Lake Rotorua Treated Wastewater Discharge: Environmental Effects Study Draft Report

2015

ERI Report XX

Draft client report prepared for Rotorua Lakes Council

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Cover: View eastwards across Lake Rotorua.

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EXECUTIVE SUMMARY

[To be completed.]

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1. INTRODUCTION

The Environmental Research Institute, University of Waikato led an environmental effects study of proposed options for discharging treated wastewater to Lake Rotorua, Bay of Plenty. The study is intended to inform a decision–making process to resolve how treated municipal wastewater should be discharged after 2019, when irrigation operations at the Land Treatment System in the Whakarewarewa Forest are scheduled to cease. The process is being led by a Project Steering Committee appointed by the Rotorua Lakes Council. Based on the outcomes of this decision making process, the Steering Committee will recommend a preferred disposal option to Rotorua Lakes Council that will be subject to a separate Assessment of Environmental Effects following preliminary design (RLC 2014).

The environmental effects study involved undertaking mass balance calculations and environmental modelling to examine water quality effects associated with the proposed options for treated wastewater discharge. The proposed options included either discharge directly to the lake, or to the lower reach of the Puarenga Stream. As such, potential effects to both of these receiving waters were considered.

2. BACKGROUND AND OBJECTIVES

2.1. Lake Rotorua

2.1.1. Background

Lake Rotorua (Map 1) is nationally iconic and represents an important resource for Rotorua, supporting a range of recreational opportunities that attract tourists to the region. The lake is highly valued by Māori, and the lake is of particularly high cultural significance to Te Arawa who are the legal owners of the lake bed.

The lake is large (≈ 80 km^2) and volcanically–formed. As a consequence of its relatively shallow depth (mean depth ≈ 10 m), the lake is polymictic and only stratifies continuously for periods of up to a few weeks during calm conditions in summer months (November–March). Since the 1960s, Lake Rotorua has experienced water quality problems associated with eutrophication (Fish 1969; Rutherford 1984; Rutherford *et al.* 1989; Burns 2009). This is the process of increased productivity caused by excessive inputs of nutrients that promote growth of plants, including both phytoplankton (microscopic plants suspended in the water column) and macrophytes (larger aquatic plants). The primary nutrients of concern are nitrogen and phosphorus. Symptoms of eutrophication include: reduced water clarity; depleted dissolved oxygen concentrations in bottom waters; unsightly blooms of cyanobacteria that may produce toxins; odours, and; extirpation of species that are adapted to less productive waters (Carpenter *et al.* 1998). The primary metric used by Bay of Plenty Regional Council (BoPRC) to monitor trophic status is the Trophic Level Index (TLI), which integrates annual mean measurements of Secchi depth and concentrations of total nitrogen, total phosphorus and chlorophyll *a* (Burns *et al.* 1999).

In response to public dissatisfaction with water quality, Lake Rotorua has been identified as a national priority for restoration (Parliamentary Commissioner for the Environment 2006). In 2008, the Ministry for the Environment committed NZ$72.1 million towards improving water quality in Lake Rotorua and three other priority lakes. This funding was subsequently matched by BoPRC and Rotorua Lakes Council (RLC). The Lakes Rotorua and Rotoiti Action Plan (BoPRC 2009) outlines actions to achieve the Lake Rotorua water quality objective of an annual TLI of 4.2, which corresponds to the lower end of the eutrophic range (4–5; Burns *et al.* 1999). A range of actions is underway and an improvement in water quality has occurred in recent years relative to the early– and mid–2000s (Figure 1), which were characterised by frequent blooms of cyanobacteria during summer and autumn (Abell *et al.* 2012; Hamilton *et al.*, 2015). As a result, annual TLI since 2011 has either been achieved or been very close to the target (Figure 1). This improvement has occurred in association with operations to dose aluminium sulphate (alum) near the mouths of the Utuhina and Puarenga streams, two major stream inflows to the lake. Aluminium ions in alum chemically bind with phosphate, removing it from the water column and thereby reducing the amount of phosphorus that is available for primary production. Dosing has been undertaken on a near–daily basis since operations began in the Utuhina Stream in mid–2006, with dosing also undertaken in the Puarenga Stream since 2010. Recent modelling work has shown that the TLI target would have been exceeded in recent years without the application of alum (Hamilton *et al*. 2015). Furthermore, this work indicates that alum is not only reducing dissolved reactive phosphorus concentrations in the inflows, but is also causing further phosphorus flocculation in the lake as excess alum is transported downstream of the dosing plant.

Map 1

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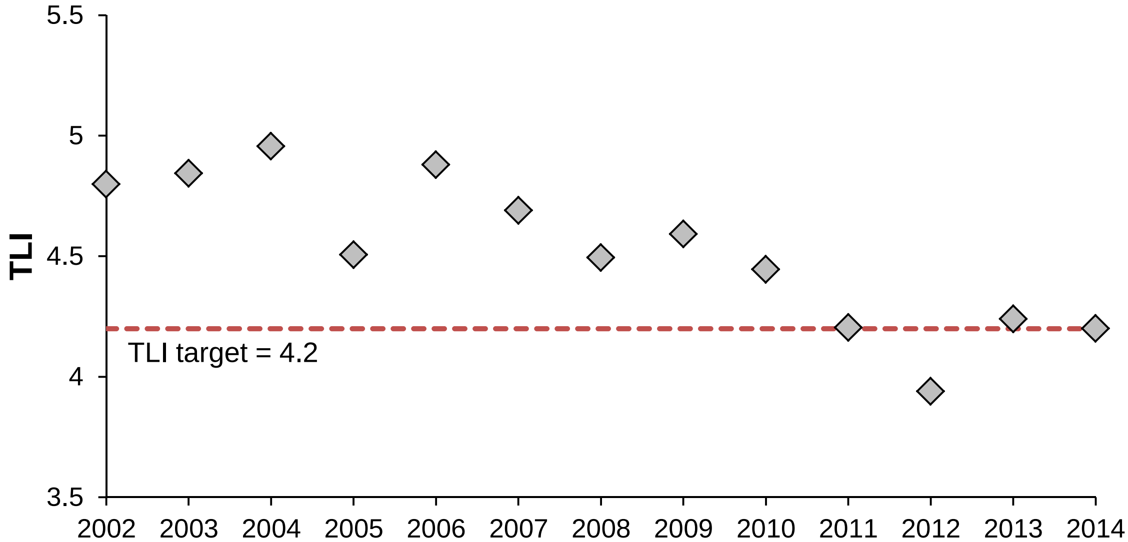

Figure 1 **Annual Trophic Level Index of Lake Rotorua.** Data for 2002–2012 are based on surface water samples only and thus values may differ slightly from by those used for BoPRC monitoring. Sources: 2002–2012 data (Abell et al. 2012); 2013 datum (Rotorua Te Arawa Lakes Programme 2014); 2014 datum (http://www.rotorualakes.co.nz/lake_rotorua_facts, accessed 29 May 2015).

2.1.2. Wastewater discharge in the lake catchment

Prior to the 1990s, municipal wastewater was discharged to the lake, contributing a significant source of nitrogen and phosphorus. Sewage–derived inputs were attributed to periods of water quality decline in the 1970s and 1980s (Rutherford 1984; Rutherford *et al.* 1989), with sewage inputs contributing to accumulation of nutrients (particularly phosphorus) in the bed sediments, in association with inputs from other sources such as farmland. These accumulated nutrients contribute to internal loading as they are recycled within the water column, particularly during stratified periods in the summer (White *et al.*, 1978; Burger *et al.*, 2007). The magnitude of such internal loads of nitrogen and phosphorus is comparable to external loads from the lake catchment (Burger *et al.*, 2007).

In 1991, discharge of treated municipal wastewater from Rotorua Wastewater Treatment Plant (WWTP) to the lake ceased. Instead, spray–irrigation of treated wastewater commenced at the Land Treatment System (LTS), located in the Whakarewarewa Forest to the south of the lake (Map 1). The forest is in the Waipa Stream catchment, which is a tributary of the Puarenga Stream. Rotorua Lakes Council currently has a Resource Consent to discharge 30 tonnes of nitrogen and three tonnes of phosphorus per annum via the LTS. Monitoring of the Waipa Stream shows that nitrogen loads frequently exceed the consent limit by a moderate amount, while phosphorus loads are typically well within the limit. Mean five-year loads for 2007–2011 were 35 t N/y and 1.7 t P/y (A. Lowe, pers. comm. 2013). Monitoring of the Puarenga Stream 2 km upstream of the lake since 1992 shows that dissolved inorganic nitrogen concentrations steadily increased over a period of approximately 10 years since operations began at the LTS, with current concentrations (~0.95 mg N/L) approximately 2.5–fold greater than those measured in 1992–1993. Compared with nitrogen, base flow phosphorus concentrations have remained relatively consistent in the Puarenga Stream, and have not exhibited a marked increase in response to the LTS operations.

2.1.3. Proposed options

The current Resource Consent for the LTS expires in 2021 and Rotorua Lakes Council (RLC) are examining the use of an alternative wastewater disposal system. The proposed system involves various options of discharging treated wastewater directly to receiving waters (Mott Macdonald 2014). The options involve permutations of different:

1) enhancements to wastewater treatment;
2) wastewater discharge locations;
3) discharge arrangements.

Six options for enhanced wastewater treatment are proposed. The options involve varying grades of treatment to enhance the removal of nitrogen and phosphorus from the wastewater relative to current treatment performance (Table 1).

The potential treated wastewater discharge locations are:

1) in the lower reach of the Puarenga Stream;
2) along the shore of Lake Rotorua close to the mouth of the Puarenga Stream;
3) on the bed of the lake.

Six potential locations for discharge to either the Puarenga Stream or the lake shore have been identified (Map 2). No specific location has been identified for the option of lake bed discharge.

The potential discharge arrangements under consideration are:

1) rock passage to direct discharge;
2) wetland;
3) rapid infiltration beds (RIB);
4) riparian/gabions;
5) pond.

Table 1 Proposed tertiary treatment options. Adapted from Table 1.1 in Mott MacDonald (2014).

Option	Description	Sub-options	Details
1	Base option	-	Upgrades to current tertiary treatment by addition of: flow balancing, P removal with chemical addition (alum) and UV disinfection.
2	Base option + basic filtration	i. Disk filter ii. Sand filter iii. Membrane filter	Addition of filtration to remove solids, including particulate N and P.
3	Base Option + filtration + denitrifying filter/bed	i. Denitrifying sand filter ii. Sand filter + denitrifying carbon bed	Addition of filtration to remove solids, in addition to final denitrification step to convert dissolved inorganic N to atmospheric N gas.

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2.1.4. Objectives of this study

The aim of this study is to assess the effects of the proposed wastewater discharge options on the water quality of Lake Rotorua and the lower reach of the Puarenga Stream. Specifically, the study examines:

- the potential instream ecological effects of discharging treated wastewater to the lower reaches of the Puarenga Stream;
- the potential effects of the proposed options on the trophic status of Lake Rotorua over multiple years;
- how mixing processes may affect how treated wastewater is transported and dispersed throughout the lake, depending on the discharge location.

3. METHODS

3.1. Overview

Three main techniques were used to inform the assessment:

1) *Mass balance calculations*. Nutrient loads were estimated for the treatment options. These were compared with estimated background loads in the Puarenga Stream to quantify how loads in the stream are expected to change, and to inform assessment of potential in–stream effects of nutrient enrichment. Estimated loads were subsequently used as forcing data to 'drive' the water quality model introduced below.

2) *One–dimensional (1–D) lake modelling*. A numerical model was configured to simulate the water quality effects of discharging treated wastewater, relative to a baseline period that represents current conditions. The lake was conceptualised as a single vertical profile in the model, i.e., vertical differences in water quality were modelled but horizontal variations were not. This 1–D assumption permitted lake processes to be sufficiently simplified so that potential effects on lake trophic status over time scales of multiple years could be examined.

3) *Three–dimensional (3–D) lake modelling*. A 3–D hydrological model was configured to examine the mixing processes that control how simulated treated wastewater inputs are transported within the lake.

3.2. Mass balance calculations to estimate in–stream nutrient loads and concentrations

3.2.1. Treated wastewater nutrient loads

The nutrient loads associated with each proposed treatment option were calculated based on information provided by Mott Macdonald (2014; Table 2). The predicted composition of the wastewater reflects upgrades to current tertiary treatment processes at the WWTP that will result in a range of improvements to the final wastewater quality. Predictions of treated wastewater composition are based on a 'combined' stream that integrates outputs from both the Bardenpho and membrane bioreactor systems. Details of any temporal variability in either wastewater discharge or composition were not provided, and therefore the assessment was based on the assumption that wastewater composition remains constant.

Table 2 Predicted final treated wastewater composition associated with each tertiary treatment option. Adapted from Table 9.1 in Mott MacDonald (2014).

Option	Sub-option	Discharge (ML/d)	Final effluent composition (mg/L)							
			TP	DRP	PP	TN	PON	DON	NO_3-N	NH_4-N
Option 1 (base option)		23.81	0.72	0.10	0.62	5.44	1.07	1.09	2.99	0.29
Option 2 (base + basic filtration)	i. Disc filter	23.81	0.37	0.10	0.27	4.86	0.49	1.09	2.99	0.29
	ii. Sand filter	23.81	0.20	0.10	0.10	4.62	0.25	1.09	2.99	0.29
	iii. Membrane filter	23.81	0.10	0.10	0.00	4.37	0.00	1.09	2.99	0.29
Option 3 (base + basic filtration + denitrifying filtration)	i. Denitrifying sand filter	23.81	0.20	0.10	0.10	2.63	0.25	1.09	1.00	0.29
	ii. Sand filter + denitrifying carbon bed	23.81	0.20	0.10	0.10	3.63	0.25	1.09	2.00	0.29

3.2.2. Puarenga Stream background nutrient loads

Nutrient loads in the Puarenga Stream were estimated for the baseline period of 2007 through 2014. Reasons for selection of this baseline period are discussed in Section 3.3.3 below.

3.2.2.1. Discharge

Discharge data for the Puarenga Stream were provided by BoPRC. Data for the period 2007 through 2010 were collected at the FRI gauge situated 2.1 km upstream of Lake Rotorua. Data for the period 2011 through 2014 were collected at the SH 30 gauge situated 0.8 km further downstream. There are no tributaries between the gauges and the data from the two sites were considered directly comparable. Discharge was recorded every 15 minutes (see BoPRC 2007 for quality assurance details). Measured data were available for 98.2% of the monitoring period (Table 3). All gaps in the record were filled using the following linear relationship (r^2 = 0.75, $RMSE$ = 0.62 m^3/s):

$$Q_{Puarenga} = 1.01 \cdot Q_{Utuhina} + 1.1301$$

where $Q_{Puarenga}$ is mean hourly discharge (m^3/s) in the Puarenga Stream and $Q_{Utuhina}$ is mean hourly discharge (m^3/s) in the Utuhina Stream, measured at the Depot Street gauge. Data are shown in Figure 2.

Table 3 **Proportion of time (%) when discharge measurements are not available for the Puarenga Stream.**

Year	%	Gaps > 1 day
2007	2.3	~3 days (July), ~2 days (September)
2008	11.5	~27 days (July), ~4 days (September)
2009	0.1	
2010	0.0	
2011	0.0	
2012	5.5	~15 days (July/August)
2013	0.0	
2014	0.0	

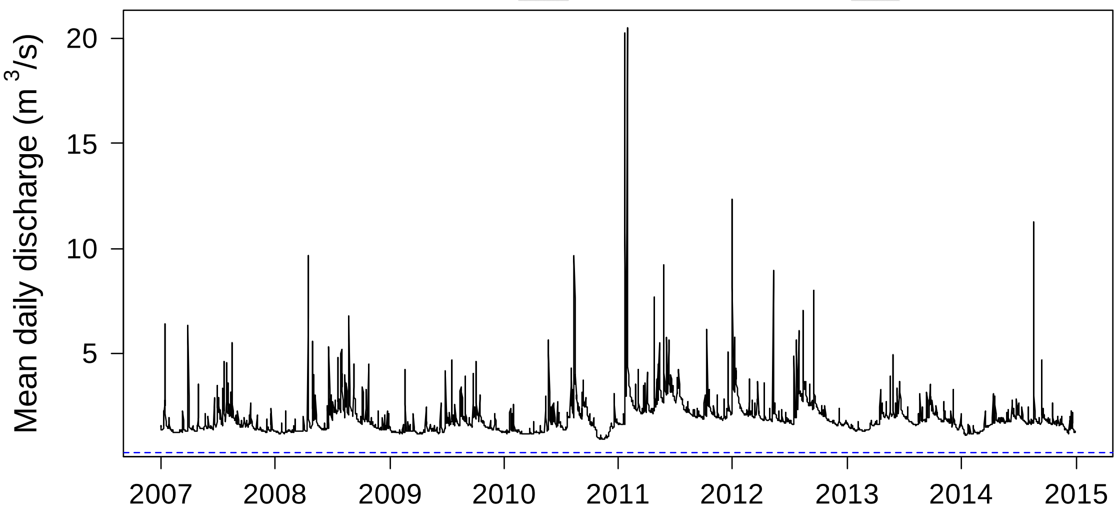

Figure 2 **Puarenga Stream mean daily discharge, 2007–2014. The dashed blue line denotes the mean discharge of treated wastewater for reference.**

3.2.2.2. Nutrient concentrations

Water quality data used to estimate baseline nutrient loads were primarily obtained from BoPRC. These data are based on monthly grab samples collected at the FRI gauge (now inactive) during 2007 through 2014. Additional data collected following storm events (Abell *et al.* 2013) were used to derive relationships between discharge and concentrations of nutrient fractions that are correlated with discharge.

Table 4 summarises the methods used to estimate baseline hourly mean nutrient concentrations. Linear interpolation of monthly measurements was used to estimate daily concentrations of nitrate, ammonium and dissolved reactive phosphorus. This was deemed suitable as concentrations of dissolved nutrient fractions are generally invariant with discharge in the Puarenga Stream. Concentrations of nitrate are a partial exception as they typically exhibit decreases during high discharge (dilution effect), although these are generally balanced by subsequent 'pulses' of elevated concentrations that are of approximate equal magnitude to the prior decreases.

For periods of hourly mean discharge > 3.0 m^3/s, concentrations of both particulate phosphorus and the non–dissolved inorganic nitrogen (DIN) fraction (i.e., TN-DIN) were estimated using linear (log_{10}–log_{10} space) relationships between concentration and discharge. Such relationships were weaker for discharge < 3.0 m^3/s, and thus linear interpolation was used to estimate concentrations of these analytes for these periods. The sum of total dissolved phosphorus minus dissolved reactive phosphorus was assumed to be zero (i.e., dissolved organic phosphorus was assumed to be negligible).

Table 4 Summary of methods used to derive baseline hourly mean nutrient concentrations in the Puarenga Stream for the period 2007–2014.

Analyte	Estimation method	Notes
PO_4-P	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements (*n* = 3) replaced with the mean of concentrations measured in that year.
PP	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between log_{10}Q and log_{10}[PP] with correction for log-transformation bias (Ferguson 1986).	Measured PP was calculated as TP minus PO_4-P. Relationship was based on data presented in Abell *et al*. (2013), collected when dischage was 3.0 to 15.6 m^3/s (maximum PP = 0.44 mg/L). Maximum mean hourly discharge for 2007-2014 was 30.4 m^3/s; maximum modelled mean hourly [PP] was 0.51 mg/L.
TP	By calculation.	PO_4-P + PP
NO_x-N	Linear interpolation of monthly measurements collected by BoPRC.	Missing (n = 2) and anomalously low (*n* = 3) measurements replaced with the mean of concentrations measured in that year.
NH_4-N	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements (*n* = 4) replaced with the mean of concentrations measured in that year.
(TN-DIN)	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between log_{10}Q and log_{10}[TN-DIN] with correction for log-transformation bias (Ferguson 1986).	This fraction includes dissolved (i.e., filterable) organic nitrogen (DON) and particulate nitrogen (PN).
DON	0.40 × (TN-DIN)	Based on the mean proportion of (TN-DIN) that comprised (TDN-DIN) in 80 samples collected during three storm events (Abell *et al*. 2013). There was no correlation between this proportion and Q.
PN	0.60 × (TN-DIN)	Based on the mean proportion of (TN-DIN) that comprised (TN-TDN) in 80 samples collected during three storm events (Abell *et al*. 2013). There was no correlation between this proportion and Q.
TN	By calculation.	NO_x-N + NH_4-N + DON + PN

3.2.2.3. Calculations to estimate in–stream loads and concentrations

Daily nutrient loads in the Puarenga Stream and the various proposed treated wastewater discharges were calculated as

$$L_x = K \cdot \sum_{i=1}^{24} \widehat{C_{x}}_i \cdot Q_i$$

where L_x is load (kg/d) of nutrient x, K is a unit conversion factor, $\widehat{C_{x_i}}$ is estimated mean concentration (mg/L) of nutrient x during hour i, and Q_i is mean discharge (m^3/s) for hour i. Daily loads were summed to calculate annual loads (t/y).

Loads for individual treatment options were compared with the nutrient reduction targets that have been set for Lake Rotorua (BoPRC 2009), in addition to the baseline loads in the Puarenga Stream to place the loads in context of downstream waters.

Daily mean nutrient concentrations that corresponded to combined Puarenga Stream and wastewater loads were estimated by dividing combined loads by the combined discharge. Thus, these estimated concentrations do not reflect any non–conservative processes such as uptake by plants or denitrification. The potential for such processes to influence nutrient concentrations in the Puarenga Stream downstream of the proposed stream discharge locations is limited given the short length (and thus residence time) of this reach (Map 2).

3.2.3. Comparison of concentrations with values designated in the NPS 2014 to assess in–stream effects on Ecosystem Health

The National Policy Statement for Freshwater Management 2014 (New Zealand Government 2014) designates values for a range of attributes that correspond to different Ecosystem Health Attribute States. Attribute States range from *A* (high ecosystem health) to *D* (low ecosystem health). Values corresponding to the 'National Bottom Line' have also been defined, which correspond to the minimum acceptable state that has been set by the government. Separate values have been defined for different aquatic ecosystems types. For rivers, values have been defined for the following attributes: nitrate (with respect to toxicity effects), ammonium (with respect to toxicity effects), dissolved oxygen, *E. coli* and periphyton.

Potential effects of the proposed options in relation to nitrate, ammonium and dissolved oxygen concentrations were assessed quantitatively by comparing baseline concentrations in the Puarenga Stream with estimated concentrations following addition of separate wastewater discharges corresponding to the six treatment options (Table 2). These differences were then considered in the context of Ecosystem Health Attribute State values for these analytes, which are reproduced in Table 5, Table 6 and Table 7. For nitrate and ammonium, these assessments were based on the time series of daily mean concentration data that were derived for each modelling scenario (described further in Section 3.3.4.4 below). For dissolved oxygen, the assessment was based on comparing monthly measurements collected by BoPRC in the lower Puarenga Stream with concentrations that were estimated for corresponding days for a scenario of anoxic wastewater discharge (i.e., a worst case scenario). Concentrations for this scenario were estimated using daily mean discharge data for the Puarenga Stream (Figure 2) and assuming conservation of mass. Degassing due to temperature effects was not considered.

Potential effects of the proposed options in relation to *E. coli* concentrations were assessed semi–quantitatively by determining the corresponding Ecosystem Health Attribute States (see Table 8) for each year in the baseline period using data collected by BoPRC, and considering these in the context of likely effluent composition.

Potential effects of the proposed options in relation to periphyton were assessed qualitatively, based on consideration of potential increases to nutrient concentrations and consequent implications for bottom–up effects on periphyton.

Table 5 Nitrate nitrogen concentrations (mg N/L) corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to nitrate toxicity (New Zealand Government 2014).

Attribute state	Numeric attribute state		Narrative attribute state
	Annual median	Annual 95th percentile	
A	≤ 1.0	≤ 1.5	High conservation value system. Unlikely to be effects even on sensitive species.
B	> 1.0 and ≤ 2.4	>1.5 and ≤ 3.5	Some growth effect on up to 5% of species.
C	> 2.4 and ≤ 6.9	> 3.5 and ≤ 9.8	Growth effects on up to 20% of species (mainly sensitive species such as fish).
National bottom line	6.9	9.8	No acute effects.
D	> 6.9	> 9.8	Impacts on growth of multiple species, and starts approaching acute impact level (i.e. risk of death) for sensitive species at higher concentrations (> 20 mg N/L).

Table 6 Ammoniacal nitrogen concentrations (mg N/L) corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to ammonia toxicity (New Zealand Government 2014).

Attribute state	Numeric attribute state		Narrative attribute state
	Annual median	Annual 95th percentile	
A	≤ 0.03	≤ 0.05	High conservation value system. Unlikely to be effects even on sensitive species.
B	> 0.03 and ≤ 0.24	>0.05 and ≤ 0.40	Some growth effect on up to 5% of species.
C	> 0.24 and ≤ 1.3	> 0.40 and ≤ 2.20	Growth effects on up to 20% of species (mainly sensitive species such as fish).
National bottom line	1.3	2.2	No acute effects.
D	> 1.30	> 2.20	Impacts on growth of multiple species, and starts approaching acute impact level (i.e. risk of death) for sensitive species at higher concentrations (> 20 mg N/L).

Table 7 Dissolved oxygen concentrations corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to ammonia toxicity (New Zealand Government 2014).

Attribute state	Numeric attribute state		Narrative attribute state
	7-day mean minimum (1 Nov to 30 April)	1-day minimum (1 Nov to 30 April)	
A	≥ 8.0	≥ 7.5	No stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites.
B	≥ 7.0 and ≤ 8.0	≥ 5.0 and < 7.5	Occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen. Risk of reduced abundance of sensitive fish and macroinvertebrate species.
C	≥ 5.0 and ≤ 7.0	≥ 4.0 and < 5.0	Moderate stress on a number of aquatic organisms caused by dissolved oxygen levels exceeding preference levels for periods of several hours each day. Risk of sensitive fish and macroinvertebrate species being lost.
National bottom line	5.0	4.0	
D	< 5.0	< 4.0	Significant, persistent stress on a range of aquatic organisms caused by dissolved oxygen exceeding tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.

Table 8 *E. coli* concentrations corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to ammonia toxicity (New Zealand Government 2014).

Attribute state	Numeric attribute state 7-day mean minimum (1 Nov to 30 April; /100 mL)	Statistic	Narrative attribute state
A	≤ 260	Annual median	People are exposed to a very low risk of infection (less than 0.1% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating).
		95th percentile	People are exposed to a low risk of infection (up to 1% risk) when undertaking activities likely to involve full immersion.
B	> 260 ≤ 540	Annual median	People are exposed to a low risk of infection (less than 1% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating).
		95th percentile	People are exposed to a moderate risk of infection (less than 5% risk) when undertaking activities likely to involve full immersion. 540 / 100 mL is the minimum acceptable state for activities likely to involve full immersion.
C	> 540 ≤ 1000	Annual median	People are exposed to a moderate risk of infection (less than 5% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating). People are exposed to a high risk of infection (greater than 5% risk) from contact with water during activities likely to involve immersion
National Bottom Line	1000	Annual median	
D	> 1000	Annual median	People are exposed to a high risk of infection (greater than 5% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating).

3.3. One–dimensional lake modelling

3.3.1. Model selection

The 1–D model DYRESM–CAEDYM was selected. The model comprises a hydrodynamic model (DYRESM[1]) that is coupled to a water quality model (CAEDYM[2]). DYRESM predicts the vertical

[1] DYnamic REservoir Simulation Model

distributions of temperature and density in lakes such as Lake Rotorua that have relatively simple morphometry and satisfy the 1–D approximation. CAEDYM can be used to model a wide range of biogeochemical state variables such as nutrient concentrations or phytoplankton abundance. The models are process–based, and are thus primarily based on representations of functional (rather than empirical) relationships between different variables. Both DYRESM and CAEDYM were developed at the Centre for Water Research (CWR) in Western Australia. Details of the model conceptualisations and equations are available in the 'science manuals' (Hipsey *et al*. 2013; Imerito 2013).

DYRESM–CAEDYM is the most widely–cited aquatic ecosystem model in the scientific literature (Trolle *et al*., 2012). The model has been applied to several lakes in New Zealand, and it has now been applied to Lake Rotorua for numerous years to understand in–lake processes and inform management decisions. Specifically, the model has previously been used to predict how Lake Rotorua water quality will respond: to reductions in external and internal loads (Burger *et al*. 2008); land use and climate changes (Hamilton *et al*. 2012), and; alum dosing (Hamilton *et al*. 2015). Thus selecting DYRESM–CAEDYM meant that this study could benefit from the extensive body of previous work that has been undertaken to configure and calibrate the model to reflect the characteristics of Lake Rotorua.

Such process–based modelling enables the simulation of a wide range of variables at high temporal resolution to provide detailed understanding of major processes in the lake. The use of process–based models allows for greater certainty in the outcome of simulated scenarios that differ from the current state, compared to the use of empirical (i.e., statistical) relationships which are generally invalid outside of the bounds of the data used for model derivation. A constraint of this approach, however, is that such process–based models are "data hungry"; they require information for a large number of forcing variables such as those that relate to weather, morphometry and inflows, in addition to field measurements of simulated variables to assist model calibration. In this regard, Lake Rotorua is a suitable candidate as it has been relatively extensively monitored and therefore there exists a large body of data to use for model configuration. Information regarding a few aspects is, however, sparse; the consequences of this for model uncertainty are outlined later in this report.

3.3.2. Model overview

DYRESM simulates multiple layers of variable thickness that change dynamically to accommodate changes in lake volume. DYRESM is primarily affected by surface exchanges of heat, mass and momentum, and resolves the vertical distributions of temperature, salinity, and density in lakes and reservoirs (Imerito 2013).

CAEDYM simulates fluxes that regulate biogeochemical variables such as nutrient concentrations and phytoplankton biomass (Hipsey *et al*. 2013). The model includes representations of cycling processes for carbon, nitrogen, phosphorus, dissolved oxygen and inorganic suspended sediments. The state variables that are simulated within CAEDYM can be adjusted depending on the study

[2] Computational Aquatic Ecosystem Dynamics Model

objectives and the availability of measured data for calibration. Accordingly, the following three generic groups of phytoplankton were represented in CAEDYM: freshwater diatoms, chlorophytes and cyanobacteria. Phytoplankton growth depends on nutrient availability and temperature. For each model time step, growth rate (μ; d^{-1}) for each phytoplankton group was estimated with CAEDYM as[3]:

$$\mu = \mu_{max} \times min[f(I), f(N), f(P), f(Si)] \times f_{T1}(T)$$

where μ_{max} (d^{-1}) is maximum growth rate at 20 °C; f(I), f(N), f(P) and f(Si) represent limitation by light, nitrogen, phosphorus and silica (diatoms only) respectively, and; f_{T1}(T) is a temperature function which allows the maximum growth rate at temperature of T_{opt} and prevents growth at temperature > T_{max}. Nutrient limitation was represented using a Monod equation which required the user to assign nutrient half saturation constants to each phytoplankton group. Photo–inhibition was not represented. Simulated phytoplankton biomass can be dynamically converted to output estimates of chlorophyll *a* concentrations in the water column and summed for each of the three phytoplankton groups for different depths, at each model time step.

Conceptual diagrams of the representations of nitrogen and phosphorus cycling within CAEDYM are shown in Figure 3. Each process in the figure was explicitly represented in CAEDYM. Higher fauna and macrophytes were not considered.

[3] From Equation 6.1 in Hipsey *et al.* (2013)

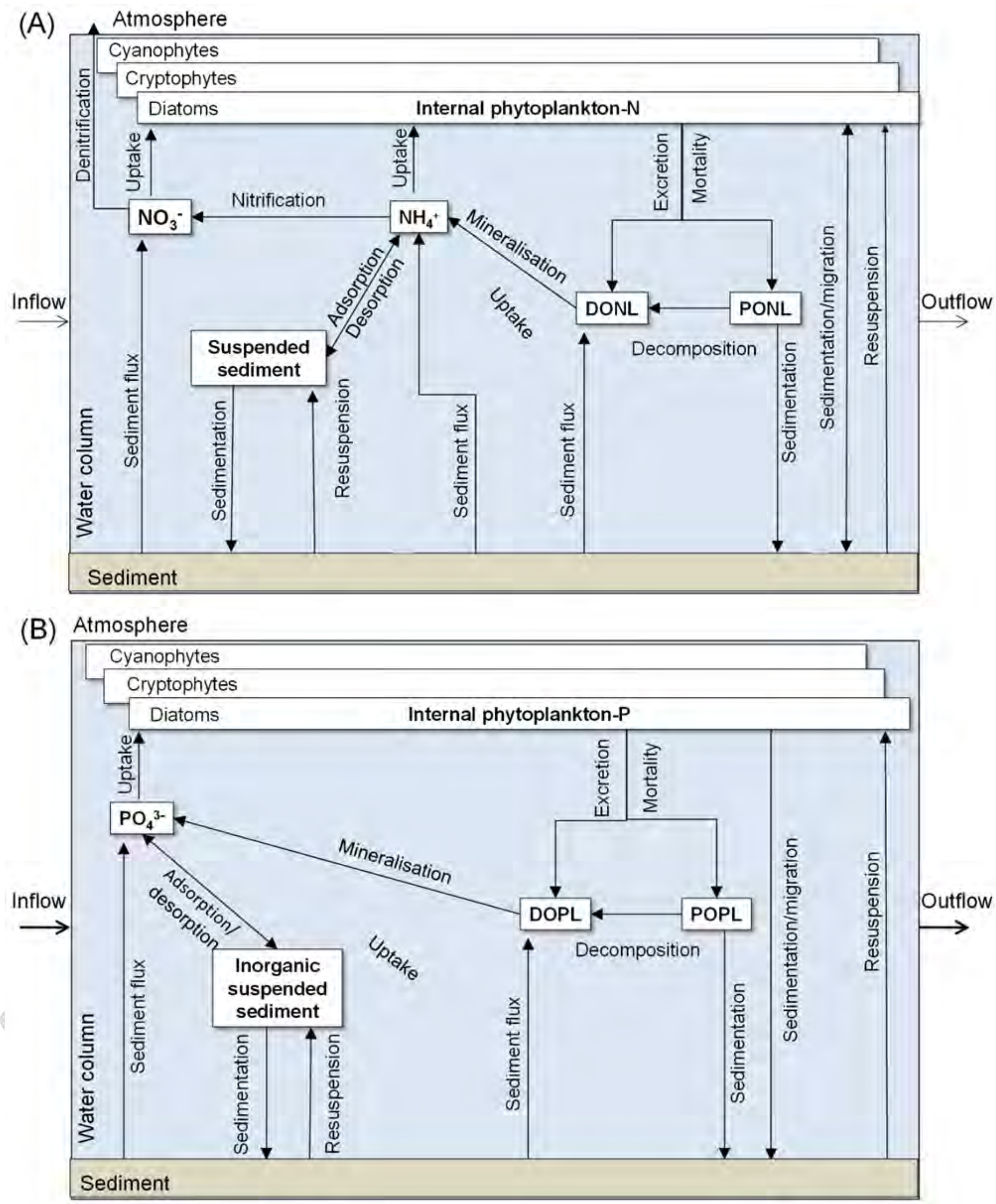

Figure 3 **Conceptual diagrams of the cycling of nitrogen (A) and phosphorus (B) within the water quality model (CAEDYM). DONL, labile dissolved organic nitrogen; PONL, labile particulate organic nitrogen; DOPL, labile dissolved organic phosphorus; POPL, labile particulate organic phosphorus.**

3.3.3. Model simulation, calibration and validation periods

An eight year baseline period of 2007–2014 was selected for the 1–D water quality modelling. This period encompasses the most recent period for which the necessary forcing data are available and it was deemed important to select a period which was as recent as possible to help to assess effects relative to current water quality. It was also desirable to select a baseline period that spanned multiple years so that it encompassed a range of forcing conditions (particularly weather) that were representative of current conditions. The first year was selected as 2007 because this corresponds to the first full year during which alum dosing was undertaken (see Section 2.1.1). Alum dosing has had a significant effect on lake water quality (Hamilton *et al.* 2015) and it was desirable to constrain the modelling period to include only the period when alum dosing was undertaken. This is because the effects of alum dosing are currently represented 'statically' in the DYRESM–CAEDYM configuration by adjusting parameters that control sediment nutrient release rates and particulate matter diameter to reflect nutrient adsorption and sediment flocculation caused by alum (discussed further in Section 3.3.4.5 below). Thus, the need to use a separate model configuration for periods with and without alum dosing currently inhibits the use of the model to simulate a single period that includes years both before and after 2007.

The calibration period was defined as 2007–2010 and the validation period was 2011–2014. Model performance for each period was quantified by comparing modelled and measured values of the following water quality parameters: temperature, dissolved oxygen, nutrients and chlorophyll *a* (Table 9). Comparisons were made with measured data collected at a range of depths by BoPRC as part of a monthly monitoring programme. For each sampling date, a mean of measurements collected at the two mid lake sites that are sampled by BoPR ('Site 2' and 'Site 5') was calculated, and these mean values were used in all comparisons with model results.

The model was run with a time step of one day. Water quality parameters were initialized based on the most recent monitoring data that corresponded to the start date. A one year 'spin up' period was modelled prior to each simulation. This was configured by 'looping' forcing data for 2007, and model outputs from this period were not considered during analysis.

Table 9 Model performance statistics

Abbreviation	Statistic	Details	Equation
r	Pearson product moment correlation	Measures the strength of the correlation between modelled and measured data, i.e. how 'in phase' the two signals are. Vales range from -1 (perfect negative correlation) to 1 (perfect positive correlation).	$\frac{\sum_{i=1}^{n}(o_i-\bar{o})\times(m_i-\bar{m})}{\sqrt{\sum_{i=1}^{n}(o_i-\bar{o})}\times\sqrt{\sum_{i=1}^{n}(m_i-\bar{m})}}$
RMSE	Root mean square error	A measure of the magnitude of the error between modelled and measured data which is disproportionately affected by large errors.	$\sqrt{\frac{\sum_{i=1}^{n}(m_i-o_i)^2}{n}}$
MAE	Mean absolute error	Measures the average error, irrespective of whether the model under– or over–predicts measurements.	$\frac{\sum_{i=1}^{n}\lvert(m_i-o_i)\rvert}{n}$

3.3.4. Model configuration

3.3.4.1. Bathymetry

Lake bathymetry was represented using a lake–area relationship provided by BoPRC. Maximum lake depth prescribed by this relationship was 25 m and therefore isolated holes present in the lake (depth ≈ 50 m) were ignored.

3.3.4.2. Meteorological input data

Meteorological data were obtained from records collected at the Rotorua Airport automatic weather station (AWS), located on the south–eastern shore of the lake (Map 1). Data collected prior to 2013 were obtained from the National Climate Database administered by NIWA (http://cliflo.niwa.co.nz/); data collected since January 2013 were provided by MetService. Mean daily data were collated for the following variables as inputs to the model:

- rainfall (m)
- wind speed (m/s)
- air temperature (° C)
- shortwave solar radiation (W/m^2)
- vapour pressure (hPa).

Daily cloud cover was estimated based on the difference between observed daily mean short–wave solar radiation and estimated theoretical minima and maxima (Luo *et al*. 2010).

3.3.4.3. Hydrologic input data

The model configuration included representation of daily mean discharge for nine major streams and nine minor streams (Table 10). Where available, stream discharge data were obtained from near–continuous records from hydrometric gauges that were operational throughout the modelling period. This was the case for the Ngongotaha Stream (operated by NIWA), and the Puarenga, Waingaehe and Utuhina streams (operated by BoPRC; see BoPRC 2007). For streams without a permanent gauge, mean discharge was estimated based on monthly measurements of discharge that were either collected by BoPRC, presented in other studies (Rutherford *et al*. 2008) or used in previous modelling applications (Abell and Hamilton 2015). Daily fluctuations of discharge in such streams were then modelled based on fluctuations measured in comparable streams.

Outflow via the Ōhau Channel (the only outlet) was configured based on daily mean measured discharge provided by NIWA. This was then amended to reflect negative values of the residual term in the water balance (see below).

Groundwater inputs to the bed of the lake were estimated as the residual term in a water balance constructed for the lake. Thus

$$Groundwater = (Q_{\bar{O}hau} + E + \Delta S) - (Q_{inflow} + rainfall)$$

where *Groundwater* is mean daily groundwater input (m^3/s), $Q_{\bar{O}hau}$ is mean daily discharge of the only lake surface outflow (m^3/s), ΔS is mean daily rate of change in lake storage (m^3/s) due to water level change (provided by NIWA, measured at the Mission Bay monitoring station), Q_{inflow} is mean daily stream discharge (m^3/s) and *rainfall* is mean daily rainfall (m^3/s) based on measurements at Rotorua Airport applied across the lake.

Note that this term will therefore reflect error in the estimation of the other terms in the water balance, in addition to unmonitored inputs such as overland flow and additional minor streams.

Hourly mean evaporation rate (E; m^3/s) was calculated based on Fischer *et al.* (1979):

$$E = \frac{A\left(\frac{-0.622 C_L\, \rho_a\, L_E U(e_a - e_s)(T_{surf})}{P}\right)}{L_v}$$

where A is the area of the lake (m^2), C_L is the latent heat transfer coefficient for wind speed (0.0013), ρ_a is air density (kg/m), L_E is the latent heat of evaporation of water (2453000 J/kg), U is measured wind speed (m/s), e_a is the vapour pressure of the air (Pa), e_s is the saturated vapour pressure of the air (Pa) corresponding to the lake water surface temperature (°C), P is the atmospheric pressure (Pa), L_v is the latent heat of vaporisation (2260000 J/kg) and T_{surf} is the surface water temperature (°C) estimated using a relationship established between day of the year and historic measurements. A value of 0 was substituted where $E < 0$ as the models do not simulate condensation effects.

e_s was calculated by the Magus–Tetens formula (Hodges and Dallimore 2011):

$$e_s(T_{0.5}) = 100 \exp\left[2.3026\left(\frac{7.5\, T_{0.5}}{T_{0.5} + 237.3}\right) + 0.758\right]$$

It was necessary to subtract a small quantity (3% of daily mean discharge) from the outflow data to optimise the fit between modelled and measured water level. This reason for the minor discrepancy is uncertain, although it may relate to minor differences in either evaporation rates or the changes in storage calculated by the model, and those estimated in the water balance

Table 10 Summary of how discharge was configured for the inflows and outflow.

Inflow type	Inflow	Mean discharge (m^3/s)	Details
Major streams	Awahou Stream	1.69	The mean discharge was set to the mean of monthly instantaneous gaugings during 2005 through 2012 (*n* = 86). Temporal fluctuations were then imposed based on fluctuations measured in the Ngongotaha Stream.
	Hamurana Stream	2.57	This is a groundwater spring-dominated stream. Monthly (approximate) instantaneous gaugings were interpolated for the period 2007 through 2012 (*n* = 51). Discharge set to the mean of gaugings (2.558 m^3/s) during 2012 through 2014.
	Ngongotaha Stream	1.84	Based on measured data (99.9% of record) at SH 30 gauge. One gap of 89 h was filled with mean value of preceding and subsequent days.
	Puarenga Stream	1.95	Based on measured data (97.2% of record) at FRI gauge (2007 to 2010) and SH30 gauge (2010 to 2014). Gaps were replaced with modelled data (2.8% of record) based on linear relationship (r^2 = 0.75) with measurements for Utuhina Stream.
	Utuhina Stream	1.81	Based on measured data (92.8% of record) at Depot Street gauge. Gaps were replaced with modelled data (7.2% of record) based on linear relationship (r^2 = 0.67) with measurements for Puarenga Stream.
	Waingaehe Stream	0.27	Based on measured data (99.5% of record) at SH30 gauge. Gaps were replaced with mean values of adjoining measurements (0.5% of record).
	Waiohewa Stream	0.38	As for the Awahou Stream. Mean discharge was estimated based on a sample of 70 measurements.
	Waiowhiro Stream	0.31	As for the Awahou Stream. Mean discharge was estimated based on a sample of 78 measurements.
	Waiteti Stream	1.23	As for the Awahou Stream. Mean discharge was estimated based on a sample of 76 measurements.
Minor Streams	Lynmore Stream	0.05	The long-term mean discharge was set to the mean of monthly instantaneous gaugings during 2005 through 2012 (*n* = 71). Temporal fluctuations were then imposed based on fluctuations measured in the Waingaehe Stream.
	Motutara (geothermal seep)	0.04	A constant discharge was assigned
	Rotokawa 1 (geothermal seep)	0.02	A constant discharge was assigned
	Rotokawa 2 (geothermal seep)	0.04	A constant discharge was assigned
	Hauraki Stream	0.01	The long-term mean discharge was set to the mean discharge reported in Rutherford *et al*. (2008). Temporal fluctuations were then imposed based on fluctuations measured in the Waingaehe Stream.
	Waitawa 1 Waitawa 2 Waimehia Drain Waiowhiro 2/ Waikuta	0.06 0.06 0.06 0.06	The long-term mean discharge in these four streams was calculated from the mean discharge reported in Rutherford *et al*. (2008) for 'minor' catchments (0.4 m^3/s), minus the mean discharge for the other five minor streams. Temporal fluctuations were then imposed based on fluctuations measured in the Waingaehe Stream.
Outflow	Ōhau Channel	19.44	Daily mean discharge was provided by NIWA (mean for period = 18.15 m^3/s). This was then adjusted to account for negative values of the residual quantity in the daily water balance.
Groundwater	Groundwater	4.68	Calculated as positive values of the residual quantity in the water balance.

3.3.4.4. Inflow water quality

Temperature and dissolved oxygen

Hourly mean temperature (°C) of precipitation was set to lake surface water temperature, estimated using an empirical relationship between historical measurements and day of year.

Hourly mean temperatures (°C) of remaining surface inflows (T_s) were estimated using an empirical model described by Mohseni *et al.* (1998):

$$T_s = \frac{\alpha}{1 + e^{\gamma(\beta - T_a)}}$$

where T_a is the average daily air temperature measured at Rotorua Airport AWS (°C), α is the maximum historic measured stream temperature (°C) and both γ and β are dimensionless parameters. Parameters γ and β were determined by fitting the model to historic spot measurements of stream temperature provided by BoPRC (n = 65 – 96) and minimising root mean squared error using the Solver add–in to Microsoft Excel 2007. Measured data were not available for most minor streams and subsequently T_s for one stream (Lynmore) was assigned to five minor streams.

Dissolved oxygen (DO) concentrations of all inflows were assumed to be 100% saturated based on estimated water temperature. Accordingly, DO concentrations were estimated using the following equation derived by Mortimer (1981)

$$DO = \exp\big(7.71 - 1.31\ ln(T_s + 45.93)\big)$$

where DO is dissolved oxygen at saturation (mg/L).

Nutrient and suspended sediment concentrations

Major streams

Nutrient and inorganic suspended sediment (ISS) concentrations were assigned to stream inflows based on measured data. Data were primarily obtained from a dataset collected by BoPRC during routine monthly sampling. Additional data obtained from a study undertaken of two major stream inflows during 2010–2012 (Abell *et al.* 2013) were used to assign concentrations during storm flows.

The nine major stream inflows (Map 1) were represented separately in the model. Details of how nutrient and ISS concentrations were assigned to these streams are presented in Table 11. For completeness, the table repeats details for the Puarenga Stream that are described above in Section 3.2.2. Briefly, daily nutrient concentrations were typically assigned by linearly interpolating monthly measurements. Exceptions were concentrations of ISS, particulate phosphorus (PP) and the non–dissolved inorganic nitrogen (DIN) fraction of total nitrogen (TN) pool (i.e., TN-DIN) in the Ngongotaha, Puarenga and Utuhina streams. Concentrations of these analytes have been shown to positively correlate with discharge (Hoare 1982, Rutherford 2008), and failure to account for this effect results in marked underestimation of long–term loads to the lake (Abell *et al.* 2013). Such storm loads were quantified for the Ngongotaha, Puarenga and Utuhina streams as these have the greatest proportion of annual nitrogen and phosphorus loads transported in storm flow (Rutherford 2008). Storm loads were not quantified for other streams as storm fluxes are less dominant for these streams, due to relatively greater dominance of groundwater inputs. In addition, there were insufficient data to robustly define relationships between concentrations and discharge for these streams, and therefore the potential for increasing error by estimating such relationships was deemed to outweigh the error associated with underestimating storm fluxes.

Table 11 Methods to assign nutrient concentrations to major stream inflows. See glossary for definitions of abbreviations.

Analyte	Stream	Estimation method	Notes
PO_4-P	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing/anomalous measurements replaced with the mean of concentrations measured in adjoining months.
PP	Puarenga	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q ≥ 3 m^3/s: Derived from a linear relationship between $\log_{10}$Q and $\log_{10}$[PP] for the Puarenga Stream with correction for transformation bias (Ferguson 1986).	Relationship was based on data collected from the Puarenga Stream when discharge was 3.0 to 15.6 m3/s (maximum [PP] = 0.44 mg/L; n = 174; r2 = 0.19). Maximum modelled mean daily [PP] was 0.38 mg/L.
	Ngongotaha and Utuhina	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q ≥ 3 m^3/s: Derived from a linear relationship between $\log_{10}$Q and $\log_{10}$[PP] for the Ngongotaha Stream with correction for transformation bias (Ferguson 1986).	Relationship was based on datacollected when discharge was 3.0 to 22 m3/s (maximum [PP] = 0.44 mg/L; n = 44; r2=0.77). Maximum modelled mean daily [PP] was 0.53 mg/L and 0.44 mg/L.
	Awahou, Waiteti, Waingaehe, Waiowhiro, Waiohewa, Hamurana	Linear interpolation of monthly measurements collected by BoPRC.	Measured PP was calculated as TP minus PO_4-P.
TP	All	By calculation.	PO_4-P + PP
NO_x-N	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing and anomalous (e.g., > TN) measurements replaced with the mean of concentrations measured in adjoining months.
NH_4-N	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements replaced with the mean of concentrations measured for adjoining months.
(TN-DIN)	Puarenga	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q ≥ 3 m^3/s: Derived from a linear relationship between [(TN-DIN)] and $\log_{10}$Q for the Puarenga Stream with correction for log-transformation bias (Ferguson 1986).	This fraction includes dissolved (i.e., filterable) organic nitrogen (DON) and particulate nitrogen (PN). Relationship was based on data collected from the Puarenga Stream when discharge was 3.0 to 15.6 m3/s (maximum [(TN-DIN)] = 1.62 mg/L; n = 223; r2 = 0.15). Maximum modelled mean daily [(TN-DIN)] was 1.60 mg/L.
	Ngongotaha and Utuhina	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q ≥ 3 m^3/s: Derived from a linear relationship between [(TN-DIN)] and $\log_{10}$Q for the Ngongotaha Stream with correction for log-transformation bias (Ferguson 1986).	Relationship was based on data collected when discharge was 3.0 to 18 m3/s (maximum [(TN-DIN)] = 1.63 mg/L; n = 38; r2=0.85). Maximum modelled mean daily [(TN-DIN)] was 1.59 mg/L and 1.48 mg/L .
	Awahou, Waiteti, Waingaehe, Waiowhiro, Waiohewa, Hamurana	Linear interpolation of monthly measurements collected by BoPRC.	
DON	All	0.4 × (TN-DIN)	Based on the mean proportions of (TN-DIN) that comprised (TDN-DIN) (to define DON) and (TN-TDN) (to define PN) in 80 samples collected during three storm events on the Puarenga Stream and 73 samples collected during three storm events on the Ngongotaha Stream. The mean proportions were the same for both streams and there was no correlation between the values for this proportion and Q.
PN	All	0.6 × (TN-DIN)	
TN	All	By calculation.	NO_x-N + NH_4-N +DON + PN
ISS	Puarenga	Derived from a linear relationship between $\log_{10}$[(TSS] and $\log_{10}$Q for the Puarenga Stream with correction for log-transformation bias (Ferguson 1986).	Relationship was based on data presented in Abell et al. (2013), collected from the Puarenga Stream when discharge was 1.5 to 10.8 m3/s (maximum [TSS] = 463 mg/L; n = 507; r2 = 0.65). Maximum modelled mean daily [TSS] was 1422 mg/L. Assumed that [ISS] = 0.68 × [TSS], based on the mean value of [ISS]/[TSS] measured during storm sampling of Puarenga Stream. (n = 234, σ = 0.12)
	Ngongotaha and Utuhina	Derived from a power function (negative exponent) between $\log_{10}$[(TSS] and $\log_{10}$Q for the Ngongotaha Stream with correction for log-transformation bias (Ferguson 1986).	Relationship was based on data collected from the Puarenga Stream when discharge was 1.4 to 22 m3/s (maximum [TSS] = 510 mg/L; n = 256; r2 = 0.85). Maximum modelled mean daily [TSS] was 663 mg/L and 295 mg/L. Assumed that [ISS] = 0.57 × [TSS], based on the mean value of [ISS]/[TSS] measured during storm sampling of Ngongotaha Stream (n = 111, σ = 0.23).
	Awahou, Waiteti, Waingaehe, Waiowhiro, Waiohewa, Hamurana	Set equal to the mean TSS concentrations measured by BoPRC in each stream since 2000 (sampling undertaken in 2002 and 2003).	
DOCL	All	Calculated as 7.29 × [DIN]	Assumed that C:N is 7.29 (by mass), based on Sterner et al (2008)
POCL	All	Calculated as 7.29 × [PN]	Assumed that C:N is 7.29 (by mass), based on Sterner et al (2008)

Minor streams

Details of how nutrient and ISS concentrations were assigned to nine minor stream inflows are presented in Table 12. The minor streams were represented in the model by a single inflow for which discharge–weighted (i.e., volumetric) concentrations were specified based on estimated loads for individual streams.

Table 12 Methods to assign nutrient concentrations to minor stream inflows. See glossary for definitions of abbreviations.

Analyte	Stream	Estimation method	Notes
PO_4-P	Minor rural surface streams (Waitawa 1, Waitawa 2, Hauraki, Waimehia Drain, Waiowhiro)	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream (smallest of the major stream inflows, drains a predominantly pastoral catchment).	Missing/anomalous measurements replaced with the mean of concentrations measured in adjoining months.
	Lynmore Stream (minor urban surface stream)	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.176 mg/L; n = 134).	
PP	Minor rural surface streams	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream.	
	Lynmore Stream	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.074 mg/L; n = 134).	
TP	All	By calculation.	PO_4-P + PP
NO_x-N	Minor rural surface streams	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream.	Missing and anomalous (e.g., > TN) measurements replaced with the mean of concentrations measured in adjoining months.
	Lynmore Stream	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.036 mg/L; n = 134).	
NH_4-N	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements replaced with the mean of concentrations measured for adjoining months.
(TN-DIN)	Minor rural surface streams	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream.	
	Lynmore Stream	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.316 mg/L; n = 134).	
DON	All	0.4 × (TN-DIN)	As for major streams.
PN	All	0.6 × (TN-DIN)	
TN	All	By calculation.	NO_x-N + NH_4-N +DON + PN
ISS	Minor rural surface streams	Set equal to the mean TSS concentrations measured by BoPRC in Waingaehe Stream 2000 (sampling undertaken in 2002 and 2003).	
	Lynmore Stream	Set equal to the mean TSS concentrations measured by BoPRC in Lynmore Stream 2000 (sampling undertaken in 2002 and 2003).	
	Groundwater seeps at the lake edge	Assumed nil.	
DOCL	All	Calculated as 7.29 × [DIN]	As for major streams.
POCL	All	Calculated as 7.29 × [PN]	

Atmospheric deposition

Wet atmospheric deposition of nitrogen and phosphorus on the lake surface was represented by configuring precipitation as a surface inflow to the lake (rather than including this in the meteorological forcing file). Precipitation was assigned constant nitrogen concentrations of 0.285 mg/L (as NO_3–N) and phosphorus concentrations of 0.013 mg/L (as PO_4–P), based on values used in previous model applications (Hamilton *et al.* 2012), which were based on typical concentrations for the Taupo Volcanic Zone (Hamilton 2005). Concentrations of other nutrient fractions were not assigned to this input.

Groundwater (residual)

A final inflow was configured that was termed 'groundwater'. This represented input associated with the residual term in the water balance (see Section 3.3.4.3) and therefore included groundwater inputs to lake bed of the lake, in addition to fluxes associated with overland flow additional minor streams and any under–estimation of hydraulic inputs in the other inflows. Daily nutrient and ISS concentrations in this inflow were assigned using discharge–weighted concentrations calculated using data for the nine major stream inflows.

Summary of assigned nutrient concentrations

A summary of nutrient concentrations assigned to each inflow is presented in Table 13.

Table 13 Summary of nutrient concentrations assigned to inflows represented in the 1–D model, 2007–2014.

Analyte	Percentile	Inflow													
		Awahou	Hamarana	Puarenga	Puarenga (-LTS)	Puarenga (-alum)	Utunina	Utuhina (-alum)	Waingaehe	Waiohewa	Waiowhiro	Waiteti	Minor	Groundwater[1]	Atmospheric deposition
PO_4-P	5	0.056	0.062	0.003	0.002	0.020	0.007	0.043	0.070	0.007	0.021	0.024	0.089	0.036	0.013
	25	0.063	0.075	0.006	0.005	0.028	0.016	0.048	0.089	0.013	0.028	0.030	0.101	0.044	0.013
	50	0.066	0.079	0.012	0.010	0.035	0.027	0.055	0.094	0.017	0.035	0.033	0.106	0.048	0.013
	75	0.070	0.082	0.042	0.033	0.045	0.035	0.060	0.098	0.021	0.039	0.037	0.109	0.051	0.013
	95	0.078	0.087	0.065	0.051	0.065	0.046	0.065	0.105	0.029	0.044	0.045	0.115	0.058	0.013
TP	5	0.07	0.08	0.05	0.04	0.07	0.05	0.06	0.10	0.07	0.06	0.04	0.13	0.07	0.013
	25	0.07	0.08	0.07	0.06	0.09	0.06	0.08	0.11	0.08	0.08	0.05	0.14	0.08	0.013
	50	0.07	0.09	0.09	0.07	0.10	0.07	0.10	0.12	0.10	0.09	0.06	0.14	0.09	0.013
	75	0.08	0.09	0.11	0.09	0.12	0.08	0.11	0.13	0.11	0.10	0.07	0.15	0.09	0.013
	95	0.10	0.11	0.14	0.11	0.14	0.10	0.14	0.15	0.15	0.11	0.09	0.17	0.11	0.013
NO_3-N	5	1.087	0.646	0.677	0.295	0.677	0.509	0.509	1.270	0.983	0.759	1.137	1.066	0.820	0.285
	25	1.240	0.695	0.774	0.295	0.774	0.591	0.591	1.379	1.162	0.854	1.300	1.167	0.888	0.285
	50	1.320	0.726	0.844	0.295	0.844	0.652	0.652	1.454	1.330	0.911	1.368	1.244	0.949	0.285
	75	1.448	0.775	0.948	0.295	0.948	0.708	0.708	1.528	1.457	0.968	1.441	1.318	0.984	0.285
	95	1.519	0.807	1.114	0.295	1.114	0.822	0.822	1.631	1.680	1.082	1.581	1.402	1.062	0.285
NH_4-N	5	0.001	0.003	0.034	0.063	0.034	0.024	0.024	0.003	0.373	0.007	0.008	0.099	0.033	0.00
	25	0.004	0.005	0.060	0.063	0.060	0.030	0.030	0.005	0.897	0.013	0.012	0.107	0.048	0.00
	50	0.005	0.006	0.069	0.063	0.069	0.035	0.035	0.007	1.221	0.019	0.015	0.113	0.061	0.00
	75	0.009	0.008	0.079	0.063	0.079	0.042	0.042	0.011	1.505	0.028	0.018	0.119	0.075	0.00
	95	0.020	0.014	0.104	0.063	0.104	0.056	0.056	0.015	1.938	0.046	0.028	0.126	0.088	0.00
TN	5	1.16	0.74	0.80	0.38	0.80	0.63	0.63	1.37	1.99	0.84	1.27	1.36	0.99	0.285
	25	1.32	0.79	0.95	0.42	0.95	0.71	0.71	1.46	2.35	0.95	1.41	1.44	1.05	0.285
	50	1.41	0.83	1.05	0.47	1.05	0.77	0.77	1.56	2.61	1.00	1.47	1.52	1.08	0.285
	75	1.50	0.86	1.20	0.54	1.20	0.86	0.86	1.64	2.89	1.08	1.55	1.58	1.16	0.285
	95	1.64	0.88	1.46	0.75	1.46	1.07	1.07	1.78	3.34	1.17	1.70	1.66	1.29	0.285

1. The 'groundwater' inflow represents the residual quantity in the water balance and thus represents groundwater inputs to the bed, in addition to all other inflows that are not otherwise accounted for. These include: additional minor streams, drains, overland flow and any inputs related to underestination of discharge in the other streams.

3.3.4.5. Alum dosing

Alum was added to the Utuhina and Puarenga streams during the baseline period, resulting in reduced dissolved reactive phosphorus concentrations in the streams and the lake (see Section 2.1.1). It was therefore necessary to represent this action in the model configuration for the baseline period.

The water quality monitoring site at the Utuhina Stream is downstream of the alum dosing plant and therefore the measured water quality data for this stream reflected the in–stream effects of alum (i.e., reduced dissolved reactive phosphorus concentrations). The water quality monitoring site at the Puarenga Stream was upstream of the alum dosing plant and therefore it was necessary to reduce dissolved reactive phosphorus concentrations in the inflow data for this stream inflow to reflect alum effects. Concentrations were reduced in proportion to the mean load of aluminum that was applied during a particular month (data provided by BoPRC). This was calculated using a linear–$\log_{10}$ relationship derived by Hamilton *et al.* (2015) between the concentration reduction factor and aluminium load, based on data collected by BoPRC at sites upstream and downstream of the alum dosing plant (Figure 4).

In addition, two changes were made to the configuration of the water quality parameters in CAEDYM to reflect the in–lake effects of alum. Firstly, internal loading associated with hypoxia was suppressed by reducing the maximum potential PO_4–P release rate from bed sediments to 0.02 g/m^2/d, which is lower than the rate assigned in previous model applications that simulated periods prior to alum dosing. Secondly, elevated in–lake flocculation of organic material caused by alum was represented by assigning a high particulate organic material diameter of 0.018 mm. Further details about the rationale for the methods used to represent in–lake alum effects are provided in Hamilton *et al.* (2015).

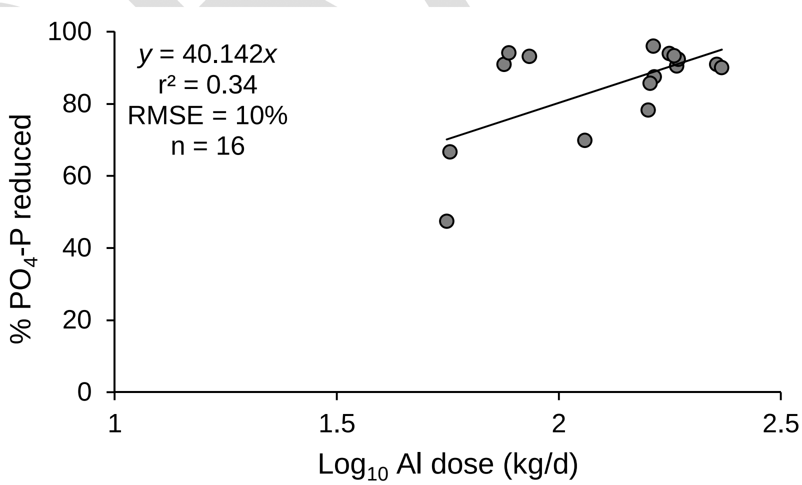

Figure 4 **Relationship between percentage reductions to dissolved reactive phosphorus (PO_4–P) concentrations and mean monthly aluminium dose in the Puarenga Stream. Data provided by BoPRC.**

3.3.5. Model scenarios

3.3.5.1. Baseline and wastewater discharge

Scenarios simulated with the 1–D model are listed in Table 14. The baseline (1D_0) scenario involved no discharge of treated wastewater and therefore provides a benchmark representative of current conditions against which the effects of the various scenarios can be compared. Separate scenarios were simulated to represent discharge of treated wastewater to surface waters following treatment using each of the six treatment options (Table 2). These scenarios were configured by adding the treated wastewater as a separate inflow that enters the lake surface. These scenarios therefore represent discharge to either the Puarenga Stream or the lake shore sites (Map 2).

Two further scenarios were configured to examine the effects of lake bed discharge. The treatment options selected for these scenarios were 2c and 3a because they provide some contrast; relative to the other options, these respectively have low phosphorus concentrations and moderate nitrogen concentrations, or low nitrogen concentrations and moderate phosphorus concentrations.

Discharge rates and nutrient concentrations were assigned to the treated wastewater using the information presented in Mott Macdonald (2014; see Table 2). Table 15 presents the mean annual nitrogen and phosphorus loads in the Puarenga Stream that correspond to the 1–D scenarios.

The treated wastewater temperature was assumed to follow an annual sinusoidal trend with a maximum of 18 °C and a minimum of 16 °C (K. Brian, pers. comm. 2015a; Figure 5). Precise specifications of dissolved oxygen concentrations were unavailable so treated wastewater was generally assumed to be 100% saturated in the scenarios (see Section 3.3.4.4), although two additional scenarios were included to simulate discharge of anoxic treated wastewater (Options 2c and 3a) to isolate the effects of varying this parameter.

No distinctions were made between the various discharge arrangements, such as gabions or rapid infiltration beds (Section 2.1.3). The purpose of these options is to convey treated wastewater, rather than to provide treatment (Mott MacDonald 2014; RPSC 2014). Consequently, no specific discharge arrangement has been specified for the scenarios.

The lake outflow volume was increased (+ 0.276 m^3/s) to balance the additional inflow for all scenarios involving treated wastewater discharge.

Table 14 Scenarios simulated using the 1–D model

#	Code	Scenario	Details
1	1D_0	Baseline with no wastewater discharge simulated.	Eight year period (2007-2014). Alum dosing effects represented.
2	1D_1_Surface	Treatment option 1, discharge to surface waters	
3	1D_2a_Surface	Treatment option 2a, discharge to surface waters	
4	1D_2b_Surface	Treatment option 2b, discharge to surface waters	
5	1D_2c_Surface	Treatment option 2c, discharge to surface waters	
6	1D_3a_Surface	Treatment option 3a, discharge to surface waters	
7	1D_3b_Surface	Treatment option 3b, discharge to surface waters	
8	1D_2c_Bed	Treatment option 2c, discharge to lake bed	
9	1D_3a_Bed	Treatment option 3a, discharge to lake bed	
10	1D_2c_Surface - DO	Treatment option 2c, discharge to surface, no dissolved oxygen in wastewater	Option 2c has the 'best' P treatment (TP = 0.10 mg/L) and 'moderate' N treatment (TN = 4.37 mg/L)
11	1D_3a_Surface - DO	Treatment option 3a, discharge to surface, no dissolved oxygen in wastewater	Option 3a has the 'best' N treatment (TN = 2.63 mg/L) and 'moderate' P treatment (TP = 0.20 mg/L)
12	1D_0 - LTS	Baseline, Land Treatment System loads removed from the Puarenga Stream	
13	1D_2c_Surface - LTS	Treatment option 2c, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
14	1D_3a_Surface - LTS	Treatment option 3a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
15	1D_0 - Alum	Baseline, alum effects (in-lake and in-stream) not simulated	
16	1D_2c_Surface - Alum	Treatment option 2c, discharge to surface, alum effects (in-lake and in-stream) not simulated	
17	1D_3a_Surface - Alum	Treatment option 3a, discharge to surface, alum effects (in-lake and in-stream) not simulated	
18	1D_0 - LTS - Alum	Baseline, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	
19	1D_2c_Surface - LTS - Alum	Treatment option 2c, discharge to surface, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	
20	1D_3a_Surface - LTS - Alum	Treatment option 3a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	
21	1D_30N_3P_Surface	Wastewater discharge of 30 t N/y and 3 t P/y to surface waters	
22	1D_30N_3P_Surface - LTS	Wastewater discharge of 30 t N/y and 3 t P/y to surface waters, Land Treatment System loads removed from the Puarenga Stream	
23	1D_30N_1.5P_Surface - LTS	Wastewater discharge of 30 t N/y and 1.5 t P/y to surface waters, Land Treatment System loads removed from the Puarenga Stream	
24	1D_0 + 'pure' wastewater	Baseline with discharge of wastewater to surface waters that contains no nutrients	Not proposed but simulated to quanity potential flushing effects

Table 15 Summary of mean annual Puarenga Stream nutrient loads for the 1–D model scenarios.

Scenario	Description	TN (t/y)		DIN (t N/y)		TP (t/y)		PO_4-P (t P/y)	
		Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.
1D_0_Stream	Baseline 2007-2014 (PO_4-P attenuated by alum)	69.8	19.8	58.6	13.5	6.0	1.9	1.3	1.1
1D_1_Stream	Baseline + Option 1	117.2	19.8	87.1	13.5	12.3	1.9	2.2	1.1
1D_2a_Stream	Baseline + Option 2a	112.1	19.8	87.1	13.5	9.2	1.9	2.2	1.1
1D_2b_Stream	Baseline + Option 2b	110.0	19.8	87.1	13.5	7.7	1.9	2.2	1.1
1D_2c_Stream	Baseline + Option 2c	107.8	19.8	87.1	13.5	6.9	1.9	2.2	1.1
1D_3a_Stream	Baseline + Option 3a	92.7	19.8	69.8	13.5	7.7	1.9	2.2	1.1
1D_3b_Stream	Baseline + Option 3b	101.4	19.8	78.5	13.5	7.7	1.9	2.2	1.1
1D_0-LTS	Baseline with LTS loads removed	33.2	10.5	22.0	4.3	4.7	1.5	1.0	0.9
1D_2c_Stream_-LTS	Baseline + Option 2c, LTS loads removed	71.2	10.5	50.5	4.3	5.6	1.5	1.9	0.9
1D_3a_Stream_-LTS	Baseline + Option 3a, LTS loads removed	56.1	10.5	33.2	4.3	6.5	1.5	1.9	0.9
1D_0 - Alum	Baseline with no alum dosing	69.8	19.8	58.6	13.5	7.0	2.3	2.3	0.6
1D_2c_Stream - Alum	Baseline + Option 2c with no alum dosing	107.8	19.8	87.1	13.5	7.8	2.3	3.1	0.6
1D_3a_Stream - Alum	Baseline + Option 3c with no alum dosing	92.7	19.8	69.8	13.5	8.7	2.3	3.1	0.6
1D_0 - LTS - Alum	Baseline, LTS loads removed, no alum dosing	33.2	10.5	22.0	4.3	5.6	1.9	1.8	0.5
1D_2c_Stream -LTS-Alum	Baseline + Option 2c, LTS loads removed, no alum dosing	71.2	10.5	50.5	4.3	6.5	1.9	2.7	0.5
1D_3a_Stream_-LTS-Alum	Baseline + Option 3a, LTS loads removed, no alum dosing	56.1	10.5	33.2	4.3	7.4	1.9	2.7	0.5
1D_30N_3P_Surface	Baseline + 30 t N/y and 3 t P/y	99.8	19.8	87.1	13.5	9.0	1.9	2.2	1.1
1D_30N_3P_Surface - LTS	Baseline + 30 t N/y and 3 t P/y t, LTS loads removed	63.2	10.5	50.5	4.3	7.7	1.5	1.9	0.9
1D_30N_1.5P_Surface - LTS	Baseline + 30 t N/y and 1.5 t P/y t, LTS loads removed	63.2	10.5	50.5	4.3	6.2	1.5	1.9	0.9

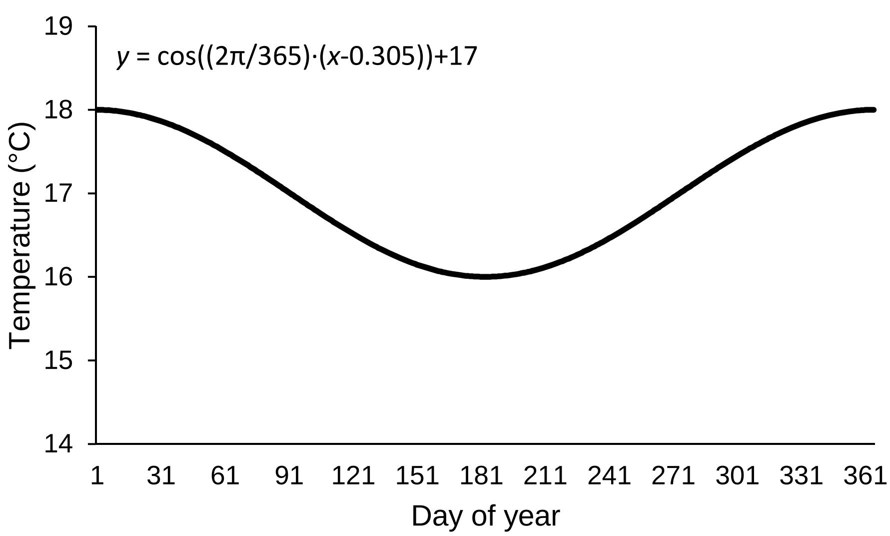

Figure 5 Water temperatures assigned to treated wastewater

3.3.5.2. Removal of LTS loads and alum dosing

A further nine scenarios (#12–20 in Table 14) were simulated to examine permutations of the following two conditions: 1) removal of nutrient loads from the Puarenga Stream associated with the LTS; 2) cessation of alum dosing.

Removal of LTS loads was simulated to reflect the decline in background nutrient loads in the Puarenga Stream that is anticipated to occur over the medium to long term following LTS closure. The rate of this decline is uncertain and background nutrient loads are expected to be higher than those in the '-LTS' scenario for several years after the LTS is closed while residual loads are 'flushed' through the catchment (see Discussion for further consideration of lag times). Scenarios comprising no LTS loads were configured by reducing concentrations of nitrogen and phosphorus fractions in the Puarenga Stream. Discharge was not reduced and it was assumed that there would be negligible decline in water yield following LTS closure as the majority of irrigated water is presumed to be lost from the catchment by evapotranspiration. Dissolved inorganic nitrogen concentrations were set to the mean of concentrations measured in the Puarenga Stream in 1992 and 1993, immediately following the initiation of the LTS in 1991. No measurements before 1991 were available and the 1992–1993 data were assumed representative of conditions prior to the marked increase in nitrogen concentrations that occurred through the mid to late 1990s (Tomer *et al.* 2000; Burns *et al.* 2009). Thus, nitrogen concentrations in the Puarenga Stream under the '-LTS' scenarios were approximately 2.5– to 3–fold less than contemporary concentrations[4]. Unlike nitrogen, phosphorus concentrations measured by BoPRC did not exhibit a marked rise in the years following LTS initiation, and contemporary concentrations are comparable with those in the early 1990s, with data exhibiting indication of a slight increase in only total phosphorus, and not dissolved reactive phosphorus[5]. Phosphorus concentrations are typically more variable than nitrogen concentrations as the particulate fraction is strongly correlated with discharge. Phosphorus concentrations in the '-LTS' scenarios were therefore configured by adjusting concentrations of all phosphorus fractions by a constant factor (0.81) to reduce the phosphorus load in the Puarenga Stream during the baseline period by an average of 1.7 t P/y, which is the 5–year 'sewage–derived' load estimated from LTS consent monitoring during 2007–2012 (A. Lowe, pers comm. 2013)[6].

Scenarios were simulated to examine the effects of removing alum dosing to examine how discontinuing this action will influence the predicted effects of discharging treated wastewater. Configuring these scenarios involved: 1) increasing dissolved reactive phosphorus concentrations in the Utuhina and Puarenga streams to 'non alum' levels; 2) adjusting the CAEDYM parameters that were specifically modified to represent the in–lake effects of alum dosing.

Dissolved reactive phosphorus concentrations in the Utuhina Stream (monitored downstream of the alum dosing plant) were amended by setting them equal to the product of the mean ratio of dissolved reactive phosphorus to total phosphorus during 2001-2005 (pre–alum dosing; 0.804) and assigned total phosphorus, with the maximum value set to 0.065 mg/L (90th percentile of 2001-2005

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[4] Assigned concentrations were: NH_4–N = 0.064 mg/L; NO_3–N = 0.295 mg/L.

[5] E.g., 1992–1993 data: TP = 0.060 mg/L, PO4–P = 0.042 mg/L; 2013–2014 data: TP = 0.090 mg/L, PO4–P = 0.036 mg/L.

[6] Thus the baseline phosphorus load was reduced by 13.6 t (8 × 1.7) over the eight years. Note that this calculation method meant that the load in each year was not reduced by exactly 1.7 t, and therefore the difference in mean annual phosphorus load between the scenarios with and without LTS loads is ~ 1.4 t P/y (Table 15).

monitoring data) to avoid anomalously high values during storms, when total phosphorus was estimated using a relationship with discharge (Table 11). Dissolved reactive phosphorus concentrations in the Puarenga Stream (monitored upstream of the alum dosing plant) were set to the concentrations determined before modifications to represent alum effects (see Section 3.3.4.5).

Removal of in–lake alum effects was represented in CAEDYM by adjusting the maximum PO_4–P release rate and particulate organic material diameter to 0.04 g/m^2/d and 0.09 mm respectively. These values correspond to calibrated values that were used in a version of the model configured for the period prior to alum dosing commencing (Hamilton *et al*. 2015).

3.3.5.3. Additional scenarios

Two additional configurations of treated wastewater discharge were simulated to examine the effects of improvements to current treatment performance. These two scenarios involved discharge of either: 1) 30 t N/y and 3 t P/y; 2) 30 t N/y and 1.5 t P/y. The first of these configurations was simulated both with and without LTS loads. The second of these scenarios was simulated without inclusion of LTS loads. These scenarios were configured by setting the dissolved inorganic nitrogen and phosphorus concentrations in treated wastewater equal to those of Options 1 and 2 (Table 2), and then varying the concentrations of the other fractions to achieve the desired loads.

A final scenario (1D_0 + 'pure' wastewater; Table 14) was configured that involved addition of wastewater that contained no nutrients. The objective of this was to isolate any potential effects that that are predicted to occur following wastewater discharge solely due to a slight reduction in residence time, rather than enhanced productivity due to nutrient addition.

3.3.6. Comparison of scenarios

Annual TLI_3 values were compared between the model scenarios to provide an assessment of the predicted effects of each scenario on lake trophic status in the context of water quality objectives for Lake Rotorua (BoPRC 2009). The TLI_3 integrates concentrations of total nitrogen, total phosphorus and chlorophyll *a*, based on the equations presented in Burns *et al.* (1999). TLI_3 values were calculated using surface water data.

The TLI_3 value is comparable with the TLI (see Section 2.1.1) although Secchi depth is omitted from the calculation, which is not calculated explicitly in CAEDYM. This omission means that TLI_3 and TLI are not identical, and the TLI target for Lake Rotorua of 4.20 (BoPRC 2009) is equivalent to 4.32 in TLI_3 units (Hamilton *et al*. 2015).

In addition, modelled concentrations of chlorophyll *a*, total nitrogen and total phosphorus for each scenario were compared with Ecosystem Health attribute values prescribed for lakes in the current National Policy Statement for Freshwater Management (New Zealand Government 2014). These values are reproduced in Table 16, Table 17 and Table 18. Ecosystem Health attribute values are also defined in relation to *E. coli* and planktonic cyanobacteria concentrations. Potential effects of the proposed options in relation to these attributes were assessed qualitatively, with reference to data collected during other studies where relevant.

Table 16 Chlorophyll *a* concentrations (µg/L) corresponding to Lake Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management (New Zealand Government 2014).

Attribute state	Numeric attribute state		Narrative attribute state
	Annual median	Annual maximum	
A	≤ 2	≤ 10	Lake ecological communities are healthy and resilient, similar to natural reference conditions.
B	> 2 and ≤ 5	>10 and ≤ 25	Lake ecological communities are slightly impacted by additional algal and plant growth arising from nutrients levels that are elevated above natural reference conditions.
C National bottom line	> 5 and ≤ 12 12	> 25 and ≤ 60 60	Lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrients levels that are elevated well above natural reference conditions.
D	> 12	> 60	Lake ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state, due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters of deep lakes.

Table 17 Total nitrogen concentrations (µg/L) corresponding to Lake Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management (New Zealand Government 2014).

Attribute state	Numeric attribute state	Narrative attribute state
	Annual median (polymictic)	
A	≤ 300	Lake ecological communities are healthy and resilient, similar to natural reference conditions.
B	> 300 and ≤ 500	Lake ecological communities are slightly impacted by additional algal and plant growth arising from nutrients levels that are elevated above natural reference conditions.
C National bottom line	> 500 and ≤ 800 800	Lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrients levels that are elevated well above natural reference conditions.
D	> 800	Lake ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state, due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters of deep lakes.

Table 18 Total phosphorus concentrations (µg/L) corresponding to Lake Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management (New Zealand Government 2014)

Attribute state	Numeric attribute state	Narrative attribute state
	Annual median	
A	≤ 10	Lake ecological communities are healthy and resilient, similar to natural reference conditions.
B	> 10 and ≤ 20	Lake ecological communities are slightly impacted by additional algal and plant growth arising from nutrients levels that are elevated above natural reference conditions.
C National bottom line	> 20 and ≤ 50 50	Lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrients levels that are elevated well above natural reference conditions.
D	> 50	Lake ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state, due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters of deep lakes.

3.4. Three–dimensional lake modelling

3.4.1. Model selection

ELCOM (Estuary and Lake Computer Model v. 2.2) was selected for the 3–D modelling. ELCOM is a 3–D hydrodynamics, thermodynamics and transport model that was developed at the Centre for Water Research, University of Western Australia. The model has been used extensively worldwide, and it has recently been used to study mixing processes in Lake Rotorua over periods of weeks to a month (Abell and Hamilton 2015; Gibbs *et al., in prep.*). Elsewhere in New Zealand, ELCOM has been used, either on its own or in combination with CAEDYM, to study systems that include Tauranga Harbour (Tay *et al.* 2013), Lake Benmore (Norton *et al.* 2009), Lake Rotoiti (Von Westernhagen 2010) and Lake Rotoehu (Allan 2014).

3.4.2. Model overview

ELCOM simulates velocity, salinity and temperature distributions in water bodies. The model solves the unsteady Reynolds–averaged Navier–Stokes and scalar transport equations, with modules for heat and momentum transfer across the water surface due to wind and atmospheric thermodynamics (Hodges and Dallimore 2011).

ELCOM was used in this study to investigate how mixing processes in the lake may affect the transport of treated wastewater that is discharged at the proposed locations (Map 2). This required configuring the model to include an inflow that represented treated wastewater. The propagation of the inflow was then examined by observing the path of a conservative tracer included in the inflow.

3.4.3. Model simulation periods and validation

Two separate periods were simulated to examine mixing under contrasting conditions; these were: summer 2013/2014 and winter 2014. The model was typically run for a two–month period, although some simulations designed to examine model sensitivity to wind forcing (see below) were run for only one month. Each simulation was preceded by a two–week 'spin up' period that was not considered in analysis. The performance of the model with regard to simulating the temperature structure of the lake was validated by comparing simulated temperatures with high frequency temperature measurements collected at the monitoring buoy operated by the University of Waikato. Further validation of mixing processes was not undertaken; the implications of this for model uncertainty are considered in the Discussion.

3.4.4. Model configuration

Model application required simplifying lake morphology by discretizing the water column into 3–D cells with dimensions: x = 50 m, y = 50 m and z = 0.5 – 2 m. Mean elevation of each cell was determined by interpolation using a bathymetry map with 5–m horizontal resolution. 'Flow' boundary conditions were specified at the lake–bottom and sidewalls. ELCOM was run independently of CAEDM and thus heat flux and storage associated with particulate material (e.g., phytoplankton cells) were not varied. Hourly discharge, temperature and dissolved oxygen concentrations were assigned to 18 separate inflows using the methods described for 1–D model configuration (Section 3.3.4).

Meteorological forcing data for the following variables were obtained from the Rotorua Airport AWS (Map 1): wind speed, wind direction, air temperature, solar radiation, atmospheric pressure and rainfall. Cloud cover was estimated from short–wave solar radiation (see Section 3.3.4.2). Meteorological data for the two modelling periods are presented in **Error! Reference source not found.** and **Error! Reference source not found.**.

The summer period was characterized by typically having moderate wind speeds (5 to 8 m/s) in the afternoon, frequently from a north–west to north–east direction, indicative of sea breezes from the Bay of Plenty (Figure 6). There was, however, a period of approximately two weeks in early January when the wind was predominantly from a south–west to westerly direction. This was approximately three weeks into the simulation period.

Wind speeds were generally higher during the winter period (Figure 7). Approximately one week after the start of the simulation period, there was a period of several days (~10–13 July) with high rainfall (~ 50 mm) and north–east winds of moderate to high speed (~5 to 13 m/s). Later, there were multiple periods of several days with consistent south–westerly winds, which are typical of winter in Rotorua.

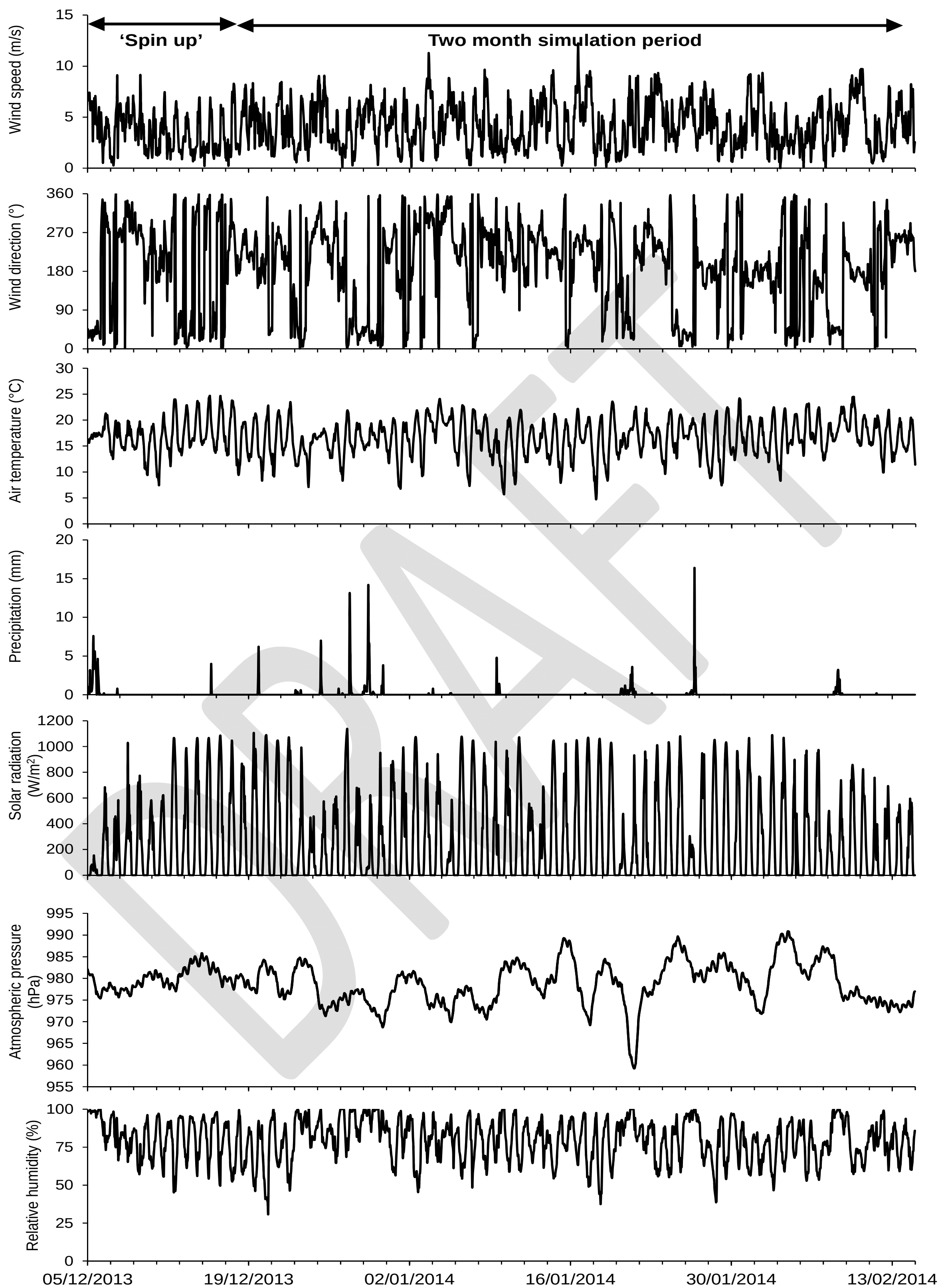

Figure 6 **Hourly mean meteorological data for the summer 2013/14 modelling period.**

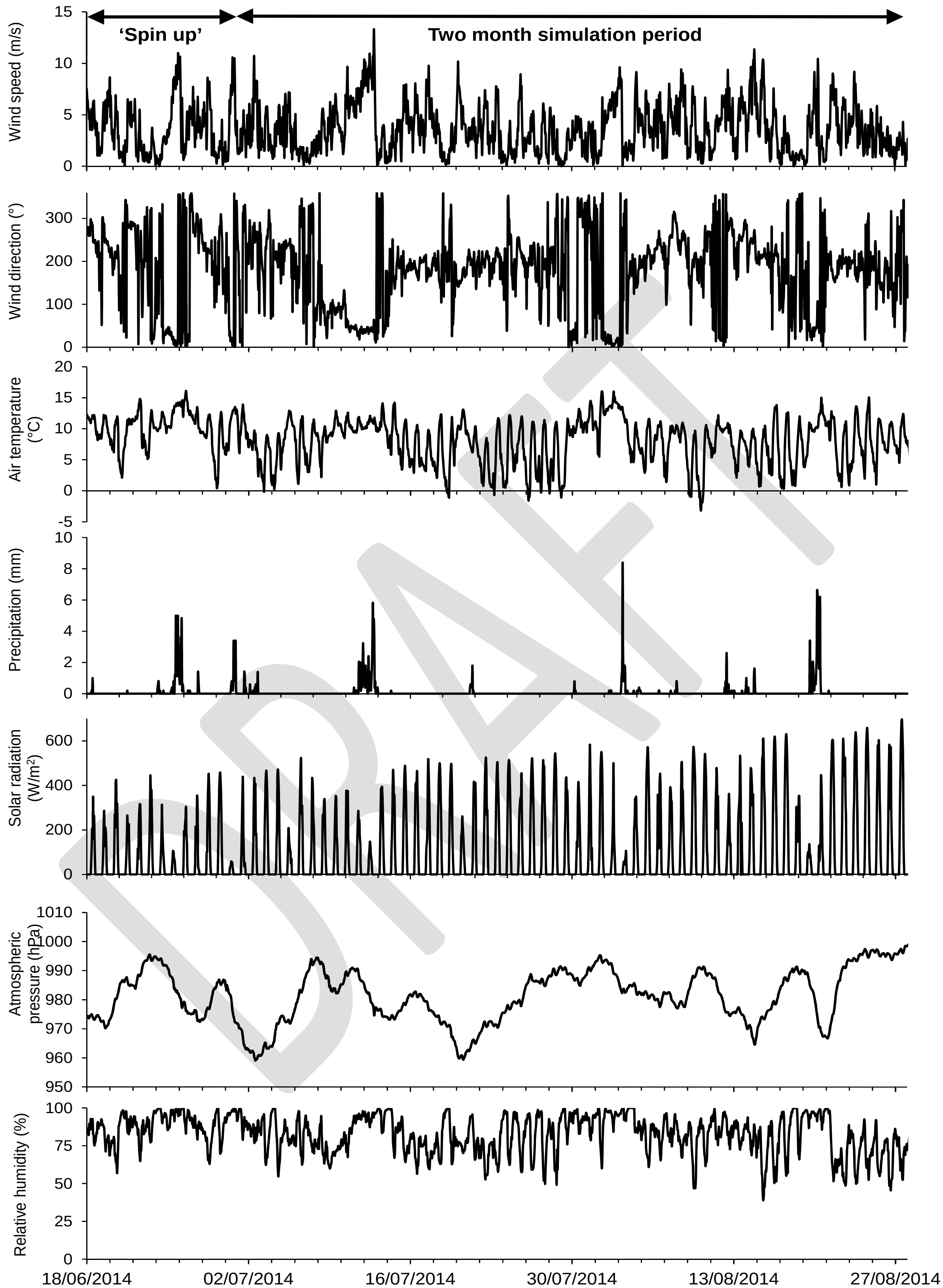

Figure 7 **Hourly mean meteorological data for the winter 2014 modelling period.**

3.4.5. Model scenarios

The 3–D modelling scenarios are listed in Table 19. Scenarios involved simulating discharge of treated wastewater to the lake at a constant rate (0.2756 m^3/s; Mott Macdonald 2014). For most scenarios (#1 to 12 in Table 19), discharge was simulated to either the Puarenga Stream (representing discharge locations 1 to 3; Map 2), or the lake bed 2 km to the north of the Puarenga Stream mouth, at a depth of ~28 m (representing discharge location 6; Map 2). These scenarios were simulated for both the summer and winter periods. Scenarios were simulated using each of the following configurations of wind forcing data: 1) measured wind speed and direction; 2) constant moderate winds (4 m/s) from the north–east; 3) constant moderate winds (4 m/s) from the south–west. The two artificial wind configurations were included because they represent the dominant wind directions in Rotorua (Figure 8), and previous work has indicated that these wind conditions establish alternate circulation patterns that have the potential to exert major and differing effects on how treated wastewater moves throughout the lake (Gibbs *et al.* 2011; Abell and Hamilton 2015; Gibbs *et al.*, in prep.).

A final scenario (#13 in Table 19) involved discharge at a location corresponding to Site 5 (Map 2), approximately 1.2 km to the north–east of the Puarenga Stream mouth. This scenario was designed to examine whether mixing of treated wastewater is likely to be different if it is discharged at the most eastern of the two proposed shoreline locations, compared to discharge via the Puarenga Stream located relatively nearby.

Table 19 Scenarios simulated with the 3–D model

#	Code	Scenario
1	3D_W_Stream	Winter, wastewater discharge to the Puarenga Stream
2	3D_W_Stream_SW	Winter, wastewater discharge to the Puarenga Stream, SW wind forcing
3	3D_W_Stream_NE	Winter, wastewater discharge to the Puarenga Stream, NE wind forcing
4	3D_W_Bed	Winter, wastewater discharge to the lake bed
5	3D_W_Bed_SW	Winter, wastewater discharge to the lake bed, SW wind forcing
6	3D_W_Bed_NE	Winter, wastewater discharge to the lake bed, NE wind forcing
7	3D_S_Stream	Summer, wastewater discharge to the Puarenga Stream
8	3D_S_Stream_SW	Summer, wastewater discharge to the Puarenga Stream, SW wind forcing
9	3D_S_Stream_NE	Summer, wastewater discharge to the Puarenga Stream, NE wind forcing
10	3D_S_Bed	Summer, wastewater discharge to the lake bed
11	3D_S_Bed_SW	Summer, wastewater discharge to the lake bed, SW wind forcing
12	3D_S_Bed_NE	Summer, wastewater discharge to the lake bed, NE wind forcing
13	3D_S_Shore	Summer, discharge to lake shore, Site 5

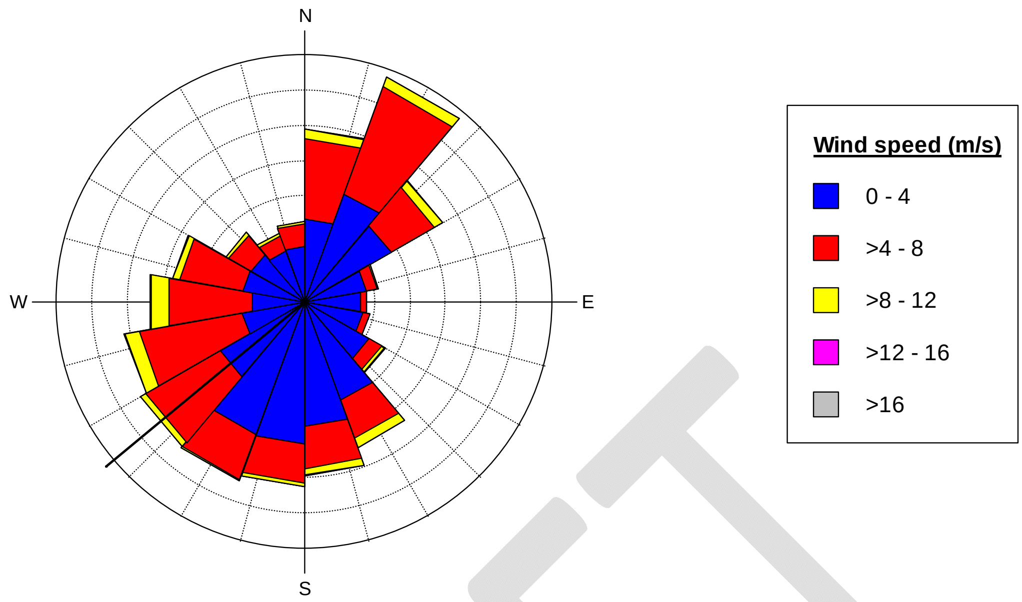

Figure 8 Summary of hourly wind measurements at Rotorua Airport Automatic Weather Station, 2007–2014.

3.4.1. Comparison of scenarios

Simulated tracer concentrations at various depths and locations in the lake were visualised for individual scenarios using ARMSLite v. 2.1.2, which was developed at the Centre for Water Research, University of Western Australia (Dallimore 2011)

4. RESULTS

4.1. Mass balance calculations to estimate in–stream nutrient loads and concentrations

4.1.1. Treated wastewater nutrient loads to the Puarenga Stream

The total nitrogen loads vary from 23 to 47 t N/y for the different treatment options, while the total phosphorus loads vary from 0.9 to 6.3 t P/y (Table 20). Relative to the 2029 external nutrient load reduction targets set for Lake Rotorua catchment (BoPRC 2009), the loads for the treatment options correspond to 9% to 19% of the nitrogen load target and 9% to 63% of the phosphorus load target (Table 21).

The loads for the treatment options correspond to approximately 33% to 67% of the mean annual total nitrogen load in the Puarenga Stream, and 13% to 90% of the mean annual total phosphorus load. There was, however, considerable between–year variability in nutrient loads conveyed by the Puarenga Stream during the baseline period, primarily reflecting differences in rainfall (Figure 9; Figure 10).

Table 20 Summary of annual nutrient loads corresponding to the six treatment options.

Source	TN (t/y)	DIN (t N/y)	TP (t/y)	PO4-P (t P/y)
Option 1	47	29	6.3	0.9
Option 2a	42	29	3.2	0.9
Option 2b	40	29	1.7	0.9
Option 2c	38	29	1.7	0.9
Option 3a	23	11	0.9	0.9
Option 3b	32	20	0.9	0.9
Puarenga Stream (mean for 2007-2014, U/S of alum dosing)	70	59	7.0	2.3

Table 21 Treatment option nutrient loads as a proportion of the external nutrient load reduction target for Lake Rotorua catchment by 2029 (BoPRC 2009).

	Proportion of annual external load reduction target (%)	
Treatment option	Nitrogen	Phosphorus
1	19%	63%
2a	17%	32%
2b	16%	17%
2c	15%	17%
3a	9%	9%
3b	13%	9%

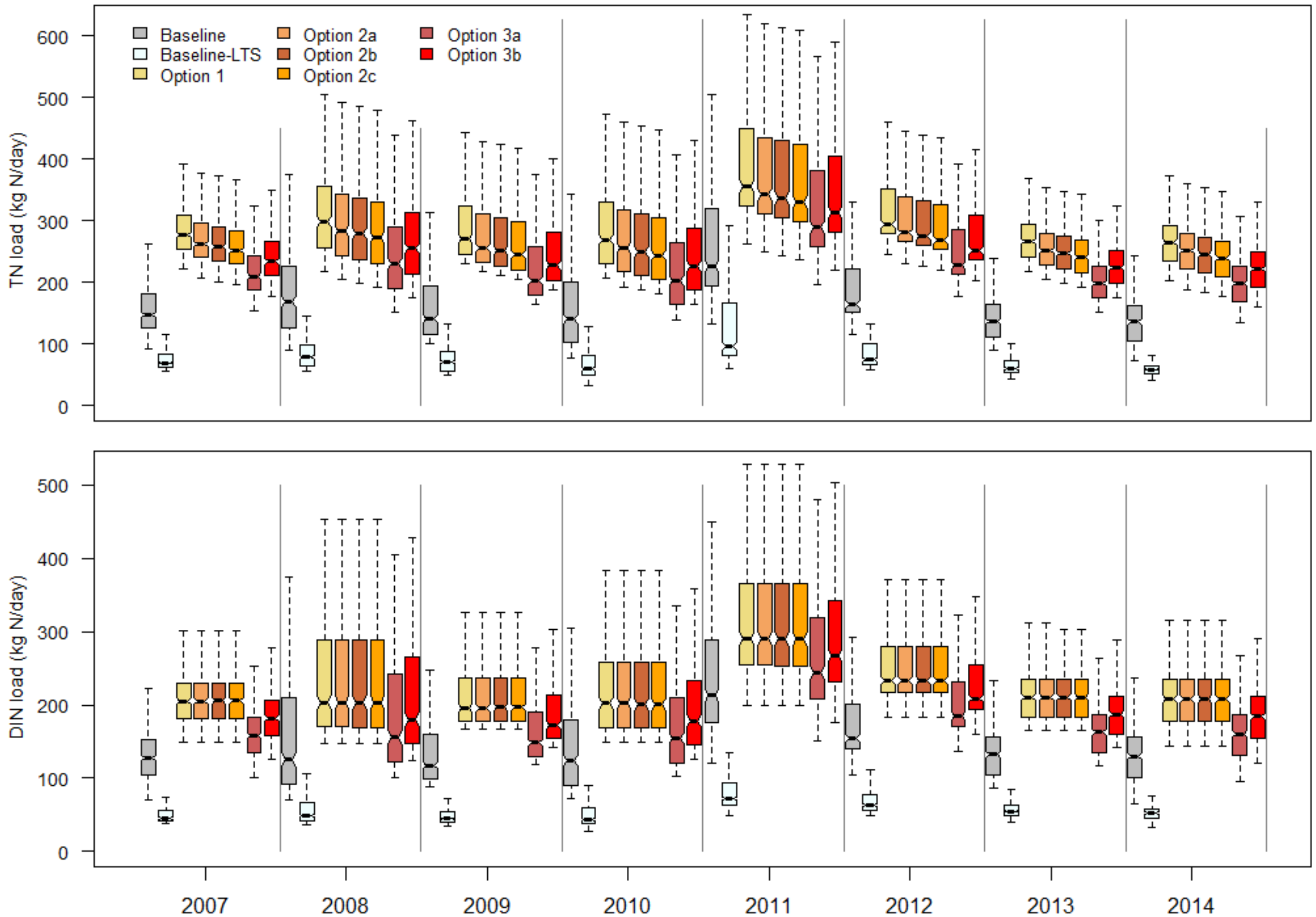

Figure 9 **Total nitrogen (TN) and dissolved inorganic nitrogen (DIN) loads in the Puarenga Stream that correspond to baseline and wastewater discharge scenarios, 2007–2014. Loads for each option include baseline loads for the Puarenga Stream. Boxplots show distributions of daily loads: notches denote median values; boxes denote 25th and 75th percentiles; whiskers extend up to 1.5 times the inter–quartile range.**

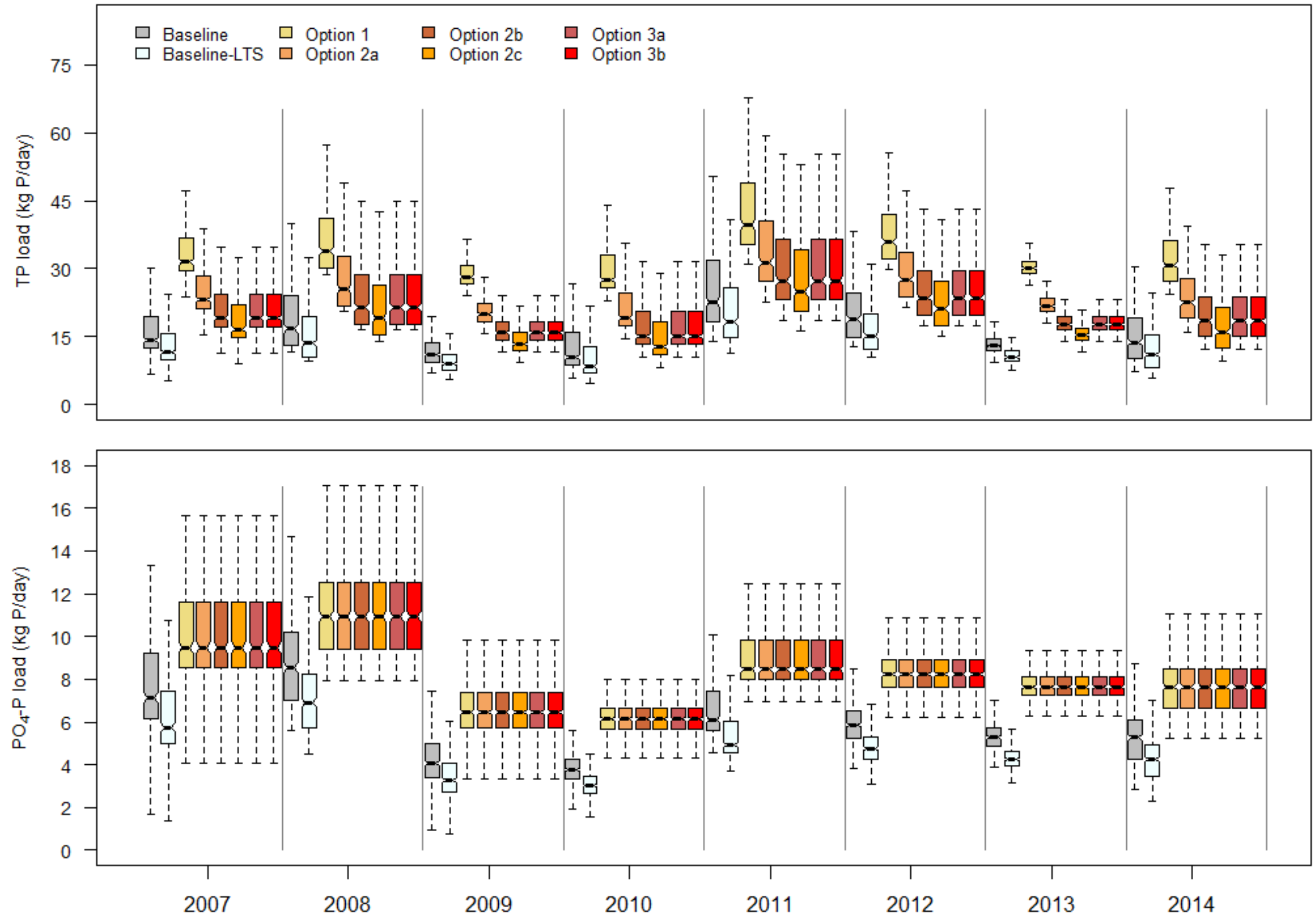

Figure 10 Total phosphorus (TP) and phosphate–phosphorus (PO4–P) loads in the Puarenga Stream that correspond to baseline and wastewater discharge scenarios, 2007–2014. Loads for each option include baseline loads for the Puarenga Stream Boxplots show distributions of daily loads: notches denote median values; boxes denote 25th and 75th percentiles; whiskers extend up to 1.5 times the inter–quartile range. Plots show loads before attenuation of baseline loads due to alum dosing.

4.1.2. Comparison of concentrations with values designated in the NPS 2014 to assess in–stream effects on Ecosystem Health

4.1.2.1. Nitrate nitrogen (toxicity)

Background nitrate concentrations in the Puarenga Stream correspond to the upper (i.e., more impacted) end of the range that is designated for Attribute State A (Figure 11; Table 22). This State corresponds to high conservation value systems (Table 5). Mass balance calculations indicate that stream discharge of wastewater following treatment with Option 3 will not change the Attribute State, although the median nitrate concentration for the 'Baseline + Option 3a' scenario is equal to the value at the boundary of Attribute States A and B (1.0 mg N/L). Discharge of wastewater following treatment with Options 1 or 2 is predicted to increase the median concentration to 1.1 mg N/L, which corresponds to the lower end of the range for Attribute State B. This State corresponds to the range at which some growth effect on up to 5% of species may occur (Table 5), although note that the 95th percentile value for these Options still corresponds to Attribute State A (Table 22).

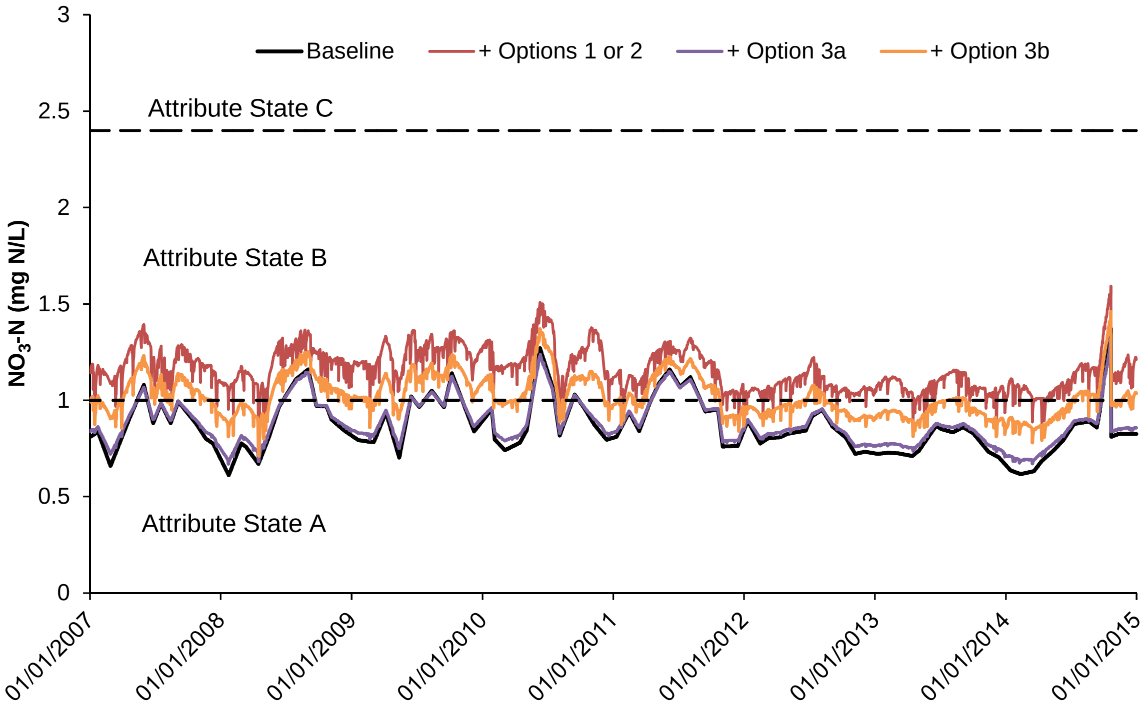

Figure 11 Estimated mean daily nitrate–nitrogen concentrations in the Puarenga Stream for baseline conditions and following addition of treated wastewater. Dashed lines denote annual median values that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.

Table 22 Statistics for nitrate–nitrogen concentrations based on: month water quality monitoring in the Puarenga Stream (2007–2014); estimated concentrations in the Puarenga Stream following addition of treated wastewater (2007–2014), and; Attribute States defined in the National Policy Statement for Freshwater Management 2014.

		Median (mg N/L)	95th percentile (mg N/L)
Puarenga Stream monthly measurements		0.8	1.1
Scenarios	Baseline	0.8	1.1
	Baseline + Options 1 or 2	1.1	1.3
	Baseline + Option 3a	0.9	1.1
	Baseline + Option 3b	1.0	1.2
NPS 2014 (annual values)	Attribute State A	≤ 1.0	≤ 1.5
	Attribute State B	> 1.0 and ≤ 2.4	>1.5 and ≤ 3.5
	Attribute State C	> 2.4 and ≤ 6.9	> 3.5 and ≤ 9.8
	National Bottom Line	6.9	9.8
	Attribute State D	> 6.9	> 9.8

4.1.2.2. Ammonium nitrogen (toxicity)

Background ammonium concentrations in the Puarenga Stream correspond to Attribute State B (Figure 12; Table 23). This State corresponds to the range at which some growth effects on up to 5% of species may occur (Table 6). Discharge of wastewater treated using the proposed options is not predicted to cause a change of Attribute State.

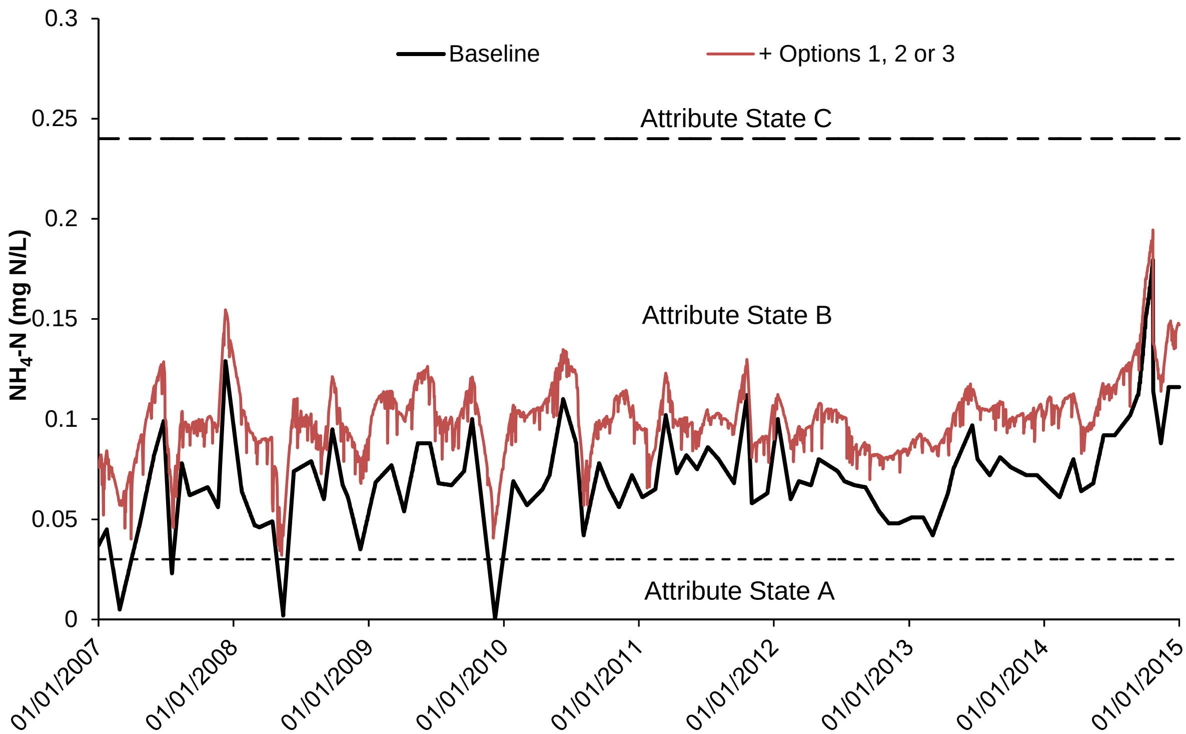

Figure 12 **Estimated mean daily ammonium nitrogen concentrations in the Puarenga Stream for baseline conditions and following addition of treated wastewater. Dashed lines denote annual median values that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

Table 23 **Statistics for ammonium nitrogen concentrations based on: month water quality monitoring in the Puarenga Stream (2007–2014); estimated concentrations in the Puarenga Stream following addition of treated wastewater (2007–2014), and; Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

		Median (mg N/L)	95th percentile (mg N/L)
Puarenga Stream monthly measurements		0.07	0.11
Scenarios	Baseline	0.07	0.10
	Baseline + Options 1, 2 or 3	0.10	0.13
NPS 2014	Attribute State A	≤ 0.03	≤ 0.05
	Attribute State B	> 0.03 and ≤ 0.24	>0.05 and ≤ 0.40
	Attribute State C	> 0.24 and ≤ 1.3	> 0.40 and ≤ 2.20
	National Bottom Line	1.3	2.2
	Attribute State D	> 1.30	> 2.20

4.1.2.3. Dissolved oxygen

Background dissolved concentrations measured in the Puarenga Stream by BoPRC generally corresponded to Attribute State A, with only 2 of the 91 measurements (2%) slightly less than the value that defines the boundary of States A and B (Figure 13). Attribute State A corresponds to a condition of "*no stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites*" (Table 7). Calculations showed that, relative to this baseline, a worst case scenario involving addition of anoxic treated wastewater would cause more frequent measurements that correspond to Attribute State B, with the majority (73%) of measurements still corresponding to Attribute State A. Attribute State B corresponds to a state of "*occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen* [causing] *risk of reduced abundance of sensitive fish and macroinvertebrate species"* (Table 7).

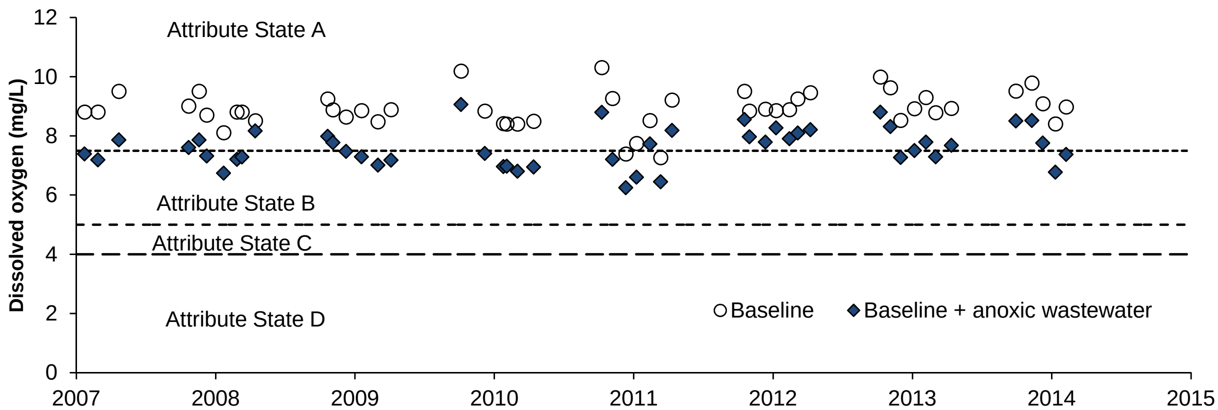

Figure 13 **Monthly measurements of dissolved oxygen concentration in the lower Puarenga Stream collected during November–April by BoPRC (circles), compared with estimated concentrations following addition of anoxic wastewater (diamonds). Dashed lines denote 1–day minimum values that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

4.1.2.4. *E. coli*

Historical E. coli concentrations measured by BoPRC show moderately–high temporal variability (Figure 14). Consequently, Attribute States were determined for individual years to characterise the baseline conditions in the Puarenga Stream with respect to this analyte (Table 24). The *E. coli* Attribute State was B for six of the eight years in the baseline period. Attribute State B corresponds to a low (<1%) risk of infection to water users (Table 8). Concentrations corresponded to either Attribute States A or D during a single year (see Table 8 for details).

No specific data were available for projected *E. coli* concentrations for each of the treatment options. Data were provided, however, of *E. coli* concentrations measured following membrane bioreactor treatment at the current WWTP (K. Brian, pers. comm. 2015b; Table 25). The median count is zero and these concentrations are very low compared with the concentrations measured in the Puarenga Stream, which have an annual median count of 29/100 mL to 185/100 mL. If these concentrations are representative of those corresponding to the proposed options, then there is predicted to be a neutral to very minor effect on the current risk to human health related to *E. coli* in the lower Puarenga Stream.

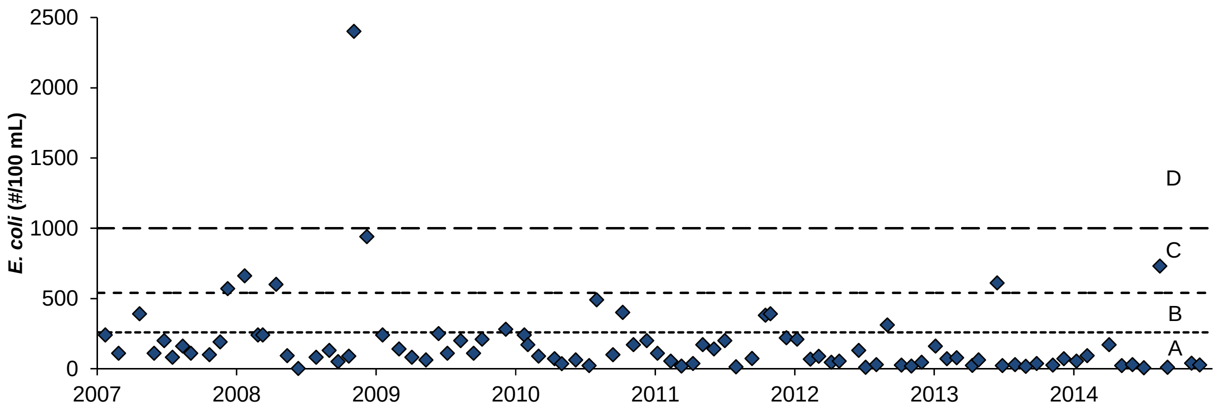

Figure 14 Monthly measurements of *E. coli* concentration in the lower Puarenga Stream collected BoPRC. Dashed lines denote values (defined as both annual median and annual 95th percentile) that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.

Table 24 *E. coli* concentrations in the lower Puarenga Stream measured by BoPRC and associated Attribute States, as defined in the National Policy Statement for Freshwater Management 2014.

Year	Annual median (#/100 mL)	95th percentile (#/100 mL)	Attribute State
2007	160	480	B
2008	185	1597	D
2009	170	266.5	B
2010	135	458.5	B
2011	125	384.5	B
2012	49.5	255	A
2013	50	362.5	B
2014	29	450	B

Table 25 Summary of E. coli concentrations following treatment with the current membrane bioreactor (K. Brian, pers. comm. 2015b).

Statistic	E.coli (#/ 100 mL)
n	277
95th percentile	6.2
Median	0
Mean	5.6
Std. Dev.	61

4.1.2.5. Periphyton

No baseline periphyton data were available for the lower Puarenga Stream to inform this assessment and the baseline Attribute State for this parameter is currently undetermined.

Bottom up control by nutrients typically exert strong control on periphyton biomass accumulation in New Zealand Rivers, particularly during summer (Biggs and Kilroy 2000). The proposed options will result in minor increases to background dissolved nutrients in a short (< 1.5 km) reach of the Puarenga Stream if wastewater is discharged to either of sites 1, 2 or 3 (Map 2). The potential for this discharge to contribute to periphyton growth will depend on the suitability of the substrate and the relative importance of other controls on periphyton growth in the stream, notably light (influenced by stream depth and optical transmissivity) and scouring (influenced by peak stream velocity during storm flow periods). The occurrence of stream alum dosing is expected to have a major effect on the potential for dissolved phosphorus additions to promote periphyton growth.

4.1.3. One–dimensional lake water quality modelling

4.1.4. Calibration and validation

4.1.4.1. Overview

Satisfactory model performance was achieved for the 2007–2014 study period with DYRESM–CAEDYM parameter values set to those assigned in a recent study by Hamilton *et al.* (2015), who calibrated the model for the period 2004–2007. The only exception was that it was necessary to reduce the maximum sediment release rate of ammonium nitrogen[7] to a value that was used in an earlier model application (Hamilton *et al.* 2012) to improve the model fit with measured total nitrogen concentrations. Details of the other parameter values are tabulated in Hamilton *et al.* (2015).

Overall, model performance was comparable with other model applications to Lake Rotorua (Burger *et al.* 2008; Hamilton *et al.* 2012; Hamilton *et al.* 2015), and with that of water quality model applications more generally (Arhonditsis and Brett 2003).

4.1.4.2. Water level

There was a very good match between modelled and measured water levels (Figure 15).

4.1.4.1. Temperature and dissolved oxygen

Similarly, there was a very good match between modelled and measured water temperatures and a good match between modelled and measured dissolved oxygen concentrations (Figure 16)

[7] Value reduced from 0.5 $g/m^2/day$ to 0.2 $g/m^2/day$.

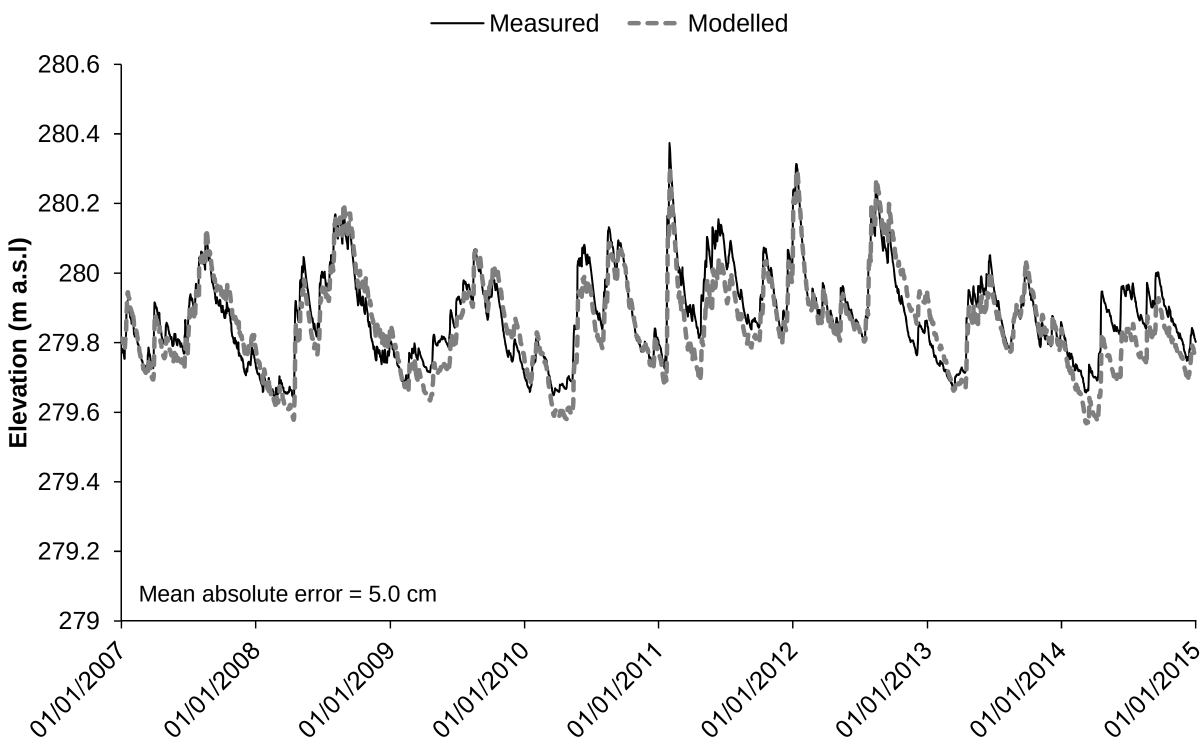

Figure 15 Modelled and measured water levels during the 1–D modelling study period

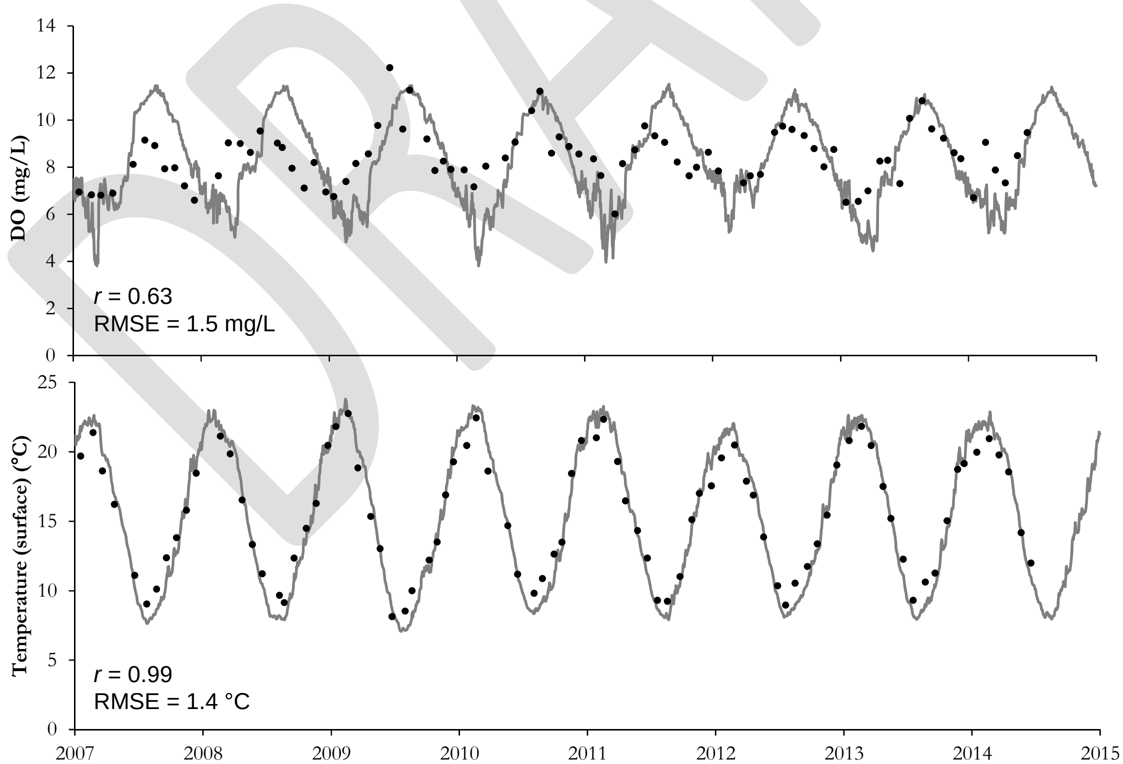

Figure 16 Comparisons of measured (circles) and modelled (line) surface concentrations of dissolved oxygen (DO) and temperature.

4.1.4.2. Chlorophyll *a* and nutrients

The model reproduced the magnitude of the chlorophyll *a* measurements reasonably well (Figure 17; Table 26) although inter–annual differences were not well–produced, most notably for the validation period (r = -0.06; Table 26). Both trends and magnitude were reproduced satisfactorily for most nutrient fractions, particularly total phosphorus and total nitrogen (Figure 17; Figure 18; Table 26). Relatively high concentrations of dissolved inorganic nitrogen fractions were observed in the measurement after 2011, and these were typically not reproduced (Figure 18).

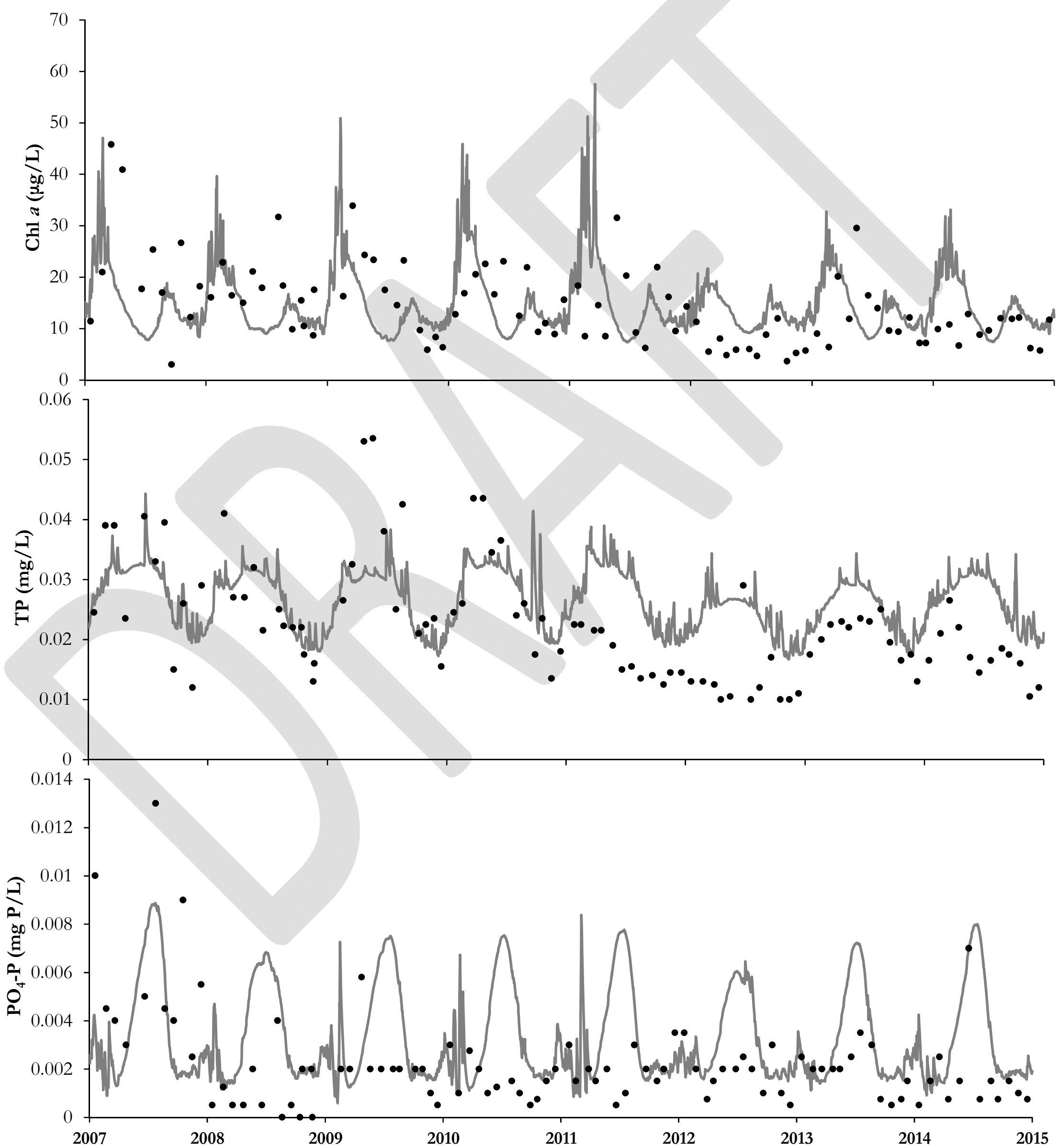

Figure 17 Comparisons of measured (circles) and modelled (line) surface concentrations of chlorophyll *a* and phosphorus fractions.

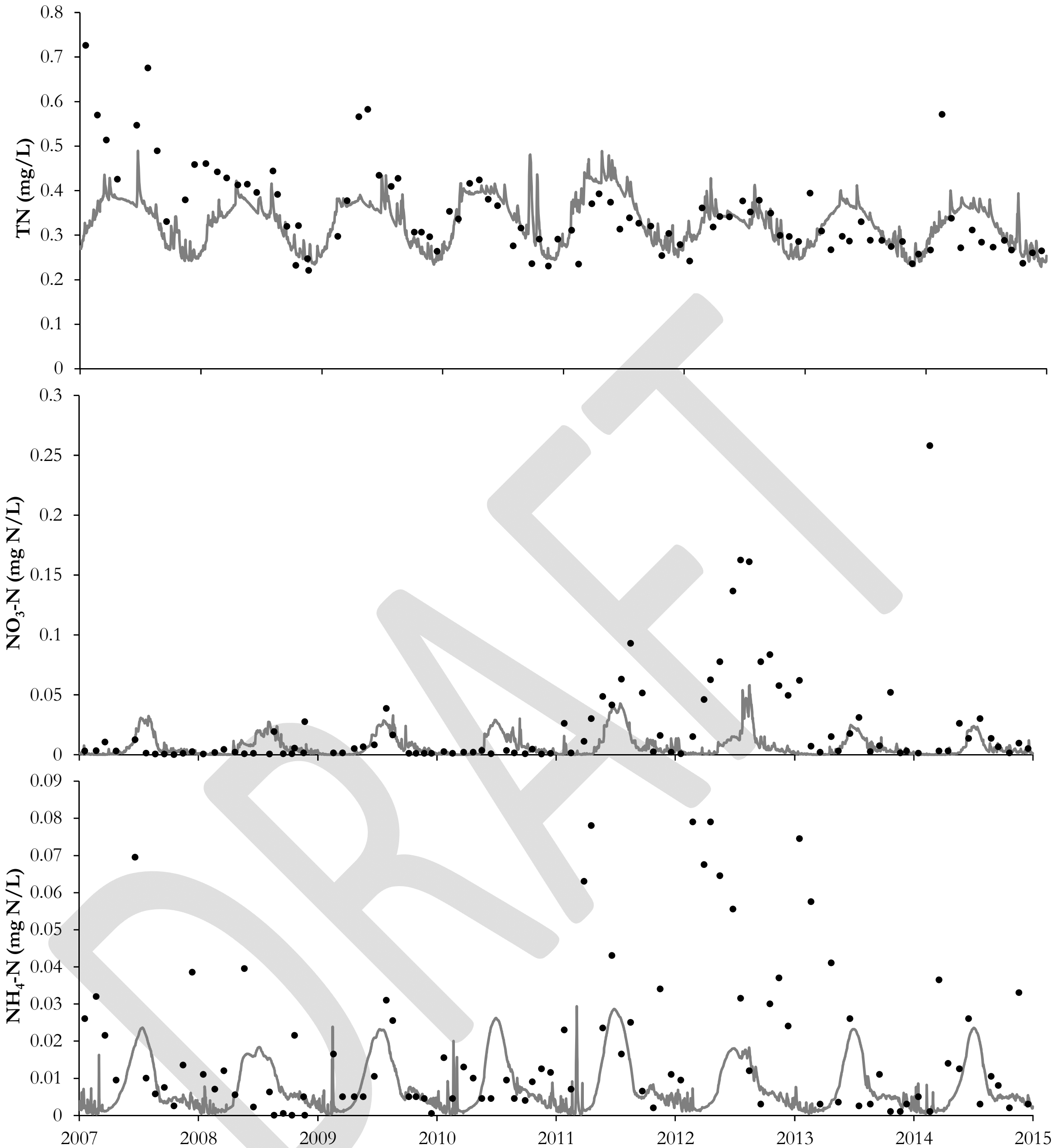

Figure 18 **Comparisons of measured (circles) and modelled (line) surface water nitrogen concentrations.**

Table 26 Model performance statistics for calibration (2007–2010) and validation (2011–2014) periods (chlorophyll *a* and nutrients).

SURFACE		2007-2010	2011-2014
Chl *a*	*r*	0.21	-0.06
(µg/L)	RMSE	10.07	8.91
	Mean error	-2.84	3.01
TP	*r*	0.67	0.49
(mg/L)	RMSE	0.01	0.01
	Mean error	0.00	0.01
PO_4-P	*r*	0.27	0.28
(mg P/L)	RMSE	0.003	0.002
	Mean error	0.001	0.001
TN	*r*	0.48	0.42
(mg/L)	RMSE	0.12	0.06
	Mean error	-0.06	0.02
NO_3-N	*r*	0.35	0.35
(mg N/L)	RMSE	0.01	0.06
	Mean error	0.00	-0.03
NH_4-N	*r*	0.22	-0.07
(mg N/L)	RMSE	0.014	0.031
	Mean error	-0.005	-0.017

[A comparison will be made of simulated and measured values for parameters measured at other depths (BoPRC provided the necessary data for this comparison on 15 June). Note though that model predictions have only been made for surface concentrations.]

4.1.4.3. TLI_3

Modelled annual TLI_3 values approximated measurements (Figure 19; Table 27), reflecting the satisfactory performance of the model with regard to simulating the three constituent parameters (Table 26). The model simulated inter–annual trends in the TLI_3, although the range of this variability was underestimated in the model simulations. In particular, error was high in 2007 when TLI_3 was underestimated by 0.36 units, and in 2012 when TLI_3 was overestimated by 0.47 units.

A measure of good model performance has previously been identified as an ability to model the measured TLI_3 value with an error of ≤ 0.1 units (Hamilton *et al.* 2012). This was only achieved for one year (2010). The eight–year average measured TLI_3 for the period was 0.8 units less than the modelled value.

Overall, the model was able to simulate both the magnitude of the TLI_3 for the eight–year period, and inter–annual trends with moderate success. This indicated that the model was suitable to examine effects of scenarios on lake trophic status over multi–year time periods.

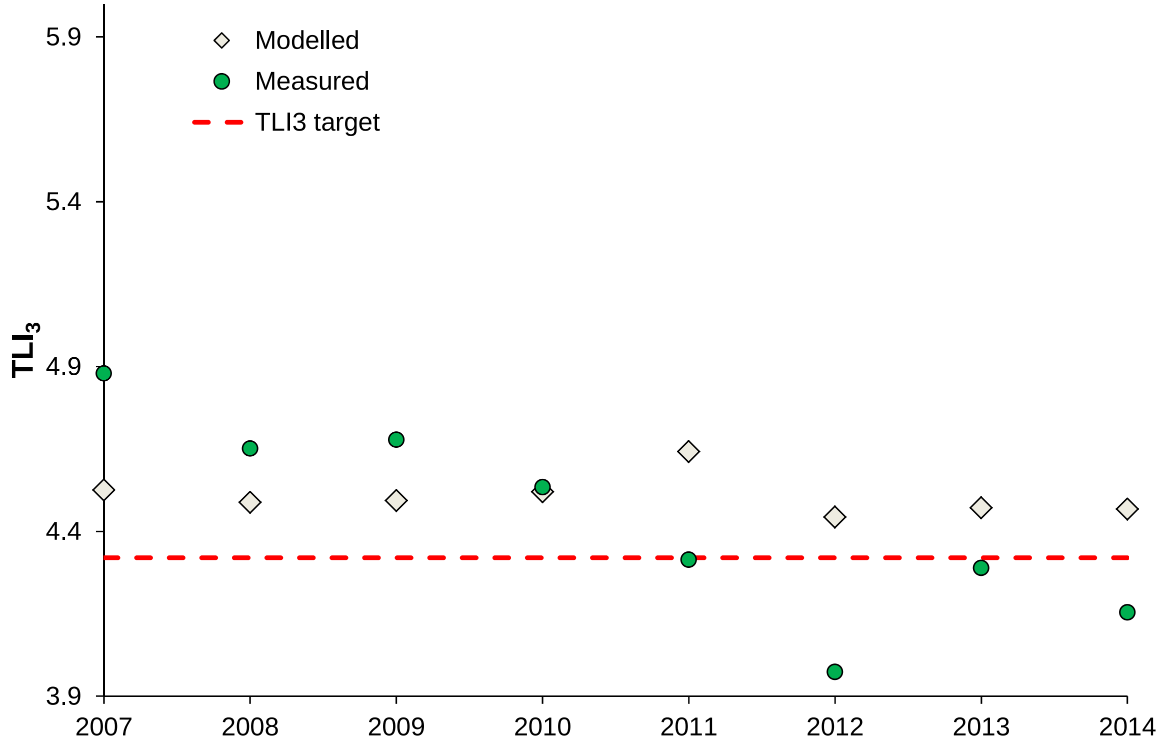

Figure 19 **Comparison of modelled and measured annual TLI_3. The dashed red line denotes the TLI_3–adjusted target for Lake Rotorua.**

Table 27 **Summary of model performance for simulation of annual TLI_3.**

	Calibration (2007-2010)	Validation (2011-2014)	Eight year period (2007-2014)
Measured TLI_3 (mean)	4.69	4.18	4.43
Modelled TLI_3 (mean)	4.51	4.51	4.51
r	0.28	0.66	0.24
RMSE	0.22	0.34	0.28
Mean error	-0.18	0.32	0.07

4.1.5. Modelled external loads

External nutrient loads that were represented in the baseline model scenario are presented in Figure 20 and Figure 21, alongside loads for individual treatment options. In broad terms, the figures show that the nutrient loads associated with each option are comparable with those of a major stream inflow.

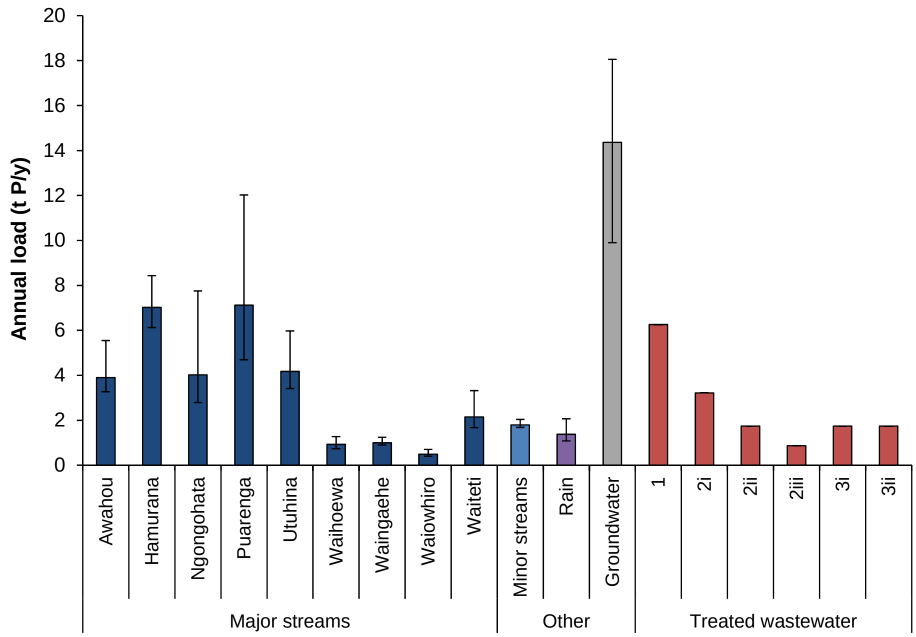

Figure 20 **Summary of external phosphorus loads used as forcing data in baseline model simulations. Puarenga Stream loads do not reflect attenuation by alum. Vertical lines denote between–year variations.**

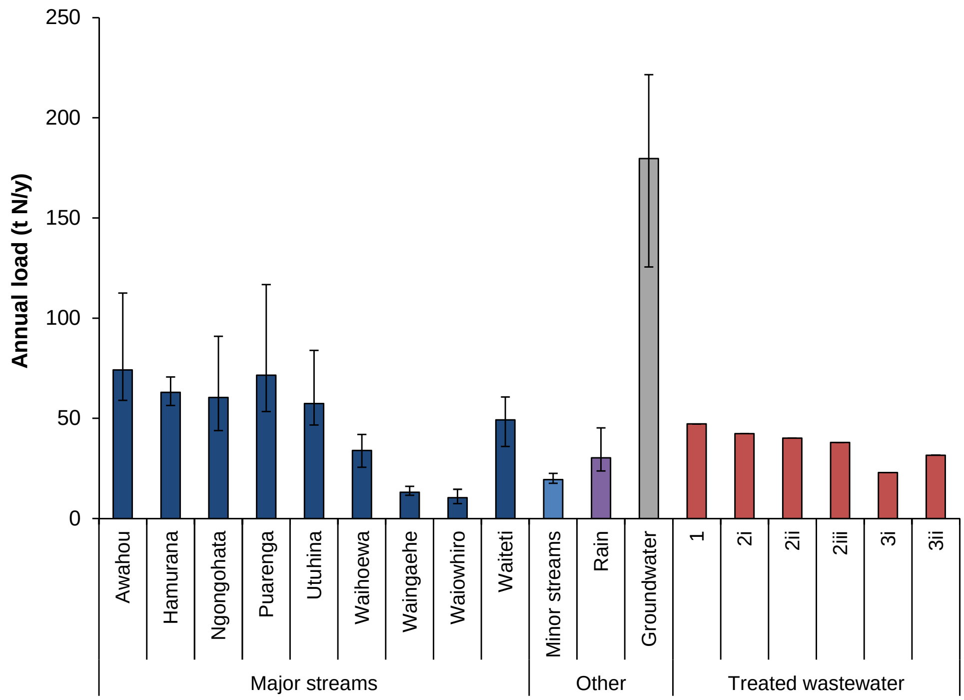

Figure 21 Summary of external nitrogen loads used as forcing data in baseline model simulations. Vertical lines denote between–year variations.

4.1.6. Simulated TLI_3 for scenarios

Simulated eight–year mean TLI_3 values for each scenario are presented in Table 28. Magnitudes of departure from the baseline scenario are presented in Figure 22.

These results indicate that all proposed scenarios of treated wastewater have a very minor effect on the TLI_3, relative to the baseline scenario. The eight–year mean results in Table 28 show that the treatment options result in an increase of between 0.01 and 0.02 TLI_3 units relative to the baseline scenario. Table 29 provides a summary for individual years of differences between model predictions relative to the baseline scenario. These data highlight differences between the individual treatment options in finer detail than the eight year–mean values presented in Table 28; however, the differences between the options are still very small, especially when compared with the magnitude of model error (Figure 19). The scenario involving addition of 'pure' water (#24; Table 28) highlights the occurrence of very minor water quality improvements associated with flushing effects. Results for this scenario provide insight into why some scenarios actually exhibit extremely minor improvements in TLI_3 for a small number of years (notably 2011) compared with the baseline period.

Neither of the scenarios involving either discharge of anoxic treated wastewater or discharge to the lake bed had an appreciable effect on modelled TLI_3. The scenarios involving removal of LTS loads highlight a very small effect due to this action; TLI_3 is 0.03 less for the baseline scenario when LTS loads are removed. By contrast, the scenarios involving cessation of alum dosing to streams had a much more substantial effect, with all of these scenarios resulting in an increase of ~0.5 TLI_3 units.

Table 28 Summary of predicted TLI_3 values. Each value is the mean of eight annual TLI_3 values for 2007–2014.

#	Scenario	Details	Mean annual TLI_3
-	Measured	Mean of annual TLI_3, 2007-2014	4.43
1	1D_0	Baseline with no wastewater discharge simulated	4.51
2	1D_1_Surface	Treatment option 1, discharge to surface waters	4.52
3	1D_2a_Surface	Treatment option 2a, discharge to surface waters	4.52
4	1D_2b_Surface	Treatment option 2b, discharge to surface waters	4.53
5	1D_2c_Surface	Treatment option 2c, discharge to surface waters	4.53
6	1D_3a_Surface	Treatment option 3a, discharge to surface waters	4.52
7	1D_3b_Surface	Treatment option 3b, discharge to surface waters	4.53
8	1D_2c_Bed	Treatment option 2c, discharge to lake bed	4.51
9	1D_3a_Bed	Treatment option 3a, discharge to lake bed	4.52
10	1D_2c_Surface - DO	Treatment option 2c, discharge to surface, no dissolved oxygen in wastewater	4.53
11	1D_3a_Surface - DO	Treatment option 3a, discharge to surface, no dissolved oxygen in wastewater	4.52
12	1D_0 - LTS	Baseline, Land Treatment System loads removed from the Puarenga Stream	4.48
13	1D_2c_Surface - LTS	Treatment option 2c, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	4.51
14	1D_3a_Surface - LTS	Treatment option 3a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	4.52
15	1D_0 - Alum	Baseline, alum effects (in-lake and in-stream) not simulated	5.06
16	1D_2c_Surface - Alum	Treatment option 2c, discharge to surface, alum effects (in-lake and in-stream) not simulated	5.05
17	1D_3a_Surface - Alum	Treatment option 3a, discharge to surface, alum effects (in-lake and in-stream) not simulated	5.06
18	1D_0 - LTS - Alum	Baseline, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	5.05
19	1D_2c_Surface - LTS - Alum	Treatment option 2c, discharge to surface, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	5.04
20	1D_3a_Surface - LTS - Alum	Treatment option 3a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	5.06
21	1D_30N_3P_Surface	Wastewater discharge of 30 t N/y and 3 t P/y to surface waters	4.52
22	1D_30N_3P_Surface - LTS	Wastewater discharge of 30 t N/y and 3 t P/y to surface waters, Land Treatment System loads removed from the Puarenga Stream	4.52
23	1D_30N_1.5P_Surface - LTS	Wastewater discharge of 30 t N/y and 1.5 t P/y to surface waters, Land Treatment System loads removed from the Puarenga Stream	4.50
24	1D_0 + 'pure' wastewater	Baseline with discharge of wastewater to surface waters that contains no nutrients	4.50

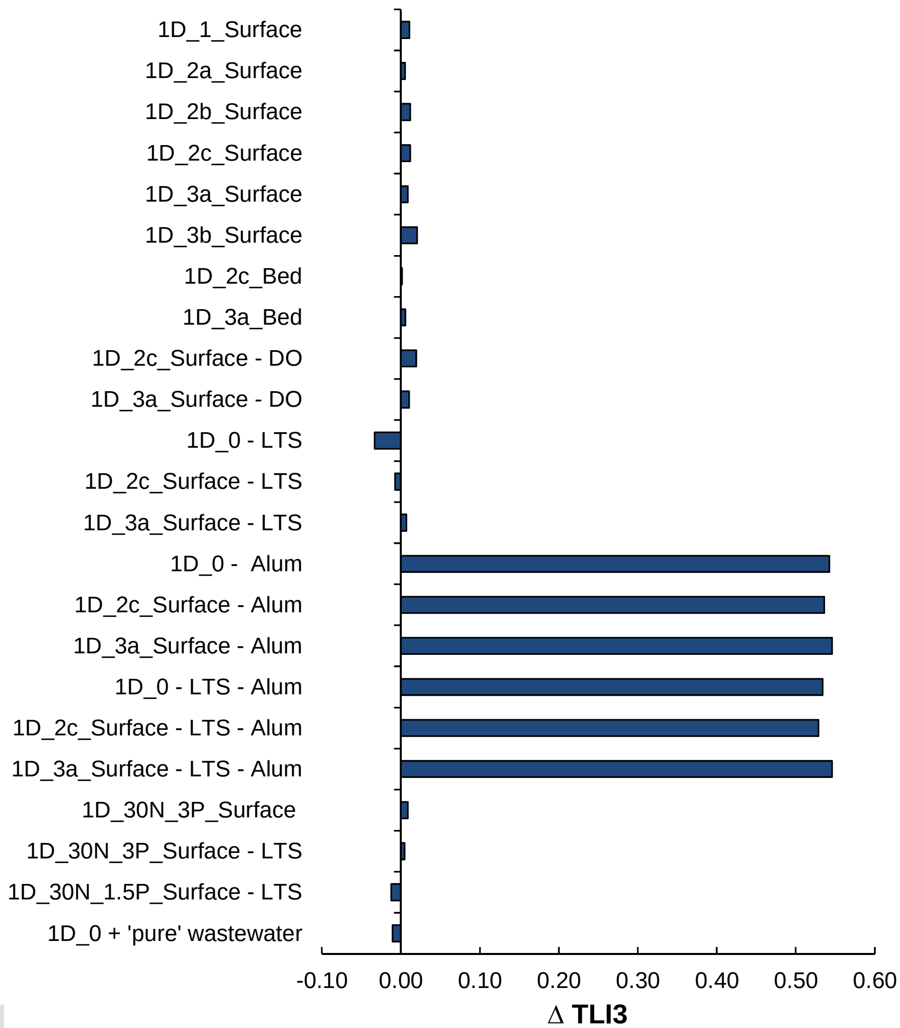

Figure 22 **Change in eight–year mean annual TLI_3 for each 1–D scenario (Table 19) relative to the baseline simulation (no wastewater added).**

Table 29 Percentage change in annual TLI_3 for each 1–D scenario (Table 19) relative to the baseline simulation (no wastewater added) for individual years. Shading is proportional to relative differences.

Year	1D_1_Surface	1D_2a_Surface	1D_2b_Surface	1D_2c_Surface	1D_3a_Surface	1D_3b_Surface	1D_2c_Bed	1D_3a_Bed	1D_2c_Surface - DO	1D_3a_Surface - DO	1D_0 - LTS	1D_2c_Surface - LTS	1D_3a_Surface - LTS
2007	0.71	-0.26	0.22	0.22	0.48	0.40	-0.04	-0.11	0.73	-0.32	-0.51	0.68	0.08
2008	0.23	0.92	0.16	0.16	0.50	1.01	0.68	0.66	1.29	0.56	-0.39	0.44	0.43
2009	1.21	0.44	1.13	1.13	0.98	0.22	0.83	0.88	0.56	1.63	-0.17	0.50	0.57
2010	-0.62	-1.25	0.07	0.07	-0.78	0.77	-1.73	0.34	-0.77	-0.45	-2.76	-1.45	-0.30
2011	-0.01	-0.14	-1.44	-1.44	-0.30	-0.03	-0.64	-1.05	-0.23	-0.82	-1.77	-1.33	-0.64
2012	-0.48	0.08	0.09	0.09	-0.02	-0.02	-0.20	-0.77	0.24	-0.30	-0.69	-0.76	0.21
2013	0.23	0.34	0.83	0.83	0.21	0.54	0.65	0.37	0.48	0.80	-0.22	0.34	0.20
2014	0.69	0.93	1.17	1.17	0.55	0.74	0.83	0.73	1.20	0.84	0.74	0.38	0.74
Mean	**0.24**	**0.13**	**0.28**	**0.28**	**0.20**	**0.46**	**0.05**	**0.13**	**0.44**	**0.24**	**-0.72**	**-0.15**	**0.16**

Table 29 continued.

Year	1D_0 - Alum	1D_2c_Surface - Alum	1D_3a_Surface - Alum	1D_0 - LTS - Alum	1D_2c_Surface - LTS - Alum	1D_3a_Surface - LTS - Alum	1D_30N_3P_Surface	1D_30N_3P_Surface - LTS	1D_30N_1.5P_Surface - LTS	1D_0 + 'pure' wastewater
2007.00	10.89	10.59	11.49	10.89	11.03	11.49	0.18	-0.50	-0.73	0.38
2008.00	11.88	11.64	12.33	11.85	12.05	12.33	0.21	0.54	0.45	0.76
2009.00	12.83	12.53	13.00	11.85	12.13	13.00	0.37	1.26	0.39	0.61
2010.00	11.38	10.90	10.58	11.42	10.11	10.58	-0.02	-0.02	-1.55	-2.03
2011.00	10.99	10.46	11.30	10.68	10.39	11.30	-0.49	-0.51	-0.99	-1.15
2012.00	13.31	12.93	13.06	13.96	12.83	13.06	-0.17	-0.42	-0.80	-0.72
2013.00	12.24	13.09	12.28	12.10	12.51	12.28	0.67	0.12	0.39	-0.06
2014.00	12.73	13.02	12.83	12.02	12.85	12.83	0.84	0.42	0.75	0.44
Mean	**12.03**	**11.90**	**12.11**	**11.85**	**11.74**	**12.11**	**0.20**	**0.11**	**-0.26**	**-0.22**

4.1.7. Predicted nutrient limitation status of phytoplankton

The values for the simulated nitrogen and phosphorus limitation functions that partly control phytoplankton growth (f(N) and f(P) respectively; see Section 3.3.2) were examined to gain insight into the relative importance of each of these nutrients in influencing phytoplankton biomass accumulation (Figure 23). Under the baseline scenario, the functions indicate that phosphorus limitation was slightly more dominant (the values were lower) for the majority of the period, although the values were frequently very similar during late summer to autumn. When the representation of alum dosing was removed, the values showed that nitrogen limitation was generally the most dominant.

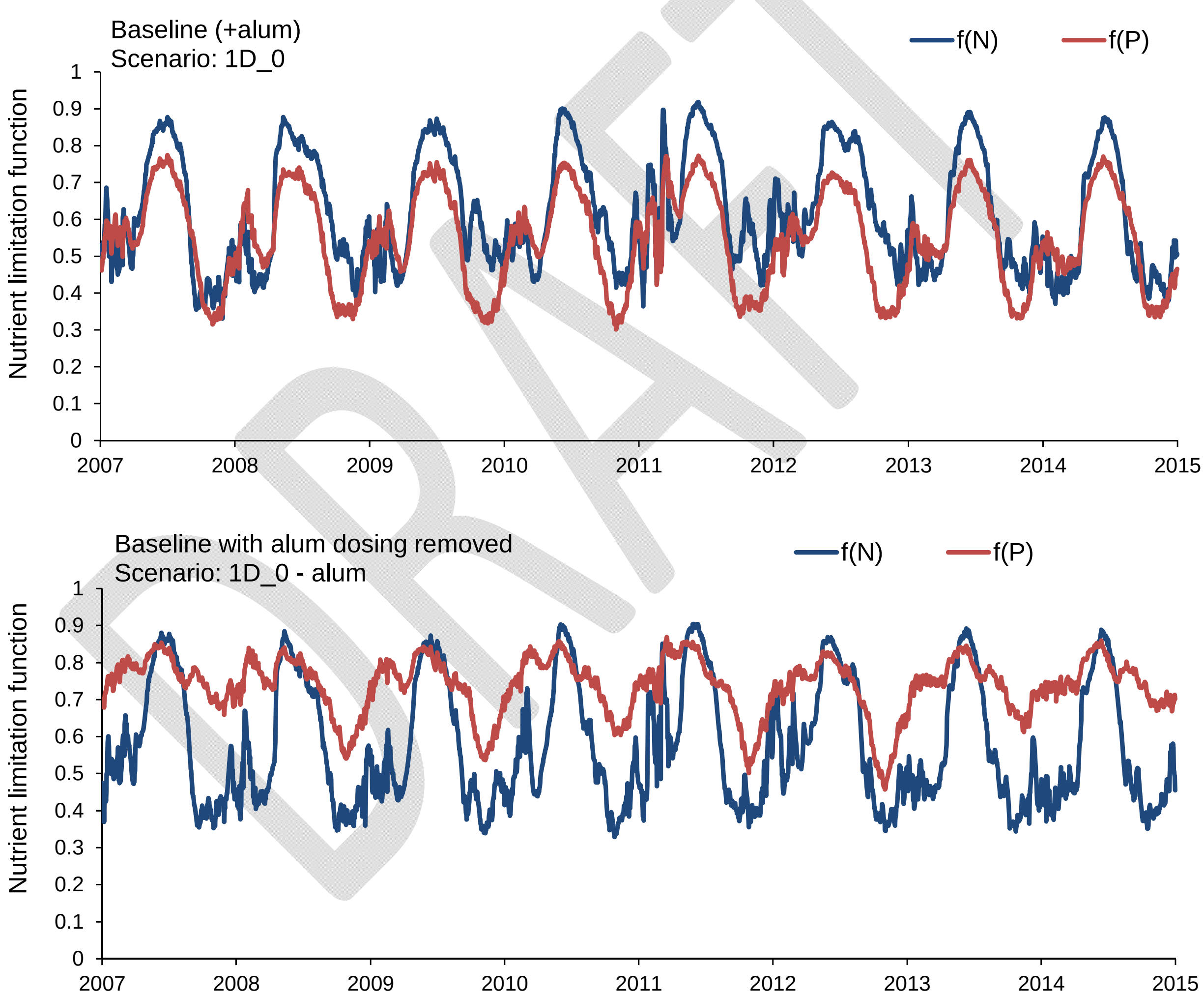

Figure 23 **Nitrogen and phosphorus limitation functions corresponding to the baseline scenario with (1D_0) and without (1D_0 - alum) alum dosing effects simulated.**

4.1.8. Comparison of concentrations with values designated in the NPS 2014 to assess in–lake effects on Ecosystem Health

Table 30 presents comparisons of model output with values designated for Attribute States for the three parameters that were assessed.

Consistent with the very minor effects on TLI_3 that were observed 4.1.6, no changes were predicted to occur to the modelled baseline Attribute States for each of the scenarios that involved addition of treated wastewater to the baseline scenario. Note that median concentrations of chlorophyll *a* were above (albeit often slightly) the designated 'national bottom line' of 12.0 µg/L for all scenarios (Table 16).

Table 30 Median surface water concentrations of chlorophyll *a*, total phosphorus and total nitrogen for each 1–D scenario (Table 19) for the period 2007–2014, with corresponding Attribute States based on the National Policy Statement for Freshwater Management 2014.

	Chlorophyll *a*		Total phosphorus		Total nitrogen	
Scenario	Median	Attribute State	Median	Attribute State	Median	Attribute State
Measured	12.13	D	0.022	C	0.32	B
1D_0	12.82	D	0.327	C	0.028	A
1D_1_Surface	12.81	D	0.336	C	0.028	A
1D_2a_Surface	12.93	D	0.341	C	0.028	A
1D_2b_Surface	12.81	D	0.339	C	0.028	A
1D_2c_Surface	12.79	D	0.333	C	0.027	A
1D_3a_Surface	12.79	D	0.334	C	0.027	A
1D_3b_Surface	12.84	D	0.333	C	0.027	A
1D_2c_Bed	12.79	D	0.333	C	0.027	A
1D_3a_Bed	12.79	D	0.334	C	0.027	A
1D_2c_Surface - DO	12.75	D	0.346	C	0.027	A
1D_3a_Surface - DO	12.76	D	0.341	C	0.027	A
1D_0 - LTS	12.60	D	0.322	C	0.027	A
1D_2c_Surface - LTS	12.70	D	0.334	C	0.027	A
1D_3a_Surface - LTS	12.79	D	0.337	C	0.027	A
1D_0 - Alum	16.03	D	0.447	C	0.060	C
1D_2c_Surface - Alum	16.13	D	0.455	C	0.060	C
1D_3a_Surface - Alum	16.03	D	0.456	C	0.060	C
1D_0 - LTS - Alum	15.65	D	0.439	C	0.061	C
1D_2c_Surface - LTS - Alum	15.85	D	0.449	C	0.060	C
1D_3a_Surface - LTS - Alum	16.03	D	0.456	C	0.060	C
1D_30N_3P_Surface	12.77	D	0.338	C	0.027	A
1D_30N_3P_Surface - LTS	12.81	D	0.332	C	0.027	A
1D_30N_1.5P_Surface - LTS	12.77	D	0.328	C	0.027	A
1D_0 + 'pure' wastewater	12.69	D	0.331	C	0.027	A

4.2. Three–dimensional hydrodynamic modelling

4.2.1. Validation of simulated temperature at monitoring buoy

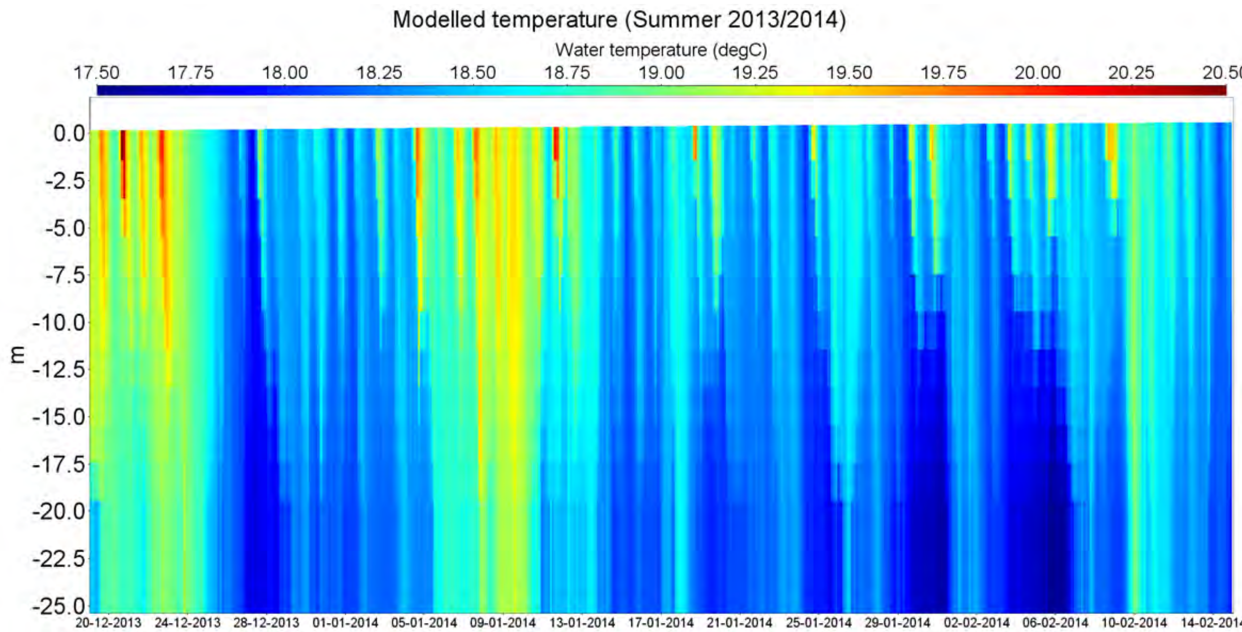

Figure 24 **Simulated water temperatures at lake monitoring buoy site during summer 2013/2014 modelling period, 19 December 2013 to 14 February 2014.**

[A comparison will be made of simulated and measured water temperatures for both the summer and winter periods]

4.2.2.Simulated tracer concentrations

Simulations showed that basin–scale circulation processes can dominate mixing processes in the lake under certain wind forcing conditions. These circulation processes exerted a strong influence on transport (advection) of the simulated tracer.

Figure 25 illustrates the alternate circulation processes that become dominant following periods of consistent wind forcing from either the SW or the NE. Simulations show that such continuous winds set up a double gyre feature within the lake. Following SW winds, currents to the north of Sulphur Bay flow to the east, and then follow the shoreline northwards towards the Ōhau Channel (Figure 25a). This flow is reversed following SW winds, with currents following the shoreline southwards from the Ōhau Channel (Figure 25b). These currents then converge with a second gyre in the western basin of the lake, with subsequent northwards distribution of water to the north of Sulphur Bay.

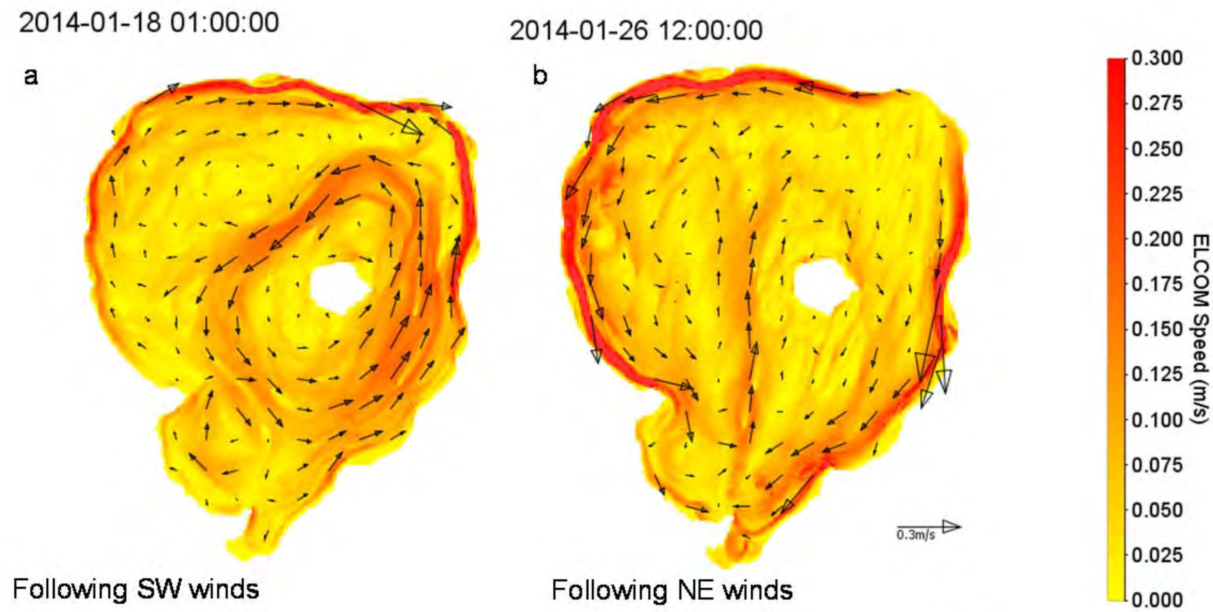

Figure 25 **Simulated water column average water speed and velocity vectors for two dates in Summer 2014. *a.* 18 January, following a 72–hour period of continuous SW winds, with a mean hourly speed of 6.2 m/s (maximum = 12.2 m/s). *b.* 26 January, following a 48–hour period of continuous NE winds, with a mean hourly speed of 5.6 m/s (maximum = 8.3 m/s; Figure 6).**

The potential effect of these two circulation processes on treated wastewater dilution was examined by simulating continuous wind forcing (4 m/s) from SW and NE directions. Simulated water column average concentrations of a conservative tracer (10 units) added to the wastewater discharge were examined to understand how the inflow is dispersed throughout the lake. When examining simulated concentration data, it is important to consider that computational constraints meant that the length of the simulation periods (two months) was considerably less than the mean hydraulic residence time of the lake (~1.5 years). This means that tracer concentrations are not at long–term equilibrium, and the mean concentration across the lake would therefore increase if the simulations were to run for longer. In addition, the conservative nature of the simulated tracer means that the concentrations are not reflective of analytes such as dissolved nutrients or microbes that are influenced by attenuation process such as biological uptake and settling, respectively.

Figure 26 shows simulated tracer concentrations that correspond to a discharge of water to the Puarenga Stream. Under a scenario of continuous SW wind, the simulated treated wastewater is predominantly transported along the eastern shoreline towards the Ōhau Channel. By contrast, under a scenario of continuous NE wind, the simulated treated wastewater is predominantly transported towards Rotorua city lakefront, with subsequent dispersal northwards into the central body of the lake.

Figure 27 shows simulated tracer concentrations that correspond to a discharge to the lake bed at a site 5 km to the north of Puarenga Stream mouth. The dispersion patterns are similar to those shown in Figure 28, with the key difference that the tracer is dispersed throughout the lake to a greater degree, i.e., it is more diluted. Thus, the scales and the simulation times are the same for both figures but the concentrations are generally lower throughout the lake in Figure 29. Furthermore, under a scenario of NE wind forcing, the lake bed discharge resulted in reduced accumulation of treated wastewater in the vicinity of Rotorua city lakefront, compared with the stream discharge option. This reflects the predicted northwards transport of treated wastewater from the lake bed discharge location.

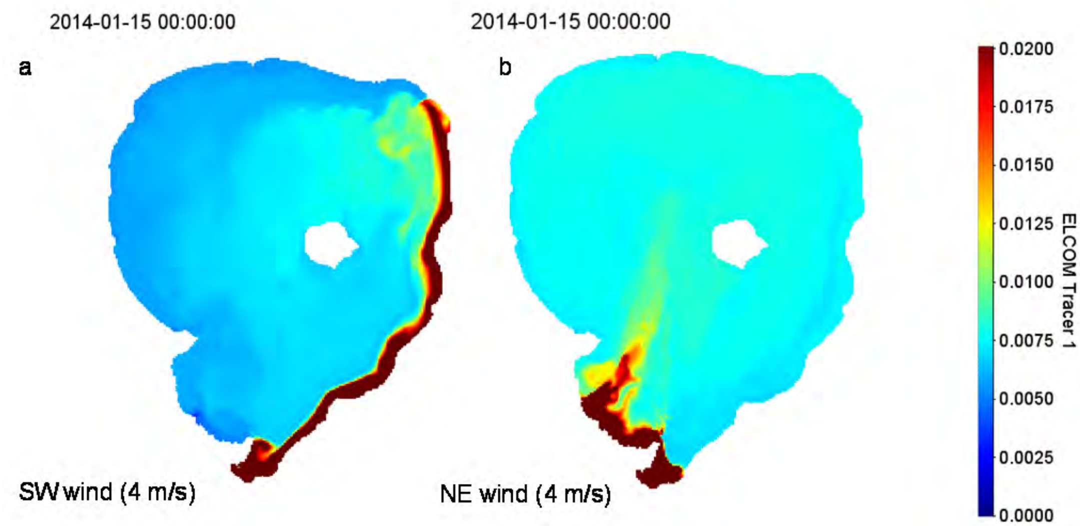

Figure 26 Comparison of simulated tracer concentrations for scenarios of consistent SW (a) and NE (b) winds during summer. The conservative tracer was assigned a concentration of 10 units, included in a discharge from a point at the Puarenga Stream mouth. Thus, dark red shading shows a water column average concentration of ≥ 0.2% treated wastewater. Plots are six weeks after the simulation started.

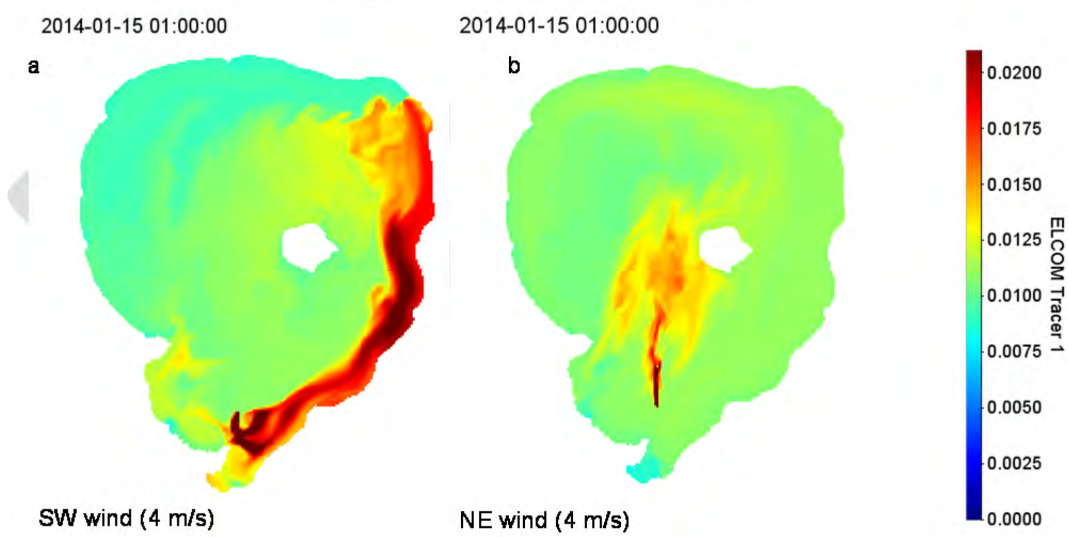

Figure 27 Comparison of simulated tracer concentrations for scenarios of consistent SW (a) and NE (b) winds during summer. The conservative tracer was assigned a concentration of 10 units, included in a discharge from a point 5 km north of the Puarenga Stream mouth. Thus, dark red shading shows a water column average concentration of ≥ 0.2% treated wastewater. Plots are six weeks after the simulation started.

Figure 28 compares simulated tracer concentrations between scenarios involving discharge to the Puarenga Stream (i.e., at Sites 1, 2 or 3; Map 2) or to a shoreline site (Site 5; Map 2).

Figure 29 shows relative differences in tracer concentration between stream discharge and lake bed discharge (Site 6; Map 2) with continuous wind forcing from either the NE or the SW.

[Further figures and text will be added to this section following completion of remaining simulations. Different ways to present the results (e.g., using log_{10} scales for tracer concentration data) will be examined. All scenarios are being re-run following information that was updated in recent weeks regarding predicted wastewater temperatures and the location of the proposed lake bed discharge site.]

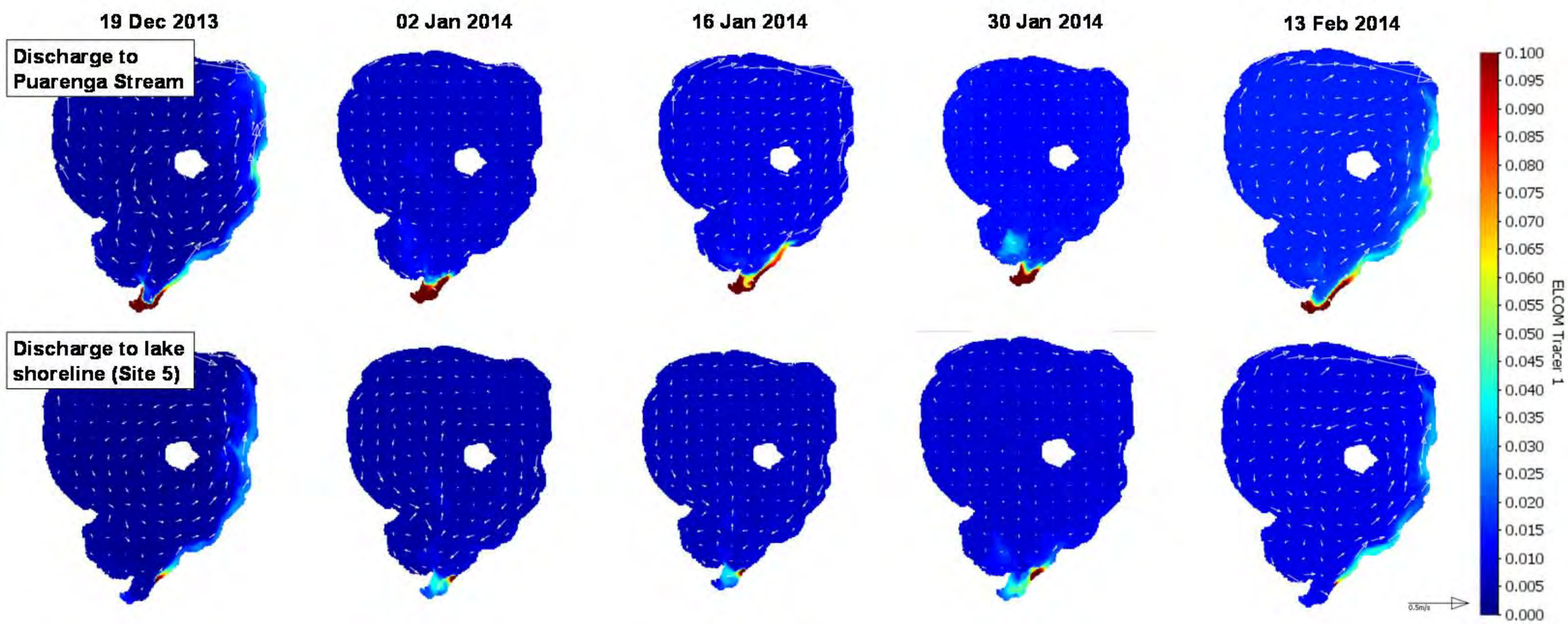

Figure 28 **Comparison of simulated tracer concentrations for scenarios of discharge to the Puarenga Stream and Lake Rotorua shoreline (Site 5; Map 2) during summer 2013/2014. Plots are at two–week intervals, commencing two–weeks after the simulation started. The conservative tracer was assigned a concentration of 10 units, thus dark red shows a water column average concentration of 1% treated wastewater.**

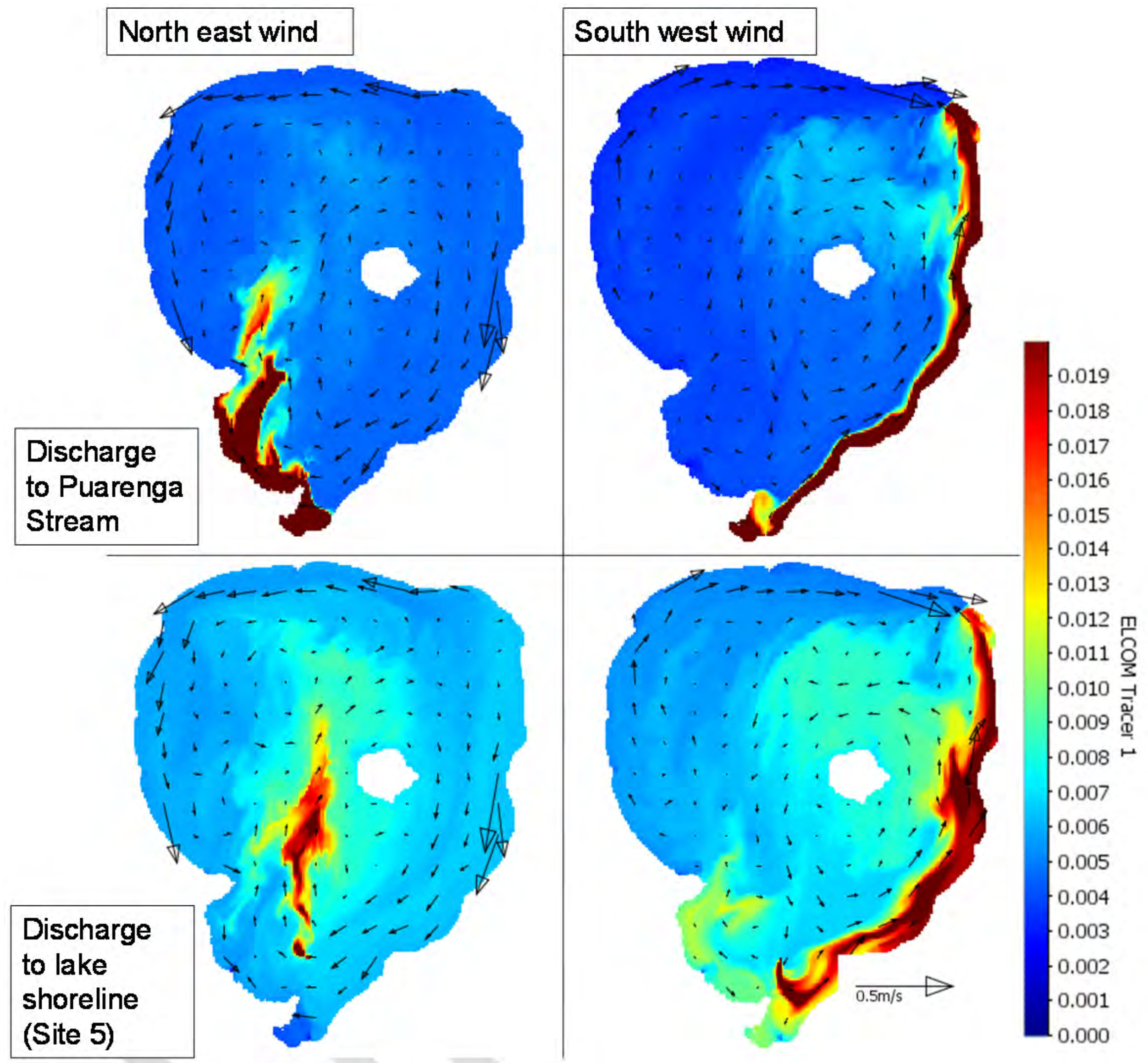

Figure 29 **Comparison of simulated tracer concentrations for scenarios of discharge to the Puarenga Stream and the lake (Site 6; Map 2) during summer 2013/2014 with consistent wind forcing (4 m/s) from the NE or SW. Images are water column average concentrations one month after the simulation started. The conservative tracer was assigned a concentration of 10 units, thus dark red shows a water column average concentration of 0.2% treated wastewater.**

5. DISCUSSION

Water quality modelling showed that treated wastewater addition is predicted to have very minor effects on lake trophic status. This result reflects the small to moderate contribution of each option to the overall external load to the lake (Figure 20; Figure 21), in addition to the high importance of internal nutrient cycling for controlling trophic status in the lake (Burger *et al.* 2007). Overall, the performance of DYRESM–CAEDYM that was quantified during validation indicates that there is relatively low uncertainty in this general result. Model validation did indicate, however, that the model underestimated the extent to which annual TLI_3 varied in response to inter–annual differences in forcing conditions (Figure 19). This suggests that the increases in TLI_3 predicted for each scenario may have been slightly under–estimated, although the magnitude of any such error is expected to be low. With regard to differentiating between the treatment options, it is important to note that the relative differences between the TLI_3 predictions for each treatment option (0.02 to 0.03 units) is much less than the mean error in annual TLI_3 predictions (Figure 19).

In terms of managing lake water quality to achieve and maintain TLI targets, the lack of marked difference between the six treatment options suggests that it is appropriate to carefully consider the economic costs associated with each of the options (not considered in this study) relative to the projected nutrient loads for each option (i.e., $/t of nutrient removed from the discharge). This information can be compared with other catchment actions designed to help achieve the target nutrient loads to the lake to differentiate between the options from a catchment–level perspective.

The 3–D simulations highlighted the potential for wind–driven basin–scale circulation processes to greatly influence how treated wastewater mixes throughout the lake, depending on prior wind conditions and the location of the outfall. The simulations showed that wind forcing can establish alternate double gyre features that are predicted to cause dominant transport of water added to the Puarenga Stream either in a north–eastern direction along the eastern shoreline (SW winds), or northwards towards the main body of the lake, with potential partial accumulation in the vicinity of Rotorua city lakefront (NE wind). Such double gyre patterns have been observed in large lakes elsewhere (Beletsky *et al.* 1999), although single gyres are more typical (Emery and Csanady 1973; Csanady 1977). In the case of Lake Rotorua, Mokoia Island apparently acts as an axis around which a second gyre rotates (Gibbs *et al., in prep.*). It is important to note that model predictions of lake currents have not been validated in the vicinity of Sulphur Bay, and the model configuration did not include fine scale details of bathymetric characteristics in the embayment, nor detailed representation of the temperatures of geothermal inflows that are likely to influence mixing process. Therefore there is moderate uncertainty in the predicted basin–scale circulation patterns, and moderate to high uncertainty regarding predictions of localized mixing processes in Sulphur Bay. Field studies undertaken elsewhere in the lake do, however, support the model predictions in relation to the gyre features. Abell and Hamilton (2015) used high frequency water sampling to study the propagation of the Ngongohatā Stream in the lake following a rainstorm. Measurements were collected up to a distance of ~5 km from the stream mouth, and they showed excellent correspondence between observed mixing processes and those simulated with ELCOM–CAEDYM. In addition, Gibbs *et al.*

(*in prep.*) validated ELCOM predictions using data collected using two acoustic Doppler current profilers (ADCPs) sited to the west and north of Mokoia Island. Modelled and measured current speeds and directions showed high correspondence, with the results also indicating that the double gyre pattern described above is highly–influential in controlling mixing processes in the lake at the basin scale. Further validation of mixing processes would require field studies in the vicinity of the proposed discharge locations. These could involve the deploying instruments such as ADCPs to measure current velocity, or drifter buoys to track currents. Alternatively, field tracer studies (cf., Gibbs *et al*. 2007) would be valuable to validate model predictions.

[Further discussion points will be developed following feedback at the TAG on 16 June. These will include discussion of nutrient limitation, in stream effects and alum dosing effects.]

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