Assessing the effects of alum dosing of two inflows to Lake Rotorua against external nutrient load reductions: Model simulations for 2001-2012

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Executive Summary

This study considers the effects on Lake Rotorua water quality of alum dosing two inflows to the lake. Alum dosing commenced in the Utuhina Stream in 2006 and in the Puarenga Stream in 2009. Dosing rates were highly variable in each stream on a daily time scale. A one-month 'rolling average' showed that the combined dose to the streams was up to 400 kg Al per day. Dosing rates were consistently higher once the Puarenga inflow dosing commenced and particularly from 2011 to the end of our study period in 2012. Alum dosing was highly effective in adsorbing ('locking up') dissolved reactive phosphorus (DRP) in the stream inflows, particularly above certain threshold concentrations, as indicated by extremely low dissolved reactive phosphorus concentrations and low ratios of DRP to total phosphorus (DRP:TP) in the stream inflows below the dosing point. Concentrations of TP remained largely unchanged in the streams below the dosing point, suggesting that turbulence in the streams maintained the adsorbed DRP as suspended particulate phosphorus.

Our study included an analysis of the time series of discharge and nutrient concentrations in the lake inflows including the nine major stream inflows, combined minor stream inflows, and rainfall. No trend analysis was carried out for the stream inflow data because its primary purpose was to generate input data and verify output from the catchment model (ROTAN) used as input to the DYRESM-CAEDYM lake model. It was evident, however, that over the 12 years (2001-2012) nitrate concentrations were increasing in some inflows (e.g. Awahou) as expected from progressive enrichment of large groundwater aquifers due to historical changes in land use and agricultural intensification. It was unexpected, however, that some inflows (e.g. Awahou and Waiteti) showed a recent period (2010-2012) of elevated and highly variable TP concentrations, which may be related to erosion and loss of particulate phosphorus from high-intensity rainfall events over this period. The exception was Puarenga, which showed a clear decrease in DRP concentrations commencing around 2009, little change in TP concentrations, and a consistent reduction in total nitrogen (TN) and nitrate (NO_3-N) concentrations over the study period. We attribute at least some of these effects to changes in treatment processes at the Rotorua Wastewater Treatment Plant.

We examined measurements of nutrient concentrations in surface and bottom waters at a central site in Lake Rotorua from 2001 to 2013. Concentrations of TP and DRP began to decrease around 2007-8. This period also corresponded to reduced TN and chlorophyll *a* concentrations, while annual TLI decreased to the point where it reached the 'target' (prescribed in the Bay of Plenty Regional Council Land and Water Plan) of 4.2 in 2012. These in-lake improvements were achieved despite the changes in inflow concentrations mentioned above.

On the basis of DYRESM-CAEDYM model simulations it was surmised that alum dosing was impacting on lake concentrations beyond simply locking up DRP in the Utuhina and Puarenga Steam inflows. This conclusion was based on the fact that simulated trophic state of the lake remained substantially above the observed level using the previously calibrated and validated model (i.e. 2001-2007 prior to alum dosing) and applying it to the period of *intense* alum dosing (2009-2012) and including removal of the DRP locked up by alum dosing through the latter period. We therefore increased rates of sedimentation of organic matter and decreased rates of sediment phosphate release, both individually and together, in order to achieve a satisfactory match of trophic state (i.e. TLI and its water constituents of TN, TP and chlorophyll *a*) for the period 2009-2012. We also justified this approach on the basis that there would be increased rates of flocculation and sedimentation of organic matter in the lake as a result of alum dosing, and rates of oxygen consumption by bottom sediments appeared to have decreased based on high-frequency monitoring data for dissolved oxygen in bottom waters. Simulation of alum effects was not dynamic (i.e. alum concentrations were not explicitly simulated in the model) but provided a satisfactory simulation of the observed average TLI over the four-year period of particular interest. On consideration of hydraulic flushing rate, estimated sedimentation of the alum floc and the time scale for changes in bottom water oxygen consumption rates, we estimated that there may be persistent effects from alum dosing lasting perhaps 2-3 years.

Alum dosing in the stream inflows is now highly regulated to maintain three-month surface TP concentrations at 20 mg m^{-3}, i.e., around one-half of the very high levels observed in the lake in the mid-2000s. Concentrations now show much less seasonal variability than before alum dosing. Of considerable importance is whether alum dosing has brought about a transition in nutrient limitation status of phytoplankton in Lake Rotorua. Recent studies of nutrient limitation have commonly shown addition of both nitrogen + phosphorus to have the greatest growth-stimulation effect on phytoplankton (i.e. ‘co-limitation’). The most recent study (Abell et al. 2012) was undertaken in during a ‘trough’ in nitrate concentration compared with periods before and after the study. This may explain the observed dominance of N limitation, and we hypothesise that had the study been conducted during adjacent periods of much higher nitrate concentration then P limitation would have been dominant. High nitrate concentrations in 2011-12 suggest that demand on dissolved inorganic nitrogen by phytoplankton was lower, consistent with phosphorus concentrations being reduced to limiting levels whereby excess nitrate remains unutilised in the water column.

Much recent speculation has considered managing nutrient loads so that *either* N or P is controlled to limiting levels whilst the other is less stringently controlled. Even with alum dosing, Abell et al.’s (2014) study and field observations suggest that there are locations and periods in Lake Rotorua where either nutrient or both is limiting. The efficacy of controlling a single nutrient to limit primary production in freshwaters is not well supported by direct measurements (e.g. using bioassays), of which there are remarkably few (see Abell et al. 2010).

The possibility that recently observed improvements in Lake Rotorua water quality are a result of a regime shift towards more frequent P-limitation is an important consideration for the management of the lake. Specifically, the intensity and sustainability of alum dosing needs to be carefully weighed against the management of present and future loads of both nitrogen and phosphorus from catchment land use.

Acknowledgements

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1. Introduction

Lake Rotorua is a major asset to the city of Rotorua. It has great cultural significance to Māori, is an important trout fishery and provides recreation and tourism opportunities for residents and tourists. Water quality of Lake Rotorua declined between the 1960s and mid-2000s (e.g. Fish 1969; Rutherford et al. 1996; Burger et al. 2007a, 2008), co-incident with urbanisation and agricultural development in its catchment.

Increasing nitrate concentrations have been observed over recent decades in many of the major stream inflows to the lake (1968-2003; Rutherford et al. 2011). This trend has been attributed to gradual nitrate enrichment of groundwater aquifers draining agricultural land in the catchment, with changes in nitrate concentration reflecting a history of agricultural development and intensification.

Treated effluent from Rotorua city was discharged to Lake Rotorua until 1991, when land-based effluent polishing was commenced in an area of the Whakarewarewa forest in the Puarenga sub-catchment. Coupled with implementation of the Kaituna Catchment Control Scheme in the 1980s, which appeared to have largely arrested increases in particulate phosphorus loads from earlier pastoral conversions (Williamson et al., 1996), there were noticeable improvements in water clarity through the early 1990s (Rutherford et al. 1996). However, bottom sediments in Lake Rotorua have been strongly enriched in nitrogen (N) and phosphorus (P) as a result of many years of eutrophication (Trolle et al. 2008). Releases of these nutrients from sediments to the water column are greatly enhanced as levels of dissolved oxygen decline when the water column stratifies, which can occur for periods of up to one month during warm, calm weather (Vant 1987). On an annual basis, loads contributed from internal release (the 'internal load') are comparable to, or may be even greater (in gross terms) than those arising from the lake catchment (see Burger et al. 2008; Bay of Plenty Regional Council 2009).

The net result of these influences has been a decline in water quality over several decades, to an extent that threatens ecosystem services and the mauri (life force) of the lake. The trophic level of Rotorua increased dramatically in the early 2000s, and the decade was characterised by frequent blooms of cyanobacteria (blue-green algae) through summer-autumn as well as prolonged bottom-water anoxia (loss of dissolved oxygen) during periods of stratification.

In response to these issues, an 'Action Plan' was developed by Bay of Plenty Regional Council (BoPRC), Te Arawa Lakes Trust and the Rotorua District Council (Bay of Plenty Regional Council, 2009). Key components of this Action Plan are considerations of suitable land use and management practices aligned with reducing catchment nutrient loads, as well as artificial nutrient controls to stream inflows and/or lake bottom sediments. The objective of the Action Plan is to restore water quality in Lakes Rotoiti and Rotorua to where major problems and issues are infrequent.

The primary tool used by Bay of Plenty Regional Council (BoPRC) to assess and report on lake water quality is the Trophic Level Index (TLI). The TLI is assigned to annual average components comprised of chlorophyll *a*, total phosphorus and total nitrogen, and transparency (Burns et al. 1999), whereby high TLI values indicate water with high nutrient and phytoplankton concentrations, and low clarity. It is generally acknowledged that water quality of Lake Rotorua began to deteriorate in the 1960s. Therefore, a target TLI value of 4.2 was set in the Action Plan as a goal for water quality restoration in Lake Rotorua, corresponding to an estimated TLI value for the 1960s.

The Lake Rotorua Action Plan recommended the dosing of inflows to Lake Rotorua with aluminium sulphate (alum) in order to reduce the phosphorus load to the lake. Water treatment with alum is a well-established lake restoration method used worldwide for controlling phosphorus (P) in the water column and/or bottom sediments, and has been used for more than four decades with varying degrees of success (Cooke et al. 1993; Welch and Cooke, 1999; Lewandowski et al. 2003; Pilgrim et al. 2007; Egemose et al. 2013). Despite the frequent use of alum in wastewater systems and directly to lakes, dosing alum to surface inflows of lakes is a relatively innovative method which has received little attention in the literature (Pilgrim and Brezonik, 2005; Churchill et al. 2009). The Utuhina stream and two other streams were proposed to be dosed with alum. To date, alum has been used in two Lake Rotorua inflows, the Utuhina (commenced 2006) and Puarenga (commenced 2009) streams (McIntosh, 2012).

Water quality in Lake Rotorua has improved since alum dosing commenced, and the annual TLI target (4.2) was reached in 2012 (Figure 1). It has been hypothesised that alum dosing carried out in the Utuhina and Puarenga steams has had subsidiary beneficial effects on the water quality in Lake Rotorua. These effects relate to adsorption of P in the water column and the settling of alum flocs to the bottom sediments of the lake, which in turn could reduce the release of phosphorus into the overlying water column. The potential for these mechanisms to explain the observed improvement in water quality is examined in detail by Özkundakci et al. (2013).

Effective management of catchment nutrient loads to lake systems is essential for the management of water quality (e.g. Schindler, 2006, Jeppesen et al. 2007, Sondergaard et al. 2007). Bay of Plenty's 'Rule 11' sets 'benchmark' nutrient export loads for properties within the catchments of five Rotorua Lakes, based on export rates from 2001 to 2004. The Lake Rotorua Action Plan suggests more aspirational 'sustainable' catchment-wide loads of 435 t N yr^{-1} for Lake Rotorua. More recently, the Oturoa Agreement between the Lake Rotorua Primary Producers Collective Inc., Federated Farmers Rotorua and Bay of Plenty Regional Council set a time frame of 2032 to attain the 435 t yr^{-1} sustainable nitrogen load. The year 2032 is designed to provide the time needed for implementation and to take account for the long groundwater residence times in the Rotorua catchment. Catchment modelling with ROTAN has shown that N loads could still continue to increase to as high as high as 750 t yr^{-1} before reducing towards the agreed target N load (Rutherford et al. 2011, Hamilton et al. 2012).

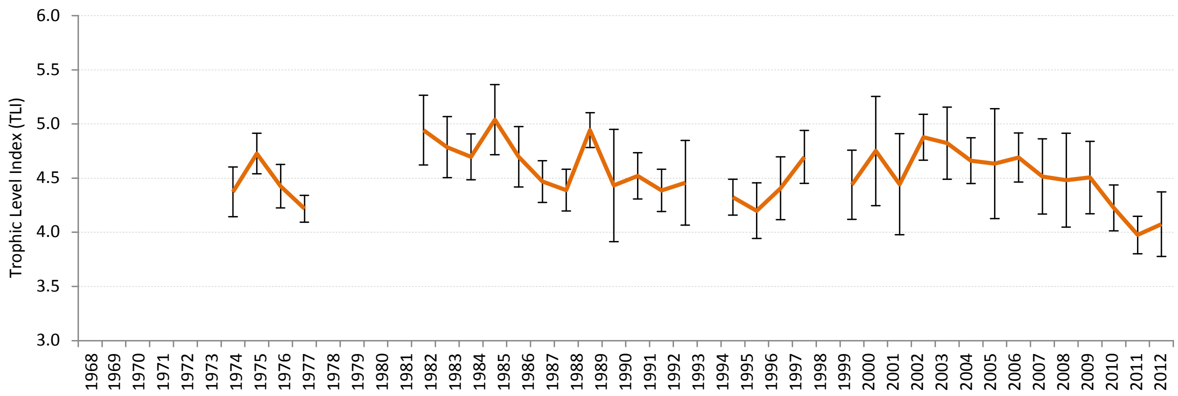


Figure 1. Trophic Level Index values from the 1970s to 2013. Values are plotted only when at least four months of data were available for all four constituent TLI variables (total nitrogen, total phosphorus, chlorophyll *a* and Secchi depth). Note also that analytical techniques have changed within the time period shown and may have affected inter-annual values. Error bas represent one standard deviation. Data from Bay of Plenty Regional Council records.

The present study utilises a process-based modelling approach to assess the effects of alum dosing and catchment nutrient export rates on Lake Rotorua water quality. The model used is the coupled hydrodynamic-ecological model 'DYRESM-CAEDYM', which has been previously applied to Lake Rotorua (Burger et al. 2008, Hamilton et al. 2012). Output from the Rotorua catchment model ROTAN (Rutherford 2011) is used to provide data for flow and nitrogen inputs from the catchment, in order to 'drive' the lake model (as in Hamilton et al. 2012). The objectives of this work are to address the following questions:

- *Can alum dosing of two stream inflows to Lake Rotorua explain the observed improvement in water quality*? Specifically, we use the lake model to simulate scenarios including no alum application, alum application with associated adsorption of dissolved phosphorus within the streams, and alum application with in-lake effects including increased sedimentation of particulate organic matter and suppression of internal releases of phosphate during hypoxia. By comparing observed field data with model output for these scenarios, insight can be gained into the processes most likely to explain recent improvements in water quality.
- *How might changes in external (catchment) nutrient loads of N and P affect water quality in Lake Rotorua?* Specifically, we simulate a range of external nutrient load

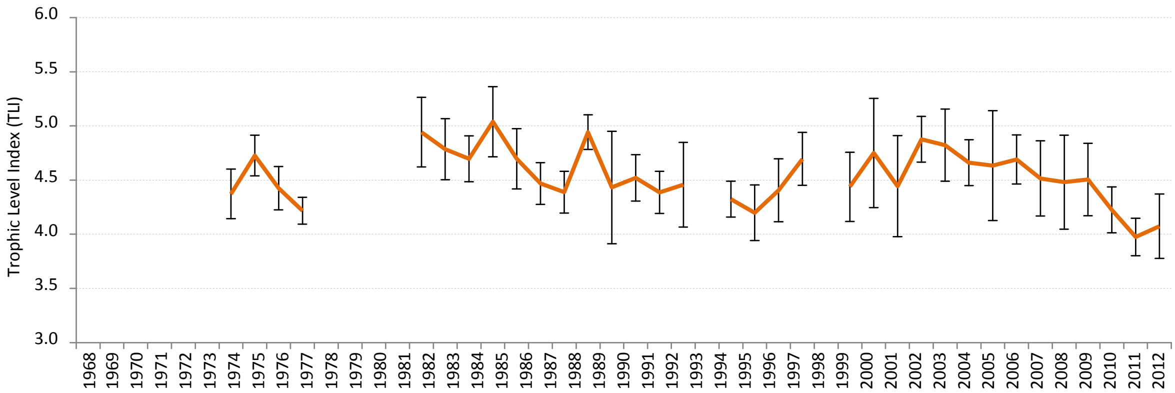
scenarios, spanning the 'sustainable' action plan load of 435 t N yr^{-1} to the 'worst case' 750 t N yr^{-1}, with corresponding changes to P loads.

The process of addressing the questions above includes by necessity a detailed consideration of N and P loads from all sub-catchments, as well as analysis of the efficacy of P adsorption in the two inflows alum dosed to date.

2. Methods

2.1 Study site – Lake Rotorua

Lake Rotorua is a large (80.8 km^2), relatively shallow (mean depth 10.8 m) lake of volcanic origin. Its catchment has an area of approximately 425 km^2 (Effective management of catchment nutrient loads to lake systems is essential for the management of water quality (e.g. Schindler, 2006, Jeppesen et al. 2007, Sondergaard et al. 2007). Bay of Plenty's 'Rule 11' sets 'benchmark' nutrient export loads for properties within the catchments of five Rotorua Lakes, based on export rates from 2001 to 2004. The Lake Rotorua Action Plan suggests more aspirational 'sustainable' catchment-wide loads of 435 t N yr^{-1} for Lake Rotorua. More recently, the Oturoa Agreement between the Lake Rotorua Primary Producers Collective Inc., Federated Farmers Rotorua and Bay of Plenty Regional Council set a time frame of 2032 to attain the 435 t yr^{-1} sustainable nitrogen load. The year 2032 is designed to provide the time needed for implementation and to take account for the long groundwater residence times in the Rotorua catchment. Catchment modelling with ROTAN has shown that N loads could still continue to increase to as high as high as 750 t yr^{-1} before reducing towards the agreed target N load (Rutherford et al. 2011, Hamilton et al. 2012).



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), with complex hydrogeology including large unconfined aquifers that retain groundwater for long and variable periods. Rutherford et al. (2011) produced a series of land use maps that indicate how land use of the Rotorua catchment has changed between 1940 and 2010. In summary, the urban area has expanded substantially during this time but with progressive reticulation of septic tanks into the centralised wastewater treatment plant. Relatively low-intensity pastoral land cover comprising mostly sheep and beef farms covered a considerable area of the lower catchment as early as 1940 but also expanded rapidly around the lake margin and then progressively into the upper catchment. Dairy farming has more recently replaced both forestry and sheep and beef farming in the north-west of the catchment, mostly in and around the Mamaku Plateau, particularly over the past 3-4 decades. In general there has been a gradual loss of forest with the expansion of pasture in the Rotorua catchment to the point where pasture now makes up about 50% of the total catchment area.

Phytoplankton biomass and production in Lake Rotorua may be limited by nitrogen (N) and/or phosphorus (P), as well as other environmental factors depending on time of year and the location within the lake (Burger et al. 2007b). Ratios of N:P are low by comparison with many other lakes in New Zealand and particularly overseas, and, based on Redfield ratios (see Abell et al. 2010) suggest that either nutrient could potentially limit phytoplankton productivity. White et al. (1977) found using laboratory based bioassays that N consistently limited algal biomass. Burger et al. (2007b) conducted similar bioassays *in situ*, which combined with a modelling study, indicated that co-limitation by N and P was common. Bioassays carried out in large-scale mesocosms in Lake Rotorua in summer 2009-10 also indicated that co-limitation was most frequent, as opposed to limitation by either N or P (Hamilton, unpubl. data). Most recently Abell et al. (2014) showed that N-limitation was dominant in the central pelagic zone in the lake while there was little evidence of nutrient limitation adjacent to the Ngongotaha Stream inflow. The bioassay experiments by Abell et al. (2014) in December 2014 occurred when nitrate concentrations were relatively low compared to periods either side of the study.

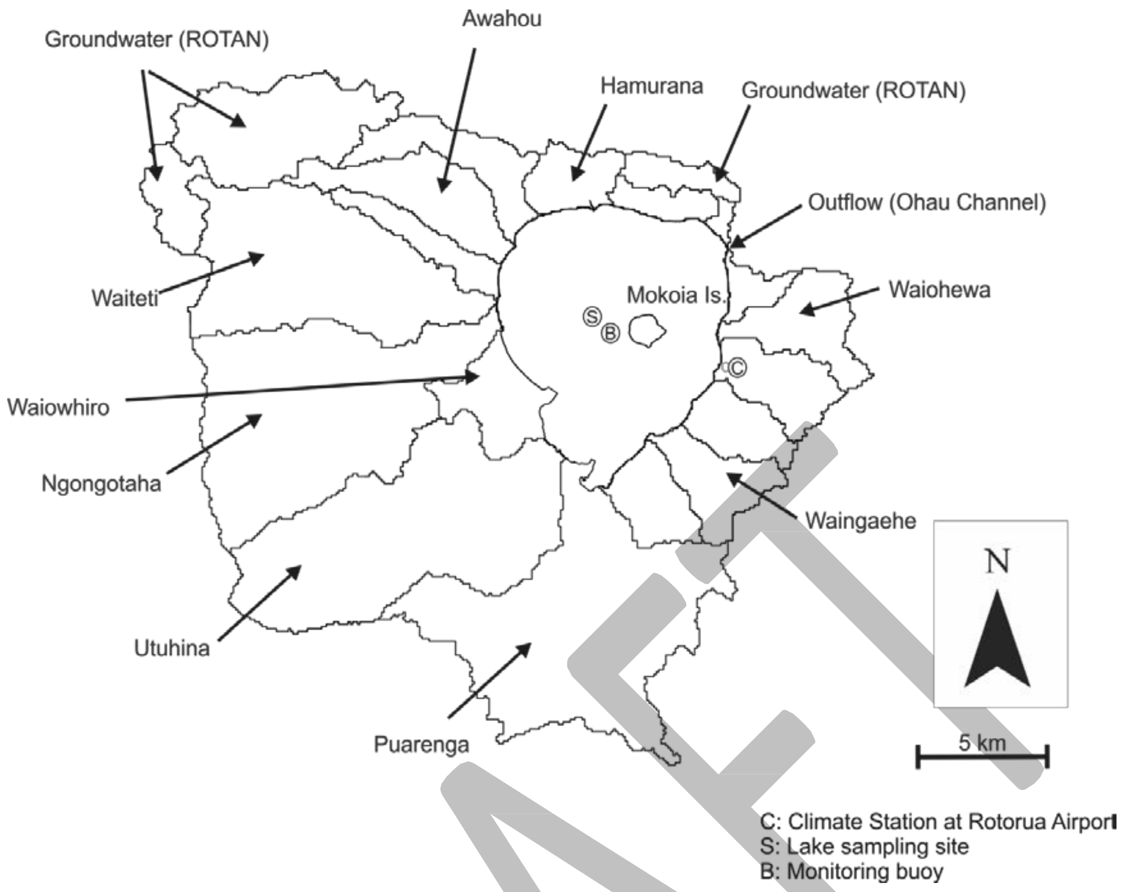

Figure 2: Map of the Lake Rotorua catchment with details of ROTAN sub-catchments (from Rutherford et al. 2011), and showing the BoPRC sampling site (S), the high-frequency monitoring buoy (B) and the Rotorua airport weather station (C). Mokoia Island is the closed circle in the centre of the lake, and unnamed subcatchments are those without permanently flowing surface streams.

2.2 Lake model description

The one-dimensional (1D) hydrodynamic model DYRESM (version 3.1.0-03) was coupled with the aquatic ecological model CAEDYM (version 3.1.0-06), both developed at the Centre for Water Research, The University of Western Australia, to simulate water quality in Lake Rotorua. DYRESM resolves the vertical distribution of temperature, salinity, and density, and the vertical mixing processes in lakes and reservoirs. CAEDYM simulates time-varying fluxes that regulate biogeochemical variables (e.g., nutrient species, phytoplankton biomass). The model includes comprehensive process representations for carbon (C), nitrogen (N), phosphorus (P), and dissolved oxygen (DO) cycles, and suspended particulate matter. Many applications have been made of DYRESM-CAEDYM to different lakes (e.g., Bruce et al., 2006; Burger et al., 2008; Trolle et al., 2011; Gal et al., 2009; Özkundakci et al., 2011) and these publications have detailed descriptions of the model equations. Hamilton et al. (2012) describes in detail the setup of DYRESM-CAEDYM for Lake Rotorua.

2.3 Model timesteps and baseline simulation period

In this study, DYRESM-CAEDYM was run at hourly time steps between July 2001 and December 2012, with daily averaged input data and daily output data at 0900 h. The period July 2004 to June 2007 was used for calibration of the model, and the validation period was July 2001 to June 2004. These periods were selected in order that the model could be calibrated and validated without the potentially confounding influence of alum application to the Utuhina (2006 to present) and Puarenga (2010 to present) inflows. Scenarios were simulated using runs of the model over the period July 2007 to December 2012, corresponding to the period of alum dosing to inflows.

2.4 Meteorology

Meteorological data required for simulations (2001 – 2012) were obtained from the National Climate Data Base, for the Rotorua Airport climate station c. 50 m from the Lake Rotorua shoreline. Variables included air temperature (°C), shortwave radiation (W m^{-2}), cloud cover (fraction of whole sky), vapour pressure (hPa), wind speed (m s^{-1}) and rainfall (m) (Figure 3). Data are collected at Rotorua airport at various time intervals from one hour to whole-day, and for the purposes of model input were standardised to daily average values except for rainfall, which was a daily total value. Airport air temperature for the entire period of 2001 – 2012 was adjusted using linear regression with air temperature measurements of 2007 – 2012 from a high-frequency monitoring buoy.

2.5 Water balance

Surface inflow discharges to the lake were obtained from output of the ROtorua TAupo Nitrogen model (ROTAN; Rutherford et al., 2011). Flows for nine streams of the major Lake Rotorua sub-catchments were included, along with a tenth stream inflow representing the sum of all minor surface flows from around the lake. Each of these surface flows accounted for both stream and groundwater inputs from the respective sub-catchments. Rainfall was removed from the meteorological input to the model, and instead daily rainfall directly on the lake was represented as a surface inflow in order to account for atmospheric deposition of N and P which would not otherwise be accounted for in the rainfall input in the present model version.

Change in lake storage (ΔS) was calculated from water level recorder data provided by BoPRC, multiplied by the water level-dependent lake area derived from hypsographic curves (also provided by BoPRC). Daily values for the outflow volume were calculated as a residual term of a water balance for the simulation period:

$$\sum(\quad) \qquad (4)$$

where:

is evaporation in $m^3\ d^{-1}$

is change in storage in $m^3\ d^{-1}$

The resulting flow was averaged over 21 days to remove any negative values. Derived outflow was used for a DYRESM simulation over the period 2001–2012. The estimated outflow (calculated from the water balance described above) and the observed Ohau Channel discharge showed very good agreement (Figure 5). Lake level output was compared to BoPRC water level recorder data, and also matched closely (Figure 4).

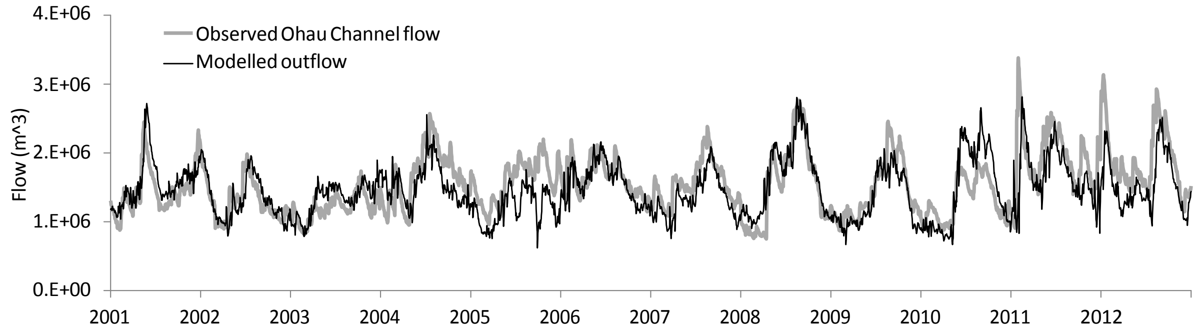

Figure 3. Simulated 15-day average Lake Rotorua outflow derived from the lake water balance, and observed Ohau Channel flow.

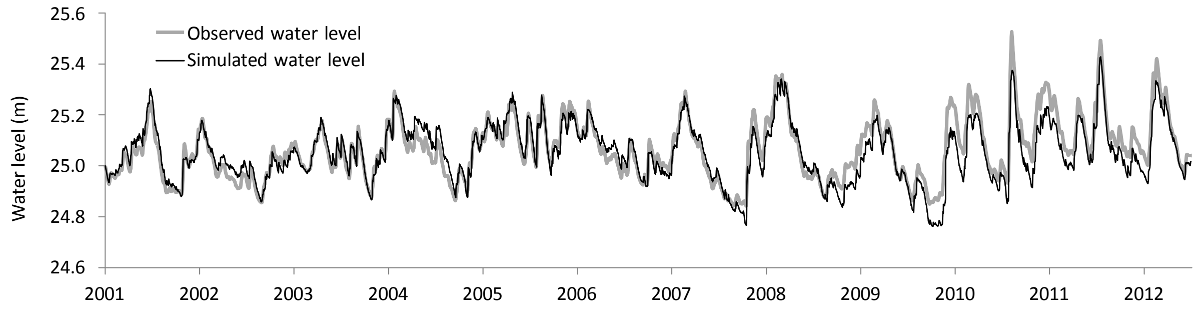

Figure 4. Simulated Lake Rotorua water level, and observed BoPRC water level recorder data.

2.6 Surface inflow, groundwater and rainfall parameterisation

A total of 11 inflows to Lake Rotorua were simulated, including nine major streams, an inflow representing the sum of all inflows from minor sub-catchments, and another representing direct rainfall to the lake surface. All non-rain inflows were assumed to represent both surface and groundwater inputs to the lake, consistent with the modelling approach within ROTAN (Rutherford et al. 2011).

Table 1. Average annual flow and nutrient loads for all inflows over the calibration, validation, and scenario periods (2001 – 2012).

Inflow	Flow	TN ($t\ y^{-1}$)	NH_4-N ($t\ y^{-1}$)	NO_3-N ($t\ y^{-1}$)	Organic N ($t\ y^{-1}$)	TP ($t\ y^{-1}$)	PO_4-P ($t\ y^{-1}$)	Organic P ($t\ y^{-1}$)	Particulate Inorganic P ($t\ y^{-1}$)
Awahou	5.17E+07	59.3	0.4	56.2	2.7	3.7	3.3	0.3	0.1
Hamurana	8.06E+07	62.4	0.8	57.7	3.9	7.0	6.5	0.3	0.1
Ngongotaha	6.70E+07	74.3	1.4	61.2	11.7	3.5	1.7	1.3	0.5
Puarenga	6.63E+07	117.8	6.6	91.7	19.5	5.1	2.5	1.8	0.7
Utuhina	7.35E+07	58.2	2.7	47.3	8.1	4.7	2.4	1.6	0.7
Waingaehe	9.87E+06	13.1	0.1	12.1	0.8	1.1	0.9	0.2	0.1
Waiohewa	1.32E+07	27.3	12.2	12.3	2.8	0.9	0.3	0.5	0.2
Waiowhiro	1.57E+07	13.2	0.3	11.7	1.2	0.7	0.5	0.1	0.1
Waiteti	4.37E+07	81.6	1.2	74.8	5.5	2.2	1.4	0.6	0.2
Minor	5.52E+07	124.4	1.3	114.5	8.6	2.8	1.9	0.7	0.3
Rainfall	1.21E+08	29.2	0.0	29.2	0.0	1.3	1.3	0.0	0.0
TOTAL	**5.98E+08**	**660.6**	**27.0**	**568.7**	**65.0**	**33.1**	**22.8**	**7.3**	**3.0**

2.6.4 Phytoplankton

The Hamurana, Awahou and rainfall inflows were given a concentration of zero for each of the three phytoplankton groups. Those inflows not dominated by groundwater springs were prescribed a 'seeding' concentration of 0.1 µg chlorophyll *a* L^{-1} for each phytoplankton group.

2.7 Model calibration and validation

DYRESM-CAEDYM was calibrated against field data for a three-year period between July 2004 and June 2007 for variables of temperature, DO, PO_4-P, TP, NH_4-N, NO_3-N and TN. Monthly samples collected and analysed by BoPRC were used to assess model performance. Near-surface samples were collected using a tube sampler from 0 – 6 m in the water column, and a Schindler-Patalas trap from between 18 and 20 m. These field samples were compared with simulation output for near surface (3 m depth), and near-bottom (19 m depth), respectively. The three simulated phytoplankton groups collectively contributed to a total simulated chlorophyll *a* concentration, but with cyanophytes dominating during summer and diatoms/chlorophytes during winter and early spring, in a sequence similar to what has been observed previously in lakes Rotorua (Paul et al. 2012) and Rotoiti (Von Westernhagen et al. 2010). The sum of the chlorophyll concentrations for all three groups was calibrated against surface chlorophyll *a* measured using an acetone extraction procedure (Arar and Collins, 1997) carried out by NIWA (on contract to BoPRC). Model parameters were adjusted manually using a trial and error approach with values set to within literature ranges (e.g., Schladow and Hamilton, 1997; Trolle et al., 2011). The model error, represented by the root-mean-square-error (RMSE) and Pearson correlation coefficient (R) for each output variable, was quantified after each simulation for which model parameter values were adjusted. Calibration continued until there was negligible improvement in RMSE and R values with repeated model simulations. RMSE and R values were also compared to values from modelling studies in the literature, to assess an acceptable model error for prediction purposes.

A three-parameter TLI value was calculated for each year of the simulation period. The relevant equations for determination of the TLI are:

$$TL_{Chla} = 2.22 + 2.54 log(Chla) \quad (11)$$

$$TL_{TP} = 0.218 + 2.92 log(TP) \quad (12)$$

$$TL_{TN} = -3.61 + 3.01 log(TN) \quad (13)$$

$$\mathrm{TLI} = \frac{1}{3}\sum\left(\mathrm{TL_{Chla}}, \mathrm{TL_{TP}}, \mathrm{TL_{TN}}\right) \quad (14)$$

where:

TL_{Chla}, TL_{SD}, TL_{TP} and TL_{TN} represent the individual level trophic level indices for the individual variables of chlorophyll *a*, total phosphorus and total nitrogen. The Secchi depth component of the traditional TLI was ignored, because CAEDYM does not explicitly simulate Secchi depth.

TLI output from the model was compared with observed data and calibration of parameters was undertaken in DYRESM-CAEDYM until a satisfactory match was achieved. We aimed to calibrate the model TLI within ± 0.1 TLI units of the measured TLI. The final model parameters from the calibration were then fixed for model validation over the period July 2001 – June 2005.

2.8 Scenarios

2.8.1 Simulating the possible effects of alum dosing the Utuhina and Puarenga inflows

A number of model simulation runs were undertaken to explore the possible mechanism of alum dosing and its impact. These scenarios were run over a period when alum dosing was used in the lake but involved keeping all other model input variables at constant levels, i.e., identical meteorology, inflow and outflow data. The four different aluminium dosing scenarios included no alum dosing (S50-Al), alum dosing on the Utuhina and Puarenga streams and assuming that the DRP was stripped out according to observations in the streams below the point of dosing (S50+Al), alum dosing of both streams with DRP stripping in the streams and elevated levels of removal of particulate organic material in the lake to reproduce flocculation effects caused by alum (S50+Al+Osed), and incorporation of in-steam DRP stripping by alum and flocculation effects, as well as a suppressed rate of phosphate release from the bottom sediments (S50+Al+Osed+intP).

2.8.2 Simulating the possible effects of varying catchment (external) nutrient loads

A number of simulations were undertaken to assess different catchment nutrient loading scenarios corresponding to different land uses and either including or not including alum dosing. Details of the different loading cases are given in Table 2 and correspond to the highest TN loads accounting for groundwater lag times and little mitigation (750 t yr^{-1}), with TP loads up to 40.3 t yr-1, as well as the current targets (435 t yr^{-1} for TN and an assumed value of 23.4 t yr^{-1} for TP).

Table 2. Summary of model scenarios, run for the period 2007 – 2012. Scenarios in grey are those designed to assess the potential effects of alum dosing of the Utuhina and Puarenga streams. Scenarios in orange are those designed to assess the possible effects of changing catchment nutrient loads.

Scenario	Description	Alum dosing?	ROTAN dairy N loss (kg N/ha/y)	Catchment TN load (t/y)	Internal N release (g/m^2/d)	Catchment TP load (t/y)	Catchment PO4 load (t/y)	Internal P release (g/m^2/d)	Sediment oxygen demand	Diameter of POM* (mm)
S50-Al	No alum dosing	None	50	641.5	0.50	34.5	23	0.0400	2.90	0.009
S50+Al	Alum dosing of Utuhina and Puarenga	Pua & Utu	50	641.5	0.50	34.5	20.3	0.0400	2.90	0.009
S50+Al+Osed	Alum, some in-lake flocculation	Pua & Utu	50	641.5	0.50	34.5	20.3	0.0400	2.90	0.018
S50+Al+Osed-intP	Alum, flocculation & suppressed internal P release	Pua & Utu	50	641.5	0.50	34.5	20.3	0.0200	2.90	0.018
Smin	Legislated N limit 2032	None	n/a	435	0.34	23.4	15.6	0.0271	0.46	0.009
S40	Decreased landuse intensity	None	40	615.3	0.48	33.1	22.1	0.0400	2.55	0.009
S50-Al	Current catchment loads	None	50	641.5	0.50	34.5	23	0.0400	2.90	0.009
S50+al+Osed+Pmit+intP				641.5			7.6	0.0066	1.47	
S60	Increased landuse intensity	None	60	696.7	0.54	37.5	25.0	0.0492	3.64	0.009
Smax	Maximum landuse intensity	None	n/a	750	0.86	40.3	26.9	0.0690	4.36	0.009
SmaxPmit	Max dairy, and on-land P control mitigation	All streams	n/a	750	0.61	30.3	20.2	0.0366	4.36	0.009
SmaxPmit+Osed-intP	Max dairy,on-land P control mitigation and full alum	All streams	n/a	750	0.58	30.3	10.1	0.0088	2.22	0.018

* Particulate organic matter

3. Results

3.1 Inflows: Field measurements and ROTAN simulations

Lake Rotorua inflow volumes, nitrate, ammonium, total nitrogen, dissolved reactive phosphorus (DRP), and total phosphorus concentrations are typically monitored on a monthly basis by Bay of Plenty Regional Council. Trends in nutrient concentrations (over the period 2001 – 2013) vary amongst the different inflows, with a trend of increasing nitrogen species' concentrations indicated for some inflows, (e.g. Awahou, Figure 5), whilst for others, concentrations appear to be relatively stable (e.g. Ngongotaha, Figure 7) or decreasing (e.g. Puarenga, Figure 8). Total phosphorus concentrations appear to have increased substantially in Waiteti Stream over 2011-2013 (Figure 13), whilst a striking decrease in DRP concentration is evident in both Puarenga (which is monitored upstream of the alum dosing location) and Utuhina (monitored downstream of dosing) Streams from 2009 onwards, (although it should be noted that DRP concentrations increased again in Utuhina Stream in 2012/2013; Figure 8 and Figure 9).

Measured inflow discharge and nitrogen concentration were also compared with relevant output from ROTAN simulations (Figures 5 - 15). The model appears to be capable of simulating the magnitude of the volume and nutrient load for most inflows, with some exceptions, such as the Puarenga, where the model appears to substantially overestimate the nitrogen load (Figure 8). Although the model does not tend to capture the timing of peaks and troughs in measured data, the ROTAN output represents daily average values, and field measurements represent c. monthly manual samples, making direct comparison difficult. Furthermore, ROTAN modelled inflow volume was used in the lake water balance, which resulted in a very good match between observed and modelled water level (Figure 4), suggesting that the gross water loads in ROTAN are appropriate for 'driving' the lake model.

3.1.1 Major inflow: Awahou

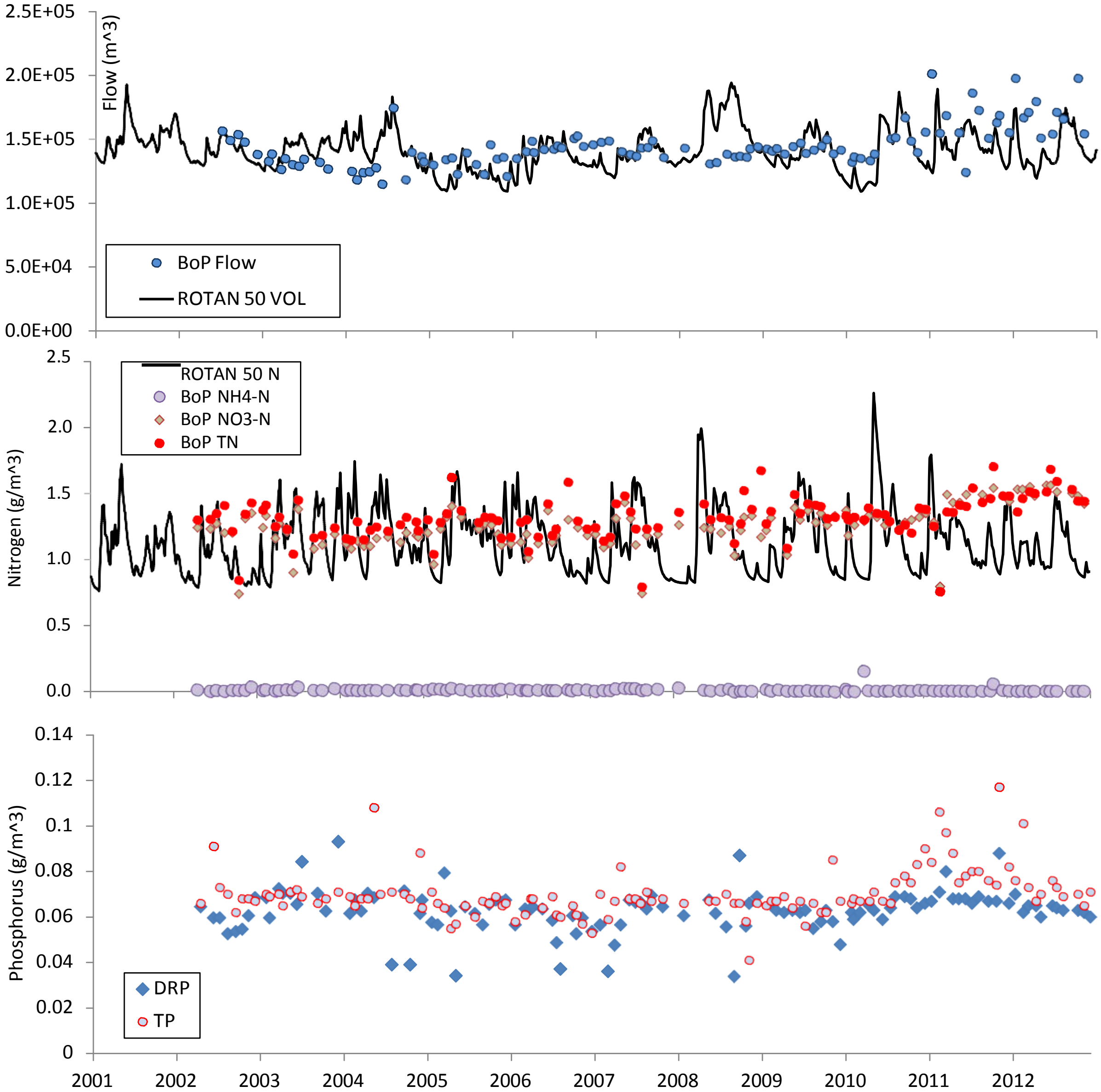

Figure 5. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Awahou stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.2 Major inflow: Hamurana

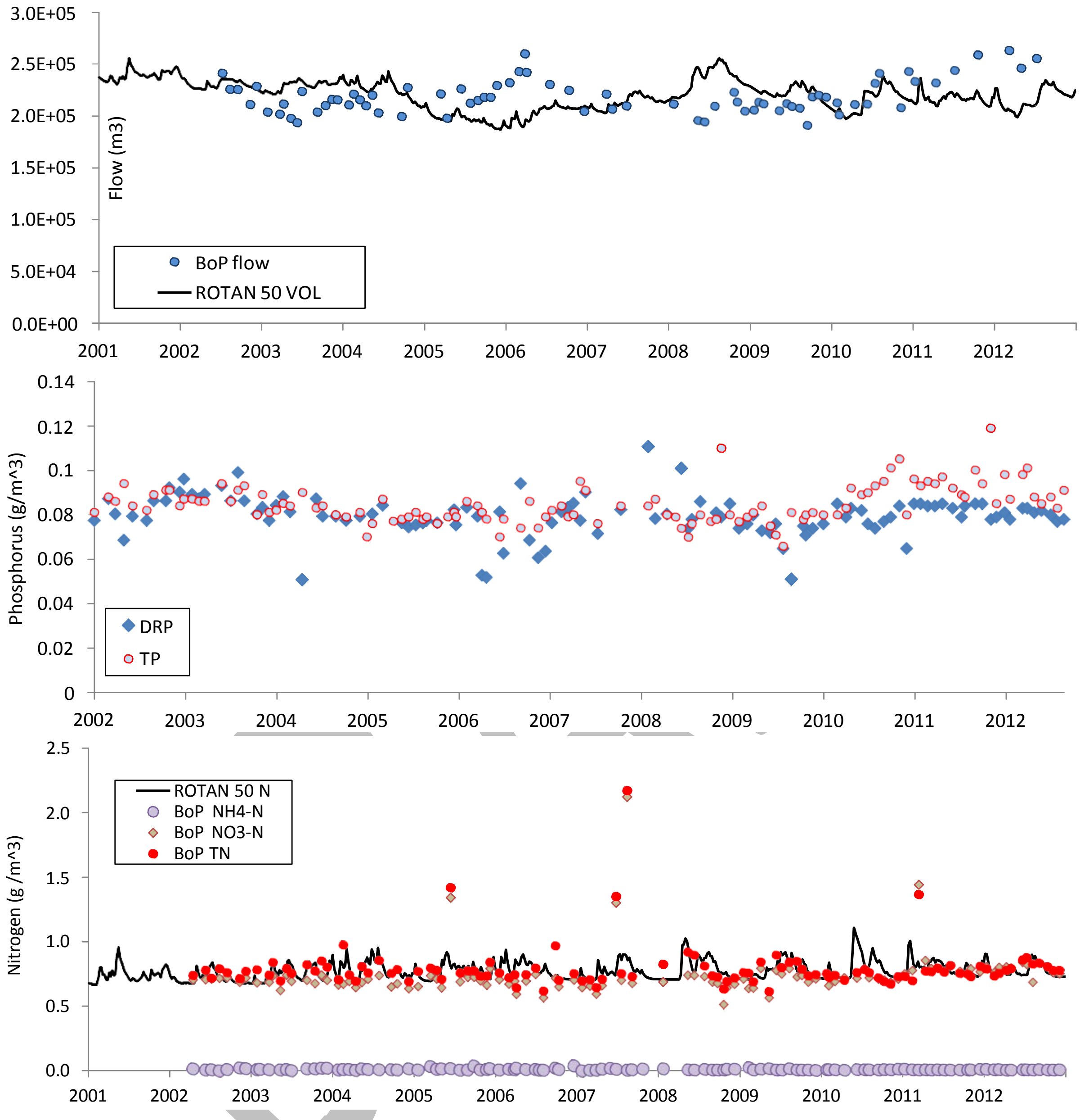

Figure 6. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Hamurana stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.3 Major inflow: Ngongotaha

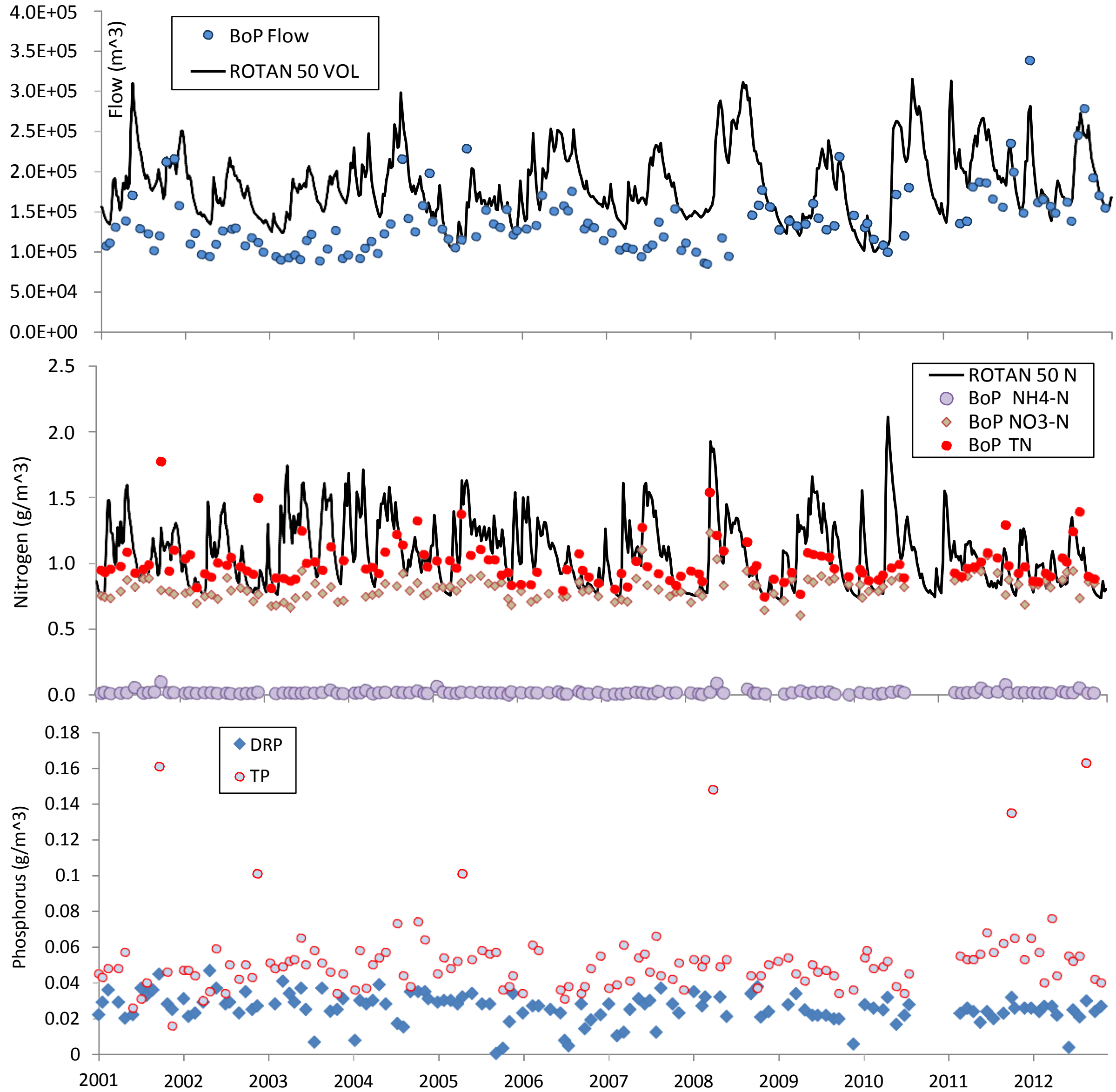

Figure 7. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Ngongotaha stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.4 Major inflow: Puarenga

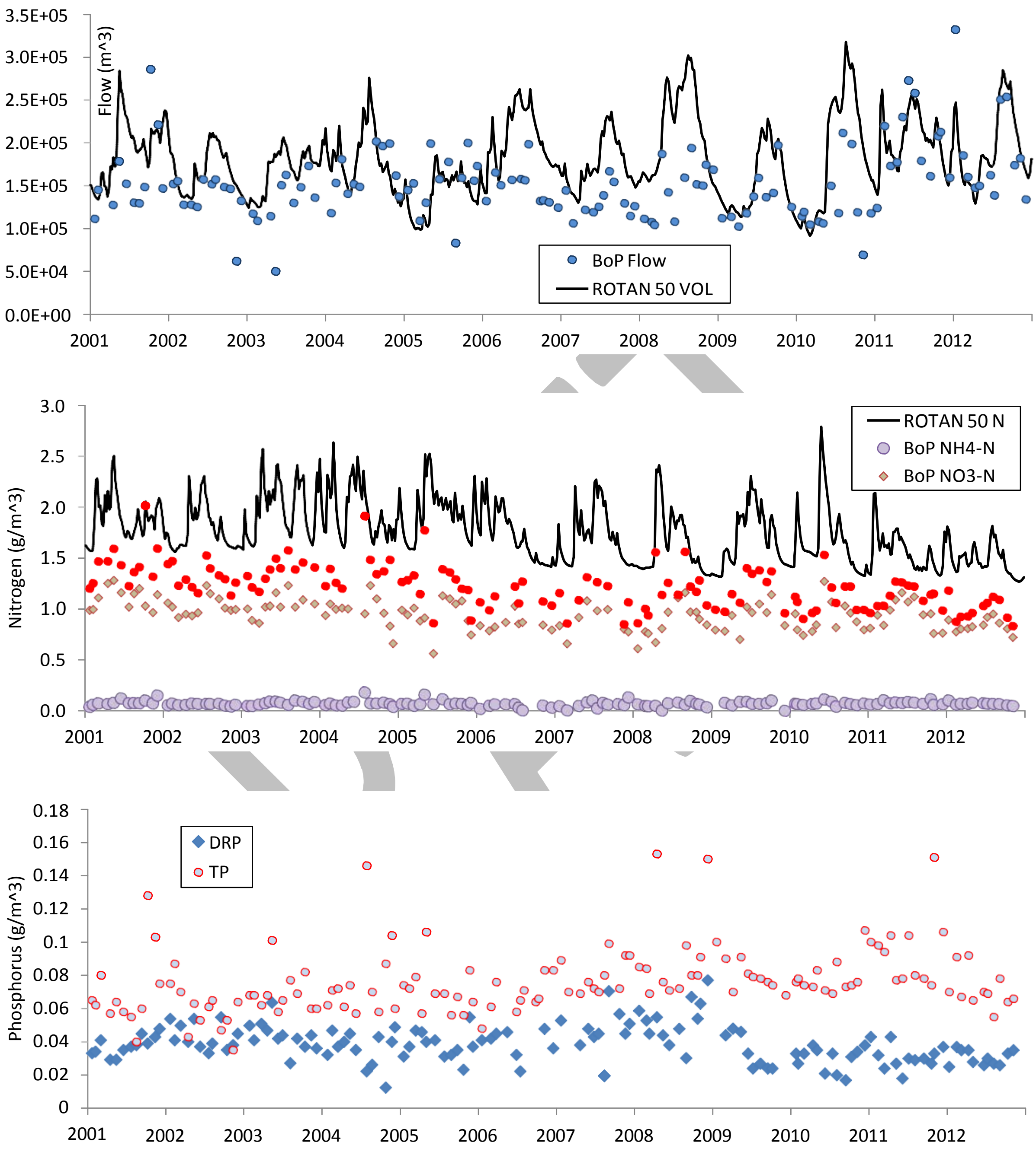

Figure 8. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Puarenga stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.5 Major inflow: Utuhina

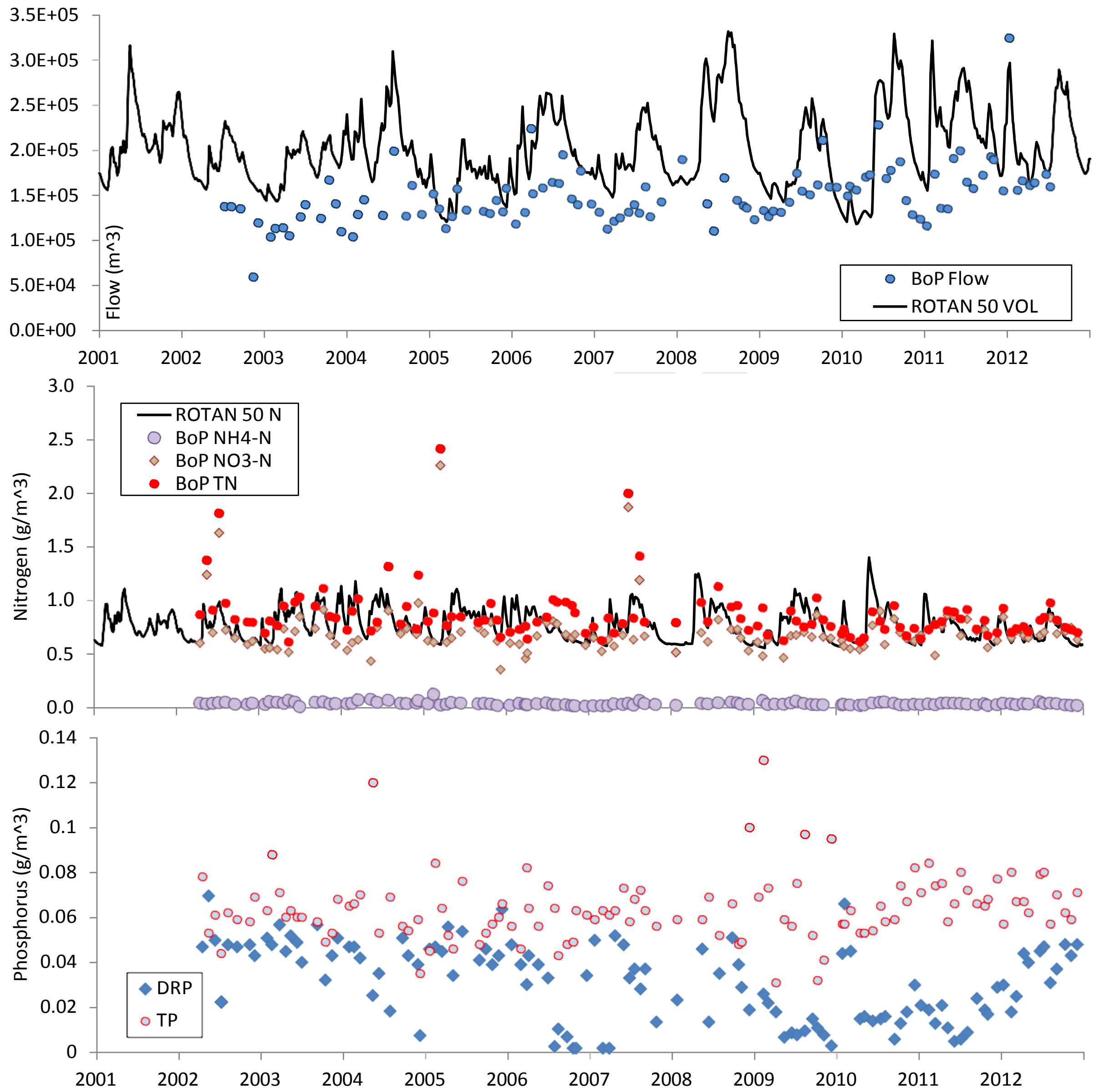

Figure 9. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Utuhina stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.7 Major inflow: Waingaehe

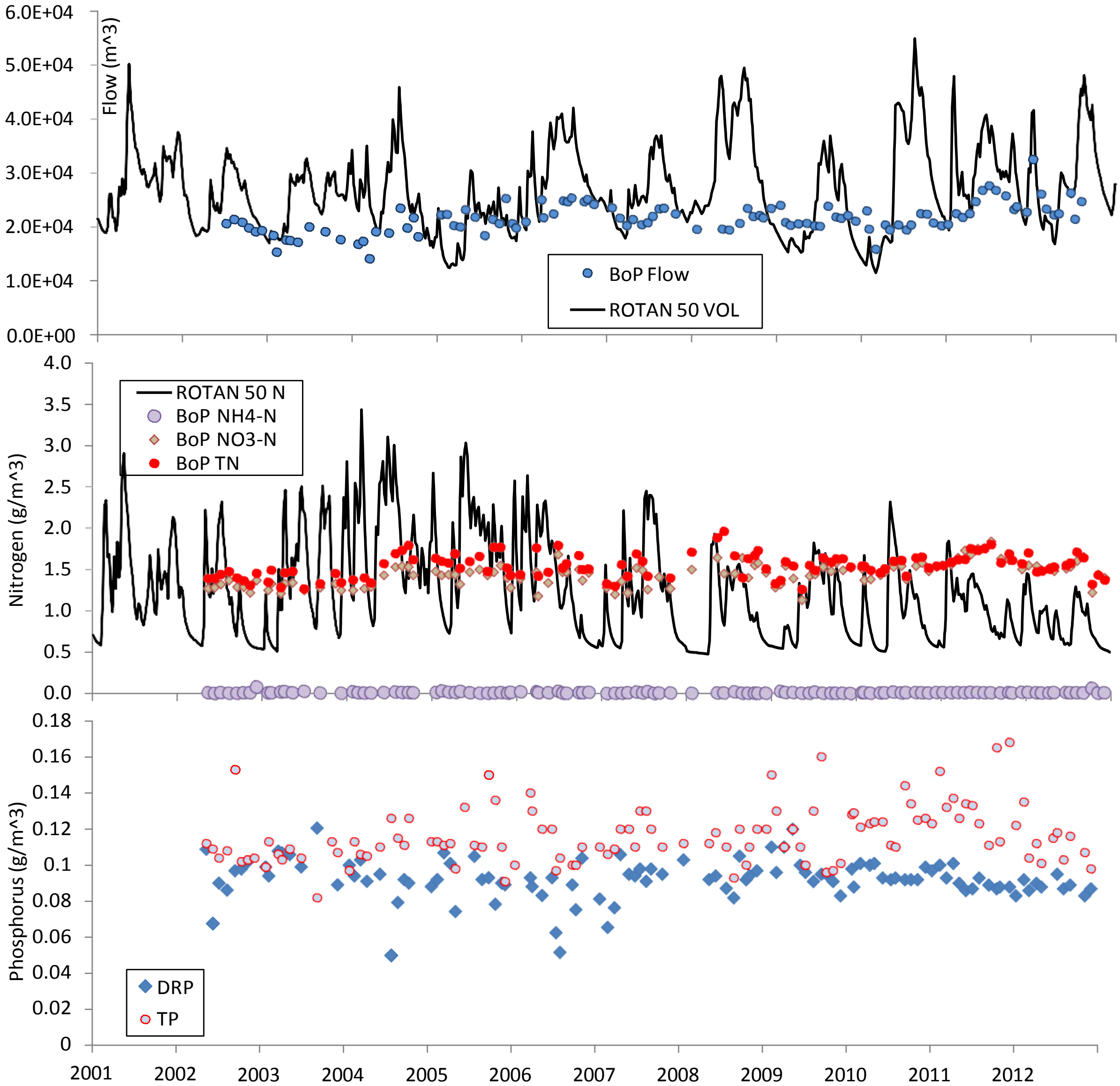

Figure 10. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Waingaehe stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.8 Major inflow: Waiohewa

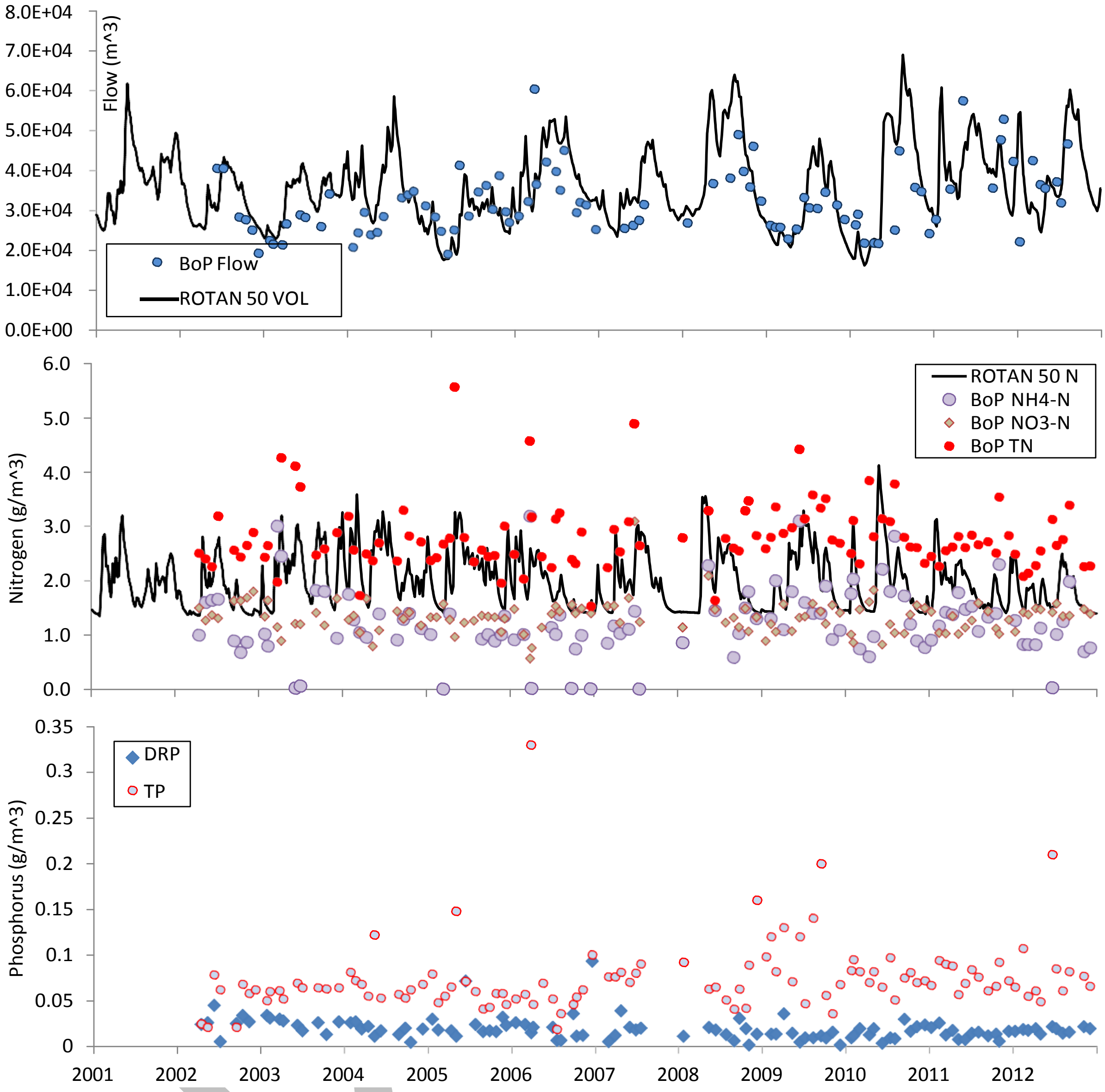

Figure 11. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Waiohewa stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.9 Major inflow: Waiowhiro

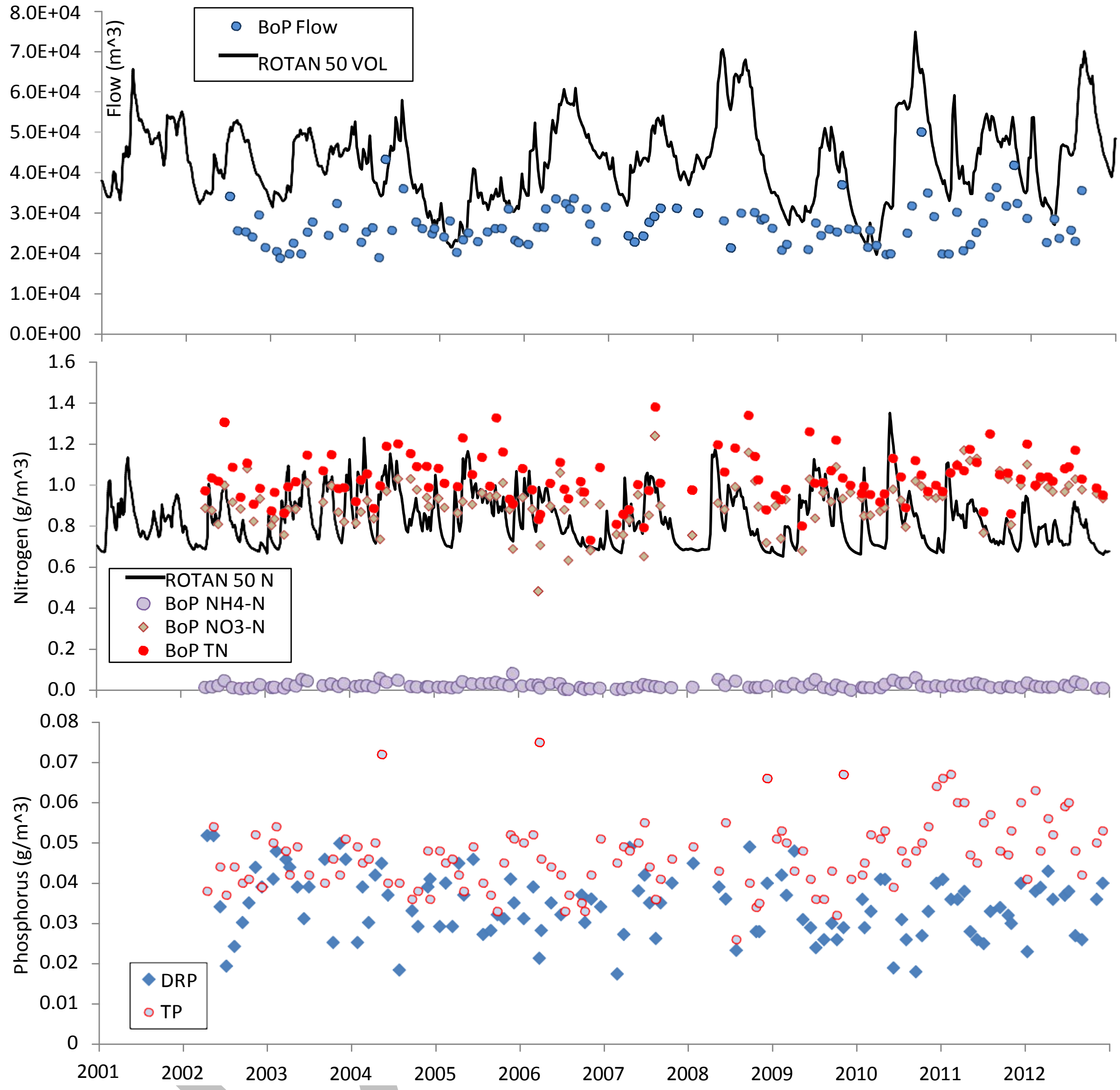

Figure 12. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Waiowhiro stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.10 Major inflow: Waiteti

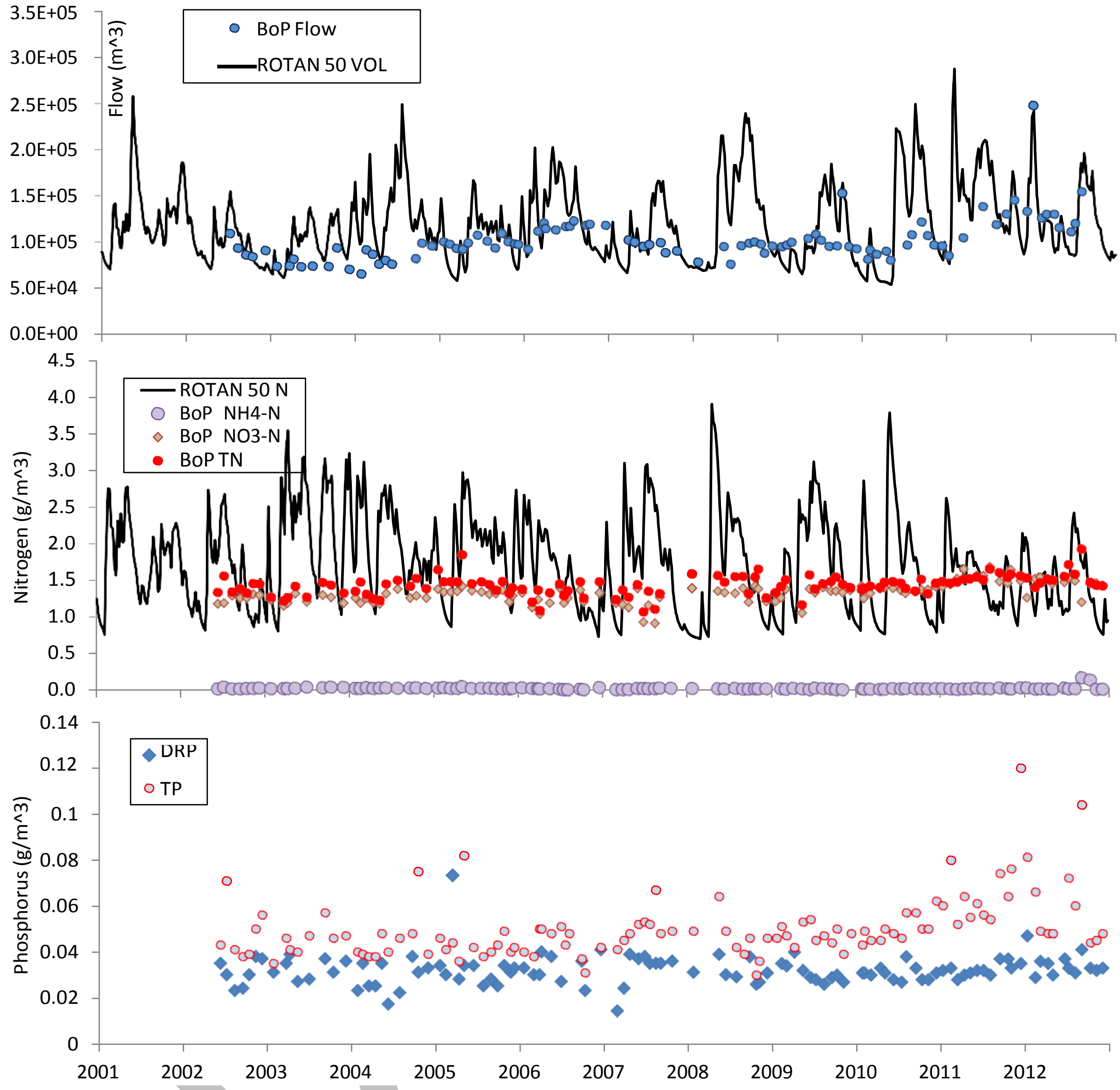

Figure 13. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for the Waiteti stream sub-catchment, for A) daily flow, B) nitrogen, and C) phosphorus.

3.1.11 Minor inflows

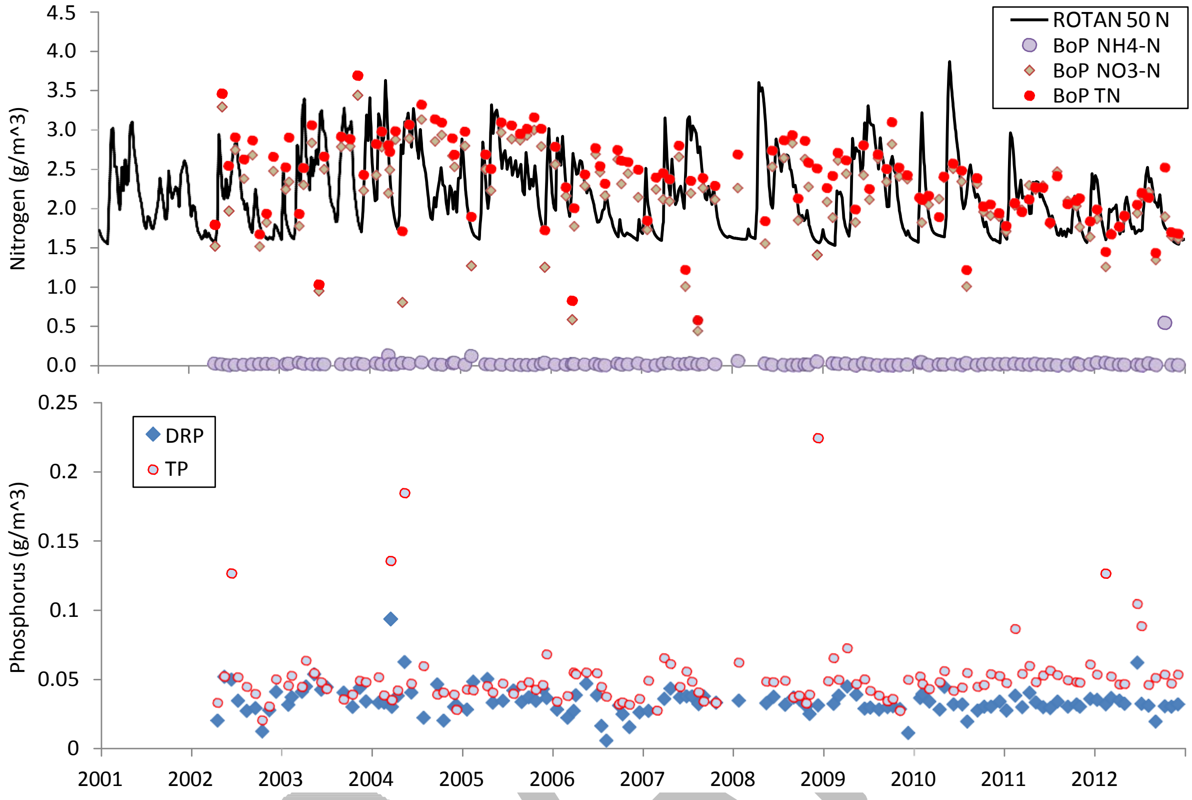

Figure 14. Time-series plots of Bay of Plenty inflow monitoring data (dots) and ROTAN output (lines) for minor inflows, for A) daily flow, B) nitrogen.

3.2 Alum application to Utuhina and Puarenga streams

Alum has been applied to two streams in an attempt to reduce bioavailable phosphorus that would otherwise fuel phytoplankton growth in Lake Rotorua. Both streams, Utuhina and Puarenga, have been dosed at a variable rate since the start of the dosing (c. mid-2006 for the Utuhina and early 2010 for the Puarenga Stream). Alum was dosed in the Utuhina stream at a rate of c. 0 – 100 kg aluminium d^{-1}, with much higher dosing (c. 200 – 600 kg aluminium d^{-1}) occurring intermittently. The Puarenga appears to have been dosed at a higher rate of c. 0 – 200/300 kg aluminium d^{-1}, bringing the combined dosing to c. 200 kg aluminium d^{-1} between 2011 and 2012 (Figure 15).

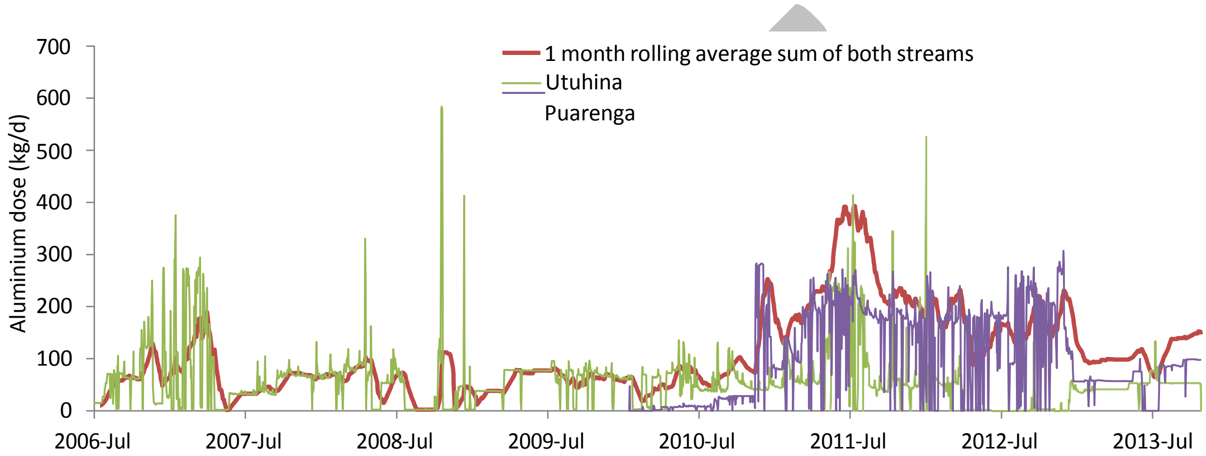

Figure 15: Alum dosing in Utuhina (green line) and Puarenga Streams (purple line), and combined dosing (1 month rolling average; red line). Data supplied by Alastair McCormack, Bay of Plenty Regional Council.

3.2.2 Utuhina stream

Monthly stream monitoring for Utuhina is carried out downstream of the alum dosing point. Periods of high alum dosing resulted in low ratios of dissolved to particulate P, presumably due to the formation of alum flocs via adsorption of dissolved P from the water column (Figures 16 and 17).

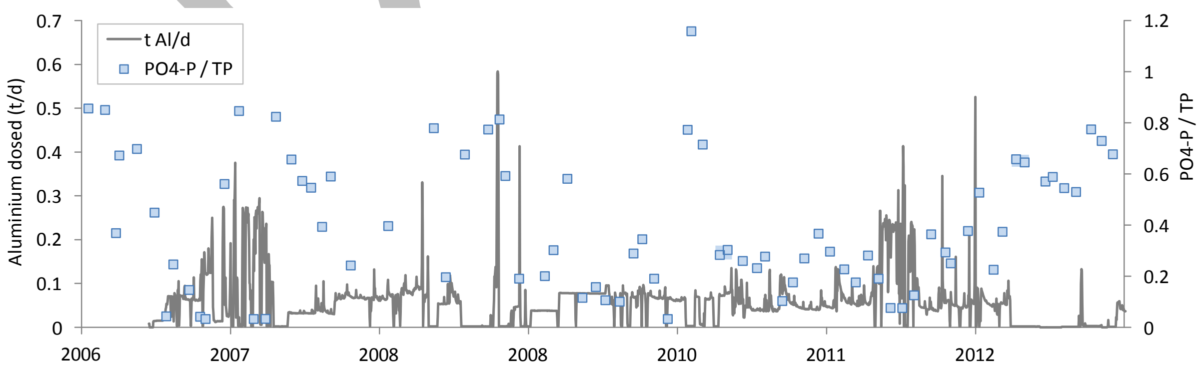

Figure 16. Aluminium dosed to the Utuhina stream (grey line) in tonnes Al per day (assuming 4.2 % of the alum product is aluminium), and the ratio (blue dots) of dissolved reactive phosphorus (DRP) to total phosphorus (TP). Data provided by Alastair McCormack and Paul Scholes, Bay of Plenty Regional Council.

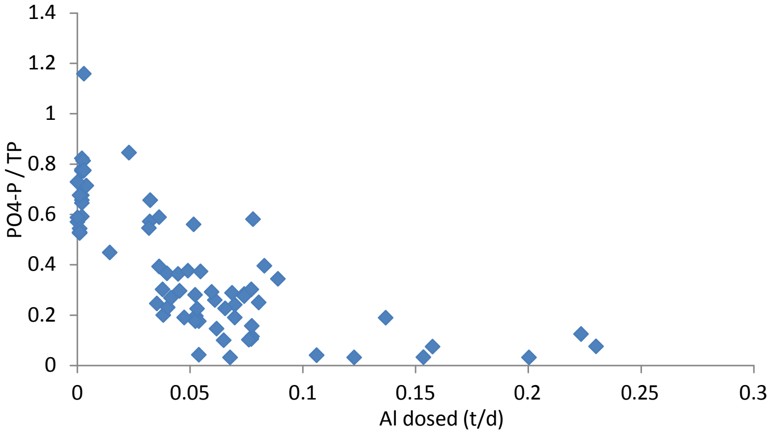

Figure 17. Relationship between daily aluminium dose and observed ratio of DRP to TP downstream of the dosing location in Utuhina Stream. Data provided by Alastair McCormack and Paul Scholes, Bay of Plenty Regional Council.

In order to simulate hypothetical scenarios with no alum dosing of either inflows, it was necessary to approximate likely dissolved P concentrations were the effects of alum absent. To this end, mean DRP:TP ratio was calculated for each month of the year using data between 2001 and 2006 (i.e., 'pre-alum'). This ratio was then applied to each measurement of TP 2006 – 2012, in order to estimate DRP concentration (Figure 18, triangles).

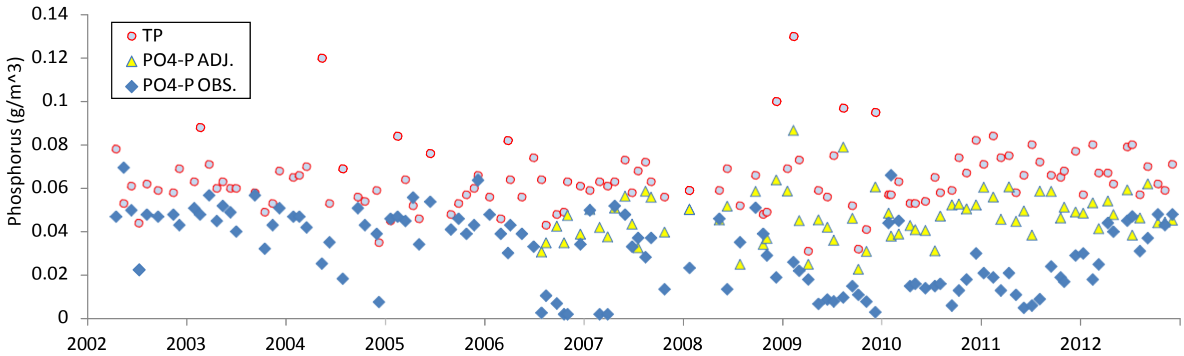

Figure 18. Time-series of inflow measurements in the Utuhina inflow for TP (red dots), DRP (blue diamonds). Green triangles represent estimated DRP if alum dosing had not been undertaken 2006 – 2012. This was estimated using an average of DRP:TP for each month of the period before alum dosing commenced.

3.2.2 Puarenga stream

Monthly monitoring of the Puarenga inflow is carried out upstream of the alum dosing point. However, BoPRC (John McIntosh) monitored dissolved and total P both upstream and downstream of dosing commencing in July 2011 (Figure 19). A dose of greater than 75 kg Al d^{-1} was highly effective, with typically >80% reduction of dissolved P. More specifically, at a dose ratio of Al:DRP greater than 20, most dissolved P was transferred to particulate form. Because observed dosed Al:DRP was as high as c. 90, it seems possible that substantial loads of Al could have been transported to the lake as free Al (Figure 20).

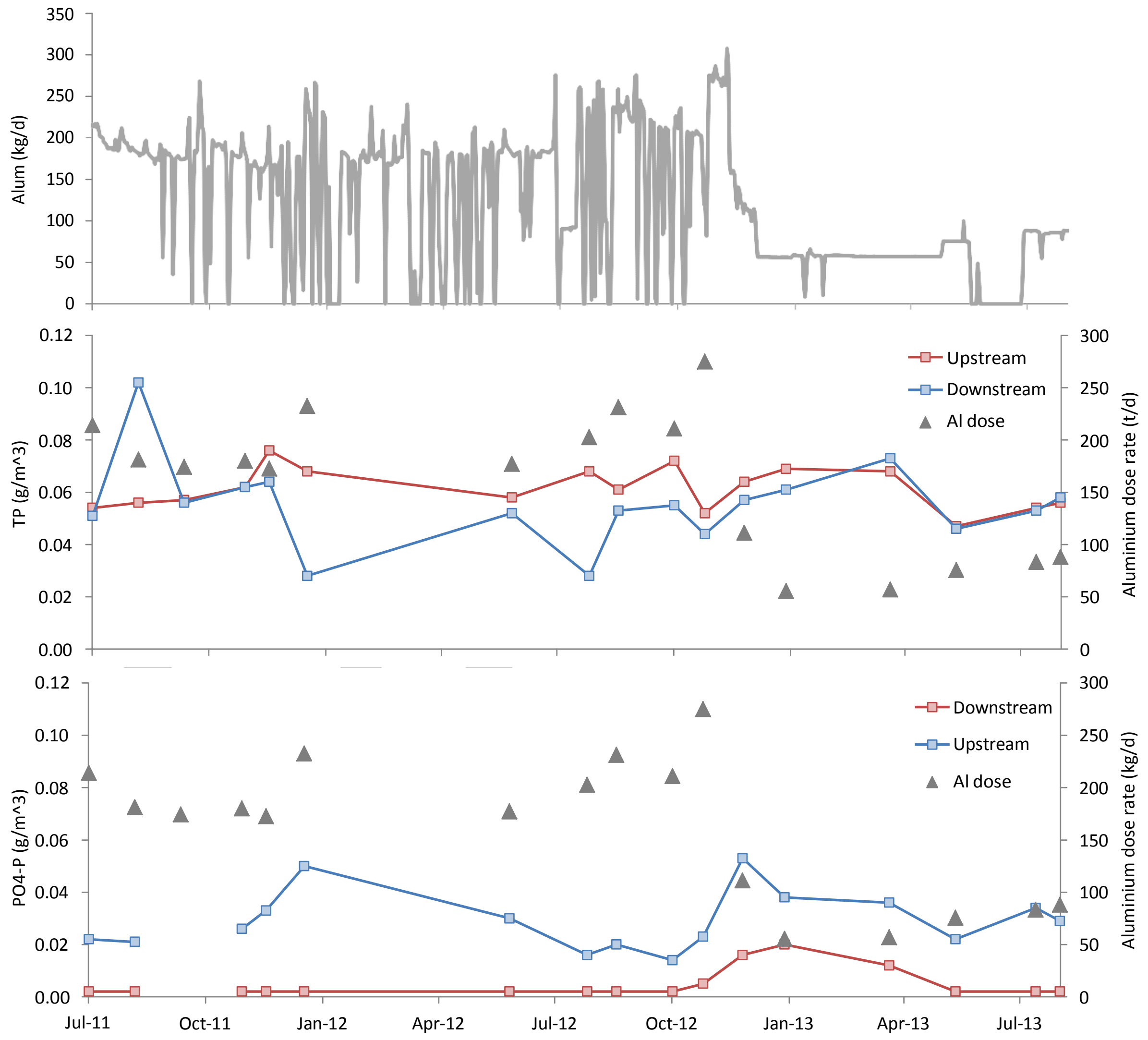

Figure 19. A) Dose rate of aluminium to the Puarenga inflow (assuming 4.2% of alum product is aluminium), B) Upstream and downstream TP concentrations with corresponding Al dose rate for spot measurements 2011 – 2013, C) Upstream and downstream DRP concentrations with corresponding Al dose rate for spot measurements 2011 – 2013. Data provided by John McIntosh and Bay of Plenty Regional Council.

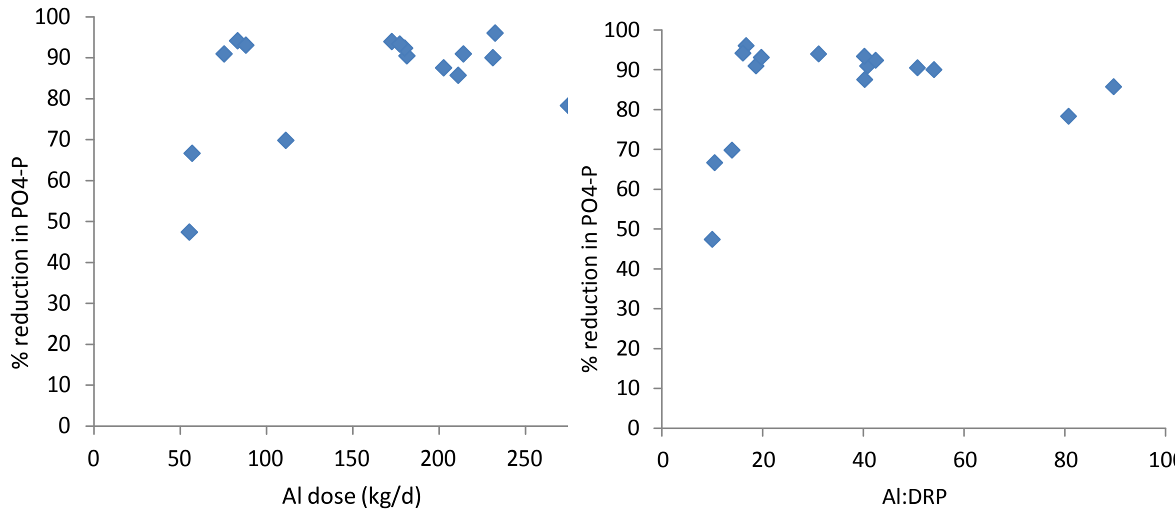

Figure 20. Percent reduction of DRP downstream of alum dosing in the Puarenga inflow, relative to A) aluminium dose rate, and B) aluminium to DRP ratio, for spot measurements 2011 – 2013. Data provided by John McIntosh.

3.3 In-lake measurements 2001-2012

Concentrations of the different nutrient species and chlorophyll *a* are shown in Figure 21 for the study period of 2001 – 2012. No attempt was made to fit trend lines to these data because of potentially confounding effects of alum dosing and the relatively short period of the record with which to generate meaningful time trends. The period in the early 2000s was characterised by relatively high concentrations of DRP including some large peaks in bottom waters in particular. Chlorophyll *a* concentrations were also elevated during this time and the summer-autumn peaks prior to 2007 likely correspond to major blooms of cyanobacteria (blue-green algae). A major transition appears to have occurred around 2007. Concentrations of DRP become close to detection limits in both surface and bottom waters whilst TP and TN concentrations generally declined and became less variable. There appear to be fewer 'spikes' in all nutrient species' concentrations and in chlorophyll *a*, and there appears to be less separation of surface and bottom nutrient species' concentrations. Of note in the last two years of the record (2011 – 2012) is a large seasonal variation in nitrate concentration which was less apparent (if present at all) in most other years. This variation was denoted by a progressive increase in nitrate concentrations through autumn, reaching a peak in early winter and then a rapid decline to very low concentrations in mid-summer.

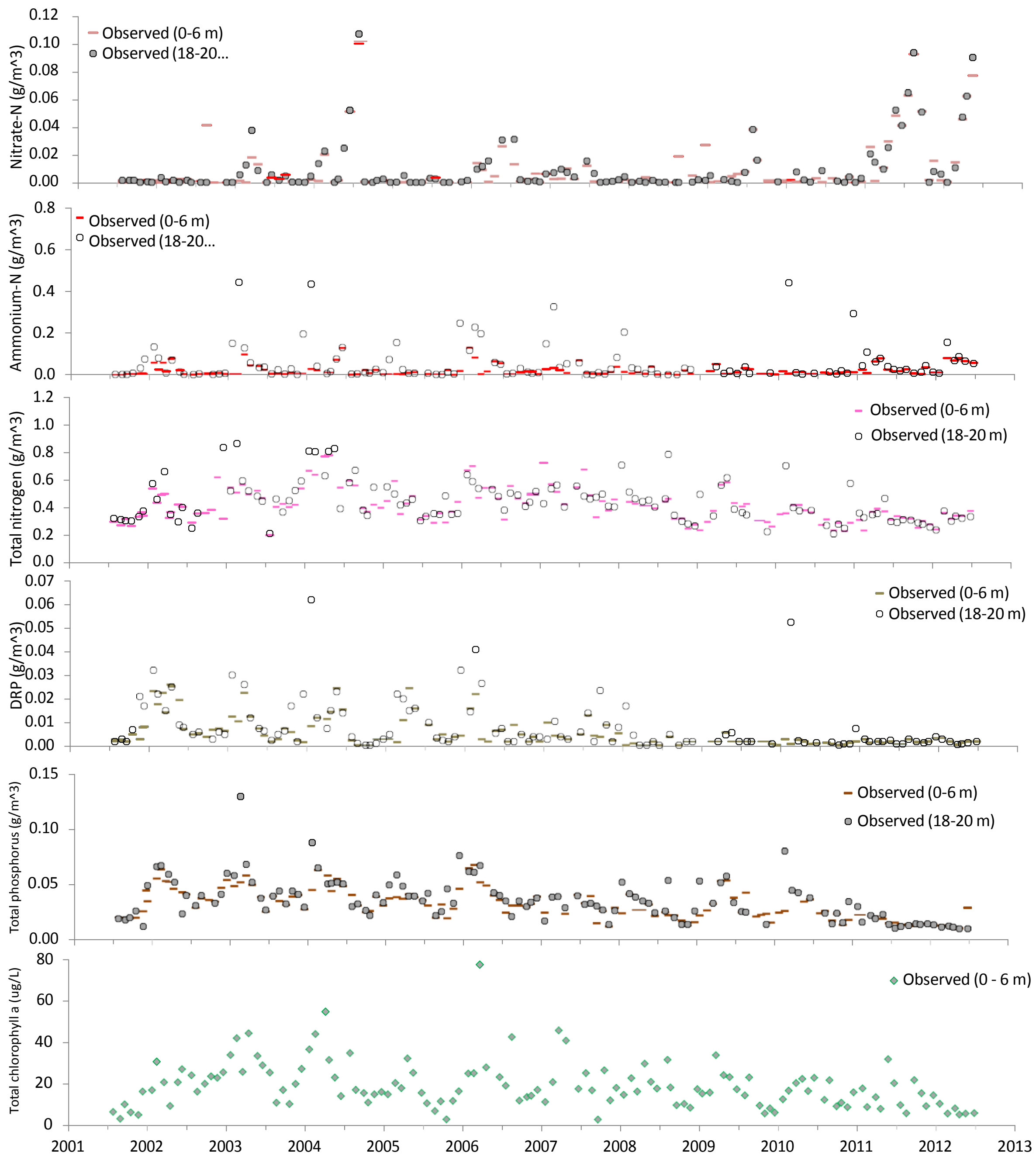

Figure 21: In-lake measurements for nitrate, ammonium, total nitrogen, dissolved reactive phosphorus (DRP), and total phosphorus at the surface (0 - 6m) and bottom (18 – 20m), and total chlorophyll a at the surface (0-6 m) from 2001 to 2013.

3.4 Model calibration and validation

Model calibration was undertaken from July 2004 to June 2007 and validation from July 2001 to June 2004 as shown for comparisons of measured and simulated data for plots of surface TP, TN and chlorophyll *a* (Figure 22) as well as surface and bottom temperature and dissolved oxygen (Figure 23). The calibration of DYRESM-CAEDYM is undertaken by adjusting a number of parameters that influence the performance of the hydrodynamic component of the model (DYRESM; Table 3) and the ecological component of the model (CAEDYM; Table 4). The calibration has been progressively refined since the first published study of the coupled model application to Lake Rotorua (Burger et al. 2008). The current statistical performance of the model may be considered as satisfactory compared with the earlier applications and relative to similar studies undertaken elsewhere (e.g. Arhonditsis and Brett 2004). Of particular note is the ability of the model to capture intermittent stratification events in Lake Rotorua, and to be able to capture the corresponding depletion of dissolved oxygen in bottom waters (Figure 23).

The target for the calibration was to obtain TLI values within 0.1 of observed values (note: both measured and modelled TLI values correspond to a TLI3 value, without Secchi depth included). Figure 24 indicates a satisfactory outcome with only small disparities between observed TLI and simulated values. Of note are the high TLI values of 2003 and 2004. These may appear as outliers in observation and also correspond to a period when there were major blooms of the cyanobacterium *Anabaena planktonica.*

Table 3. Assigned values for parameters used in DYRESM.

Parameter	Unit	Calibrated value	Reference/remarks
Critical wind speed	$m\ s^{-1}$	4.5	Spigel et al. (1986)
Emissivity of water surface	-	0.96	Imberger & Patterson (1981)
Mean albedo of water	-	0.08	Patten et al. (1975)
Potential energy mixing efficiency	-	0.2	Spigel et al. (1986)
Shear production efficiency	-	0.3	Spigel et al. (1986)
Wind stirring efficiency	-	0.23	Spigel et al. (1986)
Vertical mixing coefficient	-	500	Yeates & Imberger (2003)
Effective surface area coefficient	m^2	1.0×10^7	Standard value

Table 4. Assigned values for parameters used in CAEDYM for Lake Rotorua; DOPL and DONL are dissolved organic phosphorus and nitrogen, respectively.

Parameter	Unit	Calibrated value	Reference source
Sediment parameters			
Sediment oxygen demand	$g\ m^{-2}\ d^{-1}$	2.9	Schladow & Hamilton (1997)
Half-saturation coefficient for sediment oxygen demand	$mg\ L^{-1}$	0.5	Schladow & Hamilton (1997)
Maximum potential PO_4 release rate	$g\ m^{-2}\ d^{-1}$	0.04	
Oxygen and nitrate half-saturation for release of phosphate from bottom sediments	$g\ m^{-3}$	1.0	
Maximum potential NH_4 release rate	$g\ m^{-2}\ d^{-1}$	0.5	
Oxygen half-saturation constant for release of ammonium from bottom sediments	$g\ m^{-3}$	1.0	
Maximum potential NO_3 release rate	$g\ m^{-2}\ d^{-1}$	-0.1	
Oxygen half-saturation constant for release of nitrate from bottom sediments	$g\ m^{-3}$	1.0	
Temperature multiplier for nutrient release	-	1.05	Robson & Hamilton (2004)
Nutrient parameters			
Decomposition rate of POPL to DOPL	d^{-1}	0.01	Schladow & Hamilton (1997)
Mineralisation rate of DOPL to PO_4	d^{-1}	0.01	Schladow & Hamilton (1997)
Decomposition rate of PONL to DONL	d^{-1}	0.01	Schladow & Hamilton (1997)
Mineralisation rate of DONL to NH_4	d^{-1}	0.03	Schladow & Hamilton (1997)
Denitrification rate coefficient	d^{-1}	0.8	
Oxygen half-saturation constant for denitrification	$mg\ L^{-1}$	1.0	
Temperature multiplier for denitrification	-	1.08	
Nitrification rate coefficient	d^{-1}	0.1	
Nitrification half-saturation constant for oxygen	$mg\ L^{-1}$	1.0	
Temperature multiplier for nitrification	-	1.08	

Phytoplankton parameters		*Cyanophytes, Chlorophytes, Diatoms*	
Maximum potential growth rate at 20°C	d^{-1}	0.76, 1.28, 1.50	Robson & Hamilton (2004)
Irradiance parameter non-photoinhibited growth	µmol m^{-2} s^{-1}	200, 80, 15	Robson & Hamilton (2004)
Half saturation constant for phosphorus uptake	mg L^{-1}	0.008, 0.007, 0.007	Trolle et al. (2008)
Half saturation constant for nitrogen uptake	mg L^{-1}	0.03, 0.04, 0.04	Trolle et al. (2008)
Minimum internal nitrogen concentration	mg N (mg chl *a*)$^{-1}$	1.6, 2.0, 2.0	Schladow & Hamilton (1997)
Maximum internal nitrogen concentration	mg N (mg chl *a*)$^{-1}$	7.0, 8.0, 8.0	Schladow & Hamilton (1997)
Maximum rate of nitrogen uptake	mg N (mg chl *a*)$^{-1}$ d^{-1}	3.5, 3.0, 3.0	Schladow & Hamilton (1997)
Minimum internal phosphorus concentration	mg P (mg chl *a*)$^{-1}$	0.13, 0.2, 0.2	Schladow & Hamilton (1997)
Maximum internal phosphorus concentration	mg P (mg chl *a*)$^{-1}$	1.2, 0.8, 0.8	Schladow & Hamilton (1997)
Maximum rate of phosphorus uptake	mg P (mg chl *a*)$^{-1}$ d^{-1}	0.1, 0.2, 0.2	Schladow & Hamilton (1997)
Temperature multiplier for growth limitation	-	1.09, 1.05, 1.06	Schladow & Hamilton (1997)
Standard temperature for growth	°C	22, 19, 12	Gal et al. (2009)
Optimum temperature for growth	°C	32, 25, 20	Gal et al. (2009)
Maximum temperature for growth	°C	39, 35, 28	Gal et al. (2009)
Respiration rate coefficient	d^{-1}	0.07, 0.13, 0.135	Schladow & Hamilton (1997)
Temperature multiplier for respiration	-	1.05, 1.07, 1.07	Schladow & Hamilton (1997)
Fraction of respiration relative to total metabolic loss rate	-	0.7, 0.7, 0.7	
Fraction of metabolic loss rate that goes to DOM	-	0.3, 0.2, 0.2	
Constant settling velocity	m s^{-1}	1.2×10^{-5}, -0.05×10^{-5}, -0.35×10^{-5}	Burger et al. (2008)

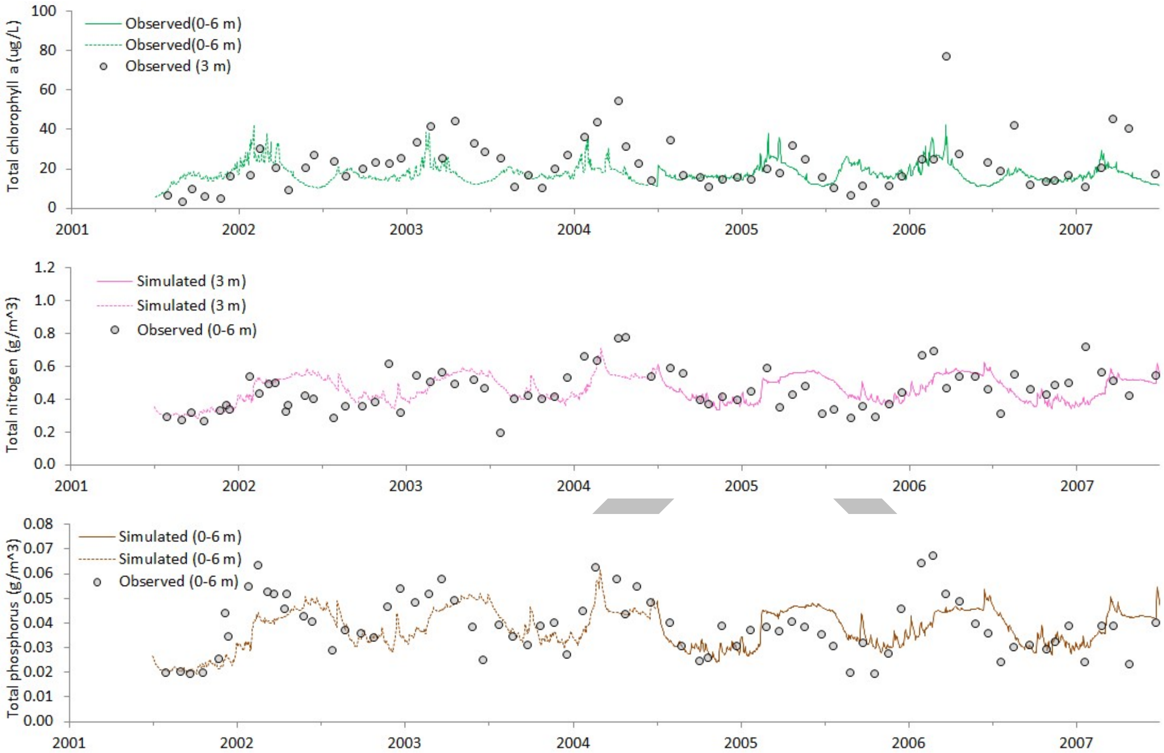

Figure 22. Comparison of model simulation results against field observations (black circles) in the surface (0 – 6 m) waters of Lake Rotorua during the calibration period (solid line) and validation period (dashed lines) for total chlorophyll *a*, total nitrogen and total phosphorus.

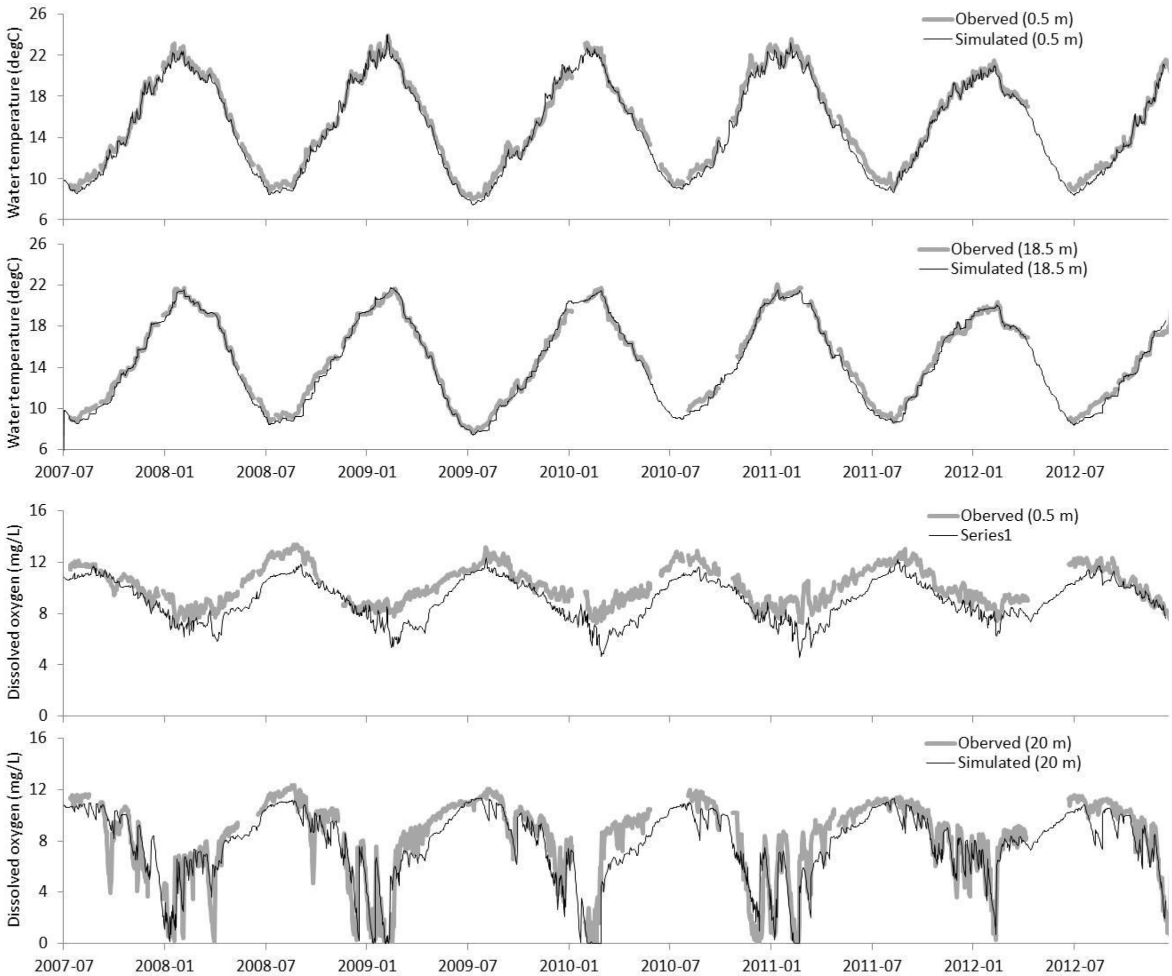

Figure 23. Comparison of model simulation (black line) results against high frequency observations (grey line) in the surface (0.5 m) and bottom (18.5 m) waters of Lake Rotorua for the period 2007 to 2012.

Table 5. Statistics for model performance against field observations for the calibration (2004-2007) and validation (2001-2004) periods. For Pearson correlation coefficient (R), mean absolute error (MAE), root mean square error (RMSE), and RMSE normalised by standard deviation of field observations (NRMSE).

		Calibration (2004 - 2007)				Validation (2001 - 2004)			
		R	MAE	RMSE	NRMSE	R	MAE	RMSE	NRMSE
NO3	0 m	0.405	0.009	0.017	0.953	0.184	0.009	0.014	1.254
	15 m	0.351	0.009	0.019	0.974	0.351	0.009	0.019	0.974
	19 m	0.294	0.010	0.020	1.011	0.182	0.009	0.014	1.241
NH4	0 m	0.200	0.017	0.029	1.015	0.168	0.020	0.032	1.027
	15 m	0.329	0.035	0.053	0.968	0.329	0.035	0.053	0.968
	19 m	0.265	0.052	0.084	1.004	0.402	0.057	0.099	0.932
TN	0 m	0.095	0.103	0.126	1.121	0.450	0.091	0.122	0.901
	15 m	0.320	0.098	0.118	1.035	0.320	0.098	0.118	1.035
	19 m	0.040	0.091	0.109	1.157	0.387	0.141	0.174	0.910
PO4	0 m	0.117	0.004	0.006	1.085	0.175	0.007	0.009	1.251
	15 m	0.152	0.006	0.010	1.052	0.152	0.006	0.010	1.052
	19 m	0.274	0.007	0.010	0.989	0.393	0.010	0.017	1.016
TP	0 m	0.317	0.008	0.011	1.009	0.573	0.009	0.011	0.891
	15 m	0.320	0.009	0.012	0.963	0.320	0.009	0.012	0.963
	19 m	0.294	0.010	0.013	0.962	0.398	0.014	0.021	0.939
Tchl *a*	0 m	0.482	8.806	12.552	0.896	0.222	10.557	13.123	1.088

Figure 24. Comparison of modelled lake TLI3, and observed TLI3 for Lake Rotorua, for the calibration (July 2004 to June 2007) and validation (July 2001 to June 2004) periods. The dashed red line is the TLI3-adjusted target for Lake Rotorua (TLI3 = 4.32).

3.5 Effects of alum dosing

Comparisons of observed TLI3 and values simulated for two scenarios with and without alum dosing are shown in Figure 25. The departure of ± 0.1 TLI3 units from the observations are shown for consistency with the target for the calibration. For the closest possible representation by the model of the observed case, we used observed DRP (and other nutrient species') concentrations for the Utuhina Stream because the monitoring station for this stream was below the point of alum dosing. For the Puarenga Stream where the monitoring station was upstream of the dosing point, we used an alum dose – DRP response relationship from the data of McIntosh (Figure 20) to estimate DRP downstream. Because at least some active component of the alum (i.e., Al^{3+}) was almost certainly entering the lake via the stream inflows, we made an assumption that there was some in-lake effect from the alum. To represent this effect, and in the absence of explicitly modelling the dynamics of alum, we increased the rate of sedimentation of particulate matter and reduced the release of DRP from the bottom sediments (see Table 2 for the parameter value adjustments associated with these assumptions). The interannual variability was not as well captured by the model for this 'scenario period' and TLI3 values were overestimated compared with observations in the last two years (2011 – 2012) and slightly underestimated in the first year (2009). We attribute these deviations to the fixed parameters used in the CAEDYM model to represent the effect of alum dosing (i.e. organic matter sedimentation rates and DRP sediment releases) whilst in practice alum dosing was highly variable on a day-to-day basis and in particular had been substantially increased in the period of 2011 – 2012 (see Figure 15). In this instance, because the model parameters were not dynamic the outcome of the simulations was an average TLI3 value over the four years that was similar to the observed one.

Simulations for the same period, but without the effects of alum dosing on stream phosphorus loads and in lake processes, resulted in a TLI3 approximate 0.5 units higher than for the simulation without alum. This suggests that the reduction of TLI in Lake Rotorua through the period of 2008 to 2012 may not have occurred without the dosing of alum, and thus the improved water quality may be attributable, at least in part, to its effects.

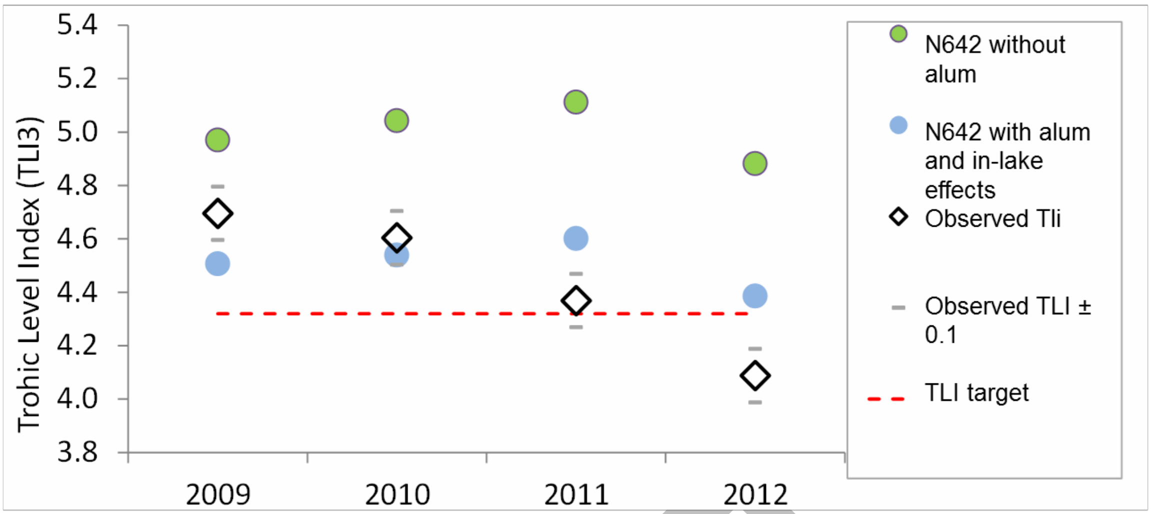

Figure 25. Comparison of observed Lake Rotorua Trophic Level Index (TLI) with simulated TLI for a scenario without any inflow alum dosing (green dots), and of alum dosing with in-lake effects (blue dots) including increased sedimentation of organic particulate matter (flocculation) and suppression of internal DRP release (50% reduction). The dashed red line is the TLI3-adjusted target for Lake Rotorua (TLI3 = 4.32).

3.6 Scenarios - effect on TLI

Figure 26 presents simulation results for a range of scenarios, encompassing possible catchment nutrient loads, alum use or absence, and additional measures to reduce the phosphorus load to the lake. For a description of the scenarios and the associated parameter adjustments, refer to Table 2. For a scenario of increased nutrient load and without any dosing of alum or phosphorus load mitigation (N730), TLI3 was substantially higher than for any other scenario (average TLI3 = 5.57) representing highly impacted water quality relative to targets for Lake Rotorua. Reduction of the catchment nitrogen load to an annual average of 435 t N yr^{-1} resulted in a corresponding reduction in TLI3, to the point where the TLI3 target was almost attained for the period July 2011 to June 2012.

Importantly, only those scenarios where alum dosing was included consistently met or bettered the TLI3 target for Lake Rotorua. In order to meet the TLI3 target whilst maintaining a nitrogen load of 730 t N yr^{-1}, it was necessary to include in the simulations a reduction of catchment TP and DRP loads of 25% (to represent on-land mitigation measures), and a transfer of 50% of the DRP load to TP load in inflows (representing alum dosing of at least half the total DRP load to Lake Rotorua, i.e. multiple inflows). It is important to note that reductions in P inputs to the lake to the extent described above may be difficult to attain in practice (see discussion). A TLI3 well below target levels was simulated by maintaining the current nitrogen load but employing the full range of P reduction measures (scenario N642_Pmit+Almax.Floc.intP; phosphorus external load mitigation and alum treatment effects on DRP in inflows, elevated in-lake flocculation of particulate organic material and reduced release of phosphate from the lake sediments).

Driving the reduction in TLI3 simulated by those scenarios including P mitigation and/or alum dosing is a shift in the simulated system from co-limitation of phytoplankton by N and P to a highly P-limited state. Some evidence of a shift of this nature was apparent in the record of field observations from Lake Rotorua after 2010, specifically very low DRP concentrations and a relatively high concentration of nitrate in surface waters for much of the subsequent period (Figure 21).

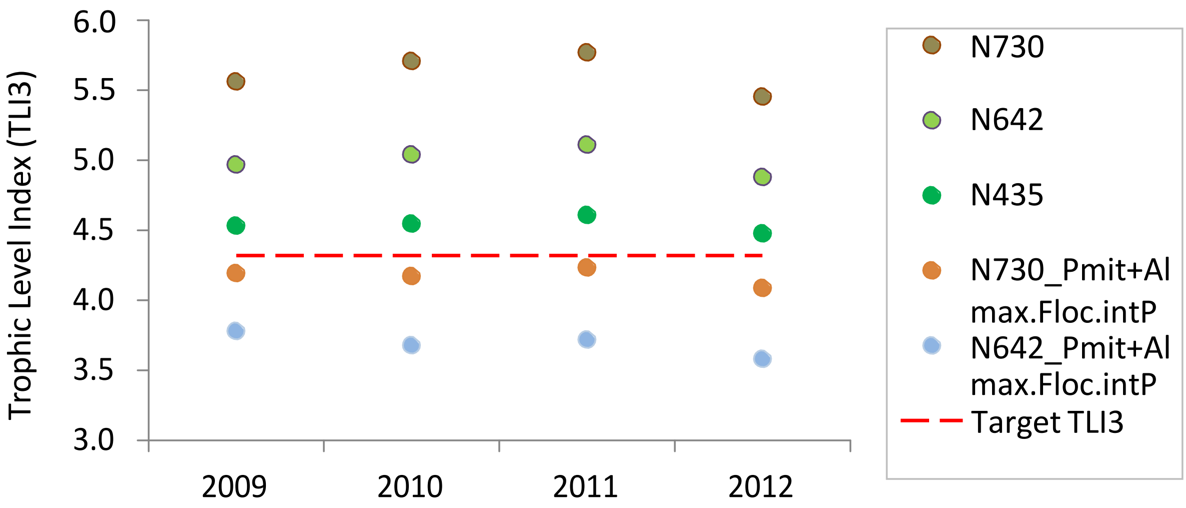

Figure 26. Yearly TLI values for scenario simulations. Refer to Table 2 for an explanation of each scenario name. The dashed red line is the TLI3-adjusted target for Lake Rotorua (TLI3 = 4.32).

Table 6. Annual average and long-term average TLI (2008 – 2012) for nutrient load and alum scenario simulations.

Year to (Jul-Jun)	N730	N642	N642_Al	N642_Pmit +Almax. Floc. intP	N435	N730_Pmit+Almax.Floc.intP	Target
2008	5.33	4.86	4.86	4.18	4.65	4.44	4.32
2009	5.56	4.97	4.97	3.78	4.53	4.19	4.32
2010	5.71	5.04	5.04	3.68	4.55	4.17	4.32
2011	5.77	5.11	5.11	3.72	4.61	4.23	4.32
2012	5.46	4.88	4.88	3.58	4.48	4.09	4.32
Average	**5.57**	**4.97**	**4.97**	**3.79**	**4.56**	**4.23**	**4.32**

4. Discussion

In-lake measurements indicate that there has been a substantial reduction in surface and bottom phosphorus concentrations from 2008 to 2013 (compared with the period between 2001 and 2007). At the same time, there has also been a reduction in chlorophyll *a*, and the annual TLI reached the BoPRC target of 4.2 in 2011-12. Reductions in water column phosphate and chlorophyll *a* concentrations have taken place despite indications that both DRP and TP concentrations in lake inflows have remained static, or in several cases appear to have increased (e.g. Waiteti Stream). The exception is of course the Utuhina Stream which was sampled below the alum dosing station and in which DRP decreased dramatically since alum dosing. By contrast TP concentrations have remained reasonably constant, indicating that DRP transitions to a particulate form with alum dosing but the particulate P remains in suspension due to turbulence in the stream. An earlier study showing slightly elevated levels of aluminium in bottom sediments adjacent to the Utuhina Stream indicates that this relatively quiescent area may be where a substantial component of the alum floc initially settles. There is, however, a possibility that deposited alum floc is resuspended and transported more widely within the lake (e.g. during strong winds generating wave action that resuspends sediment in the shallow near-shore area).

The original intent of the alum dosing was to 'lock up' the phosphorus contained in the Utuhina and Puarenga streams (J. McIntosh, pers. comm.) but the evidence from this report suggests a more extensive mode of action. Based on data for the Utuhina Stream (Fig. 17), there appeared to be a threshold of action at dose rates of around 100 kg Al day^{-1}, at which point almost all of the DRP was removed. Similarly dose rates above about 75 kg Al d^{-1} appeared to lock up almost all of the DRP in the Puarenga Stream, as indicated by very low DRP concentrations during alum dosing, whilst TP remained largely invariant. Dose rates above the respective 'thresholds' in these streams would presumably allow free aluminium to enter the lake and flocculate phosphorus in-lake as well as in-stream. In the Puarenga Stream this threshold corresponded to a mass ratio of Al: PO_4-P of approximately 20: 1.

Considering the relatively low levels of DRP in the lake compared with the two stream inflows above the respective dosing plants, it is possible that the mode of action of the aluminium may be slightly different between the inflows and the lake. Flocculation of organic material and subsequent sedimentation may increase under the relatively quiescent conditions in the lake and there is evidence for this effect in the marked decreases in both TP and TN after 2010, when high rates of alum dosing were undertaken in the streams. Our model simulations, in which we removed DRP in the inflows according to observations below the dosing plants (without transformation to TP because we considered it to be 'locked up'), accounted for little of the observed in-lake reductions in TP, DRP and TN, particularly post 2011. Increases in the rate of organic matter sedimentation, as well as a decrease in anoxia-generated phosphorus release, were required to achieve a 'satisfactory' replication of the observed TLI. High-frequency dissolved oxygen measurements from 19 m depth at the lake buoy site show a substantial decrease in the frequency of anoxia over the past 2-3 years (see Figure 23 as well as reinforcement by more recent data not shown in this report). The consistency of this reduction suggests that it is unlikely to be solely due to favourable (windy) weather conditions in recent years. We suggest that alum has either indirectly or directly altered the composition of the bottom sediments in a way that has resulted in lower rates of oxygen consumption and lower rates of phosphorus release. The direct mode of action may be due to changes in the chemical composition of bottom sediments due to alum floc deposition (with lower rates of oxygen consumption than the basal sediments) while the indirect mode of action may be linked to overall improvements in lake trophic status and reductions in the rate of organic matter deposition generally.

The DYRESM-CAEDYM model used in this study was not adapted specifically to dynamically simulate alum effects according to the daily rate of dosing. Instead we applied a rate of sedimentation for organic material which did not vary with alum dosing rate but was higher than for the period prior to alum dosing. Similarly, we applied a rate of sediment phosphorus release which was not varied with alum dosing but was lower than before alum dosing. Alum dosing rates in the two stream inflows have varied considerably since commencement of dosing in the Utuhina Stream in 2006 (Figure 23) but an especially large increase occurred in 2011, and there was high variability of dosing in this year. The lack of a dynamic response in the model to account for variations in alum dosing helps to explain why our simulated TLI in 2011-12 was higher than the observed values, while in 2009-10 it was lower; i.e., the parameters which were altered to account for the occurrence of alum dosing (i.e. organic matter sedimentation rate and sediment phosphorus release) were not varied with the rate of alum dosing. If the model is to continue to be used for making detailed evaluations of the effect of alum dosing of Lake Rotorua then it will be important for model development to be undertaken to specifically simulate alum dynamics, and to provide detailed in-lake sampling with which to validate the model.

It might now be questioned how long the beneficial effects of alum dosing would persist with cessation of dosing (e.g., if the 2017 resource consent for its use was rescinded). The water column impacts may be expected to dissipate reasonably quickly from the combined effects of sedimentation and flushing. The hydraulic residence time in Rotorua is 1.5 years, so it is likely that sedimentation would be the major mechanism leading to loss of alum impacts rather than flushing. Given that following some period of sedimentation the concentration at time t (C_t) may be given by:

$$-$$

$$(\qquad)$$

where C_i is the initial concentration, v_s is the sedimentation rate, t is time and z is water depth, it may be possible to make an approximation for the time scale on which alum persists. Assuming a conservative value (i.e., likely much lower than observed) sedimentation rate of 0.3 m d^{-1} for alum flocs and a mean depth of 10 m for Lake Rotorua, around 40% of the alum would persist after 30 days and about 7% after 90 days. Any more persistent effects (i.e., >90 days) are therefore likely to be associated with the way in which deposited alum flocs alter the bottom-sediment composition. Based on the progressive reduction in the rate of oxygen consumption in the bottom sediments over 2-3 years following intensive stream alum dosing, we hypothesise that any legacy effects of alum dosing may persist over a 2-3 year duration following termination. Only with an experimental approach involving prolonged periods of removal of alum dosing (and the implications of such an approach on lake water quality) or a mandatory cessation of alum dosing imposed by the absence of a resource consent, will it be possible to gain a very complete understanding of the longevity of its effects.

Bay of Plenty Regional Council has now adopted a process control approach to alum dosing based on in-lake TP concentrations. Alum dosing in the streams is increased or decreased when the surface TP concentration is greater or less than 20 mg m^{-3}, respectively. This mode of operation appears to be reasonably successful in maintaining a relatively constant concentration of TP, with concentrations somewhat higher than during some of the very high dosing rates of 2011 but on average around one-half of the very high levels observed in the lake in the mid-2000s. The mid-summer peaks of TP prior have also been absent under the current dosing regime, with much less seasonal variability in TP and chlorophyll *a* concentrations (Figure 21).

The Bay of Plenty Regional Council has undertaken studies and supported data collection to determine potential acute and chronic impacts from alum dosing. These include Utuhina Stream fish and aquatic invertebrate surveys (e.g. Ling and Brijs 2009), surveys of aluminium concentrations in bottom sediments (Özkundakci et al. 2013) and high-frequency measurements of pH from the mid-lake buoy. There are no obvious chemical or biological effects from alum dosing except for an area of enrichment of Al at the Utuhina Stream mouth in the lake. The pH is of particular interest because Al hydrolysis produces acidity that has in some cases led to whole-lake acidification events and fish kills. pH in Lake Rotorua does not appear to have been affected by alum dosing and remains in the range 6.5 to 8. This range corresponds to where alum is most effective in its hydrolysis and flocculation processes for phosphorus removal. Perhaps of more concern is whether high phytoplankton biomass could elevate pH through draw down of dissolved inorganic carbon concentrations (i.e., by reducing natural production of carbonic acid when bicarbonate is reduced to low concentrations in the water column as a result of high productivity). In Lake Okaro the spring bloom is characterised by a rapid increase in pH (from c. 7 to 10 within one month; C. McBride pers. obs. from high-frequency lake buoy) and the very high pH likely fuels the bloom further by destabilising metal cation (e.g., Al, Fe and Mn) binding of phosphorus, releasing phosphate to the overlying water (see Gao et al. 2012). This occurrence appears relatively unlikely in Lake Rotorua given the greater stability of pH. We recommend that pH continue to be monitored at high frequency in Lake Rotorua and that values are examined closely in relation to both alum dosing rates and variations in phytoplankton biomass. We also recommend that other good indicator biota be analysed for aluminium concentrations in tissue samples. These biota could include filter-feeding kākahi (*Echyridella menziesi*) and generalist feeders such as koura (*Paranephrops planifrons*).

Of considerable importance is whether alum dosing brought about a transition in nutrient limitation status of phytoplankton in Lake Rotorua. In recent times nutrient limitation has been measured in three different studies; in 2004-5 by Burger et al. (2007b), in 2010 by Mead (pers. comm.) and in 2012 by Abell et al. (2014). In these studies nitrogen + phosphorus additions have consistently been found to have the greatest growth-stimulation effect on phytoplankton. Burger et al. (2007b) found a much greater phytoplankton growth response to nitrogen than in earlier studies of the 1970s (White 1977) but Abell et al. (2014) found that nitrogen tended to elicit the greatest growth response on three separate occasions in December 2012. The timing of the study by Abell et al. (2012) is noteworthy. It was in a period when there was a trough in nitrate concentration compared with periods before and after the study (see Fig. 21a). This may help to explain why they found dominance of N limitation during this period of alum dosing. We hypothesise that had the study been conducted during adjacent periods of much higher nitrate concentration, then P limitation would have been dominant. The relatively high peaks in nitrate concentrations in 2011-12 are of interest and suggest that demand on dissolved inorganic nitrogen by phytoplankton was lower. This is consistent with phosphorus concentrations being reduced to limiting levels and allowing a release from nitrogen limitation – expressed as the increases in nitrate concentration in the lake water. In recent times there has been much speculation about limiting nutrients (specifically N and P) in freshwaters with a view that one nutrient may potentially be controlled to limiting levels whilst the other is less stringently controlled. Even with alum dosing, Abell et al.'s (2014) study suggests that there are locations and periods in Lake Rotorua with different limiting nutrient(s). Some of the hypotheses expressed widely in New Zealand about controlling only one nutrient to limit primary production in freshwaters are not well supported by direct measurements of nutrient limitation (e.g. using bioassays) and in fact there are remarkably few direct measurements (see Abell et al. 2010). We recommend that Bay of Plenty Regional Council considers a regular programme for monitoring nutrient limitation, at least in lakes where alum dosing is being undertaken.

5. Recommendations

Here we summarise some of the recommendations that were mentioned in the discussion section. These recommendations apply specifically to programmes of monitoring that should be considered by the Water Quality Technical Advisory Group and the Lakes Strategy Group for the Rotorua lakes.

1. We recommend that consideration be given to wider use of biota as an assessment tool for monitoring potential chronic effects from alum in the lake. Tissue sampling could be conducted on key species (e.g. kākahi, koura and trout) to encompass a broad representation of different feeding strategies and species of cultural and recreational significance.

2. We recommend that a regular, repeatable monitoring protocol be adopted for determining phytoplankton nutrient limitation in Lake Rotorua. Analysis of the data should include considerations of alum dosing rates, concentrations of inorganic and total nutrients, and time of year in relation to phytoplankton composition.

3. We recommend close examination of pH from the high-frequency lake monitoring buoy to better understand its variability and the possibility of any untoward consequences from relationships between alum dosing, phytoplankton biomass and pH variations.

4. We recommend that increased frequency of in-lake measurements of Al be complemented with development of a dynamic module for simulating Al concentration in DYRESM-CAEDYM, should this model continue to be used to provide key information on the effectiveness of alum and to generate hypothetical scenarios (e.g., if alum dosing was not undertaken).

5. Sediment oxygen demand over stratification events should be calculated from high-frequency lake buoy data. The most recent six-year period of high-frequency monitoring as well as data collected in 2004-5 can be used to examine any conspicuous trends and potential correspondence to alum dosing.

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