22	10.8	27.8	18.1	18.9	25.0	10.9	21.5
2019-2023	pH	18	6.6	9.0	7.3	7.2	8.9	6.7	7.3
2019-2023	EC sp (uS/cm)	20	132	5000	472	175	3099	137	207
2019-2023	DO (mg/L)	20	1.8	11.9	8.1	8.7	11.7	2.3	10.1
2019-2023	E coli (cfu/100ml)	20	8	420	69	43	295	9	59
2019-2023	TN:TP	20	2.8	7.3	5.2	4.8	7.3	3.0	6.9

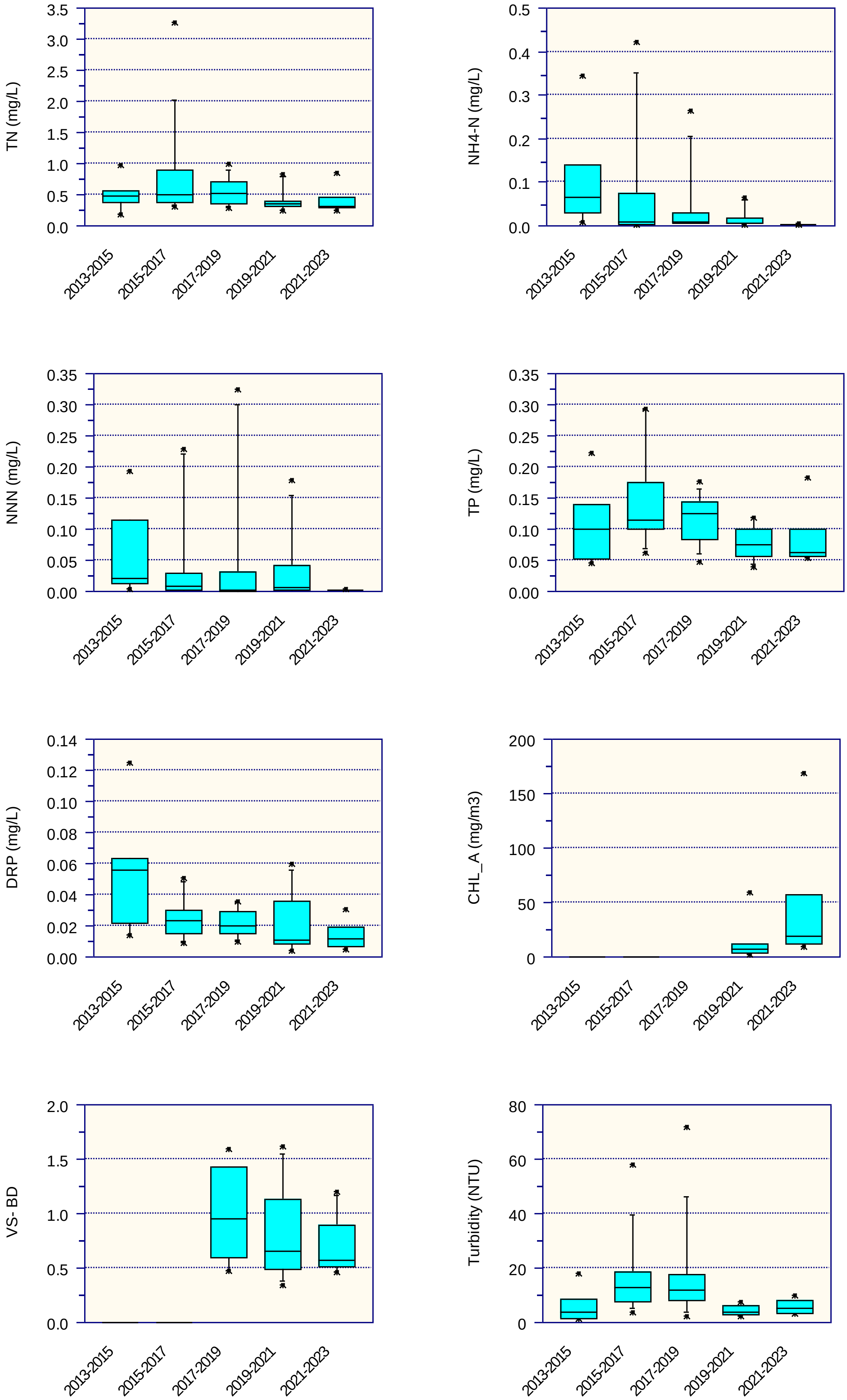


Figure 3.11a: Water quality in Awatapu Lagoon West (top) for two-year (July to June) periods.

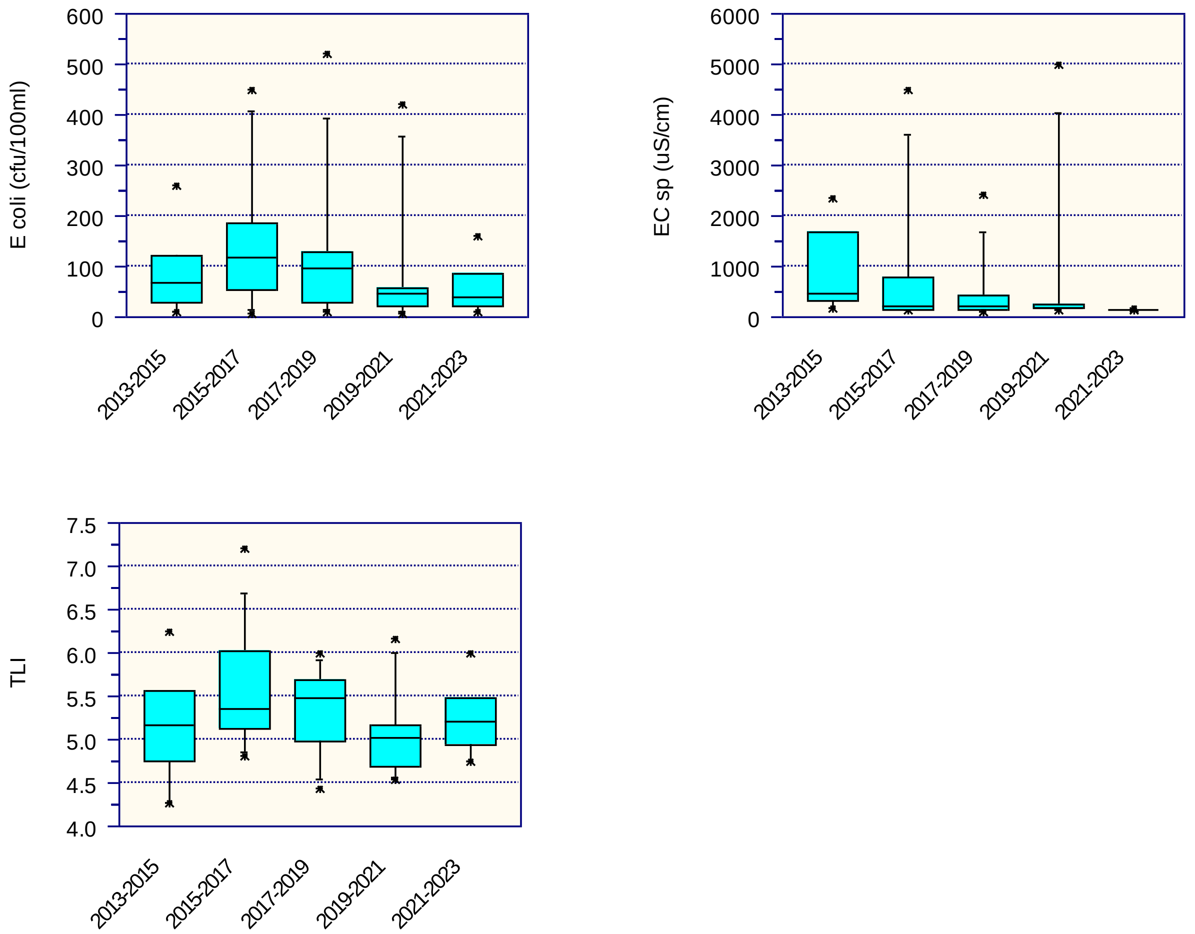


Figure 3.11b: Water quality in Awatapu Lagoon West (top) for two-year (July to June) periods. Samples prior to January 2021 were sampled from Awatapu Lagoon West at Awatapu Drive, TLI prior to March 2019 is estimated using only TN and TP.

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 common in Awatapu Lagoon during summer and autumn, often exceeding recreational use guidelines (MfE and MOH 2009). Summer monitoring of cyanobacteria in Awatapu Lagoon from 2015 to 2021 (30 samples) found that the **Alert Mode** (biovolume 0.5 to 10 mm^3/L) occurred on 30% of occasions, while the **Action Mode** trigger for total biovolume (≥10 mm^3/L) was exceeded on 23% of occasions, and the **Action Mode** trigger for potentially toxic cyanobacteria (biovolume ≥1.8 mm^3/L) was exceeded on 43% of occasions.

The 80th percentile of potentially toxic cyanobacteria biovolume for the three-year period of 2018-2020 was 8.7 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. were the dominant cyanobacteria present, but both *Anabaena* spp. and *Microcystis* spp. were prevalent during blooms; often seen as green flocs suspended in the water (**Figure 3.12**, **Table 3.3**).

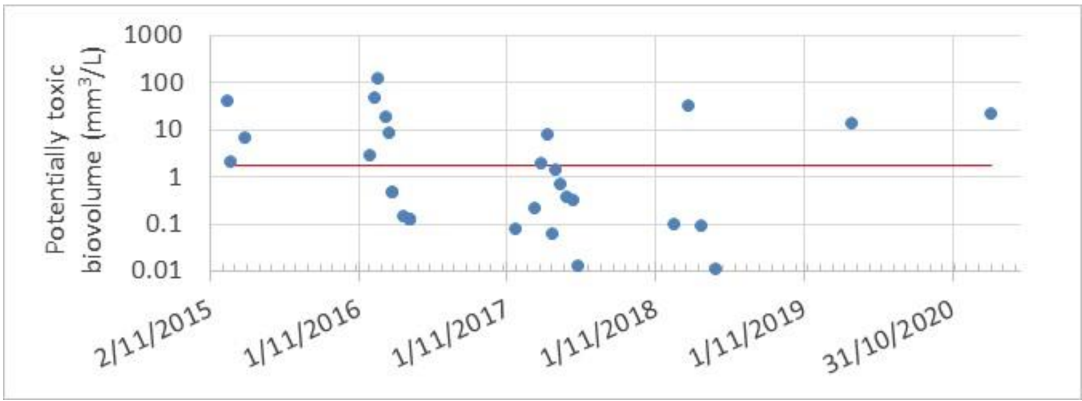


Figure 3.12: Biovolume volume of potentially toxic cyanobacteria in Awatapu Lagoon during summer (the red line is the 'Action' mode trigger).

Table 3.3: Occurrence of cyanobacteria species in Awatapu Lagoon west during summer, 2015-2021 (source BOPRC). Shading groups common genesis.

Species	Count	% occurance	% occurance when biovolume ≥10 mm^3/L
Anabaena circinalis	13	19%	11%
Anabaena lemmermannii	2	3%	
Anabaena planktonica	9	13%	22%
Anabaena sp	1	1%	
Anabaena spiroides	8	12%	22%
Anabaena torulosa	1	1%	
Aphanocapsa delicatissima	1	1%	
Aphanocapsa holsatica	1	1%	
Aphanocapsa sp	1	1%	
Chroococcus dispersus	2	3%	
Chroococcus limneticus	2	3%	
Coelosphaerium kuetzingianum	1	1%	
Geitlerinema sp	1	1%	
Microcystis aeruginosa	1	1%	
Microcystis cf panniformus	1	1%	
Microcystis flos-aquae	2	3%	
Microcystis panniformis	1	1%	11%
Microcystis sp	1	1%	11%
Microcystis sp. (small)	2	3%	11%
Microcystis wesenbergii	1	1%	
Oscillatoria sp	1	1%	
Phormidium sp	1	1%	
Planktothrix sp	9	13%	11%
Pseudanabaena galeata	1	1%	
Pseudanabaena limnetica	2	3%	
Trichodesmium iwanoffianum	1	1%	
No Cyanobacteria	1	1%	

3.6.3 Spatial variation in water quality

The Western and Southern lagoons have broadly similar water quality, but Awatapu Lagoon South has considerably worse water quality compared to the rest of the lagoon (i.e low DO, low clarity and high concentrations of TN, TP and *E.coli* bacteria). The water quality of the Wainui Te Whara Stream has similar TN, and much lower TP compared to Awatapu Central, but more of the nutrients are in a bioavailable form. *E.coli* bacteria concentrations tend to be higher in the Wainui Te Whara compared with the main lagoon (**Table 3.4**).

Table 3.4: Median water quality in the Awatapu Lagoon west, Lagoon Central, Lagoon South and the Wainui Te Whara (2021-2023, *n*= 9 to 12).

Site	EC sp (uS/cm)	DO (mg/L)	pH	BD (m)	TN (mg/L)	NNN (mg/L)	NH4-N (mg/L)	TP (mg/L)	DRP (mg/L)	CHL_a (mg/m3)	E coli (cfu/100ml)	TN:TP
Awatapu West - Top	169	9.2	7.2	0.57	0.31	0.003	0.005	0.062	0.008	12.4	27	4.8
Awatapu Central - Top	149	8.9	7.1	0.62	0.34	0.005	0.007	0.060	0.009	22.7	57	6.1
Awatapu South at Bridge St	257	3.1	6.7	0.43	0.49	0.018	0.074	0.181	0.009	13.7	350	2.2
Wainui Te Whara - Hinemoa St	128	10.2	6.9	1.15	0.33	0.156	0.012	0.037	0.019	0.8	335	8.2

3.6.4 Stratification, deoxygenation and internal nutrient release

A distinctive feature of Awatapu Lagoon is that the central lagoon and western lagoon experience both thermal stratification and salinity stratification during summer and autumn. Warmer surface water separates from the relatively cooler bottom waters by a thermocline – but this is usually weak and easily disturbed by wind. In addition, brackish water enters the western lagoon from the Whakatāne River when flows are low, which drops to the bottom of the lagoon and separates from the overlaying freshwater by a halocline – this stratification is persistent. In the western lagoon the halocline typically forms at about 1.5m depth. A halocline is less common in the central lagoon because the shallow (0.9m) water under the footbridge is a barrier to brackish bottom waters. However, on occasions when the central lagoon has brackish bottom water, the halocline is about 1.5m deep (e.g. April 2022), and it can persist in the deepest basin (below about 3m deep) even when the halocline in the western lagoon has dissipated (e.g. Nov 2022 to April 2023).

During periods of stratification, dissolved oxygen is depleted in the bottom water. Depth profiles found near-anoxic bottom water (DO < 1 mg/L) on 89% (16/18) of occasions in Awatapu Central, and 75% (15/20) of occasions in Awatapu West. In Awatapu West the pattern of bottom water anoxia can be complicated by tidal inflows of oxygenated brackish water from the river.

When anoxic conditions occur, it changes the geochemistry of the sediments and results in the release of nutrients (DRP and NH4-N) from the sediment. Elevated nutrient concentrations in the low-oxygen bottom-water is evident in the sampling from Awatapu (**Figure 3.13, Table 3.5**). Elevated nutrients from the bottom water become more available for phytoplankton growth when the top and bottom waters eventually mix. The effects of this in the western lagoon may be mitigated because the mixing that results in the loss of the halocline in the western lagoon is often associated with higher river flows and more flushing.

On 16 March 2021, the bottom water of Awatapu Lagoon West had compete anoxia (DO = 0 mg/L). Measurement of gas in the bottom water found H_2S at 984 ppm, CH4 at 0.7% and CO2 at 1.7%. This is consistent with literature showing decomposition of organic matter in eutrophic lakes being a source of greenhouse gases.

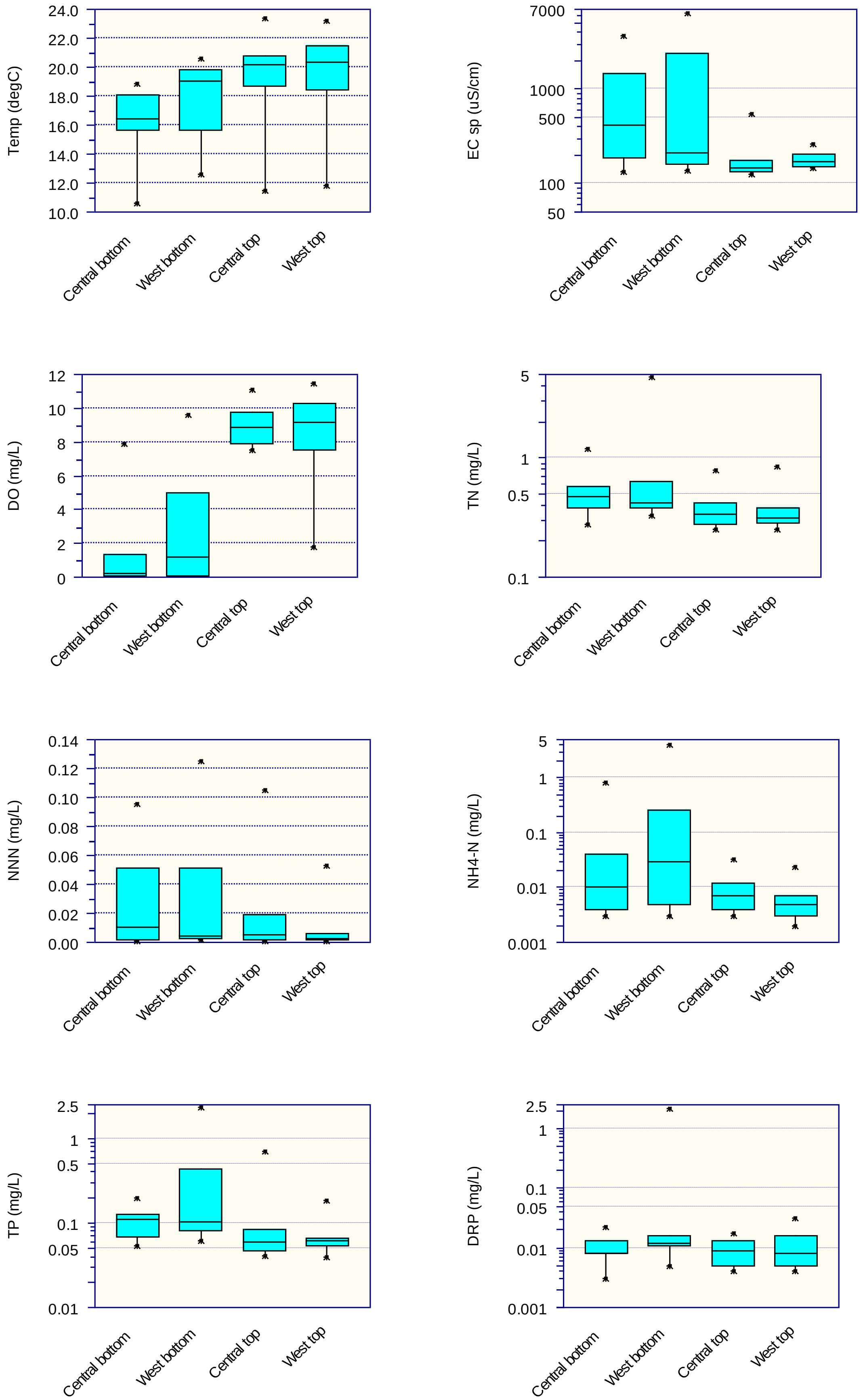


Figure 3.13: Top and bottom water quality from mid-lake sites of Awatapu Lagoon West and Awatapu Lagoon Central (February 2021 to April 2023, n= 10).

Table 3.5: Summary statistics for top and bottom water samples from Awatapu Lagoon West and Awatapu Central.

Site	Variable	n	Min	Max	Mean	Median	75 %ile	25 %ile
Central bottom	Temp (degC)	9	10.6	18.9	16.2	16.4	18.1	15.6
Central bottom	pH	8	5.9	7.1	6.5	6.4	6.9	6.1
Central bottom	EC sp (uS/cm)	9	132	3620	1071	421	1465	188
Central bottom	DO (mg/L)	9	0.0	7.9	1.4	0.2	1.3	0.1
Central bottom	DO sat (%)	9	0.9	71.0	19.0	2.6	43.3	1.6
Central bottom	TN (mg/L)	9	0.28	1.20	0.53	0.47	0.58	0.38
Central bottom	NH4-N (mg/L)	9	0.003	0.835	0.11	0.01	0.04	0.004
Central bottom	NNN (mg/L)	9	0.001	0.096	0.028	0.01	0.051	0.002
Central bottom	TP (mg/L)	9	0.053	0.201	0.109	0.112	0.125	0.069
Central bottom	DRP (mg/L)	9	0.003	0.022	0.01	0.008	0.013	0.008
Central bottom	Turbidity (NTU)	5	5.7	70.0	21.0	9.0	26.5	7.7
Central bottom	TN:TP	9	3.6	7.2	4.9	4.7	5.5	4.2
West bottom	Temp (degC)	9	12.6	20.6	17.8	19.0	19.8	15.7
West bottom	pH	8	6.2	7.1	6.6	6.5	6.9	6.3
West bottom	EC sp (uS/cm)	9	139	6482	1692	214	2404	162
West bottom	DO (mg/L)	9	0.0	9.6	2.9	1.2	5.0	0.1
West bottom	DO sat (%)	9	0.0	91.0	30.5	12.3	55.0	1.3
West bottom	TN (mg/L)	9	0.33	4.71	0.96	0.42	0.64	0.39
West bottom	NH4-N (mg/L)	9	0.003	4.06	0.559	0.029	0.252	0.005
West bottom	NNN (mg/L)	9	0.002	0.125	0.03	0.004	0.051	0.003
West bottom	TP (mg/L)	9	0.061	2.34	0.465	0.102	0.433	0.081
West bottom	DRP (mg/L)	9	0.005	2.19	0.254	0.012	0.016	0.011
West bottom	Turbidity (NTU)	5	4.6	15.0	8.6	7.3	10.4	6.5
West bottom	TN:TP	9	1.0	5.5	3.6	4.2	4.9	1.9
Central top	Temp (degC)	10	11.5	23.4	19.2	20.2	20.8	18.7
Central top	pH	9	6.1	7.4	6.8	7.1	7.1	6.3
Central top	EC sp (uS/cm)	10	126	551	190	149	176	135
Central top	DO (mg/L)	10	7.5	11.1	9.0	8.9	9.8	7.9
Central top	DO sat (%)	10	80.0	120.0	96.4	90.4	111.6	86.0
Central top	TN (mg/L)	9	0.25	0.79	0.39	0.34	0.42	0.28
Central top	NH4-N (mg/L)	9	0.003	0.033	0.01	0.007	0.012	0.004
Central top	NNN (mg/L)	9	0.001	0.105	0.02	0.005	0.019	0.002
Central top	TP (mg/L)	9	0.041	0.71	0.133	0.06	0.083	0.047
Central top	DRP (mg/L)	9	0.004	0.018	0.01	0.009	0.013	0.005
Central top	Turbidity (NTU)	5	2.6	7.7	5.2	5.3	6.4	3.8
Central top	E coli (cfu/100ml)	9	8.0	1200.0	186.0	57.0	110.0	29.3
Central top	TLI3	9	4.6	6.3	5.3	5.1	5.5	4.8
Central top	TN:TP	9	0.6	8.0	5.6	6.1	6.4	5.5
West top	Temp (degC)	10	11.8	23.2	19.3	20.4	21.5	18.4
West top	pH	8	6.6	7.5	7.1	7.2	7.3	6.9
West top	EC sp (uS/cm)	10	147	260	183	169	204	153
West top	DO (mg/L)	10	1.8	11.5	8.5	9.2	10.3	7.5
West top	DO sat (%)	10	17.6	126.2	91.3	91.4	114.3	84.7
West top	TN (mg/L)	9	0.25	0.85	0.37	0.31	0.38	0.29
West top	NH4-N (mg/L)	9	0.002	0.024	0.007	0.005	0.007	0.003
West top	NNN (mg/L)	9	0.001	0.053	0.009	0.003	0.006	0.002
West top	TP (mg/L)	9	0.04	0.184	0.072	0.062	0.066	0.054
West top	DRP (mg/L)	9	0.004	0.031	0.012	0.008	0.016	0.005
West top	Turbidity (NTU)	5	3.1	10.0	5.8	5.4	7.5	3.6
West top	E coli (cfu/100ml)	9	9.0	160.0	44.9	27.0	54.0	18.0
West top	TLI3	9	4.8	6.0	5.1	5.0	5.2	4.9
West top	TN:TP	9	4.2	7.3	5.4	4.8	6.5	4.6

3.6.5 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 facilitate the release of nitrogen (as NH_4-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.

3.6.5.1 Dissolved oxygen depth profiles

Changes in DO, temperature and electrical conductivity was measured with depth when mid-lake water samples were collected. During summer, the dissolved oxygen was often low or anoxic in the bottom waters below about 1.5m (range 1m to 2m). This is associated with both salinity stratification and thermal stratification. The higher density of salt water compared to freshwater makes salinity stratification (and its associated halocline) stronger and more stable than thermal stratification. Some of the thermal stratification measured in the profiles may be diurnal, with the strength of the stratification weakening as water cools at night.

During summer, thermal stratification and salinity stratification reinforce each other to separate bottom water from mixing with surface waters, however even when surface water temperatures cool during late autumn / winter the bottom waters are slow to mix while the salinity stratification persists (e.g. April and May 2022) (see graphs in **Appendix 1**).

Brackish is usually restricted to water deeper than about 1.5m to 2.5m deep. The persistence of brackish bottom water can differ between the central lagoon and the western lagoon depending on river conditions. The western lagoon becomes saline sooner as river flows reduce (e.g. January 2022), but the brackish water doesn't persist as long when the river levels are high and tidal inflows are fresher (e.g. see profiles from 2023 in **Appendix 1**).

3.6.5.2 Temporal Variation in DO

Dissolved oxygen loggers were installed in Awatapu Lagoon Central in February/March 2017 and in Awatapu Lagoon West from April to June 2022. Moderately large diurnal fluctuations of DO occur in the surface water of the lake in response to photosynthesis and respiration from phytoplankton and plants. When the lake is stratified the bottom water has low DO concentrations and the surface water is aerated, but when the stratification breaks down, the mixing of the bottom water reduces the DO in the surface water (**Figure 3.14** and **Figure 3.15**).

During early April 2022 there was strong salinity stratification in Awatapu West with the halocline at about 1.5m depth. Bottom waters were anoxic while the surface water had over 100% saturation. Bottom waters became progressively less brackish during April (likely due to higher flows in the Whakatāne River (to 44 m^3/s) reducing saline inflows). By 10 May 2022 the depth of the halocline had reduced to 2.0 - 2.5m. On 21 May 2022, flood in the Wainui Te Whara initiated full mixing of the western lagoon, and the lagoon was fully mixed follow further floods in early June. Following full mixing, the surface water DO range between about 40% and 70% saturation (**Figure 3.14**).

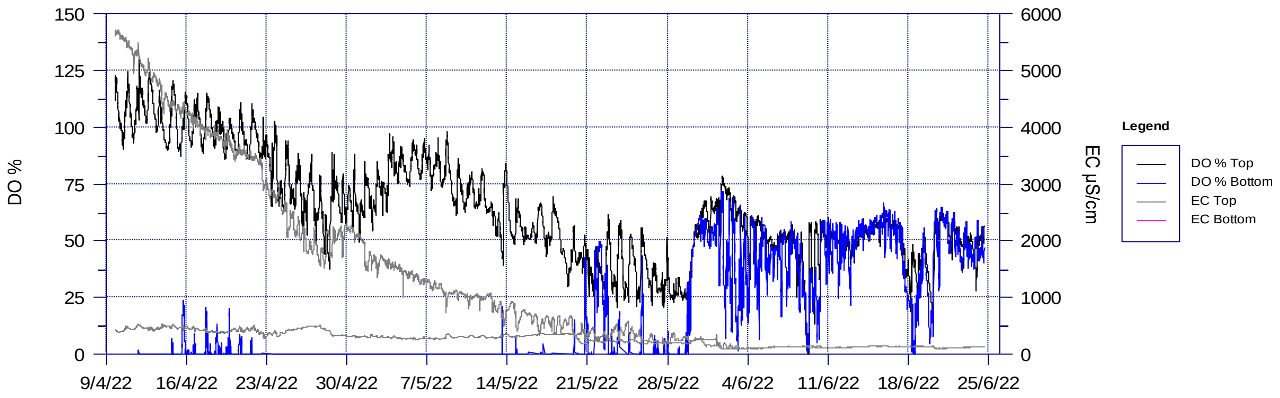


Figure 3.14: Dissolved oxygen and electrical conductivity in the top and bottom waters of Awatapu Lagoon West, autumn 2022. Bottom waters became progressively less brackish during April. Oxygen returned to the bottom water around 21 May following a flood in the Wainui Te Whara Stream.

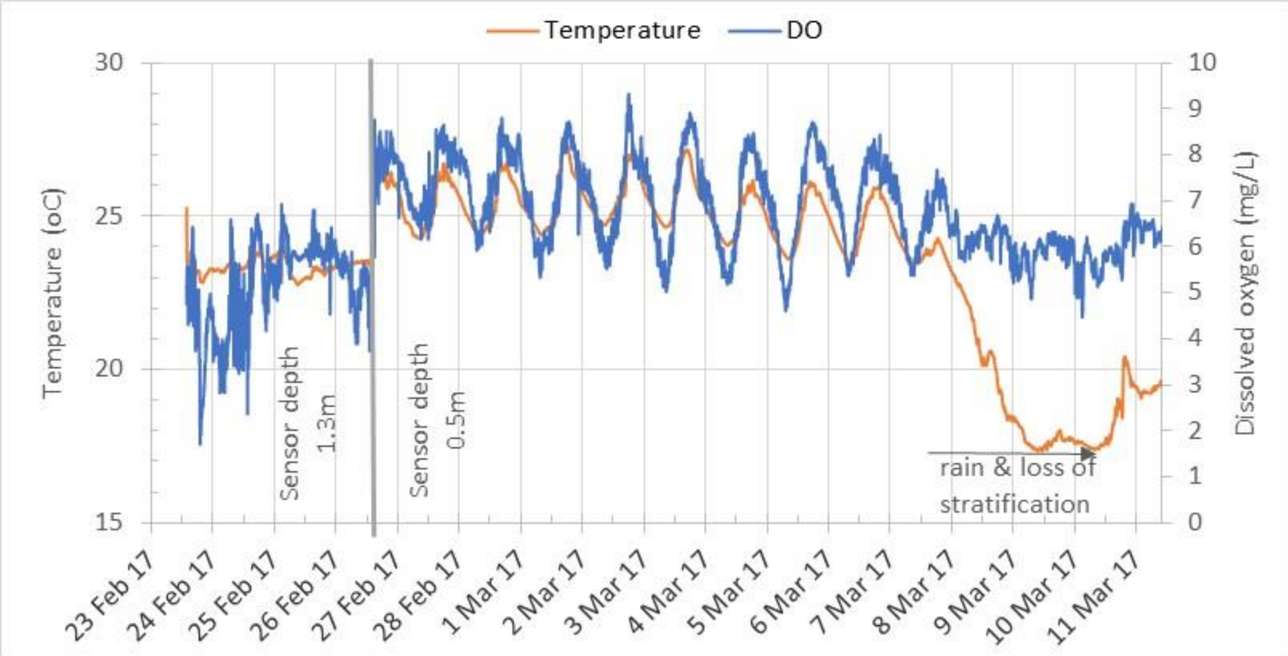


Figure 3.15: Dissolved oxygen in Awatapu Lagoon Central. The logger was initially located at 1.3m depth and moved to 0.5m depth after four days. On 24/2/2017 there was a thermocline at about 1m depth, and DO was 8.2mg/L in the surface water and 3.5 mg/L near the bottom at 1.5m depth.

3.6.5.3 Spatial Variation in DO

Awatapu Lagoon can have considerable spatial variations in DO, pH and EC. Synoptic surveys undertaken in the late afternoon on 11 April 2022 found the western end of Awatapu Lagoon generally had DO mostly above full saturation and elevated pH. In stark contrast, the southern lagoon and south of the Wainui Te Whara delta have very low DO and slightly acidic pH. These patterns are consistent with the distribution of aquatic macrophyte cover. Floating mats of hornwort and associated epiphytes had accumulated in the western end of the western lagoon and readings of DO and pH varied depending on whether the sensor was in water just above a mat (high DO and pH) or beneath a mat (low DO and pH). The southern lagoon has persistently high macrophyte cover which accumulates and decomposes. The mats are surface reaching and readings were made just below the floating mats (**Figure 3.16, Figure 3.17**).

EC was lowest near the Wainui Te Whara delta and increased towards the western end of the Western Lagoon. This reflects the inflow of brackish water from the Whakatāne River. Interestingly, EC is also slightly elevated in the Southern Lagoon, which may reflect more dissolved ions from decomposition processes (**Figure 3.18**).

The DO results from the synoptic survey are consistent with previous measurements. In 2021 the spot readings of percent DO saturation in the Southern Lagoon ranged from 3.5% to 57% saturation. In February 2021 floating rafts of hornwort (about 400mm thick) covered large areas of the Central Lagoon. The DO concentration immediately underneath these rafts were less than 5% saturation (0.45 mg/L), and got lower with depth. This is much less than the oxygen requirements of even very tolerant fish, and dead goldfish were observed in the lagoon on this occasion.

Figure 3.16: Spatial variation of dissolved oxygen saturation (%) in Awatapu Lagoon during late afternoon on 11 April 2022. High DO saturation in the western lagoon is associated with the photosynthesis of hornwort mats in this part of the lagoon. Very low DO saturation in the southern lagoon and near the Wainui Te Whara delta is consistent with decomposition of macrophyte organic matter in this area.

Figure 3.17: Spatial variation of pH in Awatapu Lagoon during late afternoon on 11 April 2022. pH is high in the western lagoon associated with the photosynthesis of hornwort accumulating in this part of the lagoon. Low pH in the southern lagoon and near the Wainui Te Whara delta is consistent with decomposition of macrophyte organic matter.

Figure 3.18: Spatial variation of electrical conductivity (uS/cm) in Awatapu Lagoon during late afternoon on 11 April 2022. EC is higher in the western lagoon due to the influence of brackish water from the Whakatāne River.

3.6.5.4 Summary of DO regime

Awatapu Lagoon West and Awatapu Lagoon Central can have persist salinity stratification due to brackish water that enters from the Whakatāne River during high tides and when the river has low flows (i.e. less than about three quarters of median flow).

When Awatapu Lagoon West and Lagoon Central are stratified the surface water is usually replete in oxygen, while the bottom water has very low oxygen. During periods with high macrophyte cover the southern lagoon can have very low dissolved oxygen concentrations. When the stratification breaks down, the bottom water becomes aerated but the mixing of the bottom water reduces the DO in the surface water.

There can be strong spatial variability in surface water DO associated with the distribution of floating mats of hornwort. Thick mats block the air from aerating the surface water and partial decomposition within the mats creates an oxygen demand. In 2021, DO was measured at less than 5% saturation directly below an expanse of thick macrophyte mats floating over deep water. Furthermore, the sinking of hornwort mats to the lake bed and their subsequent and decomposition will be a major contribution to accumulated organic matter that drives the rapid decline in DO in bottom water following stratification.

3.7 Water quality issues affecting Awatapu Lagoon

Awatapu Lagoon provides valuable habitat for birds and fish; and many fish species migrate through the lagoon to the upper catchment of the Wainui Te Whara Stream. The lagoon is an important area for passive recreation and has considerable potential for improving recreational, habitat and water quality values.

Key ecological and water quality issues identified in Awatapu Lagoon include:

- Poor water quality: Low clarity, high nutrients (particularly phosphorus), cyanobacteria blooms, and low dissolved oxygen in the southern lagoon.
- Water quality not suitable for recreational bathing due to cyanobacteria blooms. However microbial water quality (*E.coli* bacteria) is usually within contact recreation guidelines.
- During periods of stratification, the dissolved oxygen in the bottom waters of Awatapu Lagoon rapidly declines due to oxygen demand from organic matter on the lake bed. Anoxic conditions can mobilise nutrients from the lake bed sediment. Methane and sulphur dioxide is released during organic matter decomposition under anoxic conditions and can accumulate in the bottom waters.
- Floating mats of hornwort can occur throughout the lagoon and cause low DO conditions beneath them. These also impede kayaking and visual amenity values.
- Awatapu Lagoon South consistently has low DO due to extensive cover of hornwort and high organic matter load when the hornwort dies back during winter.
- The extensive cover of aquatic pest plants (hornwort and parrot’s feather) make establishment of native aquatic macrophytes more difficult.
- The exotic weed, *Glyceria maxima,* is spreading amongst raupo on the margins of the lagoon. It requires spraying to reduce its extent and to prevent further spread.
- There is opportunity to improve water quality by harvesting the hornwort and parrot’s feather. Harvesting would not only mitigate their effect on the DO regime, but also remove organic matter and nutrients from the lake system.
- Herbicide could be used to maintain low cover of hornwort between harvests, but there is a risk of causing worse water quality conditions if herbicide is the only method of control.
- There is an opportunity to improve water quality and habitat by creating in-lake wetlands.
- Rubbish is a persistent problem in Awatapu.

4 Management Actions to improve Awatapu Lagoon

4.1 Introduction

There is a strong community desire to improve the water quality and ecological values of the Awatapu Lagoon. This will require an integrated approach that reduces external and internal nutrient loads, and enhances biological processes mediated through aquatic macrophyte and wetland vegetation. It will also require managing the excessive growth of pest macrophytes that cause substantial nuisance and water quality problems in Awatapu Lagoon.

General 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**.

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 .

The current literature informed potential intervention measures to address specific water quality and ecological issues in Awatapu Lagoon. These are described in **Table 4.1**, and a sub-set of these management interventions were selected based on their potential benefits and input from WDC.

The key management interventions assessed for Awatapu Lagoon are:

- Reducing catchment sediment and nutrient loads (e.g. by use of detainment bunds in the upper Wainui Te Whara catchment).
- Reducing residence time by increasing flows into Awatapu from the Whakatāne River.
- Harvesting macrophytes to manage plant cover and remove nutrients (possibly in conjunction with herbicide to maintain low biomass between harvesting).
- Constructing wetlands to treat nutrients and improve biodiversity values. [8]
- Floating wetlands to remove nutrients and improve biodiversity.
- Sediment phosphorus (P) locking to reduce internal load of P.

[8] The 2021 Long Term Plan approved funding to creation of wetland areas in Awatapu lagoon including diverting baseflow of the Wainui Te Whara into Awatapu south. The concept design for this project has been modified to be more cost-effective and extend the water quality benefits for the whole of Awatapu Lagoon at a lower cost.

Table 4.1: Potential interventions to address ecological and water quality issues in Awatapu Lagoon.

Ecological issues	Causes	Potential management options
Supertrophic high concentration of nutrients, algae, poor clarity. Cyanobacteria blooms.	External nutrient load	o Create treatment wetlands in vicinity of Wainui Te Whara delta. o Floating wetlands for N removal and habitat. o Sediment detainment bunds in upper catchment o Educate to reduce nutrient use in catchment
	Internal nutrient load via sediment resuspension or anoxia	o Harvest aquatic macrophytes / azolla and remove from the lagoon. o Phosphorus-locking of sediments with focus on deep areas that develop halocline (west and central lagoon).
	Bottom water anoxia	o Harvest macrophytes to reduce load of organic matter.
Poor oxygen conditions	Excessive macrophyte cover as floating rafts.	o Harvest aquatic macrophytes. Option of using spray to manage biomass between harvesting.
	Halocline stratification in western lagoon	o Naturally occurring but may have limited impact if broken only during high flow events.
	Negligible flow in southern lagoon	o Pipe or syphon water from Whakatāne River to Awatapu south using tidal head. o Divert WTW Stream to flow via the south lagoon in conjunction with creating treatment wetlands.
Excessive aquatic pest plant cover affecting water quality, biodiversity, recreation and aesthetics. (contribute C and N)	Pest plants of hornwort and parrot's feather as floating rafts and rooted in shallow areas	o Harvest aquatic macrophytes o Herbicide spray of aquatic macrophytes to maintain low biomass between harvests (risks worsening low DO and high nutrients).
Pest plants on riparian zone	*Glyceria maxima* near the foot bridge	o Targeted herbicide spray of reed *Glyceria* sp. o Remove weeds on floating wetlands
Siltation	Stream inflow. Decomposition of macrophytes.	o Continue sediment removal from WTW delta o Extend WTW delta to create a wetland areas that help trap suspened sediment.
Litter	Rubbish directly and via stormwater	o 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 shallow sloping banks into water and plant with wetlands.
Poor access to the water	Lack of structures in water	o Construct jetties and boat-launching areas in western lagoon
Enhance biodiversity		o Create wetlands and floating wetlands for birds, fish and invertebrates. o Harvest pest aquatic macrophytes to assist natives. o Establish native macrophytes. 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 2**.

Detainment bunds can be a cost-effective option for reducing sediment and P that may have potential in the upper catchment of the Wainui Te Whara Stream. Detainment bunds are low earth berms placed across ephemeral storm water flow paths on farms to temporarily detain storm water run-off. The detained water allows the settling of sediment and associated phosphorus and microbial contaminants within the paddock and reduces export to waterways. They also increase water infiltration and this enhances their effectiveness at contaminant removal. The infrequent and short-term inundation does not compromise pasture production. Standard criteria have been developed for detainment bund design referred to as “Detainment Bund PS120”, which incorporates a key design element of having a minimum water storage of 120 m3 per hectare of contributing catchment (Paterson et al. 2020).

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.

Detainment bunds can be very effective at preventing the export of sediment and phosphorus and retaining it on the paddock. Studies have found that they retain 47% - 68% of the TP in stormwater run-off, 57-72% of the TN and 51% - 59% of suspended sediments. Average removal rates were 0.72 kg TP per ha of catchment per year and about 2.0 kg TN/ha/yr. In the Rotorua catchments, the detainment bunds allow about 50% of the runoff to infiltrate back into soils, in addition they reduce peak storm flows that can accelerate stream bank erosion (Levin et al 2020). Their practical application depends largely on the topography and farming practices.

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.

Detainment bunds are relatively cheap to construct which makes them a very cost-effective way to reduce the export of phosphorus. Levin (2020) estimated their cost-effectiveness as $120 - $140 /kg of phosphorus retained (based on an average cost of $20,000 per detainment bund and annualised). The costs depend on the permitted activity rules in a region. Resources may also need to be assigned to identifying suitable locations within a catchment to locate detainment bunds and to liaise with landowners.

4.2.4 Application to Awatapu Lagoon

4.2.4.1 Suitability

Potential options to reduce external nutrient loads entering Awatapu Lagoon from its catchment include:

- Use of sediment detainment bunds in the upper catchment of the Wainui Te Whara
- 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.

Lake catchments are particularly sensitive to nutrient inflows. There may be good potential for using detainment bunds in the upper catchment of the Wainui Te Whara Stream, but this would need further investigation.

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 within the lower Wainui Te Whara Stream. McDowell et al. (2007) described the use of melter slag contained 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.

One option may be to trial the use of melter slag P-socks in the Wainui Te Whara, but some additional sampling of inflows is recommended to 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.

4.2.5 Summary

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

- Use detainment bunds in the upper catchment of the Wainui Te Whara Stream.

- Educate land owners about reducing sediment and nutrient discharges to the stormwater network.
- Use P-socks at culvert outlets to bind phosphorus.

4.3 Reduce residence time by increasing flow of river water

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 reduces 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.

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.

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.

Increasing flows are unlikely to be effective at managing rooted macrophytes in lakes, but may provide a net movement of floating macrophyte rafts if the flows are sufficiently high. In Awatapu lagoon, the location of floating macrophyte appears mostly determined by wind direction/strength, but there is a noticeable movement of macrophytes towards the outlet when the Wainui Te Whara has a large flood greater than about 2 m^3/s (equivalent to a residence time of about 1.2 days).

4.3.3 Application to Awatapu

4.3.3.1 Amount of flow to be effective

Water from the Whakatāne River could be rediverted to the Awatapu lagoon south (the old inlet) by either a pump, or an augmented syphon, or via a gravity culvert. This would decrease the residence time for all of Awatapu Lagoon. Diverting some flow from the Whakatane River back to Awatapu Lagoon may also help restore aspirations of local iwi to reconnect the lagoon with the river.

The amount of flushing achieved and its potential benefits is dependent on the amount of water rediverted. Rediverting 70 L/s flow from the Whakatāne River to Awatapu south would reduce the median residence time in Awatapu South to 7 days (42,400 m^3/ 6048 m^3/day), and reduce the annual median residence time in Awatapu Central/West to 17.3 days (209,200 m^3 / 12,096 m^3/day). This would exert a small control on phytoplankton biomass in central and western lagoons, and substantially reduce phytoplankton biomass in the southern lagoon. However, it would not provide sufficient water velocity to cause meaningfully change in the cover of macrophytes in the southern lagoon.

Allowing a 70 L/s flow into the southern lagoon would improve the dissolved oxygen regime, but the extent of improvement during summer is likely to be small if there continues to be extensive macrophyte cover. Measurements of DO in the central lagoon (with flow from the Wainui Te Whara) during February 2021 found the DO immediately under a floating raft of hornwort was 5% (0.5 mg/L) while the DO in the adjacent open water was 114% (9.8 mg/L) - indicating strong oxygen demand (i.e. from sediment and the base of the floating mat) relative to reaeration from the air or stream flow.

Rediverting 220 L/s flow from the Whakatāne River to Awatapu south would reduce the median residence time in Awatapu South to 2.2 days (42,400 m^3/ 19,008 m^3/day), and reduce the annual median residence time in Awatapu Central/West to 8.3 days (209,200 m^3 / 25,056 m^3/day). This would significantly reduce phytoplankton biomass during summer throughout all of Awatapu Lagoon. A residence time of two days may be sufficient to move floating rafts of macrophytes, but not rooted macrophytes (based on observations following floods).

Further increasing the flow to 340 L/s would reduce the annual median residence time in Awatapu Central/West to 5.9 days. However, this amount of flow may start to noticeably) increase (i.e. >0.25m the water level in the southern lagoon [9].

The amount of water able to be diverted from the Whakatāne River to Awatapu lagoon is largely dependent on the pipe size, the pump size and /or the tidal head. These are discussed below for different options.

4.3.3.2 Water quality considerations

Concentrations of nutrients TN and TP are typically lower in the Whakatāne River (median 0.19 mg/L and 0.05 mg/L respectively) than in Awatapu west (median 0.31 mg/L and 0.062 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 Awatapu (median 0.004 mg/L and 0.008 mg/L respectively)[10]. The Whakatāne River is turbid during flood and any flow diversion into the lagoon should pause during flood events.

4.3.4 Diverting flow from Whakatāne River using a gravity culvert

Diverting a portion of water from the Whakatāne River to the original inlet of Awatapu (at the western end of the southern lagoon) would provide additional flow throughout the lagoon. On average, there is negligible difference in head between the original inlet and the outlet of Awatapu, but there are tidal fluctuations that causes the river level to be higher than the lagoon at high tides. These tidal fluctuations could be harnessed to drive a flow of water.

One option is to use a large culvert with a fish-friendly flap gate at the upstream end, and a normal flap gate at the downstream end to prevent backflow. A mathematical tidal model was developed to assess

[9] The flow from the southern lagoon is controlled by a partially submerged 2m diameter culvert under Bridge Street. The Bridge Street culvert invert is set at -0.033m RL, which is lower than that of the outlet culverts (0.065m RL).

[10] Hamill et al. 2020

the head difference and water flow over a 28-day tidal cycle. The model was calibrated using water level measurements from loggers placed in Awatapu Lagoon and in the Whakatāne River (described earlier in this report). Flow through the partially full pipe was calculated using the Bernoulli's equation for a culvert with inlet control. The effect of the FFG on head loss was assumed to be linear between the start of the FFG closing and when fully closed (set at 60% and 100% pipe of pipe height) (**Figure 4.1**).

The tidal model estimated that a 1.2m diameter culvert, 70m long, might achieve a monthly average inflow of about 150 L/s. This inflow would only occur during high tides, and there would be more flow during a spring tide than during a neap tide (**Figure 4.2)**.

However, constructing the culvert through the stop bank would be expensive. Piping through the stopbank could be constructed by either directional drilling or by an open-cut trench. In discussions with WDC (2019), BOPRC has expressed the view that an open-cut a trench through the stopbank would have less risk to the stopbank integrity. However, this method is much more expensive. A similar project was undertaken for McAlister Stormwater Pump Station upgrade in 2018, that installed a 3m x 1m box culvert at a location where the stopbank is about 4.1m high. The stopbank at the southern lagoon is about 7m high. The cost of open trenching a pipe through the stopbank to Awatapu south would likely be close to $1,000,000.[11]

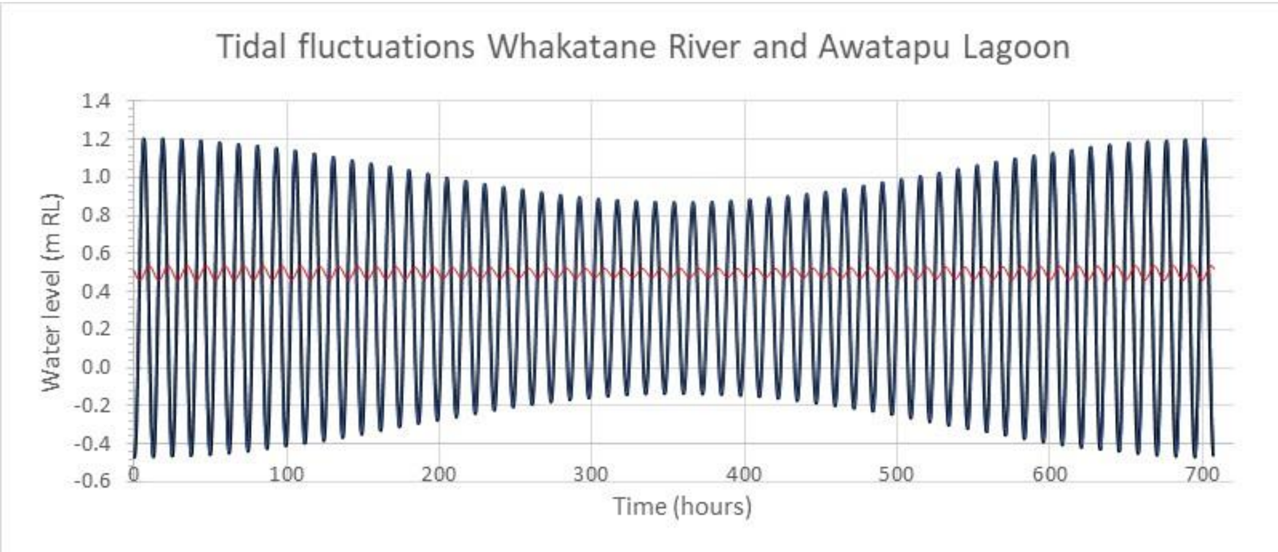


Figure 4.1: Modelled tidal fluctuations in The Whakatane River and Awatapu Lagoon over a 28-day tidal cycle, in 15-minuite increments.

[11] In 2019 WDC estimated the cost to be about $800,000, and there have been considerable cost escalation since this time.

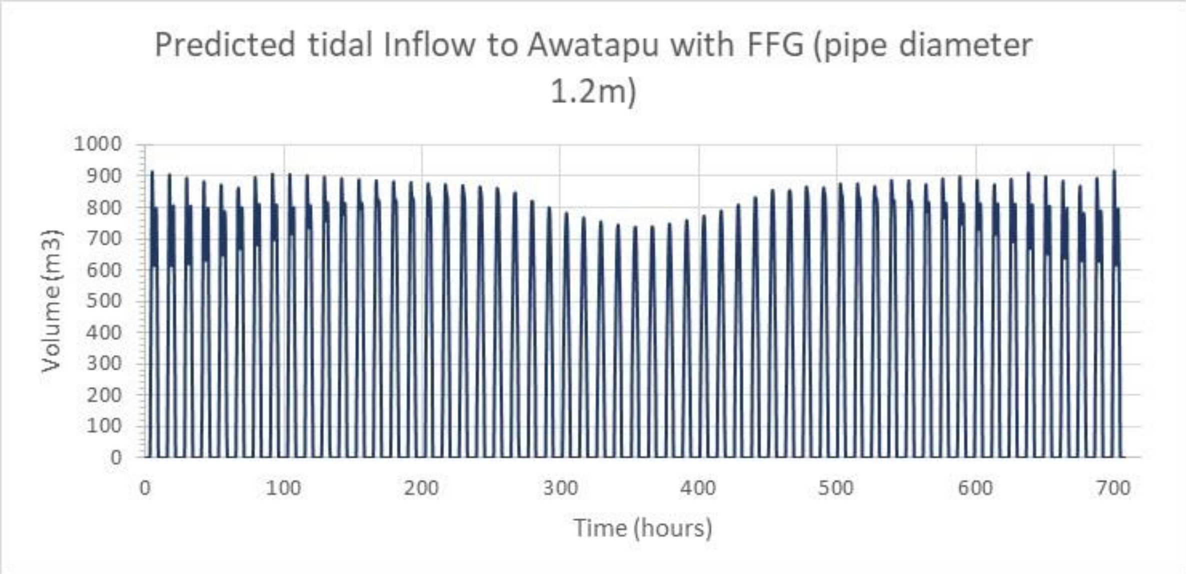


Figure 4.2: Predicted inflow of water to Awatapu through a 1.2m diameter culvert with fish-friendly flap gate (lower graph). In this scenario, the amount of water able to flow into the lagoon is restricted by the FFG closing during a high spring tide.

4.3.5 Diverting flow from Whakatāne River using either a pump or syphon

An alternative way to get flow back into Awatapu Lagoon from the Whakatāne River is to pipe water over the stop bank using either a pipe or a syphon. Both options would require constructing a river inlet structure, conveying water to the base of the stop bank by either pipe or an open channel, an inlet sump, a pipe over the stop bank, and an outlet structure in southern Awatapu Lagoon. For both options the inlet and outlet of the pipe over the stop bank will need to stay continually submerged to allow syphoning and improve pump efficiency.

Syphoning water over the stop bank has become a practical option of Awatapu with development of syphon manifolds that remove airlocks. Syphons create a vacuum inside a pipe that allows gravity to drain water to a lower elevation by going up and over an obstacle. The Water Siphon system (www.watersiphon.co.nz) removes air locks with a hydraulic vacuum primer pump run off main pressure to allow syphoning over obstacles up to 8m high. Non-return valves prevent backflow of water. To syphon over the stop bank at the southern lagoon would require about 7m of lift.

The tidal model was used to estimate flow through a syphon using the Hazen-Williams equation, and adding a module to adjust the head for water level increase in the southern lagoon. The model estimated that a syphon could provide a monthly average flow of about 74 L/s, 170 L/s, 220 L/s and 340 L/s assuming an internal pipe diameter of 381mm, 530mm, 600mm and 750mm respectively[12]. These inflows would cause tidal fluctuations in the southern lagoon of about 12mm, 69mm, 120mm and 268mm respectively, and the elevated water level reduces the syphon efficiency by about 1%, 6%, 10% and 24% respectively.

[12] Peak flow rates of 260 L/s, 600 L/s, 800 L/s and 1320 L/s respectively.

Benefits of a pump compared to a syphon include: more control over the flow rates, smaller pipe sizes required to for equivalent flows, potentially lower peak flows for the same average flow, and efficient energy use can still be achieved by utilising a syphon effect in the pipe. However, a pump requires more maintenance and running costs, requires an intake screen to prevent entrainment of debris or fish, and will require either a pipe or a deep channel (below low tide level) to get water to the base of the stop bank.

Benefits of a syphon system over a pump include it being: a robust, low maintenance system, low operational cost (just mains pressure water to assist with priming), it would be reasonably fish friendly[13], and the inlet channel from river to the base of the stop bank is more easily built and less prone to siltation. The channel can be set at a gradient towards the river (e.g. 0.5% grade) as it only needs to flow when water levels are above mid-tide (i.e. > *c*. 0m RL). The lower end of the channel could be integrated with wetlands and riparian planting for inanga spawning habitat.

In 2019 WDC estimated the cost of pumping 130 L/s of water to the southern lagoon as $280,00, but this would now be close to about $340,000. A syphon system for an equivalent average flow is expected to cost less and have lower maintenance and running costs. If implementing a syphon system it would probably be most cost-efficient to use pipe size of 530mm to 600mm which would allow average flows of 170 L/s to 220 L/s.

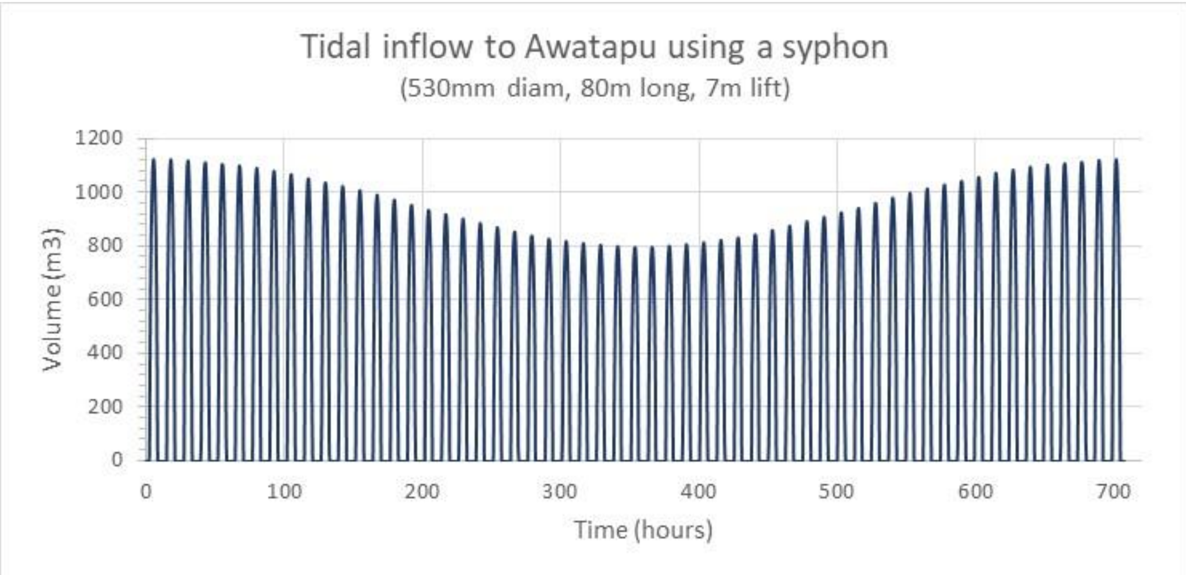


Figure 4.3: Predicted inflow of water to Awatapu through a 381mm diameter syphon during a 28-day tidal cycle.

[13] A syphon to Awatapu lagoon is unlikely to result in significant barotrauma to fish because the ratio of pressure change is moderate (about 1.7) and the maximum rate of pressure change (about 1.5s / m lift) avoids rapid decompression that occur is some hydropower turbines. Salmonids and most NZ freshwater native fish (including eel, bully and galaxiids) are physostomous (i.e. have a pneumatic duct between their swim bladder and stomach), which helps them adjust to changes in pressure.

4.3.6 Implementing by diverting flow from Wainui Te Whara Stream to Awatapu South

Water from the Wainui Te Whara Stream could be forced to enter Awatapu South before flowing through the rest of the lagoon. This could be achieved by:

- Installing a second culvert under the eastern end of Bridge Street;
- Extending the peninsula at the Wainui Te Whara delta to force the stream to flow through the new culvert, and
- Forming a peninsular in the southern lagoon to help circulate the water through the southern lagoon before flowing out the current culvert. A considerably cheaper alternative to building a peninsular would be to use floating silt curtains to prevent bypass flow. This could be used in combination with floating wetlands.

This option would reduce the residence time in the southern lagoon, but would increase the residence time in the central and western lagoons. When the Wainui Te Whara is at median flow (70 L/s), the residence time in Awatapu South would reduce to 7-days (42,400 m^3/ 6048 m^3/day), while the residence time in Awatapu Central/West would <u>increase</u> to 35-days (209,200 m^3 / 6048 m^3/day). The median flow during summer is about half of the annual median, resulting in a hydraulic residence time in the southern lagoon being closer to about 14-days.

A new culvert thrust under Bridge Street is unlikely to be sufficiently large to allow flood flows due to cost and practical constraints. This will require the peninsular to have a low point to act as a broad crest weir for bypass flows. Constraining the size of the culvert will also restrict flood flows entering the southern lagoon, and it is unlikely that water velocity will be sufficiently large to cause much movement of floating rafts of macrophytes. Management of macrophyte cover will still likely be required.

The diversion of flow from the Wainui Te Whara to the Awatapu South lagoon was considered in a concept plan for Awatapu Lagoon South. If this option is undertaken, it would be important to size the pipe sufficiently large so as to ensure fish passage can be maintained for most flows. This requires keeping the water velocity through the culvert less than the sustained swimming speed of fish required to pass, for inanga this is typically less than 0.24 m/s. To ensure fish passage during winter baseflows of about 120 L/s would require a circular culvert diameter of about 0.8m.

If a portion of water was to be diverted from the Whakatāne River than the marginal benefits of diverting the Wainui Te Whara Stream into the southern lagoon would reduce.

4.3.7 Summary

There is potential to divert river flow into Awatapu Lagoon so as to sufficiently reduce residence times so as to improve water quality. An enhanced syphon system may offer a robust and cost-effective option to provide sufficient flow at a reasonable cost. A syphon with size of 600mm could transfer an average of about 220 L/s over a monthly tidal cycle. This volume would substantially reduce the residence time in all of Awatapu Lagoon. Flow augmentation could also be applied in conjunction with installing treatment wetlands that will help remove incoming nutrients.

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) [14].

- 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,[15] 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.

[14] 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.
[15] 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. Wetlands are more effective at P removal when the P is mostly in particulate form and the sediment-bound P is removed in the sediment forebay.

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)[16].

4.4.4 Application to Awatapu Lagoon

4.4.4.1 Suitability

There is considerable potential to build in-lake wetlands in Awatapu Lagoon that would provide multiple benefits of reducing sediment accumulation, improved water quality, improved biodiversity and habitat for invertebrates, fish and birds. The most cost-effective location is near the delta of the Wainui Te Whara which is already shallow.

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

AR & Associates (2020) developed a wetland concept for the southern lagoon, including directing the Wainui Te Whara to flow via the southern lagoon by drilling a separate culvert under Bridge Street and forming peninsulas to direct the flow in the delta area and in the southern lagoon. This concept required a large amount of infill and left little open water remaining in the southern lagoon. We have refined this concept to retain more open water in the southern lagoon and to achieve more lake wetlands for less cost. This is achieved by having less infill of the southern lagoon and more wetlands near the Wainui Te Whara delta where the lagoon is shallower (**Figure 4.4**).

Key features of the proposed design are:

- Creating 1.34ha of wetland area to provide biodiversity and water treatment. This consists of about 0.95ha in the lagoon south and 0.39ha in the Wainui Te Whara entrance of the central lagoon. It would result in wetlands covering about 35% of the southern lagoon area, which is significant.
- Forming islands to support bird habitat and increase biodiversity at the land-water interface.
- Ensuring gentle (flat) gradients to allow wetland plants to expand down to their natural depth range.
- Retaining adjacent deep-water zones to facilitate the deposition of sediment and improve water clarity.
- Diverting water from the Wainui Te Whara to the Southern Lagoon using a peninsula and additional culverts under Bridge Street. Any culverts and channels will need to ensure fish passage in all seasons.
- Forming a peninsular in the southern lagoon to direct water from the Wainui Te Whara along the southern edge and set up a circulation current.
- Using sand deposited in the delta from the Wainui Te Whara to provide material required for the wetland and peninsula.

If water flow was to be diverted from the Whakatāne River to the southern lagoon (as discussed in previous section), then the marginal benefit of diverting the Wainui Te Whara into the southern lagoon would be considerably less. In this circumstance, the length of the peninsula and extent of the wetland in the southern lagoon should be reduced from what is shown in **Figure 4.4**, as flow would be coming from the Whakatāne River.

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 planting during summer when water levels are lower, and/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. Retaining deep zones are also important to facilitate the deposition of sediments and reduce the wind suspension of sediment.

Additional water quality benefits might be achieved by removing organic sediment from adjacent deepwater areas and using it as the top layer of infilled for terrestrial areas. The organic muds can be a source of sediment oxygen demand and of nutrients, so their surface area exposed to lagoon water should be minimised.

There are a number of native emergent plants suitable for Awatapu Lagoon including: *Eleocaris sphacelate, Machaerina articulata*[17], *Carex secta,* and *Schoenoplectus tabernaemontani*. *Typha orientalis* (raupō) could be considered but would need care to ensure it is contained by surrounding deep zones (F**igure 4.5**)

4.4.4.2 Cost

Establishing wetland adjacent to the Wainui Te Whara delta could be cost-effective way to remove nutrients and enhance ecological values. Priority should be given to creating wetlands in the shallow area near the Wainui Te Whara delta.

Wetland creation is much more expensive in areas where the water is already deep and more fill is required (e.g. much of the southern lagoon); in these situations, it may be more cost-effective to use floating wetlands, or use harvesting of macrophytes as a more cost-effective way to remove nutrient load.

4.4.5 Summary

Treatment wetlands are a common and often 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 Awatapu Lagoon to treat inflows and improve biodiversity values. The creation of wetlands in the lagoon should first occur in area of currently shallow water (e.g. the Wainui Te Whara delta) where they will be most cost-effective.

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[17] Formally *Baumea articulata.*

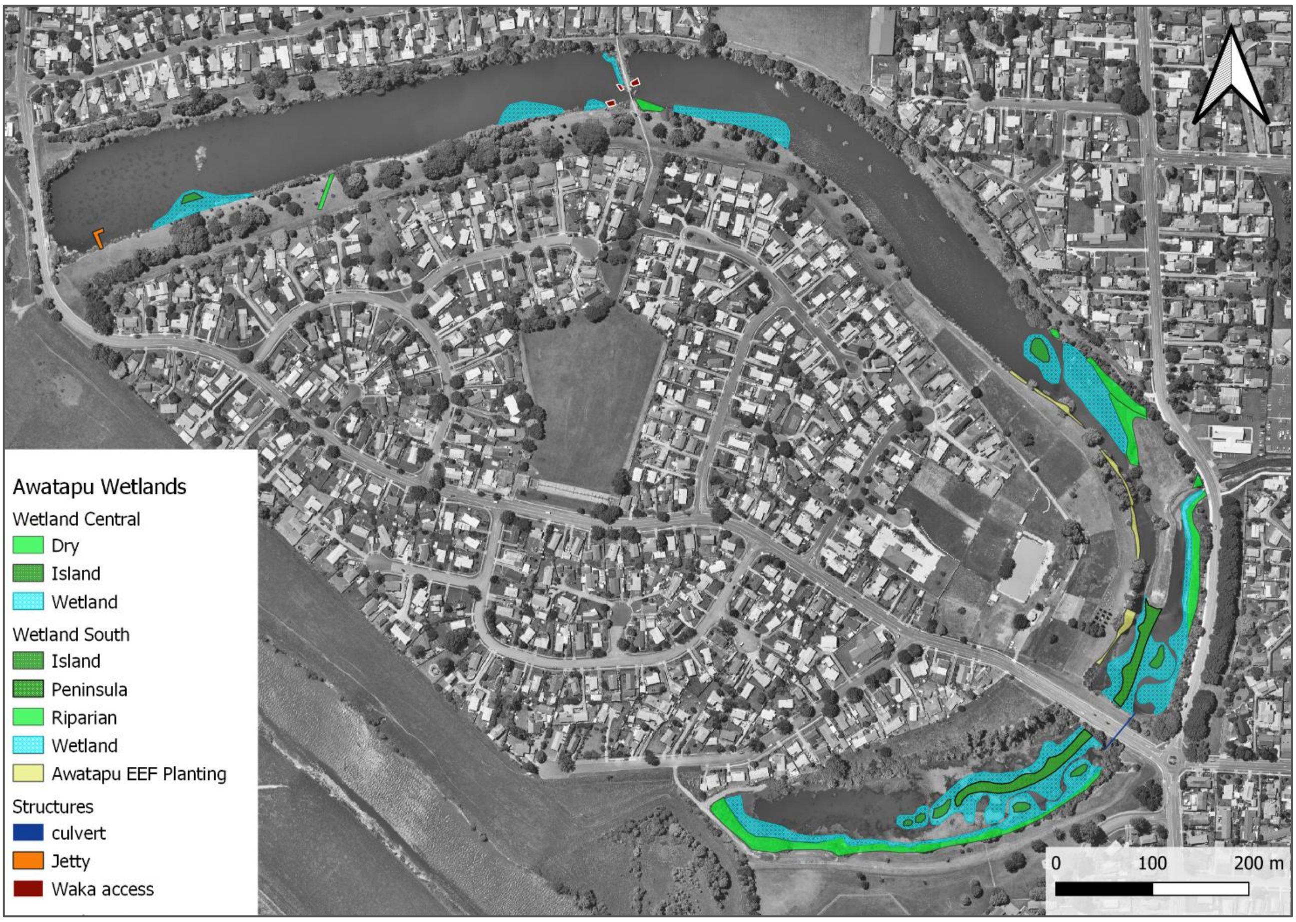

Figure 4.4 Potential layout for a treatment wetland and riparian wetlands in Awatapu Lagoon to improve water quality and provide biodiversity values.

Figure 4.5: 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 wetlands are widely used around New Zealand for water treatment and ecological enhancement. To be most effective for nutrient removal 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. >1.2 m) where the plant root systems will not reach the sediment (unless it is intended to partially block the flow).

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. This can be avoided by using rafts made of more stable material (e.g. HDPE) or natural materials (**Figure 4.6**).

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[18] 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.

[18] Annualised cost spread over 50 years.

Because of their relatively high cost, floating wetlands are better suited to situations that optimise their 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 Awatapu Lagoon

4.5.4.1 Suitability

Additional floating wetlands could be installed in Awatapu Lagoon near the stormwater inflows and on the main body of the lagoon to provide multiple benefits for nutrient removal and habitat for fish and birds.

If the wetland concept of diverting flow from the Wainui Te Whara to the southern lagoon is adopted, then floating wetlands, in association with floating silt-curtains, could be a cost-effective way to direct water flow and minimise the extent of infill required to create a peninsula.

Floating wetlands may be a cost-effective way to create wetlands in the southern lagoon due to the extensive infill required to create a large area of wetland where the water is deep.

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. They are typically more expensive than creating surface flow wetlands, but may be a cost-effective and flexible alternative in the southern lagoon where a large amount of infill is required.

Figure 4.6: An AST style floating wetland being deployed in Awatapu Lagoon in 2019.

4.6 Phosphorus Locking

4.6.1 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 it is very efficient and has 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 remove 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).

4.6.2 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 Awatapu Lagoon are alum, Phoslock and Aqual-P. PAC is a flocculant and can be used in combination with alum or Phoslock which absorb the phosphorus. For the immediate management of cyanobacteria blooms, algaecide (e.g. hydrogen peroxide) can be used before phosphorus locking to reduce the risk of algae 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.

4.6.3 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.

4.6.4 Application to Awatapu Lagoon

4.6.4.1 Suitability

Awatapu Lagoon is nitrogen limited but reducing phosphorus concentrations is important for controlling cyanobacteria. The source of phosphorus in Awatapu is via localised areas of anoxic waters. DRP may also be released with wind-induced mixing, mobilising porewater from bottom sediments.

Phosphorus locking (probably with alum) may be a useful remediation to apply in specific areas of Awatapu where bottom waters are commonly anoxic e.g. Awatapu West and in the deep section of Awatapu Central.

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 or plant material. 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 Awatapu Lagoon 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, and the suitability of specific produces in brackish water. 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.

4.6.5 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 Awatapu Lagoon. Additional investigations or trials are required to better determine suitable products and likely success. P-locking will be more successful if implemented in conjunction with actions to reduce the deposition of sediment and plant material (e.g. macrophyte harvesting and forming treatment wetlands).

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

4.7 Macrophyte harvesting to manage aquatic plants and reduce nutrients

4.7.1 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 plant 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 4.6**). 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 may in 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 4.6: A lake macrophyte harvester in operation on Lake Rotoehu (source: www.lakeweed.co.nz).

4.7.2 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**)[19].

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 parrot's feather (*Myriophyllum aquaticum*) both found in Awatapu Lagoon.

4.7.3 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 2,400 kg N /yr and 320 kg P/yr (Hamilton and Dada 2016), i.e. a cost-effectiveness of $22 / kg N and $166 /kg P. However, the cost of small-scale operations is considerably more.

Weed harvesting from Awatapu Lagoon South, Whakatāne in 2019 cost c. $48,000 for about 200 tonnes of weed which would have removed about 240 kg of N and 32kg of P with a cost-effectiveness of $200 /kg N and 1500 / kg P. This cost included consenting, establishment, harvesting, dewatering and disposal. Future costs would be less as about $20,000 was spent on site preparation.

4.7.4 Application to Awatapu Lagoon

4.7.4.1 Suitability

Harvesting of hornwort has been proven to work in Awatapu Lagoon and is very effective at reducing macrophyte cover. 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. However, herbicide might be used in alternative years to keep biomass low.

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

4.7.4.2 Cost

The cost of macrophyte harvesting can vary widely depending on the scale of the operation. The cost of macrophyte harvesting in Awatapu Lagoon South and part of Awatapu Central is expected to be in the range of $80,000 per harvest, but will depend on the amount to be harvested. The cost is likely to reduce following establishment of initial site access and as operations become more efficient, e.g. management of harvested material on-site or nearby.

4.7.5 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 lakes. It has a high certainty of being effective at addressing these issues in Awatapu Lagoon. Harvesting is more expensive than herbicide spray, but provides water quality benefits not achieved by herbicide spraying. It could be used in conjunction with herbicide spraying in alternative years to control regrowth.

4.8 Summary: Actions to improve water quality and ecology

Intervention options to improve water quality in Awatapu Lagoon are summarised in **Table 4.2**. There is no single quick fix to improving water quality in lakes and no “magic bullet”. There are however effective actions that can shift Awatapu Lagoon towards being a healthier ecosystem. The path towards sustainable improvement in lake water quality requires reducing both external and internal nutrient loads, and improving the function and diversity of aquatic habitat. Highest priority should be given to actions that would address multiple issues in a cost-effective way, and with low risk of adverse effects.

The management actions with most potential to improve water quality and ecology in Awatapu Lagoon are:

1 Harvesting and control of aquatic pest macrophytes. This is a priority to improve the DO regime, reduce organic matter load to lake sediments and reduce nutrients. It is also a priority for maintaining recreational use of the lagoon.

2 Construction of wetlands adjacent to and within the open water. These provide multiple benefits in removing nutrients, providing habitat for aquatic life and increasing biodiversity values. The most cost-effective place to locate constructed wetlands is near the shallow delta of the Wainui Te Whara Stream.

3 Phosphorus locking of sediments in deeper basins. This has considerable potential to reduce the internal load of phosphorus from anoxic bottom waters. However, it needs to be undertaken in conjunction with actions to reduce the organic matter load from hornwort.

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

- Increasing the number of floating wetlands to remove nutrients and improve biodiversity. Floating wetlands and silt curtains could be a cost-effective way to direct flow of any diverted water without infilling the lagoon.

- Increasing water flow from Whakatāne River through Awatapu Lagoon. A cost-effective option would be to harness the differential head caused by tidal fluctuations to drive a flow of water from the Whakatāne River to Awatapu South via a syphon system. This would could flushing, and reduce phytoplankton biomass throughout the lagoon, but would likely be insufficient to reduce excessive macrophyte growth. Further investigation is required in consultation with BOPRC.

- There is potential to reducing catchment sediment and nutrient loads to Awatapu Lagoon. One option for further investigation is to use detainment bunds within the upper Wainui Te Whara catchment.

Attestation should also be given to maintaining existing wetlands and biodiversity values. This should include control of *Glyceria maxima* that is spreading amongst raupo on the margins of the lagoon.

Table 4.2: Summary of intervention options to address ecological and water quality issues in Awatapu Lagoon

Priority	Management Option	Description	Effectiveness in Awatapu	Limitations
High	Harvest hornwort /parrot's feather	Harvest hornwort during summer to improve oxygen regime, remove nutrients, improve public use. Herbicide could be considered in alternative years to maintain low cover, but could exacerbate water quality issues if used as the sole method of control.	WQ - High Weeds - High (but short term). Habitat - Moderate	Requires ongoing effort. Access to equipment limited at peak times.
High	Create surface flow wetlands	Construct treatment wetlands near the Wainui Te Whara delta and central lagoon and southern lagoon. Option to include a diversion of the Wainui Te Whara Stream into Awatapu South.	WQ - High Habitat - High	Requires a large area. Moderate to high capital cost. Good design important to ensure nutrient removal and minimise maintenance. Cost are less where the lagoon is shallow (eg near the delta). If infilling deep area then floating wetlands may be more cost-effective.
High / 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. Important to first reduced organic deposition form macrophytes. pH conditions are critical. Can be culturally sensitive.
Moderate	Reduce catchment sediment and nutrient loads	Reduce external nutrient loads including: - Sediment Detention Bunds in the upper catchment of Wainui Te Whara Stream. - P-socks at culvert outlets to bind P.	WQ - High (address root causes)	Investigations required to better assess feasibility. A social challenge to achieve changes in land use or land management.
Moderate	Floating wetlands	Install additional floating wetlands to remove nutrients and provide habitat.	WQ - High Habitat - High	Can be costly compared to surface flow wetlands. Best suited to near inflows, areas with high nutrient concentrations, or where site constrains make surface flow wetlands less cost-effective.
Moderate	Increase flushing flows through Awatapu Lagoon from the Whakatāne River.	Increase flushing flows through Awatapu by directing a proportion of flow from the Whakatāne River to Awatapu Lagoon South using either a gravity culverts with flap gates, or a syphon with manifold to avoid airlocks.	WQ- Moderate Cultural benefit.	Requires a pipe either over or through the stopbank. Requires further discussion with BOPRC and iwi.

5 Conclusions and Recommendations

5.1 Conclusion

The water quality in Awatapu Lagoon is poor with low water clarity, high nutrient concentrations and high phytoplankton growth indicative of eutrophic to supertrophic conditions. The lake has internal loading of nutrients from the sediment via bottom water anoxia during periods of stratification. Aquatic pest-plants often form floating mats that result in low dissolved oxygen, and compromises values for aquatic life and recreation.

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 Awatapu Lagoon over the long term will require multiple actions over a sustained period to reduce nutrient loads (internal and external) and enhance natural processes that attenuate nutrients. Reducing the biomass of hornwort is a high priority for many water quality issues in Awatapu. However, maintaining some aquatic plants is also important for maintaining reasonable water quality in small natural lakes.

5.2 Future monitoring and investigations

Water quality monitoring of Awatapu Lagoon has been limited in recent years. While we have useful information about the current state and issues affecting Awatapu Lagoon, 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 (e.g. release of phosphorus from anoxic zones, or the extent of pest plant cover).

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 lake's 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 Awatapu Lagoon and understanding the success of any mitigation should include:

- 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.
- Undertake a bathymetry survey of Awatapu Lagoon to more accurately calculate residence time and to better estimate the cost of creating treatment wetlands in the lagoon.

- Monitoring water quality of main stormwater inflows to Awatapu Lagoon during rainfall to characterise the quality and contribution of stormwater entering the lake (including an estimate of flow from the culvert).

- Repeat synoptic surveys of DO and pH following macrophyte harvest / control operations.

- Water quality monitoring of the lake surface water with a minimum frequency of bi-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 spring/early summer to assess DO depletion in bottom waters following the start of stratification.

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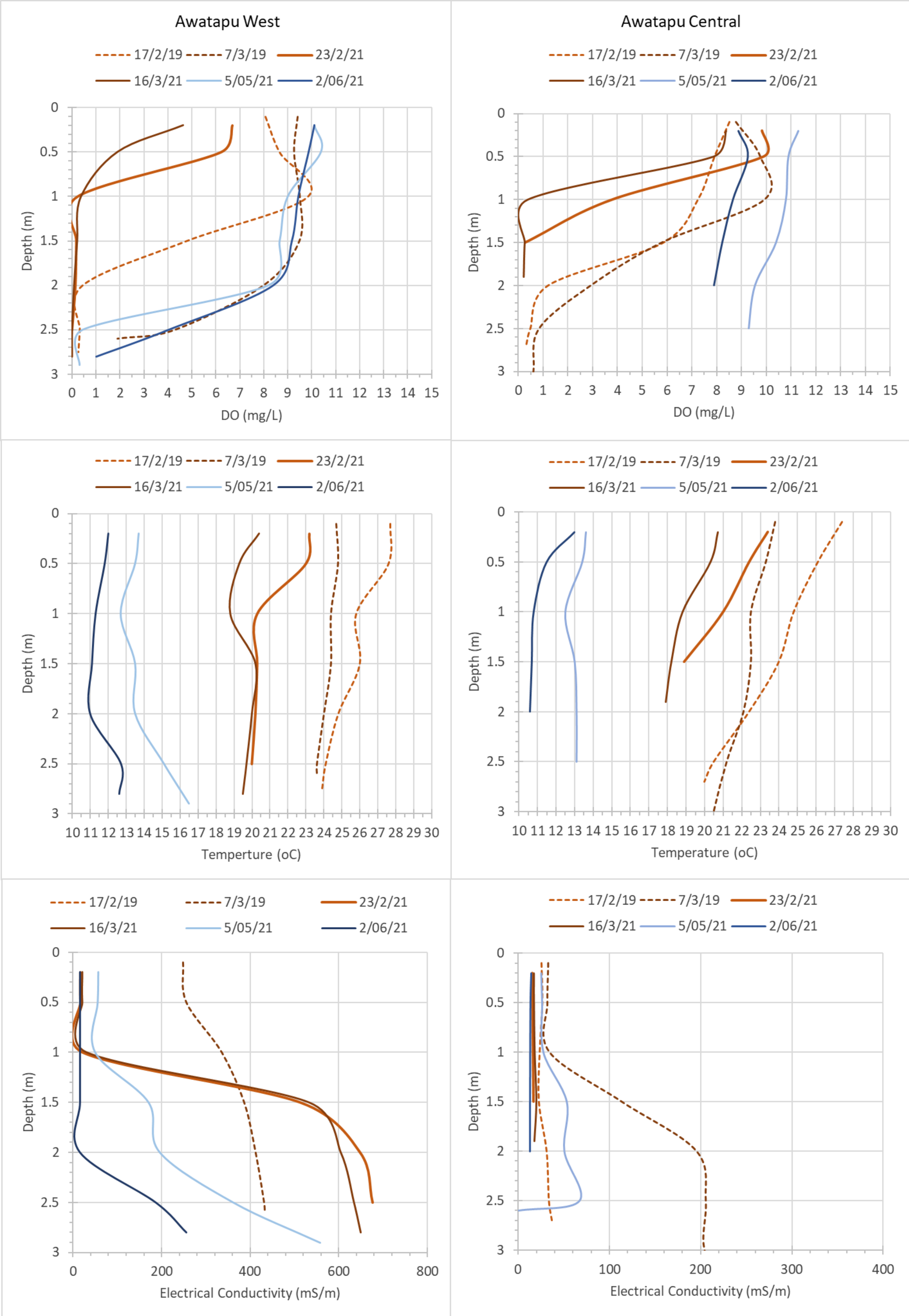
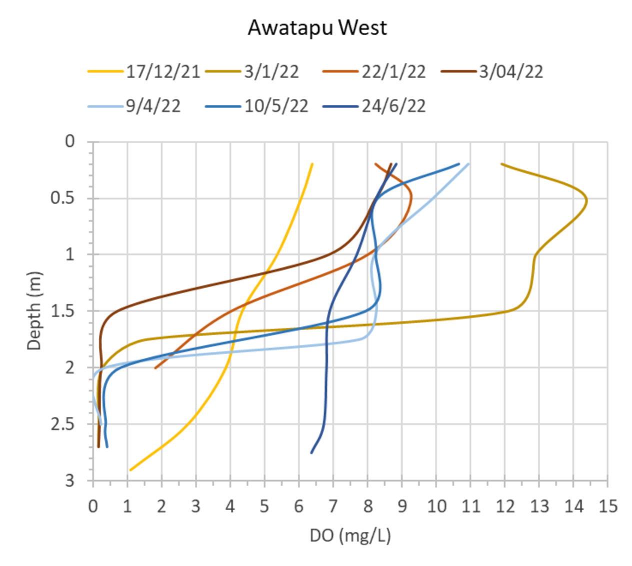
Appendix 1: Temperature, dissolved oxygen, conductivity and depth profiles for Awatapu Lagoon

The graphs below show changes in water temperature, dissolved oxygen and electrical conductivity with depth from the water surface. During summer, the dissolved oxygen was often low or anoxic in the bottom waters below about 1.5m (range 1m to 2m). This is associated with both salinity stratification and thermal stratification. The higher density of salt water compared to freshwater makes salinity stratification (and its associated halocline) stronger and more stable than thermal stratification. Some events of thermal stratification measured in the profiles may be diurnal, as diurnal temperature fluctuations of about 3^{o}C are common during summer.

During summer, thermal stratification and salinity stratification reinforce each other to separate bottom water from mixing with surface waters, however even when surface water temperatures cool during late autumn / winter the bottom waters are slow to mix while the salinity stratification persists (e.g. April and May 2022).


Awatapu West
Awatapu Central
17/2/19
7/3/19
23/2/21
16/3/21
5/05/21
2/06/21
Depth (m)
DO (mg/L)
Temperture (oC)
Temperature (oC)
Electrical Conductivity (mS/m)


Awatapu West
17/12/21
3/1/22
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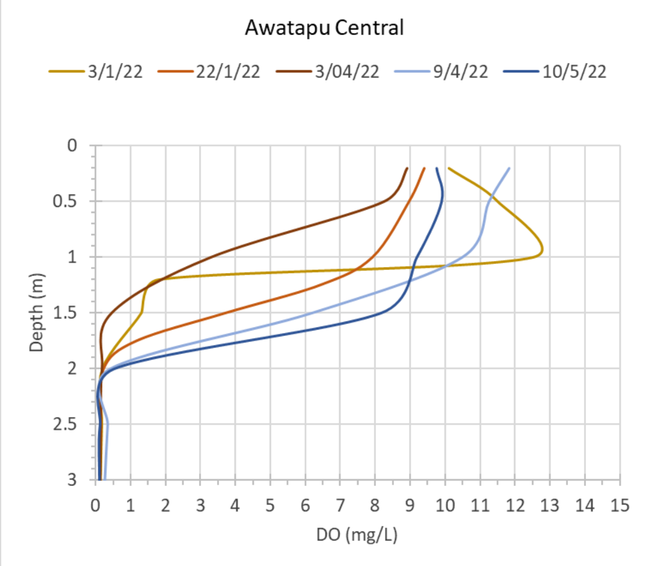
3/04/22
9/4/22
10/5/22
24/6/22
Depth (m)
DO (mg/L)


Awatapu Central
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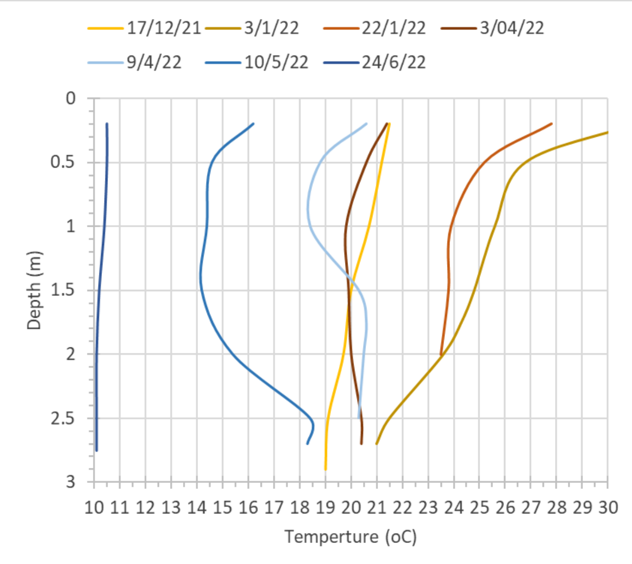
DO (mg/L)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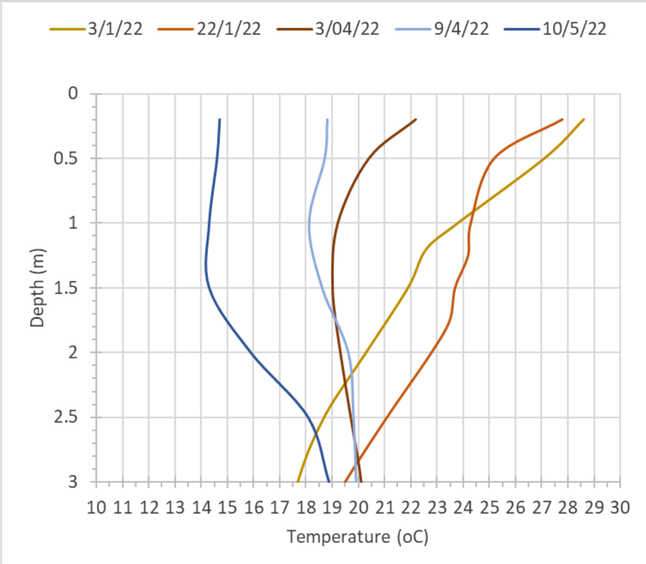
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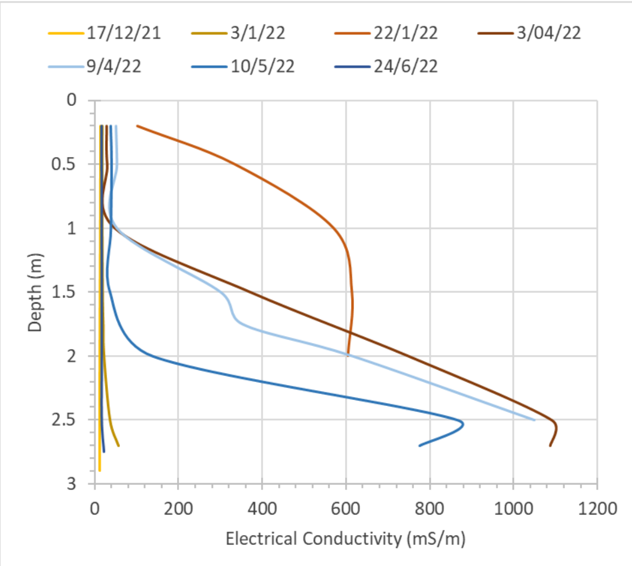
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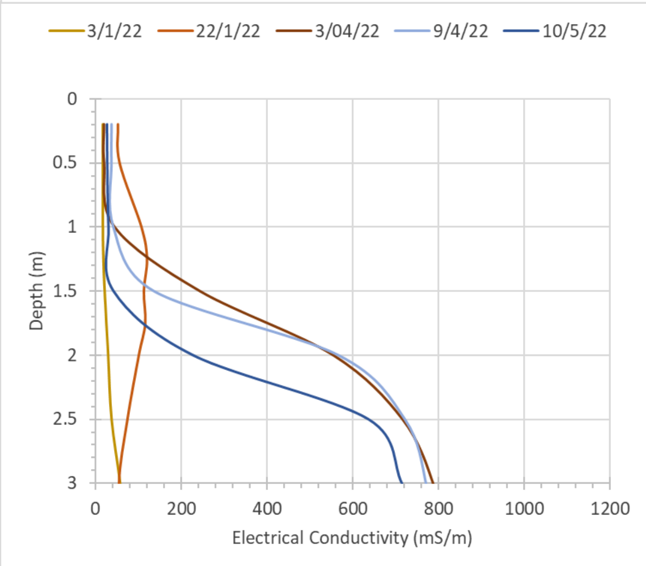
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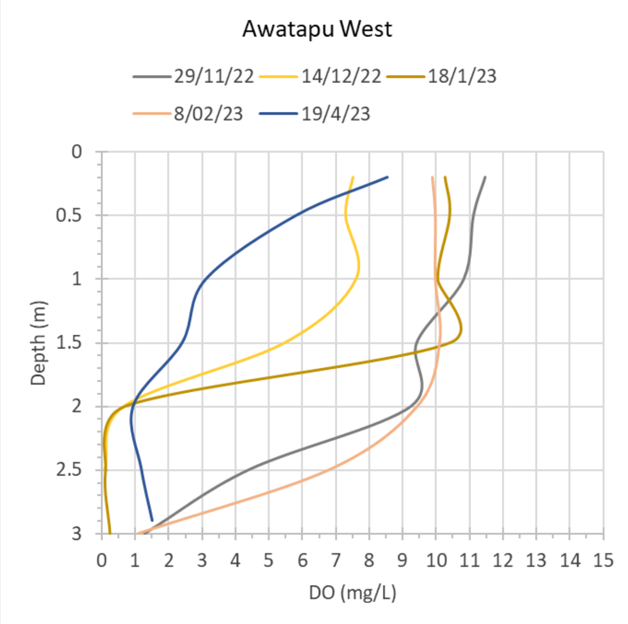
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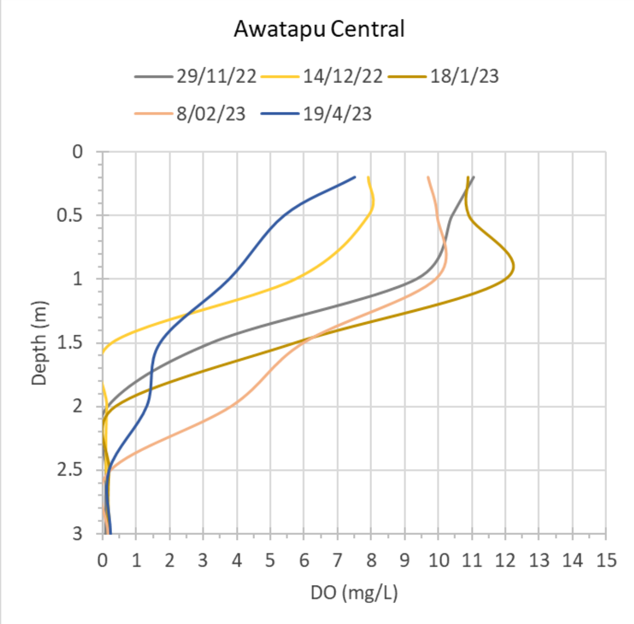
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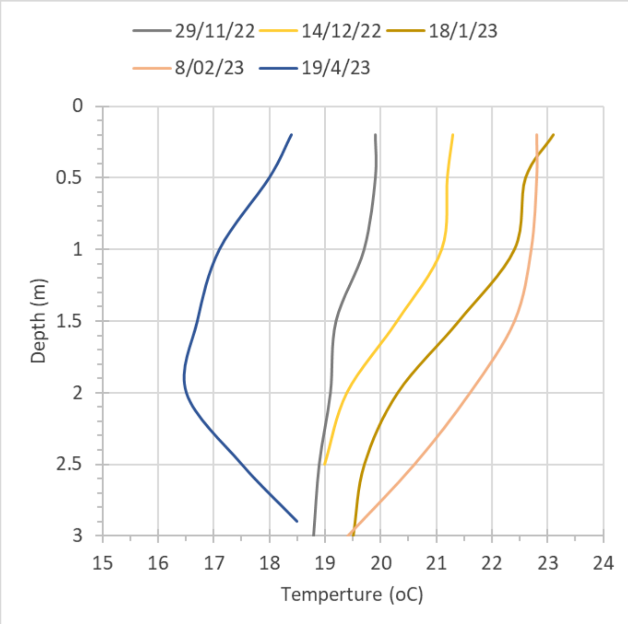
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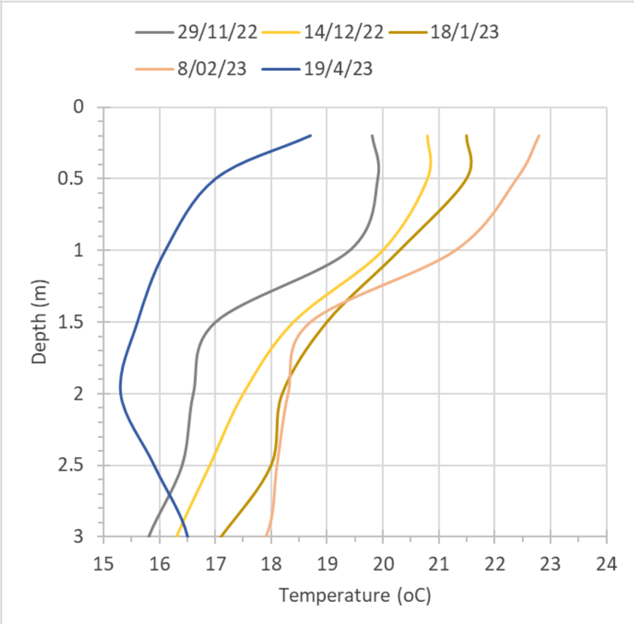
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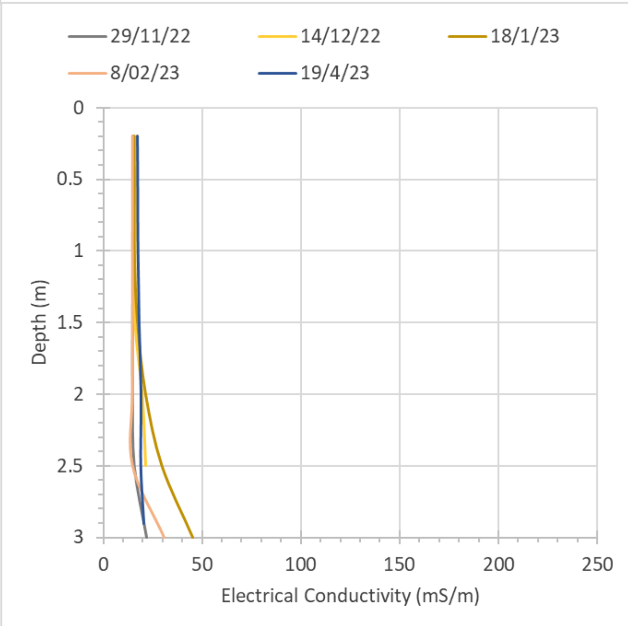
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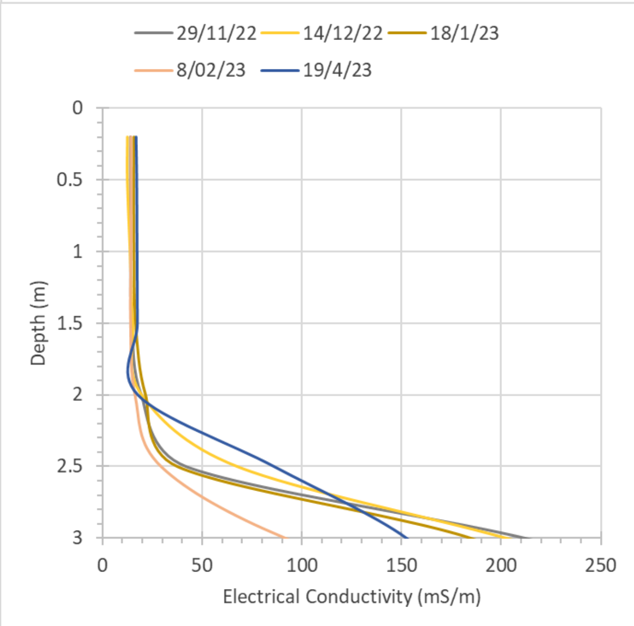
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Depth (m)
Electrical Conductivity (mS/m)

Appendix 2: 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); Bums 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.