Ecological Monitoring of Artificial Destratification Effects in Lake Rotoehu: 2014-2015

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Executive Summary

Lake Rotoehu is a shallow (mean depth 8 m), polymictic eutrophic lake in the Rotorua lakes district. Catchment land use intensification has contributed to a decline in water quality and persistent cyanobacterial algal blooms for the past 20 years and combines with a background of nutrient-rich geothermal inflows. Two artificial mixing devices were deployed in Lake Rotoehu in the spring of 2012 by the Bay of Plenty Regional Council with the aim of preventing lake thermal stratification. They were previously operated over the summer-autumn periods of 2012–13 and 2013–14 and their effects were monitored on a monthly basis until June 2014. The devices force compressed air through a diffuser near the lake bottom, and into three large vertical cylinders. Buoyancy, caused by the rising bubbles, draws water from the bottom of the lake up through the vertical cylinders and directs it horizontally into the surface mixed layer (epilimnion). This causes mixing of thermally distinct layers of the water column, and is designed to inhibit stratification and prevent hypolimnetic deoxygenation. Results from previous monitoring indicate that the mixing devices produced a limited, localised mixing effect, but had no influence on whole lake ecology. With a view to improving performance, modifications, including channelling all air through a single mixing device and increasing compressor output, were effected in preparation for the 2014–15 summer–autumn seasons. The University of Waikato was contracted by the Bay of Plenty Regional Council to conduct a monitoring programme to ascertain the effect of these modifications.

Three sites within Lake Rotoehu were sampled monthly at similar depths from October 2014 to May 2015. Sites 1 and 3 were located approximately 30 m from a mixing device and Site 2 acted as a control site as it was considered to be unaffected by the direct effects of the mixing. Zooplankton, phytoplankton and nutrient samples (total nitrogen, total phosphorus, nitrate, ammonium and dissolved phosphorus) were collected at two different depths (0.5 m and 9.0 m), at each site. In addition, CTD (conductivity, temperature, depth) vertical profiles were also taken at each site. Temperature data were also obtained from a fixed-sensor, high-frequency monitoring buoy deployed in the lake and used to determine lake stability.

The summer of 2014-15 produced conditions favourable for the formation of prolonged lake stratification. This resulted in a marked decline in water quality during January–May 2015 in comparison to the previous three years of monitoring data. Evidence of localised homogenisation by the mixing device of the dominant phytoplankton and zooplankton taxonomic groups demonstrated notable improvement in mixing device performance. Temperature profile data were more equivocal, with no clear observations of the effects of artificial mixing over an extended area of the lake. However, supplemental profiles taken from within the discharge plumes (approximately 30 m from the mixing device) demonstrated evidence of hypolimnetic water being discharged to the epilimnion.

It is concluded that the current design and configuration of the artificial mixing devices is effective in drawing water from the hypolimnion into the epilimnion. However, it appears that the thermal mass of Lake Rotoehu is too large for a single mixing device to overcome and there appears to be no substantial ecological benefit in continuing to operate the devices without substantial modification. If significant, sustained improvements in water quality are to be achieved using these devices, further capital investment will be required. Further research investigating the efficiency and area of influence of the mixing devices may help to improve their design and operation.

Acknowledgements

John Tyrrell, Christopher Dada David Hamilton and Andy Bruere are thanked for reviewing this document. Joe Butterworth, Jeremy Garrett-Walker, Kohji Muraoka, Chris McBride and Niroy Sumeran are thanked for their assistance with data acquisition and analysis. This project was funded both directly by Bay of Plenty Regional Council and indirectly the Bay of Plenty Regional Council Chair in Lake Restoration. Additional support was provided through the Lake Biodiversity Restoration programme funded by the Ministry of Business, Innovation and Employment (Contract UOWX0505).

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Introduction

Thermal stratification of lakes primarily occurs during the summer months due to increased insolation. Heating of the surface waters results in the formation of different water densities between the surface and bottom waters, thus inhibiting lake mixing (Wetzel, 2001). The decline in lake mixing is associated with deoxygenation of the benthic and subsequently, hypolimnetic layers due to bacterial respiration. The resulting hypoxic conditions decrease the redox potential in the hypolimnion, bringing about the release of phosphate and ammonium from bottom sediments (Wetzel, 2001). Once the lake mixes, these plant nutrients are released into the epilimnion where they facilitate the formation of algal blooms. In shallow (<10 m) polymictic lakes, such as Rotoehu, stratification followed by mixing may occur over relatively short timescales of hours to days, while in deeper monomictic lakes stratification is usually stable, only breaking down with the onset of cooler temperatures in the autumn (Wetzel, 2001). Prevention or disruption of lake stratification could reduce the release of nutrients from bottom sediments into the water column and reduce the likelihood of noxious algal bloom formation (Miles and West, 2011).

Artificial destratification is a potential management tool for the prevention of noxious algal blooms in eutrophic freshwater systems (Heo & Kim, 2004). Aerators and mechanical mixers have been used to artificially break down stratification and maintain oxygenation of the hypolimnion. These devices typically function in one of two ways; aerators diffuse air into the water column near the sediment to oxygenate the hypolimnion. Alternatively, mechanical mixers use impellers to mix water from the epilimnion with water from the hypolimnion breaking down the thermal gradient and allowing mixing to occur.

In the spring of 2012, two artificial mixing devices were installed in Lake Rotoehu in preparation for operation in the summer of 2012–13. Designed by Hans Burggraaf (Page Macrae Engineering), the devices employ compressed air to entrain water from the hypolimnion and draw it to the surface within a draft tube. Like other destratification devices, this approach draws cooler hypolimnetic water to the surface where it mixes with warmer surface waters, preventing the formation of different thermal layers in the lake. The advantage of the new design is the potential for the majority of compressed air to be recycled, reducing energy costs.

The effects of the artificial mixing devices were monitored over the spring-autumn seasons of 2012–13 and 2013–14. A mixture of routine monthly monitoring was combined with intensive weekly monitoring during February–March 2013. No significant effects were observed on lake nutrient concentrations, plankton community composition and physical attributes during both years of mixing device operation (McBride et al. 2015). However, there was some evidence for localised mixing in close proximity (<50 m) to the devices as evidenced by homogenisation of plankton communities and minor reductions in thermal and dissolved oxygen gradients between the epilimnion and hypolimnion in comparison to control sites (McBride et al. 2015). It should also be noted that in the 4 years prior to 2015 an improving trend in Lake Trophic Lake Index (TLI) was evident (2011 TLI: 4.4 –2014 TLI: 4.0) (Creagh 2015). This was supported by the comparatively low chlorophyll *a* concentrations (mean 3.9 µg l^{-1}) and absence of cyanobacterial blooms during the 2012–13 and 2013–14 summer months (McBride et al. 2015).

In order to ascertain the full potential of the mixing devices, an alternative approach was employed for the 2014–15 season. Rather than directing air through both devices, only one device was operated at a time, enabling the full output of the compressor to be delivered through that device. In addition, the total available output from the compressor was increased by approximately 40% (mean flow 3.16 m^3 s^{-1}). Air flow delivery was alternated between the mixing devices, with the southern device operating from 5 November 2014 to 22 January 2015 and 29 March 2015 to 29 May 2015 while the northern device was in operation from 22 January 2015 to 29 March 2015. This approach provided an opportunity to determine the effects of the mixing devices on the plankton community and whether the increased flow through a single device would have a significant localised effect in comparison to previous monitoring.

This report presents a comparative analysis of plankton community abundance, zooplankton and phytoplankton distribution, temperature profiles, nutrient composition, as well as the degree of thermal stratification at monitoring sites where mixing devices were operational at Lake Rotoehu from October 2014 to May 2015. The report also presents recommendations for consideration of continued operation of the mixing devices. Results from these studies may help inform the revision of the current restoration initiatives, and aid the design of similar artificial mixing and nutrient management programmes in other lakes.

Methods

Site description

Lake Rotoehu is a shallow (mean depth 8.2 m), moderately sized (795 ha), polymictic lake in the northeast of the Rotorua/Te Arawa lakes district (Scholes & Bloxham 2008). The catchment land use is relatively evenly distributed between exotic forestry, native vegetation and pasture (Scholes 2009), with small settlements at Ngamimiro Bay and Otautu Bay on the eastern side of the lake. Inflow to the lake is from small surface streams, ground water and geothermal springs (Donovan, 2003). Geothermal waters contribute dissolved nitrogen and phosphorus to the lake, and the trophic state of the lake has historically been classified as mesotrophic to eutrophic (Jolly & Chapman 1977; Scholes 2009). The Trophic Lake Index (TLI) (~4.5) has been relatively stable since 1994 (Scholes 2009) with an improving trend in water quality for the past three years (2012-2014). However, there was a significant decline in lake TLI during 2015 (Figure 1). Associated with elevated TLI, Lake Rotoehu has experienced regular cyanobacteria blooms since 1993 and the Lake Rotoehu Action Plan was formulated to address these problems by targeting a reduction in the TLI to 3.9 (Lake Rotoehu Action Plan 2007).

Monthly monitoring at three sites was undertaken between October 2014 and May 2015. Sites 1 and 3 were located approximately 30 m from the southern and northern mixing devices respectively. Site 2 was selected to be as distant from the mixing devices as possible but still be subject to the same lake conditions as Site 1 and Site 3 (Figure 2). The study design was such that, sites 1 and 3 each acted intermittently, either as an additional control when the adjacent mixing device was not in operation or as a treatment site whenever the adjacent mixing device was in operation.

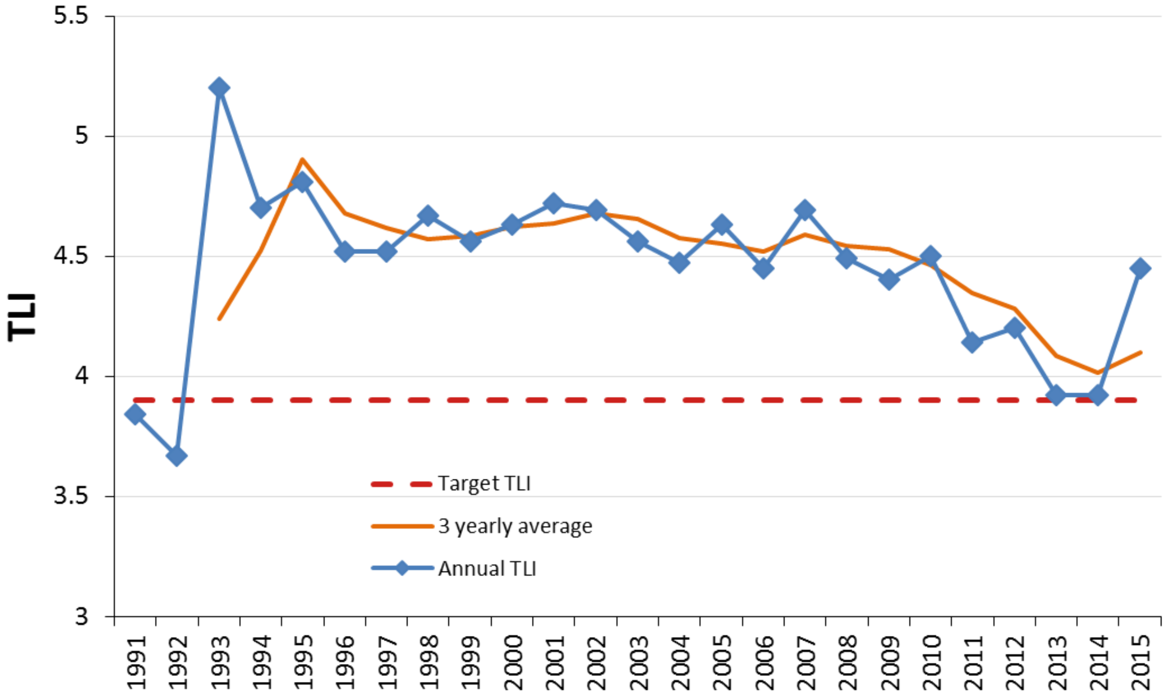

Figure 1. Historical Trophic Level Index (TLI) of Lake Rotoehu from 1991 to 2015 (BoPRC data).

Figure 2. Map of Lake Rotoehu showing sampling locations. Destratification devices are represented by white squares. The red cross marks the location the fixed sensor monitoring buoy. Note that the Site 1 and Site 3 locations are approximately 30 m from the mixing devices.

Physico-chemical analysis

Water samples were collected at all three sites, from October 2014 to May 2015, for analysis of chlorophyll *a*, total nutrients (nitrogen and phosphorus), dissolved nutrients (dissolved reactive phosphate, ammonium, nitrate) from the epilimnion (0.5 m) and hypolimnion (9 m). Water nutrient concentrations were analysed using a Flow Injection analyser 8500 Series II. Phosphate was analysed using LACHAT QuickChem method 31-115-01-1-H; ammonium was analysed using LACHAT QuickChem method 31-107-06-1-B and LACHAT QuickChem Method 31-107-04-1-A was used to analyse nitrate/nitrite. Water column profiles were taken at sites 1, 2 and 3 between October 2014 and May 2015, using a conductivity-temperature-depth (CTD) profiler (SBE 19 plus SEACAT Profiler, Seabird Electronics Inc.). Water clarity was measured using a Secchi disc comprising a 20 cm diameter disc with black and white markings which is lowered into the water until it is no longer visible.

Additional data were sourced from the fixed sensor water quality monitoring buoy located in the central area of Lake Rotoehu (Figure 2). The buoy collects quarter-hourly data for meteorological (air temperature, wind speed and direction, humidity, barometric pressure, and rainfall) and water quality (chlorophyll fluorescence, dissolved oxygen, and water temperature) variables. Water column temperature measurements from the buoy were also used to calculate a daily average 'Schmidt Stability Index' using the software 'Lake Analyzer' (Read et al. 2011). This index describes the energy required to mix surface and bottom waters, i.e., the strength of thermal stratification.

Plankton community sampling

Phytoplankton and zooplankton samples were also collected at depths of 0.5 m and 9.0 m from all three sites from October 2014 to May 2015 using a 10 L Schindler-Patalas trap. Unfiltered phytoplankton samples (400 mL) were preserved with Lugol's iodine and zooplankton were collected by filtering 10 L of lake water through a 40 µm net and subsequent preservation in 70% ethanol.

Phytoplankton analyses were carried out using Utermöhl settling chambers (Utermöhl, 1958) and an inverted microscope (Olympus, Ix71, Japan). Phytoplankton was identified to genus level, and abundance was determined for each genus. Enumeration to find densities of phytoplankton (cells ml^{-1}) used methods adapted from Hötzel & Croome (1999) and US Environmental Protection Agency (2007). Due to high cyanobacteria concentrations in the lake samples for this period, a sub-sample of 20 mL was extracted with a syringe and place under pressure, an action which allows the extraction pressure to burst gas vacuoles/aerotopes in buoyant species. A 10 mL subsample was settled in an Utermöhl chamber for 12-24 hrs and enumerated to genus level through microscopy. Phytoplankton were counted at 4009 or 2009 magnification in a single transect, including at least 100 planktonic units (cells, colonies and filaments) of the dominant species. Zooplankton were identified and enumerated by passing samples through a 40 µm mesh to remove ethanol and to attain a final known volume, dependent on the density of algae. Samples were enumerated in 5 mL aliquots in a gridded Perspex tray until counts of at least 300 cells were obtained, or the entire sample was enumerated. Species were identified using standard guides (e.g. Chapman & Lewis, 1976; Shiel 1995).

Analysis of plankton community composition was also carried out using PRIMER v.6, with the Bray-Curtis similarity metric employed. Similarity ranged from 0 (no similarity) to 100 (perfect similarity). Data were grouped by sampling site and depth and paired by sampling date. In order to account for zero inflation in the dataset, the analysis was performed using log transformed (log (x + 1)) abundance counts (Clarke and Gorley 2006).

Results

Monthly sampling was conducted from October 2014 to May 2015; mixing device operation was initiated on 5 November 2014 and was halted at the end of May 2015. During this period, air compressor faults occurred on seven occasions causing mixing to be halted for a total of 309 hours, a list of monitoring dates, device operation changes and fault periods is provided in Table 1. One notable occurrence was that following a fault, mixing device operation was only restored approximately 1 hour before monitoring was conducted on 13 March 2015.

Table 1. List of monitoring dates, mixing device stoppages and changes in device operation.

Date	Activity
21 October 2015	Lake monitoring
5 November 2014	Southern device begins operation (Site 1)
13 November 2014	Lake monitoring
19 December 2014	Lake monitoring
14 January 2015	Lake monitoring
22 January 2015	Switch to Northern device operation (Site 3)
29 January 2015	Compressor fault
2 February 2015	Compressor fault, shutdown for 102 hours
11 February 2015	Lake monitoring
13 March 2015	Compressor fault, shutdown for 12 hours
13 March 2015	Lake monitoring
29 March 2015	Switch to Southern device operation (Site 1)
31 March 2015	Compressor fault, shutdown for 117 hours
14 April 2015	Lake monitoring
28 April 2015	Compressor fault, shutdown for 22 hours
1 May 2015	Compressor fault, shutdown for 33 hours
15 May 2015	Lake monitoring
25 May 2015	Compressor fault, shutdown for 23 hours

Physicochemical results

CTD profiles

Vertical temperature profiles plotted from CTD casts showed little evidence for strong stratification on most monitoring dates (Figure 3), with a <1°C difference in temperature between epilimnion and hypolimnion on most occasions. Exceptions to this occurred on December 2014 when a 1.5°C difference was present between the epilimnion and hypolimnion and January 2015 when a >5°C difference was present. No data are presented for Site 2 October 2014 due to an instrumentation fault.

Evidence for artificial mixing of the epilimnion and hypolimnion was similarly limited, with reductions in thermal gradients at treatment sites only apparent in November 2014 and March 2015 (Figure 3). Unusually, there also appeared to be an increased thermal gradient at the treatment site compared with the control sites in December 2014. However, three distinct plumes of darker water moving away from the mixing devices were observed on a number of occasions. These surface currents had not been observed in previous years and were often visible up to 200 m from the mixing device. Temperature profiles from CTD casts conducted in these visible plumes are presented in Figure 4.

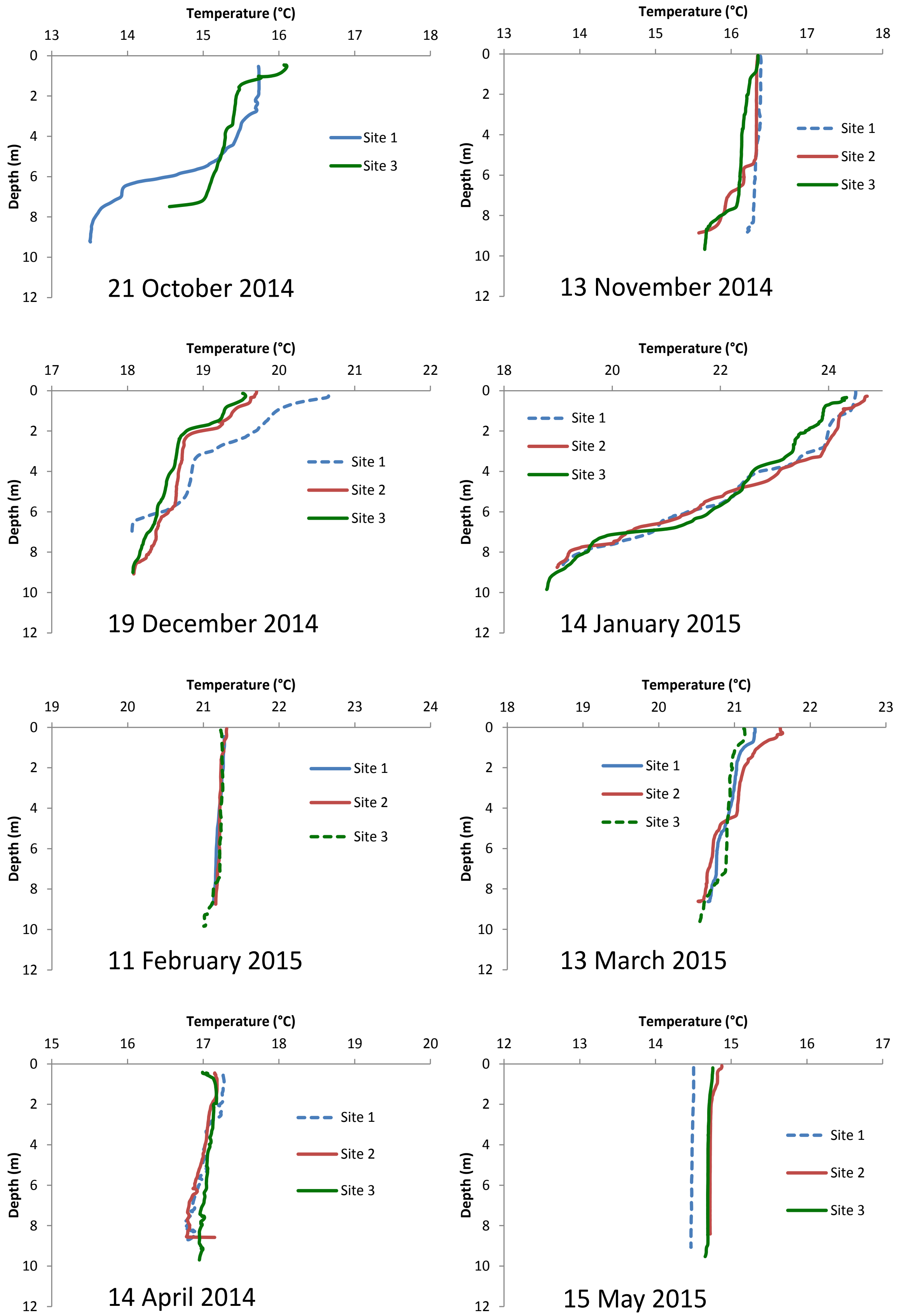

Figure 3. Vertical temperature profiles from three sites in Lake Rotoehu. Dashed lines indicate the operational artificial mixing device. Note: no Site 2 data available for October 2014.

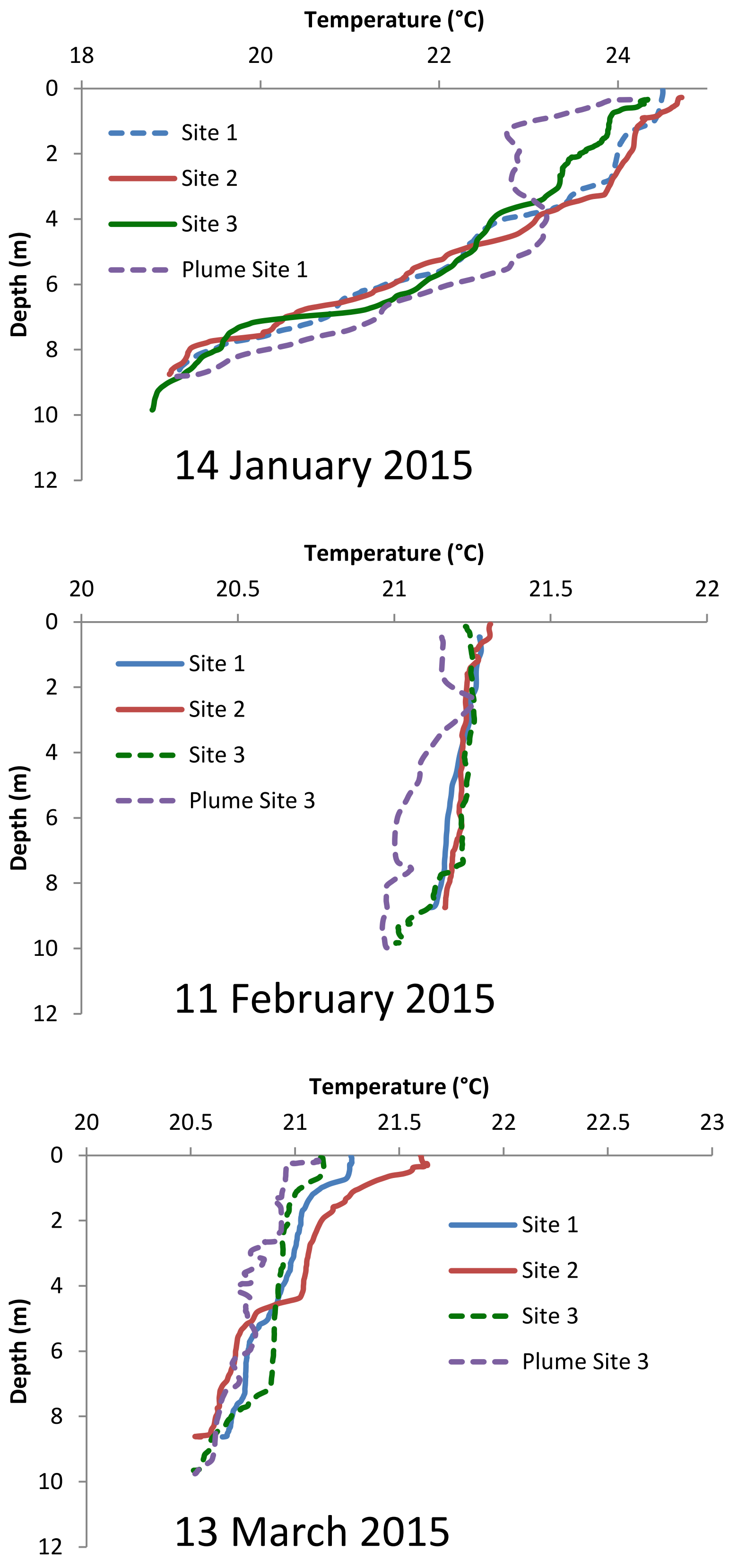

Figure 4. Vertical temperature profiles from CTD casts demonstrating the mixing effect in the plume leaving the artificial mixing device. Dashed lines indicate the operational artificial mixing device. The dashed purple line indicates the temperature profile within the water discharged from the mixing device.

Schmidt stability

Schmidt stability index was calculated from the temperature profile data provided by the monitoring buoy deployed in Lake Rotoehu. Summer stratification was generally stronger and more sustained in 2014-15 summer period compared to the three previous summer periods (Figure 5).

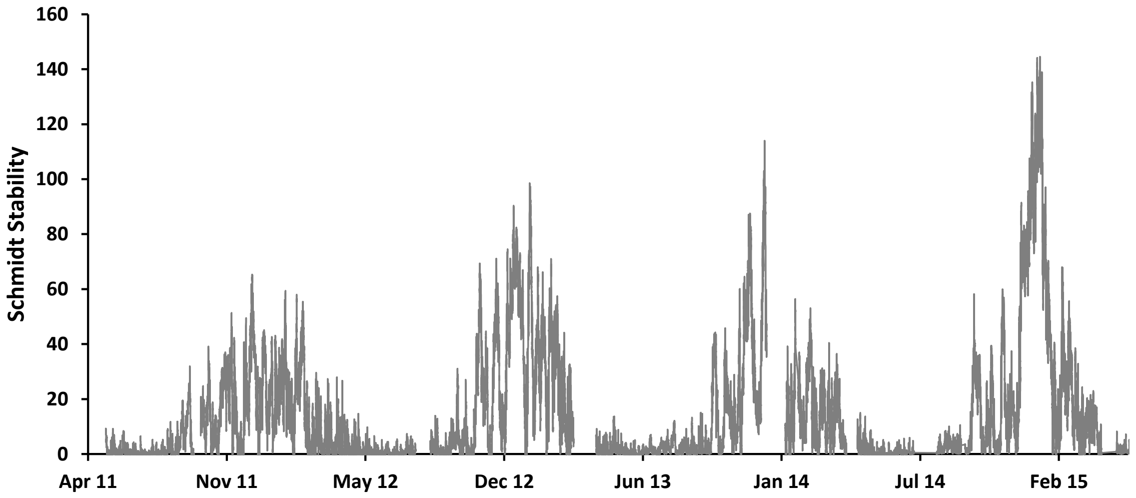

Figure 5. Schmidt stability index of Lake Rotoehu, based on records from the monitoring buoy for the period April 2011 to May 2015. Sections of data have been corrected to account for sensor drift. A higher Schmidt stability value indicates stronger stratification of the water column.

During the 2014–15 monitoring period, the primary episode of lake stratification occurred over a 6 week period from 17 December 2014 to 5 February 2015 (Figure 6). Lake stability declined in early-February and the lake mixed in mid-February before stability again increased.

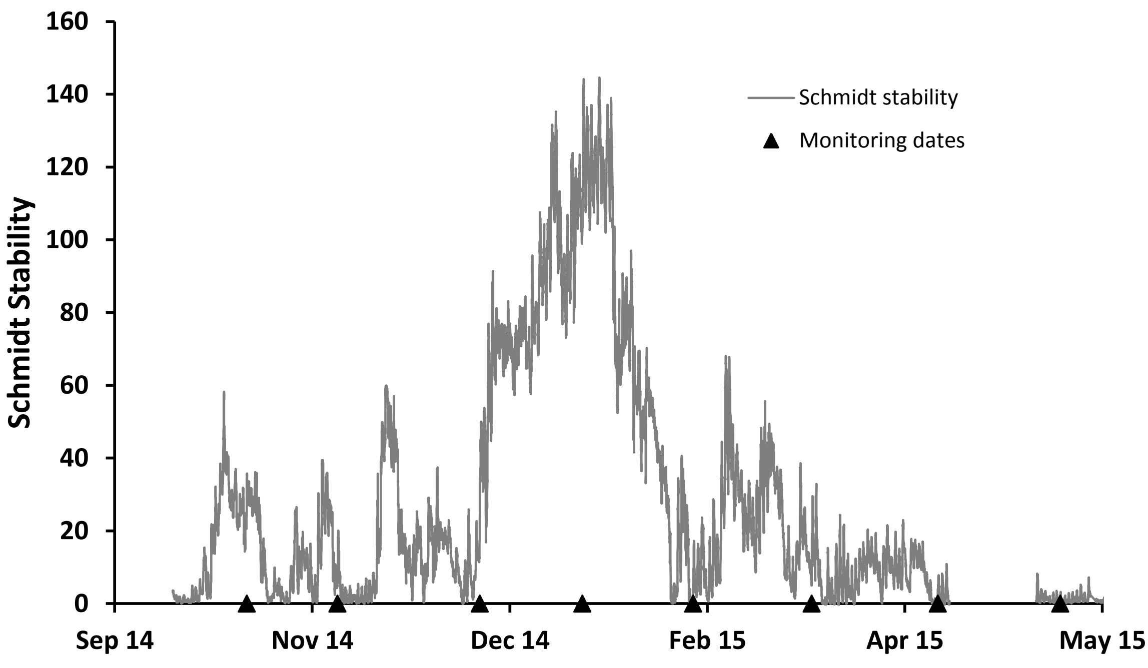

Figure 6. Schmidt stability index of Lake Rotoehu for current monitoring period 1 October 2014 to 29 May 2015. Note no data available mid-April –mid-May 2015.

Nutrients

Epilimnion and hypolimnion total nitrogen (TN) and total phosphorus (TP) concentrations for the 2014-15 monitoring period were similar to those previously observed at Site 2 (Figure 7). There were no consistent differences in nutrient concentrations between control sites and operational mixing device locations (Figure 8). Total phosphorus concentrations increased in both the epilimnion and hypolimnion over time, but this was not reflected in dissolved phosphate concentrations. Total nitrogen concentrations were highly variable in the epilimnion of treatment locations but were more consistent in the hypolimnion. Ammonium (NH_4) concentrations were comparatively consistent in the epilimnion, peaking in April 2015 and hypolimnion concentrations displayed little concordance between treatment and control sites (Figure 8).

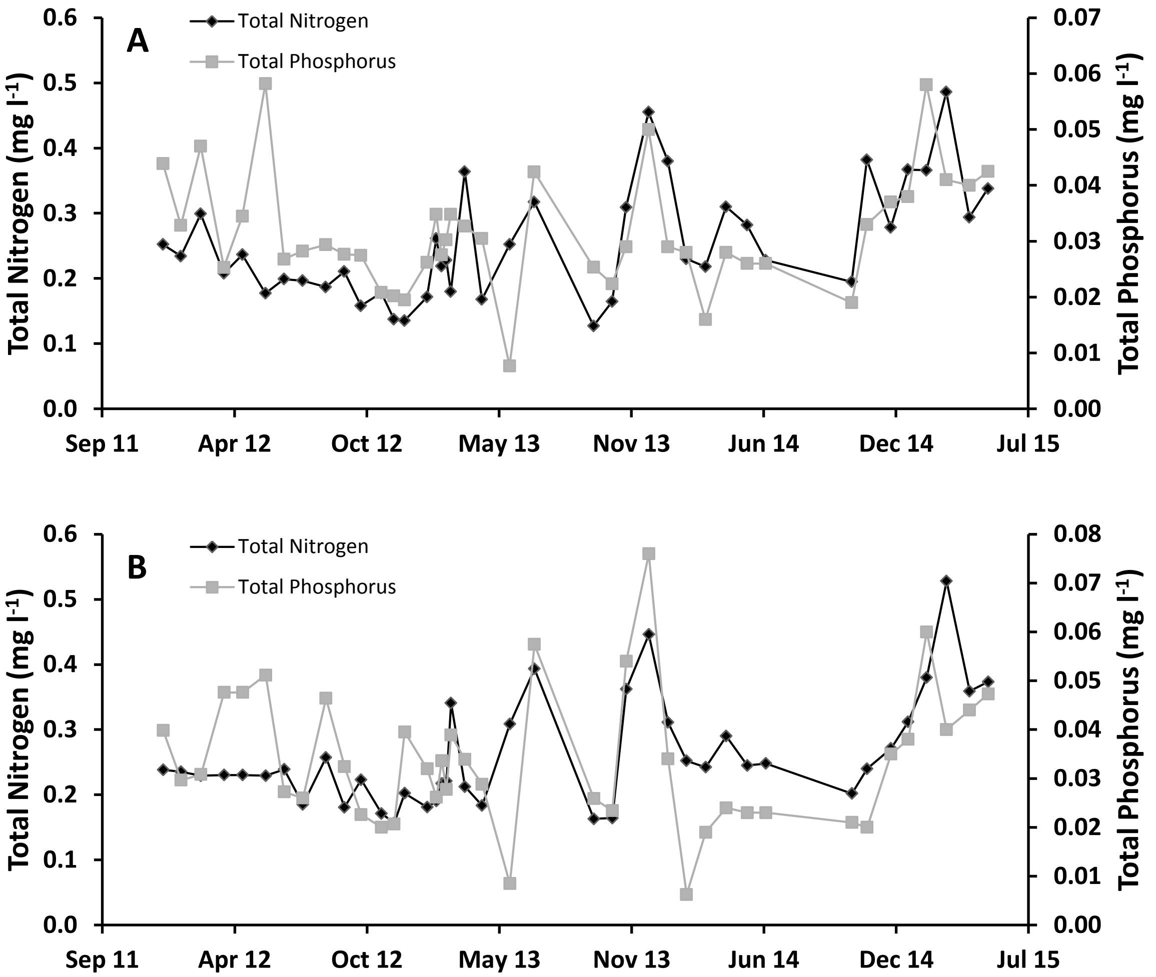

Figure 7. Lake Rotoehu A) epilimnion (0.5 m) and B) hypolimnion (9 m) total nitrogen and total phosphorus concentrations at Site 2 (control) from December 2011 to May 2015.

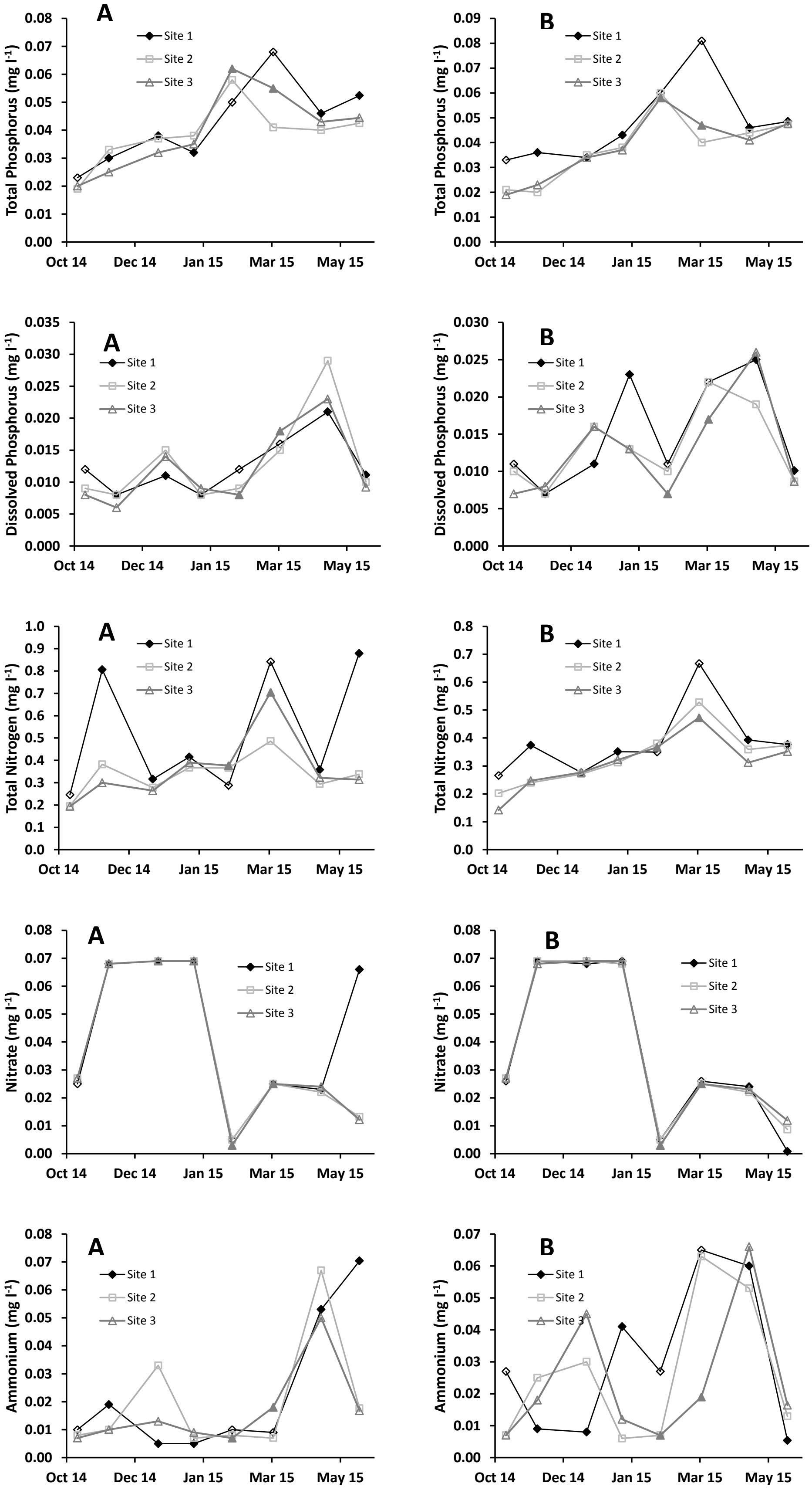

Figure 8. Lake Rotoehu nutrient concentrations from A) epilimnion and B) hypolimnion. Sites 1 and 3 are locations of mixings devices and solid symbols indicate periods when mixing devices were in operation. Site 2 was the control site and therefore only has open symbols.

Secchi disk

Mean Secchi depth was lower during the 2014-15 monitoring period (2.34 ± 0.73 m 95% CI) but was not significantly different (t-test: P >0.05) compared with the previous two years (mean 3.06 ± 0.25 m 95% CI). Secchi disc measurements for February and March 2015 were 1.2 m and 1.4 m respectively, the lowest measurements recorded since monitoring of Lake Rotoehu began in December 2011 (Figure 9).

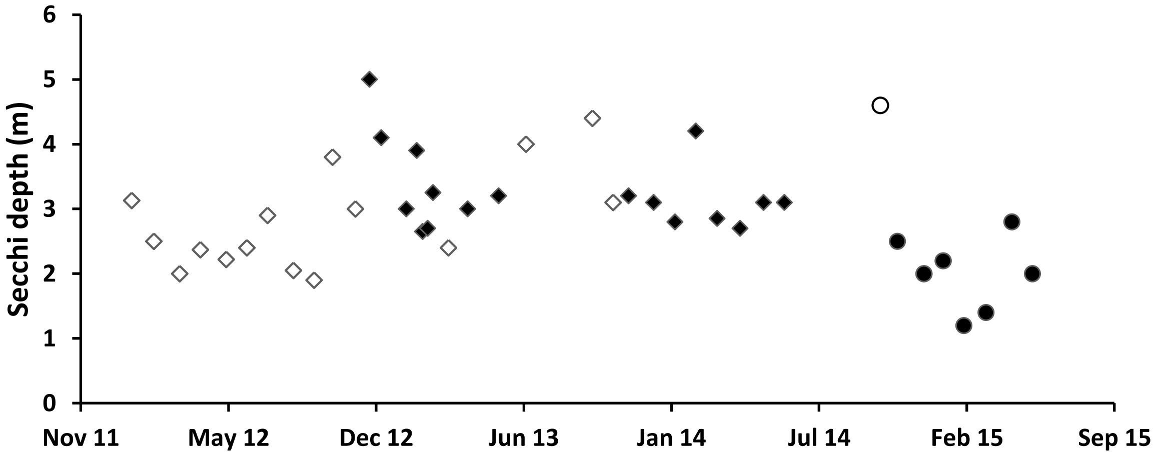

Figure 9. Lake Rotoehu Secchi disc depths, open symbols indicate periods when artificial mixing devices were not operating. Circles indicate measurements from current (2014-15) monitoring period.

Chlorophyll a

Epilimnion and hypolimnion chlorophyll *a* concentrations at the control site were generally similar to those observed in the previous three years (Figure 10). However, notably high concentrations were observed in the Site 2 epilimnion in March 2015 (45.8 µg L^{-1}) and the hypolimnion in May 2015 (42.7 µg L^{-1}). These concentrations coincided with the reduced water clarity measurements presented in Figure 9.

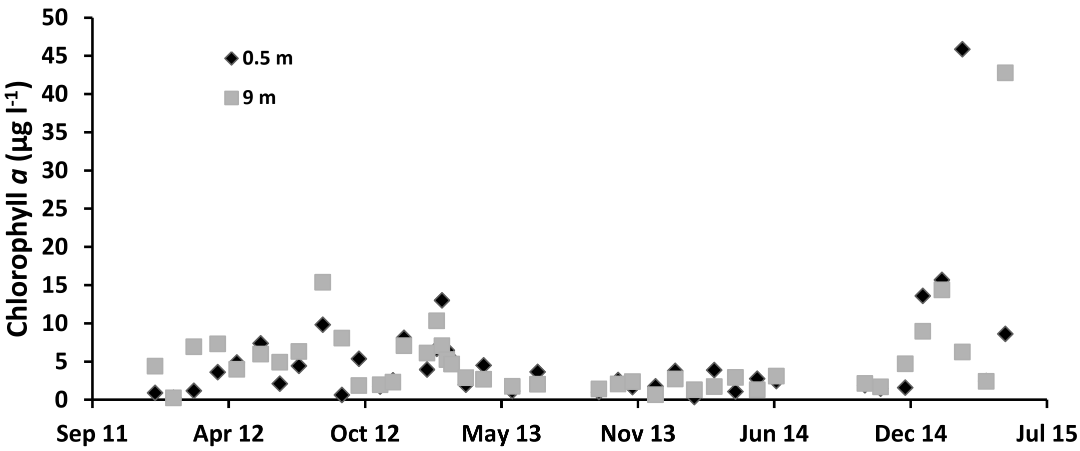

Figure 10. Lake Rotoehu epilimnion (0.5 m) and hypolimnion (9 m) chlorophyll *a* concentrations at the control site (Site 2) from December 2011 to May 2015.

Increased chlorophyll *a* concentrations were also observed during the January–May 2015 period at Site 1 and Site 3 (Figure 11). The size of the differences in chlorophyll *a* concentrations between the epilimnion and the hypolimnion was highly variable, with the largest differences occurring at Site 2. For example, chlorophyll *a* concentration differed by 39.6 8 µg L^{-1} between the epilimnion and hypolimnion at Site 2 on 13 March 2015. In contrast, the difference was 28.4 µg L^{-1} at Site 1 (mixing device not operating), and 8.4 µg L^{-1} at Site 3 which was adjacent to the operating mixing device. Interestingly, chlorophyll *a* concentrations in both the epilimnion and hypolimnion were reduced during the April 2015 sampling at all three monitoring sites before increasing again in May 2015 (Figure 11). Associated changes in the phytoplankton community assemblages which are presented below.

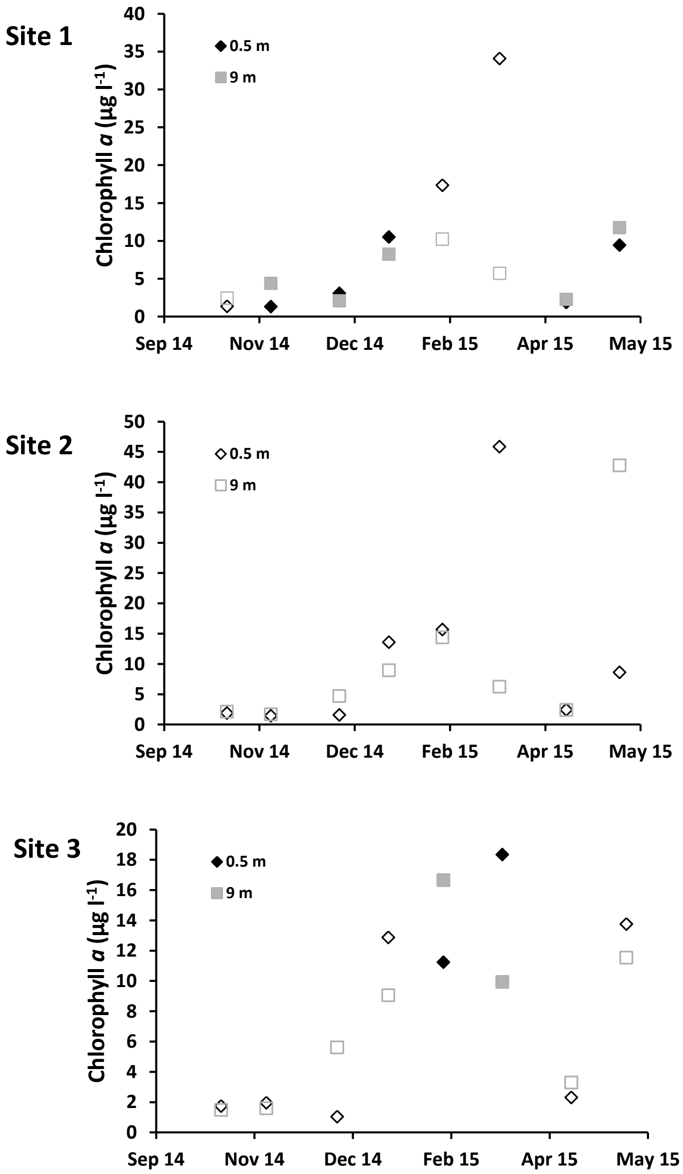

Figure 11. Chlorophyll *a* concentrations from the epilimnion (0.5 m) and hypolimnion (9 m) of Lake Rotoehu for the period October 2015 to May 2015. Solid symbols indicate periods when mixing devices were operating at the monitoring sites (Site 1 and 3). Site 2 was the control site and therefore only has open symbols.

Plankton community

Zooplankton

Changes in zooplankton taxonomic group abundance in the Site 2 epilimnion (0.5 m) and hypolimnion (9 m) for the period December 2011 to May 2015 are presented in (Figure 12). Rotifer, cladoceran and copepod abundances in the epilimnion were similar to those observed in the previous 3 years. In comparison, cladoceran abundance in the hypolimnion was unusually elevated during March 2015 relative to previous monitoring. Mean cladoceran abundance was significantly (t-test: P< 0.05) greater in the hypolimnion (21.9 individuals L^{-1}) compared to the epilimnion (10.6 individuals L^{-1}) for December 2011 to May 2015 period. This feature was shared by the copepod taxonomic group with abundance significantly (t-test: P< 0.05) greater in the hypolimnion (43.5 individuals L^{-1}) compared to the epilimnion (18.9 individuals L^{-1}). Mean rotifer abundance was higher in the epilimnion (72.3 9 individuals L^{-1}) compared to the hypolimnion (25.8 individuals L^{-1}), however, there was no significant difference (t-test: P> 0.5).

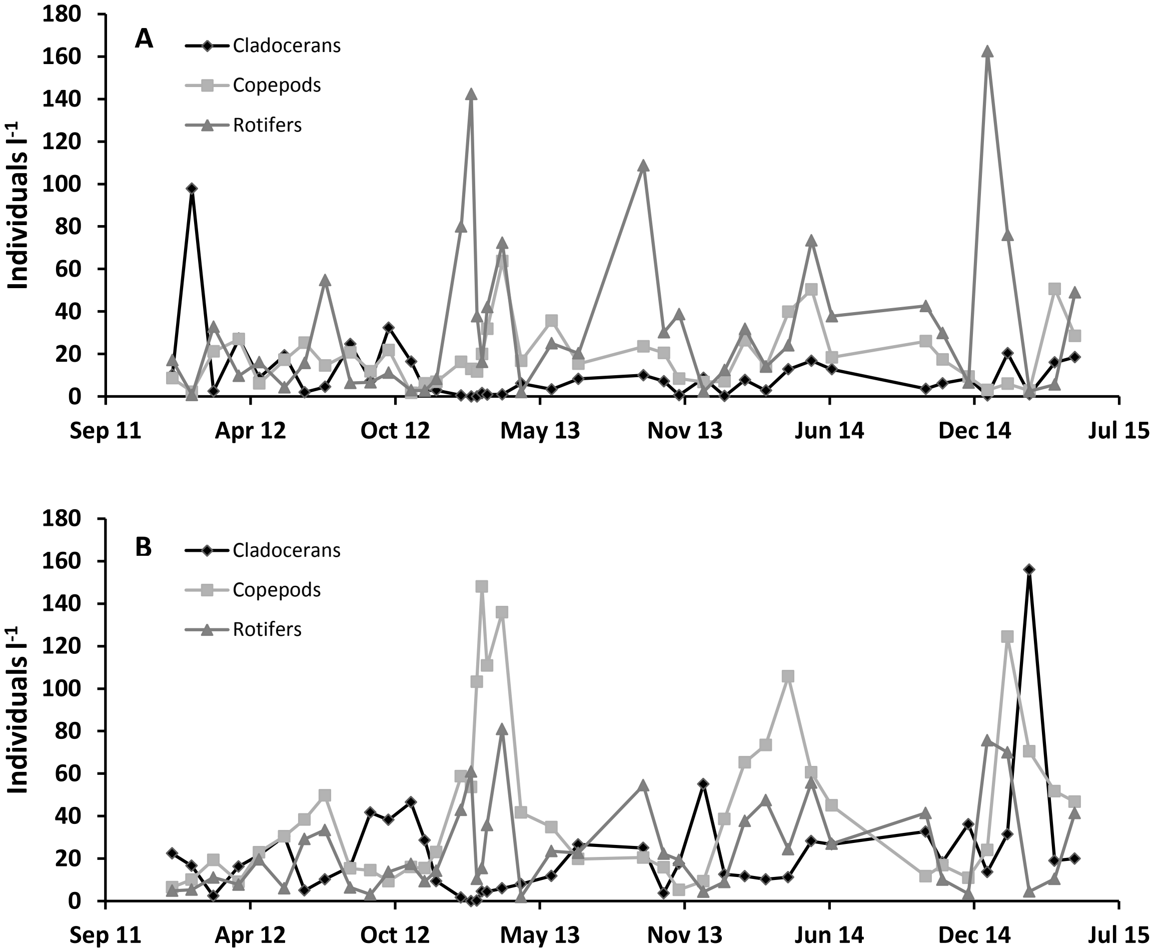

Figure 12. Zooplankton taxonomic group abundance in the Site 2 A) epilimnion (0.5 m) and B) hypolimnion (9 m) at the control site (Site 2) for the period December 2011 to May 2015.

The effect of mixing device operation on zooplankton taxonomic abundance was examined by calculating the difference in abundance between the epilimnion and hypolimnion at control and treatment sites and plotting this differential abundance over time. Cladoceran abundance was more homogenous between the epilimnion and hypolimnion at the treatment sites (adjacent to an operational mixing device) compared to control sites (no operational mixing device) (Figure 13).This was especially apparent in March 2015 when cladoceran abundance was greatest (Figure 12). Homogenisation of copepod abundance between the epilimnion and hypolimnion was not as pronounced as the cladoceran abundance under mixing device conditions. However, some evidence of homogenisation is apparent during peak copepod abundance in February and March 2015 (Figure 14).

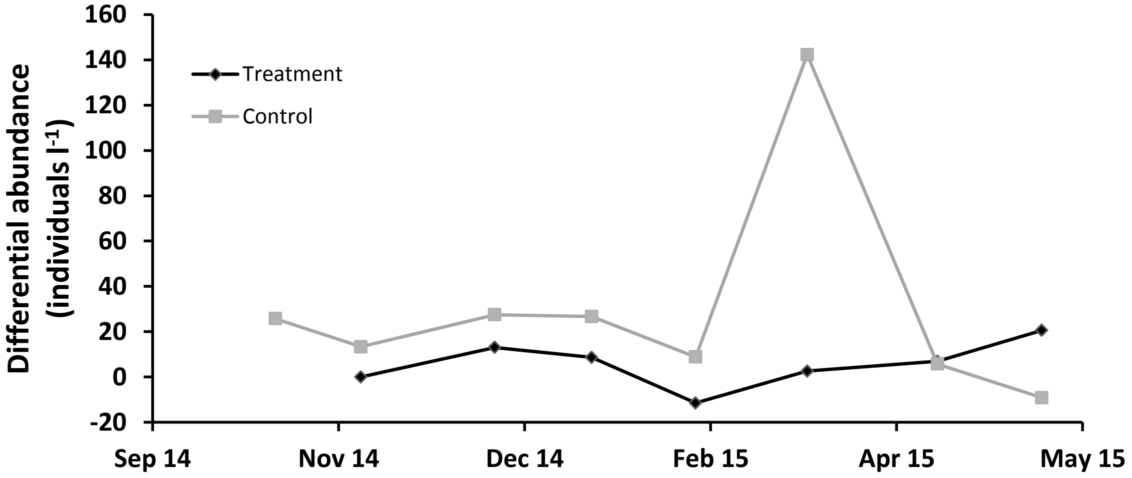

Figure 13. Lake Rotoehu cladoceran differential abundance between epilimnion (0.5 m) and hypolimnion (9 m) at treatment sites (adjacent to an operational mixing device) compared to the mean control site (no operational mixing device) abundance for the period October 2014 to May 2015.

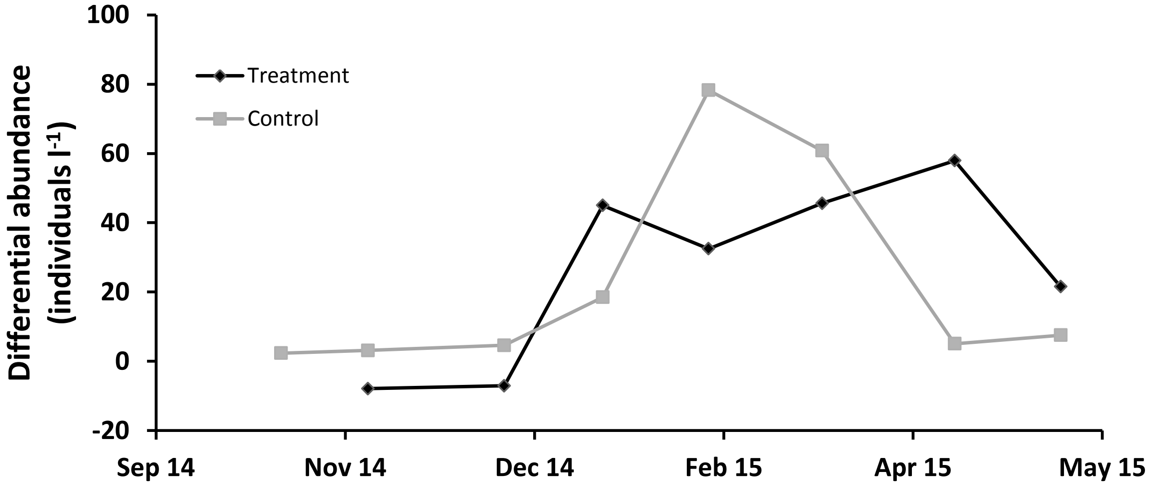

Figure 14. Lake Rotoehu copepod differential abundance between epilimnion (0.5 m) and hypolimnion (9 m) at treatment sites (adjacent to an operational mixing device) compared to the mean control site (no operational mixing device) abundance for the period October 2014 to May 2015.

In comparison to the cladoceran and copepod groups, rotifers demonstrated little evidence for depth preference, as reflected in the differential abundance plots (Figure 15).

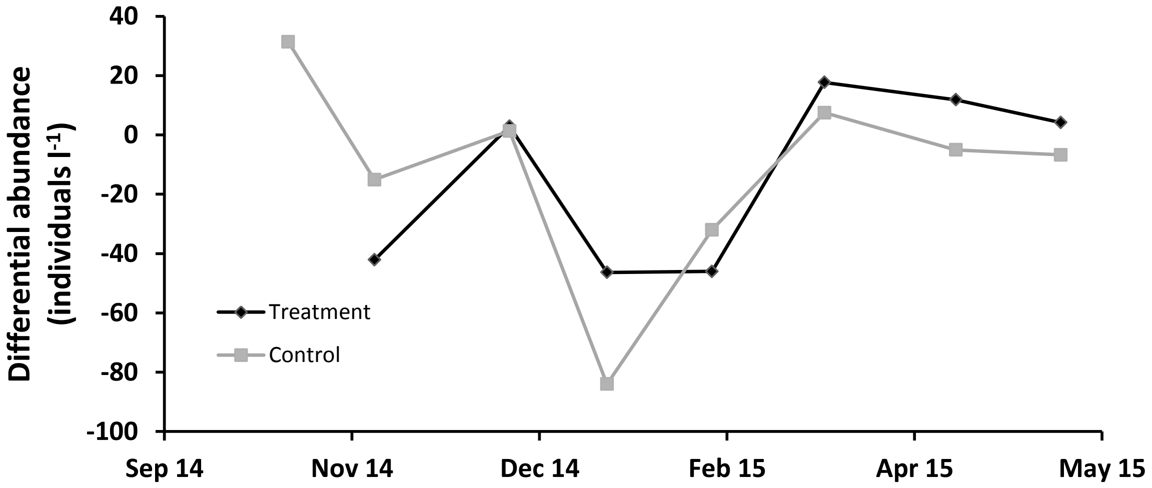

Figure 15. Lake Rotoehu rotifer differential abundance between epilimnion and hypolimnion at treatment sites (adjacent to an operational mixing device) compared to the mean control site (no operational mixing device) abundance for the period October 2014 to May 2015.

Phytoplankton

A significant summer algal bloom was observed in the Lake Rotoehu epilimnion in March 2015. Total algal abundance in the control site (Site 2) epilimnion was 91009 cells ml^{-1}, approximately five times greater than the next highest recorded abundance (July 2012 hypolimnion) and 13 times the mean (December 2011–May 2015) epilimnion abundance (Figure 16). The bloom was comprised almost entirely (97.9%) of cyanobacterial species, of which *Microcystis* sp. (65%) and *Aphanocapsa* sp. (32%) were the dominant species. This peak in abundance occurs concomitantly with the peak in chlorophyll *a* for the same period (Figure 10, Figure 11). Outside of this period, phytoplankton abundance was similar to previously observed levels, although somewhat more variable. Changes in phytoplankton taxonomic group abundance are presented in Appendix 1.

As with zooplankton abundance, the effect of mixing device operation on phytoplankton taxonomic abundance was examined by calculating the difference in abundance between the epilimnion and hypolimnion at control and treatment sites and plotting this differential abundance over time. The Euglenophyta, Cryptophyta, Chrysophyta and Dinoflagellata groups were excluded from this analysis due to low abundance (<50 cells ml^{-1}). Differences in abundance are presented for the taxonomic groups Chlorophyta (green algae) (Figure 17), Bacillariophyta (diatoms) (Figure 18) and Cyanobacteria (blue-green algae) (Figure 19). Homogenisation of the phytoplankton groups was generally evident at the treatment sites, during the latter part of the summer and early autumn. However, following periods of known lake mixing (Figure 6) in February and April 2015, homogenisation at the control sites was also evident.

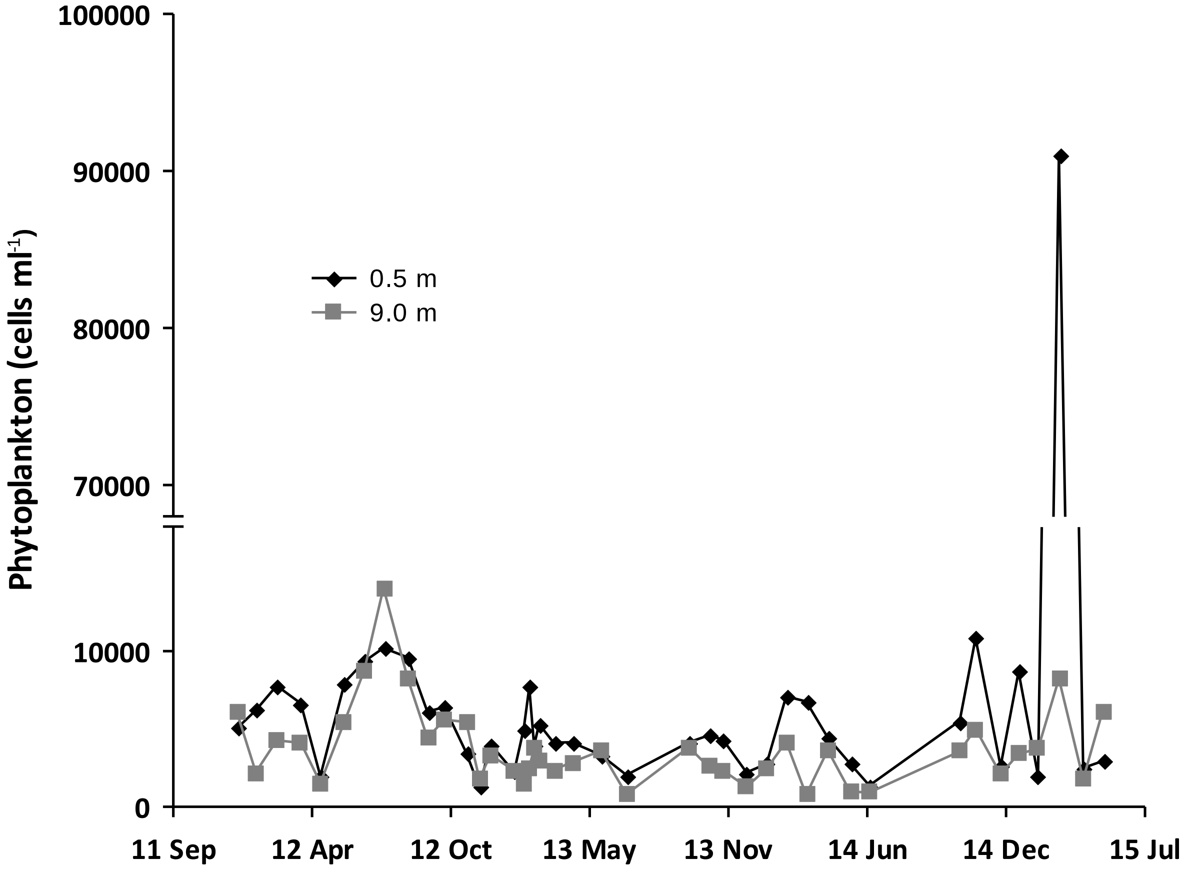

Figure 16. Lake Rotoehu total phytoplankton abundance at the control site (Site 2) epilimnion (0.5 m) and hypolimnion (9.0 m) for the period December 2011 to May 2015.

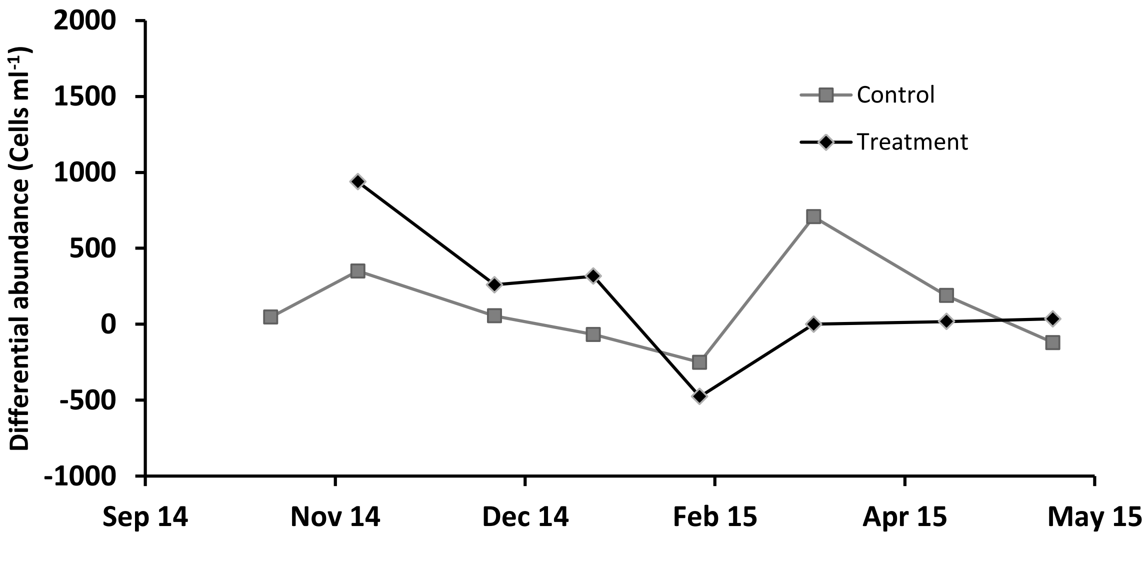

Figure 17. Lake Rotoehu differential abundance of chlorophytes (green algae) between the epilimnion (0.5 m) and hypolimnion (9.0 m) at the treatment site (adjacent to an operational mixing device) and control sites (no operational mixing device) for the period October 2014 to May 2015.

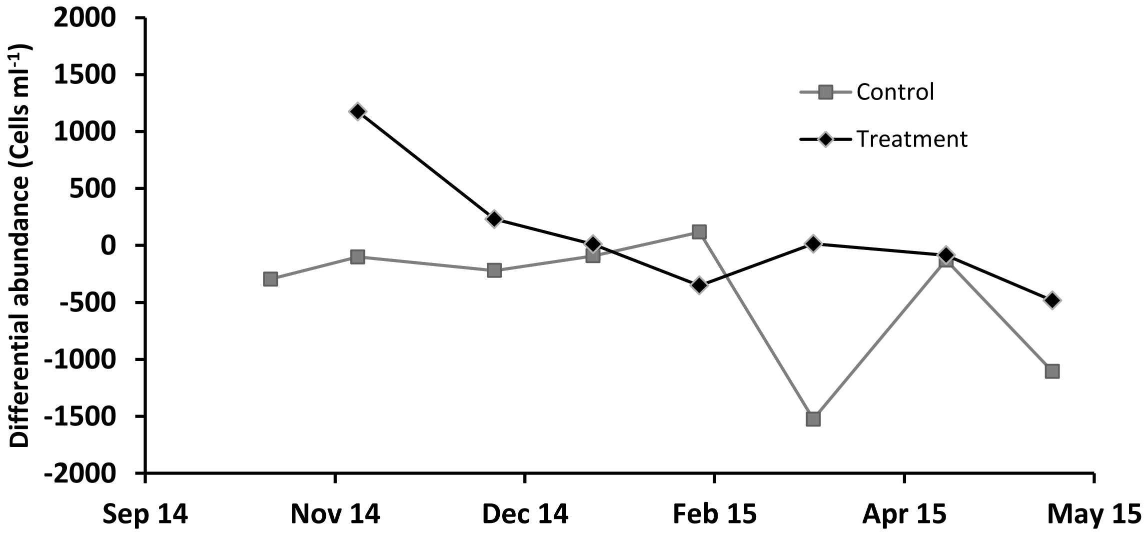

Figure 18. Lake Rotoehu differential abundance of Bacillariophyta (diatoms) between the epilimnion (0.5 m) and hypolimnion (9.0 m) at the treatment site (adjacent to an operational mixing device) and control sites (no operational mixing device) for the period October 2014 to May 2015.

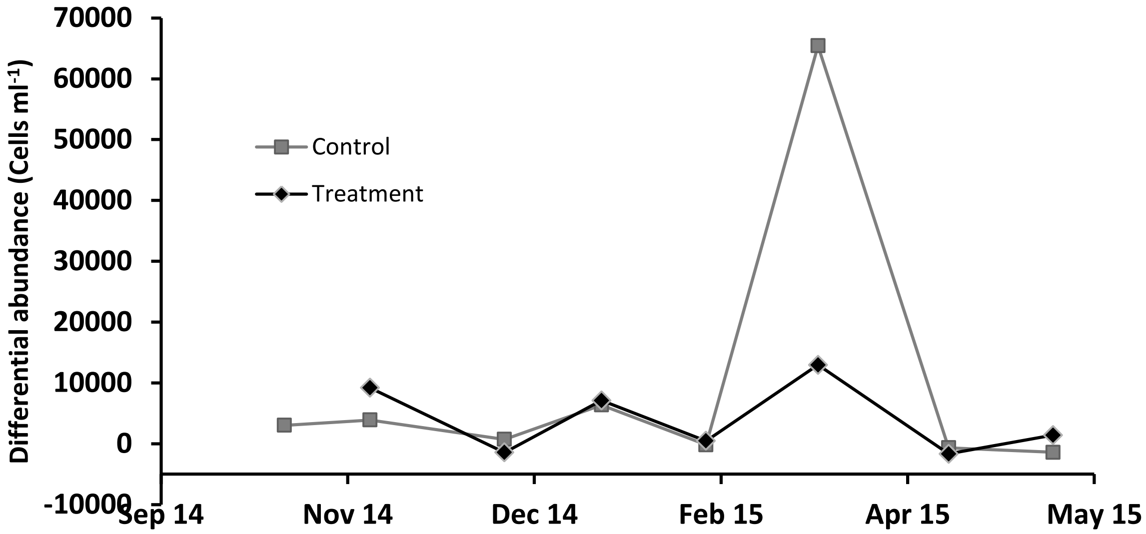

Figure 19. Lake Rotoehu differential abundance of cyanobacteria (blue-green algae) between the epilimnion (0.5 m) and hypolimnion (9.0 m) at the treatment site (adjacent to an operational mixing device) and control sites (no operational mixing device) for the period October 2014 to May 2015.

Community composition

A Bray-Curtis similarity index was used to compare the degree of similarity in the plankton (i.e., zooplankton and phytoplankton) community between treatment and control sites and the epilimnion (0.5 m) and hypolimnion (9.0 m) depths (Figure 20). Plankton communities generally displayed a high level of similarity between treatment and control sites; mean Bray-Curtis similarity between treatment and control sites was 74.5 in the epilimnion and 77.0 in the hypolimnion. Community similarity did decrease to ~65.0 between treatment and control sites in both the epilimnion and hypolimnion in April 2015 before increasing again in May 2015 (Figure 18A). Community similarity was significantly higher (t-test; $P < 0.05$) between epilimnion and hypolimnion of the control sites (mean 7.3) compared to the treatment sites (mean 65.7). Changes in community composition at the zooplankton and phytoplankton level were similar to those observed at the combined community level; the results are presented in Appendix 2.

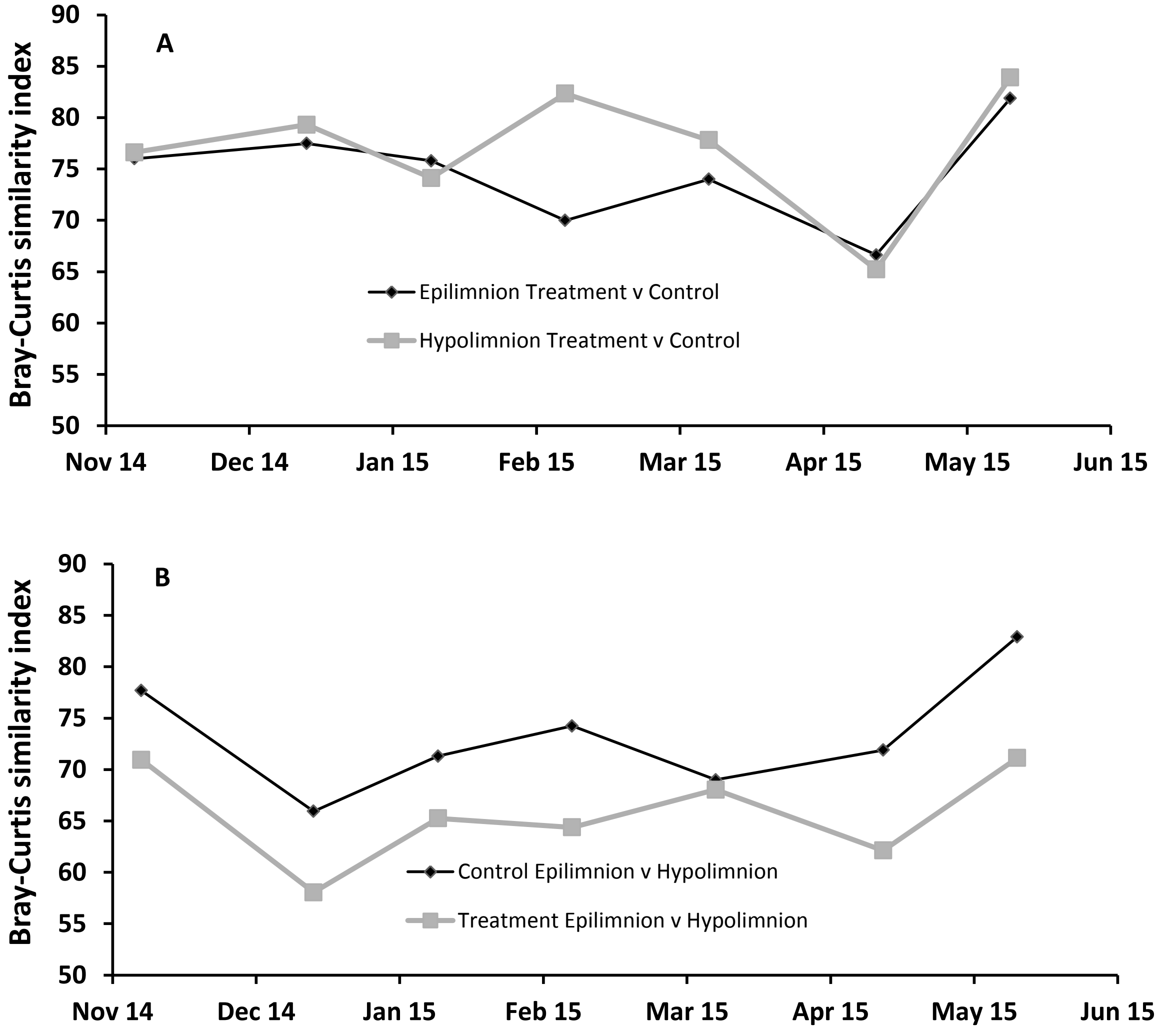

Figure 20. Bray-Curtis similarity index comparisons for A) epilimnion and hypolimnion depths between treatment and control sites, and B) control and treatment sites between epilimnion and hypolimnion depths.

Discussion

Monthly monitoring of Lake Rotoehu was carried out from October 2014 to May 2015 as part of an on-going programme to assess the effect of two artificial mixing devices installed in Lake Rotoehu during the spring of 2012. The mixing devices were designed to draw cooler water from the hypolimnion mixing it with warmer, less dense water in the epilimnion thereby breaking down thermal stratification in the lake. A number of physicochemical and biological variables have been monitored since December 2011 including, temperature, dissolved oxygen, nutrient concentrations, water clarity, chlorophyll *a* concentrations and plankton diversity and abundance. Monitoring results for the 2011–2014 period indicate that the mixing devices produced a detectable but limited localised effect. However, there was no discernible effect on the lake's ecology (McBride et al. 2015).

Following these findings, changes in mixing device operation were undertaken for the 2014–15 season. Compressed air output was set to maximum (mean flow 3.16 m^3 s^{-1}), and compressor output was directed through a single mixing device to maximise the mixing effect. The monitoring programme was adjusted from two to three sites to accommodate switching of air flow between the southern and northern devices. Monitoring objectives were to determine if increased air compressor output would increase the magnitude of effects previously observed and if a single device could influence lake ecology.

Schmidt stability (Figure 6) calculated from fixed monitoring buoy data indicated an extensive period of lake stratification during December 2014 – January 2015, although this was not as evident from the temperature profiles of the CTD casts conducted at this time. The December – January interval was the strongest and most extended period of stratification observed in four summers of monitoring and was followed by elevations in lake phosphorus and nitrogen levels when lake mixing occured in February 2015 (Figure 7 and Figure 8). There appears to be no evidence of mixing device operation reducing lake thermal stability during this time. However, localised mixing effects were detected from temperature profiles taken directly in the discharge plumes (Figure 4), indicating that the current configuration of the mixing devices is effective in drawing colder water from the hypolimnion and discharging it into the epilimnion.

As would be expected, reduced Secchi depths and observed algal blooms were associated with increased chlorophyll *a* levels (Figure 11). Differences in chlorophyll *a* concentrations between the epilimnion and hypolimnion were generally greater at the control site compared to treatment sites, indicating homogenisation of the phytoplankton biomass due to the mixing of the operational device. Natural, whole lake, mixing events were also observed (i.e. February and April 2015) and on occasion significant spatial variations in chlorophyll *a* levels were also observed. For example, differences between epilimnion and hypolimnion chlorophyll *a* at Site 2 and Site 3 were markedly different in May 2015 despite no artificial mixing device operating at either of the sites. Closer examination found Site 2 chlorophyll *a* levels and phytoplankton abundances were higher in the hypolimnion compared to the epilimnion at that site. This may be due to Site 2 being slightly more sheltered from prevailing westerly winds, resulting in less physical mixing.

Comparison of nutrient concentrations (TP, TN, DRP, NO_3, and NH_4) between monitoring sites showed generally similar levels and trends, the exception being hypolimnion ammonium concentrations which were highly variable between sites (Figure 8). Nutrient concentrations were similar to those previously observed (McBride et al. 2015) indicating no significant prevention of nutrient release by artificial mixing.

The increased phytoplankton biomass observed during January–May 2015 may have been a precursor for an increase in zooplankton biomass (Figure 12). Of particular note was the large cladoceran biomass, which was dominated by *Bosmina meridionalis* inhabiting the hypolimnion of the control site. Interestingly, this species was not abundant in either the hypolimnion or epilimnion of the treatment site, possibly indicating active avoidance or increased predation at the treatment site. Common smelt (*Retropinna retropinna*) have been observed congregating around the active mixing devices (Butterworth pers. comm.) and cladocerans are a preferred prey species (Boubee and Ward 1997).

Localised effects of the artificial mixing devices were apparent in the distribution of cladoceran and copepod species between the epilimnion and hypolimnion (Figures 13 and Figure 14). Both taxonomic groups have previously been observed to reside in greater densities in the hypolimnion (McBride et al. 2015). As with previous monitoring, reduced differences in zooplankton abundance between the epilimnion and hypolimnion were observed at sites adjacent to active mixing devices in the 2014–15 monitoring period. This effect was particularly apparent for both the cladoceran and copepod groups. There was less evidence for homogenisation of rotifer abundance by the mixing devices (Figure 15), which may be attributed to the fact that rotifers do not exhibit the same vertical migration predator avoidance behaviour as copepods and cladocerans (Folt and Burns 1999).

In March 2015 there was the first major cyanobacterial bloom observed in Lake Rotoehu for a number of years. The bloom was likely facilitated by a release of nutrients in February 2015 following mixing after the prolonged period of thermal stratification during December 2014 and January 2015. Cyanobacteria can be highly buoyant through the production of gas vesicles (Reynolds and Walsby 1975). This ability can lead to the formation of surface scums and cyanobacteria dominated communities in lake surface layers (Wetzel 2001). It was expected that these features would be disrupted locally by the vertical mixing produced by the Rotoehu mixing devices. In comparison, freshwater diatom species are generally negatively–neutrally buoyant due to their siliceous frustules, resulting in greater abundance at or below the thermocline (Reynolds 1984).

Evidence for homogenisation of algal abundance between the epilimnion and hypolimnion was not apparent during the early-mid stages of monitoring (November-February), and this may be related to the comparatively low algal abundances at the time. However, homogenisation of diatom and cyanobacteria distributions was apparent at operational mixing device locations during March–May when peak abundances occurred. Chlorophytes (green algae) were the only other abundant taxonomic group observed in Lake Rotoehu. While this group is generally considered to exhibit neutral buoyancy (Reynolds 1984), there was some evidence of increased homogenisation of abundance by the mixing device during the later period of monitoring.

Analysis of the plankton communities using Bray-Curtis similarity indices revealed no large differences in community composition between treatment and control sites (Figure 20A). However, community similarity was consistently lower between the epilimnion and hypolimnion at the treatment sites compared to the control site (Figure 20B). This was unexpected as the mixing effect produced by the devices could be anticipated to homogenise the plankton community, increasing the community similarity between depths. Despite this, community similarity was comparable to that observed during previous monitoring from 2011 to 2014 (McBride et al. 2015).

Conclusions and Recommendations

Modifications to the operation of the Lake Rotoehu artificial mixing devices operated over the spring–autumn period of 2014–15 included increasing compressor output and directing all output through a single mixing device. These changes resulted in improved (although unquantified) mixing as evidenced by observations of increased water flow leaving the devices and increased localised mixing of hypolimnion and epilimnion layers. There was also evidence to suggest that mixing of the epilimnion and hypolimnion was homogenising plankton community abundance locally. It can be concluded that the mixing devices produce a moderate, if unquantified, localised mixing effect under the current operational parameters. However, there is no evidence that this effect is being translated to the entire lake. Lake Rotoehu underwent a prolonged period of thermal stratification during December 2014 – January 2015 followed by marked algal blooms during February – May 2015. Although a partial ameliorating effect on thermal stability of the lake by the mixing device operation cannot be ruled out, the magnitude of stratification and the resulting decline in water quality in 2015 were substantial compared to the previous four years of monitoring.

The design and current configuration of the artificial mixing devices has proven effective at drawing water from the hypolimnion into the epilimnion. However, the thermal structure of a lake the size of Rotoehu appears too large for a single mixing device to be effective at whole-lake scale. To have a beneficial impact on water quality, the number of operational devices would need to be increased. Alternatively, the devices could be transferred to a smaller lake where they would be more likely to achieve a desirable outcome. While there is still some uncertainty as to the exact local dynamics in the plankton communities generated by the mixing devices, further monitoring of the plankton communities is unlikely to produce informative data.

Determination of the extent of the area of localised mixing and quantification of the amount of hypolimnetic water passing through each device will provide improved model estimates of mixing device efficiency. This could be achieved by repeating the rhodamine tracer, ADCP measurements and quantification of mixing device discharge conducted in the first year of operation. Current observations of surface currents out to 200 m from the mixing devices suggest a 16 ha maximum area of influence around each device. Assuming the main basin occupies approximately 50% (400 ha) of the lake area, 25 mixing devices would be required to dissipate thermal stratification. This estimate is based on effects observed during the 2015 summer, a period of significant lake stratification, and fewer devices may have been estimated to have lake-wide effects when thermal stratification was not as pronounced.

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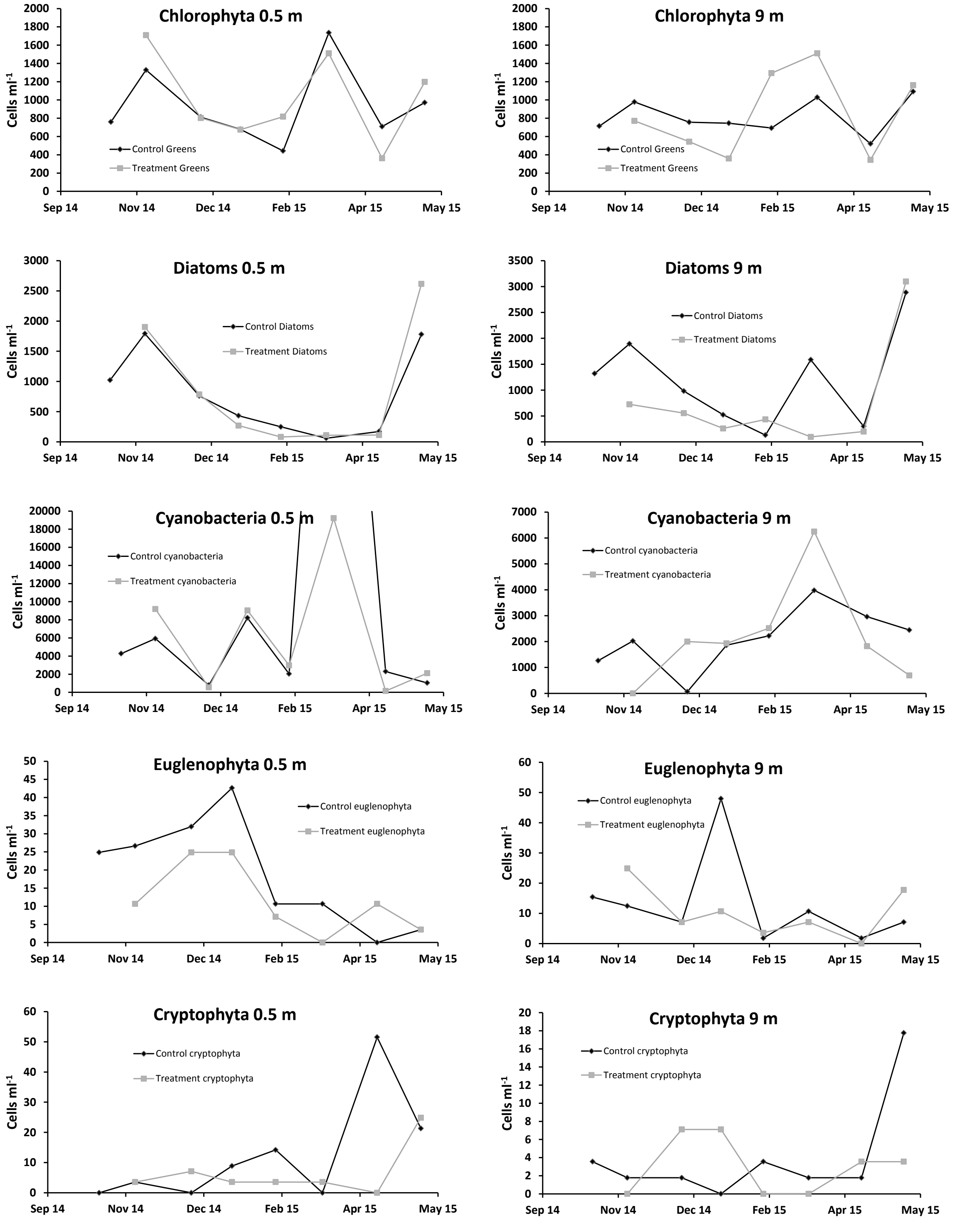
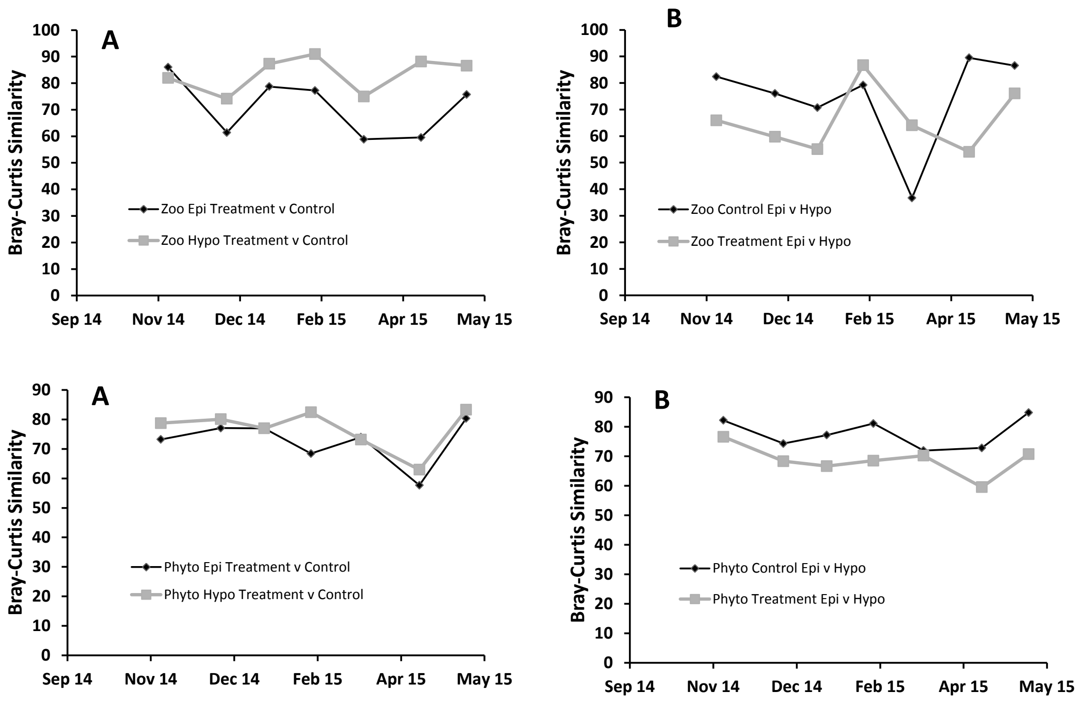
Appendix 1



Appendix 1. Changes in Lake Rotoehu phytoplankton taxonomic group abundance in the epilimnion (0.5 m) and hypolimnion (9.0 m) at the treatment site (adjacent to an operational mixing device) and control sites (no operational mixing device) for the period October 2014 to May 2015. Chrysophyta and Dinoflagellata groups have been omitted due to low numbers.

Appendix 2



Appendix 2. Bray-Curtis similarity index comparisons for zooplankton and phytoplankton communities at A) epilimnion and hypolimnion depths between treatment and control sites, and B) control and treatment sites between epilimnion and hypolimnion depths.