NIWA
Taihoro Nukurangi

Lake Horowhenua review

horizons
regional council

Assessment of opportunities to address water quality issues in Lake Horowhenua

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Prepared for:

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Cover photo: Monitoring Lake Horowhenua February 1989. Team from Taupo Research Lab., DSIR comprised Dr Eddie White (foreground), Stu Pickmere (in lake), George Payne (lake edge) and Max Gibbs (taking photo).

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Reviewed by
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James Sukias

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David Roper

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Executive summary

Lakes provide humankind with many services: aesthetic enjoyment, recreation, fish, transportation, water for irrigation and drinking. These services are impaired by exploitation of the lands of their catchments and the lakes themselves for dilution of pollutants and waste water. The goal of management is to balance the uses of lakes with conservation measures to sustain ecosystem services over time. Research can provide understanding of lakes, their catchments and the mechanisms that sustain ecosystem services and the causes of lake degradation. The focus is now on developing methods and technologies for lake restoration/rehabilitation. Each restoration project requires some level of new site-specific research, and remediation may require management actions which are difficult to implement for social or institutional reasons.

Lake Horowhenua is a shallow, hypertrophic lake which has been degraded over many years and is in need of remedial actions to improve its water quality and use for contact recreation, and to restore the lake fishery. This Ministry of Science and Innovation Envirolink report provides a review of the available information on the lake and an assessment of opportunities to address water quality issues in Lake Horowhenua. The report includes a summary of current thinking for a range of remedial actions that could be used on Lake Horowhenua and, based on this information, example management strategies are developed that, in combination, could be the basis for the rehabilitation of the lake and its fishery.

Lake Horowhenua is a coastal dune lake at the southern end of the iron sand deposits on the west coast of the North Island of New Zealand. The lake has a single outflow to the sea, the Hokio Stream, and receives water from several small streams, drains and groundwater; groundwater accounts for more than half the inflow. The lake has a mean annual hydraulic residence time of about 47 days ranging from <33 in winter to >95 in summer. Until 1987 the lake also received the treated sewage effluent from the town of Levin. Nutrients from 25 years of sewage discharge into the lake have accumulated in the lake sediment and are a major cause of the present hypertrophic condition.

The lake has high nitrogen (N) but low phosphorus (P) concentrations in winter. The N is associated with leaching of fertilizer from the horticulture, market gardening and intensive dairy farming in the catchment. These land-use practices also result in high nitrogen (N) concentrations in the groundwater. Notwithstanding the high N concentrations in the inputs, in summer the lake water column has low N but high P concentrations, the latter sufficient to favour the development of massive cyanobacteria (blue-green algae) blooms. The lake also has a lake weed problem with tall stands of *Potamogeton crispus*, which are the key to the low N and high P concentrations. Through summer the growth of *Potamogeton* reaches the lake surface interfering with contact recreation such as rowing and sailing.

This report summarises the water quality of Lake Horowhenua which is currently very poor and is declining due to increasing nutrient and sediment loads from the catchment.

The data review determined that the inflows were the major source of N and that the N concentrations in all inflows were increasing. Of concern, N concentrations in the largest inflow, the Arawhata Stream, have slowly increased from a mean of around 10,500 mg m^{-3} in 1989 to more than 13,600 mg m^{-3} in 2008. This appears to be directly linked to recent dairy intensification in the stream catchment. In contrast, the major source of P to the lake was the sediment, as an internal load. Because the release of P from the sediment requires anoxic conditions, this implies that the lake sediments become anoxic in summer. Being a very shallow lake and easily stirred and aerated by wind waves and boat wakes, bottom water anoxia requires very special conditions. In this case, anoxic conditions develop at the sediment surface when the tall weed beds collapse in late summer and fall onto the lake bed.

The annual nutrient cycle starts in winter with high N water from the streams and groundwater flushing the lake. At this time the lake has relatively low algal biomass but high wind-induced, suspended sediment-driven turbidity. In autumn the weed seeds germinate and the *Potamogeton* uses the N in the lake water for growth, substantially reducing the N concentrations in the lake water column between October and December. The oxygen demand in the sediment increases and small amounts of P are released beginning in December. The major release of P occurs during February after the tall weed beds have collapsed. The collapse and decomposition of the weed on the lake bed reduces or prevents oxygen diffusion reaching the sediment surface and P is released. Daily katabatic winds mix the upper water column and disperses the P throughout the lake. Algal biomass is dominated by cyanobacteria which are buoyant and form surface scums during the morning. Some of this biomass, which includes P, flows out of the lake down the Hokio Stream while the rest is returned to the sediment when the bloom dies in late autumn-winter (July).

The loss of P down the Hokio Stream is the natural recovery process to reduce the P in the lake sediment. Sediment analyses showed that the P load in the sediment reduced by about 47% between 1989 and 2011. Because the recovery process is exponential, at this rate it is likely to take about 100 years for the P load to reach 5% of the original load, assuming no further inputs of P occur.

About 80% of the external P load on the lake from the catchment is a single point source: the Queen Street drain.

After water quality, the single most important factor affecting the fishery in Lake Horowhenua is the weir on the Hokio Stream. The weir controls the lake at a near constant level of 30 feet (9.1 m) above mean low water spring tides at Foxton Heads, as a requirement of the Reserves and Other Lands Disposal Act, 1956. This level means there is a lake all year round but the lake is a very large settling pond with about half of its original volume filled with sediment. The source of this sediment was from sewage discharge, runoff from pasture and dairy sheds, and land-use practices which allowed drainage ditches to be dug through wetlands and swamps, and cows and pigs having direct access to the lake. This sediment has smothered the sands and cobbles of the original lake bed and reduced the habitat for fish. Kākahi cannot support themselves in soft sediment and their numbers dwindle as they drown in the mud. The weir also blocks the passage of the diadromus fish species which spend part of their life cycles in the sea. A fish passage around the weir would allow the return of species such as black flounder, grey mullet and smelt, when the water quality improves.

The lake also has a pest fish problem with goldfish, which damage the marginal plant communities that provide habitat for inanga and koura, and perch which are carnivorous. Predation by juvenile perch on zooplankton may prevent them from controlling the algal biomass in the lake.

Based on current scientific knowledge and recently developed remedial techniques, there are a number of in-lake management strategies that could be used to address water quality issues in Lake Horowhenua. These tools, which are described in detail, include:

- Lake weed control
- Stormwater and groundwater treatment
- Floating treatment wetlands
- Marginal buffer zones
- Storm water diversion
- Arawhata Stream diversion
- Flushing – external source water
- Enhanced flushing using fluctuating water levels
- Dredging
- Aeration using bubblers
- Aeration by Discing
- Aeration by nitrate injection
- Phosphorus inactivation with flocculation
- Phosphorus inactivation with sediment capping

While a combination of these tools can address the internal problems in the lake, there is also an overriding requirement to address the sources of nutrients from the catchment. A comparison of the inflow nutrient loads measured in 1988/89 and between 2000 and 2010 shows a significant increase in N and P loads from the catchment, possibly attributable to dairy intensification. In one stream catchment the mean DRP and TP concentrations have increased by 300%. The source of these nutrients to the lake need to be turned off at the same time in-lake interventions are implemented otherwise the effectiveness of the remediation measures will at best be temporary.

Groundwater contributes more than half the water to the lake and about half of the external N load. The overuse of fertilizers and discharge of nutrient rich water to the land where they can be leached into the groundwater is maintaining the high nutrient concentration in this inflow. Stock have direct access to streams raising sediment and nutrient loads through defecation into the water and disturbance of the stream bed. There are indications that dairy shed runoff and overflow from dairy wastewater ponds may be entering the streams. Management strategies in the catchment need to address these issues and be implemented concurrently with management strategies in the lake.

The high hydraulic conductivity of the gravel aquifers beneath the surface topography in the lake catchment indicates that the maximum groundwater residence time is likely to be about a year. This suggests that catchment remediation strategies are likely to be effective in reducing the nutrient loads discharging into the lake via the groundwater. Conversely, it also indicates that any increase in nutrient loading to the groundwater will rapidly impact on the lake water quality. The intensification of dairy farming in the lake catchment has the potential to exacerbate the already very high nutrient concentrations in the groundwater.

Two case studies are used to illustrate the effectiveness of these tools in lake rehabilitation and pro-active management of cyanobacteria blooms in New Zealand.

The key to the successful rehabilitation of Lake Horowhenua is a good strategic database. Without good data it is not possible to determine how the lake is changing with time until well after the time when it would be possible to made a small intervention to mitigate a developing problem. It is also not possible to differentiate between the effects of natural climatic cycles with periods of several years, e.g., Southern Oscillation Index and shifts between El Niño and La Niña weather patterns, and intensification of dairy farming and market gardening.

The Horizons Regional Council database is fragmented with large gaps in time between episodes of data collection. The data is not collated into a single database and there are numerous errors that may be able to be corrected using regression techniques and limnological knowledge to assess whether a data point is ecologically feasible. A compiled database needs to be established and maintained. Historical chemistry data exists from 1949, the 1970s, 1988to 1991, and from 1999 to 2010. Plant and fish surveys were reported for 1949, 1975 and 2002 but are not discussed in detail in this report.

The development of a management or action plan for the lake follows a well defined pattern and is illustrated in the flow diagram below (extracted from this report). For success, a similar management plan for the lake catchment should also be developed, but is not dicussed in this report.

When the issues are identified and the goals are set, the measures of success are also defined. The measures of success require monitoring programmes to determine whether the goal has been achieved. If the goal is not being achieved, there is an adaptive management loop which uses the monitoring data to adjust the management strategy until the goal is achieved.

Based on the management plan flow diagram, a six-point management strategy was developed as an example of how the rehabilitation of Lake Horowhenua might be achieved:

- Reduce N, P and sediment in surface inflows
- Reduce N from groundwater
- Stop shoreline erosion and mitigate storm water P
- Enhance contact recreation
- Improve fish access to the lake
- Reduce pest fish and increase zooplankton biomass

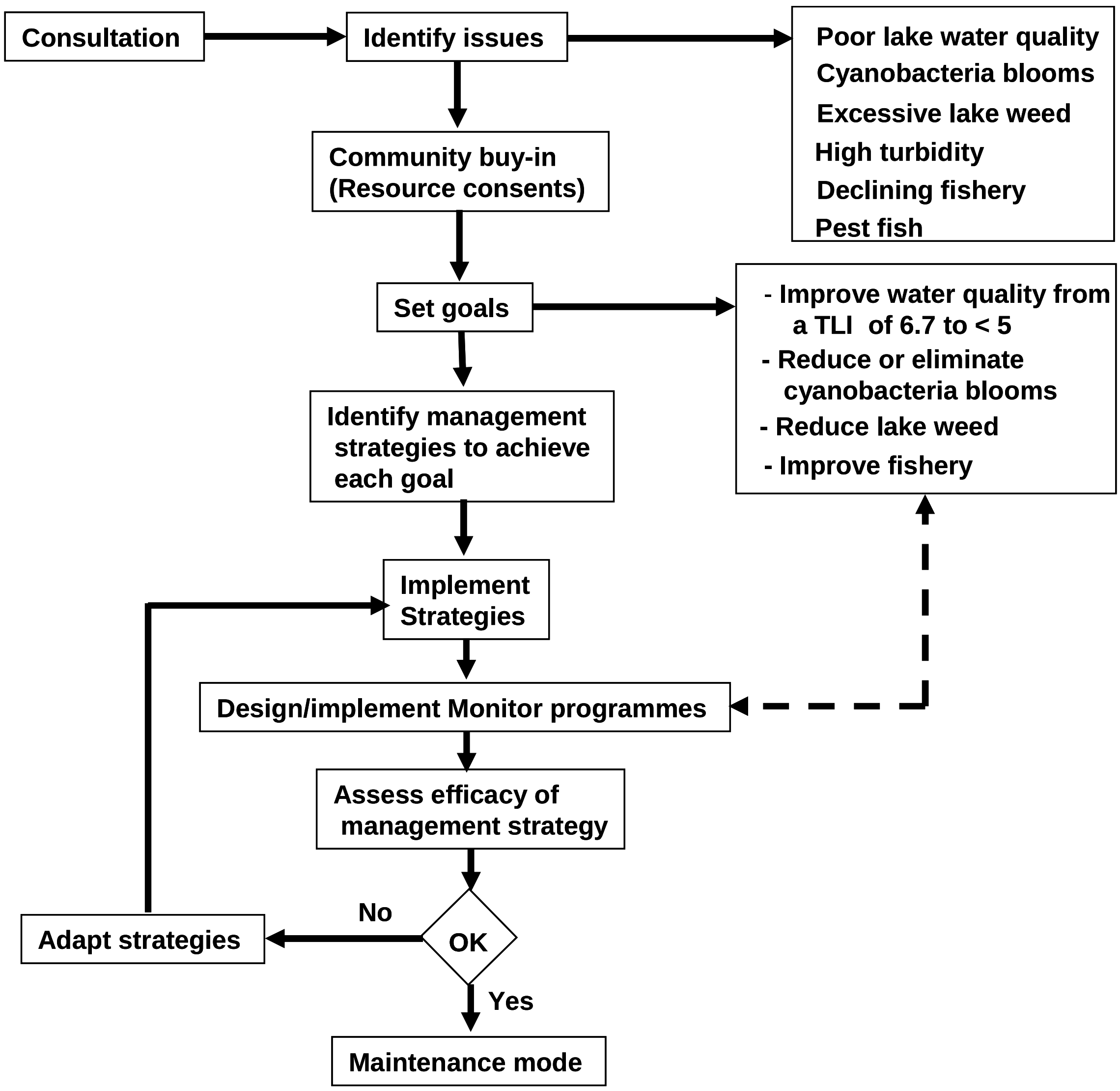
Why these management strategies were chosen and how each strategy is implemented are discussed in detail.

Currently, there is no monitoring programme on Lake Horowhenua. This lack needs to be remedied as soon as practical if lake rehabilitation is to be attempted. There are three types of monitoring required:

Strategic monitoring is required to assess the state of the environment (SOE) and allow nation wide comparisons of lakes throughout New Zealand. Ministry for the Environment information on the status of New Zealand lakes water quality lists Lake Horowhenua at 107 out of the 114 lakes monitored.

Compliance monitoring is required to manage consents.

Success monitoring is required to assess the effectiveness of management strategies implemented to rehabilitate the lake.



1 Introduction

Horizons Regional Council (Horizons) is responsible for monitoring and managing the region's waterways. Water quality is a major focus of Regional Council activities and was one of the "Big Four" issues addressed via the One Plan (Horizons second generation Regional Policy Statement and Regional Plan).

Lake Horowhenua is a shallow coastal lake that is often closed to recreation due to cyanobacteria blooms. The Regional Plan identifies Lake Horowhenua as a "rare and threatened habitat" and sets targets for lake water quality based on the identified values of the lake. Lake Horowhenua water quality is compromised in terms of those targets. Horizons seeks advice on possible combinations of actions that will improve water quality in the context of the desired goals for the lake around ecosystem health (e.g., viable native fish populations) and uses of the lake (e.g., contact recreation).

Currently, national research on lakes is providing information regarding strategies for remediation of lakes where water quality is impaired. This work includes the research on Lake Okaro, Lake Rotorua and Lake Rotoehu. Horizons looks to draw on scientifically defensible information provided by these national programmes to identify options for enhancing water quality in Lake Horowhenua. There is presently no monitoring programme on the lake although there has been sporadic monitoring in the past.

Through this Ministry of Science and Innovation Envirolink project, Horizons has asked the National Institute of Water and Atmospheric Research (NIWA) to draw together information from historic monitoring and research into Lake Horowhenua water quality and other relevant research to identify options for addressing water quality issues in Lake Horowhenua and to provide advice regarding options for monitoring lake water quality.

Scope

The advice sought by Horizons is focused on identifying remedial options, actions required for water quality improvement, and likely lake health and use-related benefits. Potentially the advice could inform or direct:

- actions in the catchment to improve water quality;
- decisions around resource consents;
- decisions around policy development;
- design of lake water quality monitoring programmes, as well as
- remedial actions within the lake.

This Envirolink project will provide a single report that will collate information available for Lake Horowhenua and identify options for rehabilitation and monitoring. A secondary output will be a presentation by NIWA experts (Drs. Max Gibbs and Chris Hickey) to Horizons staff on completion of the report.

The report and presentation will include the following components:

1. Assessment and interpretation of water quality information available for Lake Horowhenua, including data collected during 2008 as a part of Horizons intensive monitoring programme and from routine state of environment and recreational water quality monitoring programmes.

2. Where possible, inclusion of information regarding historic state and trends.

3. Information on how the specific goals of lake rehabilitation relate to numeric lake water quality target values and the existing water quality state. Examples of the goals of rehabilitation include restoring the health of the lake to ensure maintenance of viable native fish populations and increasing suitability of the lake for multiple uses e.g., elimination of health risks associated with cyanobacteria,

4. An overview of the outcomes of research related to remediation of other lakes e.g., Lake Okaro, will be provided.

5. Options for remediation of Lake Horowhenua, including options for addressing inflow sediment and nutrient sources, will be documented.

6. Recommendations for monitoring at different levels of intensity will be provided. Basic and comprehensive monitoring strategies will be defined, identifying the information likely to be provided by each of the monitoring strategies identified, as well as gaps in information that will result.

2 Background

2.1 Maori perspective

Pre-European, Lake Horowhenua (Maori name Waipunahau) was surrounded by dense, diverse forests of kahikatea, pukatea, and rata on the lake margins and wet areas, with nikau, totara, karaka, matai and rimu extending from the lake inland to the Tararua Range. To the ancient iwi, Muaupoko, it was their vital treasure (taonga) and food bowl (kumete), providing tuna (eel), inanga (whitebait), freshwater koura (crayfish), patiki (flounder), kākahi (freshwater mussels), waterbirds, and kereru (pigeon). Dense, diverse forests of kahikatea, pukatea, and rata on the lake margins and wet areas, and nikau, totara, karaka, matai and rimu extended from the lake inland to the Tararua Range. While the forest provided food, shelter and a place to live, the lake provided cultural and spiritual sustenance for Muaupoko. (MfE 2001)

"Today, no forest remains and there are no kereru. The quality of the lake water and its riparian margins have deteriorated to such an extent that the lake's mauri, its life-force or capacity to sustain life, has been seriously compromised. Locals continue to collect eels and kākahi, and the introduced trout and carp, but the fisheries are depleted when compared to their abundant past. The water is no longer drinkable and algal blooms and weed growths, and the subsequent offensive odours as the blooms die and rot, make the lake less than attractive for recreational and cultural pursuits" (extracted from MfE 2001).

2.2 Site description

Lake Horowhenua is a small (2.9 km^2) shallow (mean depth 1.3 m) coastal dune lake (volume = 3.8×10^6 m^3) in the iron sand zone on the west coast of the North Island of New Zealand adjacent to the town of Levin (Figure 1). The lake receives the runoff from intensive agriculture within its 61 km^2 catchment and, from the mid 1960s until 1987, the treated waste water from Levin, which occupies 8 km^2 or almost 14% of the catchment.

The Hokio Stream outflow has a mean flow of 0.8 m^3 s^{-1} (Gilliland, 1981) and runs 5 km to the sea. The Hokio Stream has a flow range from 0.3 to nearly 2 m^3 s^{-1}. A weir at the outlet controls the lake at a near constant level of 30 feet (9.1 m) above mean low water spring tides at Foxton Heads, a level which defines the surface area of the lake and the land boundaries for ownership as set out in the "Reserves and Other Lands Disposal Act, 1956" (Appendix 1).

Hydrological data collected during the DSIR study of the lake in 1988/89 (Gibbs & White 1994) found that the lake has several stream inflows, the largest of which is the Arawhata Stream with a mean flow of 0.26 m^3 s^{-1} and a seasonal flow range from 0.1 m^3 s^{-1} in summer to 0.7 m^3 s^{-1} in winter. Many of the streams flowing into Lake Horowhenua are spring fed and there are numerous lake bed springs along the eastern shores of the lake. The mean groundwater component of the water balance in 1988/89 was estimated to be 0.51 m^3 s^{-1}, which is more than half of the total mean annual inflow to the lake, but varied seasonally from 0.24 m^3 s^{-1} in autumn to 0.96 m^3 s^{-1} in late winter. There is a ~60 day lag between highest evaporation and lowest groundwater discharge (Gibbs & White 1994). At the 1988/89 mean discharge, groundwater contributed 16.2 million cubic metres (mcm) y^{-1} of water to the lake.

That compares favourably with a recent (2010) GNS estimate of 19.1 mcm y^{-1} (White et al., 2010).

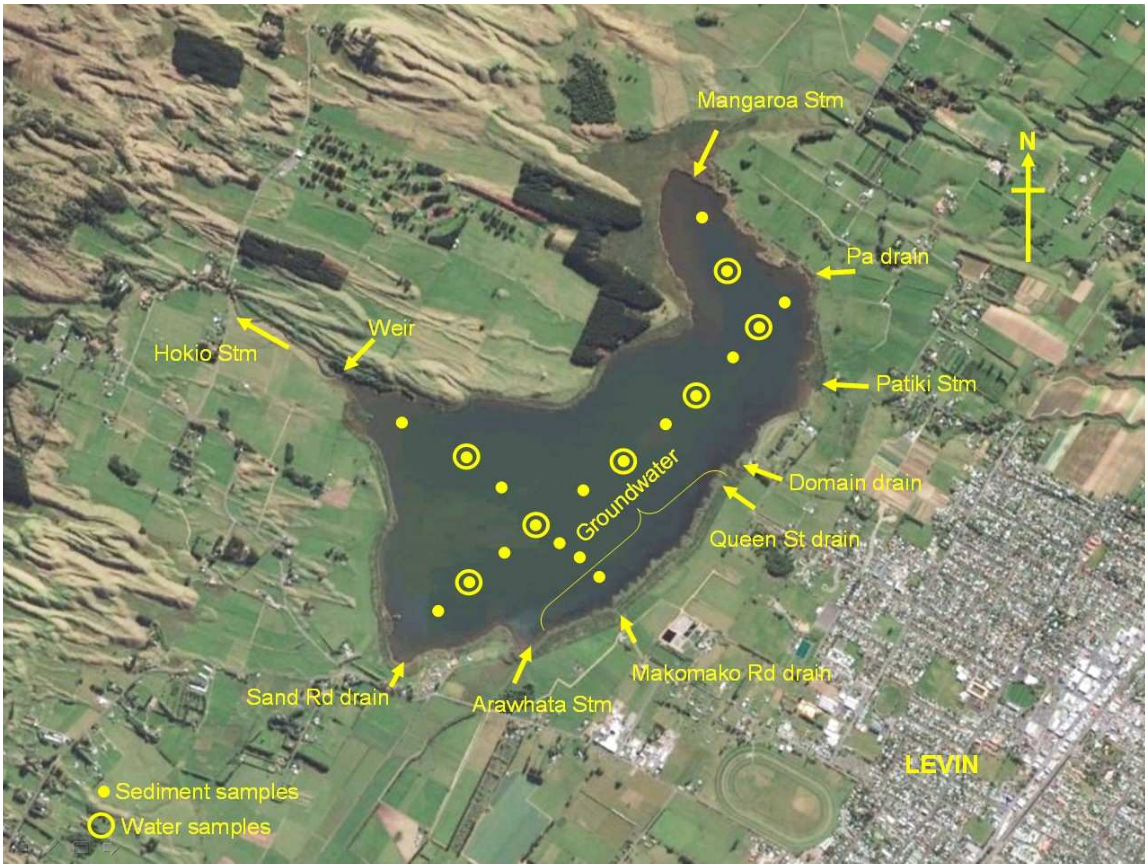


Figure 1. Site map of Lake Horowhenua showing the positions of the measured inflows, the main area of groundwater inflow and the Hokio Stream outflow. Locations for water samples (open circles) and sediment cores (solid dots) collected in July 1988 are also indicated. [Photo: Google Earth].

With an area of over 8 km^2 or 13.7% of the catchment occupied by the town of Levin, (Table 2-1) urban stormwater runoff into the lake is important and can be substantial. The largest stormwater inflow is the Queen Street drain, which can have flows of up to 0.2 $m^3 s^{-1}$. The Queen Street drain is currently the largest external source of phosphorus (P) nutrient to the lake (See Table 3-1).

Table 2-1: Land-use areas in the Lake Horowhenua catchment. These exclude the surface area of the lake (Horizons Regional Council data).

Land-use	Area (km^2)	Catchment %
Cropping	2.376	3.9
Dairy	11.555	18.9
Exotic cover	1.852	3.0
Horticulture - Other	0.725	1.2
Horticulture - Veg	1.783	2.9
Native cover	1.641	2.7
Other	0.964	1.6
Sheep and/or beef	31.782	52.0
Urban development	8.393	13.7
Total	**61.074**	

Early farming practices allowed stock direct access to the lake as well as the streams flowing into it, and stock grazing removed the marginal wetland plants that previously protected the shoreline from erosion by wave action. Land-use in the catchment is dominated by sheep and beef farming at about 52%, with the next largest land-use being dairy at about 19% (Table 2-1). If this area follows the national trend, dairy intensification is likely to increase. Cropping is the next largest land-use in the catchment at around 4%. The distribution of the different land-uses (Figure 2) indicates that dairy intensification has mostly occurred around the Arawhata Stream and in the headwaters of the Mangaroa Stream.

As a consequence of these and other catchment activities i.e., intensive agriculture, horticulture, direct stock access to the waterways, removal of the buffer zones as well as stormwater and sewage discharge into the lake, the lake was highly enriched with nutrients, which stimulated the growth of nuisance algal blooms,(Figure 3) and highly turbid with suspended sediment. Recent restoration measures have seen the lake edge replanted with flax.

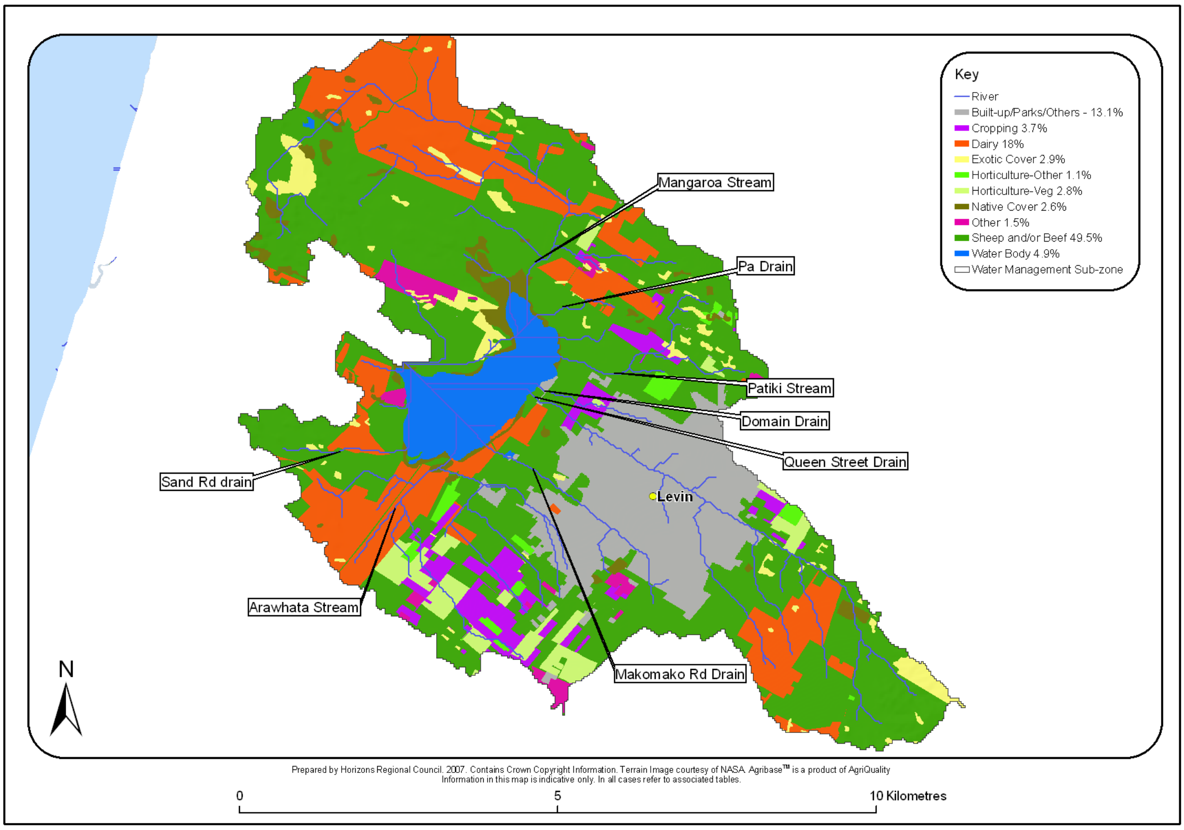


Figure 2: Land-use in the Lake Horowhenua catchment (Horizons Regional Council data). Note that the % areas on this map differ slightly from this in Table 2-1, as these include the lake area within the catchment total.

Figure 3: Cyanobacteria bloom on Lake Horowhenua. The streaking indicates wind drift with a south westerly wind [Google Earth photo date: 09/03/2011].

A key issue for Lake Horowhenua is the flushing or residence time, i.e., the time it takes to remove contaminants from the lake. The mean annual residence time of Lake Horowhenua is around 47 days ranging from <35 in winter, during high rainfall, to >95 in summer, when stream flows are low. The longer the residence time, the longer algae and aquatic macrophytes (water weeds) have to use the nutrient contaminants in the inflowing water.

The residence time of water in the ground (i.e., groundwater) is another important issue. Water travelling slowly through the ground is contaminated by the downward percolation of infiltrating rain and irrigation water leaching nutrients from stock excretion and fertiliser applications from the soil. White et al., (2010) used a hydraulic conductivity value (K) of 2 x 10^{-4} m s^{-1}, appropriate for the Pleistocene Ohau gravel fan underlying the lake catchment, to calculate the groundwater budget. This value of K indicates that groundwater flow velocity is around 17 m d^{-1} or 6.3 km y^{-1}, which means that it would take about a year for groundwater from the back of the catchment to reach the lake. This is consistent with the 60 day lag between highest evaporation and lowest groundwater discharge.

The implications for this relatively short groundwater residence time are that nutrients entering the groundwater will reach the lake within a year. Consequently, the lake will rapidly respond to changes in land-use practice. This is both good and bad – bad because the intensification of dairy farming in this catchment will exacerbate the nutrient loads already in the groundwater from other land–uses and the water quality of the lake will rapidly decline, but good because there is a reasonable expectation of seeing a rapid improvement in lake water quality if management strategies are implemented to reduce nutrient loads to the groundwater in the lake catchment.

2.3 One Plan

Horizons Regional Council has produced a new regional plan the, “One Plan”, which includes a set of recommendations for lake water quality standards. The water quality standards are intended for all lakes in the region but, because of the diverse range in size and depth, shallow lakes have been separated from deep lakes. “Deep” lakes are defined as those which undergo stable thermal stratification in summer. All other lakes are defined as “shallow”. Lake Horowhenua is a shallow lake and should meet the standards for shallow lakes (Table 2-2; extracted from Mrs Kathryn Jane McArthur-One Plan evidence pages 116-118).

Table 2-2: Recommendations for ONE PLAN lake water quality standards as they apply to Lake Horowhenua. Inflows should meet the water quality standards for rivers and streams.

Standard	Description
pH	The pH of the water shall be within the range 6.5 to 8.5
Algal biomass	The average annual algal biomass shall not exceed 12 mg chlorophyll *a* /m^3 in shallow lakes and no sample shall exceed 30 mg chlorophyll *a* /m^3
Total phosphorus	The annual average total phosphorus concentration shall not exceed 43 mg/m^3 in shallow lakes
Total nitrogen	The annual average total nitrogen concentration shall not exceed 735 mg/m^3 in shallow lakes
Ammoniacal nitrogen	The concentration of ammoniacal nitrogen shall not exceed 400 mg/m^3 and shall only apply when lake pH exceeds 8.5
Water clarity	The clarity of the water measured as Secchi depth or horizontal sighting of a 200 mm black disc shall not be less than 0.8 m and shall not be reduced by more than 20% in shallow lakes
Faecal Indicators	Escherichia coli shall not exceed 260 / 100 ml between 1 November and 30 April and shall not exceed 550 / 100 ml between 1 May and 31 October
Cyanobacteria toxins	The concentration of cyanobacteria toxins shall not exceed 20 mg/m^3

3 Limnology

Nutrients and sediment from the land accumulate in lakes and are stored in the lake sediments as a legacy from past land-use practices. Nitrogen (N) is continuously mineralised and released from the sediment in the form of ammonium (NH_4-N) which can be converted to nitrate (NO_3-N) by nitrifying bacteria or subsequently to nitrogen gas (N_2) by denitrifying bacteria. The N_2 gas is lost from the lake and the amount of N in the lake gradually reduces. Phosphorus (P) is mineralised to phosphate in the sediment but is retained in the sediment bound to iron and manganese oxides while the overlying lake water contains oxygen (aerobic conditions). This bound P can be released from the sediments in the form of dissolved reactive phosphorus (DRP) when dissolved oxygen concentrations in the lake fall to zero (anoxic conditions). This typically occurs in summer when microbial processes of decomposition are at their maximum oxygen consumption, causing the sediment to become anoxic. This consumption of oxygen for decomposition is referred to as sediment oxygen demand.

The N and P released from the sediment are readily used by algae for growth. If there is a surplus of N (i.e., P-limitation to algal growth), the dominant algal species will most likely be diatoms and green algae. If there is a surplus of P (N-limitation to algal growth), the dominant algal species will most likely be cyanobacteria (blue-green algae) which can use (fix) atmospheric nitrogen for growth. When the algae die they return the N and P in their cells to the sediment where it can be recycled again, augmenting the new N and P entering the lake from the catchment.

Nutrients released from the lake sediments are referred to as the internal nutrient load while nutrients entering the lake from the catchment via streams, groundwater and direct discharge are external loads.

3.1 Lake Horowhenua

3.1.1 Historical data

The earliest published water quality data for Lake Horowhenua was collected on 15 January 1949 (Cunningham et al., 1953). At that time the lake had a Secchi depth reading of 0.75 m and the DRP concentration was 15 mg m^{-3}. The surface dissolved oxygen concentration was 100% saturated while the bottom water at 2 m depth was 65% saturated. At a temperature of 21 ºC these oxygen values translate to 8.84 and 5.75 g m^{-3}, respectively.

Cunningham et al., (1953) also evaluated the plant, water fowl and fish communities in the lake.

Historical data collected in the 1970s by Gilliland (1978, 1981) showed that Lake Horowhenua had a seasonally dependent nutrient cycle with high DRP concentrations (up to 400 mg m^{-3}) in summer and high NO_3-N concentrations (up to 2,500 mg m^{-3}) in winter (Figure 4). This seasonal nutrient cycle caused algal P-limitation in winter and N-limitation in summer, the latter accompanied by nuisance blooms of cyanobacteria.

The high NO_3-N concentrations were associated with higher inflows and discharge (Figure 4) indicating that the N was mostly from the external load. The high DRP concentrations were associated with low inflows and discharge in summer indicating they were most likely from the internal load. This implies that the bottom waters of the lake became anoxic so that the P bound to the iron and manganese oxides in the sediment could be released into the lake water. Because Lake Horowhenua is very shallow and exposed to the prevailing winds from the west as well as katabatic winds off the Tararua Ranges to the east, wind-induced turbulence should have mixed the lake keeping it well oxygenated and thus preventing the P release observed. However, calm conditions occur at night through to morning in summer for a few hours and this is most likely when the P is released.

To achieve anoxic conditions at the sediments under these conditions, the lake would have needed a large sediment oxygen demand from decomposing organic matter. This organic matter was provided by the sewage effluent discharge into the lake, pasture and dairy shed runoff and direct stock access to the lake (Brougham & Currie 1976).

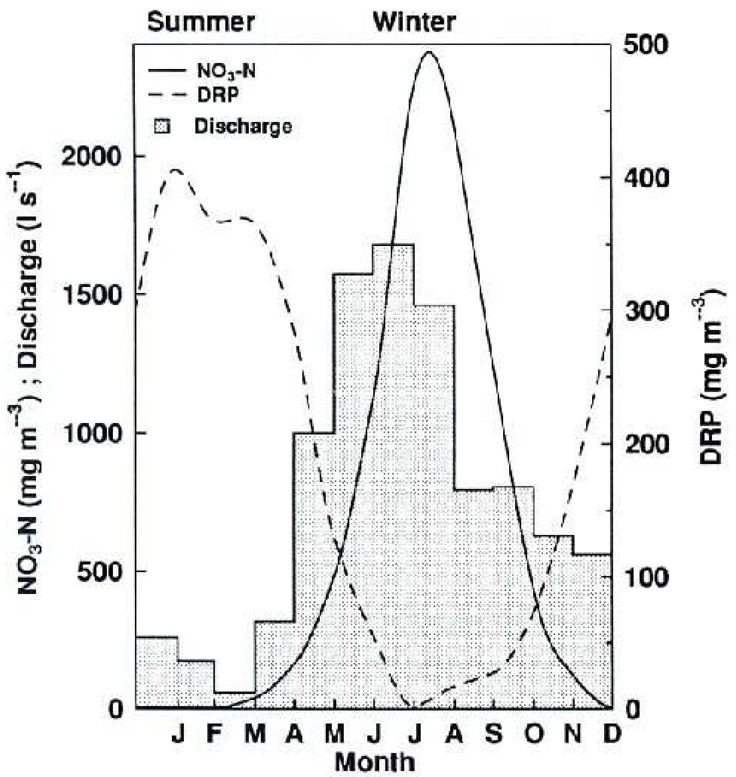


Figure 4. Annual nutrient and hydraulic cycles in Lake Horowhenua (redrawn from Gibbs & White, 1994).

3.1.2 Evidence for dissolved oxygen depletion

A mesocosm experiment conducted in Lake Horowhenua in November 1988 (M. Gibbs, unpublished data) demonstrated that the lake had the potential to develop thermal stratification under calm conditions and experience bottom water oxygen depletion (Figure 5). The mesocosm consisted of a length of 0.6 m diameter rigid PVC tube driven about 0.75 m into the lake bed at its deepest point, with the top of the tube out of the water. A series of ports let through the side of the mesocosm tube and connected via 1.5 mm ID flexible tubes to the surface, allowed the water inside the mesocosm to be sampled without disturbing the enclosed water column. Dissolved oxygen (DO) was measured by drawing the water into a syringe and injecting it into an Orion PO_2 blood oxygen micro-cell. Several sequences of measurements were made but rising winds moved the mesocosm tube mixing the water inside. The last complete sampling (Figure 5) taken 2 hours after installation, showed that the lake thermally stratified at a depth of 0.8 m and that the sediment oxygen demand was sufficient to lower the DO concentration to 4.4 g m^{-3} i.e., <50% saturation. Because these measurements were made in spring (November), it is likely that in summer, warmer water and calm conditions, especially at night when there is no photosynthesis, could allow anoxic conditions to develop very quickly.

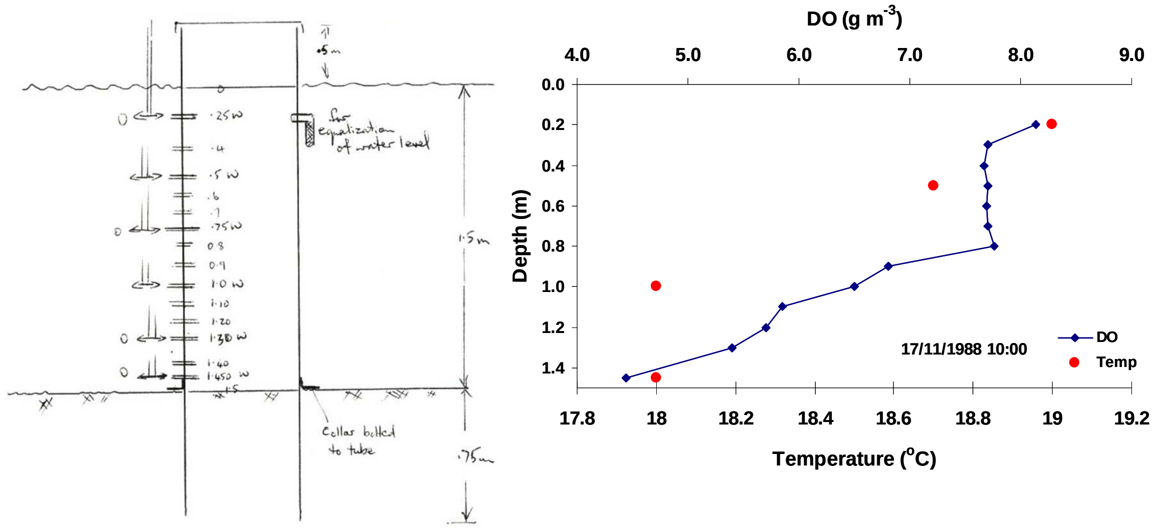


Figure 5. The mesocosm (left) (original sketch from field book), and dissolved oxygen profile obtained 2 hours after installation (right).

Temperature profile data collected in March 2011 using a thermistor chain with sensors 10 cm apart shows the rapid onset of thermal stratification on an almost daily basis (Figure 6A). Between 5-8 March, the top and bottom temperatures were the same indicating that the lake water column was well mixed. The DO concentrations 20 cm below the surface and 20 cm above the lake bed (Figure 6B) were also essentially the same indicating complete mixing and aeration at that time. However, on the 8th March, the top and bottom temperatures were widely separated (Figure 6A) indicating thermal stratification. At the same time, the 20 cm DO concentration increased to 140% saturation while the 140 cm depth DO concentration decreased to <90% saturation (Figure 6B). The increase in DO % saturation indicates high levels of photosynthesis from the algal bloom in the lake. The decrease in DO % saturation 20 cm above the lake bed indicates high sediment oxygen demand (SOD). Without the photosynthesis to counteract the SOD the DO % saturation at that depth could have been <40%. Another large excursion in DO % saturation 20 cm above the lake bed occurred over the 14th and 15th March. At this time, the oxygen depletion began as soon as the katabatic wind stopped (Figure 6C).

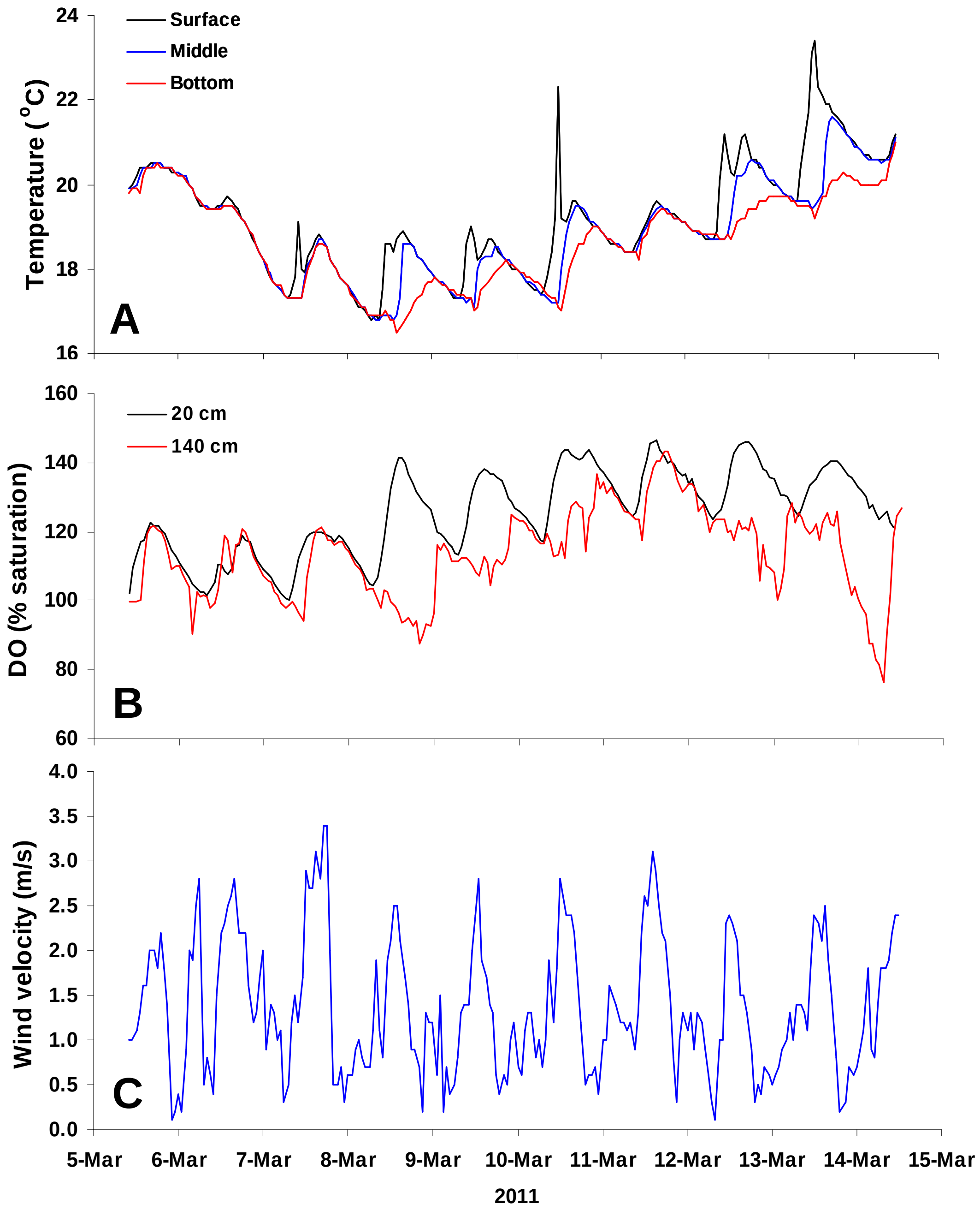


Figure 6. A: Lake water temperature at surface (10 cm), middle (90 cm) and bottom (160 cm) depth of the water column. Additional data at 10 cm depth intervals omitted for clarity.; **B:** Dissolved oxygen as % saturation at 20 cm and 140 cm depth in the lake; **C:** Wind velocity (m/s) across the lake. (Hourly data from Horizons Regional Council).

The deepest DO logger was 20 cm above the lake bed and, consequently, it was not able to record the maximum DO depletion, which would have occurred a few cm above the sediment. However, the rapid DO depletion on 15 March implies that the near-sediment water was most likely anoxic. A major limitation of these data are that they were collected late in summer when the P had already been released, rather than in January to February when the P was just beginning to be released and bottom water anoxia would have been easier to detect in the calm conditions. The response to the P release can be seen in the resultant cyanobacteria bloom at that time (Figure 3).

How rapidly dissolved oxygen would be removed from the lake can be calculated as a hypolimnetic oxygen depletion (HOD) rate from the time-series loss of DO indicated by the bottom DO logger data over the night of 14/15 March (Figure 6B). When calculating the HOD rate, the temperature should not change by >0.2ºC, and there should be no effects due to photosynthesis. The data over night on 14/15 March meet those criteria and the estimated HOD rate is 302 mg m^{-3} h^{-1}, which is 7250 mg m^{-3} d^{-1} (Figure 7). This rate is high and indicated that the bottom water would lose all DO in about a day.

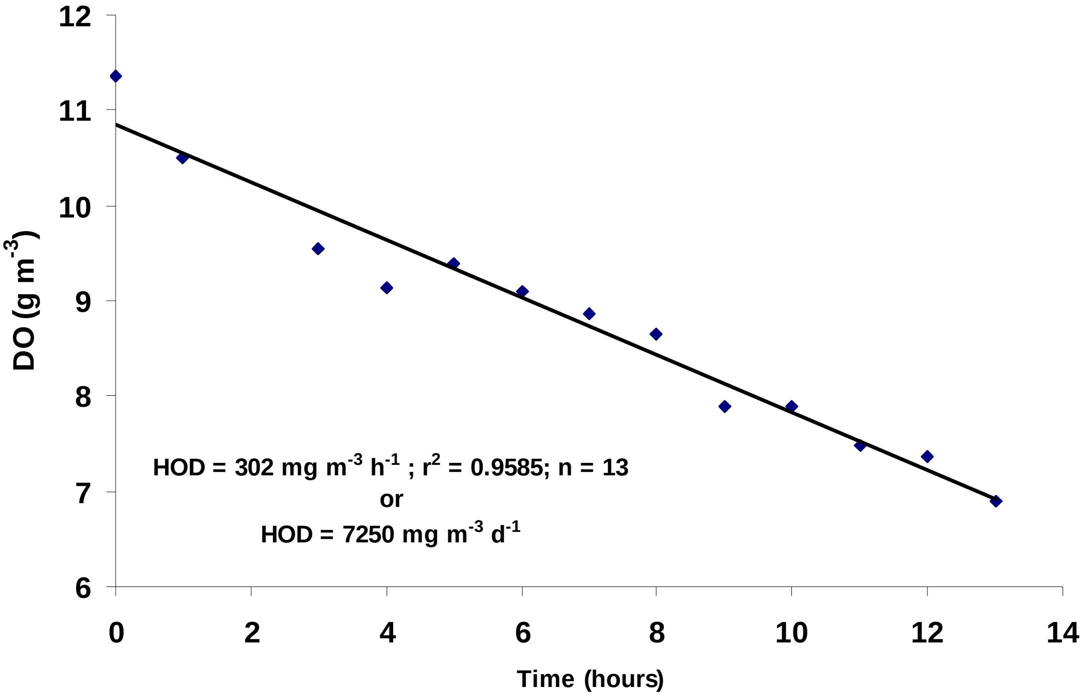


Figure 7. Hypolimnetic oxygen depletion (HOD) rate estimated for Lake Horowhenua.

Putting this HOD rate into perspective, the HOD rate for Lake Rotorua is around 820 mg m^{-3} d^{-1} (as at March 2011). If the HOD rate is converted to an areal rate (AHOD) by multiplying by the mean depth of the hypolimnion (8 m), the AHOD for Lake Rotorua is 6560 The AHOD rate for Lake Horowhenua, assuming a hypolimnetic depth of 1 m, is 7250 mg m^{-3} d^{-1}. The similarity between these AHOD estimates is consistent with the history of pollution of both lakes from sewage effluent discharge and farm nutrient runoff in the catchment. Both lakes have had the sewage effluent discharge removed for about 20 years and the lakes are beginning to recover.

3.1.3 Water quality in the late 1980s

In April 1987, the sewage effluent from Levin was diverted away from the lake reducing the external loads to the lake by an estimated 20% for N and about 90% for P (Vant & Gilliland, 1991). However, a monitoring programme established to follow the change in water quality after the diversion found that there was no improvement in winter in the first 3 years. Even though the total P (TP) and DRP concentrations in the lake water column had more than halved, they were still very high in summer and continued to stimulate cyanobacteria blooms (Figure 8). The high P was attributed to the large internal P load from the legacy of nutrients and organic matter in the sediment (Vant & Gilliland, 1991).

Figure 8. Cyanobacteria bloom on Lake Horowhenua in March 1989. Light winds caused the bloom to drift inshore along the eastern side of the lake. The pale scum was sun-bleached dead algal cells. [Photo: M. Gibbs]

In 1989 the Department of Scientific and Industrial Research (DSIR), the predecessor of NIWA, monitored the lake and all surface water inflows and outflows over a twelve month period. These data were combined with meteorological data and historical data collected by Horizons predecessors to model processes occurring within the lake. Strategies for lake remediation were also identified (Gibbs & White 1991, 1994).

The seasonal N and P cycles identified in the historical data (Figure 4) were verified (Figure 9). The Hokio Stream nutrient concentrations matched the lake concentrations confirming that the main source of water in the Hokio Stream was the discharge from the lake.

The lake data were also compared with the NO_3-N and DRP concentrations in the Arawhata Stream and groundwater inflows (Figure 10, Figure 11). From these data it is apparent that the major sources of NO_3-N to the lake in 1988/89 were from the catchment via streams and groundwater (Figure 10) while the major source of DRP was the internal load from the lake sediments (Figure 11). It is also apparent that the seasonal cycle of summer N-limitation in the lake (Figure 9) was not relieved by these catchment sources. With such an imbalance of biologically available N to P in summer, it is not surprising that the lake developed intense cyanobacteria blooms (e.g., Figure 8).

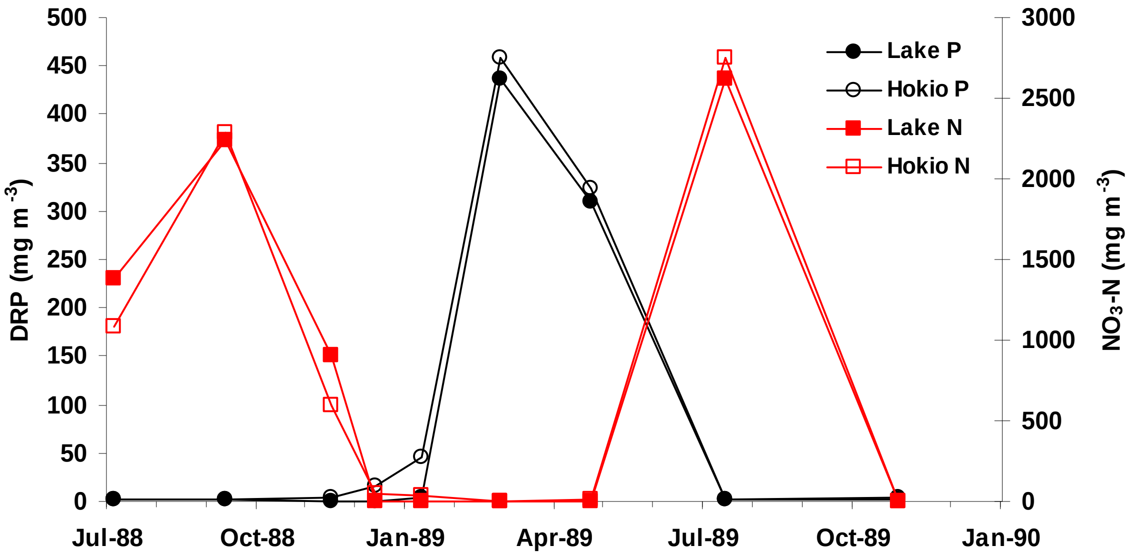


Figure 9. Seasonal cycle of dissolved reactive phosphorus (DRP) and nitrate nitrogen (NO_3-N) concentrations in Lake Horowhenua and the Hokio Stream discharge measured in 1988/1989. [DSIR/NIWA historical data]

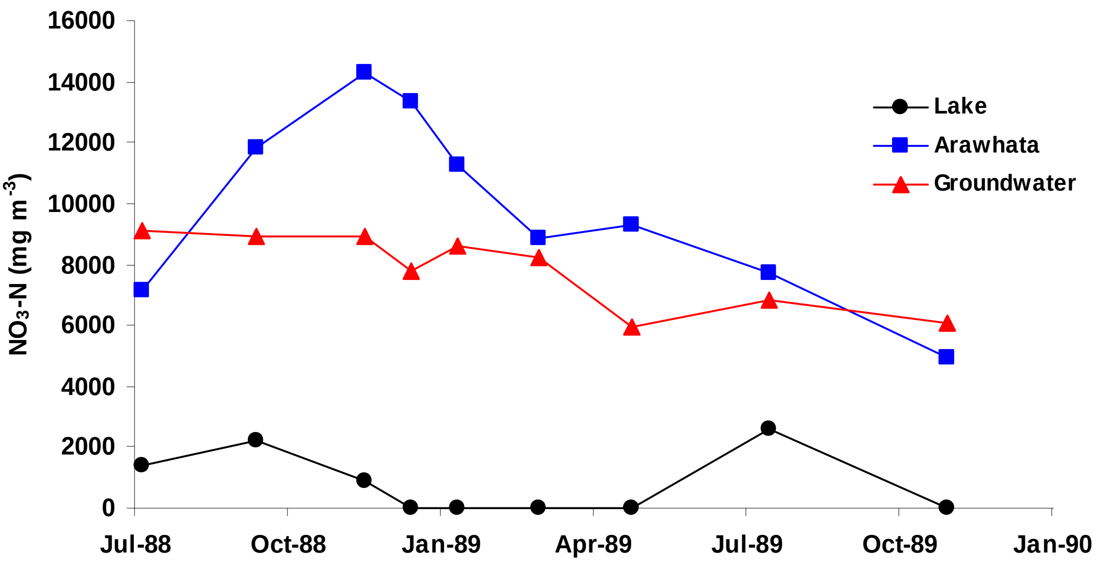


Figure 10. Seasonal cycle of nitrate nitrogen (NO_3-N) concentrations in Lake Horowhenua compared with concentrations in the Arawhata Stream and groundwater inflows measured in 1988/1989. [DSIR/NIWA historical data]

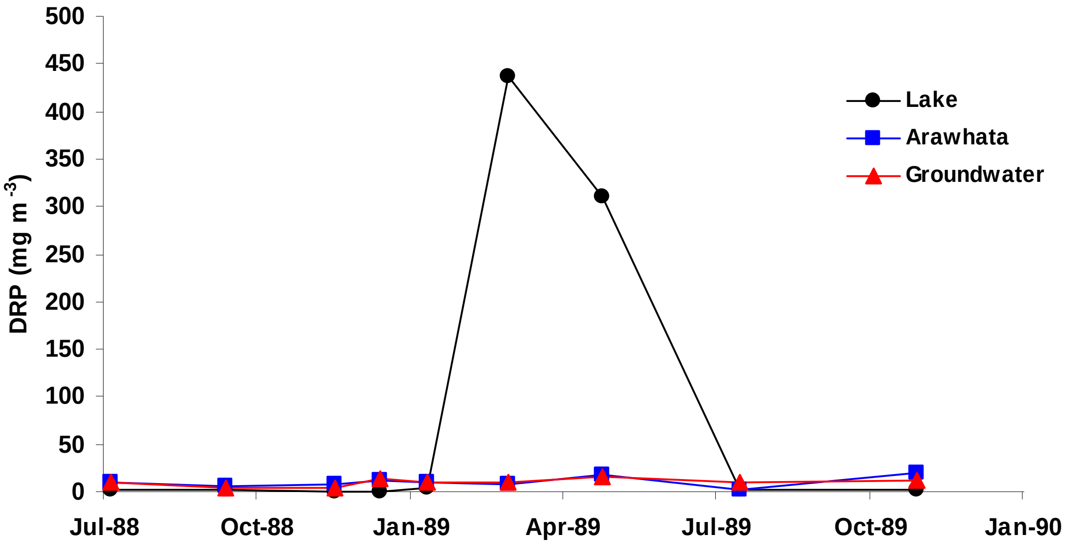


Figure 11. Seasonal cycle of dissolved reactive phosphorus (DRP) concentrations in Lake Horowhenua water column compared with concentrations in the Arawhata Stream and groundwater inflows measured in 1988/1989. [DSIR/NIWA historical data]

During the 1988/89 study, the discharge from inflows, groundwater and the Hokio Stream out flow were measured. Coupled with the concentration data, this allows an estimate of some key mass transport components (Figure 12, Figure 13) as well as an estimate of the net rate of DRP release from the sediment. Based on a period of 49 days (11 January to 1 March), a mean annual residence time of 47 days and thermal stratification isolating about ⅔ of the sediment area, the TP increased at a net rate of about 50–60 mg m^{-2} d^{-1}. This estimate allows for the DRP released from the sediment and assimilated into algal biomass before being flushed out of the lake. This net release rate is comparable with present day release rates from Lake Okaro sediment at ~38 mg m^{-2} d^{-1} (Gibbs & Özkundakci (2011) and Lake Rotorua sediment at 29-86 mg m^{-2} d^{-1} in summer (Burger et al., 2007).

The differences between the TP mass in the lake water column and the discharge rate of TP down the Hokio Stream in summer and winter (Figure 13) is a function of concentration and flow in the Hokio Stream. The soluble P components must match in the lake and in the Hokio Stream (Figure 12) because the Hokio Stream water is derived from the lake. The mass of suspended solids component discharged included both algae and sediment which, in this case (Figure 13), was enhanced by very high flows in the Hokio Stream (measured flow of 1.7 m^{-3} s^{-1} in July compared with 0.29 and 0.43 m^3 s^{-1} in March and April, respectively.

Similar flow related effects are apparent in the TN comparisons (Figure 14). In this case, the rapid infiltration of rainwater into the ground is seen as a reduction of TN in the groundwater component due to dilution. This rapid response is consistent with the high hydraulic conductivity of the soil underlying the catchment of the lake.

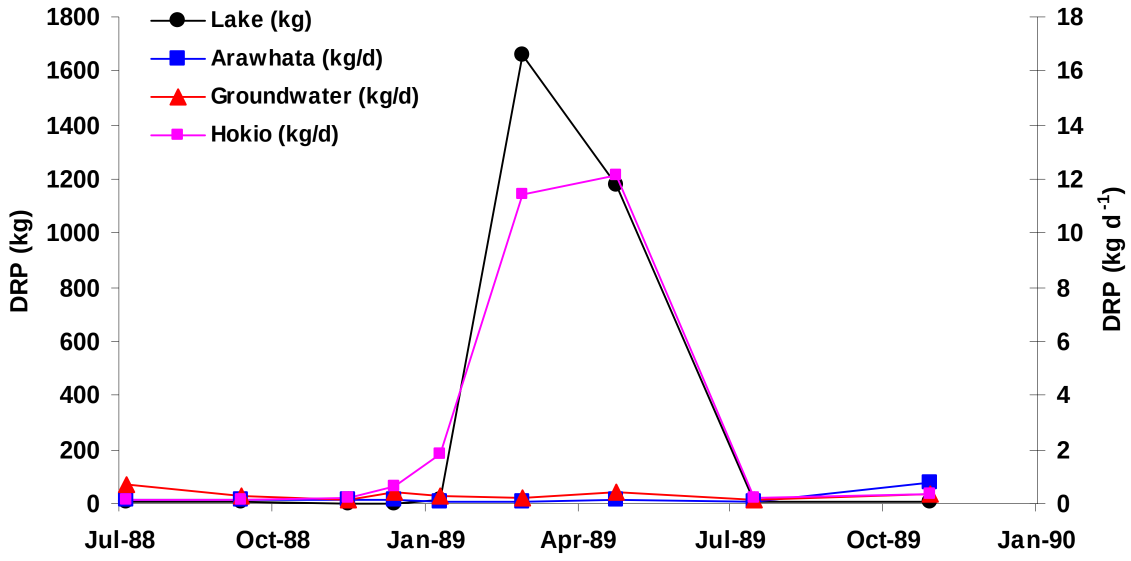


Figure 12. Mass (kg) of DRP (left axis) in Lake Horowhenua water column compared with the flux (kg d^{-1}) into and out of the lake over the 1988/89 study period (right axis)**.**

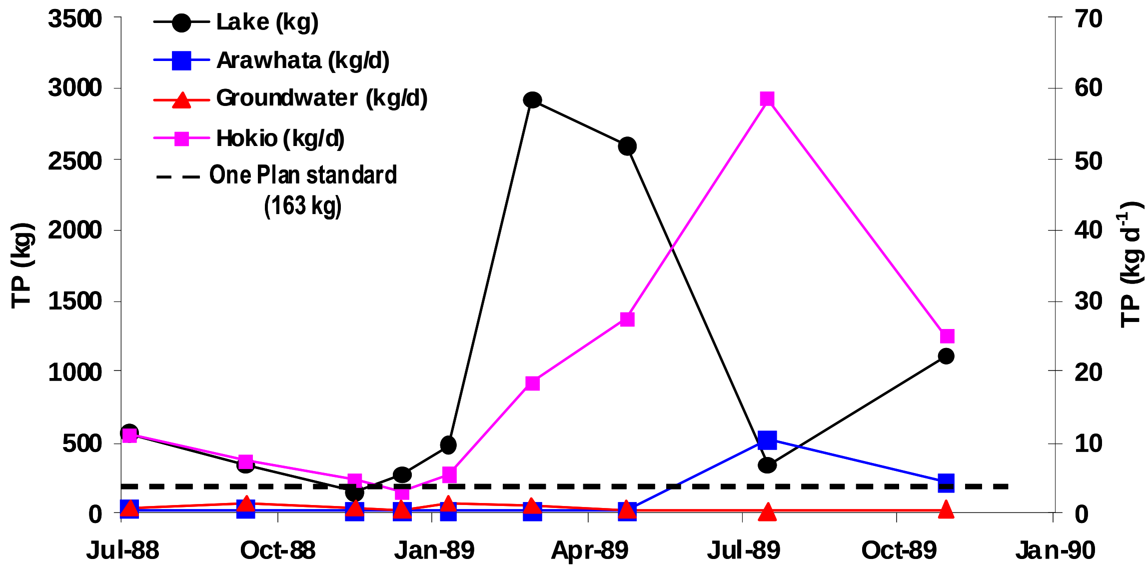


Figure 13. Mass (kg) of TP (left axis) in Lake Horowhenua compared with the flux (kg d^{-1}) into and out of the lake over the 1988/89 study period (right axis)**.** The equivalent average TP mass in the lake for the One Plan standard (broken line) is 163 kg (**Table 2-2**).

A similar mass balance for N shows a loss of about 8000 kg of TN from the lake water column between October and late December 1988 (Figure 14). This loss can be attributed to the decline in mass inflow at that time — the Hokio Stream outflow fell from 1.54 $m^{-3}s^{-1}$ to 0.47 $m^{3}s^{-1}$ — coupled with spring growth of macrophytes and denitrification, seen as a reduction in NO_3-N in the lake from ~400 mg m^{-3} to zero at that time (Figure 9). There was no increase in algal biomass at that time (Figure 15).

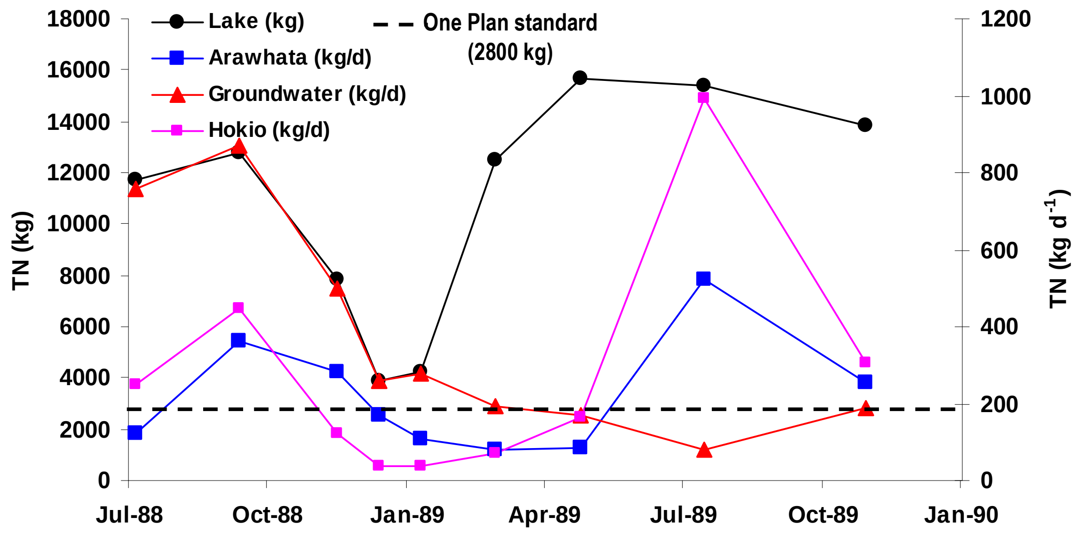


Figure 14. Mass (kg) of TN (left axis) in Lake Horowhenua water column compared with the flux (kg d^{-1}) into and out of the lake over the 1988/89 study period (right axis). The equivalent average TN mass in the lake for the One Plan standard(broken line) is 2800 kg (**Table 2-2**).

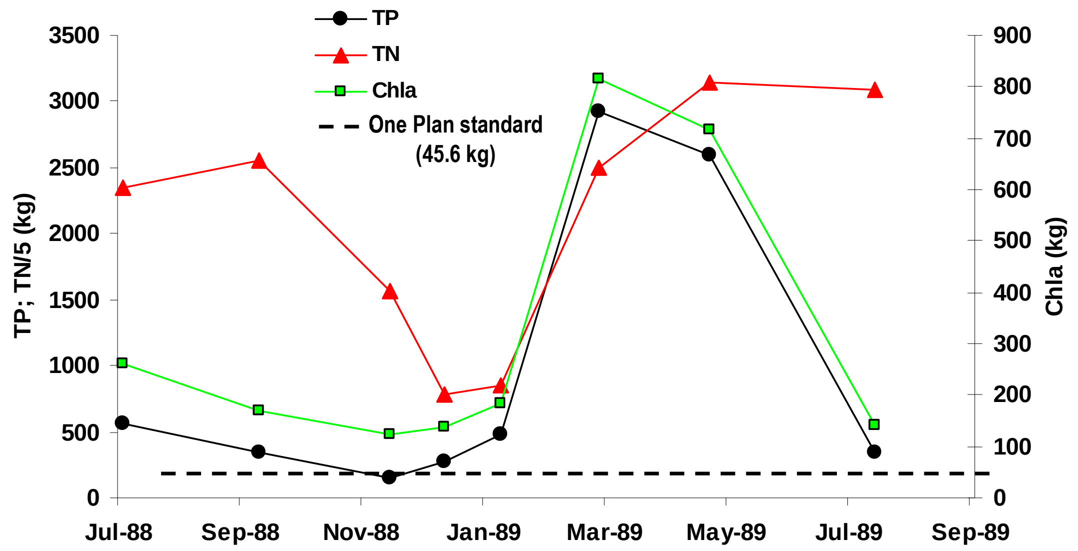


Figure 15. Mass (kg) TN (divided by 5 to fit scale) and TP (left axis) compared with Chlorophyll *a* (Chla) (right axis) over an annual cycle. The equivalent average Chla mass in the lake for the One Plan standard (broken line) is 45.6 kg (**Table 2-2**).

Algal biomass in the lake (Chla, Figure 15) was low during the spring of 1988 and increased in concert with the TP and TN in autumn 1989. Maximum chlorophyll *a* concentrations reached 215 mg m^{-3} in March 1989 but subsequently exceeded 410 mg m^{-3} in the following year — data from concurrent Manawatu, Wanganui Regional Council monitoring programme of Lake Horowhenua (Vant & Gilliland, 1991). The close relationship between Chla and TP indicates that the TP was mostly associated with algal biomass with around half being dead algal cells.

3.1.4 Nutrient budget 1988/89

From the data collected in the DSIR/NIWA 1988/89 study, hydraulic and nutrient budgets were developed for the lake (Table 3-1) using a computer model (Gibbs & White 1994).

Table 3-1. The estimated quantities of water and nutrients associated with a variety of sources and sinks over the period 29 June 1988 and 28 June 1989 (365 days) derived from a daily model output. (From Gibbs & White 1994).

Water (×10^6 m^3 y^{-1})	**Inputs**			**Outputs**	**Storage**
Rainfall (direct)	3.32				
Runoff (from Levin)	5.09				
Streams	8.57				
Groundwater	16.08			1.35	
Evaporation				2.34	
Hokio Stream (Outflow)				29.28	
Change in lake storage					0.09
Nutrients (t y^{-1})	**External inputs**	**Internal sources**	**Internal sinks**	**External outputs**	**Storage**
Phosphorus					
Rainfall (direct)	0.07				
Runoff (from Levin)	2.60				
Streams	0.27				
Groundwater	0.23				
Sediment release		5.77			
Sedimentation			4.39		
Hokio Stream (Outflow)				4.75	
Change in lake storage					-0.20
Nitrogen					
Rainfall (direct)	1.7				
Runoff (from Levin)	28.2				
Streams	71.6				
Groundwater	101.9				
Sediment release		30.0			
Sedimentation			39.7		
Denitrification			111.1		
Hokio Stream (Outflow)				77.2	
Change in lake storage					5.4

These estimates show the relative importance to the lake of the various sources. Groundwater is the largest single inflow and carries the most N but very little P. This is because the groundwater is aerobic and is travelling through an iron rich gravel aquifer which can adsorb the P from the water. The largest external source of P in 1988/89 was the runoff from Levin via the Queen Street drain. Just over half the P inputs were exported from the lake via the Hokio Stream while the rest were returned to the lake sediment. The model indicated a small net loss of P from the lake.

In contrast, there was a net accumulation of N in the lake. The N inputs were mostly offset by a large component of denitrification and the large amount of N exported from the lake via the Hokio Stream.

3.1.5 Recent water chemistry

Horizons and its predecessors have also monitored the lake within their "state of the environment" monitoring. In 2008, Horizons undertook an intensive monitoring programme over a six month period. All surface water inflows and outflows were assessed in terms of flow and water quality at monthly frequency. That monitoring programme included assessment of the storage capacity of the lake and some fish monitoring information.

Evaluation of the recent (since 2000) DRP and NO_3-N data shows that the seasonal cycling between high NO_3-N concentrations in winter and high DRP concentrations in summer continues (Figure 16). However, the magnitude of the concentrations is mostly less than in 1988/89 (Figure 9). In winter 1988 and 1989 the concentrations of NO_3-N peaked at 2500 to 2800 mg m^{-3} but in the recent data the maximum values were often less than 2000 mg m^{-3}. Higher NO_3-N concentrations occurred in winter 2006 and at different locations in the lake on occasion. For example, on 4 September 2008 NO_3-N concentrations ranged from 1140 to 4587 mg m^{-3}, mean 2630 mg m^{-3}, across 5 sites in the lake (Figure 16).

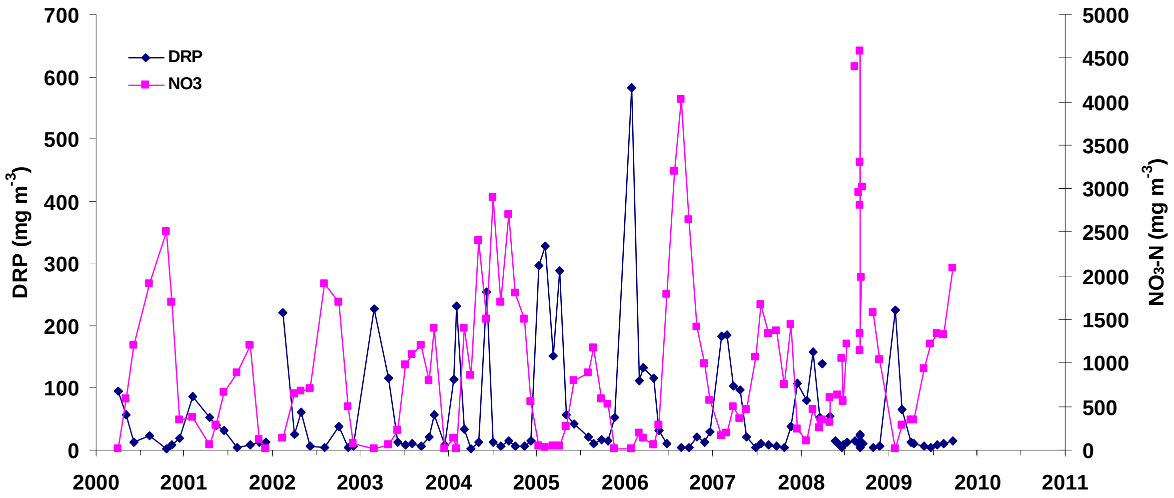


Figure 16. Time-series DRP and NO_3-N concentrations (mg m^{-3}) in Lake Horowhenua since 2000 [Horizons Regional Council data].

Reduction in the summer DRP concentrations were more substantial with the peak concentrations mostly less than half the 1988/89 concentrations – the exception being summer 2006 where the maximum DRP concentration reached 583 mg m^{-3} (Figure 16).

Chlorophyll *a* (Chla) typically increased in summer as the DRP concentrations increased. Historical data showed bloom concentrations of chlorophyll *a* at 150 to >400 mg m^{-3} (1988 to 1990 data). This general pattern continues in the more recent data, although summer bloom chlorophyll *a* concentrations range from 71 (2001) to >1300 (2002) mg m^{-3} since 2000 (Figure 17). The appearance of DRP free in the lake water column appears to trigger the summer algal bloom. The DRP is utilised by the algae for growth and TP correlates with chlorophyll *a* (Figure 15). The DRP associated with the chlorophyll *a* concentrations since 2000 (Figure 17) are residual concentrations and indicate N-limitation to algal growth.

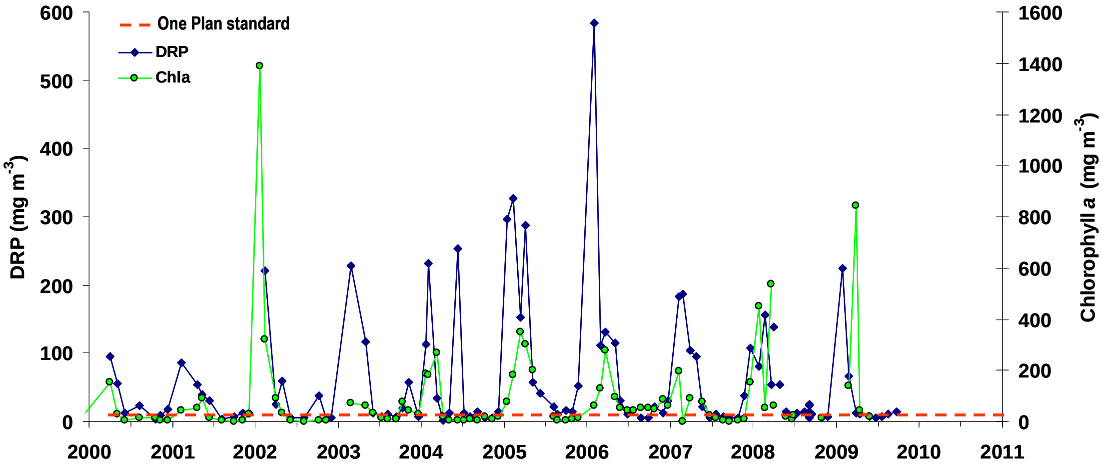


Figure 17. Time-series DRP and Chlorophyll *a* concentrations (mg m^{-3}) in Lake Horowhenua since 2000 The One Plan standard (broken line) for mean annual chlorophyll *a* is 12 mg m^{-3} (**Table 2-2**). [Horizons Regional Council data].

These data demonstrate that, although the removal of the sewage effluent from the lake in 1987 appears to have resulted in a reduction in the amount of DRP in the lake in summer, it has had little effect on the water quality as indicated by algal biomass in the summer blooms. This implies that primary production in the lake is being driven by an excess of P which favours the growth of cyanobacteria. Consequently, the development of the summer algal bloom is likely to be closely linked to the release of DRP, and thus the mechanism of that release, from the lake sediments.

3.2 Arawhata Stream

After groundwater, the largest single inflow to Lake Horowhenua is the Arawhata Stream which drains land traditionally used for horticulture and market gardens but more recently intensive dairy farming (Figure 2). The monitoring data in1988/89 shows that this inflow had high concentrations of NO_3-N with an annual mean of 9,850 mg m^{-3}. This has increased to 12,960 mg m^{-3} in the 2000/10 data. In contrast, in1988/89 DRP concentrations were low with an annual mean of 10 mg m^{-3}, These have also increased to a present mean of 22.1 mg m^{-3}. In 1988/89, TP concentrations ranged from 16 to 180 mg m^{-3}, mostly as particulate P (PP), and were directly linked with flow (Figure 18). Although the high NO_3-N concentrations dominated the TN time-series data (Figure 10), NH_4-N and particulate N (PN) were also found to be directly linked with flow (Figure 19). The implication from these data is that P-enriched soil or sediment can be washed into the lake during heavy rain or flood events. This is consistent with recent observations of up to 0.5 m depth of sediment accumulation in the Arawhata Stream channel (J. Roygard, Horizons Regional Council, pers. comm.).

It is uncertain whether the PP came from land runoff or stream bed erosion. However, the PN:PP ratios at this time averaged 3.8 (range 2.1 to 5.4), which may indicate the source was more likely to be stream bed sediment rather than fresh top soil, unless there are effluent retention ponds that could be flushed into the stream. Supporting this scenario is the linkage between NH_4-N concentrations and flow (Figure 19). The presence of NH_4-N in an aerobic system with increased fresh water implies that the increased flow has stripped the NH_4-N from otherwise anoxic stream bed sediments or wastewater ponds and there has been insufficient time for it to be nitrified to NO_3-N or assimilated into algal growth.

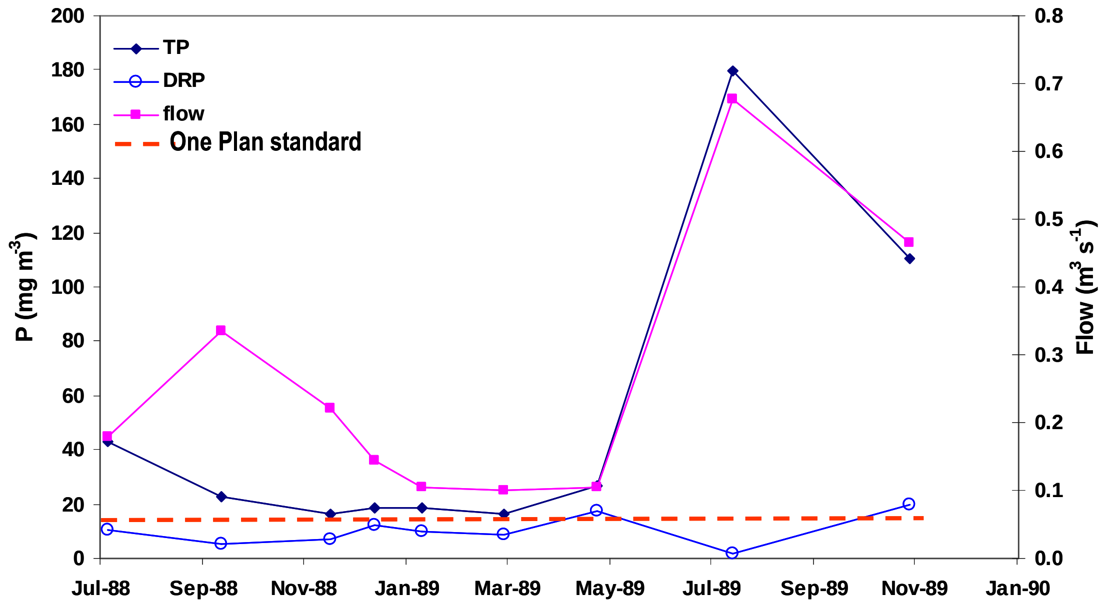


Figure 18. DRP and TP concentrations relative to flow in the Arawhata Stream during the 1988/89 monitoring period. One Plan DRP standard in streams is <15 mg m^{-3}.

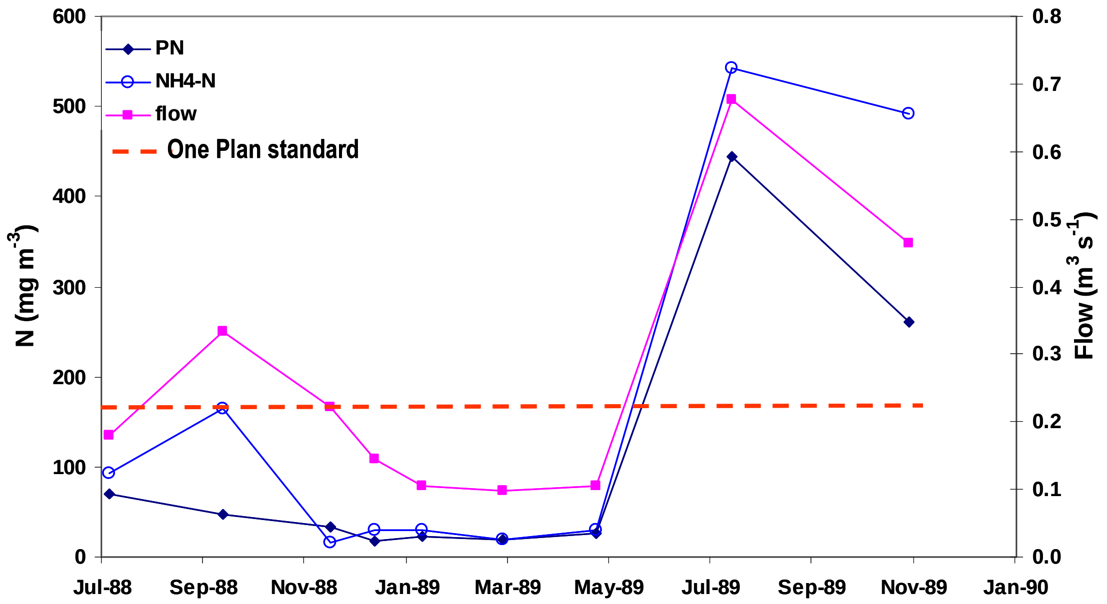


Figure 19. NH_4-N and PN concentrations relative to flow in the Arawhata Stream during the 1988/89 monitoring period. One Plan soluble inorganic nitrogen standard in streams is <167 mg m^{-3} and 400 mg m^{-3} for NH_4-N.

The response to the high flow (flood) event in the Arawhata Stream in July 1989 appears to have been an increase in algal biomass beyond the expected summer bloom period (Figure 20). This increase was about 3 times greater than the bloom driven only by the sediment release from the lake starting around April 1988. It is possible that the peak algal biomass in May 1990 includes the effects of the autumn sediment release in 1990, although the peak DRP concentration in 1990 was only 10% of that reached in March 1989.

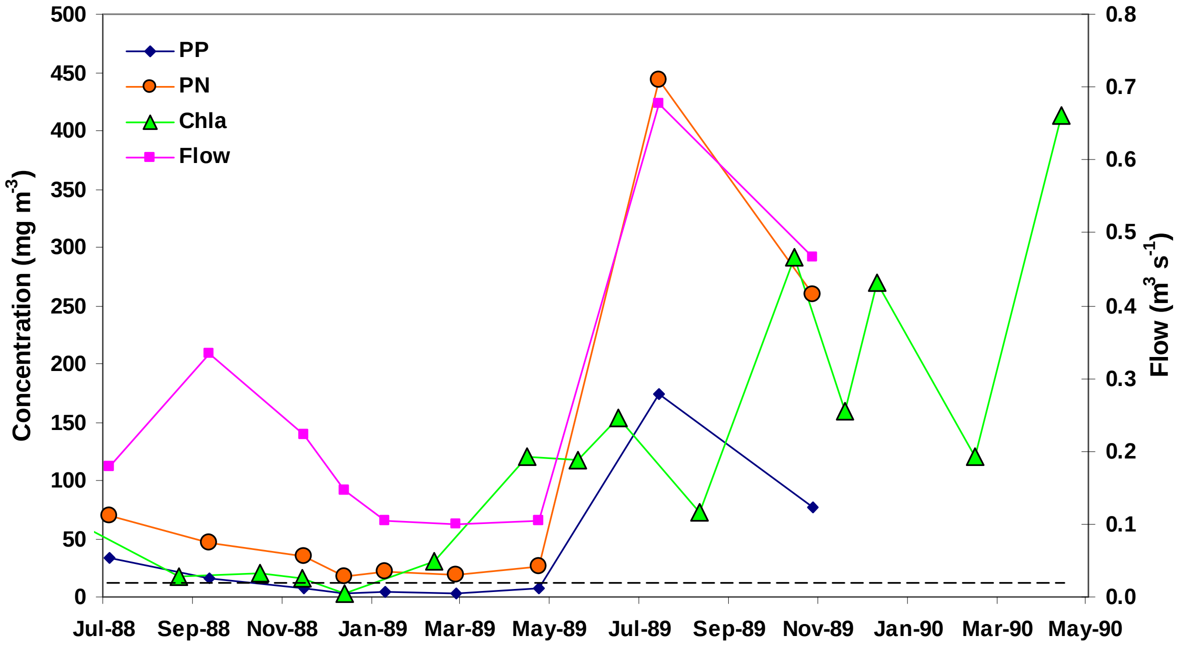


Figure 20. Changes in algal biomass (Chla) in Lake Horowhenua in response to the high flow and high PN and PP concentrations in the Arawhata Stream in spring 1989. Chla data from Vant & Gilliland (1991). The One Plan Chla standard for shallow lakes is 12 mg m^{-3} (broken line) (**Table 2-2**).

The more recent lake monitoring data also includes nutrient and DO data for the Arawhata Stream. Time-series plots of the Arawhata Stream TP data, as an indicator of high flow, do not correlate with algal biomass in the lake (Figure 21) suggesting that the 1989/90 response was coincidental or that the response was missed by the gaps in the sampling sequence.

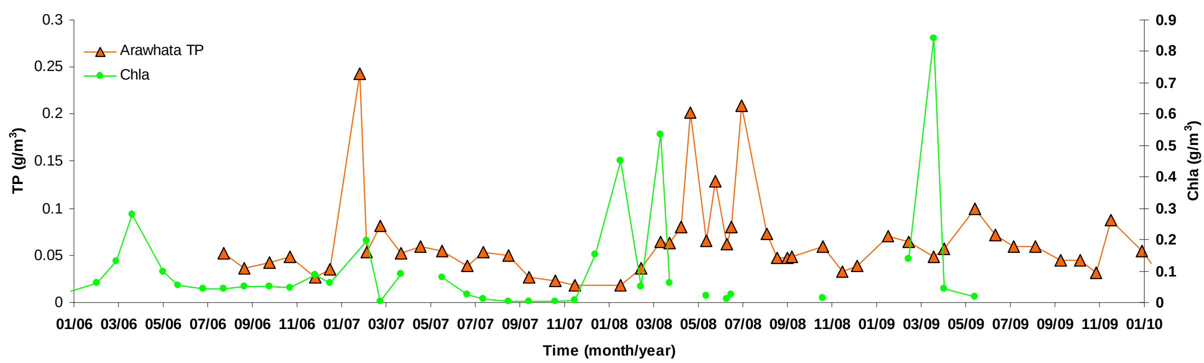


Figure 21. Time-series of TP in the Arawhata Stream, as an indicator of high flow, relative to algal biomass (Chla) in Lake Horowhenua. There is no apparent correlation.

Of some interest are the DO data in the Arawhata Stream (Figure 22). These data have considerable variability for what is reportedly a spring flow. With an annual mean temperature of around 14 ºC the expected DO concentration would be around 9 to 11 g m^{-3} for 100% saturation. Values above this can be attributed to macrophyte or periphyton photosynthesis in the stream channel during the day while values below imply respiration, presumably due to sediment oxygen demand, at night.

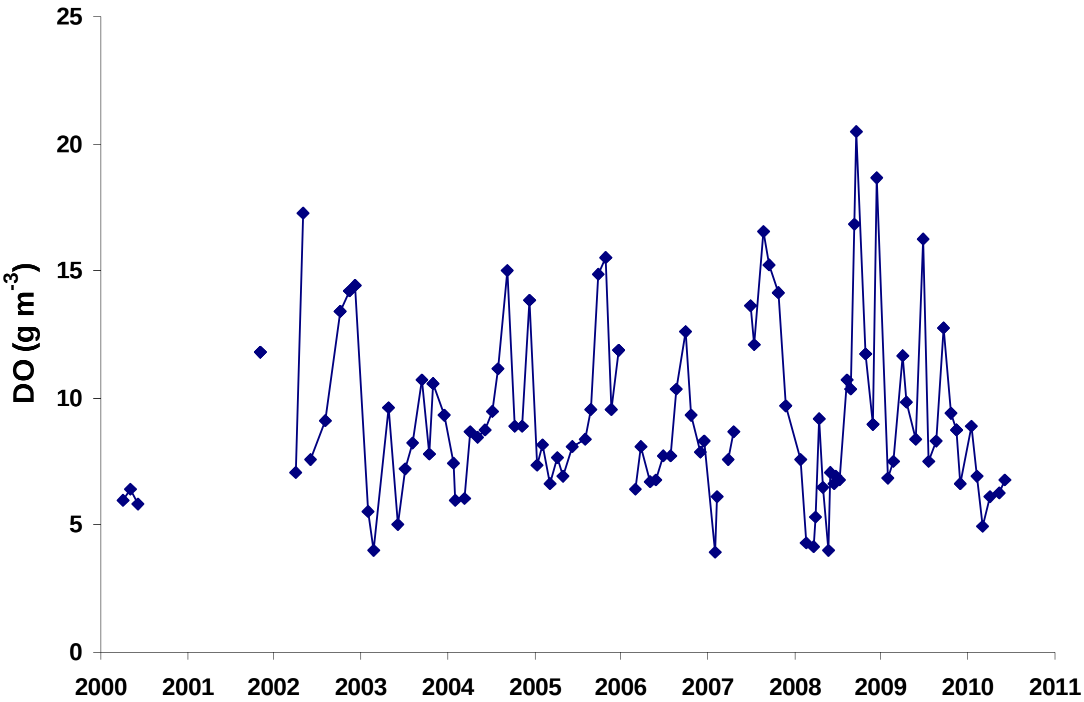


Figure 22. Time series Dissolved Oxygen (DO) concentrations in the Arawhata Stream. Extremely high DO during the day implies photosynthesis in dense plant / periphyton cover in the stream.

Typically, in lakes and rivers, the lowest DO concentrations should occur around dawn, as that is the longest period without photosynthesis. Few samples are ever collected at this time of day and few samples are ever collected at exactly the same time of day. However, it is possible to estimate the minimum DO concentration in the stream by regressing the DO concentration against the time of day sampled in summer i.e., mid December to mid April. In this case, the regression explains 60% of the variability and can be extrapolated through zero at around 6 am in the morning (Figure 23). Linear extrapolation is probably inappropriate but what this indicates is that the Arawhata Stream most likely becomes anoxic for a period at night between December and April, with the anoxia being broken by in-stream photosynthesis around sunrise. The time of the onset of anoxia each day is not known due to insufficient data. Given the rapid DO depletion in the lake (Figure 7) it is possible that the stream becomes anoxic before midnight on most days. This would be consistent with the recent accumulation of organic sediment in the stream (Jon Roygard, Horizons Regional Council, pers. comm.).

As a reality check on this scenario, there should be changes in the stream chemistry during summer which reflect these anoxic conditions. i.e., the expectation would be to see elevated DRP and NH_4-N concentrations in the stream water. In the 1988/89 data, DRP concentrations in the Arawhata Stream did not exceed 20 mg m^{-3} and NH_4-N concentrations were less than 30 mg m^{-3} during summer (December to April). In the 2000 to 2010 monitoring period, DRP concentrations were highly variable with spikes reaching up to 80 mg m^{-3} and on occasion these were matched with NH_4-N spikes of up to 400 mg m^{-3} (Figure 24). As no samples were collected at 6 am in the morning, these spikes represent the net concentrations after nitrification and plant assimilation in the stream channel and are consistent with the Arawhata Stream becoming anoxic at night in summer i.e., it is feasible, even though there is no measurement of the anoxia.

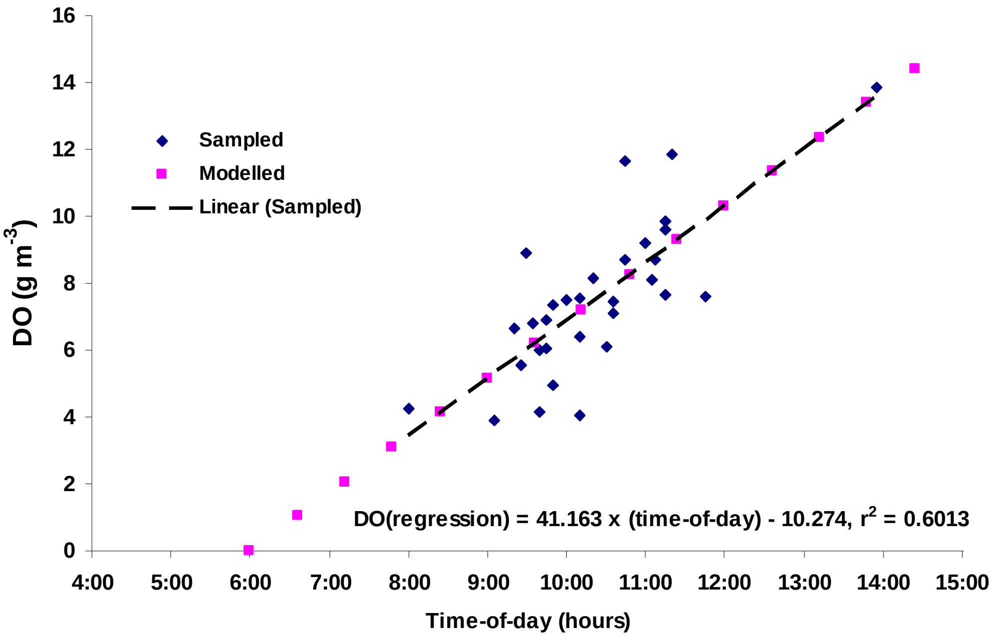


Figure 23. Dissolved oxygen data between mid December and mid April from 2000 to 2010. The linear regression through the data points (blue) was used to extrapolate the time DO values rise above zero (pink dots). Although 6:00 am is indicated, the actual time would most likely be around sunrise.

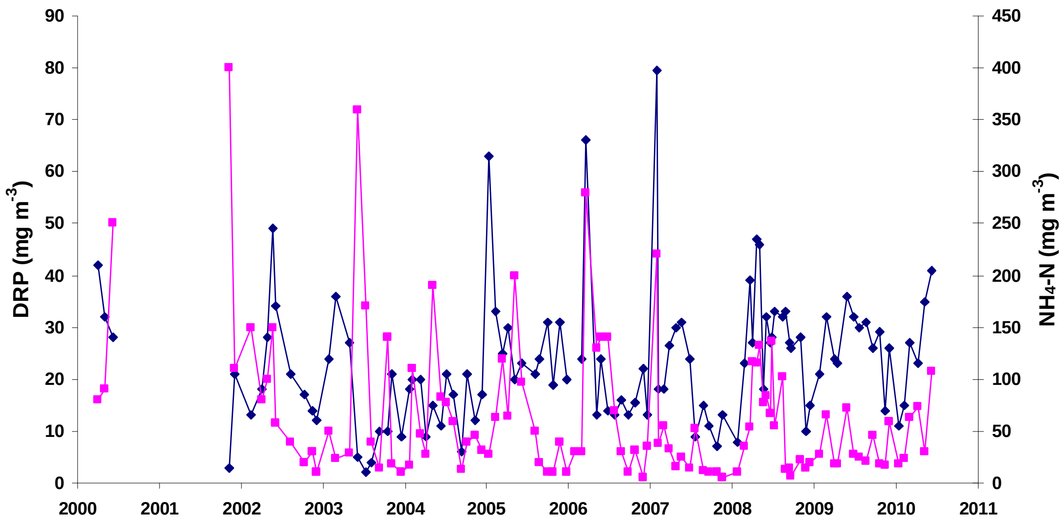


Figure 24. Time series of DRP and NH_4-N concentration data in the Arawhata Stream between 2000 and 2011.

High NH_4-N concentrations also occurred in the winter months (Figure 24) which implies contamination of the stream from sources other than sediment release. This requires further investigation if the lake is to be rehabilitated.

Because the flowing stream can become anoxic, this implies that the bottom waters in Lake Horowhenua almost certainly become anoxic at night in summer (section 3.1.2). In summer, at night the cooler stream water (Figure 25) would flow across the bed of the lake as a density current and enhance bottom water oxygen depletion. During the day, the stream water re-oxygenates but also warms so that it would only mix with the upper waters in the lake (Figure 25). The cooler stream water flowing as a density current across the lake bed at night may be the main driver of the thermal stratification observed in the lake in the mesocosm study (Figure 5) and, consequently, it may be a key element in the hypolimnetic oxygen depletion process. Although there is no DO data for the Patiki Stream during the March 2011 measurement period, the 2000 to 2010 monitoring data shows a lower mean DO with a smaller range for maximum and minimum than the Arawhata Stream (Table 3-2). Because the Patiki Stream is smaller than the Arawhata Stream, its impact on the lake would be expected to be much less. However, as it is always colder than the lake it would always enter the lake as a density current dispersing across the lake bed as a thin layer.

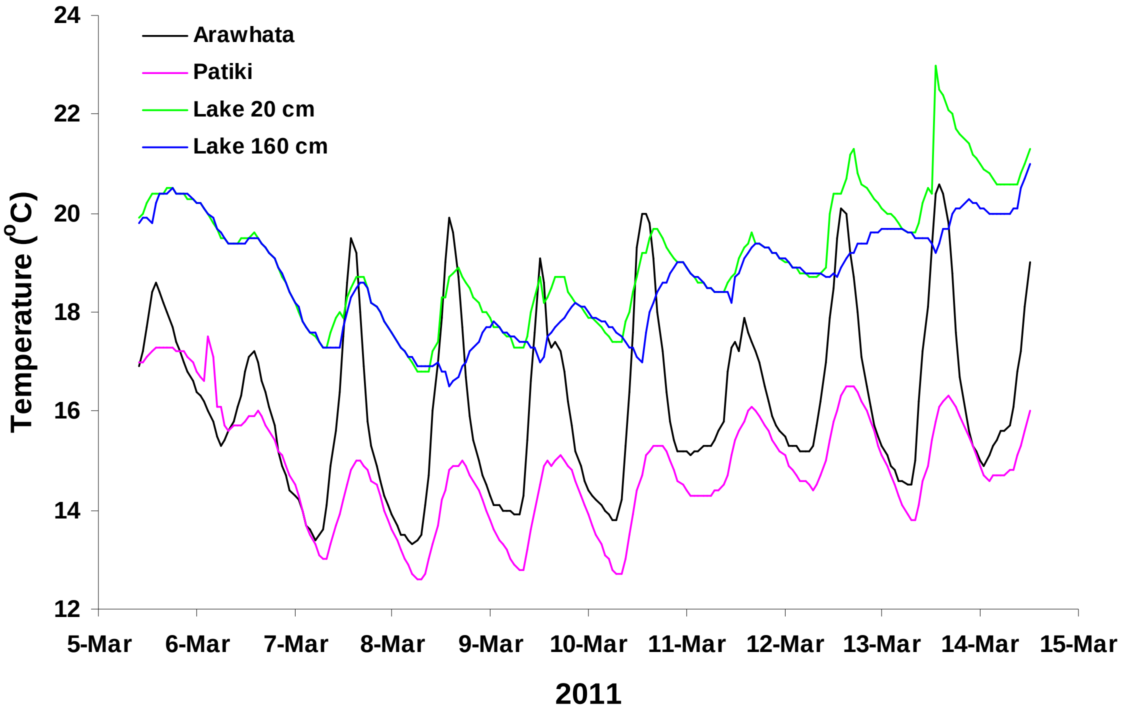


Figure 25. Temperature changes in a daily basis in the Arawhata Stream in March 2011. The temperature data are compared with those in the Patiki Stream and the open waters of the lake.

The significant difference in amplitude between the temperature fluctuations in the Arawhata Stream and the Patiki Stream can be attributed to shading, with the Arawhata is fully exposed to sunlight while the Patiki Stream is shaded by trees. Both streams have similar diurnal (day/night) temperature under cloudy or rainy weather (e.g., 5 to 8 March: Figure 25).

3.3 Other streams

A comparison of the nutrient concentrations in the major inflows to Lake Horowhenua in 1988/89 with those collected between 2000 and 2010 show significant increases in most parameters (Table 3-2). Not all streams sampled in 1988/89 were monitored during 2000/10 but there was sufficient data across all seasons for four inflows to show that catchment nutrient loads have increased. The largest increases have occurred in the Mangaroa Stream where mean DRP and TP concentrations increased by 300%, NO_3-N plus NH_4-N concentrations increased by 140% and TN concentrations increased by 130%. The decrease in NH_4-N concentrations in the 2000/10 data indicate greater nitrification converting NH_4-N to NO_3-N, which accounts for most of the increase in NO_3-N concentrations.

Table 3-2: Summary and comparison of nutrients and physical parameters of the inflows to Lake Horoewhenua in 1988/89 and between 2000/10.

Stream	Q	DRP	DOP	PP	TP	NO_3-N	NH_4-N	DON	PN	TN	Temp	pH	DO	SS
	L/s	mg/m^3	mg/m^3	mg/m^3	mg/m^3	mg/m^3	mg/m^3	mg/m^3	mg/m^3	mg/m^3	°C		g/m^3	g/m^3
1988/89 data														
Mangaroa Stream	32	57.6	16.8	65.9	140.2	2116	1472	1031	221	4841				
Pa Drain	32	12.0	5.7	15.9	33.6	5360	45	509	85	5999				
Patiki Stream	44	16.4	7.8	20.7	44.8	5444	50	554	98	6147				
Domain Drain	7	12.8	8.8	20.4	42.1	4265	87	554	95	5001				
Queen Street Drain	48	30.1	5.5	124.0	159.7	4281	58	347	533	5220				
Makomako Rd Drain	2	160.8	21.5	51.6	233.8	214	2207	579	167	3167				
Arawhata Stream	259	10.3	3.7	36.2	50.2	9851	157	391	104	10503				
Sand Road Drain	3	131.8	26.8	84.7	243.3	750	2128	767	244	3889				
2000/10 data														
Mangaroa Stream	12	173			427	5050	114			6452	10.4	7.25	12.3	
min		129			246	3110	28			4640		6.84	7.4	
max		273			1090	8825	220			9510		8.24	16.9	
Patiki Stream		31.2			82	6320	61.4			7214	12	7.13	8.56	24.1
min	20	5			18	20	13			4756	6.4	6.37	5.3	2
max	300	118			240	10800	178			9837	16.4	8.49	11.83	118
Queen Street Drain		22			66	4420	22			4535	15.1	6.4	12	
min		17			24	390	5			1200	9.4	4.53	7.52	
max	180	27			158	6490	72			6644	22.3	7.55	20.7	
Arawhata Stream	234	22.1			63.4	12960	61.8			13600	14.9	6.7	9.5	7.5
min	100	2			11	7350	5			7750	12.1	4.4	3.9	1
max	600	79.5			291	21000	400			21700	20.4	9.65	20.5	78

The Patiki Stream and Arawhata Streams also show significant increases in N and P nutrients (Table 3-2). The cause of these increases is uncertain but it is likely to be a response to dairy intensification in these catchments (Figure 2).

Another possible cause of increased nutrient concentrations in these streams is a reduction in flow. Increased groundwater abstraction for irrigation has the potential to reduce the flow in these streams. Unfortunately, there is insufficient data to properly assess this effect. However, there is data for mean flows in the Arawhata Stream in both periods which suggest about a 9% decrease in mean flow in the 2000 /10 period. Although that could account for a small increase in concentration, it does not account it all. This indicates that the increased concentrations are real and most likely due to changes in land-use since 1988/89.

3.4 Sediments

3.4.1 Historical Sediment chemistry

Sedimentation was seen as a problem in the 1970s with high turbidity in the lake. Estimates of the volume of sediment in the lake were augmented with an analysis of the TP composition of the sediment from a core 75 cm deep (Figure 26) taken in April 1971 (Gilliland, 1978). These data show the classic diagenesis of P i.e., the migration of mineralised P to the surface of the sediment where it is bound to iron oxides. Beneath the surface aerobic layer, the anoxic sediment pore water will have high concentrations of DRP. Sudden wind events / storms can suspend the surficial sediments and disperse this DRP into the lake water before it can be sequestered by metal oxides and returned to the sediment.

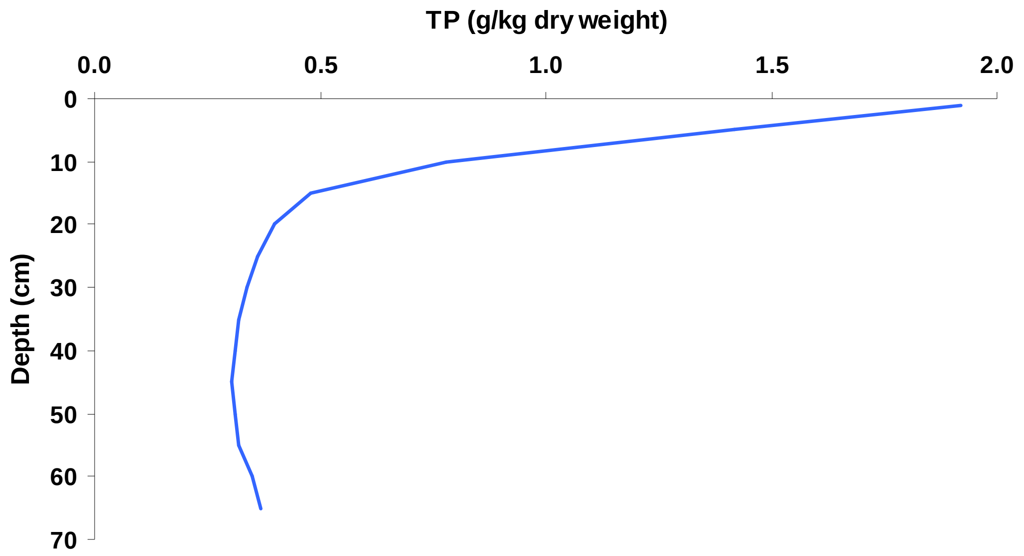


Figure 26. Estimated TP concentration (g/kg dry weight) versus depth profile in a sediment core collected from Lake Horowhenua in April 1971 (redrawn from Gilliland, 1978).

There is a linear relationship between TP and total iron (TFe) in the sediments (Figure 27) indicating that there is 1 g of TP for every 26.3 g of TFe. These data were from sediment cores collected in March 2011 to a maximum depth of 60 cm. This TFe:TP ratio holds true through the full depth of the sediment core.

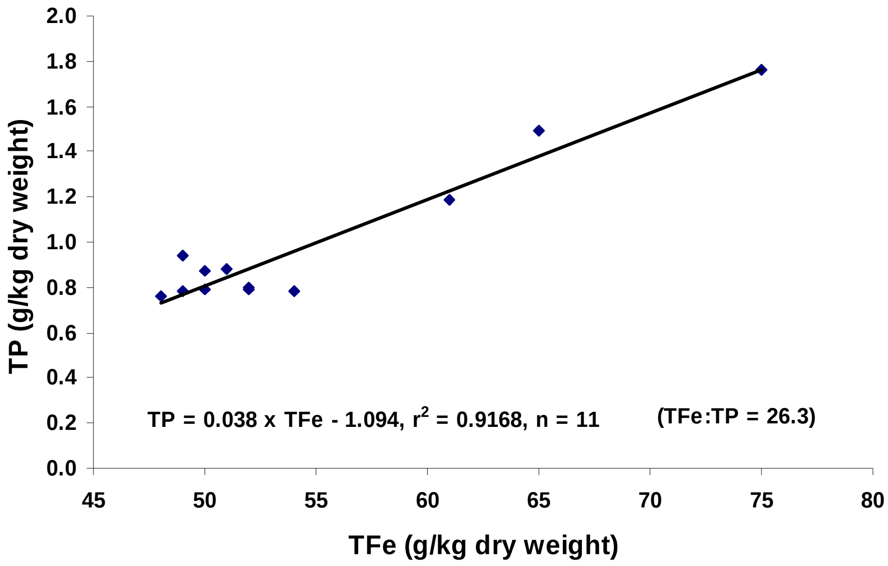


Figure 27. Total P (TP) versus total Fe (TFe) relationship in Lake Horowhenua sediments (2011 data).

During the DSIR/NIWA study in 1988/89, 19 sediment cores were taken across the lake in July 1988 (Figure 1) to determine N and P loads in the surficial lake sediment and the spatial distribution of N and P in the lake bed (Figure 28, Figure 29). The samples were sectioned and dried to estimate the total mass of P and N as g / kg dry weight of sediment and the total dissolved P and N in the pore water (Table 3-3) in the surface 1 cm layer.

Table 3-3. Mean sediment data from the upper 1 cm layer at 19 core locations in Lake Horowhenua 1988/89 (see **Figure 28**).

Parameter	Value	Units
Moisture	91.5	%
Dry weight	0.0877	g/cc wet sediment
Particulate P	1.29	mg/g dry weight
Particulate N	14.67	mg/g dry weight
Interstitial Total Dissolved P	1076.5	mg/m^3
Interstitial Total Dissolved N	11607.4	mg/m^3
Areal load of TP in top 1 cm	1.14	$g\ P/m^2$
Areal load of TN in top 1 cm	12.97	$g\ N/m^2$
Volume of lake	3.8×10^6	m^3
Surface area of lake	2.93×10^6	m^2
Area of lake bed below 0.8 m	2.35×10^6	m^2

Recent literature (e.g., Cooke et al., 2005) has shown that, although the P released from lake sediment comes from as deep as 10 cm depth, most of the P comes from the top 4-cm depth layer. Consequently, the mean potential mass of P that could be released into the water column of Lake Horowhenua could be more than 4.56 g m^{-2} (the mass in the top 4 cm).

However, the spatial distribution of P in the top 1 cm of lake sediment ranges from 0.65 g m^{-2} to over 2 g m^{-2}, which means the potential mass of P that could be released from the top 4 cm is likely to be more than 8 g m^{-2} in some areas. The spatial pattern is consistent with the discharge point of sewage effluent being near the Makomako Road drain. Based on the mean value of 4.56 g m^{-2}, there is more than 13 t P that could be released from the top 4 cm and this means that the P concentration in the lake has the potential to reach 3500 mg m^{-3}. The maximum TP concentration recorded in the lake was 760 mg m^{-3}.

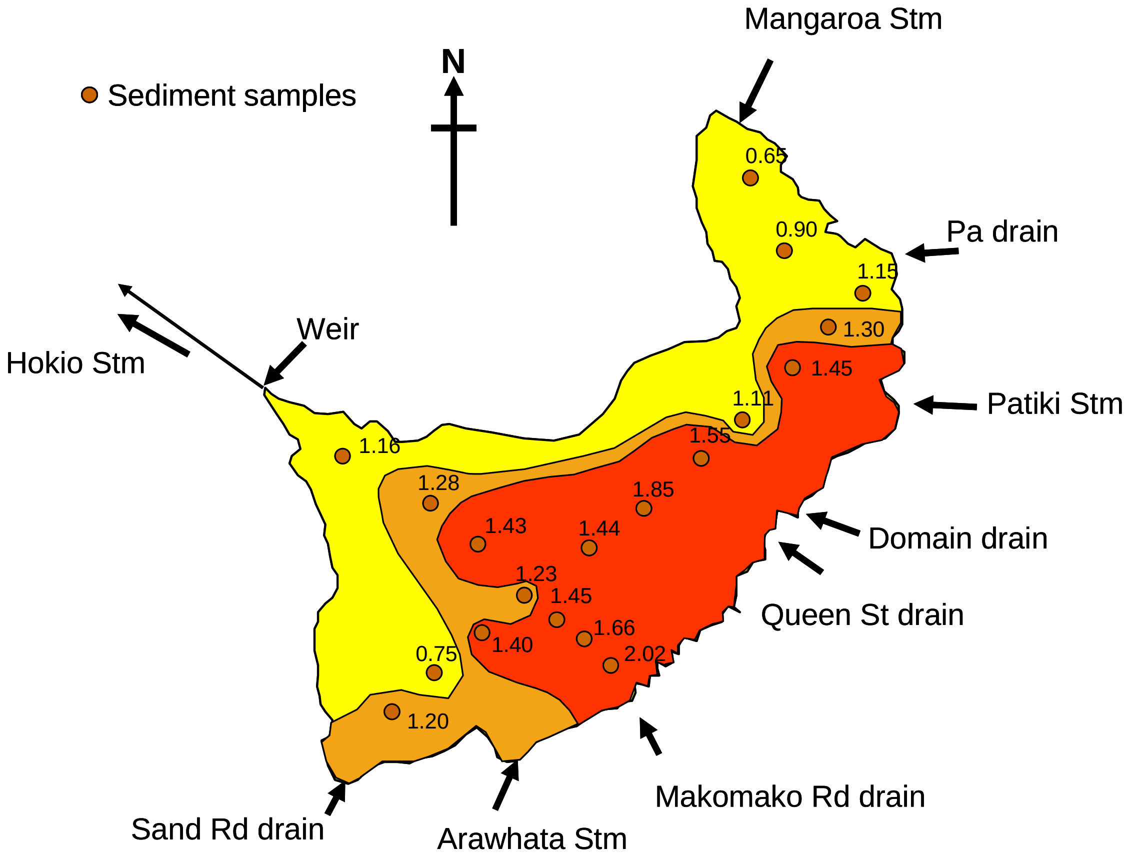


Figure 28. Best estimate of spatial distribution pattern of TP (g/kg dry weight in top 1 cm) in July 1988. This pattern draws on knowledge that the sewage effluent was discharged into the lake near the Makomako Road drain and that the prevailing south westerly wind caused a persistent drift of algal biomass onto the eastern shores (e.g., **Figure 8**).

The sudden increases in DRP concentrations in Lake Horowhenua without associated inorganic N increases (e.g., Figure 16) is consistent with P released from sediment when the lake bottom water becomes anoxic.

As a reality check, the upper 1 cm thick layer of sediment contained high concentrations of dissolved N, mostly as NH_4-N which is rapidly nitrified to NO_3-N, and P, mostly as DRP (Table 3-3). If wind events were stir the lake vigorously, it is possible for the surficial 1 cm sediment layer to be mixed up into the lake dispersing the P and the N in the interstitial water throughout the lake water column. Based on the mean interstitial water data (Table 3-3), such an event would raise the DRP and NO_3-N concentrations by 7.6 mg m^{-3} and 81.9 mg m^{-3} respectively. The DRP increase due to sediment resuspension is less than 2% of the maximum DRP value (Figure 16) while the NO_3-N increase would be several orders of magnitude greater than the observed concentrations (Figure 16). This suggests that mixing events are unlikely to be the direct source of the DRP increases in the lake in summer.

The distribution of TN in the lake sediment was similar to the distribution of TP (Figure 29) with extremely high TN values near the sewage effluent discharge point near Makomako Road drain.

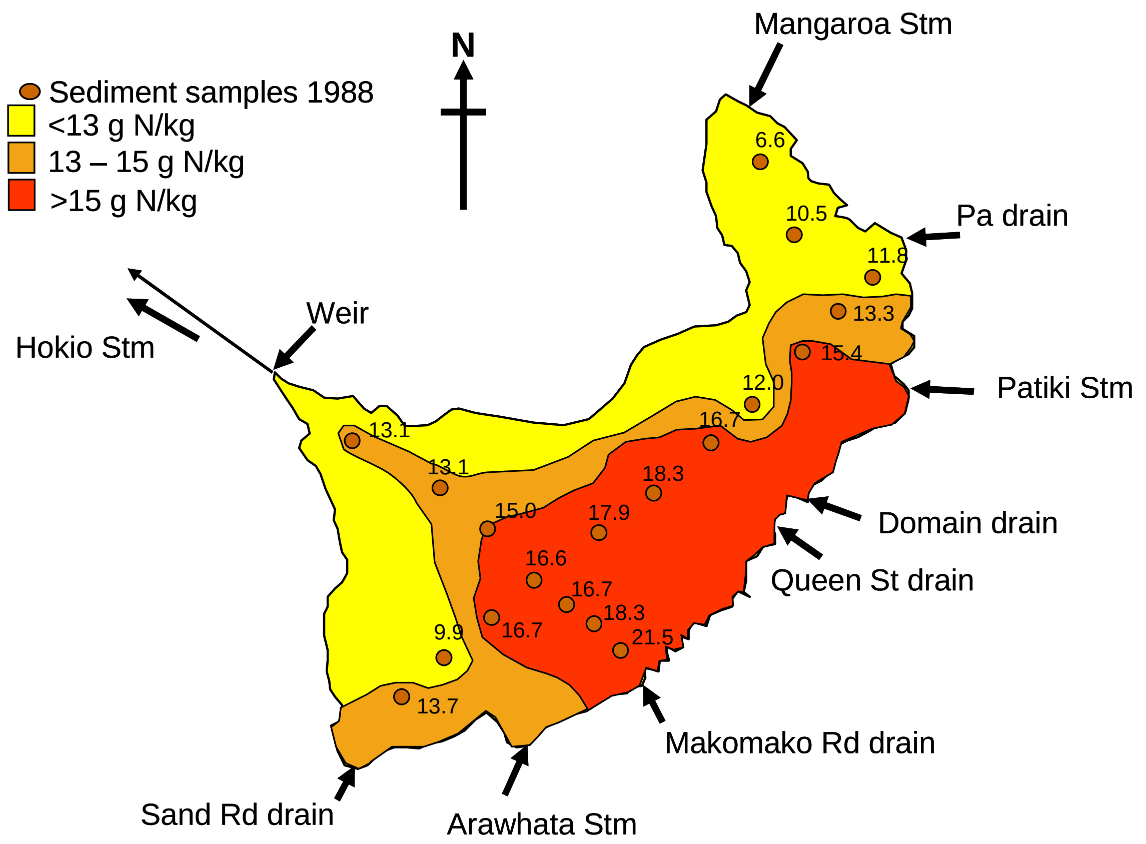


Figure 29. Best estimate of spatial distribution pattern of TN (g/kg dry weight in top 1 cm) in July 1988. This pattern is similar to the distribution pattern of TP (**Figure 28**).

The organic nitrogen in sediment is typically mineralised to NH_4-N which diffuses out of the sediment. As the NH_4-N leaves the anoxic sediments it is microbially oxidised to NO_3-N i.e., nitrification. If this NO_3-N comes into contact with the anoxic sediments it can be microbially converted to di-nitrogen gas (N_2) i.e., denitrification. When the overlying water is well oxygenated, the surface few mm of sediment are also oxygenated and the nitrification – denitrification process occurs within the sediments. Not all of the NO_3-N is denitrified and there is an accumulation of NO_3-N in the lake water column. Unlike the release of P which only occurs when the overlying water is anoxic, the efflux of N as NO_3-N from the sediment continues all year with the rate of release being mediated by the lake temperature. As a consequence of these different release rate processes, the sediments lose N and the ratio of TN:TP reduces from a natural plant ratio of around 16:1 and may fall as low as 5:1 if the sediment doesn't receive a new input of organic plant material.

The TN:TP ratios in the sediments of Lake Horowhenua average around 11.4 and range from 9.8 to 14.8. The spatial distribution of the mean TN:TP ratios separated as higher or lower than 11 shows a pattern with the higher values on the western side and the lower values on the eastern side (Figure 30). The value of 11 was chosen for this analysis as it illustrates input and removal processes associated with this lake.

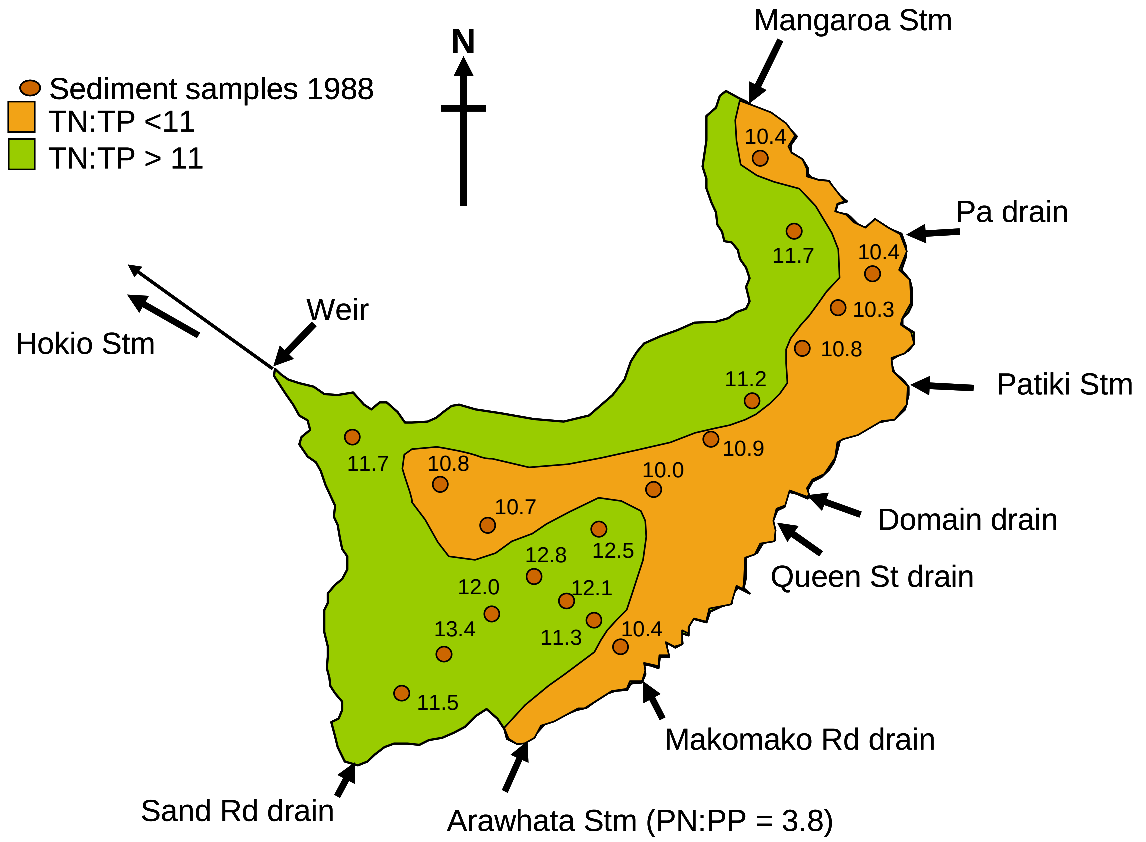


Figure 30. Spatial distribution of mean TN:TP ratios separated into more or less than 11.

The TN:TP distribution pattern is a good approximation to the distribution of macrophyte weed beds in the lake in 1988/89 (personal observations). The cluster of <11 ratios in the centre of the lake correspond with the deepest part of the lake. The inclusion of area adjacent to the Arawhata Stream input in the <11 group is based on the stream having a PN:PP ratio of around 3.8 in the suspended sediment (DSIR/NIWA 1988/89 data).

The TN:TP ratio distribution pattern indicates that sediment from the streams on the eastern side of the lake are nutrient enriched and have experienced nitrification and denitrification before reaching the lake and in the in-shore waters of the lake. In contrast, sediments on the western side of the lake may contain higher proportions of more recently deposited plant material from decaying weed beds after they collapse in autumn.

3.4.2 Recent sediment chemistry

Following the diversion of the sewage effluent out of the lake in 1987, Vant and Gilliland (1991) estimated that the P load on the lake had been reduced by 90%. Such a sudden change in the external loadings on the lake should have been reflected in a change in TP concentrations in the sediment. Sediment samples taken at 4 location across Lake Horowhenua in March 2011 had a mean TP concentration of 0.72 g kg which is a decrease to 56% of the mean TP concentration of 1.29 g kg^{-1} (Table 3-3) in the surficial sediments in 1988 just after the sewage input was stopped. The TP concentrations in a 70 cm deep sediment core collected in March 2011 (Figure 31) have a maximum concentration at a depth of between 30 and 40 cm, which may be attributed to the TP concentration in the sediment when the sewage input was stopped.

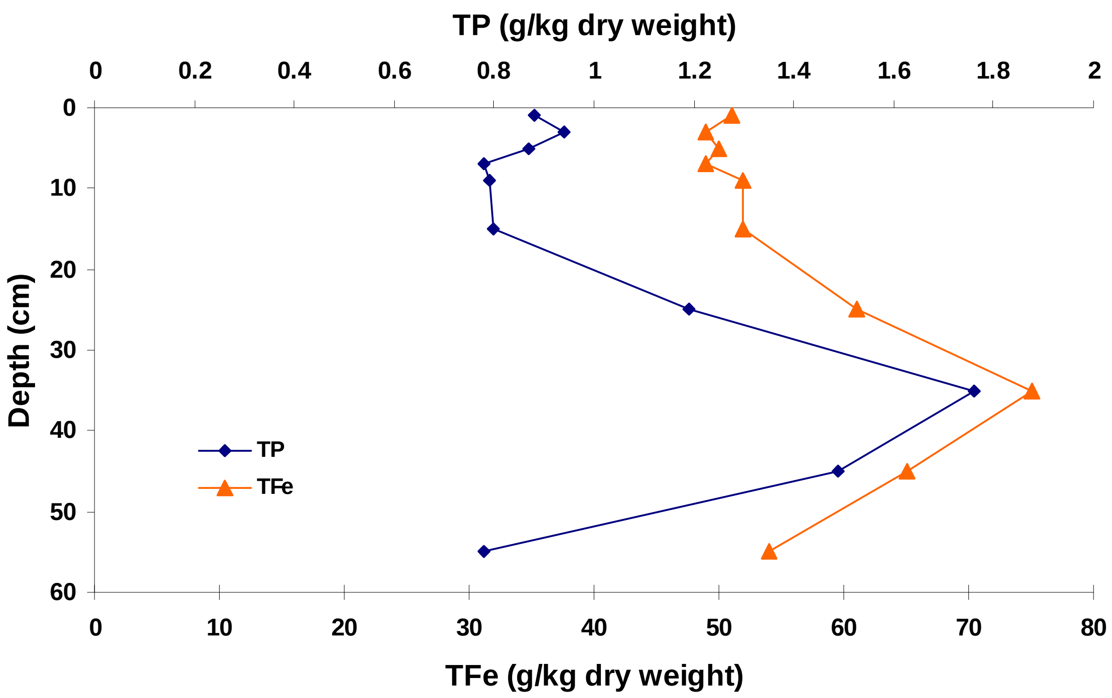


Figure 31. Depth profile of total P (TP) and total iron (TFe) from a mid-lake core collected in March 2011.

The sudden decrease in TP concentration after the sewage input was stopped suggests that there was rapid burial which prevented vertical migration of the DRP up through the sediment with time. Comparison of the March 2011 profile with that taken in April 1971 (Figure 32) suggests that peak has been buried by around 40 cm in 40 years, i.e., a mean burial rate of 10 mm /yr. This burial rate, although high by lake standards (typically 1–2 mm yr^{-1}), is feasible in the middle of the lake and would most likely be due to a combination of high suspended sediment input loads and redistribution of in-lake sediments associated with sediment focusing into the deeper parts of the lake.

As a reality check, Brougham & Currie (1976) estimated that the mean suspended solids load on Lake Horowhenua was around 232 kg/day, which represents a sediment accumulation rate of 3.3 mm yr^{-1} over the whole area of the lake, using the sediment dry weight factor of 0.0877 g/cc (Table 3-3). Sediment focusing from the shallow edges into the deeper parts of the lake could plausibly increase the local sediment accumulation rate to around 10 mm yr^{-1}.

Brougham & Currie (1976) also estimated that in the 1970s, silty sediment occupied about half of the original volume of the lake. The source of that sediment was from sewage discharge, runoff from pasture and dairy sheds, and land-use practices which allowed drainage ditches to be dug through wetlands and swamps, and cows and pigs to have direct access to the lake and streams.

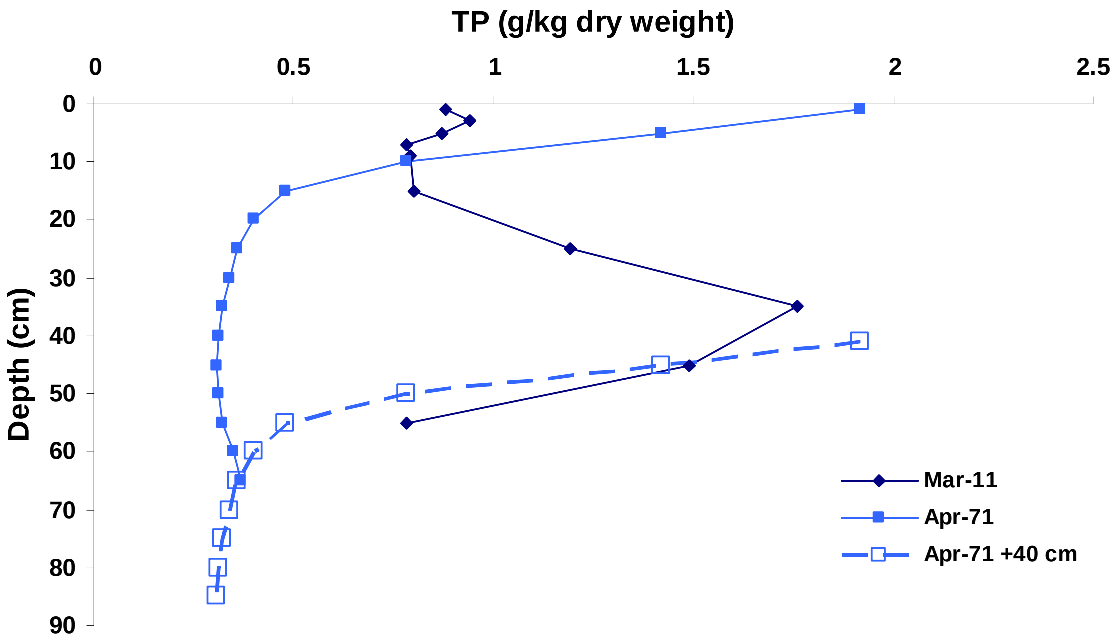


Figure 32. Comparison of the TP concentrations in sediment cores taken in April 1971 and March 2011. The broken line is the April 1971 data with a vertical offset of 40 cm to represent the effects of burial by 10 mm per year.

3.4.3 Natural recovery rate

From the sediment data, it is apparent that the internal P load in Lake Horowhenua began reducing naturally through burial by P-depleted sediments and export of P in algal biomass discharged down the Hokio Stream.

On an areal basis the mean internal P concentration in the top 1 cm has decreased from 1.14 g m^{-2} in 1988 to 0.64 g m^{-2} in 2011. Although this represents a 47% decrease, the present internal load is still too high to prevent cyanobacteria blooms in summer and, consequently, the internal load must be reduced further, possibly to around 5% of the initial internal load. The exact value would need to be modelled to estimate at what point algal biomass was reduced to a publically acceptable level.

As most natural decay processes operate at an exponential rate, i.e., they are fastest when they begin but slow down as the end point approaches zero, it can be assumed that the natural recovery of Lake Horowhenua will take many years. How many years can be estimated by applying an exponential decay curve to the known data points, assuming 1988 was 100% and 2011 has 53% of the internal load remaining, and then selecting a target load which the internal load will eventually reach – say 5% of the initial 1988 internal load. The exponential curve fitted through these points (Figure 33) suggests that it would take about 120 years to reach the target load of 5% from the start. This means that the expectation of getting to 5% of the initial internal load will take about 97 years from present.

This scenario assumes that the external P loads to the lake remain constant or do not increase across that period of time. As that is highly unlikely, the time estimate will vary depending on the inputs from the catchment. Consequently, the model prediction (Figure 33) primarily serves to illustrate that rehabilitation of Lake Horowhenua to improve its water quality in a shorter time frame than 100 years will require management interventions.

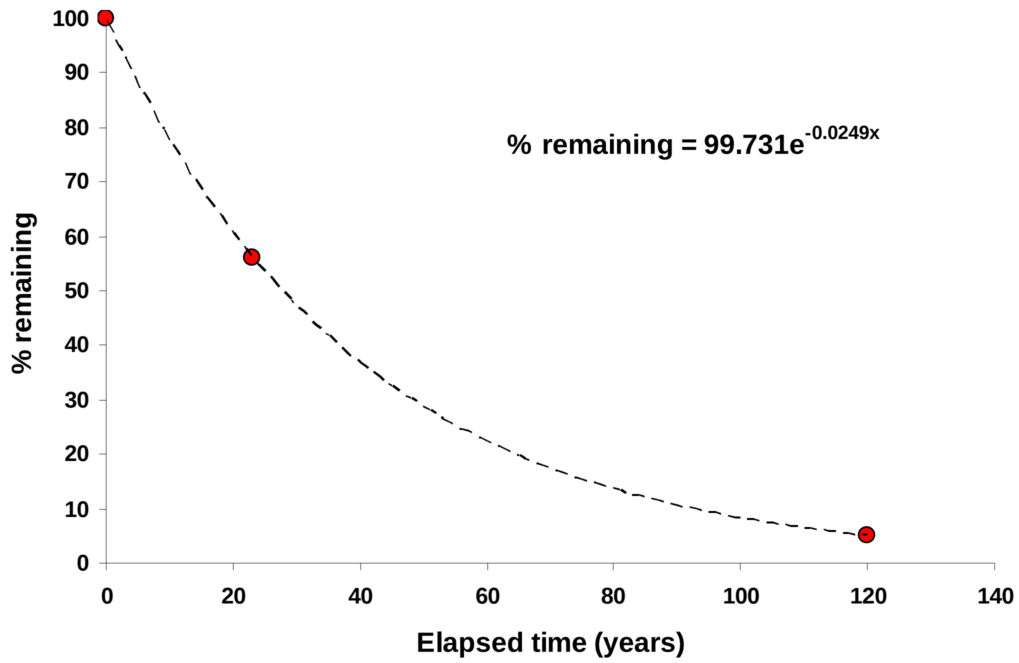


Figure 33. Exponential decay curve fitted through known data points and a target point of 5% remaining in order to estimate the time to recovery. The curve equation is shown on the figure.

3.4.4 Sediment infilling

Based on the earlier estimate of mean sediment accumulation rates of 3.3 mm per year (section 3.4.2) across the whole lake and up to 10 mm per year in the centre, a linear extrapolation would suggest that the lake could completely fill up with sediment in the next 100 to 150 years. Complete infilling is unlikely.

As the lake gets shallower, it is easier for wind waves to suspend the surficial sediment which can be flushed down the Hokio Stream. In practice this means that the lake will reach an equilibrium point at which there is a balance between a minimum depth and the level of turbidity i.e., without catchment interventions to reduce the sediment inputs, the lake will become more turbid as will the Hokio Stream. Because suspension of sediments can release DRP from the sediment pore water (Sun et al., 2006), this scenario is also likely to induce increased cyanobacteria blooms, unless in-lake interventions can inactivate the P in the sediment.

3.4.5 Overview

- Historically, Lake Horowhenua was reported to be a clean-water lake with drinkable water, and a diverse fishery.
- Now, the lake is hypertrophic and the fishery is greatly diminished.
- The water quality decline was caused by removal of the surrounding forest combined with allowing stock access to the lake, developing intensive horticulture and market gardening in the catchment, and discharging urban storm water runoff and sewage effluent into the lake.
- After 25 years, the sewage effluent input stopped in 1987.
- The lake has a large external N load via streams and groundwater.
- Groundwater contributes between 60% and 80% of the hydraulic load on the lake and about 50% of the mean annual N load.
- Contaminants in the groundwater have a long lag time before emerging and these will continue to pollute the lake for many years after the surface contaminants stop.
- The Arawhata Stream is the largest surface inflow and is the largest single source of N to the lake. Recent data suggests that the stream may become anoxic at night and may enhance oxygen depletion in the lake.
- Monitoring data shows that the nutrient loads in some inflows have increased significantly in recent years, possibly attributable to increased dairy farming.
- Oxygen depletion in the lake is most likely caused by the collapse of the macrophyte weed beds in summer decomposing in the water column and blocking oxygen diffusion into the sediment where they lie on the lake bed.
- The lake sediments have a large load of N which is being reduced by nitrification and denitrification but is being replenished by the suspended sediment input from the catchment and the weed beds in the lake when they collapse each autumn.
- The lake has a large internal P load which is being released under anoxic conditions and is driving the production of cyanobacterial blooms in summer.
- Burial of the P by new sediment and flushing the P out of the lake via the Hokio Stream outflow are the only natural mechanisms to reduce P availability for algae.
- Natural restoration processes are slow and it may take up to 100 years for the lake to recover without management intervention.
- Management interventions need to reduce external loads of nutrients and sediment at the same time as in-lake interventions reduce the internal P loads.
- Although the lake has been monitored sporadically in the past, there is currently no monitoring programme of water quality to assess the effectiveness of any management intervention to rehabilitate the lake.

4 Lake restoration

Lakes provide people with many services: aesthetic enjoyment, recreation, fish, transportation, water for irrigation, drinking and dilution of pollutants. Lake degradation results from excessive nutrient inputs, toxic substances, habitat loss, overfishing, species invasions and extirpations. The goal of management is to balance the uses of lakes with conservation measures to sustain ecosystem services over time (Carpenter & Lathrop 1999).

The scientific basis of lake degradation is generally well understood, although each restoration project requires some level of new site-specific research. Remediation may require management actions which are difficult to implement for social or institutional reasons. While there is an implicit expectation that **restoration** will return the lake to its original pristine condition, the reality is that some processes on the way to degradation are irreversible (Carpenter et al., 1999). Consequently, the expectation of restoration needs to be tempered with the knowledge that the lake may never reach its original pristine state and that the objective of a restoration project will be to **rehabilitate** the lake to improve the water quality and lake conditions to an achievable level. This rehabilitation level then becomes a management goal.

Research studies in the international literature can provide understanding of lakes, their catchments and the mechanisms that sustain ecosystem services; the causes of lake degradation; and methods and technologies for lake restoration. From a lake perspective, the ecosystem is in equilibrium and changes to any aspect of the lake ecosystem will cause the lake to adjust until a new equilibrium is established (Duarte et al., 2009).

Such changes take time and, although a remediation technique may appear to offer a “quick fix”, the resultant new equilibrium may not be reached for several years. These lags and delays may appear to be failures but may actually be unforeseen bottlenecks requiring new techniques to be developed (Gulati et al., 2008).

Traditional techniques relying on management of land based nutrient sources to reduce algal biomass in the lake are being augmented with biomanipulation techniques which use natural biological processes to target specific processes in the lake that will culminate in a water quality improvement (Jeppesen et al., 2007). The success of such techniques vary with lake size and climate.

Within these literature studies there will be a range of techniques which could be applied to Lake Horowhenua. These techniques provide management with opportunities to address the water quality in Lake Horowhenua. These techniques and various management strategies will be discussed later in this section.

A key point which is fundamental to the restoration of any degraded lake is to “turn off” the sources of nutrients entering the lake at the same time as the in-lake interventions are being implemented. This is especially important for Lake Horowhenua which receives more than 50% of its hydraulic load from groundwater. Reducing the nutrient input to the ground and thus groundwater must be part of any rehabilitation strategy used to restore the water quality of the lake.

4.1 Water quality and trophic condition

New Zealand lakes are classified by trophic level based on a combination of 4 key variables: chlorophyll *a*, TN, TP, and Secchi depth (SD) (Table 4-1).

Table 4-1. Lake classifications, trophic levels and values of the four key variables that define the different lake classifications. From Burns et al. (2005).

Lake classification	Trophic level	Concentration (mg m^{-3})			Secchi depth (m)
		Chl *a*	TP	TN	
Ultra-microtrophic	0.0 – 1.0	0.13 – 0.33	0.84 – 1.8	16 - 34	24 - 31
Microtrophic	1.0 – 2.0	0.33 – 0.82	1.8 – 4.1	34 - 73	15 - 24
Oligotrophic	2.0 – 3.0	0.82 – 2.0	4.1 – 9.0	73 - 157	7.8 - 15
Mesotrophic	3.0 – 4.0	2.0 – 5.0	9.0 - 20	157 - 337	3.6 – 7.8
Eutrophic	4.0 – 5.0	5.0 – 12.0	20 - 43	337 - 725	1.6 – 3.6
Supertrophic	5.0 – 6.0	12.0 – 31.0	43 - 96	725 - 1558	0.7 – 1.6
Hypertrophic	6.0 – 7.0	>31	>96	>1558	< 0.7

Logarithmic transformation of the mean annual data for these variables provides a numerical index i.e., Trophic Level Index (TLI) as a number to represent the lake classification. The equations used are:

- Nitrogen TLn = -3.61 + 3.01log10 (TN)
- Phosphorus TLp = 0.218 + 2.92log10 (TP)
- Chlorophyll a TLc = 2.22 + 2.54log10 (Chla).
- Secchi depth TLs = 5.10 + 2.27log10 (1/ZSD – 1/40)

The four resulting numbers are then averaged to form the TLI. This is known as the TLI4 (makes use of four variables). There is also a TLI3, which excludes Secchi depth (Verburg et al., 2010).

Lake Horowhenua is classified as hypertrophic which means that it will have chlorophyll *a* concentrations greater than 31 mg m^{-3}, TP and TN concentrations greater than 96 mg m^{-3} and 1500 mg m^{-3} respectively, and Secchi depth values less than 0.7 m. In 2005, the TLI3 value for Lake Horowhenua was 6.27, and the lake was ranked 107 out of 114 monitored lakes (Verburg et al., 2010).

Comparison of historic and present data for Lake Horowhenua (Table 4-2) shows the changes that have occurred between 1988 and 2000 and over the period of continuous monitoring from 2000 to 2009. Almost all of the data fall within the range of hypertrophic (Table 4-1). The exceptions were the Secchi depth data which were greater than expected.

The mean annual data converted to TLI values (Table 4-2) indicate that the lake had begun to recover between 1988 and 2000. The TLI value in 1988 was 6.28 and in 2000 it was 5.88. However, after 2000 there is a statistically significant (p <0.01) trend of TLI increase indicating that the water quality in Lake Horowhenua has been declining over the last 10 years (Figure 34).

Note that there is large interannual variability in the mean annual data. This variability raises the level of uncertainty about the trend observed in the TLI.

Table 4-2. Mean annual Chlorophyll *a*, TP, and TN concentrations (mg m^{-3}) plus Secchi depth (m) data for 1988 and from 2000 to 2008. These data have been log transformed to produce a TLI value.

Date	Chla	TP	TN	SD	TLc	TLp	TLn	TLs	TLI
1988	60	253	2767	0.49	6.74	7.24	6.75	4.40	6.28
2000	26.8	131	2450	0.65	5.85	6.40	6.59	4.68	5.88
2001	61	191	1990	0.72	6.75	6.88	6.32	4.78	6.18
2002	27.3	144	2160	0.8	5.87	6.52	6.43	4.88	5.92
2003	70	199	2820	0.7	6.91	6.93	6.78	4.75	6.34
2004	117	272	3780	0.73	7.47	7.33	7.16	4.79	6.69
2005	63	113	4660	0.68	6.79	6.21	7.43	4.72	6.29
2006	64	213	2810	0.63	6.81	7.02	6.77	4.64	6.31
2007	110.7	299	3730	0.73	7.41	7.45	7.14	4.79	6.70
2008	211	150	3370	0.79	8.12	6.57	7.01	4.87	6.64

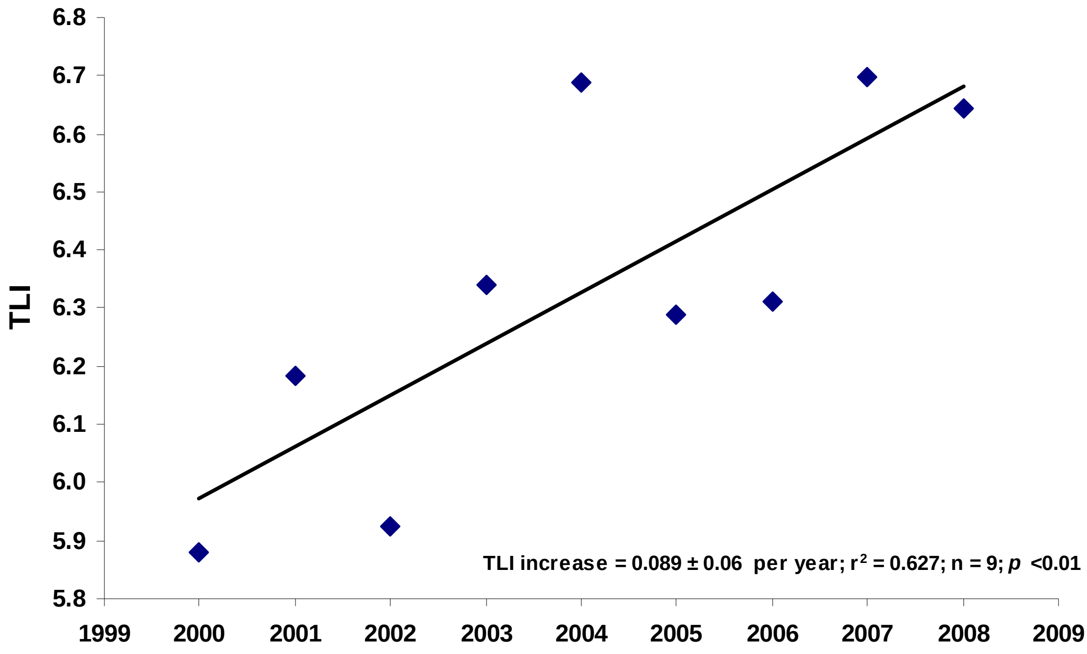


Figure 34. Trend analysis using a linear regression through the annual TLI values from 2000 to 2008 suggests that TLI is increasing and thus the water quality is declining. The trend is statistically significant (*p* <0.01). (Regression details on the figure).

4.2 Management goals

Management goals for Lake Horowhenua would include:

1. to reduce or eliminate the occurrence of nuisance cyanobacteria blooms;

2. to improve the water quality of the lake from hypertrophic to supertrophic or eutrophic i.e., reduce the TLI for the lake from its current level of 6.7 to <5;

3. to reduce the abundance of aquatic macrophytes in the lake to enable unimpeded use of the lake for contact recreation, and

4. to maintain or enhance the fishery in the lake and its tributaries.

These goals are essential to restoring the mauri of the lake. They are closely linked such that changes made to achieve one goal will interact with and affect the other goals. For example, while achieving goal 1 will be a measure of the success for goal 2, goal 3 may be negatively impacted by this success as increased light levels through clearer water will allow aquatic macrophytes to grow and spread. The overall effect on the fishery, goal 4, will depend on the new balance between competing aspects of the lake ecology.

It is unlikely that a single action will rehabilitate the lake and any management strategy will need to include the use of several management tools in the lake and in the lake catchment.

4.2.1 Improving the water quality

The trend of decreasing water quality indicated by the TLI trend analysis (Figure 34) is in contrast to the reducing pool of P in the near surface sediments indicated by the sediment core analyses. While about 45% of the P input from external sources comes from storm water (Brougham & Currie, 1976), the review of the water quality data clearly identified the linkage between the release of P from the lake sediment (the internal load) during periods of bottom water anoxia and the growth of nuisance cyanobacteria blooms. This implies that to meet the first goal, remedial action should include a focus on stopping the P release from the sediment in summer.

If TP in the near surface layers of the lake sediments follow the natural decay curve (Figure 33), the release of DRP from the sediment may not reach acceptable levels for about 100 years. Because most of P comes from the internal load, to reduce this time frame significantly, management strategies will need to include in-lake interventions, examples of which will be discussed later. Stopping the release of P from the lake sediment has the potential to lower the TP concentrations and thus the chlorophyll *a* concentrations in the lake. Applying these criteria to the 2000 to 2008 data, without changing the TN or SD data, suggests mean annual chlorophyll *a* values could reduce to around 9 to 17 mg m^{-3} and mean annual TP values would be around 52 to 83 mg m^{-3}. These values translate into a mean TLI of around 5.5, which implies a water quality improvement from hypertrophic to supertrophic.

A TLI value of around 5.5 is above the value of 5 set in goal 2. Reducing the TN would improve the water quality further and would lower the TLI. However, reducing the TN poses a problem because of the high N concentrations in the groundwater and surface water inflows. For example, in 1988/89 the mean TN concentrations in the Arawhata stream and groundwater, which account for more than 80% of the inflow, were around 10500 mg m^{-3} and 8450 mg m^{-3}, respectively. Between 2000 and 2009, the mean TN concentration in the

Arawhata Stream was around 13500 mg m^{-3} which indicates that the external N input has been increasing in recent years consistent with the TLI trend (Figure 34). This increase in the TN loads in the Arawhata Stream coincides with dairy intensification around the Arawhata Stream in recent years. The paucity of data and high seasonal variability in the available TN data in the Arawhata Stream (Figure 35) makes such a trend difficult to detect although a step change may have occurred.

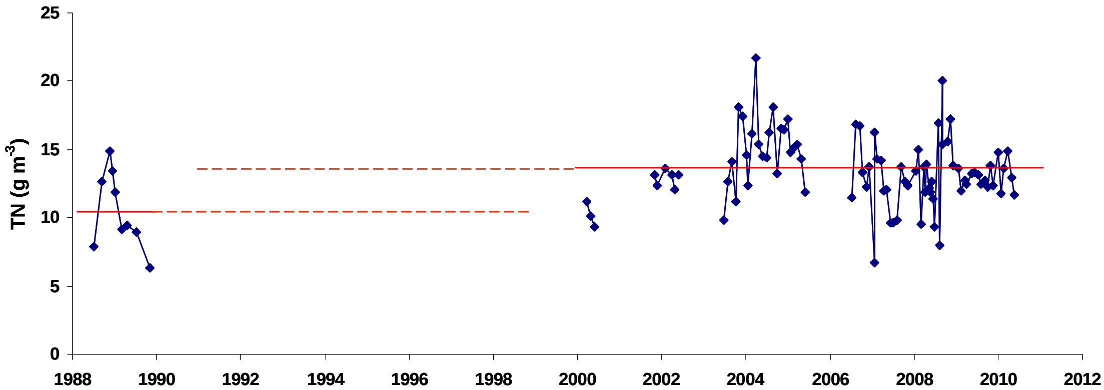


Figure 35. Time-series total nitrogen (TN) concentrations in the Arawhata Stream inflow to Lake Horowhenua. The dashed lines are means for the two data periods.

Notwithstanding this, while there is no significant trend of increasing TN concentrations in the Arawhata Stream, there is also no indication that the TN concentrations are reducing. Consequently, with a mean residence time of around 47 days and the flushing waters having such high TN concentrations, the TN component of the TLI for Lake Horowhenua is likely to remain in the hypertrophic classification without management intervention.

Because most of the N input to the lake comes from the catchment via streams and groundwater, and most of the streams receive groundwater inputs, that management intervention will need to focus mostly on the lake catchment and ways to protect surface waters and reduce contamination of the surface groundwater aquifer. Management strategies will also need to consider buffer zones to intercept the groundwater contaminants before they enter the lake until the catchment strategies become effective.

4.2.2 Reducing aquatic macrophytes

The aquatic macrophytes or “lake weeds” of concern in Lake Horowhenua are *Potamogeton crispus* and *Egeria densa* (Figure 36). Both are rooted plants with the ability to grow to the surface where the *Potamogeton* emerges to flower. While *Potamogeton* is well established in the lake, it is uncertain whether or to what extent *Egeria* has become established.

The spread of *Potamogeton* is by seed, which can rest in the sediment for several years until conditions are right for germination, but more commonly from pieces of plant broken off the main weed bed by wave action or the grazing of swans and other water fowl or during weed harvesting. Fragments of lake weed can survive considerable periods out of water and are often transferred between lakes on boat trailers and leisure activity equipment such as fishing gear. Eel traps moved from one lake to another could spread the weed, as will the misguided aquarium keeper who discards fragments in open drains or even empties a goldfish bowl into the lake. This is also often how pest fish are spread.

Management of the spread of lake weed is one of public education and making people aware of the consequences of their actions.

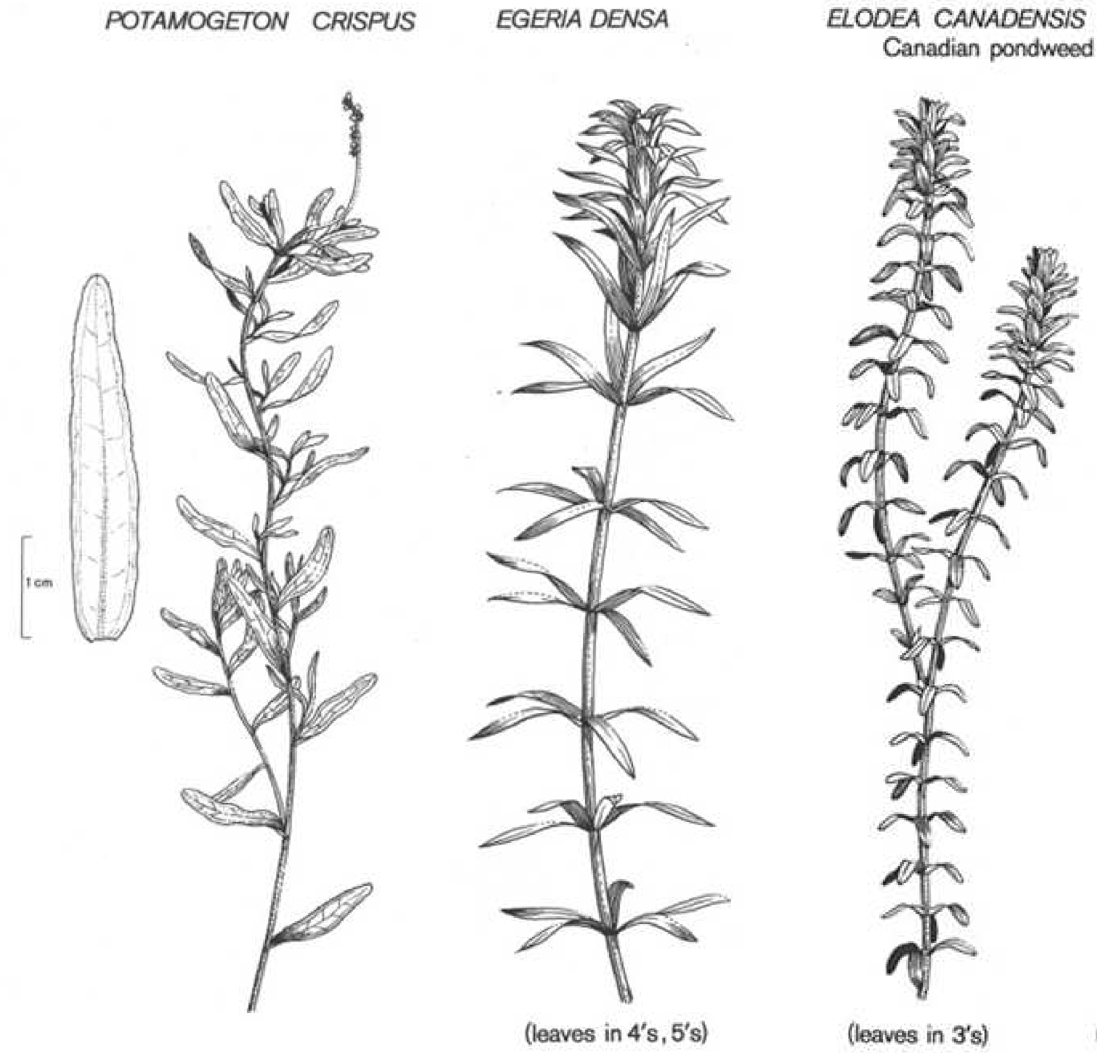


Figure 36. *Potamogeton crispus* (left), *Egeria densa* (middle) and *Elodea canadensis* (right) (drawings from Mason & West 1973).

Potamogeton crispus has been in Lake Horowhenua for many years. The annual cycle is for growth in spring when it rises to the surface to flower. This growth pattern appears to be a rapid spreading of the plant as was noted by Brougham & Currie (1976) (Figure 37). Note the similarity between the right hand distribution map (10.2.75; Figure 37) and the spatial distribution of TN:TP ratios in the lake sediments (Figure 30).

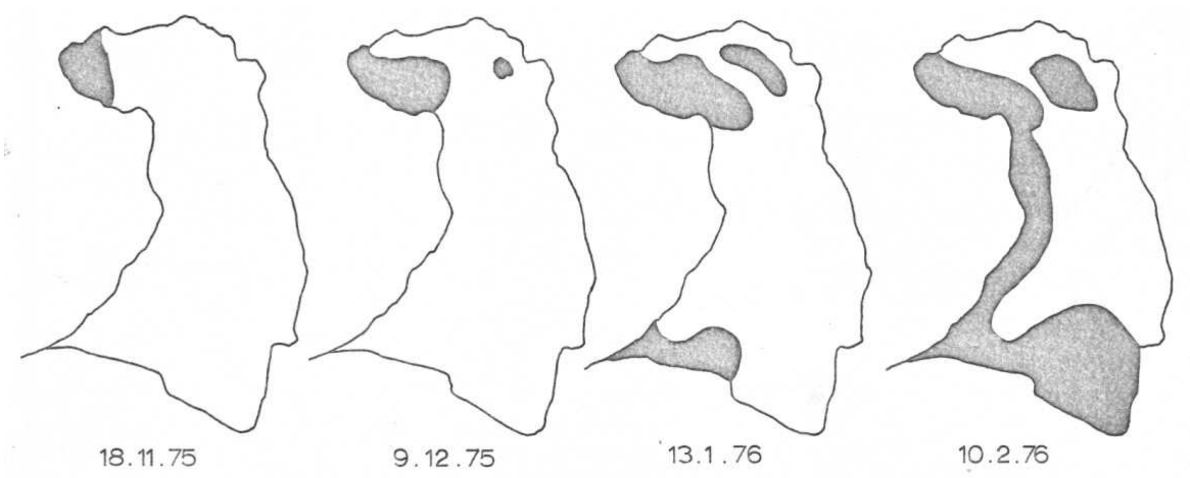


Figure 37. Time series showing the spread of *Potamogeton crispus* in Lake Horowhenua in summer 1975/76. (from Brougham & Currie 1976).

By mid summer, when the water is warm, *Potamogeton* senesces (dies back) and the growth collapses to the lake bed where decomposition releases both N and P to the water column and decomposition processes contribute to the depletion of oxygen in the bottom waters. *Potamogeton* has a low tolerance to high temperatures and low alkalinity, but a high tolerance to sediment disturbance and to most aquatic herbicides such as Diquat and Endothall (P. Champion, NIWA, pers. comm.). The plant germinates from seed so elimination would be difficult as there would be a large seed bank in the sediment. While It is highly palatable to herbivorous fish such as grass carp, these fish are non-selective in their feeding and would remove all available / accessible plant material. A weed harvester could provide a way to clear areas for recreational purposes, but will not eliminate this lake weed.

Egeria densa is a recent invader of Lake Horowhenua and was first noted in the lake in July 2001 (Edwards & Clayton 2002). Edwards & Clayton (2002) commented that the introduction is "likely to have far reaching consequences on the ecology of this lake. One of the features of many shallow lakes that have been invaded by *Egeria* is that after several years of domination by this weed, all of the submerged vegetation rapidly declines. After vegetative decline events, water becomes typically more turbid than before on account of re-suspension of bottom sediments. The ensuing turbid waters are generally too turbid to support submerged plant growth, which has been the case in many of the shallow Waikato lakes*."*

No plants of *Egeria densa* were found in Lake Horowhenua at the 13 sampling sites in August 2002 (Champion et al., 2002). However, the authors commented that where this weed has invaded other shallow Waikato lakes and Lake Omapere, the lakes have lost their diverse submerged vegetation becoming "a monoculture of often surface reaching *E. densa."* They also noted that "dense beds of *E. densa* in Lake Omapere appear to have prevented diffusion of oxygen through the water column to the lake bed and benthic respiration was responsible for benthic anoxia." The consequences of this benthic anoxia appeared to be a decline in *E. densa* health and biomass, and an increase in dissolved N and P as a result of macrophyte decomposition and release from the anoxic sediments both responsible for an increase in planktonic cyanobacterial abundance.

The projected consequences of an invasion of *E. densa* in Lake Horowhenua could be a switch from the annual cycle of summer turbidity and winter clear-water phases to a period of increasing water clarity while the weed beds establish, followed by a collapse of vegetation, benthic anoxia and sustained cyanobacterial blooms for several years.

The control of *E. densa* once established is not easy and quarantine prevention is the best approach. However, it can be controlled to some degree using the herbicide Diquat. A resource consent would be required for the use of this herbicide.

Other aquatic macrophyte species in the lake in August 2002 included *Potamogeton pectinatus* and *Nitella hookeri/cristata* (Figure 38) at <5% of the plant cover with small patches of *Chara corallina* (Figure 38), *Elodea Canadensis* (Figure 36), *Elatine gratioloides* and *Callitriche stagnalis* (Figure 38) (Champion et al., 2002). Some of these plant species e.g., *Chara corallina*, are probably remnants of the original flora of the lake which would have covered much of the lake bed.

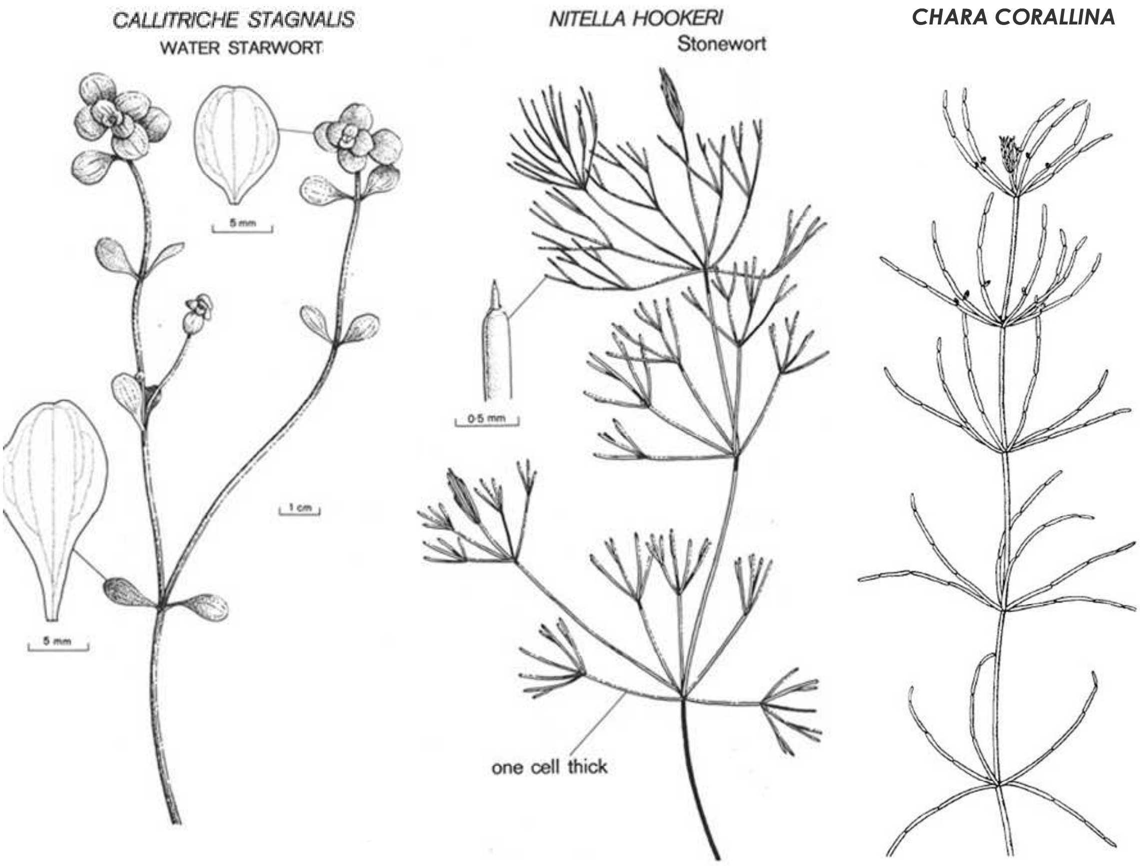


Figure 38: Native pond weeds. *Callitriche stagnalis* (left), *Nitella hookeri* (middle) and *Chara corallina* (right) (drawings from Mason & West 1973).

While macrophytes such as *Potamogeton* are rooted in the sediments, they draw much of their nutrient requirements, especially NO_3-N, from the water column. Consequently, when they begin their spring growth, they have the capacity to remove all of the free NO_3-N from the lake. This spring growth effect can be seen in the lake water nutrient concentration data (Figure 9). This translates to a reduction in TN of about 9000 kg without any increase in algal biomass (Figure 15). After the plant flowers and collapses to the lake bed, decomposition releases the N and P from the plant material back into the water column. At the same time, decomposition of the plant material lying on the lake bed, consumes oxygen and cause the sediments beneath the weed to become anoxic. Under these anoxic conditions, P is released from the sediments where it can stimulate algal growth.

The effect of the summer collapse of the weed beds can be seen as an increase in the TN and TP mass by about 9000 kg and 2500 kg, respectively, coupled with an increase in algal biomass of around 600 kg (Figure 15). The TN:TP ratio distribution plot (Figure 30) indicates lower P concentrations in the sediments beneath the weed beds. This is consistent with an annual cycle of sediment release of P from beneath the weed beds on the western side of the lake and algal assimilation into blooms which subsequently sediment preferentially on the eastern side of the lake due to wind drift.

4.2.3 Maintenance or enhancement of the fishery

As noted earlier (section 2.1 – Maori perspective) in the past, Lake Horowhenua has been renowned for its abundant food resources. Fish and other aquatic species (D Rowe, NIWA, pers. com.) that are likely to have been in the lake in the past include (Figure 39):

- eel (tuna) both short-finned (*Anguilla australis*) and long-finned (*A. dieffenbachia*)
- flounder (patiki)
- mullet (*mugil cephalus*)
- inanga (whitebait, *Galaxias maculatus*) and the other Galaxids, the banded Kokopu and giant Kokopu
- smelt (*Retopinna retropinna*)
- common bullie (*Gobiomorphus cotidianus*)
- koura (freshwater crayfish), and
- kākahi (freshwater mussel, *Hydridella menziesii*)

Of these, eel, flounder, inanga and kākahi were important fisheries. More recently, gold fish (*Carassius auratus*), perch (*Perca fluviatilis*) and to a lesser extent, trout have been additional components of the lake fishery (MWRC 1998). The eel fishery is discussed in detail in a study of the Lake Horowhenua fishery (Chisnall & Jellyman, 1999).

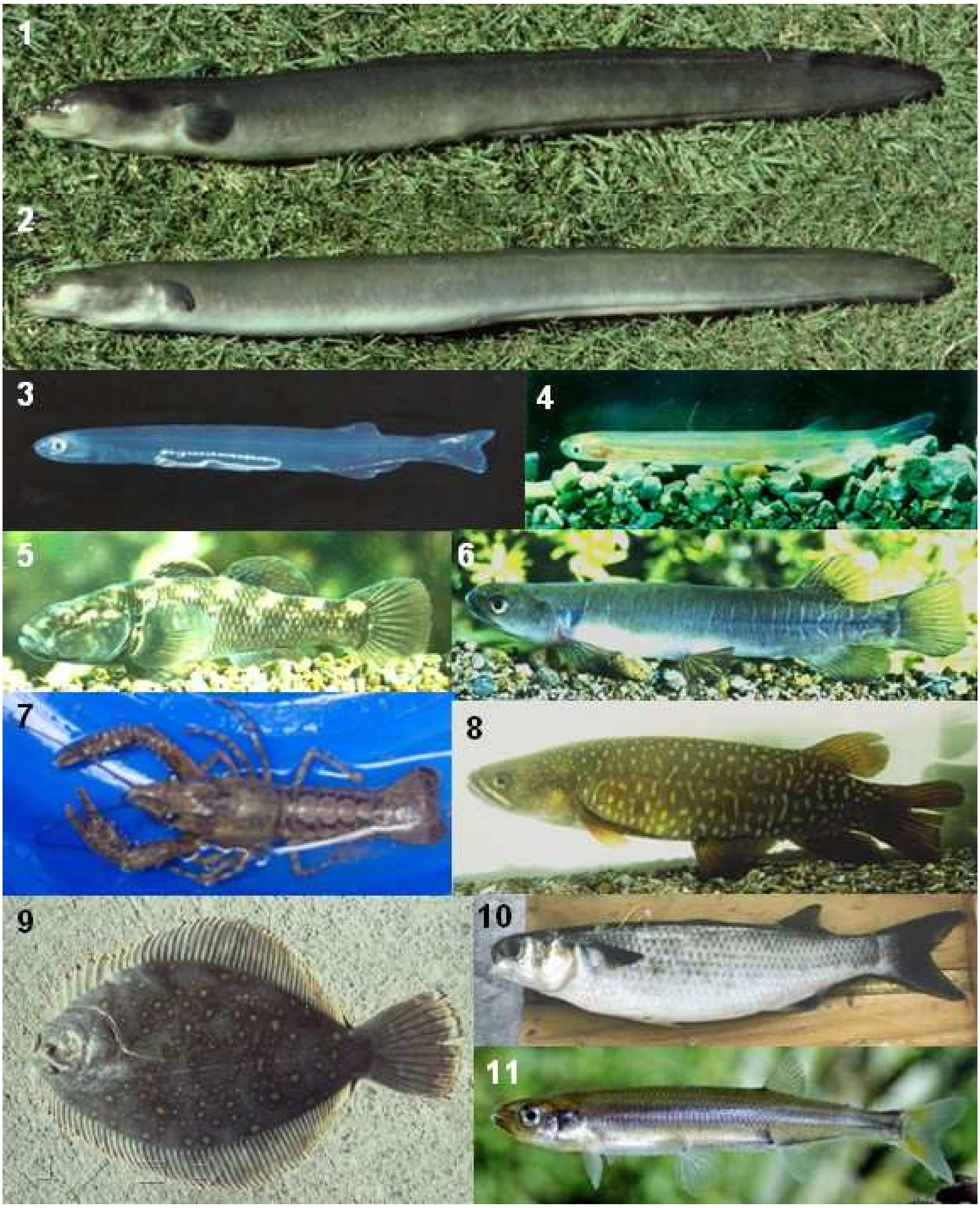


Figure 39. Fish species previously found in Lake Horowhenua: **1**) Long finned eel; **2**) Short finned eel; **3**) Inanga whitebait juvenile; **4**) Banded Kokopu whitebait; **5**) common bullie; **6**) Banded Kokopu adult; **7**) koura; **8**) Giant Kokopu adult; **9**) Black flounder; **10**) grey mullet; **11**) Common smelt. The weir is a barrier to lake access for flounder, mullet and smelt. Installing a fish pass may overcome this obstacle.

It is generally acknowledged that the food resources of the lake have diminished since pre-European times (e.g., MWRC 1998), but the extent of reduction and the causes have not been well studied or documented. Over-fishing is thought to be one cause of the depletion of fish stocks in the lake, especially eels, but more realistically, the decline is probably a response to the degradation of the lake and the installation of the weir to control the lake level and thus the land boundary for ownership.

Many of the fish species that were once found in the lake are diadromus and have an essential part of their life cycle in the sea. The weir is a barrier to smelt, flounder and mullet (Figure 39) and would affect the natural stocking of the lake with inanga. The return of those species to the lake would require a fish-pass to be constructed. However, providing access to the lake would not guarantee a restoration of the fishery if the water quality issues are not addressed.

Invertebrate species (aquatic insects) that require sand or cobble lake beds are missing from the food web although there is an abundance of larval chironomids (*Paratanytarsus grimmi*) in the silty sediments. Zooplankton grazers such as the cladoceran, *Daphnia carinata*, are super abundant during algal blooms and provide food for juvenile fish, as do water boatmen (*Sigaria arguta*).

The fishery at present is a function of the turbid lake environment. Kākahi, which are filter feeders, would have been abundant across the lake bed and their filtering would have controlled the algal biomass in the water column. Elsewhere, kākahi abundance can exceed 100 m^{-2}, but in 2002 the highest densities found were 6 m^{-2} (Champion et al., 2002). This decline is a direct response to increasing turbidity with the sediment particles interfering with feeding by reducing the relative proportion of algal food ingested at the kākahi's maximum filtering rate. Periods of benthic anoxia would also adversely affect kākahi survival and it would also affect koura, although koura could avoid anoxic zones.

Perch are carnivorous and apply top-down pressure on the fishery through predation on juvenile kokopu, goldfish and bullies. Adult perch can grow very large and in the 2002 lake survey, a 0.5 m long perch was observed at one site (Champion et al., 2002). As juveniles, perch out compete other species for the zooplankton food supply. The turbid waters provide shelter for perch from predation by shags. The turbidity also allows their juveniles to survive predation by adult perch.

Removal of the perch from Lake Horowhenua would be very difficult and is not a viable management option. However, a reduction in turbidity would allow increased predation of perch by shags and would allow the adult perch to predate juvenile perch. This would reduce the pressure on the zooplankton population which would increase and reduce the algal biomass further reducing the turbidity in a positive feed-back loop. The downside of clearer water is the growth of macrophyte weeds. A management option would be to control which macrophyte species grew by the judicious use of weed harvesting and targeted spray applications.

The weir was installed by Act of Parliament (Reserves and Other Lands Disposal Act, 1956, Appendix 1). The weir has provision for stacking timber planks on top to alter the water level. As altering the water level in the lake may be a key component of its management for rehabilitation, the regulations and operating parameters for the weir should be investigated.

4.3 Management options

Before a management strategy can be developed for the restoration of a lake, it is important to understand what the hydraulic and nutrient budgets are and how that lake works. This allows targeted management strategies which focus on key points in the seasonal cycle of the lake where a small intervention may create the greatest benefit. It will also identify management strategies that are not appropriate and should not be used on that lake.

As a starting point, the review of the limnology of the lake has identified that the lake has a large internal load of P in the sediment as a legacy from 25 years of sewage effluent input. The review also identified that the summer algal bloom is largely in response to P release from the anoxic lake sediment. From the assessment of the macrophyte component of the lake ecosystem, the weight-of-evidence indicates that the collapse of the lake weed in mid to late summer is the most likely cause of the sediment anoxia which allows the release of P from the sediments in a lake that is otherwise so shallow that it should be fully aerobic all year round. Inflows of anoxic water from the Arawhata stream may augment anoxia in the lake at night and has the potential cause thermal stratification in calm conditions.

A decision support and risk assessment framework has been produced for the management of lake sediment P release (Hickey & Gibbs, 2009). Lake Horowhenua has also been modelled to assess its prospects for restoration (Gibbs & White, 1994). The concept of collapsing weed beds causing the anoxia was not considered in Gibbs & White (1994), only that the sediment experienced anoxia. Notwithstanding this, between these two publications, there are a range of management tools that could be applied to the lake to block or reduce that P release. The decision support and risk framework considers tools for in-lake management of P but Lake Horowhenua also has a major N problem which should be addressed concurrently. Some of the tools used for P management will also serve to reduce N while others will be exclusively targeting P.

Management Tools – engineering solutions

The following is a range of options that could be used on Lake Horowhenua. In most cases there is sufficient data to enable specific scenarios to be tested using a computer simulation of the lake and an estimate of the likely benefits to the lake.

4.3.1 Lake weed control

The weight of evidence indicates that the key to restoring the water quality of Lake Horowhenua and its fishery is the management of the lake weed. There are three approaches to this issue: (1) Mechanical weed harvesting; (2) Spraying; and (3) Biomanipulation. Before undertaking any strategy to control lake weed, there needs to be a clear understanding of the goal and the consequences of each action.

The weight of evidence of the importance of lake weed includes the fact that the tall weeds remove all the NO_3-N from the lake water during spring when there are high inflows of NO_3-N enriched stream water. This evidence also suggests that under low flows the weeds become N-limited and, being unable to sustain the amount of biomass with the nutrients available, they senesce releasing the nutrients in the plants back into the water. In all management strategies, an important question to be answered is 'What happens to the nutrients?"

1. Mechanical weed harvesting is a direct method for managing lake weed. It is used in many lakes in New Zealand and around the world. It is a good way to keep waterways open such as access around boat launching ramps and swimming beaches. However, it does not stop the weed re-growing in the same season or the next. Consequently, weed harvesting is an on-going management option with on-going costs. The operational costs associated with weed harvesting depend on whether the weed is truly harvested, i.e., cut and removed from the lake, or simply cut and allowed to fall to the lake bed and rot. This latter action could initiate a release of the P from the lake sediment earlier in the seasonal cycle while there are still high levels of N in the water column.

   The advantages of weed harvesting are that it has the potential to remove significant amounts of nutrients from the lake in a short space of time. The harvested weed can be composted to off-set harvesting costs. Done in summer when the weed is just reaching maturity would not change the nutrient status of the lake water at that time but would prevent the weed from falling on the lake bed and releasing P from the sediments and the subsequent development of substantial cyanobacteria blooms.

   The disadvantages are that removal of the weed too early would also reduce the removal of NO_3-N from the lake water shifting the lake from nutrient limited in summer to nutrient replete. It is uncertain what algal species would become abundant, but it is likely that there would be a major increase in algal biomass. This, in turn, could cause light limitation to the weed beds which could completely disappear.

   This effect is known as “flipping”. Flipping has occurred in several prominent New Zealand lakes, most notably, Lake Omapere. Lake flipping can be a natural process typically associated with the collapse of extensive weed beds. The reference to the potential dangers of allowing *Egeria densa* into Lake Horowhenua refer to this effect (section 4.2.2). When a lake flips, most often it is from macrophyte dominance to phytoplankton dominance. The lake is then difficult to manage as the option for an intermediate stage of reduced macrophytes and reduced algal biomass has been essentially eliminated, and may only be achieved at great cost.

2. Spraying could have a similar effect as the weed harvesting without removal of the cut weed management option unless the timing is appropriate. The ideal time to spray would be in early spring when the weed seeds have only recently germinated and the plants are only just starting to reduce the NO_3-N concentrations in the lake. The use of Diquat on *Potamogeton* sometimes fails because the plant has a coating of periphyton and fine silt to protect it from the herbicide, i.e., the plants are “dirty”. Sprayed when the plants are young, ensures that they are clean, they are vigorously growing and adsorbing the herbicide, and there is plenty of light to sustain grown. At that time, there should only be a minimum of plant biomass to fall to the lake bed and, consequently, only a small amount of P release from the sediment. The NO_3-N concentrations would remain high in the lake and the algal species assemblage would tend to be dominated by the non-cyanobacteria species. Algal biomass would be produced with the magnitude of any bloom depending on the amount of P released.

   Spraying coupled with P inactivation technology using sediment capping (See section 4.3.14) offers a solution to the problem of P release following spraying. The P inactivation technology binds the P as it is mineralised in the sediment and permanently locks it in the sediment where it is unavailable for algal growth. Under these conditions, the lake water has a high N:P ratio which does not favour cyanobacteria. With extremely low P concentrations the algal biomass is likely to be low due to P limitation.

Another benefit of spraying with Diquat is that the native plants, the charophytes, are not affected by this herbicide and could re-colonise the lake bed, stabilising the sediment.

3. Biomanipulation is becoming more popular in European countries. The technique makes use of natural processes of grazing and predation to achieve the final goal. Zooplankton graze algae and herbivorous fish graze macrophytes. However, just introducing some herbivorous fish may do more harm than good as herbivorous fish may decide to consume desirable species, and they could go on consuming other vegetation when all of the weed has been consumed. As they grow larger, they disturb the sediment and increase the lake turbidity.

   Grass carp (*Ctenopharyngodon idella*) were introduced into New Zealand in the 1960s to combat the spread of different species of lake weed. Grass carp have been released by Northland Regional Council (NRC) into two lakes: Lake Swan in May 2009, and Lake Heather in June 2010. In Lake Swan, NIWA determined that most of the *Egeria* and about 40% of the *Hornwort* was removed within 12 months of stocking. The cost of releasing 400 grass carp in Lake Heather in 2010 was $17,000. Peter Wiessing, the Council’s Kaitaia Area Manager, said that it was “the most cost-effective and environmentally sustainable option to eradicate these weeds.” (Northland Regional Council News Archive 15 June 2010). On a proportional basis, Lake Horowhenua would require about 8000 grass carp at a cost of about $350,000. The consequence of making such a release is that all other options would be negated as the grass carp destroyed the marginal wetland plants and loosen the sediment.

   Sliver carp for reducing phytoplankton biomass is impractical in a lake this size due to the extremely large number of fish that would be required and the impact that would have on the fishery. Once silver carp, or grass carp were introduced, the fishery could not be restored to any previous state. At present there is doubt as to their efficacy at reducing algal biomass and, consequently, no compelling reason to use silver carp.

   A key point in this lake is the grazing pressure on the zooplankton which are capable of reducing algal biomass. Juvenile perch predate the zooplankton as noted above (section 4.2.3). Perch numbers could be reduced by reducing the turbidity in the lake allowing their predation by shags. Carp would increase the turbidity of the lake water. Shag perches constructed in the marginal wetlands near, but not over water, would enhance predation of perch.

The release of grass carp or silver carp into Lake Horowhenua is not recommended and other species of coarse fish such as Rudd, Tench and Koi carp should be prohibited in the lake. It could be prudent to include this prohibition in the regional plan.

4.3.2 Stormwater and groundwater treatment

Waste water treatment uses holding ponds and wetlands for reducing the nutrient load before the water is released into an open waterway. Advanced pond systems (APS) and high rate algal ponds (HRAP) have been designed for treating dairy shed effluent (Park & Craggs 2010). On a larger scale, a constructed wetland has been successfully used on two natural streams flowing into Lake Okaro (Tanner et al., 2007) (Figure 40). The main disadvantages of the technique are that, 1) to be efficient they cover a large area to give sufficient contact time for nutrient and sediment stripping, 2) they are less efficient at removing P than N, and 3) they may not cope with flood flows and the stream short circuits directly to the lake with no renovation.

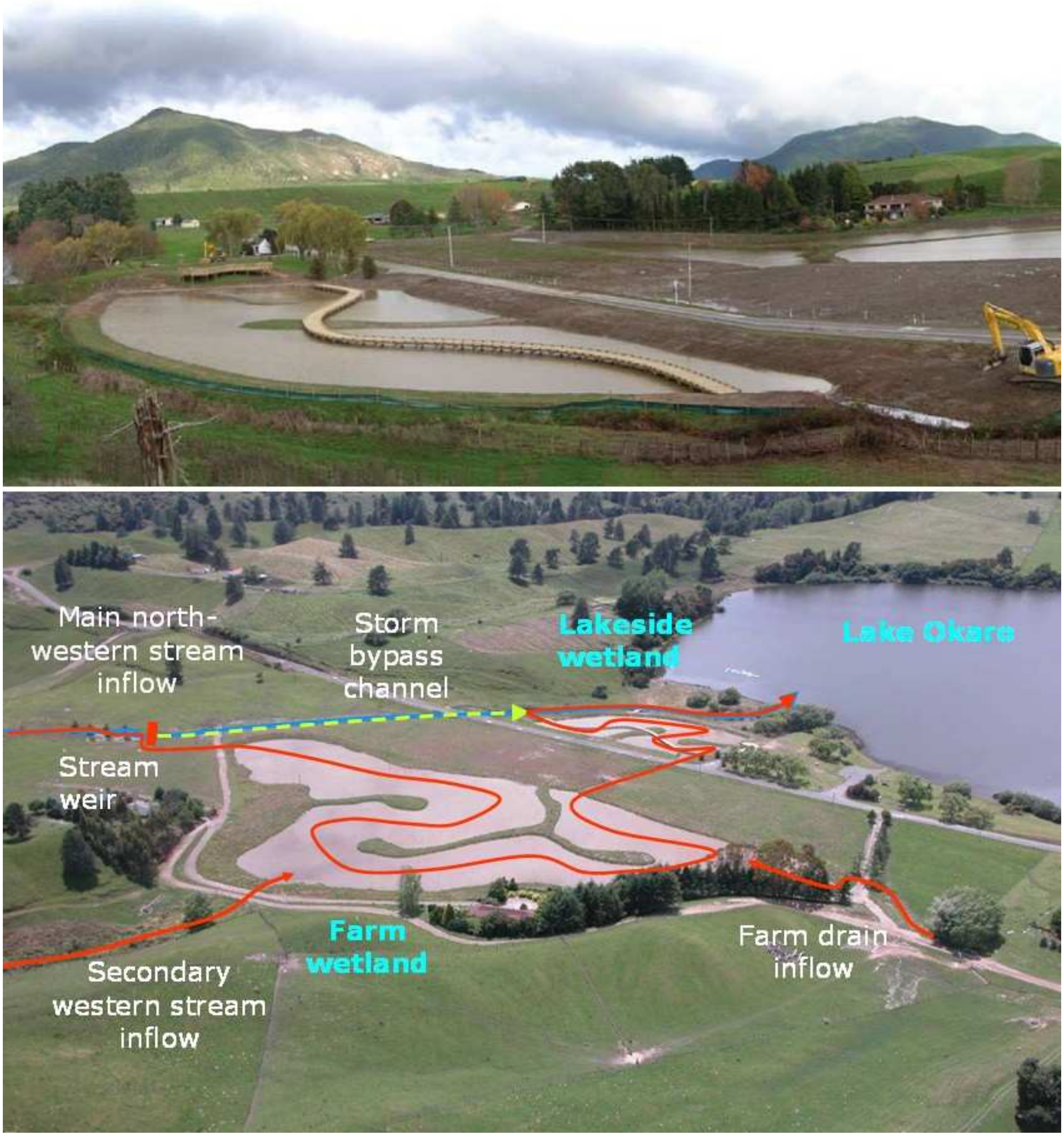


Figure 40. The constructed wetland at Lake Okaro at the time of construction. Top: Public access to the wetland is provide by a viewing walkway through the part of the wetland in the lake domain. Bottom: The stream water is forced to follow a convoluted flow path by the use of ridges within the

wetland. The broken line indicates the short circuit path for flood events. [Photos and overlay: John Quinn, NIWA].

The Okaro wetland was designed to be a publicly accessible attraction and has walkways incorporated to allow easy access for public viewing within the lake domain. Access to the portion of wetland on farm land is restricted. The wetland has a number of ridges which cause the water to follow a convoluted flow path, thus increasing the contact time for plant uptake of nutrients and for the sedimentation of particulate matter (Figure 40).

Because of its high annual P load on the lake (Table 3-1), the Queen Street drain is a contender for treatment using a holding pond to settle out sediment and rubbish, followed by passing the water through a constructed wetland. The options for locations of these constructions are on the lake shore adjacent to the lake or along the lake edge in the lake. The latter could be achieved using floating treatment wetlands (see section 4.3.2).

4.3.3 Floating treatment wetlands

Floating treatment wetlands (FTW) are a new restoration concept where emergent wetland plants are grown in buoyant rafts which are moored in a lake or stream. These rafts are constructed from recycled plastic (PET drink bottles) in various sized sections that could be 2 m by 3 m for ease of handling and joined together later. The plants are grown in recesses in the raft and their roots extend down into the water where they assimilate the N and P. The FTW concept has been tested near Lake Rotoehu using stream water with promising results (Sukias et al., 2010). An in-lake version with a surface area of 0.3 ha is being tested in Lake Rotoehu at present. The FTWs are aesthetically pleasing and blend in with the natural lake shore environment (Figure 41) and provide additional habitat for birds and koura.

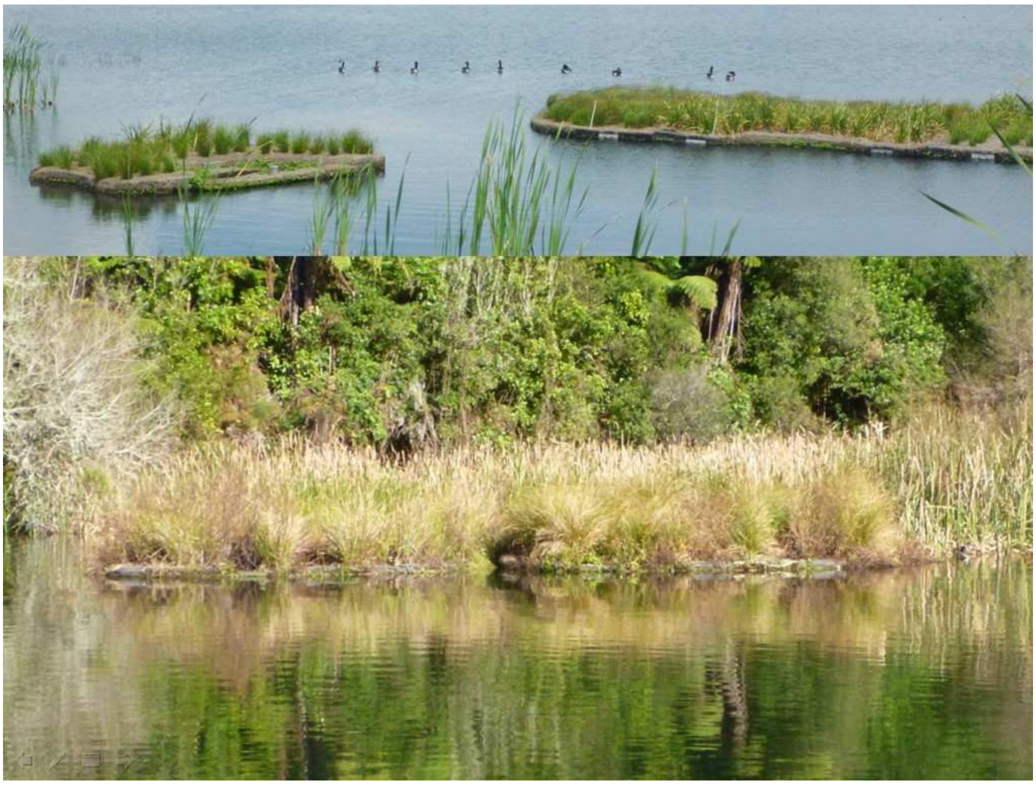


Figure 41. Top: Floating treatment wetlands (FTW) that have recently been planted and the plants are becoming established. Each of these FTWs comprised several rafts joined together. **Bottom**: Mature FTWs in Lake Rotoehu blend in with the marginal wetland plants on the lake shore.

In Lake Horowhenua, a break wall of FTWs could be established on the eastern side a few metres out in the lake between the Queen Street drain and the Makomako Road drain, a distance of about 1.5 km (Figure 42). (See also Figure 43 for cross-section). This section of lake shore has the largest groundwater inflow and consequently, a high N input. With the northern end adjacent to the Queen Street drain, the storm water flow from this drain could be channelled into the inshore water rather than directly into the lake. In this concept, the gap between the shore and the FTW wall would become a "holding pond" allowing the plants in the FTW the required contact time to assimilate the nutrients, both N and P.

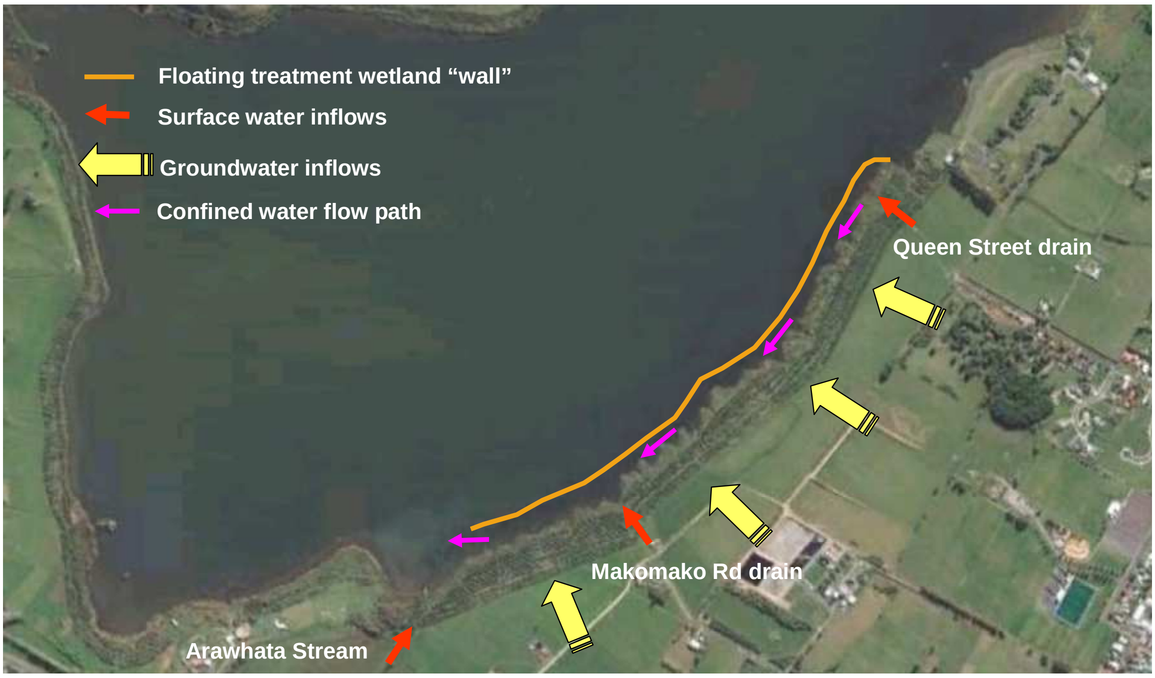


Figure 42. Schematic overlay of where a floating treatment wetland wall could be installed in Lake Horowhenua to facilitate nutrient stripping from the Queen Street drain inflow and the main groundwater inflows to the lake. [Background photo: Google Earth].

Advantages of this concept include: being able to put nutrient stripping where it is needed; providing a wave break to reduce bank erosion and sediment re-suspension from the lake bed; providing a detention zone to trap sediment and other contaminants and rubbish from catchment inflows; providing a sheltered habitat for koura and juvenile fish in the root zone below the rafts; and providing additional habitat for wetland birds and water fowl nesting. The FTWs can be installed, repositioned to improve performance, or removed if not needed, without damage to the lake. The disadvantages are the initial costs of the raft material, the plants, and the mooring system to secure the FTWs in place against wave action.

As with all wetland buffer zones, their nutrient stripping efficiency declines when the plants are mature. Consequently, the plants need to be cropped/harvested periodically to remove the organic biomass. However, with appropriate selection of plants, e.g., water cress or equivalent, the FTWs could provide a cash crop.

4.3.4 Marginal buffer zones

Onshore management to reduce nutrient inputs to the lake should look at the sources of those nutrients. However, because the lag time between the contamination of the groundwater and that contaminated groundwater reaching the lake, nutrient stripping of the groundwater at the lake edge is also essential. This is best achieved using marginal wetland buffer zones (Figure 43). The surface or unconfined groundwater aquifer is the most vulnerable to contamination in the catchment as it receives the infiltrating rainwater percolating down through the soil. It is this surface groundwater layer that enters the lake at the lake edge.

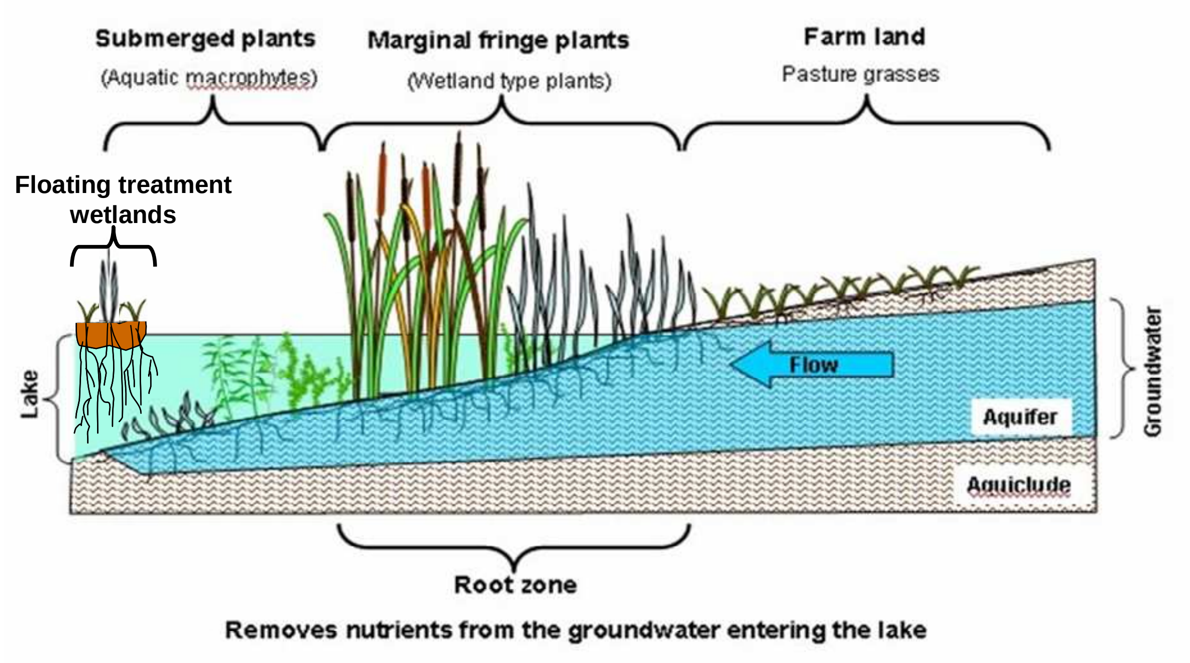


Figure 43. Schematic cross-section of a marginal buffer zone. A floating treatment wetland (FTW) is included to show the all important root structure below the raft and where the FTW would be located relative to the marginal buffer zone plants for **Figure 42**.

The most effective marginal buffer zone comprises a variety of plants which range in water tolerance from having the tips of their roots in the groundwater (e.g., kukuyu grass) through rushes and flax, to raupo (*Typha orientalis*), and reeds which are almost fully submerged with just the tips of their leaves out of the water (e.g., *Eleocharis*). These plants mostly take up nutrients through their root systems. Further into the lake submerged macrophytes take up nutrients from the water column. In Lake Horowhenua, the lake weed *Potamogeton* takes up the NO_3-N from the lake water as it grows in spring, and is capable of removing all of the available NO_3-N from the lake in that growth period. The disadvantage of marginal wetland plants is that, unless the biomass from the spring growth is harvested and removed from the lake, the nutrients in that biomass will be returned to the water when these plants senesce in late summer and winter.

The margins around Lake Horowhenua have been planted with flaxes. If these plantings are a monoculture, the nutrient stripping efficiency of the marginal buffer zone could be enhanced by augmenting the flax with a selection of other plants.

Harvesting of marginal buffer zones could be mechanical using weed harvesters in the lake and mowers with catchers (e.g., silage cutters). The alternative to mowers on land is to use short (<1 hour) grazing of the marginal buffer zone by stock. The stock are kept out of the lake with temporary electric fences on the lake edge. Because the grazing period is kept short, faecal material from the stock is mostly deposited inland from the marginal buffer zone after the stock are removed.

4.3.5 Storm water diversion

From the nutrient and water budget (Table 3-1), it is apparent that more than 80% of the external P load on the lake comes in the runoff from Levin via the Queen Street drain. The Queen Street inflow is also 45% of the annual internal P load. Diversion of the Queen Street storm water to the Hokio Stream outflow would remove that annual load of P from the lake while reducing the hydraulic load by less than 20%. The Queen Street drain also carries about 14% of the annual N load on the lake as well as faecal bacteria from animal waste on the road i.e., from free roaming dogs and effluent spills from stock trucks passing through the town. Consequently, there would be a benefit to the N load and other contaminants on the lake by diverting the Queen Street storm water flow around the lake.

This option may be more expensive and less culturally sensitive than entraining the Queen Street drain flow through a shore line holding zone behind a wall of floating treatment wetlands (see above).

4.3.6 Arawhata Stream diversion

The Arawhata Stream contributes about 50% of the external surface water N load to the lake. Diversion of the this water directly to the Hokio Stream outflow would reduce the TN load on the lake by around 50 t y^{-1} as well as reducing the NO_3-N concentration in the lake. Reducing the open water NO_3-N concentration would impact on the growth of macrophyte weed beds and potentially the magnitude of their cyclical impact on sediment release of P when the weed beds collapse. With less NO_3-N injected into the lake water column, the high proportion of denitrification identified in the nutrient budget (Table 3-1) would be more effective against the NO_3-N seeping through and across the lake bed from groundwater inflows.

Arawhata Stream diversion would be equivalent to the diversion of the Ohau Channel flow from Lake Rotorua through Lake Rotoiti to the Kaituna River outflow (Figure 44). In that case, the polluted water from Lake Rotorua was adversely impacting on the water quality of Lake Rotoiti causing Lake Rotoiti to become eutrophic with significant algal blooms of cyanobacteria. With the installation of the diversion wall in the outlet arm to Okere Falls, the water quality of Lake Rotoiti has dramatically improved and the algal blooms have largely disappeared.

4.3.7 Flushing – external source water

A simple expedient in small lakes overseas is to divert a proportion of a clean stream or river through the lake to reduce the residence time and thus flush the N and P in solution and in particulates, such as algae, down the outlet stream. Potential external water sources are the Ohau River to the south and the Manawatu River to the north. Without consideration of the water quality of either river or their capacity to deliver water, a continuous flushing volume of 0.5 m^3 s^{-1} would reduce the mean annual residence time from 47 days to 30 days. Continuous flushing might not be appropriate in winter when it could increase the risk of downstream flooding. An in-depth analysis of the water quality of the source water would be needed if this option was a serious consideration.

Figure 44. Areal view of the Ohau Channel water diversion wall in Lake Rotoiti. Eutrophic water from Lake Rotorua enters Lake Rotoiti via the Ohau Channel (lower left corner) and is deflected north by the sheet steel diversion wall (centre) to the Kaituna River outlet (top centre) instead of flowing into the body of Lake Rotoiti (lower right side). (See also **Figure 53**). [Photo: Bay of Plenty Regional Council].

The most appropriate time to introduce flushing would be during the summer low flow period when the natural residence times can be up to 95 days. At this time of year, the lake normally has high DRP from sediment release and thus high cyanobacteria algal biomass. Adding 0.5 $m^3 s^{-1}$ at this time would reduce the residence to around 45 days and help flush the algal biomass out of the lake. Higher flow in Hokio stream would be beneficial to the fishery as the water would be more likely to be well oxygenated and slightly clearer than the natural condition. The higher through-put has the potential to keep the lake at a higher dissolved oxygen content and thus reduce the P release from the sediment and the concomitant bloom of cyanobacteria.

If a pipeline was installed from the Manawatu River, it would be about 8 km long but the water quality may not be high. Alternatively, an equivalent pipe from the Ohau River would be about 6.5 km long but the volume of water abstracted in summer could adversely affect the downstream ecology of that river.

4.3.8 Enhanced flushing using fluctuating water levels

This technique requires manipulation of the weir on the lake to increase and decrease the lake water level at different times of the year to take advantage of specific parts of the lake cycle.

Lowering the water level in winter would enhance the through-put of N from the upstream catchments and would reduce sediment accumulation in the lake. Winter is the time of migratory runs of diadromus fish species such as eels and whitebait which would be moving up the Hokio, and the lower weir would provide better access to the lake. The lower lake level in spring would also provide better access for other management strategies such as spraying weed beds (see below).

Raising the weir in early summer would provide a deeper water column which is more stable and thus would allow suspended solids to settle giving a clearer water column. If the weed beds had been sprayed, this clearer water would not stimulate lake weed growth and the lake would be more suitable for summer contact recreation. The calmer, deeper water column may develop algal blooms if the sediment becomes anoxic and releases P. If such blooms develop, the weir can be operated as a skimmer to remove the floating algal bloom, selectively reducing N and P in the algal biomass. This would be most effective on calmer summer days when the katabatic winds from the east would drive the cyanobacteria blooms to the outlet.

Seasonal changes in water level are potentially good for fisheries with the most productive fisheries having low water levels in summer (J. Boubeé, NIWA, pers. comm.).

4.3.9 Dredging

Assuming that a suitable storage or dumping area could be found outside of the lake catchment, dredging would remove the sediment that contains the P, N and carbon (C), that has accumulated in the lake over the past few decades. It would also remove the seed bank for lake weeds. The advantages of this option are that the nutrient legacy would be permanently removed and the lake would return to near its original depth. The disadvantages of this option are the cost to remove the estimated 3 km^3 of sediment, the destruction of the existing ecosystem, and the release of nutrients and other toxic chemicals such as sulphides during the process of dredging. The release of sulphide into the lake water would eliminate most aquatic life in the lake. The removal of lake weed seed banks in the sediment would not be selective and desirable species for restoring the lake habitat would be removed along with the undesirable species.

This is an option which is not recommended for the lake. However, given the level of organic sediment accumulation in the Arawhata Stream, dredging of the stream channel may be an option as a measure to reduce the source of nutrients entering the lake in summer.

4.3.10 Aeration

Aeration replaces the oxygen consumed by decomposition processes and prevents the development of anoxic conditions which allow P release from the sediments. This technique is often used in water supply reservoirs to prevent the anoxic release of P, which favours cyanobacteria growth, and the release of minerals such as iron and manganese, which would stain baths, toilets and hand basins in homes and would cause black marks on washing as the water re-oxygenates and these metals precipitate.

The advantages of the technique is that it is relatively cheap, requiring an air compressor, connection hoses and an aeration bar with anchor blocks, and only needs to run in summer when low oxygen concentrations develop. For shallow lakes like Lake Horowhenua the efficiency of an aeration bar would be limited and alternative aerators such as those used in waste water treatment ponds may be more appropriate.

The issue in Lake Horowhenua is not that the lake develops anoxic bottom waters, but the cause of the bottom water anoxia. If it is caused by high sediment oxygen demand, aeration will work. However, if it is caused by the collapsing weed bed preventing oxygen diffusion to the sediments, aeration will not work.

Hazard warning: Aeration with air bars in recreational lakes should be used with caution because the water in the plume of rising air bubbles has a lower density than normal water and could cause boats to sink should the whole boat enter the bubble zone. Similarly, swimmers could drown.

4.3.11 Aeration by Discing

In shallow lakes, giant discs pulled through the lake sediments open up the sediment allowing deep penetration of oxygen from the water column. The concept behind this tool is that the P will be bound to the iron and manganese oxides in the sediment. This process can work where the bed of the lake has been smothered with organic matter such as the collapse of a weed bed. Modelling of this option (White & Gibbs 1991) indicates that the beneficial effect for P binding is short lived because as soon as the sediments go anoxic once more, the P is released.

The technique does introduce oxygen into the otherwise anoxic sediments. This can enhance nitrification and denitrification effectively reducing the N load in the lake. More importantly it supplies oxygen to the decomposition processes so that organic carbon content and thus the sediment oxygen demand is slowly reduced. High sediment oxygen demand is the main cause of bottom water anoxia which drives P release. However, to achieve a significant reduction in sediment oxygen demand, the discing would need to be repeated frequently.

There are three major disadvantage of discing through weed beds: 1) the organic matter could be driven into the sediment raising the organic carbon content, 2) every leaf node of most aquatic macrophytes can grow so the discing would most likely spread the weed more widely, and 3) the discing would devastate the benthic mussel beds destroying that part of the fishery.

Discing is not recommended for Lake Horowhenua.

Management Tools – Water treatment solutions

4.3.12 Aeration by nitrate injection

In some overseas restoration studies, concentrated nitrate is injected into the sediment with equipment similar to the giant discs to provide an oxygen source for decomposition processes in summer (Hemond & Lin, 2010). The advantages of this technique are that the release of P is reduced and thus the dominant algal species are not cyanobacteria. The release of arsenic (As) is also suppressed. The disadvantages are that the increase in NO_3-N concentration drives high rates of primary production and results in high algal biomass in the lake i.e., the lake goes very green. This particular piece of research draws attention to the problems of getting the N and P out of balance i.e., heavy metals such as As and lead (Pb) can be released from the sediments.

This is not a recommended technique for use on Lake Horowhenua.

4.3.13 Phosphorus inactivation with flocculation

The phosphorus released from the lake sediments is in the form of phosphate which is readily usable by plants, especially algae, for growth. While all plants use N and P in the ratio of 16 N to 1 P (Redfield 1958), the symbiotic bacteria inside blue-green algae, hence the name ‘cyanobacteria’, can convert N_2 gas in the atmosphere to NO_3-N which the algal host can use for growth. This gives cyanobacteria a competitive advantage over all other algal species when there is a surplus of P and a deficit of nutrient N in the lake water, and they dominate the algal species assemblage. Consequently, an excess of P in the water column is said to favour the growth of cyanobacteria and the formation of nuisance blooms.

Phosphorus inactivation is used to make that P unavailable for algal growth by binding it to a metal. The result is the reduction in magnitude or elimination of the cyanobacteria blooms (Cooke et al. 2005). There have been many documented applications world-wide and the general conclusion is that this is a highly successful method for treating lakes with high internal P loads.

Phosphorus inactivation is a naturally occurring phenomenon where, under aerobic (oxidising) conditions the P can be sequestered onto the surface of metal oxides such as iron and manganese in the sediments. Unfortunately, the process is reversible. Under anoxic (reducing) conditions the iron and manganese dissolve and release the previously bound P back into the water column.

Research has found that if the P is bound to aluminium or lanthanum, the process is irreversible under normal lake conditions and the P is not released under reducing conditions. The consequence of this is that there is very little P available for algal growth and any that appears from dying algal cells is rapidly assimilated by all algae i.e., cyanobacteria no longer have a competitive advantage and are less likely to become the dominant species.

The most commonly used P inactivation agent is aluminium sulphate, commercially know as alum. This is a flocculent in everyday use for the treatment of domestic drinking water supplies to remove sediment. It is also used in swimming pools to clarify the water. In lakes it performs the double function of binding any phosphate in the water column as well as reducing suspended solids. When the alum floc settles to the lake bed, any unused P binding capacity remains active and will bind any P released from the sediments.

Alum was sprayed on Lake Okaro (Figure 45) in an attempt to reduce the cyanobacteria blooms on that lake (Paul et al., 2007). The water column became clearer after an initial increase in algal biomass and the algal species switched from cyanobacteria to a green algae. However, at the low dose rate used there was insufficient aluminium to sequester all of the P in the lake and the cyanobacteria blooms returned in following years.

Figure 45. Spraying alum onto the surface of Lake Okaro. The air boat was fitted with GPS navigation to control where the spray was applied.

There are specific requirements for the use of alum in natural waters. Because alum is acidic (the 46% solution has a pH of 2.1) it requires to be buffered to a pH of 7 where the waters have low alkalinity (e.g., <30). Lake Horowhenua has an alkalinity of around 65 measured as g $CaCO_3$ m^{-3}, which is approaching the nominal value of 80 above which no buffer is required for normal dosing. If alum were to be used on the lake, it is recommended that sodium carbonate buffer is included with the alum to ensure the formation of a floc. Without the buffer there is a risk that the pH would fall below 6, no floc would form and toxic trivalent aluminium ions (Al^{3+}) could be released into the water column. This does not happen if the alum is correctly buffered to produce the floc. A side effect noticed with the use of alum is that some zooplankton can be caught in the floc and then be carried to the lake bed. This effect alters the grazing pressure on algae and, left unchecked, the algae may grow rapidly for a few days before the zooplankton populations become re-established.

NIWA studies on the P binding capacity of alum found that it can sequester up to 85 g P kg $^{-1}$ alum (Gibbs et al., 2011a) and that it will sequester P from the sediment as well as the water column (Gibbs et al., 2008, 2011b).

Treatment rates for the application of alum are calculated based on the areal load of TP in the top 4 cm of sediment plus the areal load of DRP in the overlying water. For Lake Horowhenua, in summer when the DRP concentrations in the 1.5 m deep water column are around 0.5 g m^{-3} and the top 4 cm of sediment hold ~2.56 g P m^{-2}, the total areal load would be around 3.3 g P m^{-2}. The estimated dose rate for alum would be around 38.8 g pure alum m^{-2}. Alum is supplied in a 46% liquid form which would require a dose rate of about 85 g m^{-2}. To treat the whole lake would require about 250 tonnes of liquid alum. The buffer requirement would be soda ash, which is also used in water treatment plants. The amount of buffer required is normally about twice the amount of alum, i.e., about 500 tonnes, but should be checked using the lake water before treating the lake.

The preferred time for treatment would be mid summer when DRP concentrations in the lake water were maximum.

4.3.14 Phosphorus inactivation with sediment capping

The alternative to alum is to use a granular P inactivation agent. These products are designed to inactivate P either in the water column or at the sediment surface before settling on the lake bed as a thin (1–2 mm thick) layer. This is the layer referred to as the sediment cap (Figure 46).

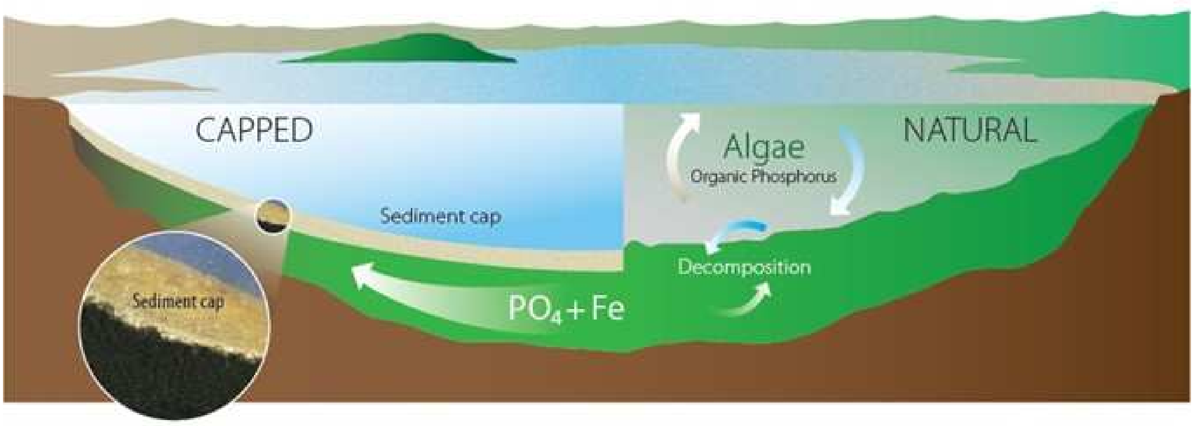


Figure 46. Schematic diagram of sediment capping. The sediment capping is only required in the zone below the thermocline to block P release when the hypolimnion goes anoxic.

The two products currently available on the market are: Phoslock™ developed in Australia by Phoslock Water Solutions Ltd., and Aqual P™ developed in New Zealand by Scion and marketed by Blue Pacific Minerals. Other future potential products include allophane which is a natural volcanic ash currently being investigated by Landcare Research Ltd., but not yet available in commercial quantities, and another product being developed by Fertco Ltd., but not yet available.

Phoslock™ is a bentonite-based product with a lanthanum salt as the active P binding agent. Phoslock™ has a maximum P binding capacity of around 12 g P kg^{-1} product and does not require a buffer. It disperses rapidly in water and has a long contact time. Because its main action is in the water column, it should be applied in summer when the DRP concentrations in the lake water are maximum. Under these conditions, Phoslock™ has the capability of binding up to 95% of the DRP in the water column (Gibbs et al., 2011a). Once it settles to the lake bed, any unused binding capacity remains available to sequester any P released from the sediment under anoxic conditions. The P is permanently bound to the lanthanum and is not available to algae for growth. Phoslock™ is beginning to be widely used overseas (European countries) and has been used in one lake trial in New Zealand, Lake Okareka.

There are two potential disadvantages of using Phoslock™: 1) Being a bentonite clay based product it is very slow to settle and may take more than a month before the water clears. However, because Lake Horowhenua has high turbidity, this may not be an issue; 2) There are questions about the potential toxicity of Phoslock™ to zooplankton and fish in low alkalinity waters (Gibbs et al., 2011b).

Aqual P™ is a zeolite-based product with an aluminium salt as the active ingredient. Aqual P™ has a maximum P binding capacity of around 23 g P kg^{-1} product and does not require a buffer. Zeolite has a natural affinity for NH_4-N meaning that an application of Aqual P™ has the capacity to adsorb both P and NH_4-N. Applied as a fine grain powder in a slurry, the product settles relatively quickly to the lake bed where it can form a cohesive capping layer. Because of the short water column contact time (<1 day), the product has a limited effect on any DRP in the water column, binding up to 25% of the free phosphate (Gibbs et al., 2011a) which leaves it with a high capacity for binding P released from the sediment (Gibbs et al., 2008, 2011b; Gibbs & Özkundakci, 2011). Consequently, Aqual P™ is designed to work on the sediment as a sediment capping agent and should be applied in winter or early spring before the sediment becomes anoxic and the P begins to release.

Aqual P™ has been used successfully on Lake Okaro where it was applied as a slurry from a barge (Figure 47). It was also successfully used pre-emptively on Okawa Bay where it was applied as a slurry from a helicopter (Figure 48)

Figure 47. Application of Aqual P™ as a slurry from a barge on Lake Okaro. The barge position was controlled using GPS navigation. [Photo: Andy Bruere, Bay of Plenty Regional Council].

Figure 48. Application of Aqual P™ as a slurry from a helicopter on Okawa Bay, Lake Rotoiti. The helicopter position was controlled using GPS navigation. [Photo: Graham Timpany, NIWA, Rotorua].

4.4 Case studies

4.4.1 Lake Okaro

Lake Okaro (Figure 47) is a hypertrophic lake (area 0.3 km^2) in the Te Arawa/Rotorua Lakes region of the Bay of Plenty. The lake has a mean depth if 12 m and a maximum depth of about 20 m. It thermally stratifies at between 5 and 8 m depth and the hypolimnion (bottom water) becomes anoxic and enriched with DRP from the internal load. The source of the nutrient legacy in the lake sediments is from land clearance and farming practices which included the over-use of superphosphate in the catchment and that the spring waters in the central volcanic plateau, where the lake is situated, are naturally enriched with P dissolved from the volcanic ash. The lake has been used as a whole lake trial for restoration/rehabilitation since 2003:

2003: Alum dosing was trialled by spraying unbuffered alum onto the lake surface (section 4.3.12; Figure 45). The results were indeterminate although there was a change in algal species dominance away from cyanobacteria the summer after treatment. The hypolimnion still accumulated high concentrations of DRP during summer stratification.

2005: A constructed wetland was built on the inflow streams which carried the most N and P into the lake (Figure 40). There was a reduction in the amount of P entering the lake but it was insufficient to prevent cyanobacteria blooms. A major difficulty was that most of the P that got into the lake was carried there during storm events in flood water which by-passed the wetland.

2007: The first Aqual P™ dose (then called Z2G1) was applied to the lake as a coarse (>3 mm) granule. About 100 t of this product formulation was applied using a fertilizer spreader on a barge (Figure 49). The coverage on the lake bed was uneven and the granules gradually sank into the sediment – but not before they had adsorbed about 50% of the internal P load in the sediment. The timing of this application was not ideal and sediment P release had already begun when the product was applied. Consequently, there was measurable DRP in the bottom water but at a lower level than in the previous years.

2009: The second Aqual P™ dose (about 45 t) was applied as <1 mm fine ground powder injected about 3 m below the surface as a slurry (Figure 50). The application was in August before the bottom water had gone anoxic and any P had been released. This application was still uneven but gave a better coverage of the sediment and did not sink into the sediment. The following summer Lake Okaro had no serious cyanobacteria blooms, for the first time in decades. Water clarity suddenly rose from <2 m to >6 m and the algal assemblage (mainly dinoflagellates) developed at the depth of the thermocline where there was access to nutrients from the hypolimnion and plenty of light.

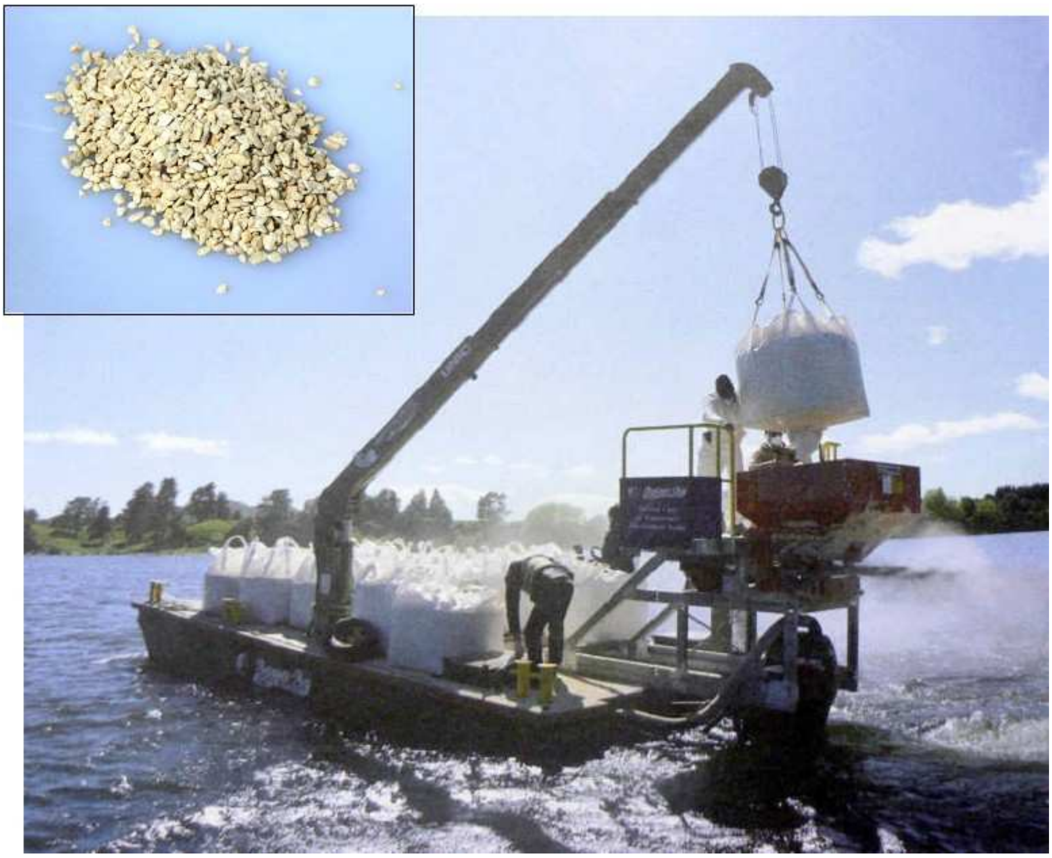


Figure 49. Application of course granular Aqual P™ on Lake Okaro in September 2007, using a fertilizer spreader on a barge. The product being applied is shown in the inset.

Figure 50. Application of fine granular Aqual P™ on Lake Okaro in August 2009. The product was injected as a slurry 3-5 m below the lake surface from a barge. Fine grain product in the inset.

The restoration sequence can be seen in the DRP concentrations in the hypolimnion (Figure 51).

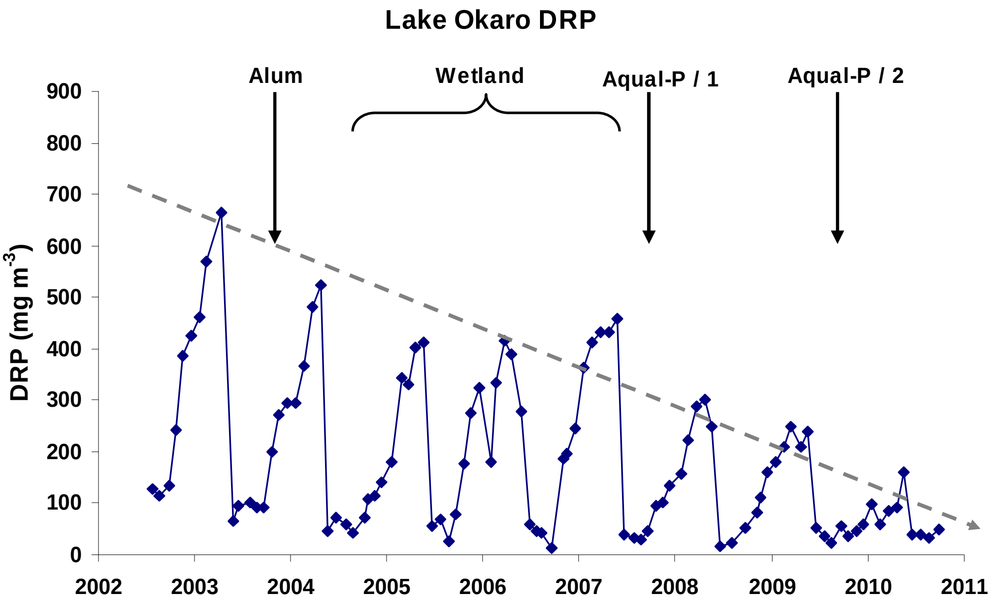


Figure 51. Time-series changes in the hypolimnetic DRP concentrations relative to the remediation measures implemented by Bay of Plenty Regional Council. There is a clear trend in the data of reducing DRP with each successive remediation action.

The point to note in these data is that each management step had an effect on the internal P load but it was not until the cumulative effect of all the remediation efforts brought the internal P load from sediment release to a critical threshold that the algal assemblage in the lake could out-compete the cyanobacteria for nutrients to grow.

The goal of this management exercise was to improve the water quality of the lake from hypertrophic to eutrophic. The target lines on the time series data for the chlorophyll *a* in the epilimnion (<12 mg m^{-3}) and for the DRP in the hypolimnion in summer (<43 mg m^{-3}) were achieved (Figure 52).

The epilogue to this sequence is that the following summer, the cyanobacteria bloom was back, very briefly. The reason is attributed to a flood event during spring bringing new particulate P into the lake and burying the capping layer. The P bound in the capping layer was not released but the capping layer was not able to sequester P released from the layer of new sediment on top of it. This reinforces the need to manage the catchment sources of nutrients at the same time as the in-lake management interventions, or before.

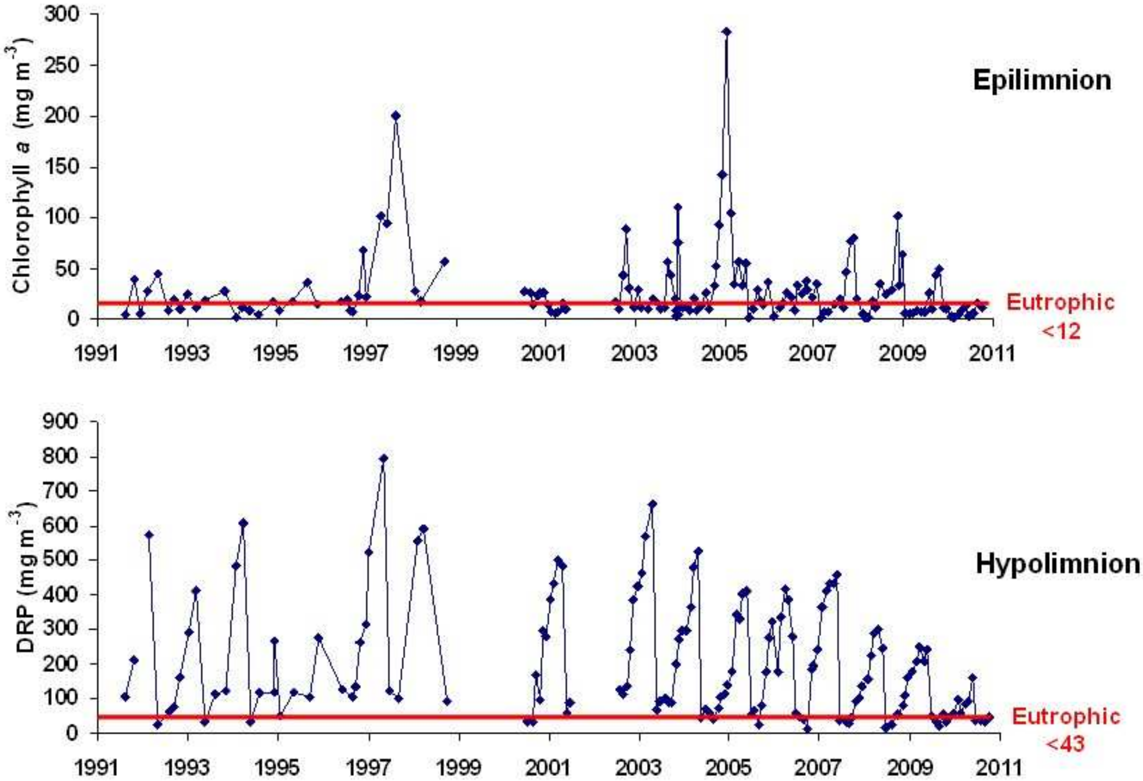


Figure 52. Lake Okaro time series chlorophyll *a* concentrations in the epilimnion relative to DRP concentrations in the hypolimnion. The horizontal lines are the target concentrations required to achieve a lake water quality goal of eutropic.

4.4.2 Okawa Bay

Okawa Bay (Figure 53) is an enclosed side arm off the western basin of Lake Rotoiti. For many years the urban development around the shores of this bay used septic tank for sewage treatment. The septic tank leachate seeped into the bay through the groundwater and massive macrophyte weed beds developed around the shore line hindering boat access and swimming.

Historically this bay was the sheltered water harbour for the Rotoiti Timber Company saw mill around 1920. Logging operations cleared much of the Lake Rotoiti catchment of native forest and the sawmill dumped the sawdust into the bay, leaving a huge legacy of organic carbon, which is still decomposing today.

The bay became a popular fishing sanctuary with the development of lake side housing serviced by septic tank waste water effluent treatment. The addition of sewage effluent nutrients to the sediment in the bay resulted in the summer release of P and the growth of cyanobacteria blooms. Recently the sewage effluent from the houses around Okawa Bay was reticulated and piped to the sewage treatment plant servicing Rotorua City. The water cleared within 12 months and the cyanobacteria blooms disappeared. Unfortunately the macrophyte weed beds continued to grow and needed periodic harvesting to keep boat access free for water skiing.

Figure 53. Areal photo of Okawa Bay relative to Lake Rotoiti, Lake Rotorua, the Ohau Channel and diversion wall, and the outlet to the Kaituna River at Okere Falls. [Photo: Max Gibbs].

A decision was made to spray the weed in summer 2011.

The decomposing weed left on the lake bed began to release DRP and there was evidence of the imminent development of a cyanobacteria bloom. To prevent this bloom, the Bay of Plenty Regional Council used an emergency pre-emptive treatment of the bay with Aqual P™. A slurry of the fine grain Aqual P™ was sprayed onto the lake surface by helicopter (Figure 48). The cyanobacteria cell counts diminished and the bloom did not develop.

The point to note in this case was that the decomposing weed left on the lake bed blocked the diffusion of oxygen into the sediment allowing anoxic conditions to develop immediately above the sediment. This was the trigger mechanism to release P from the internal load in the sediment and stimulate the growth of cyanobacteria. It is the same trigger mechanism that causes the P release from the sediment in Lake Horowhenua.

The quick thinking and decisive action of the Bay of Plenty Regional Council using the Aqual P™ as a remedial tool prevented the development of a full blown cyanobacteria bloom which would have closed the bay to all contact recreation.

5 Restoration strategy

To rehabilitate Lake Horowhenua, more than one management tool will be required. There will be a range of ways to achieve the management goals (Section 4.2). The following flow diagram (Figure 54) gives a practical example of the process towards rehabilitation of the lake. The management strategies (Figure 55) make use of selected tools discussed in section 4 with a brief explanation of the reasoning behind their selection

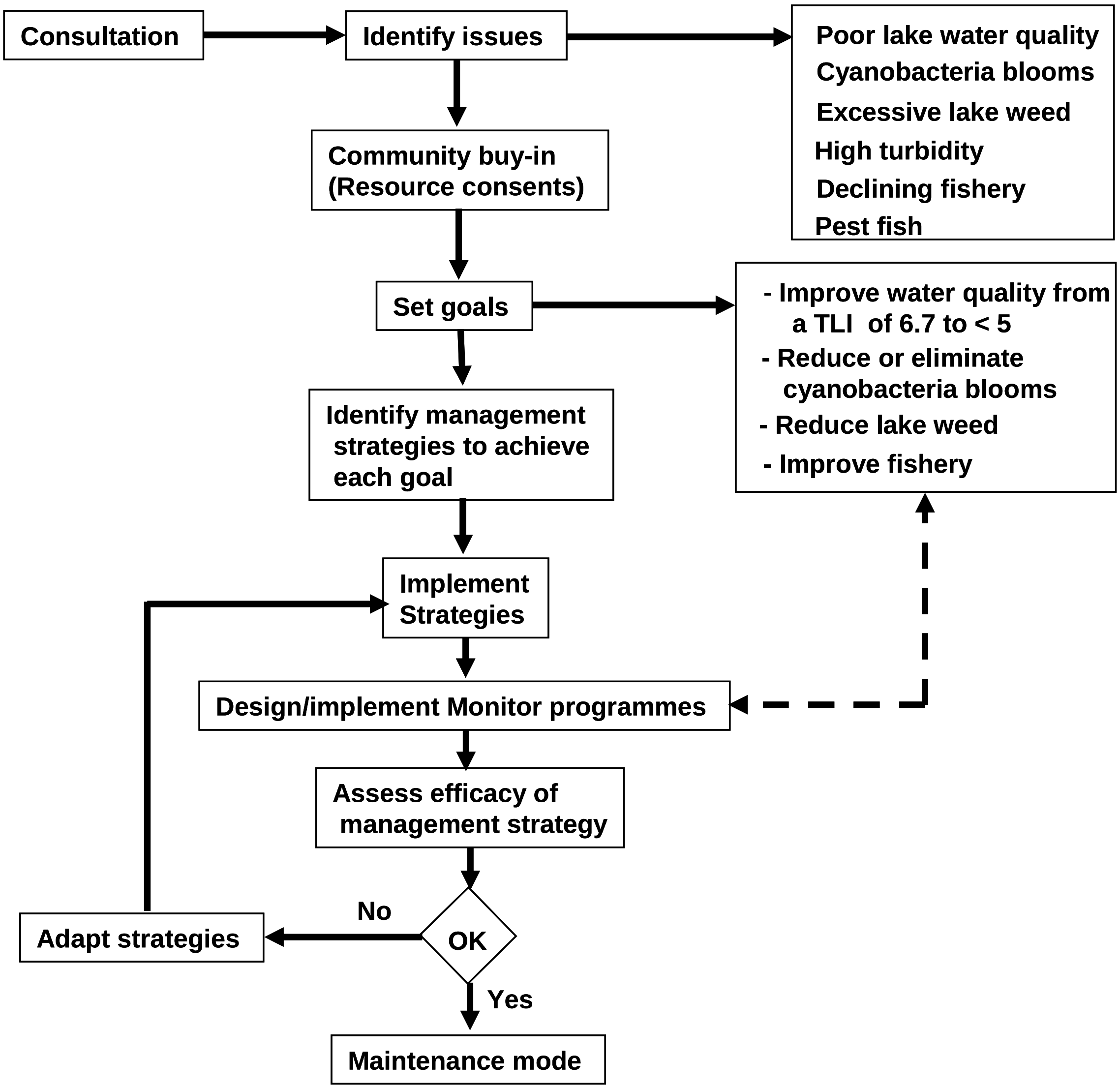


Figure 54. Management flow diagram for the rehabilitation of Lake Horowhenua: Overview example. The issues and goals shown in the flow diagram are examples and may be different or include other aspects of the lake when the actual management plan is developed.

The Management flow diagram (Figure 54) follows a logical progression starting with Consultation. The consultation process is seeking a mandate from the community to spend their money on the rehabilitation process. In the consultation process, the issues will be identified by all parties including the managers. Some of these issues may be trivial while others may appear insurmountable. The consultation process should find ways to achieve the rehabilitation which are compatible with the framework of regional policy and objectives. The desired outcome of the consultative process is an agreement by all parties on the issues and the remedial actions that are practical and affordable. It is important to the success of the rehabilitation project to obtain the community buy-in so that they accept what is to be done, understand why it is being done and they can follow the rehabilitation progress. Their buy in means that they have an input into the setting of the goals and their expectations of success can be managed. In short the community should “own” the rehabilitation project rather than having it forced on them so that they resent it.

Another critical part of the rehabilitation process is monitoring (see section 6). Without monitoring it is not possible to determine whether a specific strategy is working in the way it was intended. The measure of success is part of the management strategy and, consequently, the monitoring programme is an integral part of that strategy which allows the management strategy to be adjusted or adapted to improve its performance. The monitoring programme(s) fall inside a feed-back or adaptive management loop. The adaptive management option needs to be stated in the resource consent.

For example, the resource consent for rehabilitation work is often written in a way that is restrictive to the point that it actually prevents the success of the project. It may define the products and materials to be used and the treatment rates. These latter may have been the best estimates from a model but require to be adapted in the “real world” to accommodate environmental variability. Consequently, the resource consent needs to have “room to manoeuvre” but not be a licence to do “whatever it takes”.

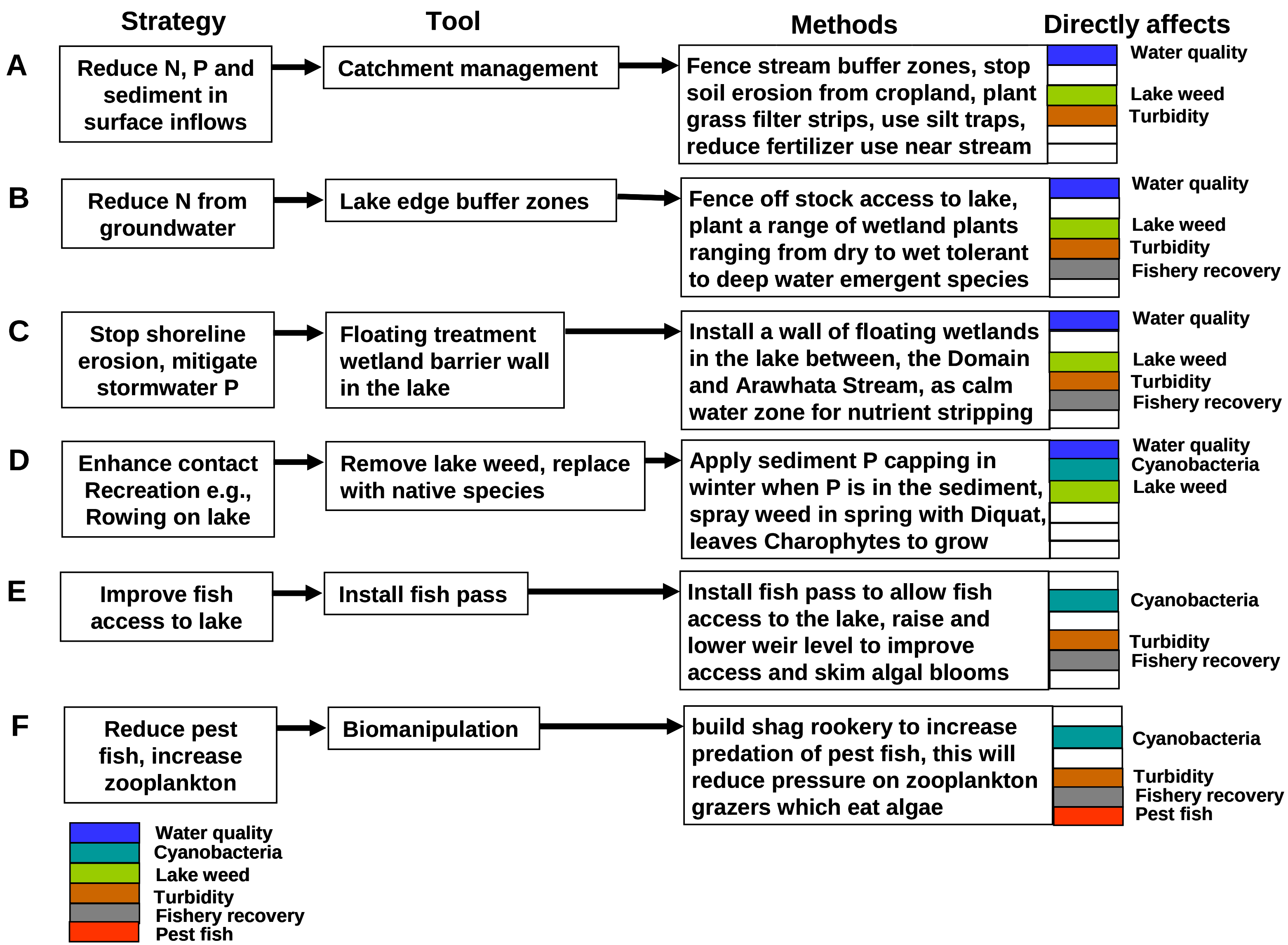


Figure 55. Management strategies required for the rehabilitation of Lake Horowhenua. These examples cover most issues but there may be others to consider.

The management strategies (Figure 55) identified from the flow diagram (Figure 54) include most issues but there may be other issues which could also be addressed in this way. The reasoning for the use of these specific management strategies is as follows:

A. **Reduce N, P and sediment in surface inflows: Why?** Before or concurrent with any in-lake management interventions, the catchment nutrient load on the lake needs to be reduced. Nutrients and sediment in the stream water are dispersed out into the lake where they support weed growth, enhance turbidity and degrade the lake water quality. **How?** On agricultural land, the exclusion of stock from direct access to the streams will reduce bed disturbance and thus sediment, bank erosion and stock defecation directly into the water. These riparian buffer zones need to be wide enough (about 5 m) to prevent over/under fence grazing destroying the plants and providing sufficient biomass of plants to assimilate nutrients in groundwater and overland flow. On market garden (crop) land, runoff from cultivated land increases sediment in the streams. It also removes arable land from production. To mitigate this soil loss, grass filters will precipitate the soil before it gets into the stream and silt traps will hold fine sediment/soil in detention ponds from which it can later be returned to the cultivated land. The riparian buffer zone will also reduce sediment transport into the stream. An alternative to clearing farm drains by digger is required (possibly spraying). Digging leaves disturbed bare soil exposed to erosion by rainfall with the resultant sediment flowing directly into the lake. Sediment detention ponds should be constructed to reduce road runoff directly entering streams and the lake. A general reduction in the use of fertilizers within the groundwater catchment of the lake by farming, horticulture, cropping and domestic gardening should be encouraged. Special attention should be given to educating home gardeners about the over-use of fertilisers. Although a relatively small industry in the catchment, the growing of vegetables, tomatoes and other crops without soil in nutrient solutions (i.e., hydroponically) has the potential to add high concentrations to the groundwater and streams There should be no permitted discharge of hydroponic fluid directly to a stream or to the groundwater table.

B. **Reduce N from groundwater: Why?** The high N concentrations in the groundwater come from land-use in the catchment. Both N and P would have originally reached the groundwater table but the high iron content of the stony subsoil has sequestered the P. Throughout New Zealand there is a tendency for the over-use of fertilisers in urban gardens and the urban area of Levin, which is above the main groundwater flow into the lake, will be augmenting the nutrients added to the groundwater further back in the catchment. **How?** These nutrients can be removed by plants at the lake edge, before the nutrients enter the lake or in the edge water by plants growing in the lake. Unlike stream water, which disperses out into the lake, the cooler groundwater flows along the lake bed and its nutrients are readily assimilated into plants through their root systems. The plants themselves provide habitats for fish and koura and thus support the recovery of the lake fishery. The habitat will be improved if the buffer zone has more than one wetland plant species. The nutrients stored in the plants should be removed by annually harvesting the leafy biomass or periodically having short (1 hour) stock grazing to crop the biomass before taking the stock out of the buffer zone again.

Another simple expedient is public education. Gardeners are notorious for the thought pattern “if one cup of fertilizer is good for the garden, then two will be better”. The second cup is not used by their plants and is leached into the groundwater with rainfall and irrigation. A change to slow release fertilizers and less of them would reduce the nutrient load in the groundwater.

C. **Stop shoreline erosion and mitigate storm water P: Why?** Not all nutrients from groundwater will be removed by the buffer zone due to wave action, which will also stir up the sediments and erode the unprotected shoreline on the eastern side of the lake. **How?** The nutrients that move beyond the buffer zone are in the water column where they can be removed via the root mass from Floating Treatment Wetlands. These root masses provide additional habitat for fish and koura. Because the FTWs are formed into a barrier wall, the wave energy that would normally stir up edge water sediment, is dissipated into the FTW wall. The calm zone between the shore and the FTW wall becomes a nutrient stripping and sedimentation zone for storm water discharges e.g., the Queen Street drain. Strategies B and C combine to remove >50% and >80% of the N and P external inputs to the lake.

   Wave action due to boat wakes is often a major problem causing shoreline erosion and high turbidity in shallow lakes. Prohibiting power boats travelling at more than 5 knots anywhere on the lake will reduce that source of wave action.

D. **Enhance contact recreation: Why?** Contact recreation includes rowing which is greatly impeded by the lake weed when it reaches the surface. Just removing the lake weed by harvesting or mulching is not the answer to this issue. For a long-term solution the tall lake weed, *Potamogeton,* needs to be replaced with a shorter macrophyte species – preferably native Charophytes. These where once a large part of the flora of the lake. As water clarity declined, the Charophytes would have become light limited allowing the taller *Potamogeton*, which can grow up to the light, to become dominant. To achieve rehabilitation of the flora of the lake, the water clarity needs to improve. Water clarity is affected by sediment and algal biomass. In Lake Horowhenua , the algal biomass in summer is driven by P release from the sediment following the collapse of the weed beds. The P release needs to be reduced which means managing the lake weed problem.

   How? There are three components that need balancing: **1)** *Potamogeton* is susceptible to the herbicide Diquat when it is young and the plants are clean of periphyton; **2)** *Charophytes* are tolerant to Diquat and their growth may be enhanced in the presence of this herbicide (J. Clayton, NIWA, pers. comm.); **3)** any plant biomass that decomposes on the sediment will cause anoxia with the concomitant release of P. To expedite the weed species exchange suggested, the P in the sediment needs to be inactivated using a sediment capping agent before the weed is sprayed. The weed needs to be sprayed soon after germination when the plants are small. Having the lake at its lowest water level at this time would reduce the amount of herbicide required and provide a higher light field to allow the *charophyte* seeds to germinate. The timing of this strategy is important with only one window of opportunity each year - early spring.

The consequences of removing the lake weed in this way is that the NO_3-N that was previously removed by the lake weed will be left in the water column. However, the P release from the sediment will not happen resulting in a P limited system and the water quality should resemble the clearer winter conditions. With calm weather in summer, water clarity could increase dramatically as the fine sediment settles out. This would support the growth of the native charophytes, which will take up the NO_3-N from the water column and reduce the algal biomass.

E. **Improve fish access to the lake: Why?** The construction of the weir blocked access to three key fish species, flounder, mullet and smelt. **How?** These species are known to use fish passes in other lakes. To restore these species to the lake requires a fish pass to be built around the weir. The lake level is controlled by the weir but there is capability for altering the water level by adding or removing planks on top of the weir. The guidelines for altering the lake level should be investigated with a mind to using the weir to skim algal biomass from the lake in summer and to aid the replacement of lake weed with native charophytes (see **D** above). Lowering the lake level in spring would reduce the flow velocities in the fish pass when the fish are migrating up the Hokio Stream.

   In conjunction with the installation of the fish pass, there should be a reduction in the netting and trapping activities in the lower Hokio Stream for one or more seasons the give the fish a chance to reach the lake without being over-fished.

F. **Reduce pest fish and increase zooplankton biomass: Why?** Pest fish can disturb the lake bed allowing sediment to be re-suspended more easily. The herbivorous species can browse the plant community and pull out plants. In searching for food, carp and especially Koi carp suck up sediment to sift out the chironomid larvae (blood worms). They then excrete the excess sediment into the water column raising the lake turbidity. Zooplankton eat algae and reduce part of that turbidity. Juvenile perch eat zooplankton preventing zooplankton populations becoming large enough to control the algal biomass. **How?** The long term solution is to use biomanipulation. Providing elevated roosts close to the lake will encourage the development of a shag population. The roosts should be away from urban developments and sited within the buffer zone but over dry land, preferably a small rise, not in a direct flow path for rainfall runoff to the lake. As the shag numbers increase, their search for food will increase predation of the perch and goldfish. The reduction in perch allows the zooplankton populations to increase and reduce the algal biomass. Less algal biomass results in clearer water making it easier for the shags to catch the pest fish. Without the cover of algal biomass driven turbidity, the juvenile perch can be predated by adult perch in a positive feedback loop. This technique will not remove all of the pest fish and a new equilibrium level will be established with a lower pest fish population density. When the food supply dwindles, the shags will search offshore as well as in the lake for fish. However, having established a rookery, they will not move away from the lake.

These six strategies are presented to show what could be done and how. Each strategy is independent but would work better in conjunction with other strategies. For example, there is little point in spending large sums of money on in-lake remedial interventions if no action is taken in the catchment to reduce the sources of the nutrients that stimulate growth.

Consequently, strategies A and B should be implemented either before or with the in-lake strategies C and D. Strategy B has already been implemented and a marginal buffer zone surrounds the lake. This may need to be enhanced to gain maximum benefit.

Strategies E and F can be implemented at any time, but sooner rather than later.

The effects of these strategies should be monitored to determine the level of success. This is very important to enable the adaptive management loop in the management flow diagram (Figure 54).

6 Monitoring

There are three types of monitoring programmes: 1) Compliance monitoring for resource consents (not covered in this report), 2) Monitoring programmes designed to assess the success of management interventions, and 3) Strategic or "state of the environment" (SOE) monitoring programmes designed to evaluate the overall water quality of the lake allowing it to be compared with other lakes nation-wide.

6.1 Sampling frequency

The frequency of sampling should be enough to include the range of variability expected from naturally occurring cycles. For management interventions, sampling frequency will be relatively short interval targeted to assess an expected effect. This monitoring will continue for the duration of the effect but the sampling frequency can be reduced as the effect becomes apparent and the monitoring is only required to confirm a trend.

For strategic monitoring, the sampling frequency should be sufficient to identify seasonal effects and to provide a meaningful annual estimate of the water quality of the lake.

In both types of monitoring, there may be specific conditions for when to sample, or not, so that event or diel driven effects (i.e., day-night cycles) are reduced. For example, sampling Lake Horowhenua during a wind storm or a flood event would put a bias towards high turbidity, when the turbidity in the lake on most other occasions may be trending lower.

6.2 Monitoring strategy success

Monitoring programmes designed to assess the success of management interventions are usually short-term focussed and designed to measure expected changes as a result of those interventions. Their design is usually part of the management strategy so that managers know whether the intervention has been successful and the degree of success achieved can be reported to the stakeholders. Baseline monitoring before the intervention is important to assess the degree of natural variability in the data to be measured. This information may come from strategic SOE monitoring (e.g., Arawhata Stream data; Figure 24) or a specific monitoring programme that will assess the success of the intervention. Suggested monitoring strategies for the six management strategies (Figure 55) include:

A. **Reduce N, P and sediment in surface inflows:** The measure of success may be the elimination of the nutrient spikes in the Arawhata Stream (Figure 21, Figure 24) and a reduction in overall concentration of all parameters measured. It should include the elimination of any anoxic events or low oxygen (<5 g m^{-3}) in summer. **Note the dissolved oxygen is expressed as a concentration, not % saturation.** Chemical transitions occur at specific DO concentrations while % saturation changes with temperature making it difficult to assess the position of the transition zone.

   The critical issues are where to sample a stream, what to measure, and what is the measure of success. Where to sample in Arawhata Stream should be at the historical SOE monitoring site. What to measure includes nutrients, suspended solids, dissolved oxygen.

   - Measure dissolved nutrients (NO_3-N, NH_4-N, DRP), total N, total P, and suspended solids in the major stream inflows affected.

- Measure the temperature and dissolved oxygen concentration, preferably with an in situ recording DO logger over a complete 24 hour cycle.

These parameters should be measured before and every month immediately after the intervention for a period of a year. They should also be measured during at least two high and two low flow events using a flow weighted auto-sampler over 24 hour periods. After a year post intervention, the data should be reviewed and the sampling frequency adjusted based on the results obtained. Sediment reductions and DO effects should be respond relatively quickly, nutrient reductions will induce a slower response.

B. **Reduce N from groundwater:** The measure of success of the buffer zones may be a reduction in dissolved nutrient concentrations in the lake edge water along the eastern shores of the lake. The measure of success in catchment interventions such as the reduction in garden fertilizer application, banning discharge of high concentration nutrient water into the ground, management of irrigation water, management of groundwater abstraction, etc., may be a reduction in NO_3-N and NH_4-N concentrations in the surface groundwater aquifer on the landward side of the buffer zone. Shallow bores holes <1 m deep with full depth screen liners would be the sampling points. Confirm the boreholes have high NO_3-N concentrations when first installed and that they have good flow characteristics. As groundwater flows are slow, sampling frequency can be every 2 or 3 months. Sampling strategies for the bore holes is to pump the groundwater out for at least 1 minute before collecting the sample.

- Measure NO_3-N, NH_4-N, DRP, temperature and dissolved oxygen.
- Also assess any changes in buffer zone plant communities annually in summer.

C. **Stop shoreline erosion and mitigate storm water P:** The measure of success of the use of floating treatment wetlands in this strategy will include a sustained reduction in dissolved nutrient concentrations in the lake edge water along the eastern shores of the lake, a reduction in suspended sediment, and an increase in the numbers of small fish and koura. Numbers of kākahi (freshwater mussels) may increase over time as food quality increases and the numbers of bullies, which are critical to the life cycle of the kākahi, increase. The nutrient and suspended sediment monitoring could be incorporated into the strategy B monitoring, assuming that proceeds. Additional monitoring is required:

- Measure changes in population dynamics of small fish using minnow traps.
- Measure population dynamics of koura using traditional Tau bunches.
- Measure changes in mussel populations using quadrats and photography.
- Measure sediment accumulation or erosion rates using buried sedimentation plates.

Monitoring would be undertaken each summer over a period of several years. Some harvesting of the buffer zone plants may be needed to encourage healthy growth.

D. **Enhance contact recreation:** The measures of success of this strategy will include a reduction in the number of complaints about lake weed by rowers/boaties, a reduction in the extent of cyanobacteria blooms which trigger a closure of the lake, a reduction in the area of lake affected by the tall weed and, ultimately, the re-establishment of charophyte meadows across the lake bed in place of the exotic tall lake weeds. The implementation of this strategy involves three parts and each can be assessed to give a measure of the success of that component. For sediment capping/P inactivation, there should be no DRP release from the sediments even if the weed beds collapse and cause anoxic conditions at the sediment surface.

- The DRP concentration in the lake at least monthly, and preferably every 2 weeks over summer, to follow the annual cycle of DRP in the lake.

After weed spraying, there should be no nuisance weed beds over the following summer

- Assess the state of the weed beds through the summer by visual inspection. This could include counting viable stems per unit area of lake bed.
- Measure the NO_3-N concentrations in the lake at least monthly, and preferably every 2 weeks during spring and summer.
- Assess the algal species assemblage at weekly intervals over the summer (cell counts and biovolume).
- Measure water clarity with a black disc system.

The charophytes should be increasing in abundance as the light levels improve.

- Survey the lake flora at least once per year by visual inspection (qualitative).
- Longer term, 5 yearly quantitative surveys should show a reduction in the area of lake bed affected by tall *Potamogeton* beds relative to charophytes.

Where the tall lake weed returns, a repeat spraying should be made the next spring.

E. **Improve fish access to the lake:** The measure of success will be the presence of any of the three species black flounder, mullet, and smelt in the lake after the installation of the fish pass. The abundance of Inanga and eels could also increase due to the reduction in fishing pressure in the lower Hokio Stream.

- Monitor whitebait catches, including bycatch.
- Survey the fishing people for observations of fish species they have seen.
- Photo-monitor the fish pass at appropriate times of year for migratory fish.
- Get accurate catch records from eel fishery.

F. **Reduce pest fish and increase zooplankton biomass:** The measure of success will be a reduction in the pest fish populations, an increase in zooplankton biomass and a reduction in algal biomass.

- Measure changes in population dynamics of juvenile perch using baited minnow traps and beach seine net hauls.

- Measure changes in population dynamics of zooplankton population using Schindler-Patalas traps.
- Measure changes in population dynamics algal species and biomass.

These monitoring programmes are intended to measure the success of the management strategy. Clearly there is overlap between success monitoring programmes for each strategy. Where multiple strategies are implemented together, the combined monitoring programme should include the relevant components from the individual monitoring programmes. It should also make use of data from the strategic monitoring programme for the lake.

6.3 Strategic Monitoring

A strategic monitoring programme is designed to assess the overall water quality of the lake. This is very different from the monitoring programme to assess success (or compliance monitoring). The success of the management strategies implemented will, however, become apparent in the strategic monitoring programme data over time.

Strategic monitoring is a long term programme. “The longer the better” as long term data enable the assessment of the natural variability in the lake water quality such as seasonal cycles, the El Niño – La Niña effect, the Southern Oscillation Index (SOI) and Interdecadal Pacific Oscillation Index (IPOI). These cycles affect climate and, consequently, the water balance of the lake via the stream and groundwater inflow volumes.

Strategic monitoring is also used to provide the annual SOE report for the lake. The SOE report typically focuses on use-related factors that would show a trend as the water quality changes. To obtain the annual SOE report value, strategic monitoring samples are collected at least 4 times a year to look at the main seasonal data. A more robust approach for Lake Horowhenua would be to tailor the strategic monitoring programme to follow changes in key elements in the lakes seasonal cycle and this may require sampling 6 times per year as a minimum until a coherent database is established. In contrast, lake weed bed surveys can be every 5 years.

In the Horizons Regional Council monitoring data from Lake Horowhenua (up to 2010), a broad range of parameters were measured but not on a regular basis. Consequently, there are gaps of key parameters in the long term database making trend analysis difficult or impossible. While the key parameters for the SOE report are measured, the monitoring programme included parameters that would assist in understanding how the lake works to enable management strategies for restoration to be developed.

Closer examination of the database shows that there are some issues with sampling time differences that generate data variability due to day-night (diel) effects on some parameters. For example almost any value of DO can be obtained for the Arawhata Stream in summer (Figure 23) depending on the time of day the stream is sampled. Similar diel cycles will occur for NO_3-N concentrations in the lake in spring, with concentrations decreasing in the afternoon due to uptake by lake weed for growth.

To eliminate these issues, it would be advisable to design a new strategic monitoring programme for Lake Horowhenua that meets the requirement of the SOE and provides good quality data on which to base decisions about management strategies for the rehabilitation of the lake.

There are three components to consider when designing a strategic monitoring programme: 1) the parameters to be measured; 2) when and how to sample; and 3) the time of day that the sample will be collected.

6.3.1 Parameters to be measured

To enable nation-wide comparisons, all lakes should be monitored for the same range of physico-chemical and microbiological water quality parameters. For simplicity these should also be the same as for freshwater rivers and streams. These parameters include onsite measurements such as dissolved oxygen, temperature, pH, conductivity and clarity as SD, as well as laboratory tests for chlorophyll *a*, turbidity, inorganic and organic suspended solids (SS and VSS), total and dissolved nutrients (TP, TN, DRP, NO_3-N, NH_4-N) and the faecal indicator bacteria, faecal coliform and *E. coli*. Phytoplankton samples should be collected for cyanobacteria identification and counts as well as abundance of other algal species on each occasion. These parameters include the information needed to calculate an annual TLI value using the original method (Burns et al., 1999, 2000) which was based on chlorophyll *a*, SD, TP, TN, volumetric hypolimnetic oxygen depletion rate (VHOD) and phytoplankton species and biomass. More recently only the first four parameters have been used for calculating the TLI value (Burns et al., 2005), and a TLI value can also be estimated where the SD data are missing (Verburg et al., 2010) (Section 4.1). This may be an appropriate measure of trophic level for Lake Horowhenua where even small breezes can re-suspend the lake sediments, due to wave action on the shore. Sampling will still need to occur over a minimum of a two-year period as the TLI is calculated on the two most recent years of data.

Biochemical oxygen demand (BOD) is often included in the analytical suite but should not be used when the BOD concentrations are consistently below the level of detection of <1 g m^{-3}. Because Lake Horowhenua is very shallow, water level should be recorded on every sampling occasion. The time of day the sample was collected should be recorded in New Zealand standard time (NZST) not daylight saving time. Other observations such as wind condition, cloud cover, presence of an algal bloom or shoreline algal scum and the abundance of water fowl should also be recorded/

Note: The water level staff gauge in Lake Horowhenua should be checked for readability and careful instruction given on how to read and record the level, as the historic database has level fluctuations that range from 0 to 3.2 m; well beyond the depth of the lake.

Analytical methods should provide the lowest detection limits possible to assist with detecting trends in water quality over time. Routine measurements of dissolved oxygen at SOE surface water sampling sites in the lake and streams are recorded as absolute concentration as well as in percentage saturation. Although Secchi depth is used to record vertical clarity in Lake Horowhenua, with a maximum depth of 1.6 m and high turbidity, it may be more appropriate to measure horizontal clarity with a black disc. However, if a trophic level index monitoring programme is implemented as part of the strategic monitoring programme then vertical clarity, i.e., Secchi depth measurements, should also be taken from the mid-lake site.

6.3.2 When and how to sample

For strategic monitoring, individual regional councils use different sampling frequencies including 4 times per year, 6 times per year, or monthly. Each sampling strategy has a field cost and an analytical cost which will be part of the consideration when designing the monitoring programme. The recommended minimum sampling regime is 6 times per year at regular 2-monthly intervals (Table 6-1) , but monthly sampling is preferred for the first year to establish the seasonality of the data. It is better to have a higher sampling frequency with fewer sites. In Lake Horowhenua, water samples from the mid-lake sites should be combined to provide a single composite sample that is representative of the whole lake. If the TLI monitoring programme is required, it would be appropriate to collect just the 4 parameters required from the mid-lake site on the intervening month (Table 6-1).

Table 6-1: Strategic monitoring programme design. Recommended monitoring programme for Lake Horowhenua showing the full range of parameters required to meet the One Plan standards. To match the seasonal cycles of the lake, one sampling date must be in mid March and that determines the other sampling dates. The programme shows 4 samplings per year (red) as the absolute minimum with the 2 extra sampling dates (pink) to make a total of 6 times per year. Reduced options for just TLI data are also included. Site details and observations should be recorded on every occasion. (* cyanobacteria toxins are only tested for when there are high levels of cyanobacteria in the lake.)

Strategic Monitoring Programme for Lake Horowhenua

Monitoring equipment

DO / Temperature	Dissolved oxygen / temperature probe on a cable
Secchi disc	Weighted black and white quartered 20 cm disc on soft tape measure
Water sampler	Composite of 3 integrated tube samples from each of 3 traditional monitoring sites (9 tube samples)
Zooplankton net	Composite net haul with 63µm mesh on cod end - OPTIONAL

Sampling time to be around 10 am or around 4 hours after sunrise

Parameter	Sample Type	Units	J	F	M	A	M	J	J	A	S	O	N	D
On site observations														
Start and finish time	Date and time	yyyy:mm:dd hh:mm												
Lake level	Staff guage	m												
wind condition	estimate	text												
cloud cover	estimate	%												
Algal bloom / scum	Shoreline observation	Yes/No												
In lake measurements														
Temperature	Instrument profile	°C												
Dissolved oxygen	Instrument profile	g m^{-3}												
Clarity	Secchi depth	m												
Depth	Tape measure	m												
Laboratory measurements														
Algal species	Water sample	cell counts, biovolume												
Chlorophyll *a*	Water sample	mg m^{-3}												
SS / VSS	Water sample	g m^{-3}												
Turbidity	Water sample	NTU												
E. coli	Water sample	MPN / 100 ml												
Conductivity @ 25°C	Water sample	µS cm^{-1}												
pH	Water sample	pH												
DRP	Water sample	mg m^{-3}												
TP	Water sample	mg m^{-3}												
NH_4-N	Water sample	mg m^{-3}												
NO_3-N	Water sample	mg m^{-3}												
TN	Water sample	mg m^{-3}												
Cyanobacteria toxins*	Water sample	mg m^{-3}												

TLI parameters | 4 times per year | Additional for 6 times per year

For each parameter there is a protocol of how the sample should be collected.

The water sample from Lake Horowhenua should be collected using an ‘integrated-tube’ sampler. This consists of a wide bore (~20 mm ID) plastic tube lowered through the full depth of the water column to just above the lake bed (See Appendix 2). This diameter tube will collect about 450 ml from a 1.5 m deep site. Ideally, water should be collected from each of the three monitoring sites and then combined to produce a single composite sample. This would be more representative than a sample from single site. The historical database indicates some water quality differences between the three monitoring sites but there is insufficient data to determine whether the differences are ‘real’, as in “being a characteristic of that site”, e.g., consistently different wind stress at each site, or just an artefact of sampling on the day e.g., differences in the time of day the sample was collected.

To ensure a representative sample is collected from the lake, the composite sample is made up of 3 integrated tube collections from each of the 3 traditional monitoring sites (i.e., a total of 9 integrated tube collections) and these are fully mixed in a 5-litre plastic sample bottle. Mixing is critical before any subsample is taken from the bulk composite sample. Suspended material will settle or float if the sample is not mixed, and this will introduce an error.

Bacteria samples need to be collected separately using protocols for that type of sampling.

Aliquots for algal species enumeration and biomass need to be decanted into a specific bottle for this purpose immediately after fully mixing the bulk composite sample. Normally the algal species sample is preserved with Lugols iodine. This can be done in the field or laboratory.

A missing link in most lakes data bases is the estimation of zooplankton biomass and species composition. Zooplankton graze the algae and thus impact on the chlorophyll *a* concentrations. At the beginning of an algal bloom, zooplankton biomass will be low. As the bloom develops, zooplankton biomass increases rapidly until their abundance stops the bloom. Zooplankton biomass would be “nice” but it is a discretionary parameter. If collected, composite net haul samples should be preserved with isopropanol.

6.3.3 Time of day effects

Shallow eutrophic lakes like Lake Horowhenua are more susceptible to diel cycles than deeper lakes which have a large hydraulic inertia / dilution capacity. Consequently, the time of day, relative to sunrise, is very important.

- Temperature follows a continuously cycle from low at night to high around 2 hours after midday. Ideally a temperature logger should be used to capture the full diel cycle.

- Dissolved oxygen is continuously consumed by respiration processes in the water 24 hours per day. Photosynthesis replenishes the oxygen during the hours of daylight proportional to the amount of light and the mass of plant biomass. Photosynthesis compensates for respiration overwhelming it in the morning and can be continuously variable reaching maximum levels around midday before declining again in the late afternoon. On cloudy days the maximum DO levels will be lower than on a bright sunny day. Maximum DO levels are also affected by the presence of weed beds and phytoplankton blooms. Ideally a dissolved oxygen logger should be used to capture the full diel cycle.

- Plant nutrients, especially NO_3-N, are consumed by the lake weed for growth. The Lake Horowhenua database indicates that the weed beds can substantially reduce the free NO_3-N from the water column. With high NO_3-N concentrations in the stream and groundwater inflows, there is a potential to have a time-of-day effect on the NO_3-N concentration in the lake sample.

- Sediment in Lake Horowhenua is easily disturbed by wind-induced wave action against the shore. The lake experiences katabatic winds daily during summer starting late morning around 11:00 am and reaching maximum velocities in late afternoon before stopping around 18:00 pm (Figure 6C). Sediment resuspension at the time of sampling will also temporally affect nutrient concentrations.
  - Sampling during the afternoon has the potential to produce a bias towards high suspended solids, dissolved oxygen, turbidity and nutrients than samples collected in the morning.
  - For Lake Horowhenua, it is recommended that water sampling is completed before
    10 am or 4 hours after sunrise, whichever is the earlier time.

6.3.4 Data integrity

Another important consideration is the maintenance of the database, the checking of data being loaded into the database, and the correction of errors in the database when these are found. This is outside the scope of this report.

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Appendix A Act of Parliament pertaining to Lake Horowhenua

Reserves and Other Lands Disposal Act 1956

Public Act 1956 No 53, Date of assent 25 October 1956.

18. Special provisions relating to Lake Horowhenua

Whereas under the authority of the Horowhenua Block Act 1896 [Repealed], the Maori Appellate Court on the twentieth day of September, eighteen hundred and ninety-eight, made an Order determining the owners and relative shares to an area of thirteen thousand one hundred and forty acres and one rood, being part of the Horowhenua XI Block: And whereas the said area includes the Horowhenua Lake (as shown on the plan lodged in the office of the Chief Surveyor at Wellington under Number 15699), a one chain strip around the lake, the Hokio Stream from the outlet of the lake to the sea, and surrounding land: And whereas certificate of title, Volume 121, folio 121, Wellington Registry, was issued in pursuance of the said Order: And whereas by Maori Land Court Partition Order dated the nineteenth day of October, eighteen hundred and ninety-eight, the lake was vested in trustees for the purposes of a fishing easement for all members of the Muaupoko Tribe who might then or thereafter own any part of the Horowhenua XI Block (in this section referred to as the Maori owners): And whereas the minutes of the Maori Land Court relating to the said Partition Order recorded that it was also intended to similarly vest the one chain strip around the lake, the Hokio Stream from the outlet of the lake to the sea, and a one chain strip along a portion of the north bank of the said stream, but this was not formally done: And whereas the Horowhenua Lake Act 1905 declared the lake to be a public recreation reserve under the control of a Domain Board (in this section referred to as the Board) but preserved fishing and other rights of the Maori owners over the lake and the Hokio Stream: And whereas by section ninety-seven of the Reserves and Other Lands Disposal and Public Bodies Empowering Act 1916 the said one chain strip around the lake was made subject to the Horowhenua Lake Act 1905, and control was vested in the Board: And whereas subsequent legislation declared certain land adjoining the said one chain strip, and more particularly firstly described in subsection thirteen of this section, to form part of the recreation reserve and to be under the control of the Board: And whereas as a result of drainage operations undertaken some years ago on the said Hokio Stream the level of the lake was lowered, and a dewatered area was left between the margin of the lake after lowering and the original one chain strip around the original margin of the lake: And whereas this lowering of the lake level created certain difficulties in respect of the Board's administration and control of the lake, and in view of the previous legislation enacted relating to the lake, doubts were raised as to the actual ownership and rights over the lake and the one chain strip and the dewatered area: And whereas a Committee of Inquiry was appointed in 1934 to investigate these problems: And whereas the Committee recommended that the title to the land covered by the waters of the lake together with the one chain strip and the said dewatered area be confirmed by legislation in ownership of the trustees appointed in trust for the Maori owners: And whereas certain other recommendations made were unacceptable to the Maori owners, and confirmation of ownership and further appointment of a Domain Board lapsed pending final settlement of the problems affecting the lake: And whereas by Maori Land Court Order dated the eighth day of August, nineteen hundred and fifty-one, new trustees were appointed for the part of Horowhenua XI Block in the place of the original trustees, then all deceased, appointed under the said Maori Land Court Order dated the nineteenth day of October, eighteen hundred and ninety-eight: And whereas agreement has now been reached between the Maori owners and other interested bodies in respect of the ownership and control of the existing lake, the said one chain strip, the said dewatered area, the said Hokio Stream and the chain strip on a portion of the north bank of that stream, and certain ancillary matters, and it is desirable and expedient that provision be made to give effect to the various matters agreed upon: Be it therefore enacted as follows:

a. For the purposes of the following subsections:

Lake means that area of water known as Lake Horowhenua enclosed within a margin fixed by a surface level of 30 feet above mean low water spring tides at Foxton Heads

Dewatered area means that area of land between the original margin of the lake shown on the plan numbered SO 15699 (lodged in the office of the Chief Surveyor, at Wellington) and the margin of the lake as defined aforesaid

Hokio Stream means that stream flowing from the outlet of the lake adjacent to a point marked as Waikiekie on plan numbered SO 23584 (lodged in the office of the Chief Surveyor, at Wellington) to the sea

b. Notwithstanding anything to the contrary in any Act or rule of law, the bed of the lake, the islands therein, the dewatered area, and the strip of land one chain in width around the original margin of the lake (as more particularly secondly described in subsection thirteen of this section) are hereby declared to be and to have always been owned by the Maori owners, and the said lake, islands, dewatered area, and strip of land are hereby vested in the trustees appointed by Order of the Maori Land Court dated the eighth day of August, nineteen hundred and fifty-one, in trust for the said Maori owners.

c. Notwithstanding anything to the contrary in any Act or rule of law, the bed of the Hokio Stream and the strip of land one chain in width along a portion of the north bank of the said stream (being the land more particularly thirdly described in subsection thirteen of this section), excepting thereout such parts of the said bed of the stream as may have at any time been legally alienated or disposed of by the Maori owners or any of them, are hereby declared to be and to have always been owned by the Maori owners, and the said bed of the stream and the said strip of land are hereby vested in the trustees appointed by Order of the Maori Land Court dated the eighth day of August, nineteen hundred and fifty-one, in trust for the said Maori owners.

d. Notwithstanding the declaration of any land as being in Maori ownership under this section, there is hereby reserved to the public at all times and from time to time the free right of access over and the use and enjoyment of the land fourthly described in subsection thirteen of this section.

e. Notwithstanding anything to the contrary in any Act or rule of law, the surface waters of the lake together with the land firstly and fourthly described in subsection thirteen of this section, are hereby declared to be a public domain subject to the provisions of Part 3 of the Reserves and Domains Act 1953: Provided that such declaration shall not affect the Maori title to the bed of the lake or the land fourthly described in subsection thirteen of this section: Provided further that the Maori owners shall at all times and from time to time have the free and unrestricted use of the lake and the land fourthly described in subsection thirteen of this section and of their fishing rights over the lake and the Hokio Stream, but so as not to interfere with the reasonable rights of the public, as may be determined by the Domain Board constituted under this section, to use as a public domain the lake and the said land fourthly described.

f. Nothing herein contained shall in any way affect the fishing rights granted pursuant to section nine of the Horowhenua Block Act 1896 [Repealed].

g. Subject to the provisions of this section, the Minister of Conservation shall appoint in accordance with the Reserves and Domains Act 1953 a Domain Board to control the said domain.

h. Notwithstanding anything to the contrary in the Reserves and Domains Act 1953, the Board shall consist of—

(a) Four persons appointed by the Minister on the recommendation of the Muaupoko Maori Tribe:
(b) One person appointed by the Minister on the recommendation of the Horowhenua County Council:
(c) Two persons appointed by the Minister on the recommendation of the Levin Borough Council:
(d) The Director-General of Conservation, ex officio, who shall be Chairman

i. Notwithstanding anything in the Land Drainage Act 1908, the Soil Conservation and Rivers Control Act 1941, or in any other Act or rule of law, the Hokio Drainage Board constituted pursuant to the said Land Drainage Act 1908 is hereby abolished, and all assets and liabilities of the said Board and all other rights and obligations of the said Board existing at the commencement of this Act shall vest in and be assumed by the Manawatu Catchment Board, and until the said Catchment Board shall have completed pursuant to the Soil Conservation and Rivers Control Act 1941 a classification of the lands previously rated by the said Drainage Board, the said Catchment Board may continue to levy and collect rates in the same manner as they have hitherto been levied and collected by the said Drainage Board.

j. The Manawatu Catchment Board shall control and improve the Hokio Stream and maintain the lake level under normal conditions at thirty feet above mean low water spring tides at Foxton Heads: Provided that before any works affecting the lake or the Hokio Stream are undertaken by the said Catchment Board, the prior consent of the Domain Board constituted under this section shall be obtained: Provided further that the said Catchment Board shall at all times and from time to time have the right of access along the banks of the Hokio Stream and to the lake for the purpose of undertaking any improvement or maintenance work on the said stream and lake.

k. The District Land Registrar for the Land Registration District of Wellington is hereby authorised and directed to deposit such plans, to accept such documents for registration, to make such entries in the register books, and to do all such other things as may be necessary to give effect to the provisions of this section.

l. The following enactments are hereby repealed:

(a) The Horowhenua Lake Act 1905:
(b) Section ninety-seven of the Reserves and Other Lands Disposal and Public Bodies Empowering Act 1916:
(c) Section sixty-four of the Reserves and Other Lands Disposal and Public Bodies Empowering Act 1917:
(d) Section fifty-three of the Local Legislation Act 1926

m. The land to which this section relates is particularly described as follows:

Firstly, all that area in the Wellington Land District, being Subdivision 38 and part of Subdivision 39 of Horowhenua 11B Block, situated in Block I, Waiopehu Survey District, containing thirteen acres three roods and thirty-seven perches, more or less, and being all the land comprised and described in certificate of title, Volume 165, folio 241, Wellington Registry: as shown on the plan marked L and S 1/220, deposited in the Head Office, Department of Lands and Survey, at Wellington, and thereon edged red (SO Plan 15589).

Secondly, all that area in the Wellington Land District situated in Block XIII, Mount Robinson Survey District, Block II, Waitohu Survey District, and Block I, Waiopehu Survey District, containing nine hundred and fifty-one acres, more or less, being part of the land comprised and described in certificate of title, Volume 121, folio 121, Wellington Registry, and being more particularly the bed of the lake, the islands therein, the dewatered area, and the strip of land one chain wide around the original margin of the lake: as shown on the plan marked L and S 1/220A, deposited in the Head Office, Department of Lands and Survey, at Wellington, and thereon edged blue, and coloured orange and red respectively (SO Plan 23584).

Thirdly, all that area in the Wellington Land District situated in Block IV, Moutere Survey District, and Block II, Waitohu Survey District, containing forty acres, more or less, being part of the land comprised and described in certificate of title, Volume 121, folio 121, Wellington Registry, and being more particularly the bed of the Hokio Stream together with a strip of land one chain wide along a portion of the north bank of the said stream: as shown on the plan marked L and S 1/220A, deposited in the Head Office, Department of Lands and Survey, at Wellington, and thereon coloured blue and sepia respectively (SO Plan 23584).

Fourthly, all that area in the Wellington Land District situated in Block I, Waiopehu Survey District, being that portion of the dewatered area together with so much of the one chain strip of land herein secondly described as in each case fronts Subdivision 38, Horowhenua 11B Block, herein firstly described, and being parts of the land coloured orange and red respectively on the plan marked L and S 1/220A, deposited in the Head Office, Department of Lands and Survey, at Wellington (SO Plan 23584).

Subsection (7) was amended, as from 1 April 1987, by omitting the word “Lands” and substituting the words “Conservation” pursuant to section 65(1) Conservation Act 1987 (1987 No 65).

Subsection (8)(d) was amended, as from 1 April 1987, by omitting the words “Commissioner of Crown Lands for the Land District of Wellington” and substituting the words “Director-General of Conservation” pursuant to section 65(1) Conservation Act 1987 (1987 No 65).

Appendix B Integrated tube sampler

For Lake Horowhenua with a maximum depth of 1.6 m and being sampled from a small boat, the integrated tube sampler consists of a 1.8 m length of ~20 mm diameter **transparent flexible** plastic (PVC) tube (Figure B-1). Considered vertically, the top is trimmed to be a smooth right-angled end and fitted with a plastic plumping ball-valve which can close the tube with a 90º turn. A length of cord is tied around the tube just below the tap leaving about 2 m of free end – this is the safety tether. Another length of cord is run the full length of the tube to the bottom, leaving about a metre spare at each end – this is the retrieval cord. The lower end of the cord is tied through one small hole drilled through the wall of the tube just above the bottom so that the cord has minimal effect on the inside diameter of the tube. The remaining free end of this cord is tied to a lead weight (e.g., one or two 8 oz fishing sinker(s) or equivalent) so that the weight hangs about 20 cm below the bottom end of the tube. Allow enough length to accommodate soft sediment so that if the weight sinks into the sediment, the bottom end of the tube is still 20 cm above the lake bed. (Trial and error first time only). Tie a small plastic float to the top end of the long retrieval cord so that it can be easily recovered if it falls out of the boat. Before and after use, wash the tube thoroughly with tap water and store it out of sunlight.

The integrated sample is ready to use. A 5-litre sample bottle and a funnel with a wide spout that will fit into the sample bottle, are also needed.

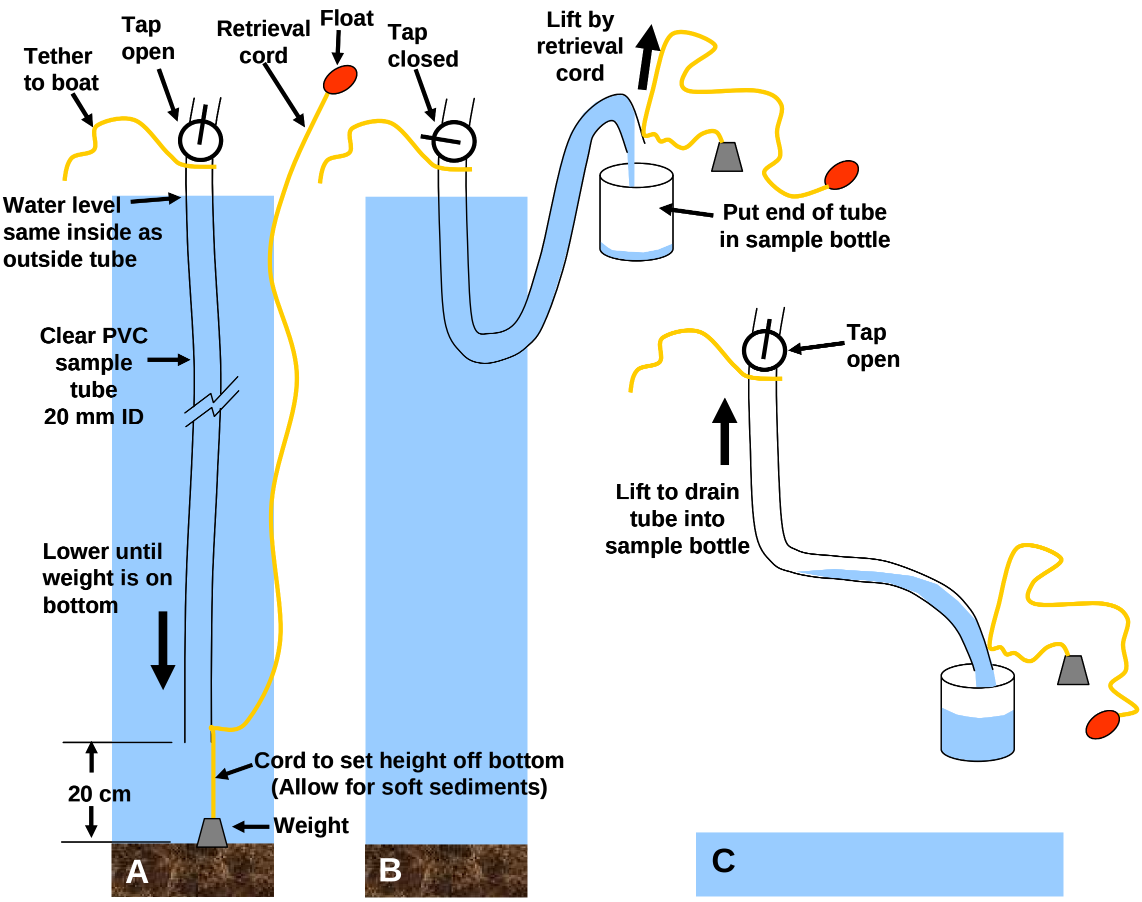

Figure B-1: Integrated tube sampler operation schematic. A) collecting sample, B) retrieving sample, and C) transferring sample to sample bottle. See text for details.

Operating the integrated tube sampler on the boat is as follows:

1 Rinse all sampling gear and the sample bottle with surface lake water.

2 **Open the tap** and tie the safety tether cord to the boat leaving plenty of slack line.

3 Lower the weighted end of the tube slowly through the water column **so that the water level inside the tube remains at the same level as the water outside the tube**. (Figure B-1 A).

4 Stop when the tube feels lighter as the weight reaches the lake bed. The bottom end of the tube will be within 20 cm of the lake bed.

5 **Close the tap** and let the tap end lie in the bottom of the boat.

6 Use the retrieval cord to lift the bottom end of the tube to the surface and immediately put the bottom end of the tube in the sample bottle (Figure B-1 B).

7 Raise the top end of the tube above the rest of the tube before **opening the tap**.

8 Drain the water into the sample bottle (Figure B-1 C).

9 Repeat steps 3 to 8 twice more to collect three integrated tube samples and combine them in the sample bottle to form a composite sample.

10 Mix well before filling the sample bottle via the funnel.

If more than one site is being sampled, unless spatial differences are required water from all sites can be combined in the bucket before filling the sample bottle. A composite sample from all sites would be appropriate for Lake Horowhenua as being more representative of the whole lake for the TLI estimate.

To prevent the spread of nuisance algal species, the integrated tube sampler for Lake Horowhenua should not be used on any other lake.

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