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# Modelling the efficacy of in-lake and catchment remediation actions for restoration of a small, eutrophic lake

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# ABSTRACT

Land conversion from native vegetation to agriculture has been widespread in New Zealand over the past century. The resulting increases in nutrient runoff have led to eutrophication of many lakes and have resulted in harmful algal blooms, deoxygenation of hypolimnetic waters, and reductions in biodiversity. These problems occur in many of the Rotorua lakes including Lake Okaro, which is the most degraded of the twelve lakes. The lake has been subject to numerous restoration efforts over the past decade (2005-2014).

In this study, I linked a catchment and a lake model to address questions around the efficacy of restoration measures. The model was also used to compare the relative benefits of catchment and in-lake restoration actions on lake trophic status. The catchment model INCA (Integrated Nitrogen Catchment model) was applied to the Okaro catchment. The output from INCA was linked to a lake water quality model DYRESM-CAEDYM (DYCD). Catchment model simulations with INCA provided a satisfactory fit to measured discharge values in the two inflow streams (Pearson R=0.77-0.82). The model adequately simulated the major variations in nitrate ($NO_3$), total phosphorus (TP) and soluble reactive phosphorus (SRP). However, ammonium ($NH_4$) concentrations were not represented well for both the calibration and validation phases of the assessment.

Three scenarios were simulated using the calibrated version of INCA. This model was implemented to investigate the effectiveness of the artificial wetland (established 2006) and associated riparian planting by removing it in the catchment model simulation. Another scenario included fertiliser reductions and a third was to convert all pastoral land to dairy farming. The scenario involving the removal of the wetland/riparian area resulted in increases in total and dissolved nutrients. The fertiliser reduction simulations resulted in decreased $NH_4$ and SRP, both of which are a major input to dry-stock and dairy farms. The conversion of all existing pastoral land to dairy farming resulted in a 160% increase in $NO_3$-N concentrations while SRP increased by 500%.

The inputs from the INCA model were fed into the DYCD inflow file and compared with the previous analysis performed by Özkundakci et al. (2011). The calibrated

model based on INCA inputs represented an improvement over earlier published versions as a result of continuous simulation of catchment inputs and enhanced calibration.

The Trophic Level Index (TLI) was used to represent water quality changes in the lake in response to the different catchment management scenarios. The wetland/riparian removal increased TLI, with higher total nitrogen (TN) and total phosphorus (TP) concentrations in the surface and hypolimnetic waters, in addition to increased chlorophyll *a* concentrations in the surface waters. Average TLI for this scenario was 5.21, slightly greater (i.e. increased trophic status) from the baseline scenario result of 5.17. The fertiliser reduction scenario showed decreases in TN (average 26.9 mg $m^{-3}$) over the seven-year time frame (2005-2012). Concentrations of TP showed minor reductions, averaging 9.6 mg $m^{-3}$. Chlorophyll *a* concentrations declined indicating lower phytoplankton biomass, particularly lower cyanophyte concentrations. The TLI over the seven-year time frame (5.07) decreased in response to lower nutrient concentrations. The increased dairy scenario resulted in large increases in TN (average 39.4 mg $m^{-3}$) and TP (average 4.6 mg $m^{-3}$). These increases enhanced chlorophyll *a* levels indicating higher phytoplankton abundance due to increased nutrient availability. The TLI in this scenario was 5.25 compared with 5.17 in the baseline scenario.

From 2003-2014 chemical flocculants (alum) and sediment capping agents (Aqual-P®) were applied to Lake Okaro in an attempt to decrease phosphorus (P) concentrations and reduce lake trophic status. To achieve a better understanding of the efficacy of these applications required a model to assess P removal in the surface and hypolimnetic waters. Assessment of alum and Aqual-P was evaluated through a comparative analysis between the original calibrated DYCD model run and the measured data. From 2008-2012 the total average $PO_4$-P removal in the hypolimnion was 81 mg $m^{-3}$. This comparison was followed by a sensitivity analysis of nutrient release parameters in the model based on values in the literature. Reduction of the $PO_4$ release parameters from 0.016 to 0.008 g $m^{-2}$ $d^{-1}$ resulted in close fit of modelled outputs to measured data for $PO_4$-P concentrations in the water column. This alteration resulted in an average $PO_4$-P removal of 108 mg $m^{-3}$ from the hypolimnetic waters over the four-year simulation period from 2008 to 2012.

The results from the catchment and lake modelling simulations provide a basis for a detailed examination of the efficacy of internal and external lake remediation applications. Future management techniques should be aimed towards the limitation of external nutrients from the catchment. INCA simulations indicate that an additional artificial wetland installation could attenuate 60-100 kg $yr^{-1}$ of $NO_3$-N and $NH_4$-N as well as removing 5-20 kg $yr^{-1}$ of TP. This research provides scientists and lake managers with a tool enabling the relative impacts of individual lake management actions. This tool allowed for assessments at a time when multiple actions were being enacted in the catchment and in the lake.

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# General introduction and study site

## 1.1 Catchment processes

Catchments are natural drainage basins which contribute water to a specified section of a channel network based around topographic features (Wagener et al. 2007). They integrate aspects of the hydrological cycle, as well as a range of physical, chemical, and biological processes which can influence ecosystem integrity and functionality (Wagener et al. 2004).

### 1.1.1 *Hydrological processes*

Catchments are open systems driven by hydrological inputs and outputs (Dooge 1986; Black 1997). There are various ways to assess the hydrological cycle in catchments, but the conceptual model by Wagener et al. (2007) is a simplified method to classify hydrological stores and the transmission of water between each store. In this model the initial stage of the hydrological cycle pertains to precipitation (i.e. rainfall, snow and hail). Inputs are first partitioned into respective flow paths which include interception, infiltration, and percolation (Wagener et al. 2007). Interception is regulated by vegetative cover which can accumulate, absorb or transpire the water, before it falls to the catchment surface (Puncochar et al. 2012). Water that reaches the catchment surface will then infiltrate and percolate into soil and groundwater stores or move into other storage sites including channels and lakes (Wagener et al. 2007). The rate of water percolation into ground water stores is highly dependent on soil moisture characteristics including moisture deficits and soil composition (Franzluebbers 2002). If the infiltration capacity is limited then water will accumulate and form surface runoff which contributes to stream and rivers flows. Runoff, stream flows, and groundwater are examples of hydrological outputs from catchment sites (Wagener et al. 2007).

### 1.1.2 *Erosion processes*

Erosion is a common process by which materials are transported from land to water. Precipitation, wind, and surface flows can influence rates of erosion (Morgan 1995;

Morgan et al. 1998). Erosion influences nutrient and sediment stores in the landscape and deposits both into aquatic systems (Bilotta et al. 2010). Runoff is the most influential erosion process as it promotes dislocation and mobilisation of various materials (Dunne & Leopold 1978). This mobilisation can allow for nutrient input directly into aquatic sites (Elwell & Stocking 1988; Stoorvogel & Smaling 1990).

Phosphorus (P) is typically associated with direct runoff as it is generally mobilised in particulate form (McDowell & Wilcock 2008). Nutrient leaching most directly affects nitrate ($NO_3$) as this compound is highly soluble and readily leached into groundwater reservoirs (Stoorvogel & Smaling 1990).

### 1.1.3 *Anthropogenic alterations of catchment processes*

Modifications of catchment dynamics have become a prominent issue over the last century (Wagesho 2014). These modifications are often related to climate changes, urban growth, and agricultural intensification and expansion, all of which have reduced naturally occurring vegetation including forests and wetlands (Budyko 1974; Khoi & Suetsugi 2014). The removal of these vegetative features has limited precipitation interception capacities, resulting in increased surface runoff, modified base flows and increased periodic flooding events (Elfert & Bormann 2010). Modifications of discharge and flow cycles have enhanced erosion patterns resulting in increased sediment loss above natural deposition rates (De Roo 1998; Verstraeten et al. 2003). This pertains especially to agricultural catchments where sediment output may commonly be 1-2 fold higher than natural rates (Montgomery 2007). Increased sediment yields can influence a range of ecological processes including water characteristics, nutrient cycling, groundwater infiltration rates and available habitats for biological colonization (Ffolliott et al. 2013)

## 1.2 Catchment and hydrological modelling

Computer models have become an integral component of improving catchment management practices (Glavan & Pintar 2012). The aim of these models is to reproduce hydrological and biogeochemical dynamics in catchment sites and provide an in-depth interpretation of land use changes and their influence on lotic environments (Whitehead et al. 1998a). There are three principal types of models in common use; process based, empirical/statistical and conceptual (Das et al. 2008). Process based models range in complexity from simple input-output equilibrium models to more complex representations of multiple processes (Perrin et al. 2001; Hejzlar et al. 2009). The Hydrologiska Byråns Vattenbalansavdelning model (HBV) (Bergström 1976) is an example of a conceptual rainfall-runoff model which outputs discharge and nutrient characteristics based around various subroutines for meteorological data (Bergström 1976). The majority of process based models implemented in assessments utilize temporal and spatial data to assess hydrological and pollutant characteristics in catchments (Glavan & Pintar 2012). Many of these models incorporate Geographical Information System (GIS) interfaces which allow for specified analysis of various land uses and catchment characteristics (Glavan & Pintar 2012). The most accurate hydrological catchment models are reliable and robust, simulating the appropriate catchment processes without large scale parameterization which will otherwise limit the certainty of model simulations. The Soil and Water Assessment Tool (SWAT) (Gassman et al. 2007) and the Integrated catchment model (INCA) (Whitehead et al. 1998a) are examples of such hydrological models. SWAT and INCA have been used heavily because of their user friendly interface and ability to simulate discharge, nutrients, and sediment yield accurately in catchments (Whitehead et al. 1998a; Panhalkar 2014).

## 1.3 Lakes and their associated processes

### 1.3.1 *Eutrophication*

Eutrophication pertains to the enrichment of freshwater systems with nitrogen (N) and P, resulting in increased primary production (Conley et al. 2009). This is a natural occurring aging process (Carpenter 1981; Anderson et al. 2002) but has often become exacerbated by anthropogenic influences including diffuse and point source discharges from agricultural and urban land use (Carpenter et al. 1998; Chislock et al. 2013). Anthropogenic eutrophication is sometimes referred to as cultural eutrophication and has now become a major issue for aquatic systems globally (Di Tullo et al. 1993). Cultural eutrophication can be characterized by a range of symptoms including excess algal and plant growth (Huisman et al. 2005), reduced water quality and clarity (Conley et al. 2009), fish kills (Schindler et al. 2008) and anoxia in benthic and near-benthic layers (Leng 2009). Perhaps the most prominent issue pertains to the formation of prolific cyanobacterial blooms which produce a range of toxins (e.g. neurotoxins and hepatotoxins) which can harm both humans and animals (Anderson et al. 2002; Wood et al. 2005). These symptoms of cultural eutrophication have resulted in significant pressures to develop management practices and technological solutions that will avoid or mitigate nutrient enrichment and provide water of sufficient quality for consumption, economic development, recreation, and maintaining ecological integrity.

### 1.3.2 *Eutrophication of New Zealand lakes*

Cultural eutrophication has become a notable issue in lakes around New Zealand (Abell et al. 2011). This issue stems around the expansion and intensification of agricultural activity for over 100 years, resulting in excessive input of organic and inorganic N and P into waterbodies (PCE 2004). Increased nutrient inputs have resulted in algal blooms, anoxia in hypolimnetic waters and reduced biodiversity, all of which reduce ecosystem functionality (Abell et al. 2010). This is a regular occurrence in many of the Te Arawa (Rotorua) lakes of the central North Island. Lakes in this region have undergone widespread water quality changes due to land use conversions, resulting in elevated trophic level index (TLI) values in lakes. TLI is a New Zealand lake classification system based around primary production. It is calculated from nutrient concentration and chlorophyll *a* concentration, and Secchi disk depths (Burns et al. 1999). TLI for lakes in the region range from oligotrophic (TLI 2-3, low productivity) such as Lake Okataina to eutrophic (TLI 4-5, high productivity) such as Lake Okaro.

### 1.3.3 *External loading*

External loading refers to the input of organic and inorganic materials from catchments into surrounding waterbodies (Smith & Schindler 2009). This naturally occurring process has been exacerbated through anthropogenic modifications including land use conversions and utilization of fertilizers (Carpenter 2008). This process has negatively impacted lake trophic states by increasing primary production through the provision of excess nutrients (Abell et al. 2011). Urban and agricultural developments are responsible for large-scale releases of N and P through diffuse and point sources into aquatic systems (Carpenter 2005; Schindler 2012). Phosphorus has become a primary issue as many fertilizers have a high P content which can accumulate in soils (Mackay et al. 2014). These accumulations become mobilised during large rain events, which results in loss of P through transport of dissolved P or P adsorbed on sediments (Bennett et al. 2001).

The input of N has been used heavily influenced by artificial fertilizers as well because N is a major constituent of animal wastes. It can also degrade surface waters as well as leach into underground reservoirs, resulting in legacies of accumulated N (Abell et al. 2011). Nutrient contributions from external sources can directly influence water column concentrations or accumulate in lake bed sediments, contributing to further degradation through subsequent internal loading.

### 1.3.4 *Internal loading (Biogeochemical processes)*

Internal loading from lake sediments is a self-enhancing biogeochemical process which influences nutrient availability in lakes (Nurnberg & Peters 1984; Pettersson 1998). Internal loading occurs primarily in stratified water bodies including monomictic and polymictic lakes which undergo stratification continuously or periodically, respectively, during summer months. Stratification in this case is the result of temperature differences between the lake surface (epilimnion) and bottom (hypolimnion). The different temperatures mean the density of the water in the epilimnion is different to that in the hypolimnion and prevents mixing. Without mixing, biological respiration depletes oxygen in the hypolimnion and may cause the water to become anoxic (devoid of dissolved oxygen) if stratification is sufficiently long (Cooke et al. 1993). Anoxia results in the release of dissolved P and $NH_4$ from lake sediments into the overlying waters (Pettersson 1998; Łukawska-Matuszewska et al. 2013). The degree of internal loading is influenced by several mechanisms including resuspension, temperature, redox, pH, mineralization rates, and microbial processes (Søndergaard et al. 2003a). The most important of these factors pertains to redox reactions and microbial processes (Jensen et al. 2006; Smolders et al. 2006; Zamparas & Zacharias 2014). Redox reactions are reliant on P particles bound to iron, with P released in both oxic and, to a much larger extent, anoxic environments (Jensen & Andersen 1992). Microbial processes are related to rates of organic and inorganic particle accumulation in the sediment. Organic materials undergo conversion through mineralization processes which occur at higher rates at high redox potential (Søndergaard et al. 2003a). The consequences of elevated internal loading include eutrophication, harmful algal blooms, and reduced water clarity. Time lags of eutrophication can have large impacts on restoration activities rendering them ineffective for short to medium time scales.

### 1.3.5 *Restoration and management*

Lake eutrophication has become a significant global issue (Søndergaard et al. 2007) requiring the establishment of restoration methods to reduce external inputs from catchment land uses in addition to mitigating internal loading during stratification events (Carpenter & Cottingham 1997). Mitigation can include the utilization of artificial wetlands, riparian margins, detainment bunds, and management of nutrient loads through regulation of fertilizer applications (Cooke et al. 1993). The first three of these methods are known as resilience mechanisms as they maintain a static state when subjected to range of disturbances (Carpenter & Cottingham 1997). Artificial wetlands are becoming a popular restoration technique as they act as a natural filter for nutrient species and facilitate various processes to remove nutrients, including microbial denitrification (Johnston 1991).

The process of internal loading can directly influence the efficacy of external remediation applications. Various techniques have been used to reduce nutrient releases from the benthic layer. They include the application of chemical flocculants (e.g. alum), sediment capping materials (Hickey & Gibbs 2009), aeration, macrophyte harvesting, bio-manipulation (Shapiro & Wright1984; Carpenter & Kitchell 1993), and dredging (Schindler 2006). Both chemical flocculants and sediment capping materials have been utilized as they provide a nutrient removal mechanism in the water column in addition to capping releases from the bottom sediments. There are multiple considerations for these applications to be effective including environmental factors (pH and alkalinity) and the necessity for continual dosing to provide more effective results while external loading remains elevated. Remediation applications exhibit different efficacies which can be assessed through modelling technologies.

## 1.4 Lake modelling

### 1.4.1 *Introduction to lake modelling*

The utilization of lake models has become prominent in ecological evaluations (Trolle et al. 2012). Mathematical models have gained widespread popularity as they provide a quantitative representation of environmental processes which was previously unobtainable through generalized assessments (Trolle et al. 2012). Models of this nature provide temporal and spatial resolution which can represent ecological changes (Trolle et al. 2012). Original model developments in the late 1960's typically utilized empirical based models to simulate interactions between TP and chlorophyll *a* concentrations as well as applying steady state input-output systems which calculate nutrient concentrations based on mass balance equations (Mooij et al. 2010), negating other vital aspects such as light, food web interactions, and internal loading (Hamilton & Schladow 1997: Mieleitner & Reichert 2006). A general assumption of these models is that there is complete mixing, which in reality is only applicable for a certain period of time in most lakes (Imberger & Patterson 1981). Modern-day enhancements have resolved these issues by updating hydrological constituents which have incorporated layer components in addition to inflows, outflows, and mixing constituents which directly influence particle deposition (Hamilton & Schladow 1997). Ecological models have been modified around numerical representations which can simulate nutrient cycling, phytoplankton growth, and dissolved oxygen, all of which were not accurately simulated in previous models (Robson & Hamilton 2004). There is a range of hydrodynamic and ecological models applications including DYRESM-CAEDYM and CE-QUAL-W2. CAEDYM is a process based ecological model which simulates various biological and chemical processes (Robson & Hamilton 2004). This model is used with a hydrodynamic driver (DYRESM (1D) or ELCOM (3D)) to simulate nutrient cycling, phytoplankton biomass, and the microbial loop (Burger et al. 2008; Trolle et al. 2008). CE-QUAL-W2 is a coupled hydrodynamic model which simulates ecological processes through lateral averages similarly to DYRESM-CAEDYM (Cole & Wells 2008). This model has been widely used for management purposes as well as simulating key ecological processes aimed towards food web dynamics (Saito et al. 2001). Lake models provide ecologists with the necessary tools to forecast potential changes in lake water quality as well as formulating specific strategies aimed for future management.

### 1.4.2 *DYRESM-CAEDYM*

One of the most prominent and cited lake models is DYRESM-CAEDYM (DYCD) (Trolle et al. 2012). DYCD was one of the first models to incorporate numerical equations to represent various phytoplankton groups as well as stipulating various specifications relating to sediment release parameters for $PO_4$ and $NH_4$ (Hamilton & Schladow 1997). DYRESM is a one-dimensional hydrodynamics model based on vertical variations in temperature, density and salinity of the lake. This model can be run over varying time frames ranging from hours to decades. CAEDYM is the ecological component of the simulation and is based on simulations of time varying fluxes which influence different biogeochemical cycles. The ecological component of the model simulates different nutrients such as N, P, and carbon (C), as well as suspended sediment (SS) and dissolved oxygen (DO). The two models, DYRESM and CAEDYM, can be coupled to simulate water quality and provide vertical variations in variable concentrations, as well as formulating different scenarios for management uses.

## 1.5 Study site: Lake Okaro

Lake Okaro (38$^o$17’60”S, 176$^o$23’59.98”E) (Figure 1.1) is a small (0.32 km$^2$), shallow (max. depth 18 m), highly eutrophic crater lake in the Bay of Plenty region in the central North Island of New Zealand (Özkundakci et al. 2011). It is located in the Waiotapu geothermal area in the Central Volcanic Plateau, 27 km southeast of Rotorua and 2 km north of Rainbow Mountain (Özkundakci et al. 2011). The lake formed as a result of large-scale hydrothermal eruptions 900 y BP followed by a succession of lake filling events (Lloyd 1959; Healy 1975). The lake is monomictic, stratifying for about nine months of the year from spring through late autumn (Özkundakci et al. 2011). The catchment (3.89 km$^2$) is made up of moderate to steep terrain with undulating hills (Birchall & Paterson 2011). Current land use is 95% agriculture with sheep/beef, dairy and deer farms being the most prominent (Environment Bay of Plenty 2006). Other land uses include production forestry (native and exotic species), artificial wetlands and riparian margins along stream sites (Hudson & Nagels 2011; Özkundakci et al. 2011). The catchment contains a range of soil types including Kaharoa ash and Rotomahana silt loam (Birchall & Paterson 2011). Two low-discharge surface streams arise from the northwest part of the catchment. The lake has one outflow (Haumi stream) in the southeastern end of the lake. This outflow discharges into Lake Rotomahana (Environment Bay of Plenty 2006; Özkundakci et al. 2010).

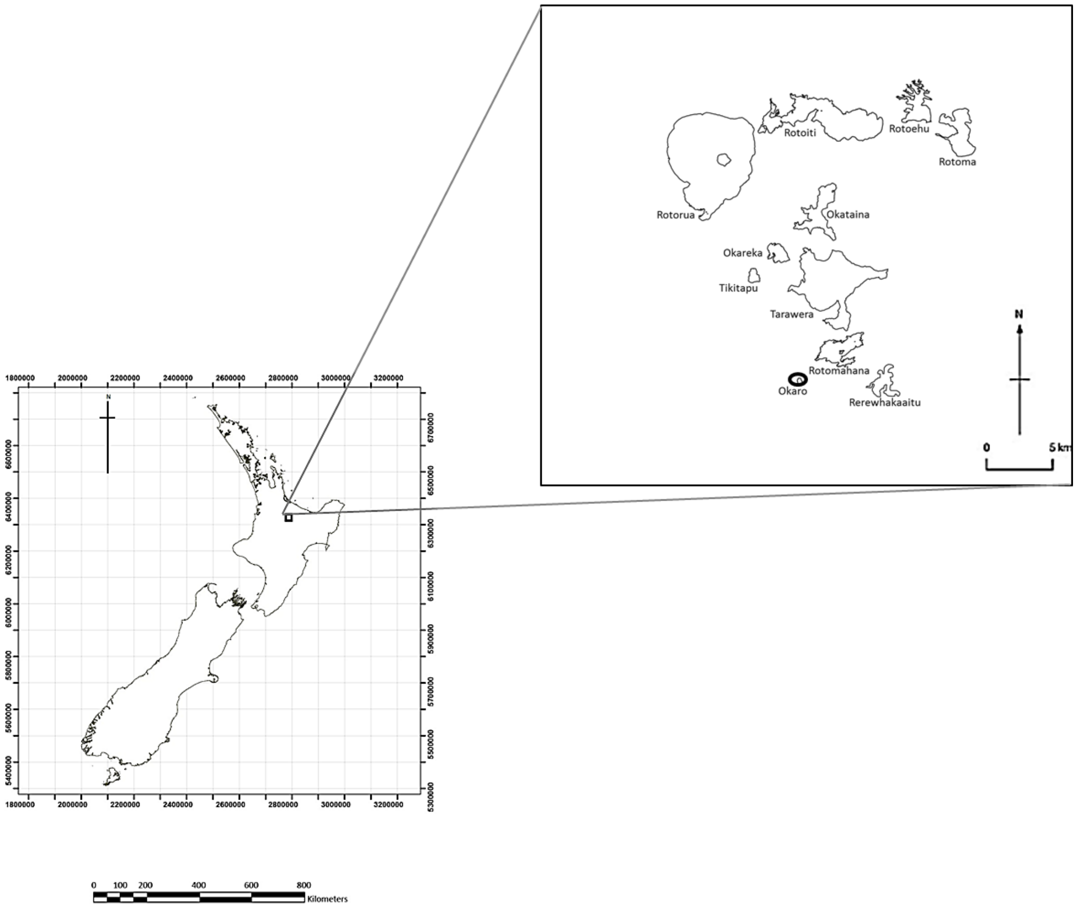



**Figure 1.1:** Location of the Rotorua lakes in New Zealand and the study site Lake Okaro (image adapted from Abell et al. 2011).

### 1.5.1 *Environmental degradation (1950-2015)*

Significant degradation of many of the Te Arawa lakes of Rotorua has become apparent since the 1950s, primarily due to land use change and increasingly intensive agricultural practices in the region (Özkundakci et al. 2010; Abell et al. 2011). Changes in the catchment include large-scale conversion of native vegetation into productive pasture, increased stocking rates, and increased use of fertilizers (Özkundakci et al. 2010). Expansion of agriculture has increased external loading of N and P into the Rotorua lakes. This loading is increasing due to the legacy of nutrients in the groundwater and soil water which have arisen in response to past land use change. An additional legacy occurs from internal loading from sediments, particularly with the occurrence of anoxia in the hypolimnetic waters during stratification (Paul et al. 2008; Özkundakci et al. 2010). In Lake Okaro, a 9-month stratification period beginning in spring (September to November) results in depletion of available DO in the hypolimnion, which initiates the release of nutrients, particularly P from the bottom sediments (Özkundakci et al. 2010). These nutrients are eventually distributed into the overlying water column when the lake mixes during winter months (June to August). Reductions in nutrient concentrations indicate general improvements in lake water quality. However, further investigation has been proposed to assess sporadic and harmful algal blooms and the formation of dead zones which are responsible for reduced biodiversity (Paul et al. 2008; Özkundakci et al. 2010).

### 1.5.2 *Environmental management and restoration*

In 2007, the TLI for Lake Okaro was 5.5 indicating it was highly eutrophic (Burns et al. 1999; Özkundakci et al. 2010). The Bay of Plenty Regional Council (BoPRC) implemented an action plan for the lake and the surrounding catchment (Environment Bay of Plenty 2006; Özkundakci et al. 2010) in conjunction with the National Institute of Water and Atmospheric research (NIWA). This action plan comprised multiple restoration measures including 2.3 ha of artificial wetland (Tanner et al. 2007), applications of aluminium sulphate (alum) and other sediment capping material (Aqual-P®) (Özkundakci et al. 2010), riparian planting, and the formation of the Okaro Catchment Lake Restoration Group (OCLRG) which focused on improving land use practices in the catchment (Paul et al. 2008; Özkundakci et al. 2010; Birchall & Paterson 2011). Water quality targets have been specified for the long-term (15 to 20 years) and are aimed towards reducing external loading, as well as short-term (4 to 15 years) actions aimed at reducing internal loading during anoxic periods (Environment Bay of Plenty 2006; Özkundakci et al. 2011).

*Artificial wetland*

A 2.3 ha artificial wetland (Figure 1.2) was implemented in 2006 to mitigate nutrient and sediment inputs from the two Lake Okaro stream inflows (Hudson & Nagels 2011). The aim was to attenuate contaminants including $NO_3$, $NH_4$, $PO_4$, and SS, as well as mitigating major storm-event discharges to the lake. A target was to remove an estimated 348 kg of TN and 16 kg of TP from entering the lake per year (Hudson & Nagels 2011). Several thousand plants were added to the wetland site including rush species *Eleocharis sphacelata* (tall spike rush), *Baumea articulate* (twig rush), and *Bolboschoenus fluviatilis* (lake clubrush) (Özkundakci et al. 2010). Estimates by BoPRC and NIWA indicate the wetland attenuated 41% of catchment TN (including $NO_3$, $NH_4$, and particulate materials) and 60% of TP to the lake in 2008.

a) b)

**Figure 1.2:** Artificial wetland (a) and riparian vegetation (b) inputs in the Okaro catchment (photos by David Hamilton).

*Alum and Aqual-P dosing*

In-lake remediation applications (intended to reduce internal nutrient loading) have included the application of the chemical flocculent aluminium sulphate (alum) and a sediment capping material known as modified zeolite (Aqual-P) (Özkundakci et al. 2010). These materials have been applied intermittently from 2003 to 2015. Alum dosing was first used in December 2003 when 4.59 tonnes of

$Al_2(SO_4)_3 \cdot 14H_2O$ was applied by boat to the surface of the lake in one day (Özkundakci et al. 2010). The concentration of 0.6 g $m^{-3}$ Al in the lake surface waters was at the lower end of the range compared with other applications globally (typically 0.05 g $m^{-3}$ to 30 g $m^{-3}$) (Paul et al. 2008). A low dose was implemented due to lake alkalinity which is typically low in Okaro (22 g $m^{-3}$ as $CaCO_3$) (Paul et al. 2008). The original alum application was associated with a rigorous monitoring program to identify effects on lake water quality and potential adverse effects to biological life. Additional alum dosing was applied in July and August 2012 (22.6 tonnes). Aqual-P (total 159 tonnes) was applied in August 2007 (110 tonnes), September 2009 (44 tonnes), and December 2011 (5 tonnes), as a slurry from boat or by helicopter.

*Okaro restoration group*

The Okaro catchment is one of five in the Rotorua district undergoing nutrient loss regulation (Rule 11) (Birchall & Paterson 2011). The rule implements a benchmark cap on N and P losses from different land use types. In Okaro, in response to this regulation the six catchment property owners collectively agreed to form the Okaro Catchment Lake Restoration Group (OCLRG). They focused on changing farming practices to limit nutrient loss from the catchment. The group was funded from the Sustainable Farming Fund (SFF). Overseer® is used as the primary farm-scale nutrient model to assess nutrient losses from the catchment sites (Özkundakci et al. 2010). Properties in the catchment consist of deer, dairy, and sheep/beef farms, each of which has its own unique fertilizer application rate. The Land-owners have hired private contractors to utilize the Overseer® program to adjust fertilizer application rates for individual farms. Property owners have limited livestock grazing as well access to riparian margins, to reduce nutrient exports to the lake (Özkundakci et al. 2010).

*Lake water quality restoration objectives*

The primary goal from the collective remediation applications was to reduce TLI from 5.5 (supertrophic) to 5.0 (eutrophic) (Environment Bay of Plenty 2006). It is specified under the Lake Okaro Action Plan (Environment Bay of Plenty 2006) that a reduction must occur across the four parameters of the TLI system (including Secchi disk depth (SD), TN, TP, and Chl *a*) (Özkundakci et al. 2010). In 2014 the lake TLI was 4.75, which is a notable improvement in lake water quality (Pers. Comm A. Bruere 2014).

## 1.6 Thesis objectives

This study has three primary objectives centred on catchment dynamics and water quality in small eutrophic Lake Okaro. The first involves simulating catchment discharge and nutrient and sediment loads at daily time scales and using this model output to evaluate the effectiveness of catchment remediation measures such as wetland construction and riparian planting. The second is to use the data from the catchment model as input to an existing hydrodynamic-ecological lake model to assess the impact of the wetland and riparian planting on lake water quality. Additional scenarios pertaining to fertilizer reductions and anthropogenic land use change were used to evaluate impacts of different catchment management regimes on lake trophic states. The final objective was to determine the efficacy of chemical flocculants and sediment capping on in-lake water quality improvements using DYRESM-CAEDYM. I used this small, eutrophic lake as the site to test these catchment and in-lake remediation methods using outputs from the catchment model which provided input to the lake model.

## 1.7 Thesis outline

The three primary objectives stated above have been addressed in four chapters. Each chapter explores different aspects of the research with some overlap to aid in general understanding. Chapter 1 provides a literature review of the catchment and lake ecosystem and examines factors responsible for their degradation. This chapter identifies issues affecting lakes globally and in New Zealand. Chapter 2 describes the application of the Integrated Catchment model (INCA) to Lake Okaro and an assessment of catchment remediation actions (specifically artificial wetland construction and riparian planting of streams). Scenarios involving fertilizer reductions and changes in catchment land use were simulated to understand their impacts on the lake ecosystem. Chapter 3 describes the coupling of output from the INCA catchment model with the previously calibrated DYRESM-CAEDYM model of Lake Okaro developed by Özkundakci et al. (2011). This work involved some recalibration of the model and extends the period of the model validation to further examine the effect of the remediation applications from the catchment to the lake scale, as well as evaluating in-lake remediation actions of alum and Aqual-P dosing. The final chapter provides a synthesis of the results in each of the previous chapters and helps to inform further restoration in Lake Okaro.

# Chapter 2
# Catchment modelling

## 2.1 Introduction

### 2.1.1 *Land use conversions globally and in New Zealand*

Pronounced land use changes worldwide have directly modified catchment dynamics and ecosystem functionality (Lake et al. 2000). Typically, modifications have been associated with urban and agricultural expansion which has removed indigenous vegetation resulting in increased runoff, reduced groundwater infiltration, and increased erosion rates. These factors have directly altered sediment and nutrient loads (Lake et al. 2000; Ramos & Martinez-Casasnovas 2006; Wagesho 2014). Globally, concern revolves around the loss of fertility due to the loss of organic materials which contribute to soil fertility and enhance plant growth (Hartanto et al. 2003; Glavan & Pintar 2012). Organic and fine sediments are removed through surface and direct runoff which can carry a range of sediment sizes into receiving waters, therefore impacting streams, rivers, lakes and estuaries (Khoi & Suetsugi 2014). Nitrogen (N) and phosphorus (P) are regularly applied to land to enhance crop growth, but are readily lost either through direct runoff or from leaching into underground reservoirs. Both sediment and nutrient relocation has directly influenced physical, chemical, and biological characteristics in catchments as well as affecting downstream habitats (Elfert & Bormann 2010).

New Zealand provides a good case study to assess the overall impacts of land use conversions on catchment dynamics (Abell et al. 2011). The country was once heavily covered in native vegetation including forests and wetlands, but has recently (particularly within the last 100 years) undergone significant agricultural expansion and intensification in lowland areas resulting in notable alterations of hydrological and erosion processes (Quinn & Stroud 2002; PCE 2013). Expansion of agriculture has resulted in large-scale inputs of N and P associated with artificial fertilizer applications and livestock excrement (PCE 2004). Dairy farming is one of the major contributors to increased nutrient loading to streams and rivers (McDowell & Wilcock 2008). Phosphorus fertilizers used in agricultural and dairy farming, in particular, have been important contributors to eutrophication (Conley et al. 2009). They tend to be cohesive on sediment. The phosphorus-enriched sediments may then be eroded and transported into streams, rivers, and lakes. Nitrogen fertiliser use is mostly associated with percolation into underground reservoirs and may create a legacy of groundwater-enriched N that hinders attempts to reduce N loads through restoration (Hamilton 2005).

### 2.1.2 *Management applications*

Increased discharge, nutrient, and sediment yields are responsible for the widespread degradation of streams, rivers, and lakes globally and in New Zealand (Abell et al. 2011). Implementation of various restoration actions are often required to combat erosion, as well as to limit other forms of nutrient releases into surrounding water bodies. There is a range of management techniques that can be applied to mitigate external loading of nutrients and sediments. The main methods utilized in New Zealand include land use change restrictions (Meneer et al. 2004; McDowell & Wilcock 2008), wetland enhancement or construction (Tanner et al. 2007), planting of riparian margins (Birchall & Paterson 2011), and the utilization of better land use management practices (Environment Bay of Plenty 2006; Schipper et al. 2010).

Construction of artificial wetland and creation of riparian margins are a mechanism to enhance resilience (Carpenter & Cottingham 1997) and have been implemented to limit both nutrient and sediment inputs into streams and rivers. They attenuate nutrients through sedimentation and microbial processes (Hudson & Nagels 2011). The application of better land use practices is focused on reducing N and P that would otherwise support productivity in aquatic sites. Each restoration measure can aid in nutrient and sediment retention, but their individual efficacy is difficult to predict and specific to the case at hand. To assess management applications, the implementation of catchment models has become important to understand the impact of restoration measures on catchment dynamics and to forecast future environmental changes under various changes.

### 2.1.3 *Modelling applications*

Catchment models incorporating hydrological and biogeochemical aspects have been utilized to identify changes in nutrient and sediment dynamics (Glavan &

Pintar 2012). These models can indicate the ecological state of catchment systems, the effects of land use change and climate change, and provide the necessary outputs for policy makers to formulate specific management plans (Todini 2007; De Girolamo & Lo Porto 2012; Glavan & Pintar 2012).

Catchment models incorporate various forms of environmental and spatial data to investigate hydrological, nutrient, and sediment processes in catchments (Glavan & Pintar 2012). Models can range in complexity from simple conceptual models such as Hydrologiska Byråns Vattenbalansavdelning model (HBV) to process based representations such as the Soil and Water Assessment Tool (SWAT), and the Integrated Catchment model (INCA). A range of other catchment models are available for specified usage including the Swedish Transport, Retention, and Källfördelning (TRK) model (Brandt & Ejhed 2003) and the ICECREAM model which focuses on P constituents (Rekolainen & Posch 1993)

Historically, modelling research was based around input-output systems, but the focus more recently has been on process based representation (Todini 2007). These process based models reflect hydrological, chemical, and biological processes which gives the user more interpretative power to assess ecological changes (Glavan & Pintar 2012). Models such as SWAT and INCA simulate time-varying hydrological and nutrient constituents in catchments, but differ with regard to the users focus and requirements. SWAT and INCA have been used in global applications to simulate discharges and nutrients in catchments including assessments by Granlund et al. (2004) and Zhu et al. (2012). For example, Granlund et al. (2004) utilized INCA to simulate hydrological and nutrient responses to varying land cover in a small Finnish catchment. This assessment focused on how land use affected nutrient generation. A SWAT-based assessment by Zhu et al. (2012) was directed towards the simulation of hydrological and nutrient characteristics in a watershed in Southern British Columbia, Canada. The primary focus of this analysis was to examine nutrient loads arising from different manure and fertilizer applications.

*Objectives of catchment modelling*

The aims of this component of the study were to identify the influence of remediation applications and hypothetical land use scenarios on catchment nutrient loads in the Okaro catchment. This required the utilization of a simple temporal based catchment model (INCA) to determine nutrients exported from the catchment. Simulation outputs then became the input to a pre-existing lake model to assess changes in-lake trophic level index (TLI) (see Chapter 3).

## 2.2 Methods

### 2.2.1 *Model descriptions*

The INtegrated CAtchment model (INCA) is a dynamic, stochastic, semi-distributed, process-based computing program developed at the University of Reading in the United Kingdom (Whitehead et al. 1998a; Wade et al. 2002; Wade et al. 2007). The INCA mode was created in 1995 and modified in 1998 (Jin et al. 2013), to provide an assessment tool to analyse N and P dynamics in catchments (Whitehead et al. 1998a; Crossman et al. 2013a). The model provides insight into both terrestrial and aquatic components, accounting for variations in N (Figure 2.1) and P (Figure 2.2) between each ecosystem type (Rankinen et al. 2004a).

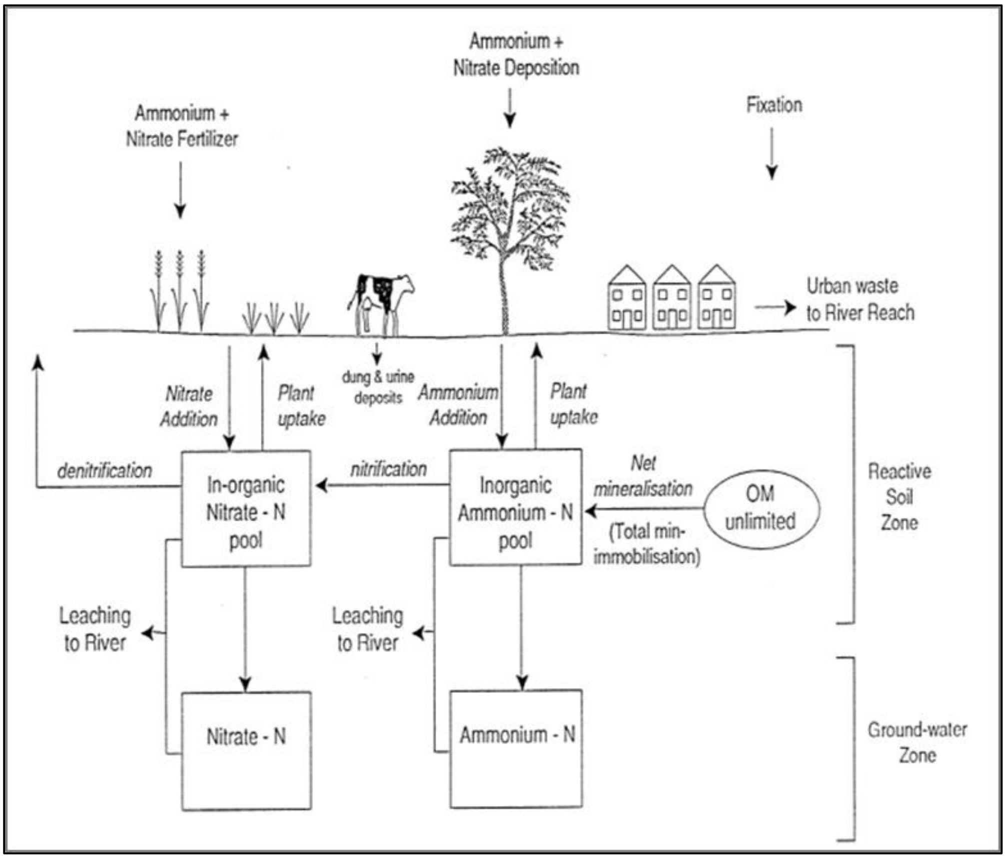



**Figure 2.1:** Land phase structure of INCA-N (Whitehead et al. 1998a; Wade et al. 2002).

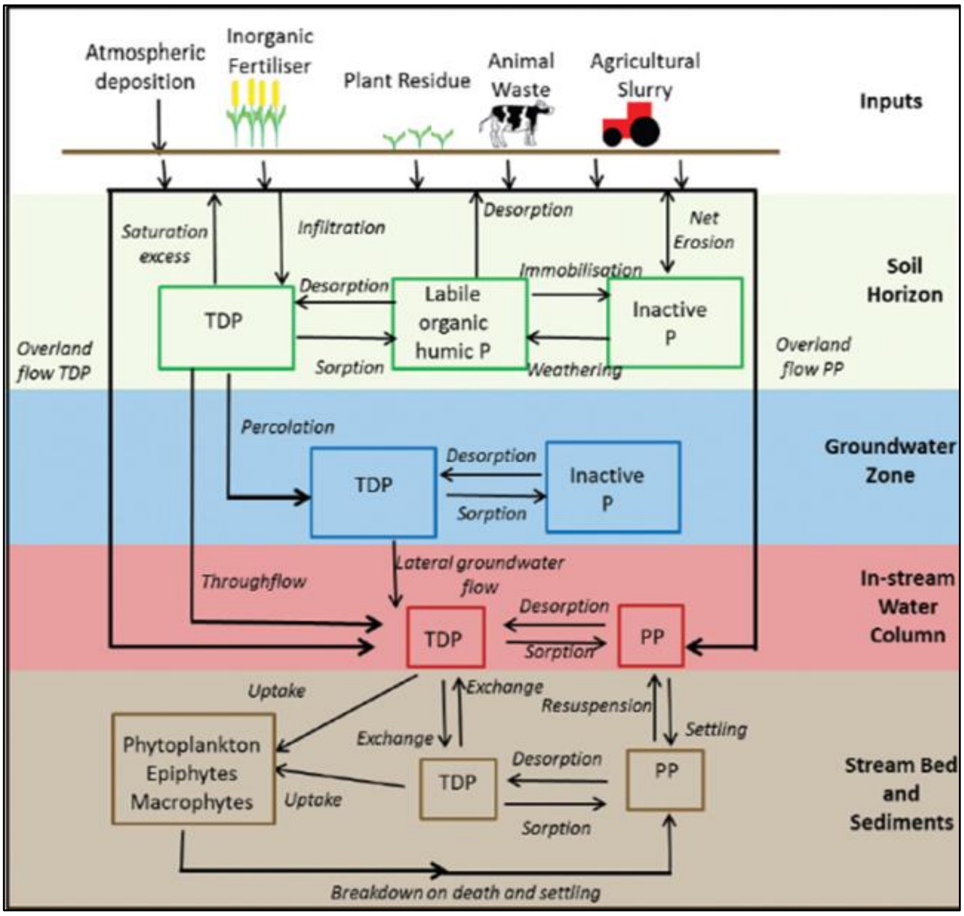



**Figure 2.2:** Land phase structure of INCA-P (Jin et al. 2013).

The INCA-N program includes a nitrogen input model, a hydrological model, and a stream nitrogen process model to simulate vertical and horizontal variations of $NO_3$ and $NH_4$ across catchments (Whitehead et al. 1998a; Flynn et al. 2002; Wade et al. 2002). The ability to analyse temporal variations provides a more in-depth perspective of plant/soil dynamics, hydrological systems, and biogeochemical fluxes (Whitehead et al. 1998a; Rankinen et al. 2004a). Additional simulated processes which directly influence N concentrations in the model include denitrification, nitrification, nitrogen fixation, and mineralisation (Wade et al. 2002). INCA-P simulates soluble reactive phosphorus (SRP), total phosphorus (TP), suspended sediment (SS), sediment constituents, discharge velocities, and proportions of particulate matter in stream discharge. Both INCA-N and INCA-P allow for a maximum of six land use types, providing flux and concentration estimates for each land use as well as discharge and nutrient time series for individual sub-catchments (Rankinen et al. 2004a). Both models provide output which includes daily and annual time series concentrations for $NO_3$, $NH_4$, SRP, and TP as well as discharges for the specified analysis time frame, and nutrient contributions for each reach and flow velocities (Whitehead et al. 2002). The overarching goal of the INCA models is to provide an easy-to use program which assesses changes in N and P distributions throughout the terrestrial and aquatic system when there are notable alterations to surrounding land uses, lotic structure, and global climate. The INCA models have typically been applied to large river catchments (e.g. River Thames, U.K.) with few applications to smaller stream catchments (Whitehead et al. 1998a; Whitehead et al. 2013). The models have been successfully implemented in sites throughout Europe (mostly in the UK, but also Finland and Spain), Canada, the United States, and Australia (Rankinen et al. 2002; Whitehead et al. 2013). Applications of the model have been aimed typically at examining land use change or global climate change effects on catchments (Jin et al. 2013).

### 2.2.2 *Model input data*

INCA requires input on physical catchment features, along with meteorological data, and measured data from stream sites for comparison with modelled outputs (Whitehead et al. 1998a). Catchment features for input to the model include watershed, catchment and sub-catchment boundaries, land uses, vegetation type, elevation, geology and reach length (Whitehead et al. 1998b). For this study, these features were obtained from GIS files provided by the Bay of Plenty Regional Council (BoPRC) and from the University of Waikato GIS mainframe. Climate data were sourced from CLIFLO meteorological database maintained by the National Institute of Water and Atmospheric Research (NIWA) (http://cliflow.niwa.co.nz), specifically the Rotorua airport climate station (1770) at 38.1092° S, 176.3172° E. Data extracted from CLIFLO included soil moisture deficit (SMD; mm), temperature ($^{o}$C), and precipitation (mm) spanning January 1, 2004 to May 8, 2014 (Figure 2.3). A hydrologically effective rainfall (HER; mm) measurement is typically provided by meteorological services in Europe, e.g. MORECS (Whitehead et al. 2002) but not in New Zealand. Thus the following equation from Rankinen et al. (2002) was used to derive HER to take into account changes in soil moisture storage, precipitation, and potential evapotranspiration:

$$HER = P - P_{ET} - \Delta S \quad \text{(Eq. 1)}$$

where,

HER= hydrologically effective rainfall (mm/day)

P= actual precipitation (mm/day)

$P_{ET}$= potential evapotranspiration (mm/day)

$\Delta S$= Change in soil moisture storage (mm/day)

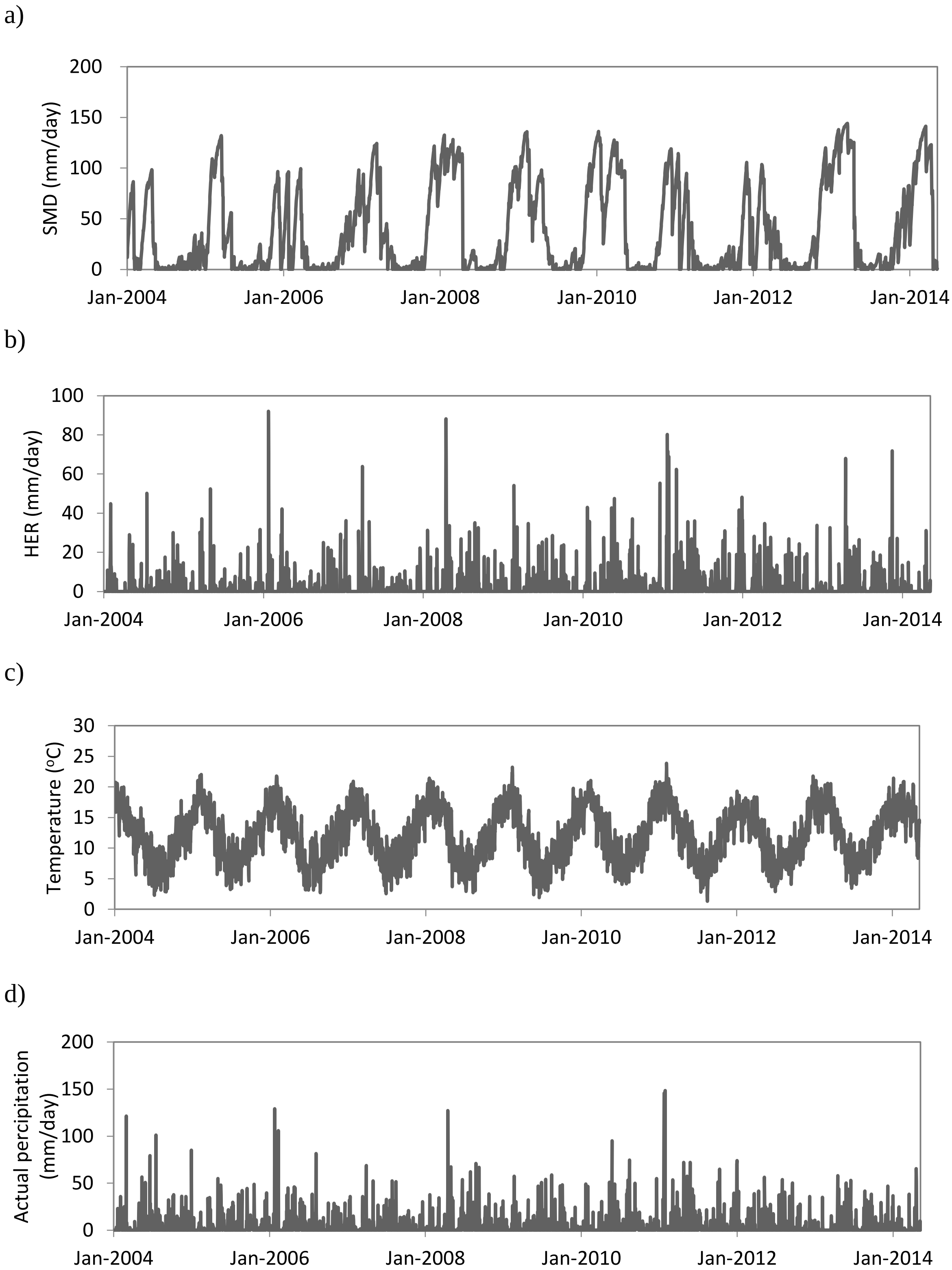



**Figure 2.3:** a) soil moisture deficit (SMD; mm/day), b) hydrologically effective rainfall (HER; mm/day), c) temperature (°C), and d) actual precipitation (mm) between January 1, 2004 and May 8, 2014 from the Rotorua airport climate station.

Measured stream data included discharge and concentrations of $NO_3$, $NH_4$, TP, SRP, and SS collected at monthly intervals from early 2007 to 2011. This data was provided by Dr Chris Tanner (NIWA) for calibration of the models. The data was reported previously in analyses of the nutrient removal capabilities of the wetland (Hudson & Nagels 2011) and for my study were used for calibration of the INCA model. Validation of the model was carried out with additional nutrient data provided by the BoPRC but was available only for the Northern stream site for the period 2011 to 2014. Validation data provided by the BOPRC was only available for the Northern stream site.

### 2.2.3 *Model setup*

INCA-N (version 1.11.10) and INCA-P (version 0.1.31) were applied to the catchment of Lake Okaro. The catchment contains six sub-catchments with six individual reaches (Figure 2.4). Sub-catchments were separated based on their respective elevations and watershed boundaries. A Southern stream and Northern stream represent surface discharges in the catchment. The Southern stream contained only one sub-catchment which directly fed into its respective stream site, while the Northern stream contained the remaining five sub-catchments (Table 2.1). Discharges in the initial simulation, using default values in the INCA models, differed from measured data, indicating the need for model calibration. This included the adjustment of discharge parameters in each sub-catchment as well as modification of stream width and reach characteristics to the specifications of the two Okaro streams. The Southern stream discharge was overestimated over the entire time frame after calibration. Discharges were overestimated due to a discrepancy between the Bay of Plenty's catchment boundaries and the actual watershed boundaries utilized in this analysis. As a result, this required an adjustment of the watershed boundary to emulate the Bay of Plenty's catchment boundary. This modification followed closely to topographical features presented in GIS resulting in resolution of Southern stream discharges. Six land uses were classified based on vegetative land cover from the GIS record. Land use included native vegetation, exotic vegetation, sheep/beef, deer, dairy, and wetland/riparian vegetation (Table 2.2). Each land use classification was allocated its own fertilizer application rates based on literature values.

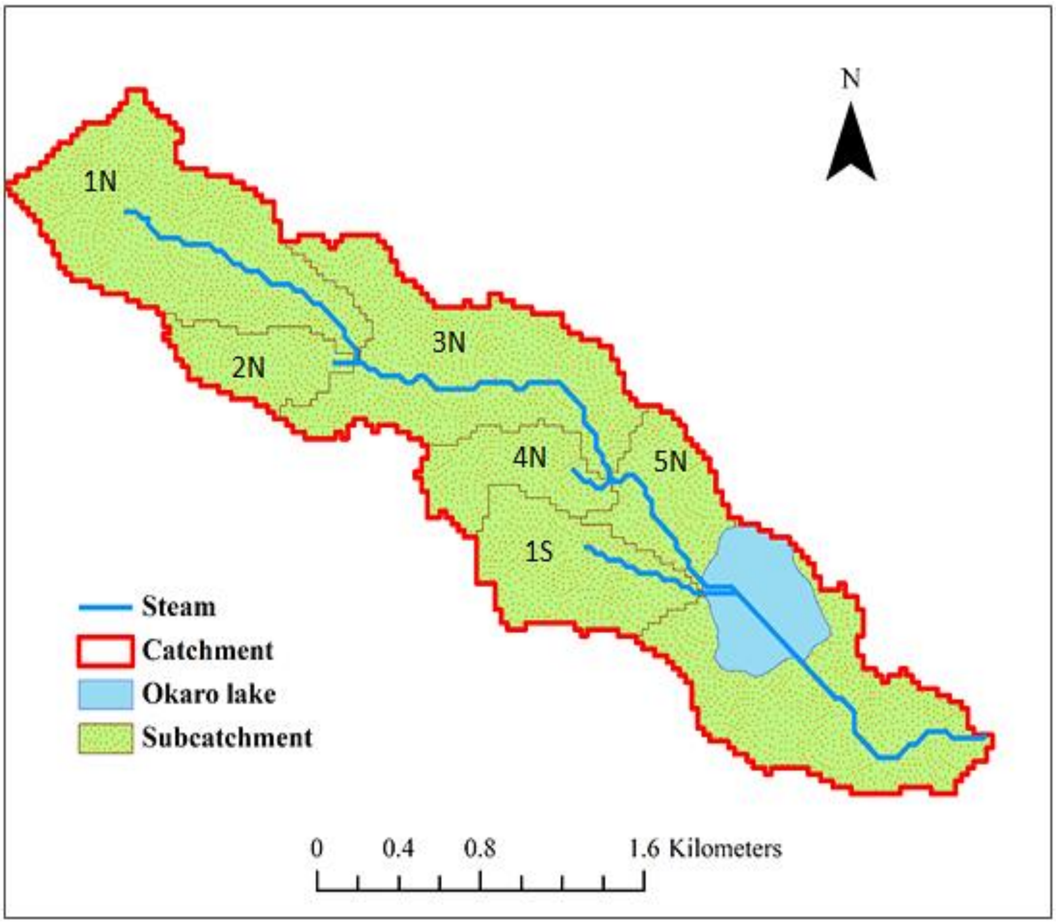



**Figure 2.4:** A GIS map of the five watersheds feeding into the Northern stream (1N-5N) and one watershed feeding into the Southern stream (1S) in the Okaro catchment (GIS data provided by BOPRC).

**Table 2.1:** Morphological characteristics of Lake Okaro sub-catchments.

| Stream | Size ($km^2$) | Reach length (m) |
|---|---|---|
| **Northern stream** | | |
| Upper catchment (1N) | 0.98 | 1275 |
| Second catchment (2N) | 0.24 | 90 |
| Third catchment (3N) | 0.93 | 1460 |
| Fourth catchment (4N) | 0.29 | 210 |
| Fifth catchment (5N) | 0.33 | 150 |
| **Southern stream** | | |
| South catchment (1S) | 0.475 (0.11 contributes directly to stream site) | 1070 |

**Table 2.2:** Percentage of land uses in the Lake Okaro catchment utilized in the INCA model.

| Land use | % in catchment |
|---|---|
| Native | 6 |
| Exotic | 0.7 |
| Sheep/Beef | 65 |
| Deer | 13 |
| Dairy | 10 |
| Wetland/riparian | 5.3 |

The meteorological data file contained the four climate based variables: SMD (mm), HER (mm), temperature ($^{o}C$), and precipitation (mm). Each of these variables was put into individual columns and spanned from the allotted time frame of January 1, 2004 to May 8, 2014. The measured discharge and nutrient data were formatted into sub-catchments in the file used for observation data.

### 2.2.4 *Calibration parameters*

The default parameters utilized for INCA-N and P runs were obtained from Whitehead et al. (1998a). These calibrations visually and statistically provided a weak match to observed discharges and nutrients ($R^2$<0.50) (Moriasi et al. 2007) indicating a necessity to calibrate the parameters (Eckhardt & Arnold 2001). Parameterization of flow variables through trial and error included alterations of baseline flows (proportion and velocity), individual land uses, and runoff coefficients. Velocity adjustments were set to the minimum values allowed by the model for the Southern stream to reduce discharge. Runoff coefficients were increased from the default values for each land use such as exotic vegetation, sheep/beef, deer, and dairy and reduced for native vegetation and wetland/riparian sites. All parameter adjustments were constrained within ranges given in the INCA manual (Butterfield et al. ND) and related research (Whitehead et al. 1998a; Wade et al. 2002).

Calibration of nutrient state variables was required for each of the six land uses. Each land use was given its own initial stream nutrient concentrations and fertilizer application rates derived from literature based values (when applicable) (Table 2.3 and 2.4). Natural vegetation, exotic vegetation, and wetland/riparian land use did not require an assigned fertilizer application rate. Biogeochemical processes including denitrification, nitrification, and nitrogen fixation were adjusted for each land use and values were compared with guidelines given in the INCA user manual (Butterfield et al. ND) and in Whitehead et al. (1998a).

**Table 2.3:** Nitrogen parameters utilized for each land use in the INCA-N model.

| Process | Native | Exotic | Sheep/beef | Deer | Dairy | Wetland/riparian | Literature | Manual range |
|---|---|---|---|---|---|---|---|---|
| **INCA-N** | | | | | | | | |
| Denitrification (m/day) | 0.003 | 0.003 | 0.030 | 0.030 | 0.030 | 0.680 | (Barton et al. 1999; Rutherford & Wheeler 2011) | 0.01-19.0 |
| Nitrogen fixation (kg N/ha/day) | 0.001 | 0.001 | 0.150 | 0.082 | 0.150 | 0.001 | (Parfitt et al. 2012) | 0-0.0001 |
| Nitrification (m/day) | 0.010 | 0.010 | 0.040 | 0.040 | 0.040 | 0.020 | | 1-54.00 |
| Mineralisation (kg N/ha/day | 0.001 | 0.001 | 0.030 | 0.030 | 0.030 | 0.040 | | 1-292.00 |
| Immobilisation (m/day) | 0.010 | 0.020 | 0.005 | 0.020 | 0.020 | 0.002 | | 0.0-1.0 |
| **Fertiliser** | | | | | | | | |
| Total addition over year (kg/ha/yr) | 0 | 0 | 80 | 78 | 136 | 0 | (Statistics New Zealand 2006; Pers. comm from M Allan 2014) | |

**Table 2.4:** INCA-P parameters for each of land use type.

| Process | Native | Exotic | Sheep/beef | Deer | Dairy | Wetland/riparian | Literature | Manual range |
|---|---|---|---|---|---|---|---|---|
| **INCA-P** | | | | | | | | |
| Freundlich isotherm constant (θ) | 9 | 9 | 15 | 15 | 15 | 9 | | N/A |
| Weathering factor (m/day) | 0.0001 | 0.0001 | 0.005 | 0.005 | 0.005 | 0.0001 | | N/A |
| Sorption coefficient ($dm^3/kg$) | 50 | 50 | 50 | 50 | 50 | 70 | | N/A |
| Equilibrium phosphorus concentration (mg P/l) | 0.50 | 0.50 | 1.4 | 1.4 | 1.6 | 1 | | N/A |
| **Fertiliser** | | | | | | | | |
| Yearly addition (kg/ha/yr) | 0 | 0 | 30 | 30.60 | 53.72 | 0 | (Statistics New Zealand 2006; Pers. comm M Allan 2014) | |

### 2.2.5 *Calibration and validation*

The INCA-N and INCA-P models were calibrated against the field data provided. Field measurements were provided for both the Southern and Northern streams allowing for four individual simulations (two N and P simulations each). For each stream site discharge, $NO_3$, $NH_4$, SRP, and TP were calibrated. Each model run was given an additional warm up period spanning from 2004 to 2006 eliminating any bias due to initial conditions and model 'spin up'. Calibration output statistics were compared to validation statistics to test the accuracy of the model under different forcing conditions corresponding to a different time period. Validation runs involved comparing all four nutrient variables provided by BoPRC spanning from December 2011 to December 2014. Data was only available for the Northern stream site so validation was not undertaken for the Southern stream.

Accepted modelling approaches include the utilization of statistics to identify overall model performance (Hamilton et al. 2012). Each statistic provides information on a different error constituent and contributed towards understanding of model accuracy. In my analysis the Pearson correlation coefficient (R), mean absolute error (MAE), root mean square error (RMSE), normalized mean square error (NMSE), and the normalized root mean square error (NRMSE) were calculated for the discharge and nutrient variables. The Nash-Sutcliffe efficiency (NSE) and coefficient of determination ($R^2$) was calculated for the discharge component of the calibration and was not included for nutrient assessments. The value of R (Eq. 2) indicates the strength of the linear relationship between two specific variables. Values of R lie between -1 and 1, with more accurate results closer either to -1 or +1 (Moriasi et al. 2007). The MAE (Eq. 3) is a statistical value utilized to measure the fit or proximity of forecasts to the calculated outcomes (Moriasi et al. 2007). The RMSE (Eq. 4) is the square root of mean squared error and assesses the overall variations in the model estimates against the measured data (Hamilton et al. 2012; Willmott & Matsuura 2005). NMSE (Eq. 5) is used to determine the amount of scatter in the comparison. This value provides an unbiased representation of model performance. Smaller values imply better model performance. The NRMSE (Eq. 6) can be utilized to investigate variations between the RMSE values. It was calculated through normalizing the RMSE value to the measured data. The NSE (Eq. 7) compares the extent of residual variations to that in the measured data. This value can range from -∞ to 1.0, with acceptable results occurring towards 1.0 (perfect fit) (Nash & Sutcliffe 1970). The coefficient of determination ($R^2$) (Eq. 8) is used to indicate variances between variables through the utilization of linear fit. Results for this statistic can be compared to guidelines set by Moriasi et al. (2007). Statistical performance criteria found in Table 2.5 are set using monthly estimates and differ for daily resolutions. The equations relating to each of the statistical variables are:

$$R=\frac{\sum_{i=1}^{n}(x_i-\bar{x})\times(y_i-\bar{y})}{\sqrt{\sum_{i=1}^{n}(x_i-\bar{x})^2\times\sum_{i=1}^{n}(y_i-\bar{y})^2}} \qquad \text{(Eq. 2)}$$

where;

*R*= pearson correlation coefficient,

*X*=values in the first set of data,

*y*=values in the second set of data,

*n*=total number of values.

$$MAE=\frac{1}{n}\sum_{i=1}^{n}|x_i-y_i| \qquad \text{(Eq. 3)}$$

where;

*MAE*=mean absolute error (same unit as variable assessed),

$x_i$= actual observations in time series (same unit as variable assessed),

$y_i$=estimated or forecasted time series (same unit as variable assessed),

*n*= number of non-missing data points.

$$RMSE=\sqrt{\frac{\sum_{i=1}^{n}(y_i-\bar{x})^2}{n}} \qquad \text{(Eq. 4)}$$

where;

$RMSE$= Root mean square error (same unit as variable assessed),

$Y$= observed values (same unit as variable assessed),

$x$ = modelled values (same unit as variable assessed),

$n$=number of values.

$$NMSE=\frac{1}{\sigma^2}\frac{1}{N}\sum_{k=1}^{N}(y_k\text{-}\hat{y}_k)^2 \qquad \text{(Eq. 5)}$$

where;

$NMSE$= Normalised mean square error

$\sigma 2$= estimated variance of the data,

$y_k$= mean of data (same unit as variable assessed),

$n$= total number of values.

$$NRMSE=\frac{RMSE}{X_{obs,max-}X_{obs,min}} \qquad \text{(Eq. 6)}$$

where;

$NRMSE$= Normalized root mean square error,

$RMSE$= Root mean square error (same unit as variable assessed),

$X_{obs,\ max}$= measured maximum (same unit as variable assessed),

$X_{obs,\ min}$=measured minimum (same unit as variable assessed).

$$NSE = 1 - \frac{\sum_{n=1}^{N}(o_n - s_n)^i}{\sum_{n=1}^{N}(o_n - \bar{o})^i} \quad i = 2 \qquad \text{(Eq. 7)}$$

where;

NSE= Nash Sutcliffe efficiency,

$o_n$= $n^{th}$ observed datum,

$s_n$= the $n^{th}$ simulated datum,

$\bar{o}$= the observed mean value,

$\bar{s}$= is the simulated mean on the day when the observation corresponds to,

$N$= the total number of observed data.

$$R^2 = \frac{\{\sum_{n=1}^{N}[(s_n-\bar{s})(o_n-\bar{o})]\}^2}{\sum_{n=1}^{N}(o_n-\bar{o})^2 \times \sum_{n=1}^{N}(s_n-\bar{s})^2} \qquad \text{(Eq. 8)}$$

where;

$R^2$= Coefficient of determination

$o_n$= $n^{th}$ observed datum,

$s_n$= the $n^{th}$ simulated datum,

$\bar{o}$= the observed mean value,

$\bar{s}$= is the simulated mean on the day when the observation corresponds to,

$N$= the total number of observed data.

**Table 2.5:** Statistical guidelines for hydrological model performances derived from Moriasi et al. (2007) (Table provided by Wang Me).

| Statistic equation | Constituent | Performance ratings | | | |
|---|---|---|---|---|---|
| | | Unsatisfactory | Satisfactory | Good | Very good |
| $\boldsymbol{R^2} = \frac{\{\sum_{n=1}^{N}[(s_n-\bar{s})(o_n-\bar{o})]\}^2}{\sum_{n=1}^{N}(o_n-\bar{o})^2 \times \sum_{n=1}^{N}(s_n-\bar{s})^2}$ | All | < 0.5 | 0.5 – 0.6 | 0.6 – 0.7 | 0.7 – 1 |
| $\boldsymbol{NSE = 1 - \frac{\sum_{n=1}^{N}(o_n - s_n)^i}{\sum_{n=1}^{N}(o_n - \bar{o})^i} \quad i = 2}$ | All | < 0.5 | 0.5 – 0.65 | 0.65 – 0.75 | 0.75 – 1 |

### 2.2.6 *Scenarios*

Scenario runs were based around remediation strategies and included fertiliser reductions and an additional hypothetical land use conversion scenario. This included the removal of a 2.3 ha artificial wetland and riparian margin (SC1), an external load reduction by altering fertilizer application rates (SC2), and conversion of the entire catchment into dairy (SC3). These scenarios were applied to assess nutrient fluctuations in catchments including their distribution into aquatic compartments which directly influence stream and groundwater quality, as well as their potential impacts on the lake. Data from each scenario was extracted and compared to baseline INCA runs to indicate variations between the two simulations.

## 2.3 Results

### 2.3.1 *Discharge calibration*

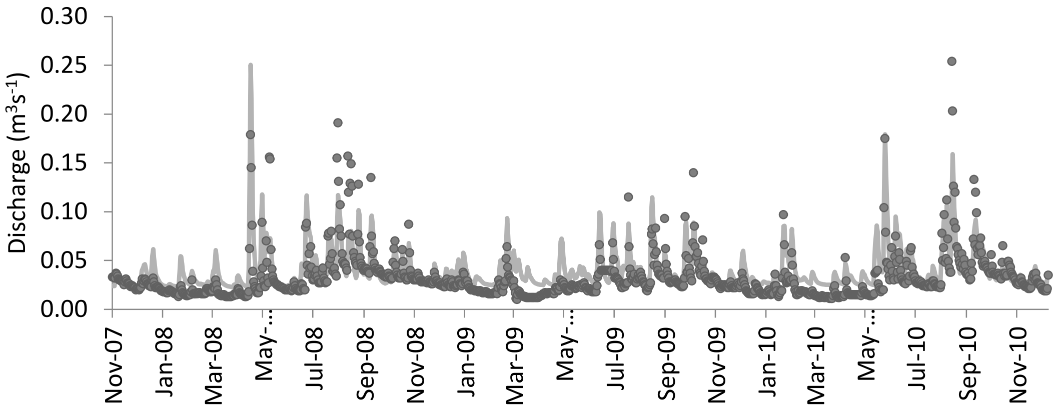
*Northern stream*

Northern stream discharge parameters were calibrated from November 1, 2007 to November 1, 2010. The most influential variables included direct discharge and discharge proportions. Comparative assessment of the measured and model outputs in the newly run hydrograph indicated a satisfactory final calibration (Pearson R correlation coefficient, 0.776) (Figure 2.5a). Additional error statistics (Table 2.6), most notably MAE, was 0.012 in the final calibration. The mean value for the modelled data was 0.04 $m^3s^{-1}$, deviating slightly from the mean of the measured data which was 0.03 $m^3s^{-1}$. The NSE was 0.613, which is considered to be a fair performance result under categories set by Moriasi et al. (2007) (Table 2.5). There was no additional discharge data for the Northern stream and therefore validation runs were not possible.

*Southern stream*

The Southern stream discharge was calibrated over the same time period as the Northern stream (2007-2010). The final calibration result with R=0.828 can categorized as good under assessment criteria set by Moriasi et al. (2007). Discharges were overestimated in April 2008 by 0.073 $m^3s^{-1}$ and underestimated between May and July 2008 (Figure 2.5b). Both of these results can be correlated with rainfall. The MAE and RMSE were acceptable for the simulation result (Table 2.6), however the NSE value was 0.334 which can be classed as a poor performance.

a)




b)

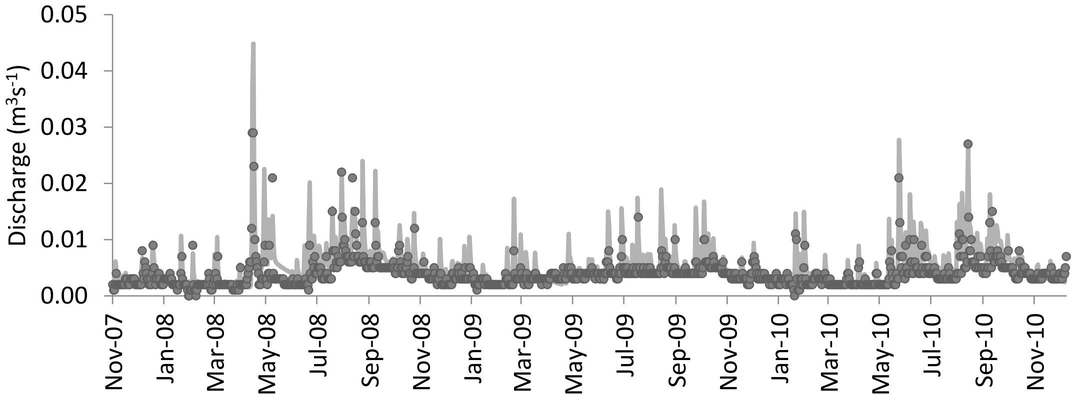



**Figure 2.5:** Calibrated model run results (black line) and measured data (grey dots) for the Northern (a) and Southern stream (b) discharges from November 2007 to November 2010.

**Table 2.6:** Calibration statistics for Northern and Southern stream discharges in the Okaro catchment.

| | **Calibration (2007-2011)** | |
|---|---|---|
| | **Northern stream** | **South stream** |
| MAE ($m^3s^-1$) | 0.012 | 0.001 |
| RMSE ($m^3s^-1$) | 0.017 | 0.002 |
| Pearson R | 0.776 | 0.828 |
| $R^2$ | 0.602 | 0.685 |
| Mean Observed ($m^3s^{-1}$) | 0.032 | 0.004 |
| Normalised mean abs error | 0.404 | 0.411 |
| Nash Sutcliffe efficiency | 0.613 | 0.334 |

### 2.3.2 *INCA-N and INCA-P calibration and validation*

INCA-N nutrients ($NO_3$ and $NH_4$) were calibrated from November 2007 to November 2010. Visual assessment of Northern and Southern stream $NO_3$ results (Figure 2.6a and Figure 2.7a) indicated acceptable fit to measured $NO_3$ levels, with the general exception of reduced concentrations (0.005 mg N $l^{-1}$) at the end of the 2007 and 2008 simulation periods. Observed increases in $NO_3$ were captured for the winter months and major reductions in the summer. Calibration statistics including R of 0.716 for the Northern stream and 0.677 for the Southern stream signify a satisfactory fit of the simulated data to the measured values (Table 2.7), but additional error statistics (NMAE and NRMSE) exhibited higher values indicating more probable error (Table 2.7). No combination of parameter adjustments allowed for a good match of simulation output to the extremely low concentrations of $NO_3$ during the summer. This mismatch of concentrations is responsible for increased MAE for $NO_3$. There was lower standard error in $NO_3$ for both streams. Mean concentrations for the Northern stream $NO_3$-N were higher for the model simulation (0.11 mg $l^{-1}$) compared to the observed data (0.08 mg $l^{-1}$). The Southern stream $NO_3$-N averages differed between the measured values (0.185 mg $l^{-1}$) and modelled outputs (0.198 mg $l^{-1}$). Variations between measured and modelled data were apparent, but the model simulation was able to reproduce the pattern seen in measured data. The final Southern stream R value was 0.716 which can be categorized as a satisfactory result (Moriasi et al. 2007).

Additional data provided by the BoPRC (2011-2014) was used to validate the 2007-2011 Northern stream model run. The simulation exhibited an acceptable fit to the newly input measured data and followed the majority of the patterns over the three-year time frame. The validation simulation reproduced low-level concentrations of $NO_3$-N (ca. 0.005 mg $l^{-1}$) during autumn for all three years. No data were available to validate nutrient simulations for the Southern stream and the comparison was limited to the calibration period.

The $NH_4$-N in the calibration (Figure 2.6b & 2.7b) deviated from the measured data which spanned from detection limits (0.005 mg $l^{-1}$) to very high concentrations (0.554 mg $l^{-1}$). The Pearson R value was -0.119 for the Northern stream and 0.072 for the Southern stream. Mean measured (0.04 mg $l^{-1}$) concentrations corresponding to the model calibration period (0.05 mg $l^{-1}$) were higher in the southern location. Validation runs for the Northern stream $NH_4$ displayed a similar pattern to the calibration. Calculated statistics exhibited an increased R value (0.253) for validation compared with the calibration result of -0.119, indicating a closer model fit for the validation period. Other error statistics also showed improvements in the validation period compared with the calibration (Table 2.7). As stated in the Methods, validation data were not available for the Southern stream.

*INCA-P (Northern and Southern streams*)

The simulation for SRP from the Northern stream nutrient run (Figure 2.6c) followed the majority of the patterns exhibited in the measured data including capturing large peaks which occurred during the 2007 summer. Statistics included a low R value (0.419) for the SRP variable (Table 2.7) with an increased MAE and NRMSE values. The Southern stream calibration for SRP exhibited similar results to the Northern stream, but differed by better resolving the large concentration peaks. Even though the model reproduction of the observed peaks appears better, the errors statistics were not representative of improvements for this stream. The R value for the Southern stream SRP run was 0.397 which is lower than the statistic calculated in the Northern stream run. Additional statistics were unsatisfactory according to Moriasi et al. (2007).

Validation results of 2011-2014 for the Northern stream SRP differed from the calibration results of 2007-2010. Figure 2.6c indicates model SRP outputs were able to replicate low-concentration periods presented in the measured data, but overestimated these during various time frames over 2010-2014. Data provided for validation was more sporadic than the data for the calibration period and the R value was low (0.201) compared to that for the $NO_3$ calibration (0.453). The NMAE and NRMSE statistics for SRP and TP are higher than the INCA-N calibrations, indicating a weaker model performance for P compared with N.

The calibration runs (2007-2010) for TP indicated an acceptable graphical fit to the measured data and accounted for the large peaks in summer 2007 and 2008. The R value for the Northern stream was poor (0.380) compared to that for SRP (0.491), but had decreased error (Table 2.7). The Southern stream R value (0.321) for TP was lower than that for the Northern stream. Outliers were responsible for the reduction of Pearson R value which is directly affected by scatter in the data. Other error statistics (Table 2.7) indicated a better fit of simulated TP to measurements for the Southern stream than the Northern stream.

The validation run for TP resulted in a small improvement in model fit based on statistical results, compared to the calibration phase. The model output (Figure 2.6d) displayed a similar pattern to the measured data, as observed in the calibration phase. The output captured the same overall pattern but with some overestimates during some phases of the model run. The differences between the two measured data sets are substantial showing a marked difference in minimum, average, and maximum values.

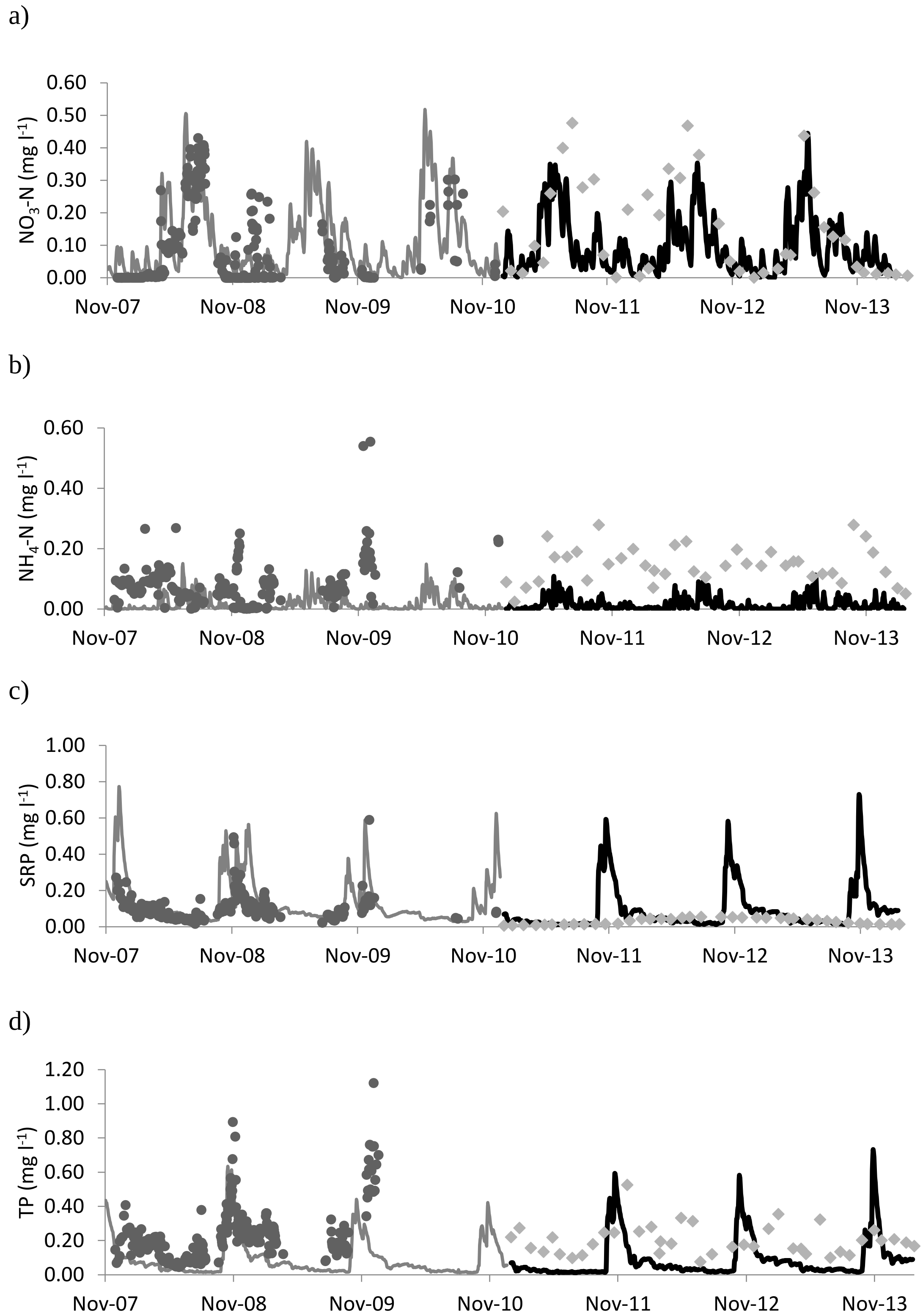



**Figure 2.6:** Model calibration (grey line) and validation runs (black line) and measured data from November 2007 to November 2009 (black points) and November 2010 to February 2014 (grey points) for Northern stream nutrients including a) nitrate ($NO_3$-N), b) ammonium ($NH_4$-N), c) soluble reactive phosphorus (SRP), and d) total phosphorus (TP).

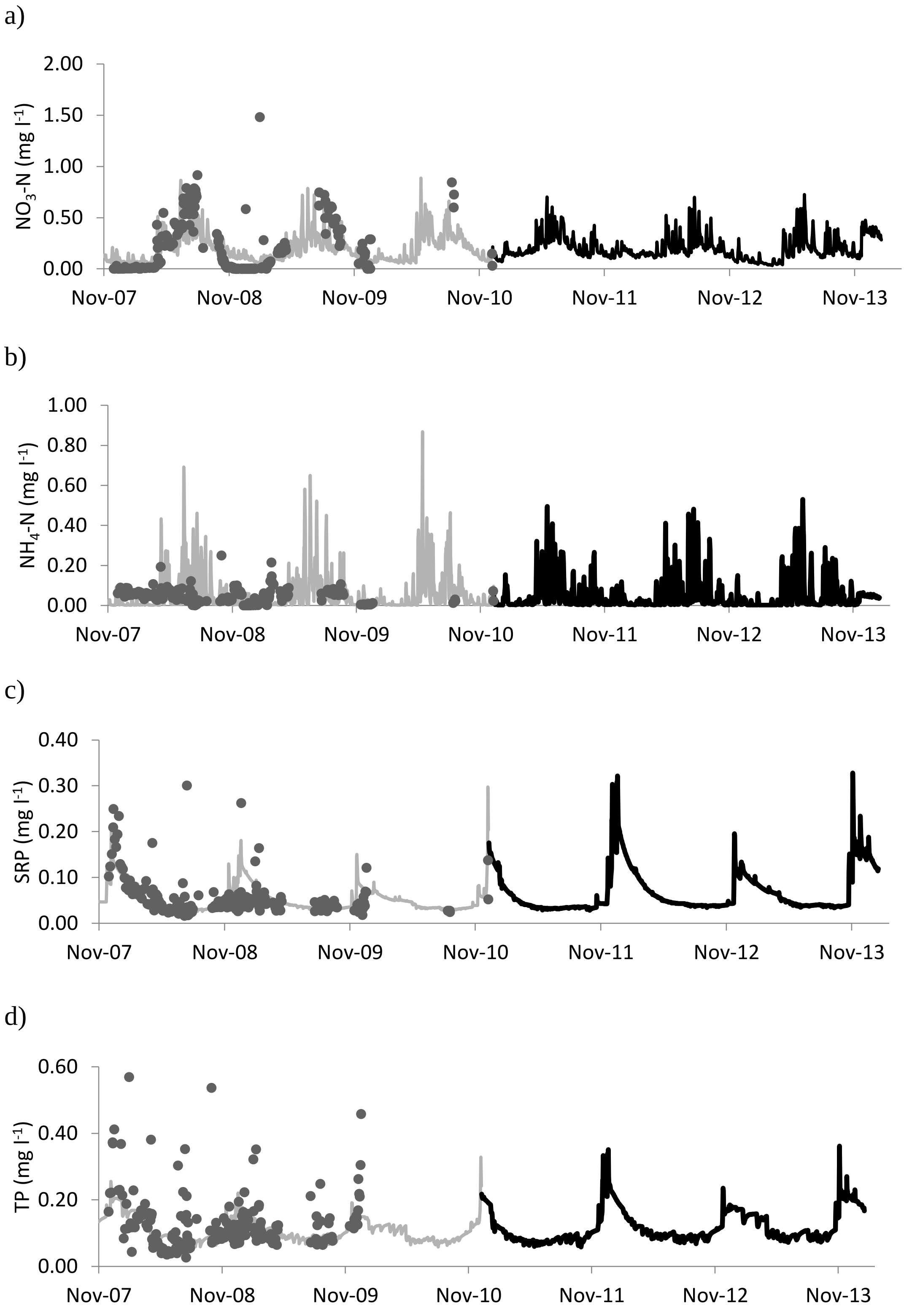



**Figure 2.7:** Calibrated model runs (grey line) and measured data (solid points) and additional validation (black line) for Southern stream nutrients including a) $NO_3$-N, b) $NH_4$-N, c) SRP and d) TP from November 12, 2007 to Feb 28, 2014.

**Table 2.7:** Statistical results for the calibration and validation runs for the Northern stream and calibration statistics for Southern stream. The R, MAE, RMSE units are the same as variable under assessment and remaining are unitless.

| | Northern stream | | | | |
|---|---|---|---|---|---|
| | **Calibration (2007-2011)** | | | | |
| | R | MAE | NMAE | RMSE | NRMSE |
| Nitrate ($NO_3$) | 0.716 | 0.073 | 0.843 | 0.096 | 1.099 |
| Ammonium ($NH_4$) | -0.119 | 0.059 | 0.909 | 0.091 | 1.402 |
| Soluble reactive phosphorus (SRP) | 0.419 | 0.070 | 0.672 | 0.134 | 1.283 |
| Total phosphorus (TP) | 0.380 | 0.143 | 0.596 | 0.187 | 0.781 |
| | **Validation (2011-2014)** | | | | |
| | R | MAE | NMAE | RMSE | NRMSE |
| Nitrate ($NO_3$) | 0.453 | 0.106 | 0.721 | 0.143 | 0.974 |
| Ammonium ($NH_4$) | 0.253 | 0.132 | 0.903 | 0.144 | 0.988 |
| Soluble reactive phosphorus (SRP) | 0.201 | 0.065 | 1.091 | 0.128 | 2.144 |
| Total phosphorus (TP) | 0.440 | 0.153 | 0.771 | 0.176 | 0.889 |
| | Southern stream | | | | |
| | **Calibration (2007-2011)** | | | | |
| | R | MAE | NMAE | RMSE | NRMSE |
| Nitrate ($NO_3$) | 0.677 | 0.144 | 0.773 | 0.195 | 1.051 |
| Ammonium ($NH_4$) | 0.072 | 0.058 | 1.348 | 0.092 | 2.133 |
| Soluble reactive phosphorus (SRP) | 0.397 | 0.029 | 0.552 | 0.045 | 0.844 |
| Total phosphorus (TP) | 0.321 | 0.064 | 0.451 | 0.167 | 1.173 |

*Scenario runs*

The combined calibration and validation runs are herewith referred to as the baseline scenario (BS). It is compared to scenarios pertaining to the removal of the wetland/riparian vegetation (SC1), reduction in fertilizer applications rates (SC2), and conversion of the entire catchment into dairy based land use (SC3) for each of the output variables.

*Wetland/riparian margin removal (SC1)*

The model output for a scenario pertaining to the wetland/riparian removal for the Northern stream (Figure 2.8) and Southern stream (Figure 2.9) exhibited increased nutrient concentrations for each of the four variables. Average annual load for the two inflows for the baseline (BS) scenario (wetland in place) over the ten-year period was 0.202 tonnes of $NO_3$-N $yr^{-1}$ and 0.046 tonnes $yr^{-1}$ of $NH_4$-N $yr^{-1}$. The SC1 scenario increased by an average of 0.103 tonne $NO_3$-N $yr^{-1}$ and 0.037 tonnes $NH_4$-N $yr^{-1}$. The P loads for both scenarios differed slightly (0.037 and 0.103 tonnes yr-1) (Table 2.8), but were not as significant as the changes seen for the N.

An analysis of model outputs and estimates from Hudson & Nagels (2011) indicated variations attributable to wetland effectiveness. INCA simulations showed >10% removal rate for TN over the three-year period (Table 2.8) compared with those presented in Hudson & Nagels (2011) who approximated 10-41% TN removal from 2008-2010 (Table 2.8). TP removal rates in the model simulations were small, ranging from 5-13% (Table 2.8) compared to those presented in Hudson & Nagels (2011) (Table 2.8).

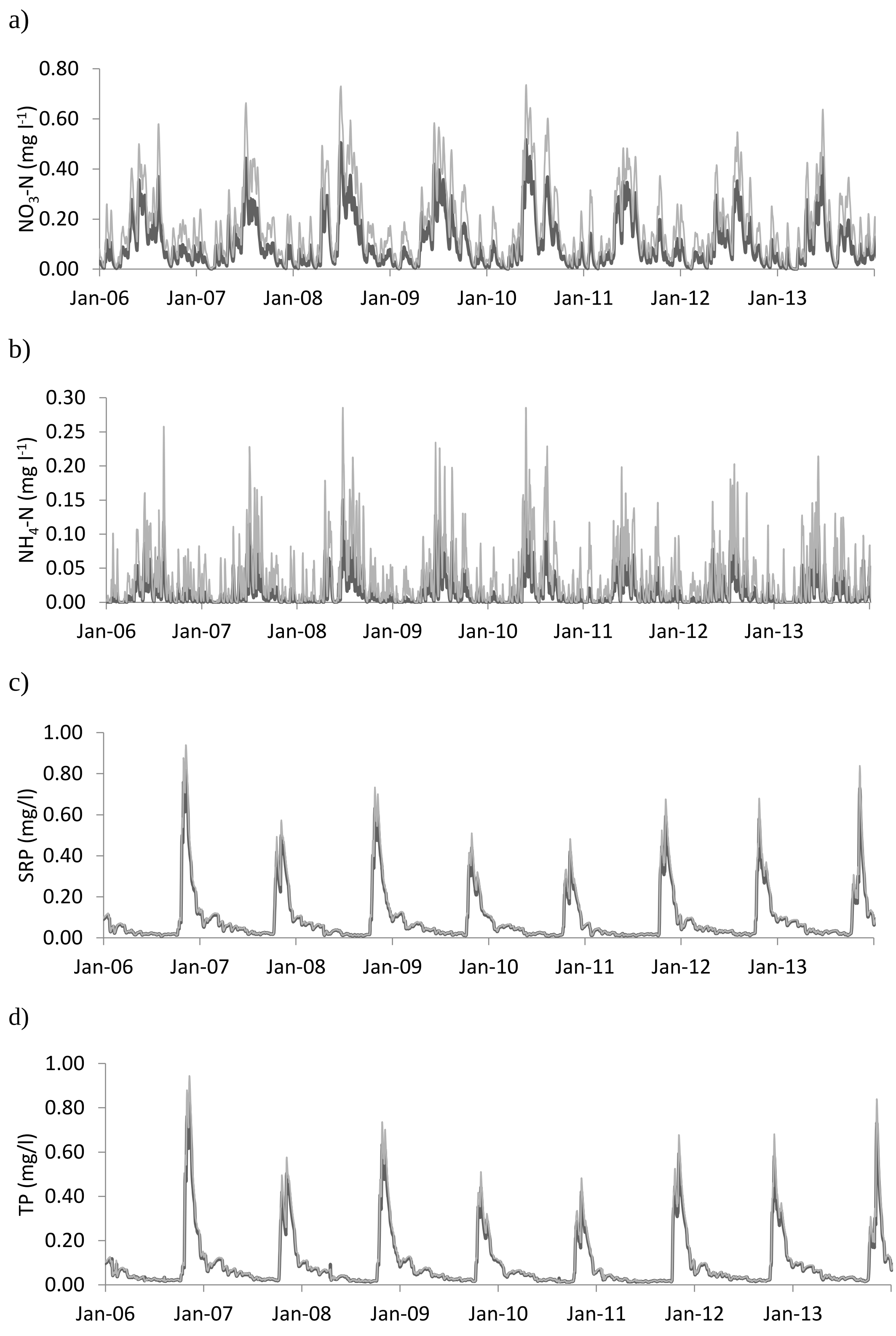



**Figure 2.8:** Concentrations a) $NO_3$-N, b) $NH_4$-N, c) SRP, and d) TP for the baseline (BS; dark grey line) and wetland removal (SC1; light grey line) scenario in the Northern stream.

a)

b)

c)

d)

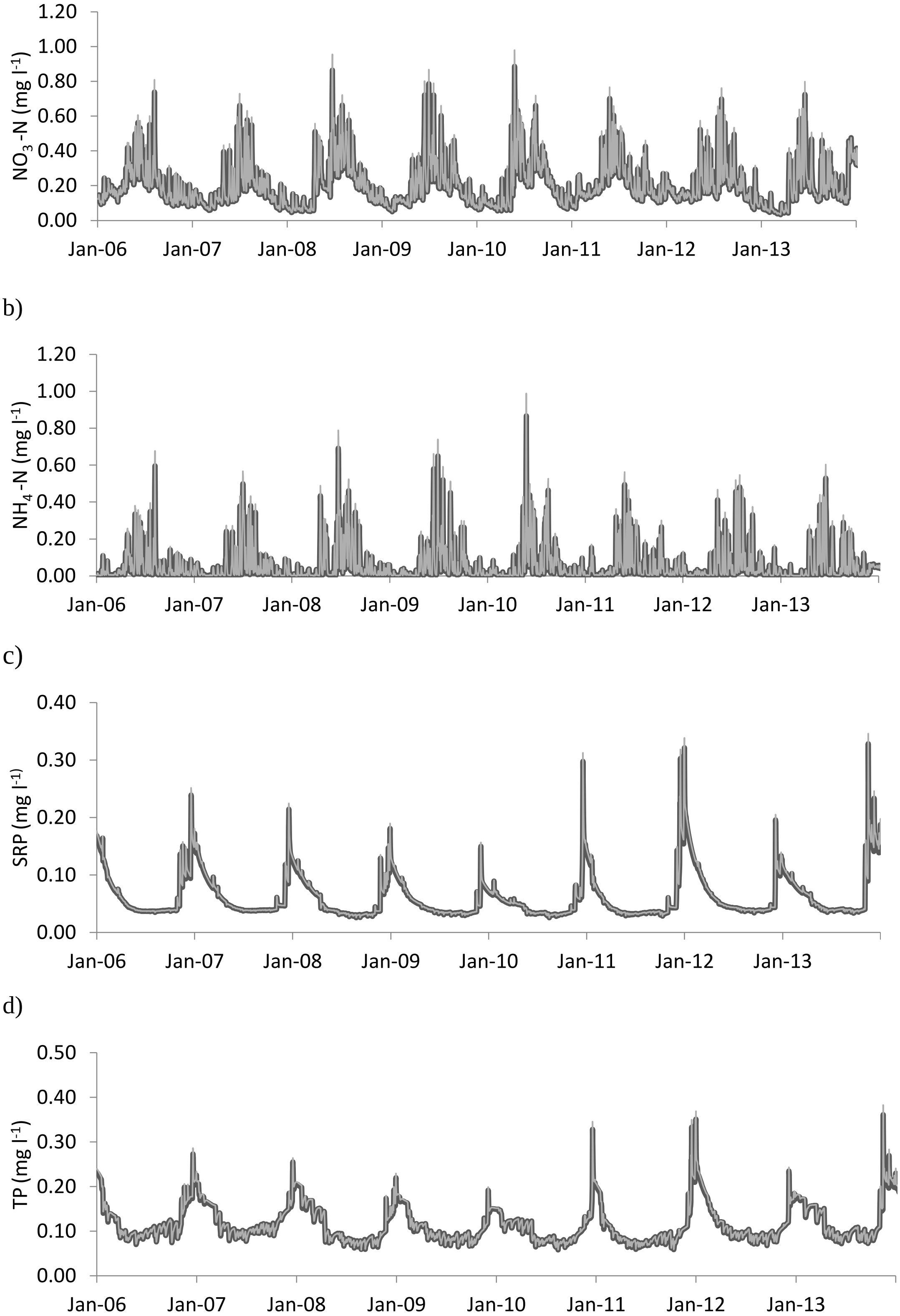



**Figure 2.9:** Concentrations of a) $NO_3$-N, b) $NH_4$-N, c) SRP, and d) TP for the baseline (BS; dark grey line) and wetland removal (SC1; light grey line) scenario in the Southern stream.

**Table 2.8:** Load and wetland retention (amount removed from stream) estimates from the INCA- N and INCA- P model runs and the Hudson & Nagels (2011) NIWA study.

| Date | TN load (tonne $yr^{-1}$) | Load retained TN (tonne $yr^{-1}$) | TP load (tonne $yr^{-1}$) | Load retained TP (tonne $yr^{-1}$) |
|---|---|---|---|---|
| 2008 (this study) | 0.42 | 0.095 | 0.16 | 0.02 |
| 2008 (NIWA) | 1.444 | 0.597 | 0.504 | 0.3 |
| 2009 (this study) | 0.314 | 0.064 | 0.126 | 0.01 |
| 2009 (NIWA) | 0.876 | 0.146 | 0.251 | 0.06 |
| 2010 (this study) | 0.426 | 0.088 | 0.11 | 0.005 |
| 2010 (NIWA) | 1.25 | 0.149 | 0.249 | 0.03 |
| | **TN retained by wetland (%)** | **TP retained by wetland (%)** | | |
| 2008 (this study) | 15 | 13 | | |
| 2008 (NIWA) | 41 | 60 | | |
| 2009 (this study) | 16 | 11 | | |
| 2009 (NIWA) | 17 | 23 | | |
| 2010 (this study) | 16 | 5 | | |
| 2010 (NIWA) | 12 | 12 | | |

*Fertilizer reduction practice (50%) (SC2)*

In the simulation with a reduction in the rate of fertilizer application of 50% on arable land (Table 2.3 and 2.4), all nutrient concentrations declined in both stream sites. Concentrations of $NO_3$-N and $NH_4$-N in the Northern stream (Figure 2.10a and b) peaked at 0.295 mg $l^{-1}$ and 0.040 mg $l^{-1}$ respectively. This is a considerable difference from the BS scenario which exhibited maximum concentrations of 0.550 mg $l^{-1}$ for $NO_3$-N and 0.190 mg $l^{-1}$ for $NH_4$-N. Average concentrations for the Southern stream showed declines of 0.033 mg $l^{-1}$ for $NO_3$-N and 0.023 mg $l^{-1}$for $NH_4$-N. Both streams displayed the same pattern as the original BS which is directly related to the fertilizer application timing, not the adjusted applications rates. Changes in the SRP and TP variables for both streams (Figure 2.10 and 2.11c, d) resulted in general reduction across the entire time frame. In the Northern stream the peak and baseline concentrations were both reduced substantially compared the Southern stream. The average concentrations of SRP in the Southern stream indicated overall declines from 0.06 mg $l^{-1}$ to 0.03 mg $l^{-1}$.

a)

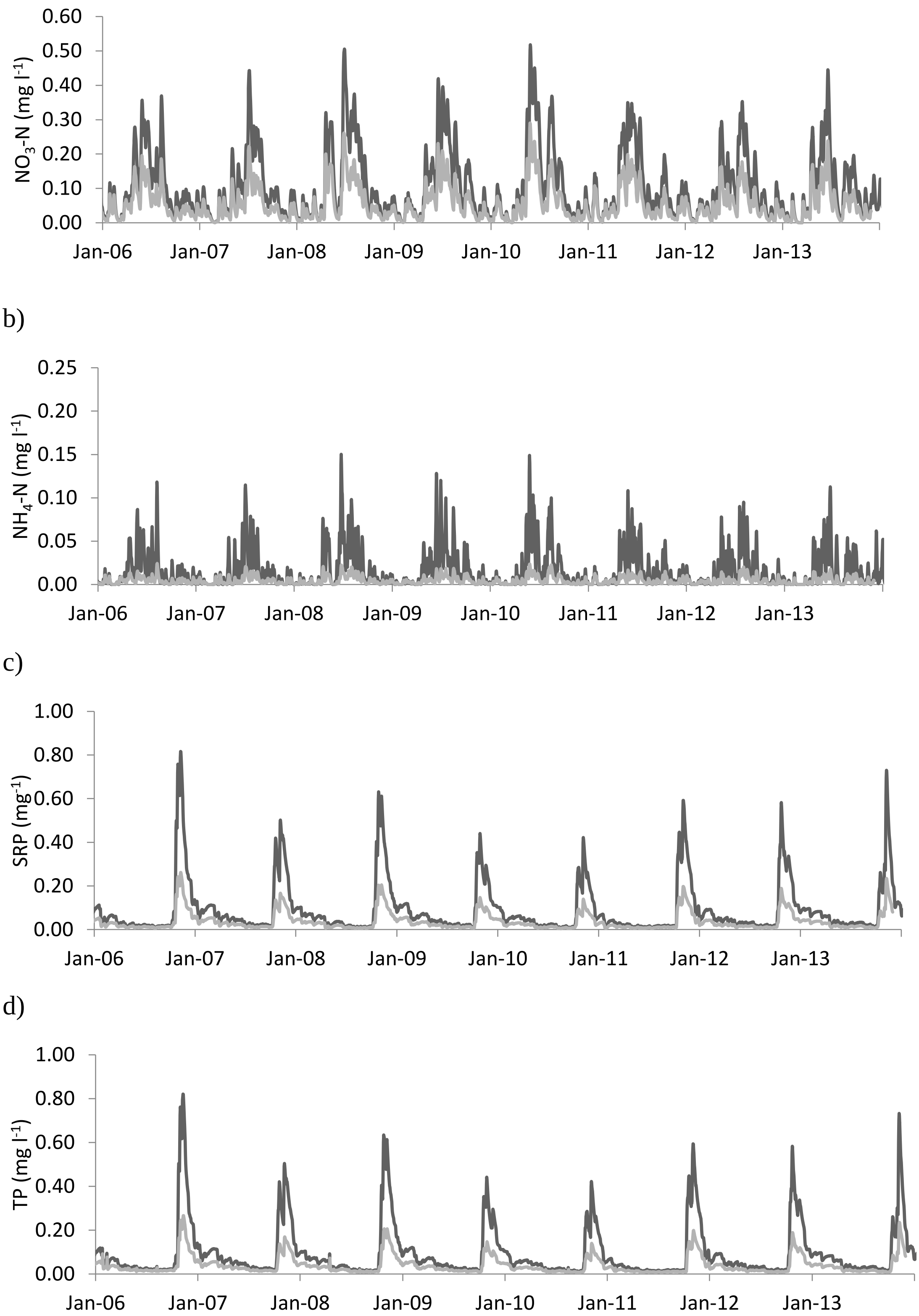



**Figure 2.10:** Concentrations of a) $NO_3$-N, b) $NH_4$-N, c) SRP, and d) TP variables during the baseline (BS; dark grey line) and fertilizer reduction (SC2; light grey line) in the Northern stream.

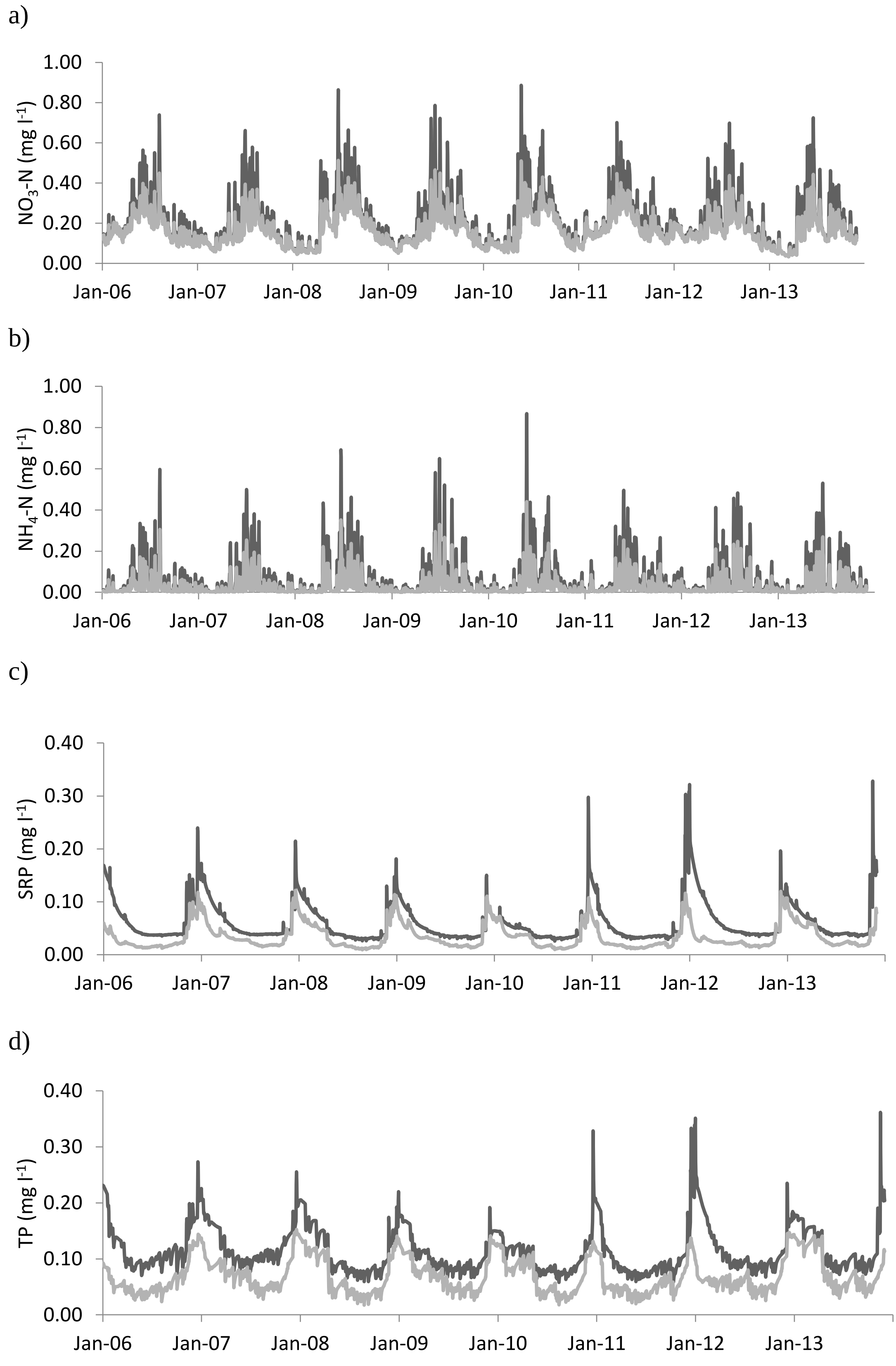



**Figure 2.11:** Concentrations of a) $NO_3$-N, b) $NH_4$-N, c) SRP, and d) TP variables during the baseline (BS; dark grey line) and fertilizer reduction (SC2; light grey line) in the Southern stream.

*Total dairy (SC3)*

The effect of conversion of the entire Lake Okaro catchment from 10% diary (Table 2.2) to 100% dairy was most prominent for the Northern stream $NO_3$. Concentrations of this variable increased from 0.105 mg $l^{-1}$ to 0.286 mg $l^{-1}$. Concentrations of $NH_4$-N also increased by an average of 0.020 mg $l^{-1}$ to 0.080 mg $l^{-1}$ over the simulation time frame (2006-2013) (Figure 2.12 and 2.13). Mean concentrations of SRP and TP in the Northern stream increased from 0.143 mg $l^{-1}$ to 0.632 mg $l^{-1}$ for SRP and 0.177 mg $l^{-1}$ to 0.681 mg $l^{-1}$ for TP over the simulated period (Figure 2.12). The stream site also played a factor in the magnitude of the increase in nutrient concentrations and Southern stream had smaller increases than the Northern stream.

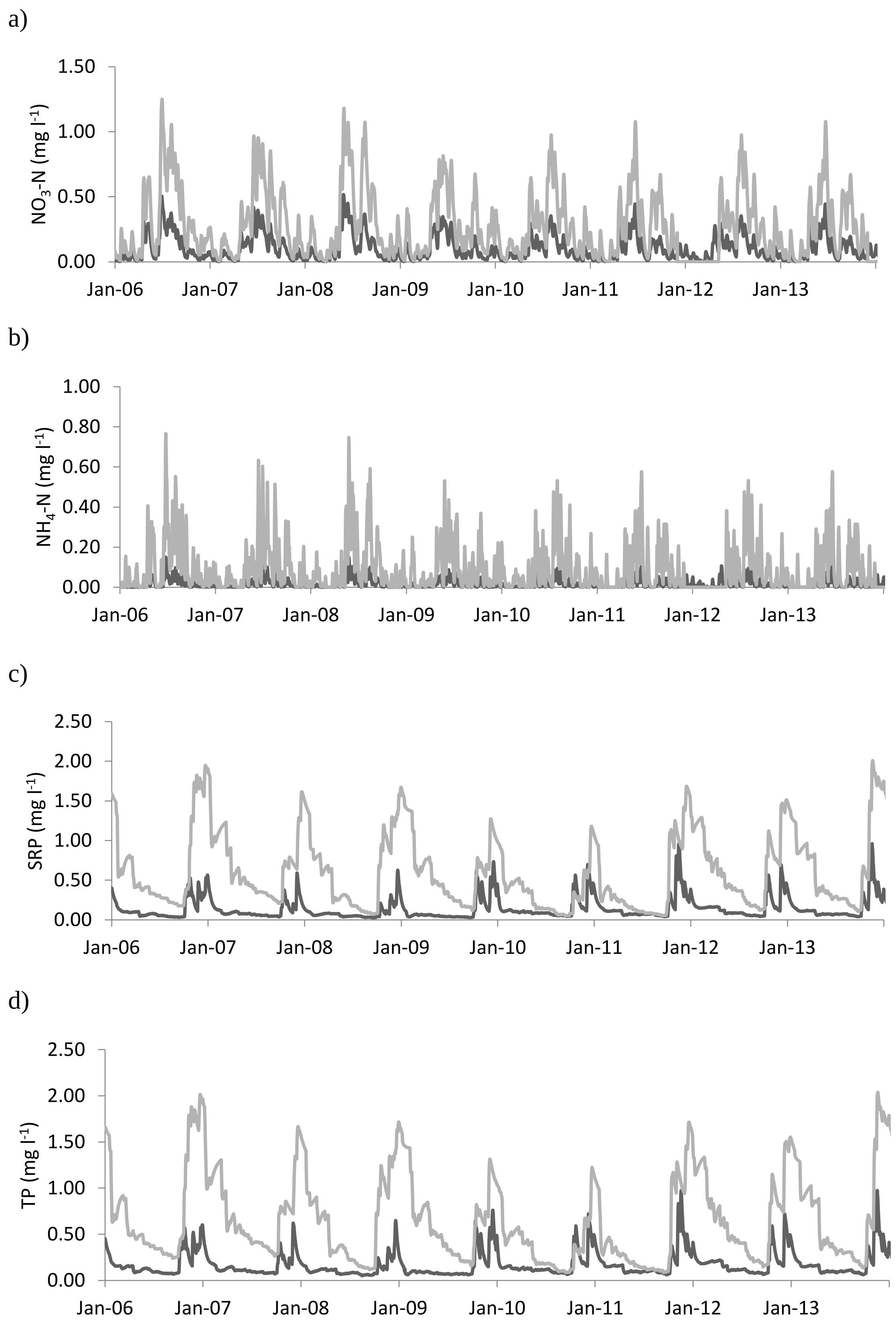



**Figure 2.12:** Concentration of a) $NO_3$-N, b) $NH_4$-N, c) SRP, and d) TP variables during the baseline (BS; light grey line) and total dairy conversion (SC3; dark grey line) in the Northern stream.

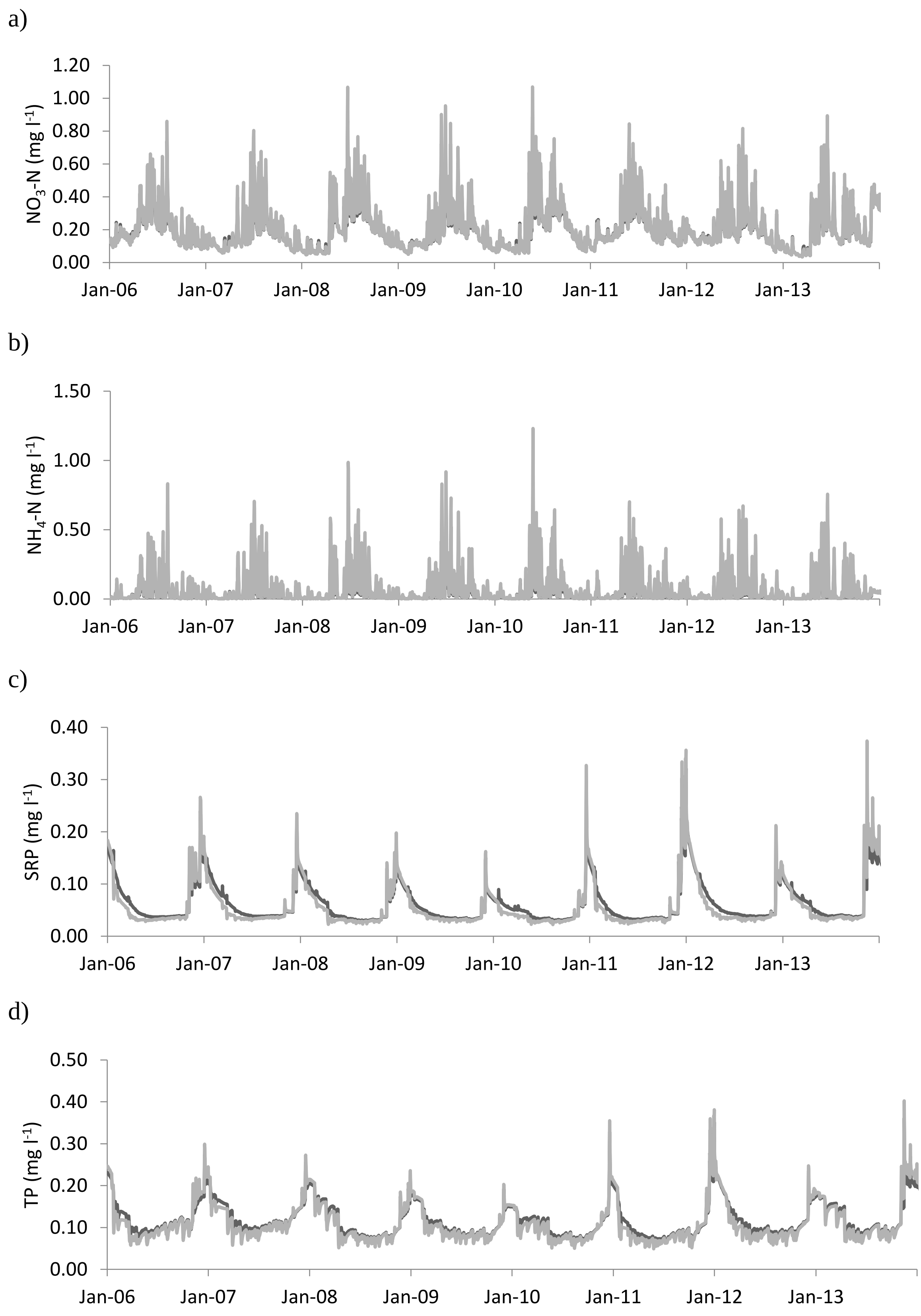



**Figure 2.13:** Concentrations of a) $NO_3$-N, b) $NH_4$-N, c) SRP, and d) TP variables during the baseline (BS; dark grey line) and total dairy conversion (SC3; light grey line) in the southern stream.

## 2.4 Discussion

The dynamic catchment model INCA was implemented to identify changes in nutrient concentrations in the Lake Okaro catchment for three different scenarios. These scenarios included the removal of a current artificial wetland and riparian planting, reduction of current fertilizer applications by 50%, and conversion of the entire catchment into a single land use type, dairy. Calibration and validation of these types of models are required to improve the ability to forecast environmental changes (Hamilton & Schladow 1997).

### 2.4.1 *Validity of results*

Discharge estimates from INCA reproduced the temporal variations in the measured data with the exception of overestimations in April 2008 and underestimations in May and July 2008. This result was quantified statistically using hydrological modelling performance criteria which indicated satisfactory model performances for the Northern stream ($R^2$=0.602 and 0.613 NSE) and good model performance for the Southern stream (0.685 $R^2$ and 0.334 NSE). Discharge estimations from 2007-2010 were within 0.01 $m^3s^{-1}$ of measured values. Published values of $R^2$ and NSE for INCA catchment model applications range from 0.54 to 0.90 and 0.58 to 0.78, respectively (Rankinen et al. 2002; Bernal et al. 2004; Granlund et al. 2004; Whitehead et al. 2011; Crossman et al. 2013a). The $R^2$ value of 0.602 and NSE value of 0.613 indicate that the Okaro INCA model application compares favourably with other published model results.

INCA-N model runs simulated $NO_3$ concentrations accurately for both the Northern and Southern streams, with the exception of underestimations during May and July 2008. The statistical performance was satisfactory for the Northern stream represented by an $R^2$ value of 0.512 and unsatisfactory for the Southern stream ($R^2$=0.458). Each stream presented unsatisfactory NSE results (<0.5), but values were still above zero, suggesting model outputs were more acceptable than using the mean of the observed data.

Statistical assessment of the validation runs produced values ($R^2$ and NSE value <0.500) indicating unsatisfactory model performance (Moriasi et al. 2007). The $NH_4$ validation was poor, with $R^2$ values close to 0 for both the Northern and

Southern stream. These results closely match the INCA-N assessment in a Mediterranean catchment (Fuirosos) by Bernal et al. (2004), who indicated a poor statistical result for its entire analysis time frame ($R^2$=0.100). In summary, the INCA-N model results for both the Northern and Southern streams exhibited unsatisfactory statistical results for $NH_4$ and satisfactory performances for $NO_3$ in the Northern stream.

INCA-P simulations of TP produced statistics that indicated average to poor performance ($R^2$<0.20, NSE<0.50). Other INCA-P assessments have indicated better results compared to this analysis, including the research performed on the River Thames by Crossman et al. (2013a) where they achieved an $R^2$ value of 0.90. INCA-P results for discharge and $NO_3$ for Lake Okaro can be considered acceptable based on performance statistics.

The statistical performance of INCA-N and INCA-P in this assessment ranged from poor to satisfactory over the period of interest (2007-2012) (Moriasi et al. 2007). Errors in the model fit can be attributed to a range of factors including the data utilized, the accuracy of the calibration, and the application of the model to a catchment considerably smaller than those where the model had been developed and applied before (Crossman et al. 2013a). High resolution data can be highly suitable for model calibration (Abell et al. 2011), but should also span for a longer time frame (2007-2010). The calibration and validation component of this research was applied over a period of only three years (2007-2010) and could be improved on if time constraints were not in place. The applicability of model to smaller catchments is acceptable, but absence of HER calculations and sub-catchment boundaries can make the assessment more difficult.

### 2.4.2 *Scenario runs*

The model was useful for evaluating the efficacy of catchment-based remediation techniques implemented in the Okaro catchment Three alternative regimes of management were simulated: the assessment of the 2.3 ha artificial wetland and riparian margin inputs, external nutrient management, and a hypothetical extreme land use conversion (100% total dairy).

*Wetland/riparian margin removal (SC1)*

A constructed wetland and riparian plantings were implemented in the Okaro catchment to reduce external loading of N and P to the lake (Tanner et al. 2007). Predicted outcomes from the Lake Okaro Action Plan (Environment Bay of Plenty 2006) suggested a 2.3 ha wetland could theoretically remove 0.193 tonne $yr^{-1}$ of N and 0.016 tonne $yr^{-1}$ of P. Estimates from INCA indicated wetland TN ($NO_3$-N and $NH_4$-N, no particulates) removal rates greater than 10% (0.095 tonne $yr^{-1}$) each year for the 2008-2010 period. This is considerably less than the 40 % (0.597 tonne $yr^{-1}$) TN removal rate calculated for 2008 by Hudson & Nagels (2011). The variation between these estimates can be partly attributed to the fact that INCA-N only calculates two primary N species ($NO_3$ and $NH_4$) and excludes other organic and particulate-bound N components. This indicates that the TN calculated in this assessment does not account for particulate materials and is therefore not in the proper order to estimates given in Hudson & Nagels (2011). An analysis of wetland efficacy completed by Hudson & Nagels (2011) indicated TN was composed of 60 to 64% organic and particulate bound materials. The efficacy of the wetland exhibited in the INCA simulations as well as the 2011 assessment by Hudson & Nagels (2011) estimated increased attenuation of $NO_3$ and $NH_4$ indicating the wetland was effective at removing nitrogen species. However, TP removal rates are likely to be lower than those (60%) reported as wetlands have a limited capacity to absorb P and eventually reach a point where they begin to act as P source.

Simulated INCA-P TP loads and removal rates were also notably smaller for the 2008-2010 assessment period compared to the estimate of 0.504 tonne $yr^{-1}$ by Hudson & Nagels (2011). Hudson & Nagels (2011) also stated that 60% of TP would be removed by the wetland. In INCA-P simulations only 13% was removed by the wetland. The TP removal estimates by Hudson & Nagels (2011) exceed the targets specified in the Lake Okaro Action Plan, which specified a modest target removal rate of 16 kg $yr^{-1}$ (Environment Bay of Plenty 2006). This removal is relatively small (0.04%) in the context of TP loads in the streams (Environment Bay of Plenty 2006). The efficacy of P removal in wetlands is typically limited, occurring primarily though plant uptake and sediment settling (Reddy et al. 1999). Accumulation of P in wetlands can result in the formation of pools which can become a source of P as the wetland ages (Richardson & Craft 1993). Hudson &

Nagels (2011) used Load Estimator (LOADEST) to indicate a substantial reduction in TP removal effectiveness after 2008. Fertilizer applications rates were not known in the 2008-2010 period, so average applications rates were used from 2014 to represent this period. These estimates may differ substantially from the models applied in the Hudson & Nagels (2011) assessment.

The TN and TP removal estimates for the 2008-2010 periods were predicted more accurately in the NIWA assessment, but fertiliser application rates and the modelling program utilized may have impacted nutrient outputs. Fertiliser application rates are important as they can reflect the total load entering a catchment in addition to influencing nutrient removal estimates. The INCA model used in this assessment does not account for important N components which directly influence the overall removal rates.

*Fertilizer reduction scenario (SC2)*

Lake Okaro is one of five catchment sites currently under a regional government-formulated action plan (Environment Bay of Plenty 2006; Abell et al. 2011). The Lake Okaro Action Plan specifies the utilization of various land management practices including the restriction of fertilizer applications, livestock grazing, and complete exclusion of livestock from flowing surface water (Özkundakci et al. 2010). The SC2 scenario was based on a 50% reduction of current fertilizer application rates in the Okaro catchment, specifically focusing on nutrient concentration changes in a realistic management scenario. The aim of this scenario was to identify its overall applicability for management purposes which may be required if lake TLI remains high. The results for the fertilizer reduction scenario indicated large-scale decreases (0.15-0.25 mg $l^{-1}$) from the BS scenario for both N and P species in the Northern and Southern streams. These reductions were more prominent in the Northern stream exhibiting average reductions of 0.250 mg $l^{-1}$ for $NO_3$-N and 0.150 mg $l^{-1}$ for $NH_4$-N. The $NH_4$ reduction was a reflection of current dairy fertiliser applications by the land owners in the catchment which are currently based on $NH_4$ applications. Reductions in both SRP and TP stream concentration can be correlated with reductions of P based fertilizer utilized in deer and sheep/beef farms in the catchment (McDowell & Wilcock 2008).

Reductions of nutrient inputs into catchments have also been assessed in a SWAT model application by De Girolamo & Lo Porto (2012). In their study, 20% fertilizer reductions produced moderate changes in TN (5-11%) and TP (12-19%) for different land uses, driven by vegetative type and soil conditions. For Lake Okaro the change in nutrient loads for 50% fertiliser reductions was a decrease in TN (46-50%) and TP (43-50%).

*Total dairy conversion (SC3)*

The total dairy scenario was applied to assess nutrient fluctuations based around the expansion of dairy land use in the Okaro catchment. Agricultural land uses in the Okaro catchment consist primarily of dry stock (65% sheep/beef) which is known to be less nutrient intensive than dairy (PCE 2013). Dairy is highly important for New Zealand's economy, and has undergone considerable expansion and intensification in the past decade (Abell et al. 2011). The hypothetical dairy conversion in this analysis resulted in a 36% increase for $NO_3$-N (0.16 mg $l^{-1}$) and $NH_4$-N (0.06 mg $l^{-1}$) as well as a 22-26 % increase for SRP (0.48 mg l-1) and TP (0.50 mg $l^{-1}$) in the Northern stream. These increases relate to limited infiltration capacities based around high scale erosion associated with land conversion (PCE 2013). This conversion indicated pronounced concentrations of each nutrient indicating the potential for large scale impacts on stream and lake water quality. This assessment is hypothetical, but is relatively realistic in a New Zealand context given continued high export prices for milk products.

# Chapter 3
# Lake modelling

## 3.1 Introduction

### 3.1.1 *Overview of lake degradation*

Eutrophication of lakes by diffuse and point source nutrients such as phosphorus (P) and nitrogen (N) has become a major environmental issue both in New Zealand and globally (Smith 2003; Schindler 2006; Abell et al. 2011). Increased nutrient levels stimulate primary production resulting in prolific and harmful algal blooms (Abell et al. 2010), formation of anoxic water in the hypolimnion, and reductions in biodiversity, all of which impact ecosystem functionality (Søndergaard et al. 2007). Nutrient enrichment of aquatic environments has resulted predominately from anthropogenically induced changes in surrounding catchments. These catchment modifications have principally been directed towards the expansion and intensification of agricultural production, which generates substantial losses of nutrients and sediments (Trolle et al. 2008). Mitigation of nutrient losses has become an increasing priority for governing bodies (Land and Water Forum 2010). Internationally, research has focused on P loss from agricultural production and has directed management projects towards limitation of this nutrient (Schindler 1977; Wang & Wang 2009). However, limitation of P can result in enhanced nitrogen (N) fixation by some phytoplankton species (Schindler et al. 2008). Co-limitation of N and P has been shown to be commonplace within aquatic ecosystems and positive outcomes have also been achieved through reduction of both N and P to freshwater ecosystems (Elser et al. 2007).

The Te Arawa (Rotorua) lakes in the North Island of New Zealand have been affected by increases in anthropogenic loads of N and P. Lake water quality has declined significantly over the last fifty years (Hamilton 2003; Abell et al. 2011). The Bay of Plenty Regional Council (BoPRC), who oversee management of these lakes, have adopted various management tools aimed at reducing external and internal loading of N and P into lake ecosystems. This includes applications of sediment capping materials to reduce nutrient release from sediments (Paul et al.

2008), optimising nutrient utilization by agricultural producers, and best practice procedures by drainage and storm water managers (Özkundakci et al. 2011). Each of these mitigation measures is aimed at restoring lake water quality to a set Trophic Level Index value (TLI). The TLI is a modified reference criteria derived from the Trophic State Index (TSI; Carlson 1977) which rates water quality based on key indices including total phosphorus (TP), total nitrogen (TN), chlorophyll *a*, and Secchi disk depth (Burns et al. 1999). The TLI is then utilized in the formulation of lake guidelines and lake action plans. Each action plan indicates a specific TLI limit relating to achieving water quality approximating that of the 1960's. If a TLI exceeds a specified threshold then a plan will be implemented to restore water quality to an assigned target.

A current action plan is in place at Lake Okaro, the smallest and most degraded of the Rotorua lakes. Its catchment land use is 95% agriculture. Various mitigation methods have been implemented to restrict external nutrients including an artificial wetland (Tanner et al. 2007), planting of riparian margins (Özkundakci et al. 2010), and improved land use practices (Birchall & Paterson 2011). In-lake remediation including applications of aluminum sulphate (alum) and modified zeolite (Aqual-P®) have also been implemented to restrict P releases from bottom sediments. These nutrient releases are increased as a result of anoxia during the stratified period which has been prominent since the early 1950's (Forsyth et al. 1988). Alum is a chemical flocculant utilized for P removal (Zamparas & Zacharias 2014). When lake pH is in the effective performance range (pH 6-8), it will precipitate particulates in addition to capping further releases from the sediment. Aqual-P is a modified zeolite derived from natural occurring alumina-silicates. These materials have been implemented based around their large surface areas which provide highly absorbent metal cations which aid in P removal (Zamparas & Zacharias 2014).

The efficacy of the applications of alum and Aqual-P for Lake Okaro remediation is unknown. An assessment of the restoration efforts through the utilization of technological tools such as hydrological and ecological models could provide valuable information on the relative effectiveness of different methods that have been used to attempt to control internal and external loads, as well as the scale of control necessary to meet the target TLI.

### 3.1.2 *Numerical modelling of lake ecosystems*

Many numerical models of lakes incorporate major ecological and hydrodynamic processes and are valuable in accounting for the interactions between physical, chemical, and biological processes. These models can be utilized for a range of different analyses including the effects of climate change on lake trophic levels (Trolle et al. 2011), assessment of remediation effectiveness in lake sites, as well as analyzing spatial and temporal variability pertaining to different rating values (Gal et al. 2009). There is a range of lake models currently in use including DYRESM-CAEDYM, ECOPATH with ECOSIM, CE-QUAL-W2, and Delft 3D-ECO (Mooij et al. 2010). Each of these models was designed around different foci, which has influenced their overall complexity and structure. DYRESM-CAEDYM is a process-based one-dimensional hydrodynamic model coupled with an ecological model aimed at simulating nutrient processes in lakes and reservoirs (Hipsey 2011). Various ecological processes are represented in the model through the use of mass conservative equations (Mooij et al. 2010). This model is typically applied to water bodies undergoing stratification (e.g. Burger et al. 2008). DELFT 3D-ECO differs from DYRESM-CAEDYM as it based around two and three-dimensional (2D-3D) representations aimed at simulating hydrodynamic and nutrient characteristics in lakes, reservoirs, and estuaries. This model utilizes grid based representations aimed at higher spatial resolution. The model is able to simulate various nutrient cycles in addition to phytoplankton characteristics (Los 2009). With the development of ELCOM-CAEDYM, 3D simulations can now be applied to assess spatial fluctuations which follow closely to DELFT 3D-ECO.

There is a range of effective lake modelling applications globally including the application of a multi-element model to Lake Washington in the United States (Arhonditsis & Brett 2005). This assessment examined management scenarios relating to plankton alterations including zooplankton and phytoplankton. Additional findings included more accurate parameterization pertaining to plankton kinetics and hydrodynamic constituents which provide useful details for external modelling research**.** Another example is the application of DYRESM-CAEDYM to three morphologically different lakes in New Zealand (Trolle et al. 2011). The primary objective of this research was to indicate fluctuations in trophic state in response to climate change. The results indicated enhanced eutrophication derived from increases in TN, TP and chlorophyll *a* with increasing air temperature. Each of these potential scenarios provided a useful forecasting tool for management purposes.

The focus of this chapter is to utilize a coupled hydrodynamic-ecological model (DYRESM-CAEDYM) to simulate in-lake water quality for various scenarios pertaining to remediation applications implemented in eutrophic Lake Okaro, Bay of Plenty Region, New Zealand. The remediation options tested included an assessment of an artificial wetland and planting of riparian margins, and in-lake dosing with alum and Aqual-P. Catchment-based remediation applications were assessed through the time-resolved catchment model INCA, which provided the necessary daily outputs to run water quality simulations with the lake model (see Chapter 2). Water quality changes were assessed mostly through the utilization of the TLI for each remediation scenario.

## 3.2 Methods

### 3.2.1 *Model description*

Lake Okaro water quality was analyzed using an existing coupled hydrodynamic-ecological model (DYRESM-CAEDYM) for the lake, which had previously been applied by Özkundakci et al. (2011). Both DYRESM and CAEDYM were developed at the Centre for Water Research at the University of Western Australia (Hipsey & Hamilton 2008). DYRESM (version 3.1.0-03) is a one-dimensional hydrodynamic model which calculates vertical variations in temperature, density, and salinity by setting up horizontal sections of a lake or reservoir (Hamilton & Schladow 1997). The horizontal layers of the model undergo changes in size due to variations in heat, momentum, and mass (Burger et al. 2008). The model can be run over varying time frames ranging from hours to decades (Hipsey & Hamilton 2008). CAEDYM (version 3.1.0-06) is the ecological component of the DYRESM-CAEDYM model. It uses process representations of biogeochemical cycles which influence nutrient fluxes, productivity, and oxygen dynamics within the lake (Hipsey 2011). Biological, physical, and chemical aspects of the lake can be assessed, including water quality variables such N, P, carbon (C), suspended sediment (SS), dissolved oxygen (DO) and various phytoplankton species including cyanophytes (Hipsey & Hamilton 2008). The model also allows for different configurations of variables according to user needs and precise calibration can help users to achieve their end goals of such actions as catchment and lake remediation (Hipsey 2011).

### 3.2.2 *DYRESM-CAEDYM model input data*

The Lake Okaro DYRESM-CAEDYM model previously calibrated and validated by Özkundakci et al. (2011) was run from July 2004 to June 2012 using daily input data with daily outputs generated using a one-hour internal time step. The meteorological component of the model required a data set which spanned the eight year time frame (extra year included to avoid model spin up). The meteorological data were obtained from climate station 1770 run by the New Zealand Meteorological Service located at Rotorua Airport (38.1092° S, 176.3172° E) 20 km north of Lake Okaro. The variables collected included rainfall (m), cloud cover (%), air temperature (°C) and wind speed (m $s^{-1}$). Vapour pressure (hPa) was calculated from air temperature and relative humidity (TVA 1972) and solar radiation was collected from virtual climate (VCN) station data and products.

Interpolation processes were previously applied to derive daily inflow volumes, DO, temperature, and nutrient concentrations including nitrate ($NO_3$), ammonium ($NH_4$), phosphate ($PO_4$), and N and P particulates (Özkundakci et al. 2011).

In this study, model runs were implemented daily from July 2004 to June 2012 with an internal time step of 1 hour. This required an extension of the previous meteorological data (Özkundakci et al. 2011) to 2012 (Figure 3.1 and 3.2). Vapour pressure was calculated from 2004-2012 using an equation based on wet and dry bulb temperature (Eq. 9) (TVA 1972). Inflow volumes and nutrient concentrations were sourced from the INCA model. This model was used to provide daily discharges and nutrient concentrations for $NO_3$, $NH_4$, soluble reactive phosphorus (SRP), and TP. Additional variables such as temperature (Eq. 10) and DO (Eq. 11) were calculated for the entire simulation period for the two stream sites from Mohseni et al. (1998) and Benson & Krause (1980), respectively:

$$e_a = e'_{as} - 0.00066\,(1 + 0.00115\, q_w).\, P.\, (q_D - q_W) \quad \text{(Eq. 9)}$$

where;

ea= partial water vapor pressure of the air (hPa),

$e'_{as}$= saturation vapour pressure at wet bulb temperature,

$q_w$= wet bulb air temperature ($^oC$),

$q_D$= dry bulb air temperature ($^oC$),

P= atmospheric pressure (hPa).

$$T_S = \frac{\alpha}{1+e^{y(\beta - T_a)}} \quad \text{(Eq. 10)}$$

where;

$T_s$= estimated stream temperature ($^oC$),

$T_a$= measured air temperature ($^oC$),

α= coefficient for the estimated maximum stream temperature,

γ= measure of the steepest slope of the function,

β= air temperature at the inflection point.

$$DO=\exp (7.71-1.31 \mathit{ln}(T + 45.93) \quad \text{(Eq. 11)}$$

where;

DO= dissolved oxygen at saturation (mg $l^{-1}$),

T= water temperature ($^oC$).

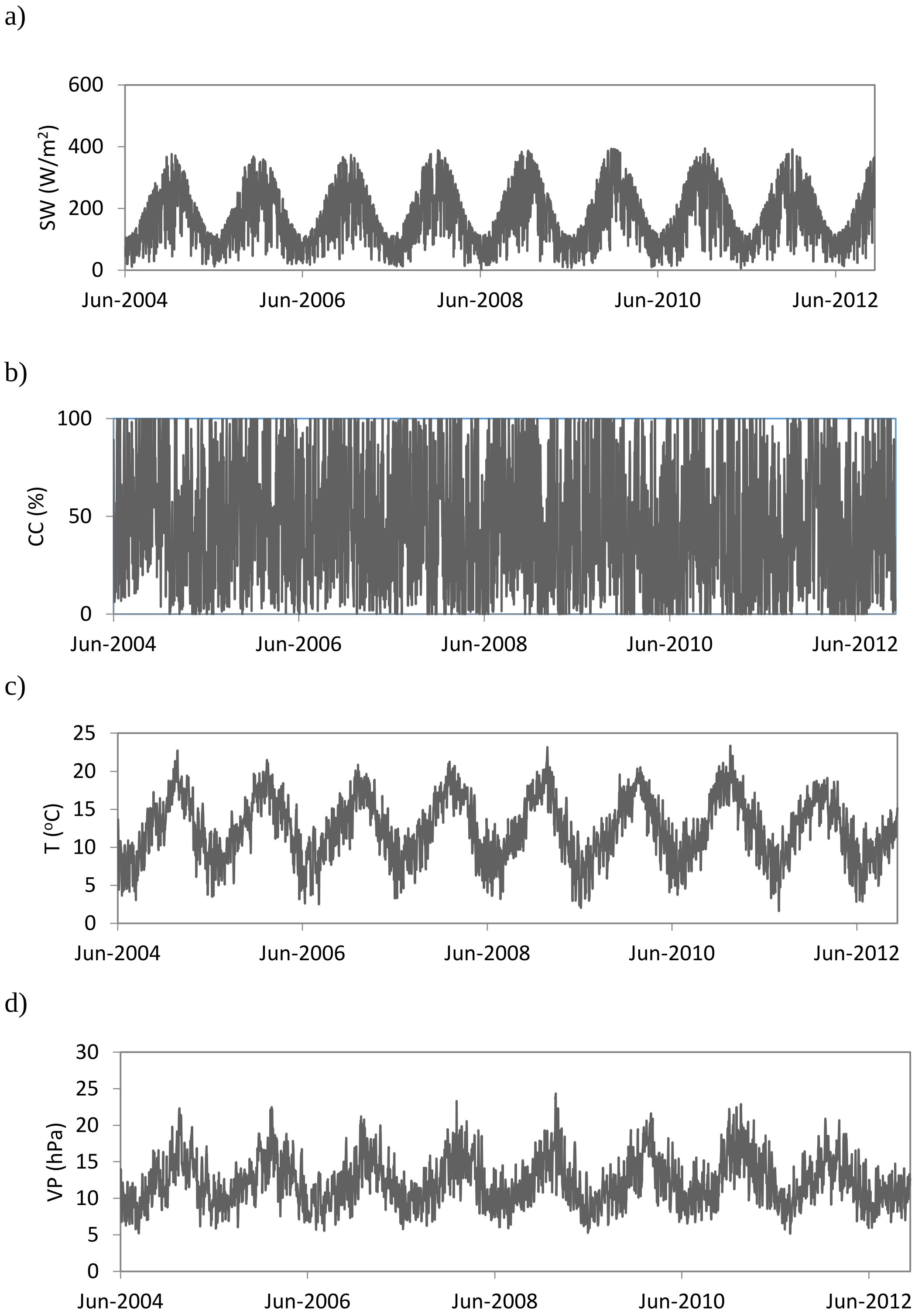



**Figure 3.1:** Meteorological data utilized as input to DYRESM (January 2004-July 2012). a) short wave radiation (SW; W/m$^2$), b) cloud cover (CC; %), c) air temperature (T; $^o$C) and d) vapour pressure (VP; hPa).

a)

b)

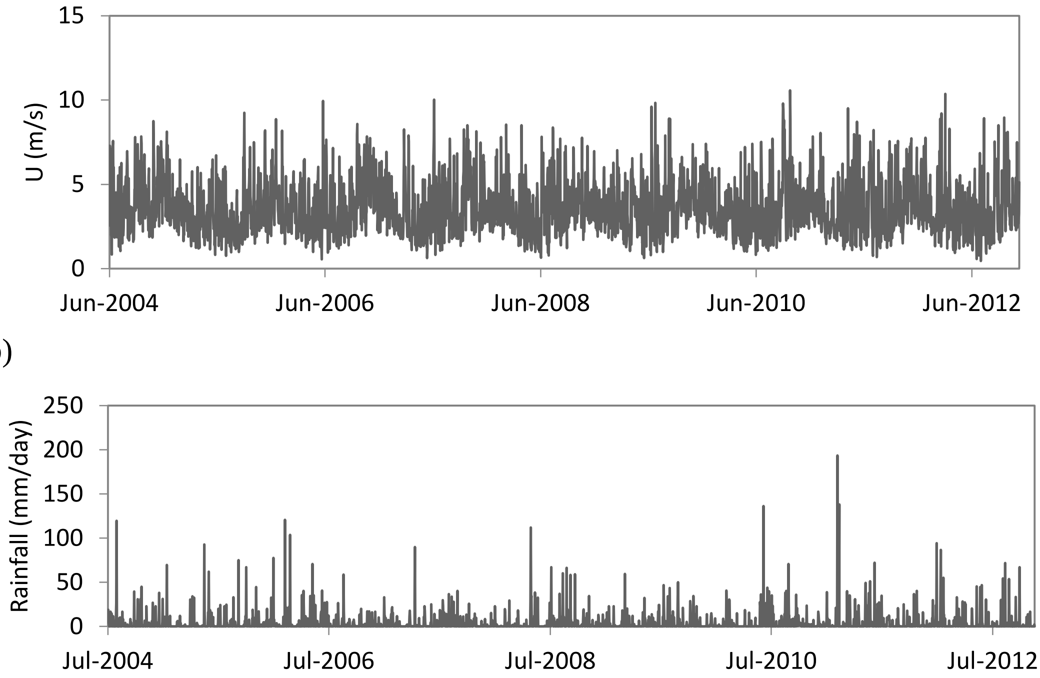



**Figure 3.2:** Meteorological data utilized as input to DYRESM (January 2004-July 2012). a) wind speed (U; m $s^{-1}$) and b) rainfall (mm/day).

*Water balance*

The two major stream inflows were added to the model in addition to a residual inflow based on surrounding land use, and precipitation. Inflow data for the two stream sites was obtained through a temporal based catchment model (INCA; Chapter 2). Residual inflows were derived from proportioning catchment model runs to simulate areas around the lake that did not contribute to the two streams. Precipitation was assigned as an inflow based on meteorological data spanning over the allotted time frame of 2004-2012. This was added to the inflow file in order to account for atmospheric deposition of nutrients (Pers. Comm C. McBride 2014). Precipitation data from the meteorological file were removed to avoid doubling of rainfall into the lake. The water balance was extended from 2004 to 2012 using terms from Wetzel & Likens (2000). Outflows were derived from residual values calculated from the latent heat flux and evaporation (Wunderlich 1972; Fischer et al. 1979; Özkundakci et al. 2011). Water levels in the model simulation were compared with measured data provided by the Bay of Plenty Regional Council.

### 3.2.3 *Calibration and validation*

Model simulation output was compared with field measurements collected from 2005-2007 for calibration purposes. Data provided by the BoPRC included temperature, DO, $NO_3$, $NH_4$, TN, $P0_4$, and TP from the surface (0m) and 14m depth. Two different phytoplankton groups (with parameters approximating cyanophytes and chlorophytes*)* were simulated in the model. Cyanophyte half- saturation constants for nitrogen uptake were set to low values to reflect lesser nitrogen limitation than other groups (and possible N-fixation). All parameters were retrieved from the original 2011 assessment (Özkundakci et al. 2011) with exception of minor adjustments to sediment oxygen demand (SOD). Error statistics were calculated for each output variable including the Pearson coefficient (R) and the root mean square error (RMSE).

Model validations were applied following the calibration run from 2005 to 2007. Validation runs were applied from 2007 to 2008 to assess the accuracy of the model calibration to measured data without additional calibration of parameters.

### 3.2.4 *Scenarios*

Scenarios were applied to the calibrated model to assess the effectiveness of both external (catchment) and internal (in-lake) remediation applications. The three external scenarios included the removal of the 2.3 ha of artificial wetland and riparian vegetation (SC1), 50% reduction of fertilizer applications for pastoral land use classifications (SC2), and conversion of the entire lake catchment land use into dairy (SC3). Each of these scenarios was first run through INCA (see Chapter 2) which then fed input data into DYRESM-CAEDYM by directly modifying the inflow file. The in-lake scenarios include an assessment of the effects of the chemical flocculent (alum) and sediment capping material (Aqual-P) applications (Table 3.1) and their overall effect on water quality. Comparison of the model baseline run (i.e. wetland included) to measured data may indicate the influence of alum and Aqual-P on water quality parameters. An additional simulation was applied to sediment release parameters in CAEDYM. This included a sensitivity analysis of $PO_4$ release rates and SOD as part of a process to more accurately simulate nutrient concentrations in bottom waters. Both parameters were based on estimates provided in Hamilton et al. (2014).

**Table 3.1:** Timing, material, and application rates for dosing to Lake Okaro.

| **Date** | **Material** | **Application method** | **Dose (tonnes)** |
|---|---|---|---|
| Dec-2003 | alum | Spraying from a moving boat as aluminium sulfate solution. | 4.59t |
| August-2007 | Aqual P | Applied using a fertilizer spreader on a barge. | 110t |
| September-2009 | Aqual P | Applied as a fine powder (<1mm) injected at 3m below surface as a slurry. | 44t |
| Dec-2011 | Aqual P | A slurry was applied by helicopter. | 5t |
| July-2012 | alum | Spraying from a moving boat. Lake water was mixed from top to bottom during application. | 8t |
| August-2012 | alum | Spraying from a moving boat. Lake water was mixed from top to bottom during application. | 14t |

### 3.2.5 *Trophic level Index (TLI) assessment*

The four-variable TLI (Burns et al. 1999) was used to assess differences in water quality for each scenario compared with the baseline. The TN, TP, and chlorophyll *a* (Chl*a*) levels were retrieved from each DYRESM-CAEDYM run and the Secchi disk depth (SD) was derived using a specified formula (Eq. 12) (Özkundakci et al. 2011).

$$z_{SD}=\frac{\alpha}{K_d} \qquad \text{(Eq. 12)}$$

where;

zSD= Secchi depth (m),

$K_d$=diffuse attenuation coefficient ($m^{-1}$),

α= constant determined by comparing field measurements of Secchi depth with value of $K_d$ (Lake Okaro, α=1.54)

Values were averaged for each year and converted from mg $l^{-1}$ to mg $m^{-3}$ for each variable with exception of Secchi disk depth. Average values for each index was utilized in their respective TLI equation (Eq. 13 to 17) and compared with lake classification guidelines provided in Burns et al. (1999) (Table 3.2):

$$TL_{Chla} = 2.22+2.54\log(Chla) \qquad \text{(Eq. 13)}$$

$$TL_{SD} = 5.1+2.27\log\left(\frac{1}{SD} - \frac{1}{40}\right) \qquad \text{(Eq. 14)}$$

$$TL_{TP} = 0.218+2.92\log(TP) \qquad \text{(Eq. 15)}$$

$$TL_{TN} = -3.61+3.01\log(TN) \qquad \text{(Eq. 16)}$$

$$TLI=\frac{1}{4}\sum(TL_{Chla},TL_{SD},TL_{TP},TL_{TN}) \qquad \text{(Eq.17)}$$

**Table 3.2:** Lake classifications based on the TLI ratings for Chlorophyll a (Chl*a*), Secchi disk depth, total phosphorus (TP), and total nitrogen (TN) (Burns et al. 1999).

| **Lake type** | **Trophic Level** | **Chla ($mg \cdot m^{-3}$)** | **Secchi Depth (m)** | **TP ($mg \cdot m^{-3}$)** | **TN ($mg \cdot m^{-3}$)** |
|---|---|---|---|---|---|
| Ultra-microtrophic | 0.0 to 1.0 | 0.13-0.33 | 33-25 | 0.84-1.8 | 16-34 |
| Microtrophic | 1.0 to 2.0 | 0.33-0.82 | 25-15 | 1.8-4.1 | 34-73 |
| Oligotrophic | 2.0 to 3.0 | 0.82-2.0 | 15-7.0 | 4.1-9.0 | 73-157 |
| Mesotrophic | 3.0 to 4.0 | 2.0-5.0 | 7.0-2.8 | 9.0-20 | 157-337 |
| Eutrophic | 4.0 to 5.0 | 5.0-12 | 2.8-1.1 | 20-43 | 337-725 |
| Supertrophic | 5.0 to 6.0 | 12-31.0 | 1.1-0.4 | 43-96 | 725-1558 |
| Hypertrophic | 6.0 to 7.0 | >31.0 | <0.4 | >96 | >1558 |

## 3.3 Results

### 3.3.1 *Water level*

There were minor differences between the simulated water level and measured water level over the seven-year period of simulation (2005-2012). The model simulation captured the overall fluctuations exhibited in the measured data, resulting in an R value of 0.76.

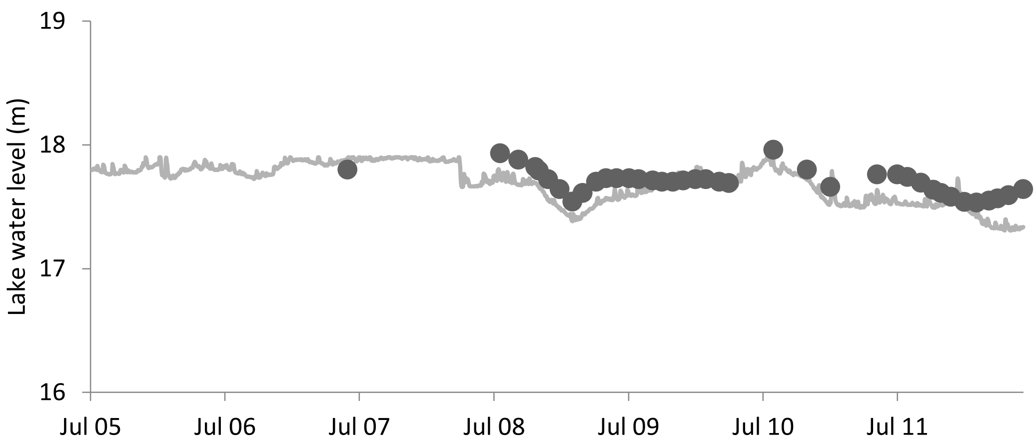



**Figure 3.3:** Comparison of DYRESM-CAEDYM water level outputs (grey line) and measured data (solid circles) for 2005-2012.

### 3.3.2 *Model performance*

The calibration (2005-2006) period of the DYRESM-CAEDYM simulation generally reproduced accurately the temporal variations in the measured data in both the surface (0 m) and hypolimnion (14 m) for temperature, DO, and dissolved nutrients (Figure 3.4-Figure 3.7). The temperature result for the surface calibration was acceptable indicated by a Pearson R value of 1 but a comparatively low hypolimnetic value of 0.64. The RMSE for both the surface (0.46) and hypolimnion (0.91) was low. Variations in temperature were reflective of seasonal changes, resulting in stratification in the hypolimnetic waters during summer months (Figure 3.4a). This stratification was responsible for reduced DO concentrations in the hypolimnion (Figure 3.4b). There was a reasonable match between measured and modelled data for hypolimnetic DO (R 0.88 and RMSE of 2.22) (Table 3.3). The formation of anoxic water resulted in substantial releases of $PO_4$ (Figure 3.5a) and $NH_4$ (Figure 3.7a) from benthic sediments. Simulated $PO_4$ in the hypolimnion produced a satisfactory statistical result (R 0.91) compared to $PO_4$ measured in the surface layer (R=0.83) (Table 3.3). Concentrations of $NH_4$ in the surface water was simulated with high R (0.92) and low RMSE (0.12), which were statistically improved compared to the hypolimnetic results (Table 3.3). The timing of the $NH_4$ release in the hypolimnion was poorly represented reflected by the poor statistical results (i.e. large error). TP in the surface layer and hypolimnion (Figure 3.5b) were simulated with reasonable accuracy, indicated by a moderate R value of 0.87 and 0.83 and low RMSE of 0.02 and 0.09, respectively. The two groups of phytoplankton simulated exhibited a dominance of cyanophytes (82% of total phytoplankton population) compared to chlorophytes (collectively 18%).

The validation component of the assessment (July 2007- July 2008) was able to simulate the majority of the seasonal variations observed in the measured data and were mostly well represented in the model. The statistical results differed from the calibration with discrepancies occurring for DO, $PO_4$, $NO_3$, TP, and TN in the surface and the hypolimnion (Table 3.3). This was most obvious for $PO_4$ with a relatively poor R statistic (0.42) in the surface layer but a high R value in hypolimnion (0.82) (Table 3.3). The surface $NO_3$ (R 0.89) was substantially better than the original calibration (R 0.70). Most hypolimnetic R-values were increased compared to the surface layer (Table 3.3), due to strong seasonal variations.

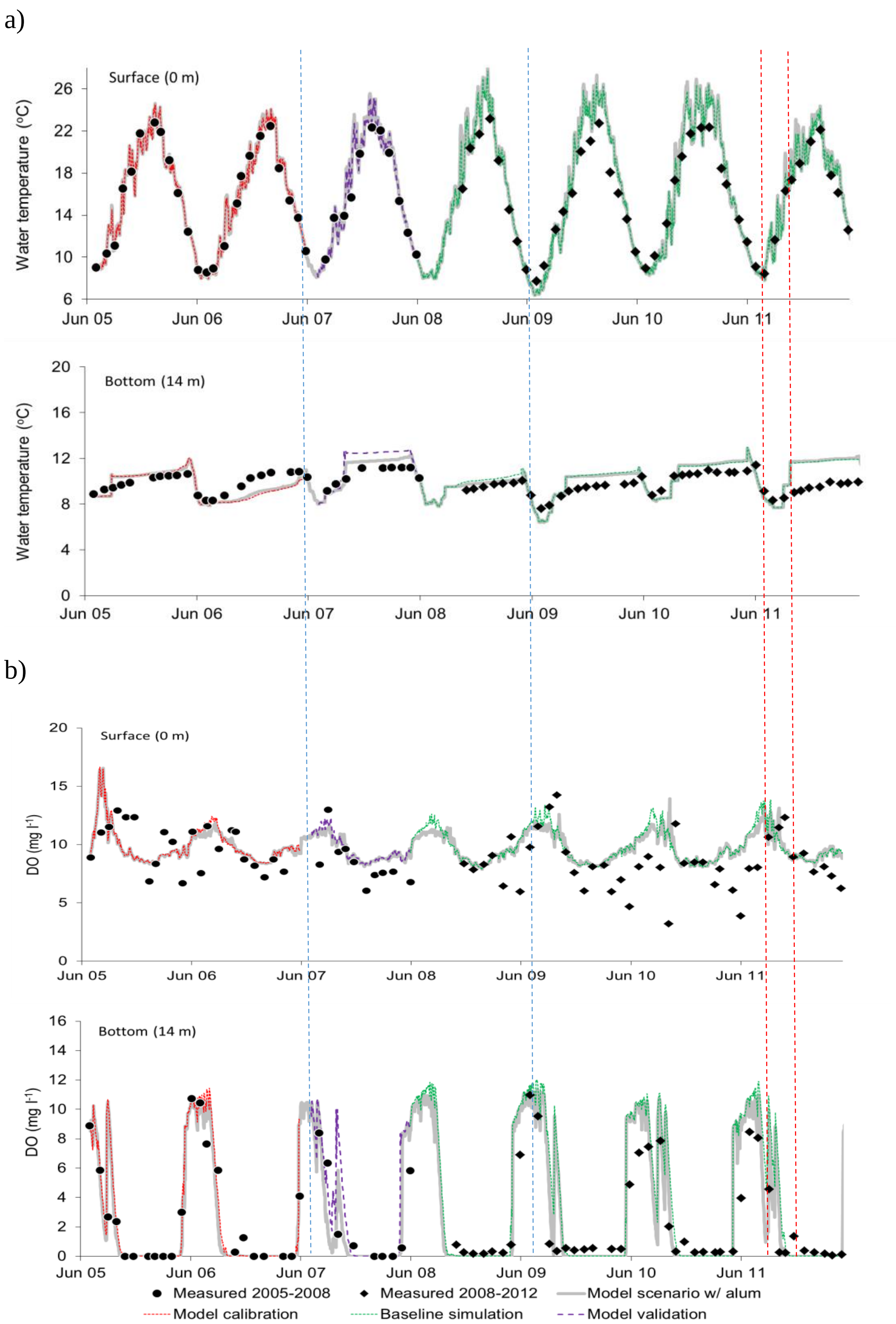
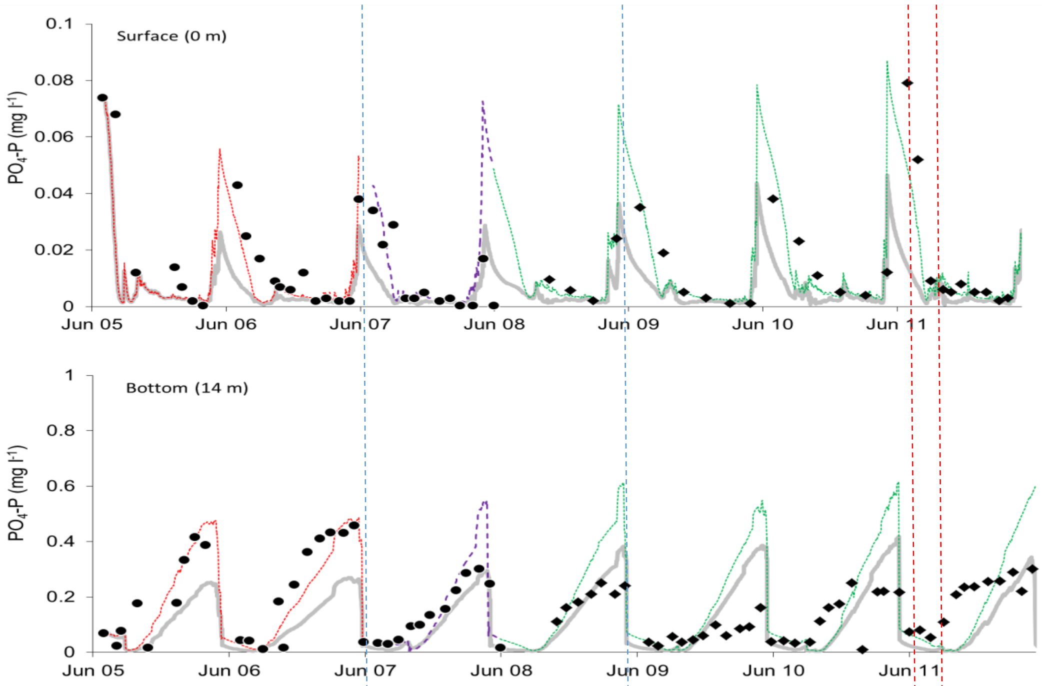



**Figure 3.4:** DYRESM-CAEDYM simulations for a) temperature and b) dissolved oxygen (DO) in the surface (0m) and hypolimnion (14m) from 2005-2012. Blue vertical lines indicate Aqual-P dosing and red vertical lines indicate alum dosing.

a)




b)

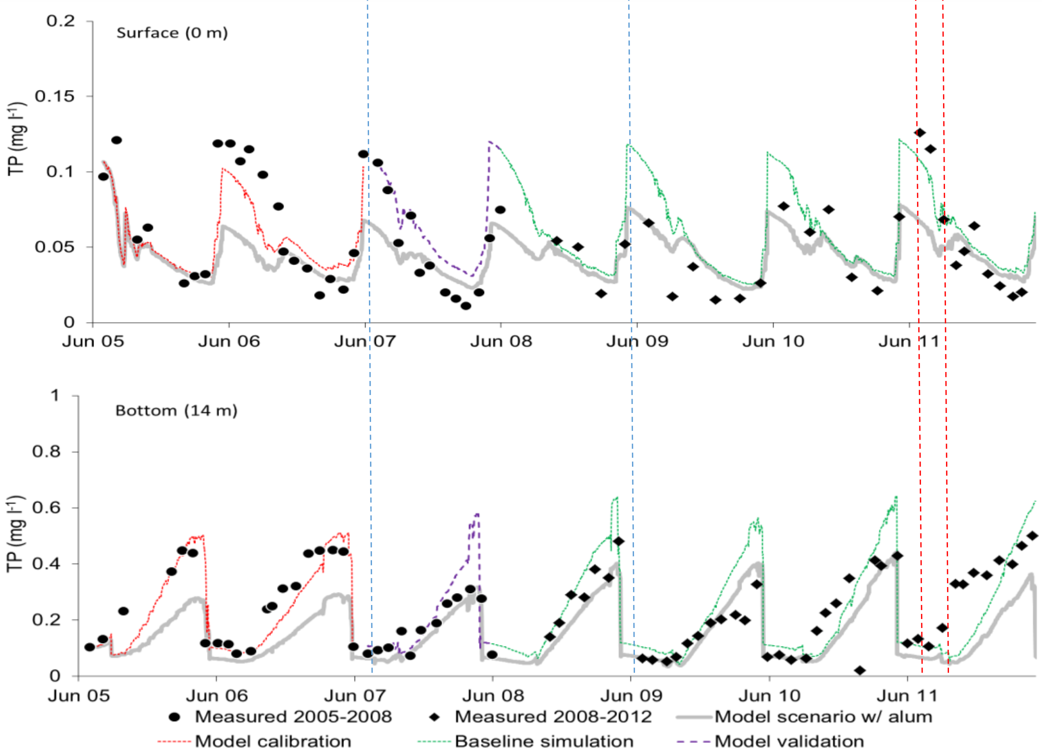



**Figure 3.5:** DYRESM-CAEDYM simulations for a) phosphate ($PO_4$-P) and b) total phosphorus (TP) in the surface (0m) and hypolimnion (14m) from 2005-2012. Blue vertical lines indicate Aqual-P dosing and red vertical lines indicate alum dosing.

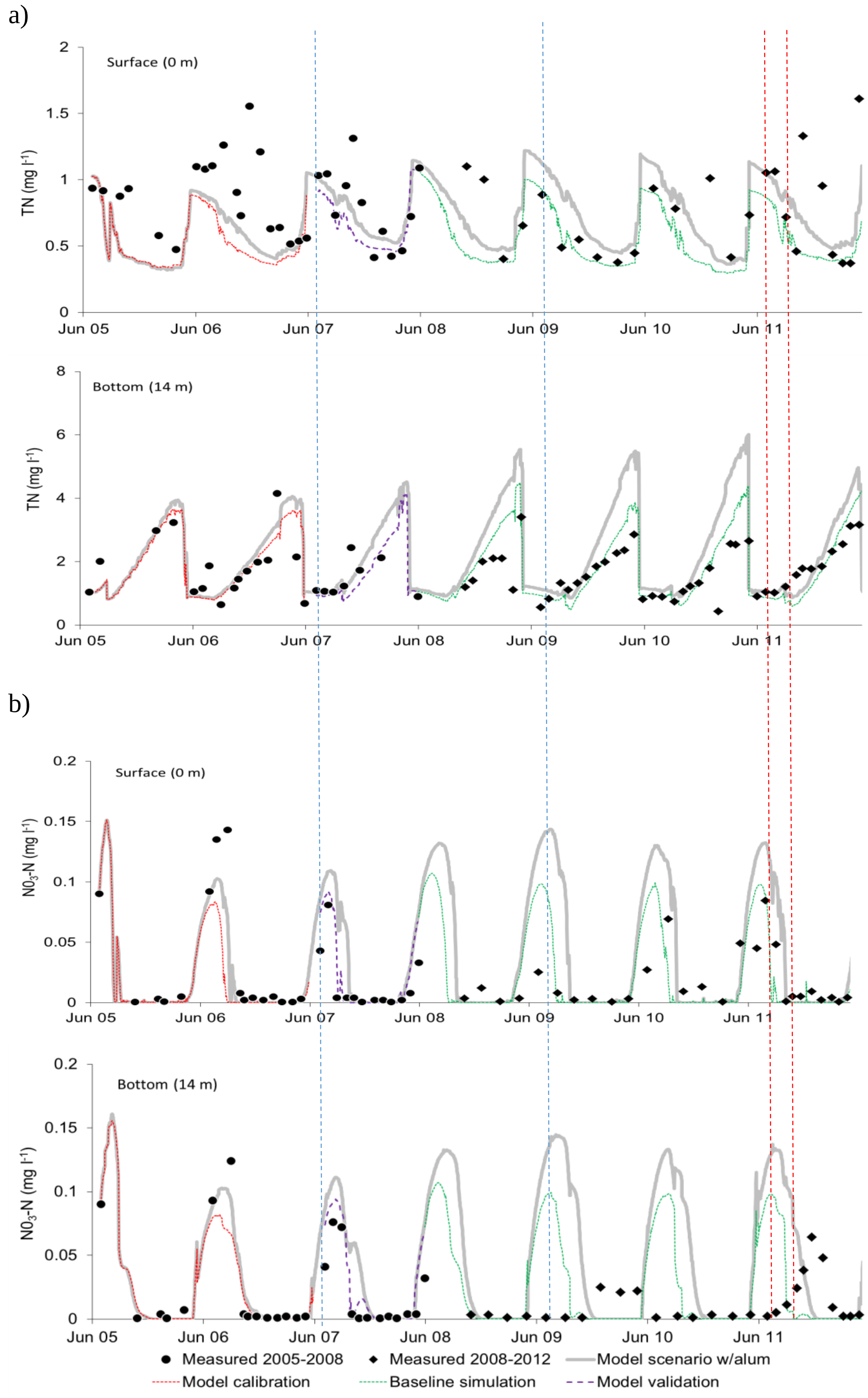



**Figure 3.6:** DYRESM-CAEDYM simulations for a) total nitrogen (TN) and b) nitrate ($NO_3$-N) in the surface (0m) and hypolimnion (14m) from 2005-2012. Blue vertical lines indicate Aqual-P dosing and red vertical lines indicate alum dosing.

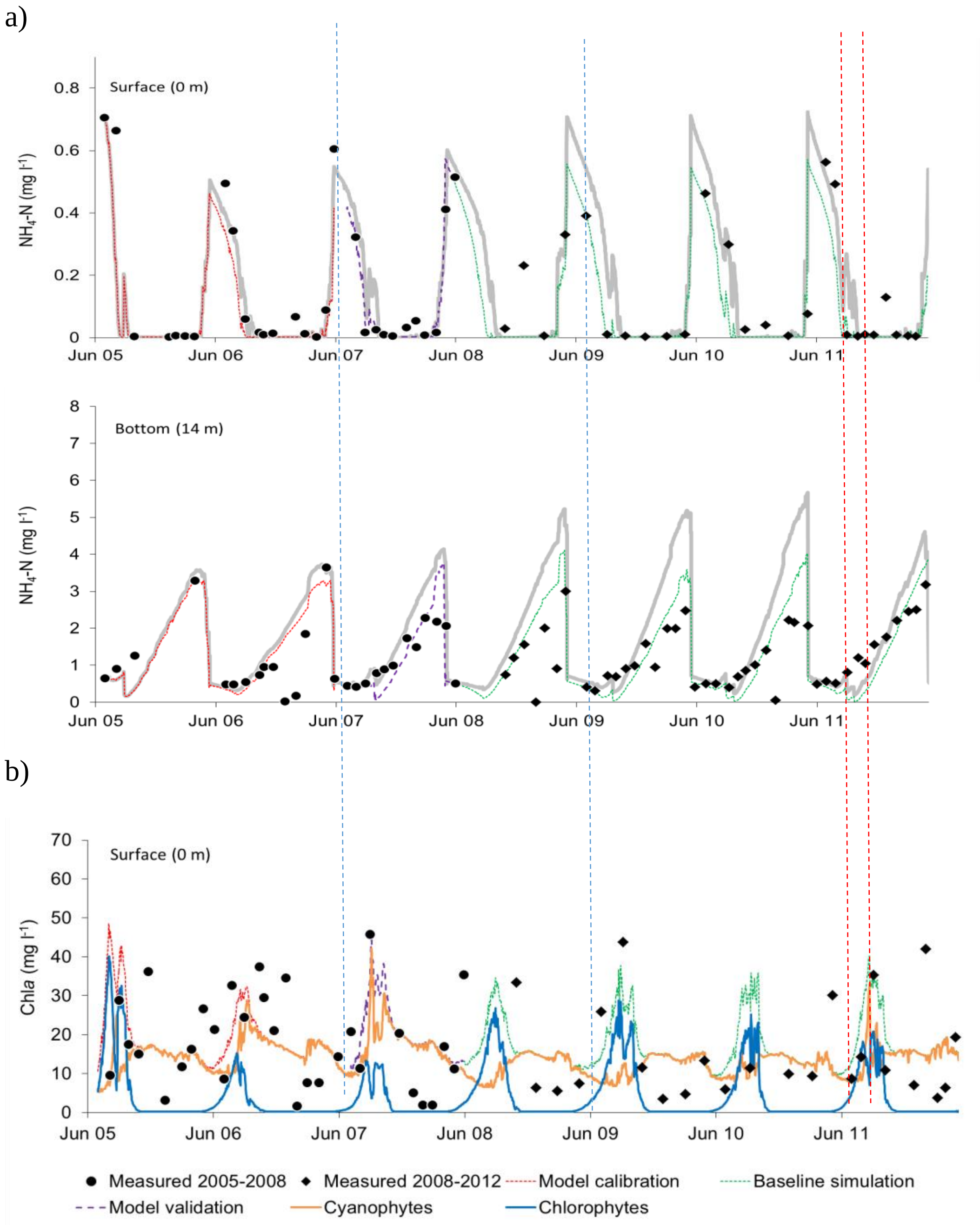



**Figure 3.7:** DYRESM-CAEDYM simulations for a) ammonium ($NH_4$-N) and b) chlorophyll *a* (Chl*a*) in the surface (0m) and hypolimnion (14m) from 2005-2012. Blue vertical lines indicate Aqual-P dosing and red vertical lines indicate alum dosing.

**Table 3.3:** Calibration and validation statistics (RMSE and R) from this assessment (2014) and Özkundakci et al. (2011).RMSE units are the same as variable under assessment and R is unit less.

| **CALIBRATION 2005-2006 (2014 assessment)** | | | | | **CALIBRATION 2005-2006 (2011 assessment)** | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **0m** | | **14m** | | | **0m** | | **14m** | |
| Variable | RMSE | R | RMSE | R | Variable | RMSE | R | RMSE | R |
| Temperature | 0.46 | 1.00 | 0.91 | 0.64 | Temperature | 0.82 | 0.98 | 0.78 | 0.98 |
| Dissolved oxygen (DO) | 1.93 | 0.34 | 2.22 | 0.88 | Dissolved oxygen (DO) | 1.81 | 0.39 | 2.20 | 0.83 |
| Phosphate ($PO_4$) | 0.01 | 0.83 | 0.09 | 0.91 | Phosphate ($PO_4$) | 0.01 | 0.82 | 0.06 | 0.92 |
| Ammonium ($NH_4$) | 0.12 | 0.92 | 0.64 | 0.70 | Ammonium ($NH_4$) | 0.11 | 0.92 | 0.50 | 0.82 |
| Nitrate ($NO_3$) | 0.04 | 0.70 | 0.02 | 0.94 | Nitrate ($NO_3$) | 0.03 | 0.82 | 0.01 | 0.96 |
| Total phosphorus (TP) | 0.02 | 0.87 | 0.09 | 0.83 | Total phosphorus (TP) | 0.01 | 0.90 | 0.06 | 0.8 |
| Total nitrogen (TN) | 0.44 | 0.24 | 0.59 | 0.80 | Total nitrogen (TN) | 0.38 | 0.27 | 0.46 | 0.78 |
| Chlorophyll (Chl*a*) | 13.1 | 0.09 | - | - | Chlorophyll *a* (Chl*a*) | 13.5 | 0.07 | - | - |

| **VALIDATION 2006-2008 (2014 assessment)** | | | | | **VALIDATION 2006-2008 (2011 assessment)** | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **0m** | | **14m** | | | **0m** | | **14m** | |
| Variable | RMSE | R | RMSE | R | Variable | RMSE | R | RMSE | R |
| Temperature | 0.55 | 1.00 | 1.26 | 0.90 | Temperature | 1.16 | 0.98 | 0.72 | 0.90 |
| Dissolved oxygen (DO) | 1.78 | 0.74 | 3.74 | 0.65 | Dissolved oxygen (DO) | 1.78 | 0.72 | 2.95 | 0.82 |
| Phosphate ($PO_4$) | 0.02 | 0.42 | 0.09 | 0.82 | Phosphate ($PO_4$) | 0.01 | 0.43 | 0.06 | 0.85 |
| Ammonium ($NH_4$) | 0.05 | 0.97 | 0.64 | 0.89 | Ammonium ($NH_4$) | 0.04 | 0.96 | 0.5 | 0.96 |
| Nitrate ($NO_3$) | 0.02 | 0.89 | 0.02 | 0.94 | Nitrate ($NO_3$) | 0.01 | 0.96 | 0.01 | 0.92 |
| Total phosphorus (TP) | 0.03 | 0.84 | 0.09 | 0.78 | Total phosphorus (TP) | 0.02 | 0.87 | 0.06 | 0.79 |
| Total nitrogen (TN) | 0.27 | 0.53 | 0.59 | 0.54 | Total nitrogen (TN) | 0.27 | 0.50 | 0.46 | 0.67 |
| Chlorophyll *a* (Chl*a*) | 23.0 | 0.70 | - | - | Chlorophyll *a* (Chl*a*) | 23.8 | 0.59 | - | - |

### 3.3.3 *Catchment scenario runs*

#### *3.3.3.1 Wetland/riparian removal (SC1)*

The lake model indicated changes in nutrient concentrations when the wetland was removed. Concentrations of TN increased on average by 28.8 mg $m^{-3}$ over seven years (2005-2012). Increases can be attributed mostly to the increased inputs of $NO_3$ through diffuse sources as a result of the absence of the wetland. Concentrations of $NH_4$ showed little change in both the surface and hypolimnion waters, however concentrations in this scenario tended to be higher than the baseline when the lake stratified (i.e. indicating increased internal loading). Concentrations of TP and $PO_4$ showed small increases (Table 3.4). Mean concentrations of chlorophyll *a* were higher over the seven years in the absence of the wetland (2005-2012). Clarity of the water indicated by Secchi disk depth was reduced, with an average depth of 2.64 m over the seven years compared to the baseline average of 2.69 m. Lake TLI over the seven-year time frame averaged 5.21 for this scenario (SC1), slightly higher than the baseline (BS) average of 5.17. Both of TLI values are still in the supertrophic category (Table 3.4), but this assessment and current BS values (TLI 5.17) indicate an improvement in water quality since the wetland was implemented.

#### *Fertilizer reduction scenario (SC2)*

A scenario of reduced fertilizer application rates showed decreases in total nutrients as well as chlorophyll *a* concentrations. In-lake TN decreased by an average of 26.9 mg $m^{-3}$ over the 2005-2012 simulation period (Table 3.4). Concentrations of TP were lower than the baseline for all seven years, by an average of 9.7 mg $m^{-3}$. The maximum reduction was in 2008, with a decrease by 11.4 mg $m^{-3}$. There was a decrease in chlorophyll *a* concentrations (average 0.6 mg $m^{-3}$) from 2005 to 2012. The TLI index for this scenario was still in the supertrophic range (5.07) (Figure 3.8), but was closer to the goal of 5.0 set in the BoPRC Lake Okaro Action Plan (Environment Bay of Plenty 2006).

*Total dairy (SC3)*

The current land use in the Okaro catchment is 95% agriculture with the most prominent nutrient contributor relating to dairy based practices. The results from this analysis, which assumed land use in the catchment to be 100% dairy, indicated a significant increase in TN (average 610 mg $m^{-3}$) and a minor increase in TP concentrations (Table 3.4) in the surface layer and hypolimnion over the seven-year simulation period. Concentrations of TN increased (average 39.4 mg $m^{-3}$). Increases in P were minimal with an average increase of 4.6 mg $m^{-3}$.

Increased external nutrient inputs in this scenario resulted in increases in chlorophyll *a* levels. Secchi disk depths averaged 2.62 m over the seven-year simulation period, differing from 2.69 m in the baseline assessment. The average TLI was 5.25 (Figure 3.8) which is markedly higher than the other two scenarios as well as the baseline.

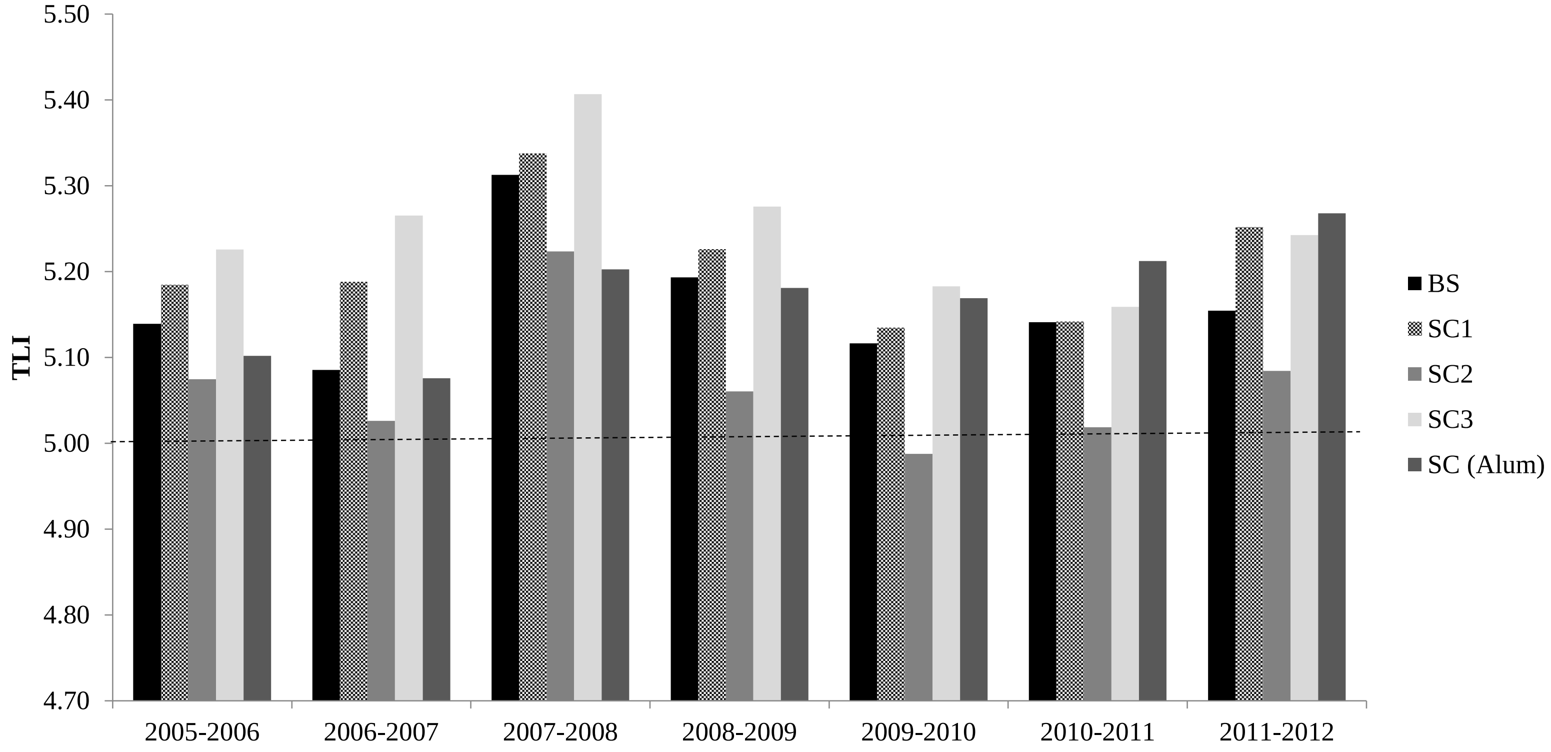



**Figure 3.8:** Lake Okaro TLI indexes for a seven-year period calculated from baseline scenario (BS), wetland/riparian removal scenario (SC1), fertilizer reduction scenario (SC2), total dairy conversion scenario (SC3), and alum scenario (SC Alum). Dashed line indicates the current goal stated in the Lake Okaro Action Plan (Environment Bay of Plenty 2006).

**Table 3.4:** Annual estimates for the four TLI constituents from the original baseline (BS) model run, wetland/riparian planting removal scenario (SC1), fertilizer reduction scenario (SC2), conversion of catchment to total dairy scenario (SC3), and alum dosing scenario (SC Alum) .

| **INCA BS** | **2005-2006** | **2006-2007** | **2007-2008** | **2008-2009** | **2009-2010** | **2010-2011** | **2011-2012** |
|---|---|---|---|---|---|---|---|
| Total Nitrogen (mg $m^{-3}$) | 556.73 | 531.34 | 680.54 | 606.68 | 547.97 | 529.90 | 542.68 |
| Total Phosphorus (mg $m^{-3}$) | 58.12 | 56.03 | 67.87 | 63.78 | 56.80 | 63.33 | 61.58 |
| Chlorophyll *a* (mg $m^{-3}$) | 19.03 | 18.55 | 19.40 | 16.93 | 17.40 | 16.33 | 18.69 |
| Secchi disk depth (m) | 2.58 | 2.58 | 2.81 | 2.79 | 2.86 | 2.65 | 2.59 |
| **INCA SC1** | **2005-2006** | **2006-2007** | **2007-2008** | **2008-2009** | **2009-2010** | **2010-2011** | **2011-2012** |
| Total Nitrogen (mg $m^{-3}$) | 564.14 | 564.92 | 693.81 | 633.52 | 573.53 | 568.57 | 599.49 |
| Total Phosphorus (mg $m^{-3}$) | 58.23 | 59.25 | 67.80 | 63.89 | 55.80 | 58.50 | 63.21 |
| Chlorophyll *a* (mg $m^{-3}$) | 19.75 | 19.50 | 19.66 | 17.31 | 17.75 | 17.31 | 20.08 |
| Secchi disk depth (m) | 2.51 | 2.55 | 2.76 | 2.74 | 2.75 | 2.54 | 2.59 |
| **INCA SC2** | **2005-2006** | **2006-2007** | **2007-2008** | **2008-2009** | **2009-2010** | **2010-2011** | **2011-2012** |
| Total Nitrogen (mg $m^{-3}$) | 536.92 | 512.20 | 635.07 | 557.07 | 514.02 | 525.29 | 527.11 |
| Total Phosphorus (mg $m^{-3}$) | 51.01 | 48.02 | 59.05 | 52.49 | 46.71 | 50.09 | 52.50 |
| Chlorophyll *a* (mg $m^{-3}$) | 18.06 | 17.96 | 18.76 | 16.10 | 16.62 | 16.27 | 18.35 |
| Secchi disk depth (m) | 2.63 | 2.62 | 2.87 | 2.81 | 2.83 | 2.67 | 2.59 |
| **INCA SC3** | **2005-2006** | **2006-2007** | **2007-2008** | **2008-2009** | **2009-2010** | **2010-2011** | **2011-2012** |
| Total Nitrogen (mg $m^{-3}$) | 574.57 | 600.92 | 734.62 | 646.11 | 575.72 | 558.57 | 580.98 |
| Total Phosphorus (mg $m^{-3}$) | 61.58 | 62.99 | 73.29 | 69.73 | 61.43 | 65.38 | 65.25 |
| Chlorophyll *a* (mg $m^{-3}$) | 20.25 | 20.83 | 20.28 | 17.51 | 18.03 | 17.32 | 19.59 |
| Secchi disk depth (m) | 2.42 | 2.52 | 2.76 | 2.72 | 2.74 | 2.57 | 2.59 |
| **SC (Alum)** | **2005-2006** | **2006-2007** | **2007-2008** | **2008-2009** | **2009-2010** | **2010-2011** | **2011-2012** |
| Total Nitrogen (mg $m^{-3}$) | 546.31 | 622.01 | 774.70 | 747.10 | 733.81 | 698.73 | 741.02 |
| Total Phosphorus (mg $m^{-3}$) | 53.04 | 39.30 | 41.44 | 43.63 | 41.74 | 48.58 | 45.62 |
| Chlorophyll *a* (mg $m^{-3}$) | 18.23 | 20.55 | 21.05 | 19.51 | 20.32 | 20.35 | 23.73 |
| Secchi disk depth (m) | 2.44 | 2.48 | 2.61 | 2.55 | 2.57 | 2.34 | 2.59 |

### 3.3.4 *Alum and Aqual-P ® assessment*

*Comparative assessment*

Alum and Aqual-P applications were initially assessed through analyzing differences between the baseline (BS) run and measured data using variables for the surface (0 m) and hypolimnion (14m) from 2008-2012. Surface concentrations of $PO_4$-P and TP from the model output were on average (2008-2012) 0.87 mg $m^{-3}$ and 14.1 mg $m^{-3}$ higher, respectively, than in the observed data. At 14m, average (2008-2012) simulated hypolimnetic concentrations were substantially higher at 81.4 mg $m^{-3}$ for $PO_4$-P and 39.60 mg $m^{-3}$ for TP (Figure 3.5a and b). As there is a notable variation between the modelled and measured outputs, the difference between the two could be identified as the average removal rate due to the flocculant applications, which was 81.4 mg $m^{-3}$ for $PO_4$-P and 39.6 mg $m^{-3}$ for TP over 2008-2012. Model variations and environmental factors should be taken into account when assessing these values.

*Sensitivity analysis*

The sensitivity analysis included the alteration of sediment parameter values according to the range found in the literature. The original parameter settings of 0.016 g $m^{-2}$ $day^{-1}$ resulted in excessive release of $PO_4$ and TP into the hypolimnion during 2008-2012, reflected in concentrations of these two nutrients that were strongly elevated above the observed values. Based on this result, $PO_4$ release was altered to 0.0088 g $m^{-2}$ $day^{-1}$ as specified in Hamilton et al. (2014) without alteration of SOD which differs in Lake Okaro. This resulted in lower surface and hypolimnetic concentrations of $PO_4$ and TP. Surface concentrations decreased by an average of 16.5 mg $m^{-3}$ for TP and 8.4 mg $m^{-3}$ for $PO_4$-P. Hypolimnetic concentrations were on average 108 mg $m^{-3}$ lower for $PO_4$-P and 119 mg $m^{-3}$ lower for TP (Figure 3.5a and b).

## 3.4 Discussion

This modelling analysis was implemented to identify the efficacy of external and in-lake remediation applications on lake water quality. This included an assessment of the effectiveness of 2.3 ha of artificial wetland and riparian margins, as well as in-lake dosings of alum and Aqual-P. A second objective was aimed at assessing and comparing the performance of the newly formatted lake model (based on catchment outputs) (INCA) to the results of Özkundakci et al. (2011) which was based on an earlier DYRESM-CAEDYM application. Both objectives were designed to provide an assessment of potential changes in lake water quality as well as the applicability of INCA and its utilization for other lake catchment simulations that include coupling with DYRESM-CAEDYM.

### 3.4.1 *Model performance and constraints*

The assessment of ecological changes in lakes sites using specific dynamic process based modelling methods requires the utilization of appropriate guidelines. This included a reassessment of the objectives, indicating the viability of the measured data, analysing performance criteria's utilized in the calibration and validation, as well as identifying potential constraints of the model as suggested by Bennett et al. (2013) and Jakeman et al. (2006).

The present modelling study of Lake Okaro followed closely to the previous assessment conducted by Özkundakci et al. (2011). Slight deviations occurred in the calibration and validation components of the assessment. The present study showed improvements in surface temperature and hypolimnetic DO model statistics, underestimation of TN in both the surface layer and hypolimnion, and poor resolution of $NO_3$ concentrations in the surface layer. Seasonal fluctuations were well represented in the hypolimnion reproducing low DO concentrations during seasonal stratification events in the summer months resulting in increased TP and $PO_4$ release. Low DO levels have been associated with nutrient release from the bottom sediments during summer stratification (Søndergaard et al. 2003a). Deviations between the original model of Özkundakci et al. (2011) and the current assessment can be attributed to differences in the forcing data in the inflow files. In the study of Özkundakci et al. (2011) discharge and nutrient data were interpolated from measured data. In my study, data was sourced from a temporal catchment model (INCA). Utilization of interpolated data formulated from sporadic measurements may not adequately account for short term (e.g., rain events) variations. Final lake simulation results indicated an acceptable fit for the majority of the variables, comparing well with other DYCD modelling assessments including those by Burger et al. (2008) and Trolle et al. (2011). Although these assessments were focused on different lakes, each assessed various scenarios to test responses of eutrophication.

Phytoplankton biomass and composition varied in the surface layer of the lake over the simulation period (2005-2012). The simulated cyanoptytes group (82%) dominated over chlorophytes (18%) (Figure 3.7b). This cyanobacterial dominance was most prominent during the winter months when the lake underwent mixing. This mixing resulted in elevated concentrations of nutrients in the surface layer. Cyanobacterial blooms have occurred in Lake Okaro since the 1950's reflecting increased external and internal loading. *Anabaena* sp., which can potentially fix nitrogen gas, have made up a substantial component of the cyanobacteria population (Wood et al. 2009). Careful consideration needs to be given to the balance of N and P as restoration actions are carried out, in order to ensure that nitrogen fixing species do not become more dominant.

The primary model limitations pertain to sediment release parameters in CAEDYM, specifically the overlying layer characteristics (temperature, DO, and $NO_3$) which influence sediment releases. These release rates do not take into account changes in sediment P concentrations which is a simplified interpretation of sediment fluxes in natural environments (Özkundakci et al. 2011). An additional factor is the exclusion of zooplankton, fish, and macrophytes from the simulation. This negates trophic cascades which can directly influence water quality in deep and shallow lakes (Schindler 2006). The model outputs from this assessment should therefore be interpreted with caution, particularly in situations where the forcing data are well beyond the bounds of what was used to calibrate the model (Jakeman et al. 2006).

### 3.4.2 *Efficacy of external and internal remediation applications on lake water quality*

*Wetland/riparian margins*

Catchment based scenarios directed towards the limitation of external loads through current remediation applications and hypothetical land use changes indicated varying results. The wetland/riparian margin removal was effective at removing $NO_3$, $NH_4$, and to a smaller degree, TP. The most significant change occurred in the form of TN which increased on average by 20-40 mg $m^{-3}$ from 2008 to 2012 when the wetland was removed. This follows the theoretical principles of artificial wetland implementations which are aimed towards the reduction of $NO_3$ and $NH_4$ by microbial denitrification in addition to other transformation processes of organic and inorganic compounds (Ballantine & Tanner 2010; Hudson & Nagels 2011). The wetland attenuated >10% of N and P loads from the two streams entering the lake. This is a relatively low N removal rate which can be attributed to an absence of particulate materials constituents in the catchment model. Even though the reduction of nutrient loads by the wetland was not completely accounted for by the catchment model, the impact on lake water quality was still significant resulting in increases in $NO_3$ and $NH_4$ concentrations throughout the water column. Reductions in P were less substantial than for N due to inaccurate sedimentation rates. A lack of P removal was predictable as wetland efficacy is reduced for TP. Even though P reductions for this scenario were lower than estimates provided in Hudson & Nagels (2011), the wetland still acted as a sink. Wetlands can also potentially act as P sources, particularly as they age, become shallower and have a large P pool (Tanner et al. 2005a). Increases in both TN and TP in lake water were responsible for increased chlorophyll *a* levels in the lake, indicating increased phytoplankton presence (specifically cyanophytes). Additional artificial wetland applications in New Zealand have been focused towards tertiary treatment of municipal wastewater, but have been used more often in lake restoration projects based around degraded water quality. The results from this analysis indicate that the input of the wetland has been effective, but monitoring should be continued to identify that the site does not become a P source.

*Fertilizer reduction*

External nutrient load reductions have been implemented in the Okaro catchment previously (Birchall & Paterson 2011). However, in this assessment a hypothetical scenario was designed around reduction of current application rates to identify the effect on water quality and its potential utility for lake management. The model outputs indicated overall reductions in nutrient concentrations. Average declines in lake water TN occurred over the seven year time frame which were in the order of 26.9 mg $m^{-3}$. The results were variable, however, and exhibited similar characteristics to the original 2011 assessment (Özkundakci et al. 2011) as well as the analysis performed by Burger et al. (2008). Both Özkundakci et al. (2011) and Burger et al. (2008) indicated a general lack of improvement in lake water quality when external reductions were the only management application. Burger et al. (2008) stated that a continuous reduction of external loads was required over extended time frames (several years) to limit cyanobacterial blooms as well as build-up of nutrients in the lake. The lack of improvement in lake TLI (average 5.07) with fertilizer reduction could be attributed to internal metabolic process, including N and P release from the sediments, which could counteract external load reductions (e.g. Søndergaard et al. 2003a). Reductions in N fertilizer without concurrent reductions in P fertilizer have the potential to alter N and P mass ratios in the lake, which could result in nitrogen fixation by cyanobacteria resulting in subsequent blooms which are already an issue in Lake Okaro (Abell et al. 2010). This is a good example of why carefully planned remediation applications should be applied in lake as well as in the external catchment. This focus should aim towards the limitation of both nutrients as individual limitations (i.e. by one nutrient) can have major draw backs.

*Total dairy*

The Okaro catchment is currently 95% agriculture, most of which is made up of sheep/beef farms. The biggest nutrient contributor on a per unit area basis is dairy farming. There is a tendency for this land use to have very large leaching rates due to excessive urea and P fertilizer inputs (McDowell & Wilcock 2008). Currently, regulation of fertilizer inputs has been implemented in the Lake Okaro catchment including better land use practices on dairy farms which entail the utilization $NH_4$

based fertilizers as well as reducing livestock numbers (Birchall & Paterson 2011). The results from the dairy increase scenario indicated large-scale increases in TN in the lake, with minor increases occurring for TP. These increases were on average 39.4 mg $m^{-3}$ higher for TN and 4.6 mg $m^{-3}$ higher for TP, compared to the BS scenario. Over the duration of the dairy scenario the overall increases in both TN and TP resulted in an average TLI of 5.25 compared with 5.17 for the baseline. This can be related to a range of factors including artificial fertilizer applications and their role in diffuse runoff and increased leaching to ground water.

*In-lake remediation applications*

Subsequent to external nutrient reductions, the application of internal remediation measures was implemented to restrict P availability in Lake Okaro (Paul et al. 2008). This included the utilization of a chemical flocculent (alum) and sediment capping material (Aqual-P) geared towards the removal of water column P in addition to capping future releases during summer stratification events (Özkundakci et al. 2010). In this analysis, there was significant variation between the BS concentrations in the model outputs and measured data over the 2008-2012 periods. Simulations over the four-year time frame were on average 30-85 mg $m^{-3}$ lower for the measured data. This may partially be caused by use of a continuous release rate in the model, which does not simulate build-up of nutrients in the sediment layer. The second portion of the analysis included the use of an altered sediment $PO_4$ release rate, which was adjusted from the original value of 0.016 g $m^2$ $day^{-1}$ (Özkundakci et al. 2011) to 0.0088 g $m^2$ $day^{-1}$ (Burger et al. 2008). It is important to note that the original SOD was not altered to parameters found in Hamilton et al. (2014) as the lake site in that analysis (Rotorua) had varying DO content. As sediment release parameters for Lake Okaro were not available, values were extrapolated from a lake with similar internal loading issues. The results from my reduction scenario indicated large decreases in nutrient values which were 119 mg $m^{-3}$ lower for TP and 108 mg $m^{-3}$ lower for $PO_4$-P compared to the BS run. Both the comparative assessment and sensitivity analysis indicated effective nutrient removal by alum and Aqual-P. This means that flocculants and sediments capping materials have effectively reduced internal loading by binding to P particles. However, it is difficult to determine what proportion of bio-available P is actually removed and or capped (Hickey & Gibbs 2009). The fit of the reduced sediment release run to measured data indicated that this is a relatively accurate assessment of P removal in the hypolimnion.

### 3.4.3 *Implications of scenario applications for lake restoration*

The results from this modelling assessment indicated both external and internal remediation activities have been effective at restoring lake water quality. The installation of the artificial wetland and planting of riparian margins have reduced TN concentrations in particular, leading to associated reduction of $NO_3$ and $NH_4$ in lake concentrations. These external nutrient reductions took place simultaneously with in-lake alum and Aqual-P applications which were effective at limiting TP and $PO_4$ release from the lake sediments into the hypolimnion during summer stratification events. The adoption of concurrent nutrient reduction of N and P was an effective strategy to restore Lake Okaro water quality. The current TLI (4.75) is a reflection of dual nutrient removal methods and has surpassed the goal of 5.0 specified in the Lake Okaro action plan (Environment Bay of Plenty 2006). This research supports the recommendations by Abell et al. (2010) that simultaneous reductions should be applied to all New Zealand lake restoration projects as limitation of both nutrients provides a more reliable method to restrict phytoplankton growth.

# Chapter 4
# General Discussion and Conclusions

The primary objective of this research was to determine the efficacy of different external and internal remediation applications on lake water quality in a small eutrophic lake (Lake Okaro, Bay of Plenty, New Zealand). Previously, restorative measures in the lake and its catchment had been assessed by Paul et al. (2008), Hudson & Nagels (2011), and Özkundakci et al. (2010 & 2011). Each of these research projects obtained nutrient estimates for specific restoration measures, but did not investigate the impacts of individual remediation methods on lake nutrient concentrations and their impacts on TLI.

This research was conducted to identify specific water quality changes for individual remediation applications and their impact on the lake TLI. To assess these changes, a two-stage process was implemented which incorporated the utilization of a dynamic (time-resolved) catchment model (INCA) to evaluate external remediation applications that altered nutrient delivery from the catchment. Simulations from this model produced outputs that were then fed into the lake model DYRESM-CAEDYM (DYCD) to identify the overall impact on lake trophic state (measured using TLI). In-lake restoration methods were investigated through the same modelling approach, but were analysed through comparative and sensitivity analysis. This analysis was applied to assess deviations between the calibrated model and the measured data, to indicate the possible impacts of restorative measures. The simulated TLI values were then compared with trophic status guidelines in Burns et al. (1999) and the Lake Okaro Action Plan (Environment Bay of Plenty 2006).

## 4.1 Catchment assessment

The INCA applications in the Okaro catchment simulated discharges and nutrient concentrations with reasonable accuracy, as quantified through a series of statistical analyses. Hydrological processes including base flow and periods of elevated rainfall-runoff were well represented in simulations of both streams as indicated by statistical fit (e.g, $R>0.50$) (Moriasi et al. 2007). Discharges were highly sensitive to periodic rainfall events possibly as result of limited interception due to the lack of vegetation. Discharge influenced nutrient inputs into the stream and increased concentrations of nitrate ($NO_3$), ammonium ($NH_4$), soluble reactive phosphorus (SRP) and total phosphorus (TP) during high-flow periods. The model was able to emulate the majority of the patterns seen in the measured data, except for $NH_4$, which was generally a relatively small component of the total dissolved inorganic nitrogen.

The first of three management scenarios was implemented to simulate the overall effects of the wetland and riparian planting implemented in 2006. These applications reduced TN and TP concentrations in the stream site preventing further degradation of the lake (Tanner et al. 2007). The results indicated notable decreases in nutrient concentrations indicative of attenuation processes for $NO_3$ through microbial denitrification. The Northern and Southern stream TP and SRP concentrations showed minor increases when the wetland was removed. The decreases in nutrients as a result of the wetland were assessed from the simulations to be less substantial than the estimates provided in Hudson & Nagels (2011). The results from the model simulations are in accordance with evidence of nutrient removal in wetlands from elsewhere. Specifically, there is increased attenuation of N (due to high affinity for $NO_3$) and limited P retention which is regulated by vegetative decomposition and sedimentation (Reddy et al. 1999; Ballantine & Tanner 2010).

The fertilizer reduction scenario indicated an overall decrease in instream nutrient concentrations for each of the simulated nutrients under assessment. Fertilizer applications in the Lake Okaro catchment are currently restricted according to Rule 11 set by the Bay of Plenty Regional Council and applied by the Okaro Catchment Lake Restoration Group (OCLRG). These reductions were far more prominent for $NH_4$, TP, and SRP, reflecting the limited use of $NO_3$ in fertilizer applications currently (Table 2.3). The results for the fertilizer reduction scenario indicated nutrient fluctuations based around altered nutrient application ratios.

The conversion of land use from 10% dairy to total dairy (100%) in the catchment produced increases for each nutrient within the streams. Nutrient loads increased by 30-36 % for dissolved inorganic nitrogen ($NO_3$ and $NH_4$), 22% for SRP and 26%

for TP. These increases are reflective of the current land use in the catchment, which is mostly sheep/beef (65%), deer (15%), and smaller areas of dairy. The catchment has been highly altered with widespread conversion of native forest and scrub present in the 1950s, to its present state of 95% agriculture. The effect of conversion to pastoral land is to alter soil and erosion characteristics to produce higher rates of sediment and nutrient loss than native vegetation. Conversely, erosion and nutrient runoff from pastoral vegetation are highly amenable due to good land use practices.

*Catchment modelling constraints*

The INCA model is an effective hydrological-biogeochemical program which provides estimates of daily concentrations and loads for a range of nutrients. However, the absence of TN in the INCA-N model is a notable omission as it limits the ability to simulate particulate forms of nitrogen. Additional limitations in the model include two separate model applications (N and P) which require individual calibrations, lack of in-channel nutrient dynamics for N, as well as the absence of hydrologically effective rainfall measurements in New Zealand.

## 4.2 Lake assessment

The Lake Okaro DYCD assessment produced a satisfactory calibration and validation in addition to identifying notable changes in nutrient concentrations for each of the scenarios. The calibration and validation followed the same parameter values set in Özkundakci et al. (2011), but differed based on modified inflow files which utilized data from the INCA catchment model in the present application. The inflow data used by Özkundakci et al. (2011) was sourced through interpolation between measured values which would have directly influenced the lake model results.

The lake model simulation represented the seasonal fluctuations well, particularly in the summer months when stratification resulted in dissolved oxygen (DO) reductions in the hypolimnion, increasing $NH_4$ and $PO_4$ releases from benthic sediments. This internal loading has been noted in stratified lakes in New Zealand (Burger et al. 2008) and globally (Søndergaard et al. 2003a).

Both external and internal restoration measures were implemented in Lake Okaro to reduce eutrophication and its associated impacts (Özkundacki et al. 2010). The artificial wetland and stream riparian margin removal scenario indicated increases for $NO_3$, $NH_4$, TP and SRP concentrations in the water column over the seven-year simulation time frame, resulting in an increased TLI of 5.21 compared with 5.17 for the baseline scenario. This indicates the wetland has been effective at removing nutrients as a result of sedimentation and microbial processes (e.g. denitrification). Additional scenarios pertaining to reductions of fertilizer applications and land use changes indicated predictable patterns including elevated lake water concentrations of N and P with the total dairy scenario and reductions of these nutrients with fertilizer reduction. The fertilizer reduction scenario indicated moderate decreases of TN (26.9 mg $m^{-3}$) and TP (9.7 mg $m^{-3}$) concentrations resulting in an overall decrease in lake TLI (5.07), but still in the supertrophic category. Total dairy resulted in increases of TP (4.6 mg $m^{-3}$) and TN (39.4 mg $m^{-3}$) resulting in increased phytoplankton biomass, directly increasing the lake TLI to 5.25 (supertrophic).

Mitigations aimed at restricting internal loading from lake sediments indicated large-scale changes over the analysis time frame (2008-2012). Removal rates were calculated using a novel method to identify differences between the original baseline simulation and an altered sediment nutrient release simulation based on literature values.  Collectively, simulations indicated that the applications of alum and Aqual-P removed over 200 mg $m^{-3}$ of $PO_4$-P from the hypolimnion from 2008 to 2012. As dosing applications differed in size it was very difficult to estimate individual application efficiencies, and this was compounded by uncertainty in the longevity of the effect of the materials. Both capping agents have been effective at removing P and suspended sediment from the water column, but consistent dosing will likely be required on a continuous basis to avoid the potential for relapses (i.e., increased trophic status) seen in other applications globally (Marsden 1989; Phillips et al. 2005; Søndergaard et al. 2007), unless catchment nutrient loads are severely constrained.

*Lake modelling constraints*

There are constraints with the lake model which should be addressed for future DYCD applications. These include the resolution of sediment parameters in the model. The current model formulation does not account for P accumulation in the benthic layer. The current program has sediment release parameters that are solely dependent on the composition of the overlying water layers (i.e. temperature, pH and dissolved oxygen). This is rudimentary as natural accumulation processes strongly regulate this release rate (Søndergaard et al. 2003a). Decomposition rates require further analysis as phytoplankton and zooplankton death rates can directly influence contributions into the lake site.

## 4.3 Recommendations

### 4.3.1 *Lake restoration*

The simulations of restoration applications implemented in Lake Okaro have indicated both positive and negative changes in overall water quality. The utilization of dual nutrient removal of N and P (wetland/riparian planting) and P specifically (flocculants) has reduced nutrient concentrations and restricted phytoplankton biomass, restoring lake water quality past the current TLI guideline of 5.0 (Environment Bay of Plenty 2006).

Future remediation applications in this lake should focus primarily on limiting internal loading. This includes additional applications of alum and Aqual-P. If chemical dosing is utilized further then dosing rates need to be more consistent for each application. This will aid in calculating the overall effectiveness of each dose. Stringent monitoring techniques will be required further both pre and post application.

Additional remediation applications in the catchment could include the implementation of wetlands associated with the Northern stream site. Recently, a new detainment bund was implemented to retain sediment and particulate P material which may otherwise be transported to the lake and influence water quality further. The bund will require monitoring to evaluate its overall efficacy.

The INCA model was effective at providing suitable nutrient estimates for the lake model. However, the INCA-N and INCA-P models differ in overall structure as well as complexity. It was stated by Bruen & Mockler (2012) that unnecessary complexity can hinder assessments, but in this assessment parameter complexity in the P model allowed for a more accurate calibration compared to $NH_4$-N results. By contrast the N model is relatively simple and the absence of particulate N was a gap that likely increased the level of error in the model fit to observed data. INCA-N outputs $NH_4$ and $NO_3$ concentrations, but does not account for various particulate materials which are incorporated in the P model. The absence of particulate materials in INCA-N could be resolved by modifying the organic N representation to be similar to that of P in INCA-P. This will require the incorporation of additional parameters for organic N and representations of associated processes (e.g. organic N mineralisation).

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