A modelling approach to assist with managing water quality in a catchment subject to rapid urbanisation: Lake Rotokauri, Hamilton, New Zealand

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Abstract

The objective of this study was to apply a coupled hydrodynamic-ecological model to a peat-stained and shallow (~4 m maximum depth) eutrophic lake whose catchment is likely to be subject to urban expansion associated with the development of Hamilton city, Waikato, New Zealand. The in-lake modelling was designed to increase understanding of the lake ecosystem and potentially to influence planning and management decisions associated with the prospective urban development project being undertaken by the Hamilton City Council (HCC). The overarching goal of the development is to accommodate urban expansion whilst retaining and enhancing the existing natural resources of Lake Rotokauri and Waiwhakareke Lake, and to restore the ecological value of the Rotokauri catchment. The main objective of this study was to understand the relationship between lake water quality and the effects of change of land-use from pastoral to urban within the Rotokauri catchment. This study incorporated results from a twelve-month programme of field work undertaken independently to the present study, into empirical calculations and computer modelling related to the catchment water budget and nutrient load, as well as the lake water quality. The fieldwork included the collection of water samples at set depths from Lake Rotokauri for the analysis of total and dissolved nutrients, chlorophyll *a* and dissolved oxygen concentrations, and water temperature. On each sampling occasion a Secchi depth was measured. The surface flow measurements and nutrient loadings via the inflows were obtained as part of a water budget calculation for the lake as well as from previous studies that used both field measurements and models to derive nutrient concentrations and loads.

An empirical water budget for Lake Rotokauri was developed to estimate the groundwater and outflow discharge as there were no gaugings that could be applied to input these variables into the lake model. Meteorological data for Lake Rotokauri was obtained from the National Institute of Water and Atmosphere Limited database, based on measurement at the Ruakura meteorological station. Meteorological data, inflows (including empirically estimated groundwater and measured surface water discharges to the lake) and the calculated outflow were entered as daily inputs to the DYRESM-CAEDYM lake model for the period of 2009. The available data relating to 2009 were looped for 2010 to check the stability of the model and its ability to capture repeated inter-annual dynamics that would be expected with identical annual forcing data input. DYRESM is a one-dimensional hydrodynamic model that predicts the vertical distribution of temperature, density and salinity. CAEDYM is an aquatic ecological model which was coupled with DYRESM as its hydrodynamic driver to simulate transport and mixing, and output temperature and biogeochemical parameters associated with lake water quality. The model satisfactorily simulated both the surface (0 m) and bottom (3 m) water temperature and the seasonal trends including the occasional stratification periods observed through spring to autumn. The model simulations showed greater departures from field data in simulating the dynamics of biogeochemical variables, particularly the seasonal dynamics of phytoplankton. The conceptual seasonal succession in phytoplankton communities depicts dominance of cyanobacteria in summer and diatoms in winter. In the observed data for Lake Rotokauri diatoms were found to be the dominant group throughout the year. The calibrated model was able to show diatoms to be the dominant group over cyanobacterial blooms. The agreement between concentrations of nitrate and dissolved reactive phosphorus in the water column was better than for chlorophyll *a*, and the observed magnitude and seasonal fluctuations at both depths (0 and 3 m) were captured reasonably well by the model simulations. The total nitrogen (TN) and total phosphorus (TP) concentrations were under and over-estimated, respectively. Dissolved reactive phosphorus (PO_4) was overestimated perhaps as a result of insufficient uptake of phosphorus by the two phytoplankton groups. As the present model does not contain a dynamic description of sediment dynamics, the sediment phosphorus release rates were influenced by user-defined maximum phosphorus release rate, temperature and the oxygen concentration in the overlying waters. Concentrations of ammonium were underestimated but it represented a relatively small proportion of TN. Due to wind-induced mixing and sediment resuspension, as well as convective sediment-water heat exchanges, phosphorus may be released from the bottom sediments where it has previously sedimented out. The model simulations may not have captured these internal loads of phosphorus adequately as sediment resuspension, for example, was not explicitly included in the model configuration.

To depict the future water quality of Lake Rotokauri when subjected to urbanisation, three scenarios were developed which involved simulations with altered nutrient loads to DYRESM-CAEDYM and comparisons with the calibrated model which represented a 'base' or present case of water quality. The scenarios considered the water quality that could evolve during and after urban development, and with a range of mitigation measures, from relatively modest treatment to best management practices to reduce nutrient loads and attenuate water flows to the lake.

The predicted nutrient load contributed from future urban run-off was less than the nutrient load from the pastoral run-off in all scenarios. The model indicated that the nutrient loading from a future catchment with little or modest treatment of the urban area (Scenario I) would be only slightly poorer water quality than Scenario II which examined the water quality during the construction phase. Scenario III (treated water) was most effective in reducing nutrient loads to Lake Rotokauri. At 3 m depth dissolved oxygen (DO) concentrations showed large fluctuations throughout the year for the both the base and untreated discharge scenarios. Chlorophyll *a* (chl *a*) concentrations for the untreated scenario were greater than in the base scenario. The timing of peak chl *a* concentrations between base and untreated discharges differed by a few days. The TP, TN and nitrate (NO_3-N) concentrations of the base scenario were greater than the untreated scenario.

Scenario II represented the intermediate stage towards Scenario 3 which was the optimal treatment case for the catchment. The greatest difference in DO at 0 m between the base case and scenario II was in March (i.e., base-intermediate = 2.76 mg L^{-1}). At 3 m depth, Lake Rotokauri was predicted to be anoxic on 4 July 2011 (0.18 mg L^{-1}) for scenario II. Chlorophyll *a* concentrations for scenario 2 were lower than the base case and PO_4-P concentrations were higher. Concentrations of NO_3-N and NH_4-N at 3 m depth for scenario II were lower than the base case.

Scenario III involved simulating water quality from with best management practices implemented. These practices included detention basins (grass-lined), constructed wetlands, biofiltration swales and floodways. At 3 m depth, fluctuations in DO concentration for both the base and scenario III were similar at the beginning of the simulated period, but for the months of May to November DO was lower in Scenario III than the base case. The maximum chl *a* for scenario 3 peaked at 30.8 μgL^{-1} compared to 38.9 $\mu g\ L^{-1}$ observed in the base model. The TP and TN concentrations were substantially lower in Scenario III than the base model. Concentrations of PO_4-P at 3 m depth were low for most of the year except in March. At 0 m depth the NH_4-N concentrations were greater than the base model from mid-June to July. Concentrations of NO_3-N for the treated scenario at 0 m depth were approximately 25% less than the base model.

Future studies should consider an ongoing comprehensive and consistent monitoring plan that would emphasise any change in the water quality of Lake Rotokauri during and/or after high-density urban developments within the catchment. Future works should involve regular monitoring that would not only limit the uncertainties in the data but also account for any effects that may be attributable to the management plan. Restoration plans should also be considered to explore the effects of biomanipulation and re-establishment of submerged vegetation. The DYRESM-CAEDYM model may also be used to examine the effects of climate change on in-lake processes and external loads to the lake.

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1.0 General Introduction

1.1 Shallow lake ecology

Shallow lakes are considered to be permanent or semi-permanent water bodies that do not stratify for long periods of time and can be colonised by submerged vegetation (Sheffer, 1998). The ratio between the catchment area and the lake area is generally high, which results in elevated nutrients and high productivity compared with deep lakes (Osmon, 2008). Shallow lakes also have intense interactions between the water column and the bottom sediments with relatively low water volume in relation to the lake bottom (Osmon, 2008). Thus in shallow lakes there is increased internal loading from lake sediments due to abiotic factors such as wind driven waves or biotic factors such as stirring of sediments by bottom feeding organisms (Dokulil et al., 2000). In shallow waters bottom shear stresses are greater than expected compared with deeper waters under similar wind stress. Waves created by wind-driven forces, set water particles into elongated elliptical orbits. When the water depth is shallow and the wave height large, the elliptical orbit creates a shear force that is strong enough to move the bed sediment (Pond & Pickard, 1983). This orbital movement is then translated to the lake bottom and is converted into an oscillatory motion (Pond & Pickard, 1983). The stress caused in the shear zone along the bottom is a function of wave depth and length and is sufficient for resuspension even when the water depth is less than one-half the wave length (Pond & Pickard, 1983).

The presence of large populations of benthivorous fish such as koi carp and catfish can also disturb the sediments (Vant, 1987). This leads to the resuspension of any organic matter into the water column that may have settled during a period of calm and stable weather. Thus biological processes can also recycle nutrients such as nitrogen and phosphorus, increasing phytoplankton productivity (de Winton et al., 2002).

The ecology of many shallow lakes has been related to the “alternate stable states” theory (Dokulil & Teubner, 2000). Shallow lakes can exist in one of two states commonly differentiated as the clear water macrophyte dominated state or the turbid state dominated by phytoplankton (Blindow et al., 1993). The clear water state includes abundant aquatic vegetation, low frequency of sediment resuspension and phosphorus recycling, low algal densities (including blue-green algae populations) and a diverse biotic community (Breukers et al., 1997). On the other hand, the turbid state is dominated by phytoplankton communities with sparsely spaced aquatic plants or none at all, undesirable populations of bottom-feeding fish communities and high bottom-sediment resuspension and phosphorus recycling (Blindow et.al., 1993). The shift from clear water to turbid water state usually occurs due to eutrophication.

Eutrophication is related to organic enrichment as a result of increases nutrient inputs (Anderson et al., 2002). Increases in anthropogenic nutrients such as phosphorus (P) and nitrogen (N) are the major cause of what has been termed “cultural eutrophication”. Over the past few decades cultural eutrophication has been associated with increases in algal biomass and blooms (Di Tullo et al., 1993). The most common harmful bloom-forming algae in freshwater lakes are cyanobacteria (blue-green algae) that are stimulated especially by phosphorus enrichment (Smith, 1983). Many species of cyanobacteria produce neurotoxins and hepatoxins that can be harmful to humans and animals. There are also other non-toxic species that can cause harm through high biomass leading to formation of surface scums, and depletion of dissolved oxygen due to algal decay, with degradation of fish habitat (Burkholder et al., 1998a). Sewage and animal wastes, industrial and agricultural runoff, contaminated groundwater inflows and atmospheric deposition of nitrogen can result in eutrophication (Cuker et al., 1990). These inputs arise as point sources and diffuse sources. Diffuse discharges are of major concern as they are increasingly prevalent in intensive agricultural systems and are more difficult to monitor and control (Howarth et al., 1996). Fertiliser usage on agricultural land is the biggest diffuse nutrient pollution source (Vitousek et. al., 1997) though its effect may be indirect, i.e., in providing a capacity to increase stock numbers whose effluent (i.e., dung and nutrients) then provides a diffuse source of nutrients. With increasing human population and an increasingly globally interconnected trade in food, there is increased loading of nutrients to many agricultural systems, particularly in New Zealand, which is reflected in greater diffuse nutrient pollution (Parliamentary Commissioner for the Environment, 2006).

1.2 Shallow lakes of the Waikato region

In the Waikato Region, lakes vary widely in physical, chemical and biological characteristics. The majority of the lakes are small in size (<10 ha) and shallow (Barnes, 2002). Many of the shallow lakes were formed during the late glacial period and their catchments were either densely covered with forests or consisted of peat-bogs (McGraw, 2002). Māori first arrived in the Waikato region in the 14th century and began to alter the landscape with burning and clearing practices. However, after the arrival of Europeans vegetation changed rapidly due to the clearing and burning of forests for timber, food cultivation and settlement (Newnham et al., 1995). With European settlement in the 20th century, dairy farming was established. Lakes and wetlands were drained for pastoral and personal usage. This abrupt change from forest to agriculture substantially increased the external loads of nutrients and sediments into the lakes (Environment Waikato, 2008) thereby increasing the production of phytoplankton.

Many shallow lakes in the Waikato are prone to wind exposure that regularly stirs up the bottom sediments of the lakes via wind-driven wave resuspension, especially when the lakes lack vegetation cover, i.e., are in the turbid state. Past records show that most of the shallow lake beds in the Waikato region were extensively vegetated (usually with charophytes and milfoils) (Kirk, 1980). However, it is difficult to establish if the complete disappearance of these vegetated beds occurred due to the rapid change of land use in the 20th century along with the changes to the inflows and water levels of the lakes, or whether other factors have been involved in the devegetation of weed beds from nearly all shallow Waikato lakes (Clayton, 2002).

The introduction of invasive plant species such as oxygen weeds (e.g. *Egeria densa* and *Elodea canadensis*) also had a negative impact on the native submerged plant communities (Wood & Mason, 1977). These weeds colonised many lakes from the 1950s to the 1980s, forming tall dense compact growths that outcompeted the naturally occurring native submerged macrophytes and ultimately helping to contribute to the complete eradication of all submerged weed beds from many lakes (de Winton et. al., 2009). Shallow lakes in New Zealand and in the Waikato region in particular have been subject to activities such as drainage for agriculture, man-made stop-banks and various flood-control measures (such as weirs and flood control gates). These artificial interventions have a strong effect on the natural seasonal variations in water levels (Hamilton et. al., 2009) and in many cases lake water levels are artificially manipulated by a weir. This potentially allows for the maintenance of higher water levels in summer than in winter. For example, in Lake Ngaroto elevated levels in summer assist in recreation such as boating, kayaking and duck shooting, while lower winter levels assist with draining of pastoral lands and wetland areas (Ministry for the Environment, 2001). With the establishment of weirs and flood gates the seasonal lake level fluctuations have been shifted substantially from the naturally occurring variation (Hamilton et al., 2009).

1.3 Lake models

Lake models have been established to simulate the different biological, chemical and physical processes. They can also be used to examine the interactions of different land uses within a catchment on the receiving waters, specifically lakes (Bruce et. al., 2006; Trolle et al., 2008a, b; Gal et. al., 2009). These models in turn simulate these processes to provide insights into strategies to either improve or stabilise lakes from further degradation. The models ELCOM, DYRESM, and CAEDYM were developed by the Centre for Water Research in Perth, The University of Western Australia. DYRESM (Dynamic RESvoir Simulation Model) is a one-dimensional (1D) hydrodynamic model which can be coupled with CAEDYM (Computational Aquatic Ecosystem Dynamics Model) to simulate important biogeochemical variables that are usually associated with lake water quality (Trolle et al., 2009). A detailed understanding of the theories used in DYRESM-CAEDYM can be found in a precursor model by Hamilton & Schladow (1997), while Imerito (2007) describes the new version in detail, with applications to a range of lakes given by Trolle et al. (2009).

DYRESM was designed to simulate the vertical distribution of temperature, density and salinity in lakes (Imerito, 2007). The model is based on the one-dimensional approximation assumption, that is, that the vertical variations in the lake play a more important role than the variations in the horizontal direction (Hipsey, 2006). The one-dimension approximation is valid when physically destabilising forces such as wind stress, surface cooling or plunging inflow do not act over prolonged periods of time (Imerito, 2007). The model therefore slices the lake of interest into a horizontal layered structure from bottom to top that allows vertical profiles of variables to be obtained from each layer (Hipsey, 2006). These horizontal layers of uniform property but varied thickness can expand, contract, move up and down and amalgamate due to the affects of inflows entering and outflows leaving the system, or from evaporation and rainfall (Imerito, 2007). The layer thickness changes to accommodate the volume changes. The advantage of the horizontal layer structure is that it gives flexibility to accommodate the varying vertical density structure of the lake (Imerito, 2007). DYRESM can simulate mixing of the bottom waters in the event of drastic climatic conditions such as storms and floods. One drawback of this model is a large amount of field and meteorological data inputs are required at relatively high resolution that may far greater than that of the measurements. The benefit, however, is that greater confidence can be placed in the outcome, e.g., understanding the seasonal variability in the hydrodynamics of a lake and prediction of long term changes in environmental and/or anthropogenic factors that could be detrimental to the lake's trophic state. CAEDYM is an aquatic ecological model system which has been designed to be coupled with a hydrodynamic driver such as 1-D YRESM or 3-D ELCOM. The coupling is essential as the thermal structure of the water column is dependent on the water quality parameters (Pilgrim, 2007). Hence, the model has been configured to simulate the major elemental cycles, inorganic suspended solids and phytoplankton groups of the water body (Ambrose et al., 1988).

1.4 Lake management

Most catchments comprise of a variety of land uses, landscape designs, soil types, climates and vegetation, making them a unique, inter-related, complex system (Gburek et. al., 2000). Different parts of the catchment are therefore predisposed to different levels of runoff of nutrients and sediments (Keipert et. al., 2008). By pin-pointing the critical source areas, adverse effects of these inputs can be mitigated via application of best management practices (BMPs). Analysis of 134 New Zealand lakes showed that eight out of the 18 most hypertrophic lakes are shallow lakes within the Waikato region (Hamill & Lew, 2006). Shallow Waikato lakes have many attributes such as high pastoral cover within their catchments, depth of less than 10 m, warm regional climate and low altitude, which in many cases contribute to the poor water quality condition (Sorrell & Unwin, 2007).

The basis of documenting effects of any management strategy is monitoring of the water quality. It determines the starting point and is specific to the questions being addressed (Osmon, 2008). Water level management also helps to promote and facilitate aquatic plant growth. Aquatic plants have a positive influence on the shift to clear water state by promoting piscivorous fish populations, competition with algae and limiting sediment resuspension (Scheffer et al., 1993). Lakes that are maintained at high levels may facilitate various forms of recreation while lowering water levels could allow light to reach the lake bottom and promote aquatic plant growth provided there is no additional sediment resuspension (Reeves et al., 2002).

Sediment resuspension is a source of nutrients and turbidity as suspended sediments intercept and limit light entering the water column and introduce nutrients (James et al., 2002). The key management options for reducing resuspension may be by reducing the impacts of wind-induced waves and removal of invasive bottom-feeding fish communities such as carp. Controlling external loading by managing nutrient sources in the catchment is the key to long-term improvements in water clarity and quality of shallow lakes (Osmon, 2008). It is important to develop water and nutrient budgets as they can be used to identify different land uses and practices that contribute to the load. This then allows implementation of different BMPs to regulate the nutrient loads entering from different land uses.

1.5 Lake Physical Limnology

1.5.1 Significance of lakes

Lakes are important ecosystems for water supply, agriculture, irrigation, recreation and hydropower (McGinnis & Wüest, 2005). Many support increasing urban populations that rely on these resources to perform day-to-day activities. As a result of the proximity of human settlements to many lakes, and due to human-induced land use change, the ecological integrity of lakes can be threatened by excess nutrients, stimulating growth of algae and water plants, and invasive species (Osmon, 2008).

The hydrodynamics of a lake vary across the globe because of many different physical features such as surrounding topographies, lake surface areas, meteorological conditions, and hydrological and geochemical loadings (O'Sullivan & Reynolds, 2004). Variations in lake surface area play a key role via heat fluxes to the lake surface, whilst biological and chemical constituents in the water can also affect the stratification and water movement of a lake (Lehman et al., 1995). Lakes become stratified when the dominant heat source from solar energy creates vertical energy differentials that exceed those related to the breakdown of stratification. As water expands, this thermal energy is converted into mechanical stability that checks the motion in the lake, hence, the stronger the thermal insolation the greater the potential for vertical density stratification (Imberger & Patterson, 1990). The stratification of a lake, however, can be modified by wind, stream inflows and outflows. Therefore, the internal hydrodynamics of a lake are important in understanding the biological, physical and chemical processes that can be used to effectively manage lake ecosystems (Lehman et al., 1995).

1.5.2 Lake thermal stratification

The key factor in the stratification of any given lake is the temperature dependence of water density (Schwab & Beletsky, 2003), allowing limnologists to classify and distinguish between different lake types. Thermal stratification in shallow lakes ranges on a time scale of a few hours to a few days up to a few weeks. The duration of stratification is related to the ratio of surface area to water depth, as well as higher levels of 'peat-stained' organic matter and chlorophyll that commonly occur in shallow lakes (Hamilton et al., 2009). Light penetration may be significantly reduced from high levels of coloured dissolved organic matter that leaches from the surrounding peat, as well as light absorbing phytoplankton and inorganic suspended sediments that result in high heat entrapment of radiation in surface waters, thus increasing the occurrence of thermal stratification (Davies-Colley et al., 1993). Stratification occurs in warmer months and, depending on the duration of the stratification and oxygen consumption rate (influenced by the organic loading to the bottom waters and the bottom sediments), can strongly deplete dissolved oxygen (DO) in bottom waters (Hamilton et al., 2009). Shallow lakes are mostly susceptible to mixing by the wind and therefore do not generally experience prolonged thermal stratification and therefore remain well oxygenated despite higher nutrient and phytoplankton concentrations (Hamilton et al., 2009).

1.5.3 Dissolved Oxygen (DO)

The levels of dissolved oxygen (DO) in a lake are the net result of metabolic processes and transfers across the air-water interface. Concentrations of DO are balanced between atmospheric oxygen supply, photosynthesis and the metabolic processes that require oxygen (Thornton, 1987). Levels of DO not only affect the distribution and growth of aquatic fauna but also have a major influence on the partitioning of inorganic nutrients between water and sediment. The solubility of DO in a freshwater system depends mainly on the temperature of the water (Thornton Kimmel & Payne, 1990). Other factors that influence DO solubility are salinity and pressure (Thornton, Kimmel & Payne, 1990). In lakes, wind-induced turbulence generally plays a major role in the rapid gain or loss of DO. Photosynthesis by phytoplankton, benthic algae and submerged vegetation adds DO in the photic zone, whereas community respiration consumes DO. Therefore, DO and the ratio of photosynthesis to respiration show substantial variation during seasonal stratification (Talling, 1969)

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1.5.4 Phosphorus (P) Cycling

The eutrophication of a freshwater ecosystem can be controlled via nutrient inputs such as P and N. Many studies conducted over the past few years have also focused on increasing the N:P ratio by reducing P in the aquatic system (Kalff, 2002). The justification for this approach is that atmospheric fixation by cyanobacteria can increase N concentrations, particularly at low N:P ratios making it potentially difficult to control N concentrations in lakes. The removal of P via chemical and biological processes is considered to be more feasible than the removal of N as P usually enters an aquatic system from land surfaces.

Based on the early research by Mortimer (1941), ortho-phosphate (PO_4^{3-}) under aerobic conditions is mostly either associated with iron oxyhydroxides (e.g. FeOOH) or is precipitated as $FePO_4$. The phosphate in the lake is then removed from the water column through sedimentation, including organic particles that are generally associated with algal cells (Prairie et al., 1988). The FeOOH flocs then serve as an effective adsorption surface at the oxidized sediment surface by trapping phosphate that is released after decomposition of the organic matter on the sediment bed, preventing PO_4 diffusion upwards. Phosphate and iron may diffuse into the hypolimnion, however, when anoxic conditions occur (Prairie et al., 1988). This is known as internal loading and is associated with presence of other reduced redox species.

It was proposed that the release of phosphate from anoxic sediments is not only affected by iron but also can be affected by the presence of sulfate (Kalff, 2002). They argued that with the microbial reduction of sulfate, the end product is sulfide in the form of insoluble FeS and FeS_2 which removes and reduces iron from the system allowing increased release of phosphate (Rigler, 1973). The Fe:P paradigm has continued to dominate the scientific thinking on P availability from sediments for most of the 20th century (Bostrom et. al., 1982) rather than the S:P paradigm. It has not been until the last three decades of the 20th century that a modern model has been developed that highlights that some lake systems do not fit the classical view of Mortimer (1941).

Recent research has postulated that phosphorus release rates from sediments are most affected by biological mechanisms (i.e., microbial decomposition) rather than chemical mechanisms (Prairie et al., 2001). It was proposed that the bacteria are directly involved by releasing soluble reactive phosphorus (SRP) in the water which is followed by cell lysis and release of solubilized polyphosphate granules produced under aerobic conditions (Prairie et. al., 2001). According to the work done by Prairie et al. (2001), the classic model did not apply on the basis that there was no relationship between iron and phosphorus release in the anoxic hypolimnion of Quebec lakes. They also found that oligotrophic lakes with anoxic hypolimnia may not necessarily exhibit P release from bottom sediments.

With changes made to the classic model, the modern model has allowed consideration of some alternative hypotheses that can account for little or no release of phosphorus in lakes with anoxic hypolimnia (Caraco et. al., 1991). The classic model also highlights the behaviour of phosphorus only in the hypolimnion rather than the whole lake system. The classic model fails to explain the P release from turbulent littoral zones, which can occur as a result of the P concentration gradient at the sediment-water interface (Bostrom et. al., 1982). Therefore, to re-iterate, many factors such as physical and biological processes are responsible for phosphorus release from lake sediments. Elevated internal loading in shallow lakes with oxic sediments has been suggested to be linked to greater temperature in shallow lake sediments (Søndergaard, 1989). Respiration and decay rates of macro-invertebrates can also increase P release from oxic sediments of lakes (Søndergaard, 1989).

1.5.5 Nitrogen (N) Cycling

Nitrogen plays a pivotal role in an aquatic system. Nitrogen and phosphorus are the two elements that are in high demand relative to their respective supply to plants and heterotrophic microbes (Vitousek et. al., 1997). Therefore, nitrogen supply is important to determine the primary productivity of freshwater systems and microbial recycling of organic matter. Nitrogen exists in various oxidation states that allow it to serve as both an electron donor and acceptor in different oxidation-reduction reactions that affect nutrient cycling and biochemistry (Boring et.al., 1988).

Nitrogen becomes part of the particulate organic nitrogen (PON) pool when fixed by photosynthesis or heterotrophic microbes along with nitrate (taken up by photosynthetic organisms) via a process called assimilative nitrate reduction (Reddy et al., 1989), more commonly known as nitrogen fixation.

In this process photosynthetic energy is captured and then used to incorporate nitrogen into the protoplasm of cells. When DO is absent from the water column, the oxidized forms of nitrogen (NO_3 and NO_2) act as the final electron acceptor in oxidizing organic matter at the oxic-anoxic interface, often at the sediment surface (Rudd et al., 1988). These oxidized forms of nitrogen are reduced via denitrification, with N_2 and N_2O as end products that are released into the atmosphere. A part of NO_3 may also be reduced to NH_4 in dissimilative ammonium production, but, the largest source of ammonium comes via a process called ammonification. It begins with detrital particles in the epilimnion, which sediment out and are mostly deposited at the sediment surface (Rutherford et al., 1987). In eutrophic lakes, large quantities of inorganic nitrogen in the form of ammonium are released from the sediment into the water column.

Nitrification is a biologically-mediated oxidation conversion of NH_4 often to NO_3 and is catalysed by a number of micro-organisms that in return receive energy from the conversion for their metabolism (Boring et.al., 1988). Nitrification plays a key role in nitrogen cycling as the oxidized forms of nitrogen, NO_3 and NO_2, take part in denitrification reactions, yielding N_2 that is lost to the atmosphere. The nitrification reaction is:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+ \qquad (1)$$

This reaction shows that two moles of dissolved oxygen are required to convert one mole of NH_4^+ to 1 mole of NO_3^+. Therefore, nitrification places huge demands on the supply of DO that is in the hypolimnion. Nitrification takes place in oxygenated environments (Reddy et al., 1989). In the anoxic zone, due to ammonification, there may be high concentrations of NH_4^+ in the sediments. Nitrification generally occurs within the sediments at depth ranges between zero to a few centimetres but if the penetration of DO is low or absent then concentrations of NH_4^+ tend to build up. This depth of DO penetration is usually determined by the availability of DO from the overlying waters and by the thickness of the diffusive boundary layer (DBL) (Reddy et al., 1989). Hence, nitrification is a product of three essential substrates: the supply of oxidized forms of nitrogen, the supply of DO and the availability of CO_2, as well as water temperature (Prosser, 1986). Carbon-dioxide is a requirement because nitrifying microbes chemosynthetically reduce CO_2 into organic carbon. Nitrification rates are not only affected by the presence/absence of particular substrates and temperature but also by pH, with rates reduced at pH<5 in acidified waters. Two groups of bacteria carry out the nitrification process. Both the ammonium oxidizing bacteria (better adapted to low DO concentrations) and nitrite oxidizing bacteria are chemoautotrophs and are capable of obtaining energy released from organic matter oxidation to CO_2, to satisfy their carbon demand (Prosser, 1986).

Denitrification on the other hand is a bacteria-mediated process of reduction of nitrogen oxides initially into gaseous NO^- and N_2O and then dinitrogen gas, N_2. This process is carried out by various heterotrophic anaerobic bacteria and by fungi at oxycline interfaces in lakes (Knowles, 1982). The microbes use NO_3^- or NO_2^- as an electron acceptor in the oxidation of organic matter. Even though denitrification and nitrification are coupled together, denitrification is responsible for the removal of fixed nitrogen to the atmosphere, mainly as N_2.

1.5.6 Phytoplankton

A great deal of research has been done on the community structure and productivity of phytoplankton in relation to environmental aspects of lakes. Seasonal variations stimulate changes in phytoplankton dominance, mechanisms, processes and composition in lakes (Reynolds, 1984a). Individual lakes have their own repetition of seasonal and inter-annual phytoplankton dominance that has been related to the environmental drivers. Many factors such as light, nutrient availability, CO_2, pH, temperature, turbulence, competition and predatory selection pressures determine the dominance of different phytoplankton genera.

The main determinant of phytoplankton species' growth rates is cell size and concentration. Small-sized phytoplankton generally dominates the mean community biomass but lakes with high P concentrations are more likely to have species with large-celled algae (Reynolds, 1984a). At the onset of summer stratification, phytoplankton communities adapted to low-light are gradually replaced with slow-growing, large cell-sized phytoplankton.

1.6 Lake Rotokauri

1.6.1 General Description

Lake Rotokauri is a polymictic, peat-stained lake, located 7 km north-west of Hamilton City (Warr, 1998). It is connected to the Waipa River via the Ohete Stream and has lake levels maintained by a weir at the outlet (Boswell et. al., 1985). Rotokauri is one of the largest shallow lakes surrounding Hamilton and is one of more than 40 lakes between Te Kauwhata and Te Awamutu (Environment Waikato, 1998). The catchment now comprises of residential, pastoral, forest and open waters. In the past, large parts of the catchment were high-producing peat land with high fertility soils, while the remaining parts were sandy hills with fertile ash-derived soils (Waikato Valley Authority 1979). Drainage from the peat adds tea-stained water to Lake Rotokauri. The lake is of great value as it has more marginal vegetation than most lakes even despite continuous modification and a variety of catchment land uses (Rotokauri Management Plan, 2000).

The lake surface area has previously been reported as ~77 ha (Barnes, 2002; Boswell et. al., 1985; Cooke & Parkyn, 2005) but also ~44 ha (Jenkins & Vant, 2007). These two surface areas are different due to the former being set by GIS, including wetland margins and the latter based on an open-water area (Diffuse Sources, 2010). In the current study the area was found to be ~37 ha (377 800 m^3) excluding the marginal wetland and some inaccessible regions of the lake. This lake surface is therefore closer to the surface area described in Jenkins and Vant (2007).

Figure 1.1: Aerial View of Lake Rotokauri (www.waikatoregion.govt.nz).

1.6.2 Origin and changes to the lake and its catchment through time

Lake Rotokauri was formed when the migrating Waikato and Waipa Rivers accumulated large sediment deposits at the end of the glacial period. The accumulated alluvial deposits blocked the valley mouths causing water to build up and Lake Rotokauri to form as a shallow lake (Green, undated). Lake Rotokauri is in a temperate climate, has a lowland position, shallow waters and large catchment area relative to other lakes in the region (Green, undated). Furthermore, Green (undated) suggested that in the past in its pristine state, the lake would have been moderately clear, supporting a large, diverse community of plant and animal species. An extensive fringe of wetland vegetation likely supported a healthy lake margin (Green, undated; Rotokauri Management Plan, 2000).

The Rotokauri catchment prior to human settlement was dominated by peat bog vegetation and forests. However, over the last 150 years frequent human interventions have resulted in substantial changes to the lake and its catchment (Warr, 1998). Today, agriculture is the main land use in the catchment. On the catchment's eastern boundary lies an area of industrial and residential settlements, while a small area of rural residential land is located on the southern margin of the lake (Fig. 2.2, Rotokauri Management Plan, 2000).

The catchment hydrology of Lake Rotokauri has changed significantly with agricultural and urban development. Prior to the modifications made to the inflows, runoff (diffuse) and groundwater flow from the catchment would have been the sole water input (Warr, 1998). Now, the lake is fed by runoff that is directly channelled by a number of open drains as well as via groundwater. Rotokauri Stream (in other reports referred as Te Rapa Drain) is the main inflow into Lake Rotokauri and drains about 60% of the total catchment (Diffuse Sources Ltd. Report, 2008). Other drains are sourced from Hamilton Zoo pond system, and a number of small farms. The changes in catchment vegetation and the introduction of drainage systems have resulted in increased nutrient and sediment levels entering the lake (Environment Waikato, 1998).

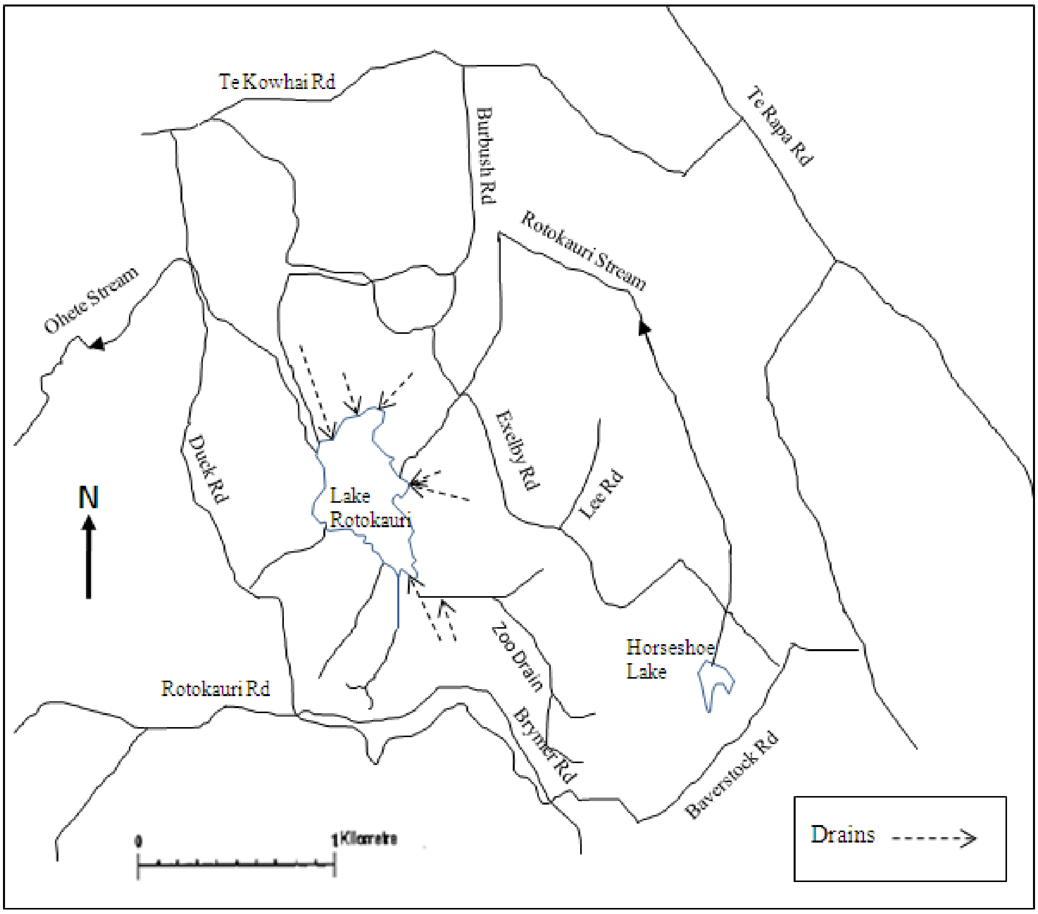


Figure 1.2: Map of Lake Rotokauri and surrounding roads and drains (www.maps.google.co.nz/rotokauri).

Lake Rotokauri discharges into the Waipa River via Ohete Stream (Figure 2.3). However, due to the continuous deepening of this stream there has been a ~5 m decrease in the lake level, resulting in the lake now covering only about half of the area it did in the 1860s (Norris, undated). The reduction in the lake area and associated changes to the lake and its catchment has had a significant impact on the biology and water quality (Warr, 1998).

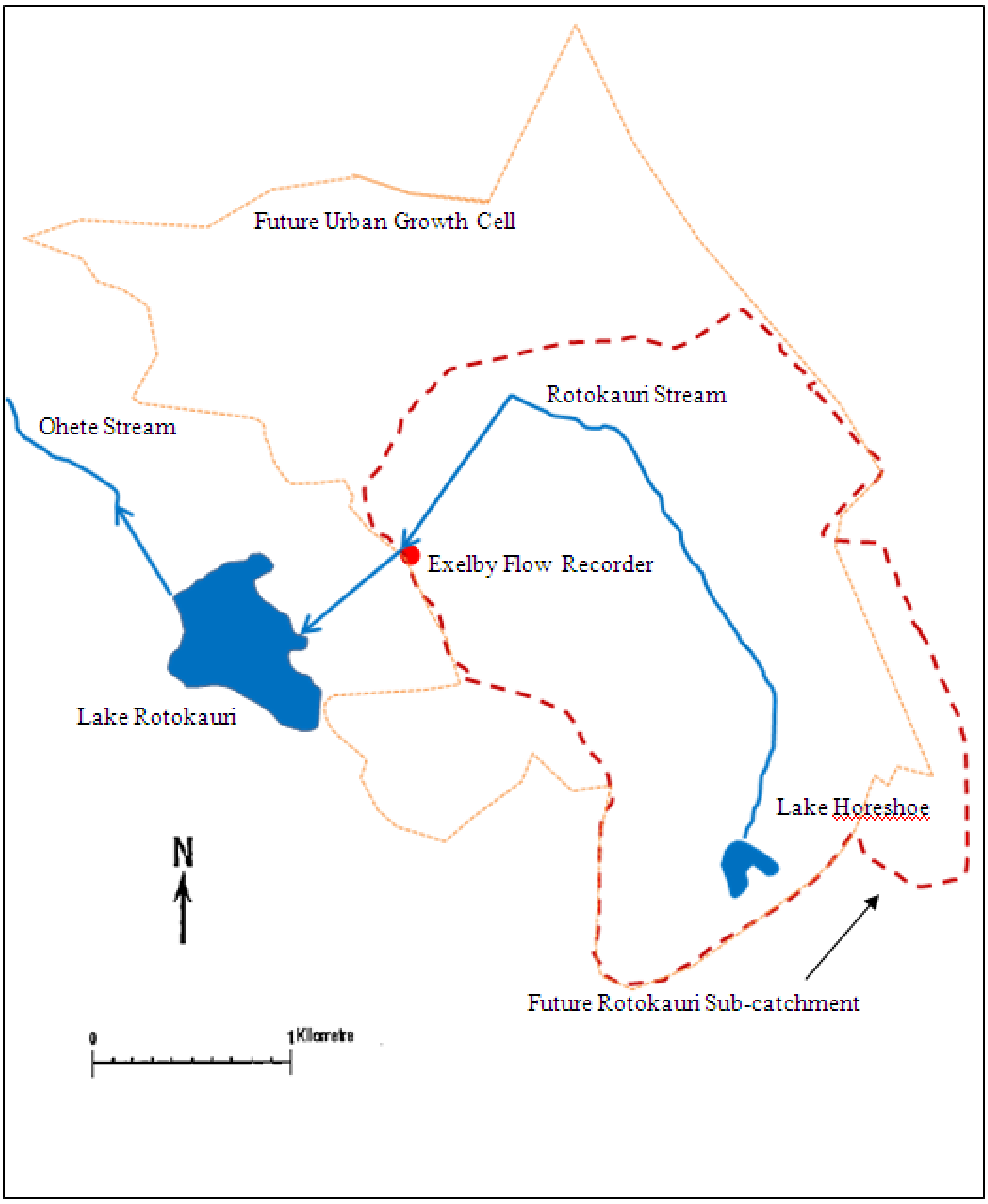


Figure 1.3: Map of Ohete Stream outlet and Rotokauri Stream (formerly known as Te Rapa Drain) and the flow recorder at Exelby Road. Adapted from Diffuse Sources Limited (2008).

1.6.3 Aquatic Flora and Fauna in the 1980s – 2000s

Twenty-two species from six different phytoplankton genera have been were identified in Lake Rotokauri in the 1980s. A thin band of marginal vegetation of raupo (*Typha orientalis*) and rushes (*Eleocharis sphacelata*) surrounded the whole lake (Boswell et. al., 1985). In September 1979, a survey of the lake indicated that ~60% of the lake had *Egeria densa* beds that reached the surface (Champion et al., 1993). Stands of *Leptospermum* represent the former indigenous marginal vegetation before it was cleared for pastoral land use. In November 1982, the Waipa County Council was granted permission to use Diquat herbicide for the control of submerged weeds in Lake Rotokauri (Boswell et al, 1985). The Waikato Valley Authority monitored the spraying programme (Barnes, 2002). Even though no side-effects were detected, the programme failed to achieve the intended reduction in weed growth (Barnes, 2002). During late summer of 1996-97 a decline of *Egeria* beds was observed by the residents living in close proximity to the lake (Warr, 1998). The lake appeared to be at this time becoming devegetated and observations made in February 2002 (Barnes, 2002) indicated that the lake bed was devoid of submerged macrophytes. It has been proposed that the collapse of submerged vegetation might have accounted for the observed change in the water quality observed from records in the 1980s and between 1995 -2001 (Barnes, 2002). There are no other records of the presence of any submerged vegetation other than those mentioned in the study conducted by Boswell et al. (1985). Therefore, it is hard to report in this thesis about any recent changes in the submerged vegetation.

The marginal wetland zone of the lake supports various diverse and large populations of waterfowls and wetland birds (Boswell et.al., 1985). In the past, the *Egeria densa* beds provided refuge and support to a large black swan population (Boswell et. al., 1985) with waterfowl hunting also a major sport in the catchment.

1.6.4 Water Quality

The margins of Lake Rotokauri are swampy with soft, deep mud. The water that enters from the peat area of the catchment has been reported to be discoloured and acidic. A water quality sampling programme was carried out from January to March 1980 and again in August, 1980, as reported in Boswell et al. (1985).

The summer data demonstrated that the surface water temperature peaked up to 25 ^{o}C, and that the lake stratified for sufficiently long periods for bottom waters to deoxygenate, even though the depth was only 4 m (Boswell et al., 1985). Dissolved oxygen was also noted to be zero on one occasion at the lake bottom. Water quality samples were also collected monthly between August 1997 and December 2001, as reported by Barnes (2002) (See Appendix I). Chlorophyll *a* concentrations peaked in summer and showed an increase of 22 mg m^{-3} yr^{-1} with an average value of 79 mg m^{-3} through the sampling period (Barnes, 2002). The lake was considered to be in a hypertrophic state and had high levels of TP (average = 125mg m^{-3}), TN (average = 165 mg m^{-3}), phytoplankton biomass (average = 70 mg m^{-3}) and low water clarity (average Secchi depth = 0.60 m) with a mean TLI value of 6.36 (Barnes, 2002). Other variables analysed included: nitrate, total organic nitrogen, ammonium, dissolved reactive phosphorus, and total Kjeldahl nitrogen. While nitrate, ammonium and dissolved reactive phosphorus showed no significant change in time, total organic nitrogen showed a gradual increase (132 mg m^{-3} yr^{-1}, $p < 0.02$), with a mean concentration of 1375 mg m^{-3}.

Barnes (2002) indicated that high concentrations of nitrate were due to the release of ammonium which was subsequently oxidised and that the increase in concentrations of total organic nitrogen and chlorophyll *a* were from increased assimilation of nitrogen by an increasing phytoplankton biomass. Total suspended sediments were also found to be high (average = 20 g m^{-3}) and comprised approximately equal amounts of organic and inorganic material. During the study the lake became increasingly dominated by organic matter, and the total suspended solids to total volatile solids ratio (TSS:VSS) showed a change where VSS had increased at a rate of 2.3 g m^{-3} yr^{-1} while TSS remained unchanged ($p > 0.1$) (Barnes, 2002). Secchi depth ranged between 0.6-2.6 m in Boswell (1985) compared with 0.25-1.35 m in Barnes (2002). The conclusion of Barnes (2002) was that the lake was in a turbid state that was clearly dominated by phytoplankton in contrast to the dominance of the submerged vegetation observed in the 1980s. Jenkins and Vant (2007) assessed the extent of the current loading of nutrient and sediment in 14 Waikato lakes (including Lake Rotokauri) using lake catchment land cover and the farming practices. The four land-use types observed were pasture, dry-stock pasture, surface water, and other land excluding agriculture, forested or urban developed areas.

1.6.5 Future Urban Catchment

Rotokauri catchment was once dense forest and raised peat-bogs, but is now mainly pasture. It will be subjected in the future to significant growth in the north-west part of Hamilton. Hamilton City Council (HCC) has developed an urban structure plan that includes residential, recreational, commercial and industrial zoning within the catchment. The primary objective of the plan is to facilitate sustainable growth of Hamilton City. One of the goals to achieve this overall objective is to provide and promote for industrial development of the Rotokauri catchment area in a sustainable manner so that it meets the social, economic and environment necessities of the communities and all other stakeholders eventually leading to integration within the existing urban infrastructure and communities (Rotokauri Structure Plan, 2007). Another goal outlined by the Hamilton City Council in the structure plan (2007) is to establish a distinct but inclusive local identity for Rotokauri by utilizing a unique urban design to define the character area and open breaks between zones and sub-areas by incorporating significant landscaping, and, integration of cultural features of the area & existing facilities. However, the main and the most prominent goal enshrined by the Council are "to sustain and enhance existing natural resources and features, and, restore the ecological value of the Rotokauri catchment" (Rotokauri Structure Plan, 2007). The primary objectives of maintaining its existing natural resources are to safeguard & sustain the areas of indigenous vegetation and water features & habitats, maintain the water table within peat soils area, and, develop linkages between Horseshoe Lake (Waiwhakareke), Lake Rotokauri and the reserve network.

These main objectives and primary goals will facilitate and ensure that there is economic prosperity, social progress, environmental protection and sustainable use of the natural resources through planning and implementation processes, thereby assisting the communities to preserve and enhance the distinctive identity and heritage of the area (Rotokauri Structure Plan, 2007). A variety of techniques are planned to be utilised to successfully implement the Council's "Structure Plans' Goals and Objectives" which includes the phasing of development and infrastructure. To achieve sustainable management of natural and physical resources "staging of development" is also planned to be used.

The Rotokauri Urban Structure Plan Area comprises approximately 1048 ha of growth area most of which is currently pastoral land. Development and urbanisation within the city and agricultural activity across the majority of structural plan area have affected the catchment areas. Consequentially, the water and environmental quality of Lake Rotokauri have gradually and progressively degraded over a period of time.

Lake Rotokauri is also highly valued as a recreational resource. However, concerns have been raised as to the effects of urbanisation and the greater nutrient loads to the already hypertrophic lake. To assess these concerns HCC commissioned Diffuse Sources Limited to further investigate the present and future nutrient loads and suggest steps to mediate any effects. Previous nutrient estimates had been made using the Nitrogen and Phosphorus Loading Assessment System (NPLAS) model (Storey and MacCaskill, 2007). NPLAS is not ideally suited to estimate nutrient loads for a large-scale urbanisation project such as Lake Rotokauri. Therefore, Diffuse Sources Limited monitored the Rotokauri Stream (which drains 60% of the catchment and is the main source of nutrients) over the course of one year in 2009. The monitoring was conducted to estimate the "current" nutrient loads and to establish a means to predict the future nutrient loads, before and after the urbanisation phase in the lake.

The project was carried out by Diffuse Source Limited in a number of phases to determine the progressive effects of urbanisation and to address ways in which nutrient runoff can be mitigated. Phase 1 looked at the current state of Lake Rotokauri and posed the question about whether altered nutrient loads would make any difference to the state of the lake. Diffuse Sources Limited (2008) concluded that even though the catchment nutrient loads were reduced it would have little or no effect on the lake's water quality unless lake restoration measures were also imposed. It was also stated that if an increase in the nutrient loads is observed, an increment in the growth of phytoplankton under the new artificial selection pressures and a shift in phytoplankton dominance could also take place. Phase 2 reported on different techniques that could be used to sample Rotokauri stream to obtain nutrient loads. It was concluded that the most cost-efficient way was a weekly sampling of the concentrations. This would allow a comprehensive examination of the relationship between flow and nutrient concentrations.

Phase 3 examined pre-urban monitoring of water quality of Rotokauri Stream to establish baseline estimates of nutrient loads, calculating mass flows and predicting effects of urbanisation on the relationship. This goal was achieved by the continuous monitoring of the flow rates (as concluded in Phase 2) and measuring nutrient concentrations in the collected water samples taken at weekly time intervals over the course of a year (2009). A flow recorder was installed by the National Institute of Water and Atmospheric Research Ltd (NIWA) at Exelby Road culvert (fig. 2.3). Water samples were manually collected at weekly intervals and were analysed for the TSS, nitrate (NO_3-N), nitrite (NO_2-N), ammonium (NH_4-N), total nitrogen (TN), dissolved reactive phosphorus (DRP), total dissolved phosphorus (TDP) and total phosphorus (TP) (Diffuse Sources Ltd Report, 2008).

Phase 4 reported on the future physical changes that will be made to the catchment to accommodate urbanisation and to predict changes to nutrient loads after the implementation of the Rotokauri Structure Plan (Diffuse Sources Limited, 2010). The revised Structure Plan (July 2010 version) reported that there would be three urban sub-catchments, with only one draining into the lake. The total area of the catchment was determined to be 1048 ha by Diffuse Sources Limited.

The Exelby Road Catchment will be formed by the surrounding Exelby, Burbush and Te Kowhai Roads. It will be ~263 ha in area including ~33 ha of the current Rotokauri catchment (Phase 4 Report, 2010). The catchment water will not drain into the lake but will be diverted into Te Kowhai Stream and Ohete

Stream. The Ruffle Road catchment, which will be bound by Ruffel Road, Te Kowhai Road, and the railway line, will drain into the Waikato River and would have an expected area of about 70 ha. The Rotokauri sub-catchment would be the only one that would drain into Lake Rotokauri. Although ~40 ha of the original Lake Rotokauri catchment area will be lost to other urban catchments, ~140 ha of land has been proposed that would be gained from other areas (Diffuse Sources Limited, 2010).

This thesis will provide the information that might be used towards Phase 5 of this project and will report on the water quality of Lake Rotokauri monitored for 2009. This information will assist with protecting the natural heritage and features of the lake and also provide mechanisms to improve the water quality by implementing best management practices during and after the urban development. Presently there is no reticulated stormwater system within the Rotokauri catchment. There are, however, several existing open drainage channels that have been provided to drain the farmland in the catchment area. Management practices could be used to increase the amount of impermeable surface and increase soakage and infiltration rates. The principal issues regarding the storm water are the quality and quantity of discharge to local water bodies, ground water levels, effects on peat soil and lack of fall across low lying areas. To achieve an effective and sustainable stormwater management the Hamilton City Council proposes that the storm water within the Rotokauri catchment discharges into three sub-catchments (Rotokauri Structure Plan, 2007). A network of floodway channels will therefore be required across the low lying areas to provide for an outlet for storm water which will assist with reducing groundwater levels, and reduce impacts on building foundations, the infrastructure in general and flooding in the proposed development area (Rotokauri Structure Plan, 2007). The Rotokauri stream will continue to serve as the outfall for the Horseshoe Lake, forming the principal storm water discharge from the Rotokauri catchment into Lake Rotokauri (Rotokauri Structure Plan, 2007).

Hamilton City Council also plans to minimise the flood risk by reticulating storm water within the low lying areas. Overland flow swales, wetlands (which ca be effective in removing NO_3) and conventional piped drains will collect storm water and discharge it to the floodways (Rotokauri Structure Plan, 2007). The floodways will be sized for storm water storage during storms, with controlled release to Lake Rotokauri. Conventional piped drains will also be used to discharge hill terrain flow to the collecting swales on the flat land. Wetland swales will collect and attenuate storm water flows to enable controlled natural soakage, and, will have a shallow grade that will minimise the risk of erosion (Rotokauri Structure Plan, 2007).

1.7 Aims and Objectives

The main focus of this study was to investigate whether the computer model DYRESM-CAEDYM was capable of simulating the water quality of Lake Rotokauri. This lake is being subjected to rapid catchment urbanisation under plans of the Hamilton City Council. While Lake Rotokauri is a shallow lake and is less readily subject to vertical stratification, the model DYRESM-CAEDYM was nevertheless considered to be a useful tool with which to capture the complex ecology and the occasional stratification in the lake.

DYRESM-CAEDYM was set up to reproduce the water quality of Lake Rotokauri for the year 2009. A major secondary objective was to run scenarios that would give an insight to the water quality of Lake Rotokauri during and after the urban development takes place. The simulations would also allow for an examination of whether phytoplankton communities would change in the lake with the urbanisation. The effort of setting up the input data for the model was considered worthwhile to contribute insights into the lake response to climate, its water budget and the nutrient levels in groundwater and surface water entering the lake.

2.0 Methods

The methodology used in this study comprised of field work, empirical calculations and computer modelling using the DYRESM-CAEDYM model.

2.1 Field Samples

Samples were collected monthly by University of Waikato staff in the year 2009. Samples were taken at 0 -3.8 m in Lake Rotokauri using the Ministry for the Environment protocol as discussed in Barnes et al. (2002). The samples were collected in two 1-litre van Dorn sampling bottles from the surface and from a single van Dorn sample at 3.8 m depth as no chlorophyll *a* samples were taken from the lower depth. The nutrient samples were sent directly to RJ Hill Laboratories, Hamilton, for analysis of nutrients and chlorophyll *a*.

On each sampling occasion Secchi depth was measured using a disk of 20 cm diameter with alternate black and white quadrants and a viewing chamber. Temperature and dissolved oxygen concentrations were recorded at 20 cm intervals through the water column with a Yellow Springs Instruments DO meter (Model I50). Samples of the surface flow at the Exelby Road site were taken weekly by Hamilton City Council (HCC) laboratory staff in 2009. These samples were collected in clean sample bottles provided by Hill Laboratories (Hamilton). Samples were analysed for total suspended solids (TSS), nitrite-N (NO_2-N), nitrate-N (NO_3-N), ammonium-N (NH_4-N), total nitrogen (TN), total dissolved phosphorus (TDP), total phosphorus (TP) and dissolved reactive phosphorus (DRP). The NH_4, NO_3, PO_4 concentrations for the months of January till March were not available and hence are not included in this research.

2.1.1 Bathymetry

A bathymetric survey of Lake Rotokauri was conducted on 13 April 2010. Depths in the lake were recorded using a BioFish (Garmin sonar, USA) with an attached Global Positioning System . The depths were recorded in a series of transects across the lake (Figure 3.1).

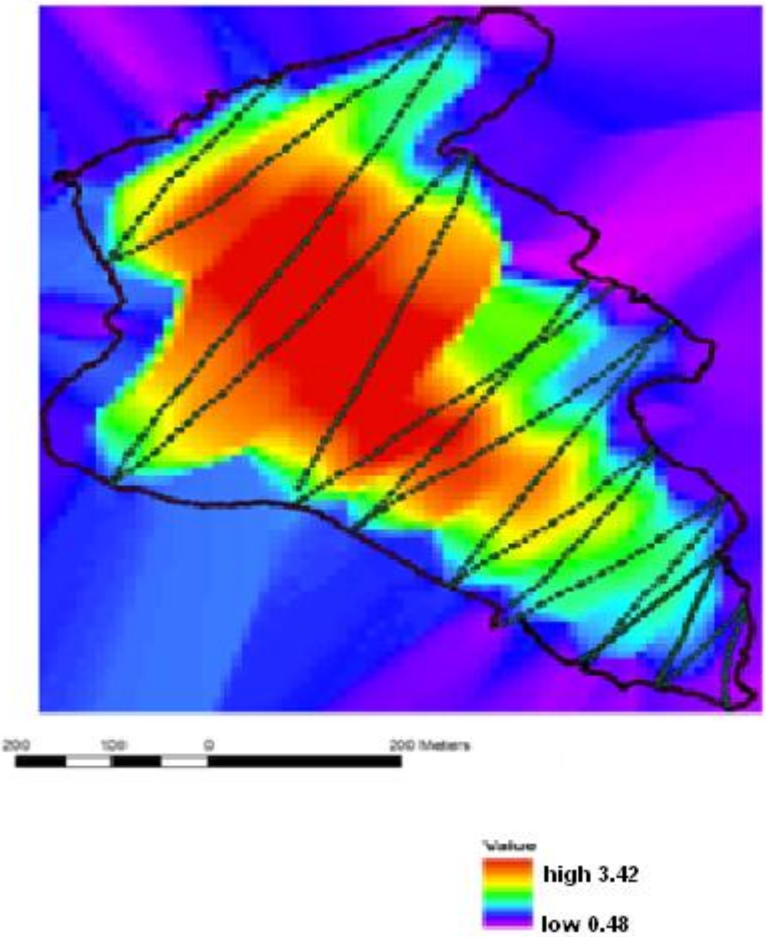


Figure 2.1: Bathymetry of Lake Rotokauri, showing lake outline (continuous dark line), transects (dashed line) and the different depths (coloured contour within the lake outline).

2.1.2 Groundwater Samples

Groundwater samples were taken on 16 November 2010 from nine boreholes out of the 16 identified boreholes installed within the Rotokauri catchment. The groundwater levels were recorded by BECA Consultants (Hamilton). Temperature and conductivity measurements of the groundwater were recorded with a YSI DO meter (Yellow Springs Instruments, Model 30, Ohio, USA) that was placed in the borehole. Samples were taken mid-way (i.e. half-way from the surface) with a 50 mL plastic tube attached to a 3m rope. (to suspend the plastic tube in the borehole to collect the water samples) Nutrient analyses were conducted by University of Waikato on groundwater samples, as described in Table 2.1.

Table 2.1: Methods with references relevant to the analysis of ground water nutrient samples.

Sample Type	Method Description	References
Total nitrogen/total phosphorus	Simultaneous acid-persulphate digestion	Ebina,Tsuyoshi & Shirai (1983)
Total oxidised nitrogen	Nitrate, nitrite colorimetric automated hydrazine reduction	EPA method 353.1 (APHA 1981)
Total phosphorus		EPA method 365.3 (APHA 1981)
Total oxidised nitrogen (dissolved)		EPA method 353.4 & SMWW/APHA Standard method 4500-NOx-E (APHA 1981)
Phosphate		EPA method 365.1 (APHA 1981)
Ammonium		EPA method 365.1(APHA 1981)
Nitrite		EPA method 365.1 (APHA 1981)
Nitrate	Given by total oxidised N – nitrite	

Groundwater boreholes were located at different proximities to Lake Rotokauri. Therefore an adjustment was applied to the incoming nutrient concentrations according to the distance of each borehole from the lake, i.e., providing a greater weighting to samples closer to the lake.

2.2 Laboratory Analysis

2.2.1 Phytoplankton Counting

Phytoplankton samples were collected monthly in 2009 from immediately below the water surface of Lake Rotokauri. The samples were preserved by adding

Lugol's iodine at room temperature (20 ⁰C). The retrieved sample container was slowly inverted ~12 times for about 30 seconds before it was sub-sampled for the purpose of phytoplankton identification and counts. A four mL sub-sample was then transferred into an Utermöhl chamber using a 10 mL pipette. Six mL of reverse osmosis (RO) water was also added to the chamber using a 10 mL pipette. The Utermöhl chamber was covered with a glass slide and the sample allowed settling overnight. A count of 100-120 planktonic units was then made to determine the dominant taxa in each of the monthly sub-samples, following University of Waikato protocol (Paul et al., 2007). Only algal/cyanobacterial cells with visible chlorophyll were counted. The cells were then counted at 600x magnification and on 1-4 transects. The whole chamber was also scanned for the less abundant taxa (i.e. taxa with less than five cells observed in a transect). The cell concentrations per taxa were calculated using:

$$N = Cf\left(\frac{A}{bAV}\right) \qquad (2)$$

Where N = number of algal cells per mL in the original water sample, C = total number of algal cells counted in all transects, A = total area of the transects (mm^2), b = number of transects counted, f = dilution factor and V = volume of the lake water that was settled (mL).

2.3 Calculations

2.3.1 Water Balance for Lake Rotokauri

A water balance was used to describe the flow of water in and out of Lake Rotokauri:

$$\Delta S = P + Q + Gr - E - O \qquad (3)$$

Where ΔS = change in volume ($m^3\ d^{-1}$), P = precipitation ($m^3\ d^{-1}$), Q = surface flow ($m^3\ d^{-1}$), G = groundwater ($m^3\ d^{-1}$), E = evaporation ($m^3\ d^{-1}$) and O = outflow ($m^3\ d^{-1}$).

For Lake Rotokauri the groundwater and the outflow were estimated using calculations explained below as there were no recorded data available for the year 2009.

The evaporative heat flux (Qlh) was calculated according to Fischer et al. (1979), with the condition that evaporative heat flux (Qlh) ≤0 so that there are no condensation effects:

$$Qlh = \left(\frac{0.622}{P} Cl.Pa.Le.Ua[ea - es(Ts)]\right) \qquad (4)$$

where P is the atmospheric pressure (hPa), Cl is the latent heat transfer co-efficient (=0.0013), Pa is the density of air (1.168 kg m^{-3}), Le is the latent heat of evaporation of water (2.453×10^{6} J kg^{-1}), Ua is the wind speed (m s^{-1}), ea is the atmospheric vapour pressure (hPa) and es is the saturated vapour pressure (hPa) at the water surface temperature Ts (°C). The latter was calculated by applying a linear equation deduced by plotting the atmospheric temperature to the field lake surface water temperature:

$$Ts = 1.158 \times Tair + 2.676 \qquad (5)$$

where Tair is air temperature. The atmospheric vapour pressure, *ea*, was calculated based on the relative humidity and air temperature (TVA 1972, equation C2):

$$ea = ({}^{RH}/_{100})exp[2.303\left(\left(a.\frac{qD}{qD+b}\right)\right) + c] \qquad (6)$$

where RH = relative humidity (%), qD = dry bulb air temperature (°C) and the coefficients for over-water calculations are a = 7.5, b = 237.3 and c = 0.7858.

The saturated vapour pressure, *es*, was calculated as per the Magnus-Tetjens formula (TVA 1972, equation 4.1):

$$es = exp[2.3026(\frac{7.5\times Ts}{Ts+237.3} + 0.7858)] \qquad (7)$$

The change in the mass of the lake surface layer (Δm) was calculated as:

$$\Delta m = \frac{-Qlh\times An}{Lv} \qquad (8)$$

where An = the surface area of the lake and Lv = latent heat of vaporisation for water (2.26 J kg^{-1}).

Therefore,

$$E(lake) = \Delta m \times 3600 \times 24/1000 \qquad (9)$$

Where E (lake) is measured in m^3 day^{-1}.

Three different land use types were recognized within the Lake Rotokauri catchment: pastoral, urban and forest. Part of the catchment also contains Horseshoe Lake. Land Cover Data Base2 (Jenkins and Vant 2007) was used to determine the different land use areas. In order to calculate the residual of groundwater, we first calculated the evaporation and evapotranspiration index, respectively, for both pastoral and forest land uses (Rutherford et al. 2008). A modified version of the Penman-Monteith model was used to estimate evaporation:

$$E = s\ (Rn\text{-}G)/[Pw\lambda(s + \gamma(1 + \frac{rs}{ra}))] \qquad (10)$$

where s = slope of the saturated vapour pressure vs. temperature line (~ 0.1 kPa $^{o}C^{-1}$), Rn = net radiation (MJ m^{-2} d^{-1}), G = soil heat flux (MJ m^{-2} d^{-1}), Pw = density of water (kg m^{-3})**,** λ = latent heat of vaporization (28.35648148 Ws kg^{-1}), γ = psychometric constant (0.067 kPa $^{o}C^{-1}$), rs = bulk canopy surface resistance (s m^{-1}) and ra = bulk aerodynamic resistance (s m^{-1}).

The soil heat flux (G) was calculated based on the assumption that the root zone depth, soil heat capacity and the soil temperature lags air temperature up to three days (Burman & Pochop 1994):

$$G = 0.3768[\check{T}d - \left(\check{T}d - 1 + \check{T}d - 2 + \check{T}d - 3\right)/3] \qquad (11)$$

Where *d* is the calendar day number, and Ť is the daily average air temperature. The evapotranspiration term (*Et*) is given by:

$$Et = (Pa.\frac{Ca(es-ea)}{ra}) \div (Pw\lambda(s + \gamma\left(1 + \frac{rs}{ra}\right))) \qquad (12)$$

Therefore:

$$Total\ Evaporation = E + Et \qquad (13)$$

To calculate the evaporation for urban land use the formula used was as it was assumed that there is 90% runoff from the land:

$$Eurban = 0.1 \times Area(urban) \times P \qquad (14)$$

The groundwater residual inflow of Lake Rotokauri was calculated using the formula:

$$Gr = P\ (catchment) - Total\ E - Q - E(lake) \quad (15)$$

Outflow:

As there were no records of outflow, values were calculated using the formula:

$$O = Gr + Q + P\ (lake) - E\ (lake) - \Delta S \quad (16)$$

Assumptions:

It is important to develop water and associated nutrient budgets as they quantify the nutrient concentrations that enter a lake from different land uses (Osmon, 2008). To match the frequency of the data required by the model few assumptions were made to balance the water budget for Lake Rotokauri. It was assumed that the total rainfall from the land use took an average 3 days to reach the lake. It was observed that the days when there was no rainfall (i.e. it was 0); the water balance was still experiencing large amounts of water being evaporated from the different land areas used within the Rotokauri catchment. To prevent this from happening, and, to stabilize the water budget it was assumed that if the evaporation was greater than the rainfall for any particular day, then evaporation would be equal to the rainfall for that particular day. Since no historic or current records of groundwater residual were available, the groundwater residual data was extracted from the equation:

Gr = Rainfall – Total Evaporation - Surface Flows

It was also observed that the days when the sum of total evaporation and surface flow was greater than the rainfall, a negative value for groundwater residual was generated. As the model does not recognize negative values it was assumed that if the residual value less than 0, then for those particular days the groundwater value would equal to 0 (zero). To minimize the flushing rate and high spiky peaks in the groundwater inflow, a 30 day running average was applied to the groundwater residual for Lake Rotokauri. This was done to enable the model to capture the phytoplankton succession dynamics for Lake Rotokauri. Outflow for Lake Rotokauri was established as the residual of the complete lake water balance due to non existence and non availability of any historic data. For some days when the total output was greater than the total input, the calculations arrived at showed the outflow values to be negative. This destabilized the water budget and the model. To obviate this destabilization, it was assumed that if on any particular day the outflow was less than zero, then, the outflow would be assumed to be equal to zero for those particular days. Despite the fact that the model does not recognize negative values, the negative values cannot be completely discarded from the water budget. Therefore to account for the water lost in the water budget, the lost water (i.e. the negative outflow values) were accounted for in the groundwater residual. These assumptions were made to balance the water budget for Lake Rotokauri and the model. It is recommended that the outflow and the groundwater flow along with the surface flow be monitored in future to avoid the input data being determined based on assumptions as it lowers the credibility of the model's output data.

2.4 Configuration Files of DYRESM-CAEDYM

Meteorological data were created for the simulation period for 2009. The data included daily short wave and long wave radiation, wind speed, rainfall, vapour pressure and air temperature measured at Ruakura meteorological station (Latitude -37.77657ºS and Longitude 175.3506 ºE) and accessed from the NIWA database (http://cliflo.niwa.co.nz/pls/niwp/doc/terms.html). The meteorological data was looped for the year 2010.

The morphometry file specified the height above mean sea level (279 m) and number of inflows (2) and outflows (1), and, points of elevation (401) and area values (401) that described the lake's morphometry, crest elevation (3.32 m) and outlet elevation (1.80 m).

Data in the initial profile file included the vertical profile of water temperature and salinity at different elevations on the first simulation day (21 January 2009).

The parameter file contained the mean albedo of water (0.10), critical wind speed (3.0 m s^{-1}), emissivity of the water surface (0.96), effective surface area coefficient (1.45 x 10^7), benthic boundary layer dissipation coefficient (7.5 x 10^{-6}) and vertical mixing coefficient (450).

The configuration file gave the simulation start date, length of the simulation and the time-step for simulations. Simulations for Lake Rotokauri were run over period of nearly two years starting from 21 January 2009 until 30 December 2010. Only one year of data were available so these data were repeated for the year 2010 in order to check the stability of the model and its ability to capture interannual dynamics. This file also contained a number of output selections (16), light extinction coefficient (0.4 m^{-1}) and minimum (0.2 m) and maximum layer thickness (0.5 m). Non-neutral atmospheric stability was switched on for the simulation run as specified in the configuration file.

This configuration file for CAEDYM included the biological variables, nutrient/chemistry variables and the miscellaneous variables. Two phytoplankton groups were simulated for Lake Rotokauri (cyanobacteria and freshwater diatoms).

The initialisation file layed out the initial conditions required for the CAEDYM simulation. This contains the initial water quality parameters for Lake Rotokauri.

2.5 Calibration

The model was calibrated against the field data for just under one year 2009 commencing 21 January 2009 till 30 December 2010, using comparisons of temperature, dissolved oxygen, nutrients (TP, TN, PO_4-P, NO_3-N, NH_4-N) and chl *a* concentrations. The parameter values used in CAEDYM for the calibration are given in the Table 2.2. After each calibration step, the root-mean-square error

(RMSE) value for each output variable was calculated. Pearson correlation coefficients were also calculated to evaluate the performance of the model for each of the variables. The stepwise manual calibration of the model continued until the RMSE and Pearson correlation coefficients were not markedly altered by successive calibrations.

Table 2.2: Biogeochemical parameter values used in the CAEDYM model.

Parameter	Unit	Value	
Dissolved Oxygen Parameters			
Temperature multiplier for sediment oxygen demand	Dimensionless	1.05	
Half-saturation constant for sediment oxygen demand	mg L^{-1}	0.6	
Sediment oxygen demand	g m^{-2} d^{-1}	2.5	
Nitrogen Parameters			
Denitrification rate co-efficient	d^{-1}	0.75	
Half-saturation constant for DO for denitrification	mg L^{-1}	2.0	
Temperature multiplier for nitrification	Dimensionless	1.08	
Nitrification rate-coefficients	d^{-1}	0.02	
Half saturation constant for DO for nitrification	mg L^{-1}	2.0	
Temperature multiplier for nitrification	Dimensionless	1.08	
Maximum potential sediment rate for NH_4	g m^{-2} d^{-1}	0.2	
Maximum potential sediment release rate for NO_3	g m^{-2} d^{-1}	-0.001	
Phosphorus Parameters			
Maximum potential sediment release rate for PO_4	g m^{-2} d^{-1}	0.02	
Phytoplankton Parameters		**Cyanobacteria**	**Diatoms**
Maximum potential growth	d^{-1}	0.6	1.5
Irradiance parameter non-photoinhibited growth	µmol m^{-2} s^{-1}	100	50
Half-saturation constant for phosphorus uptake	mg L^{-1}	0.002	0.005
Half-saturation constant for nitrogen uptake	mg L^{-1}	0.016	0.006
Minimum internal nitrogen concentration	mg N (chl *a*) $^{-1}$	5.0	5.0

Table 2.2 (cont.) Biogeochemical parameter values used in the CAEDYM model.

Maximum internal nitrogen concentration	mg N (chl *a*) $^{-1}$ d^{-1}	18.0	18.0
Maximum rate of nitrogen uptake	mg N (chl *a*) $^{-1}$ d^{-1}	2.0	3.85
Minimum internal phosphorus concentration	mg P (chl *a*) $^{-1}$	0.2	0.09
Maximum internal phosphorus concentration	mg P (chl *a*) $^{-1}$	1.0	2.5
Maximum rate of phosphorus uptake	mg P (chl *a*) $^{-1}$ d^{-1}	1.4	1.0
Temperature multiplier for growth limitation	Dimensionless	1.07	1.025
Standard temperature for growth	ºC	17	14
Optimum temperature for growth	ºC	21	19
Maximum temperature for growth	ºC	29	25.5
Respiration rate coefficient	d^{-1}	0.75	1.06
Settling velocity	m d^{-1}	$0.1x10^{-7}$	$-0.26x10^{-5}$

2.6 Scenario Runs

2.6.1 Scenario Justification

To predict the future nutrient loads three scenarios were generated. These scenarios were designed to range from minimal active management of nutrient loads through to ‘best practice’ to minimise losses of nutrients from the landscape.

Scenario I was designed to estimate the future nutrient loads to Lake Rotokauri for the rural land subjected to lifestyle blocks. Cooke & Parkyn (2005) used the Nitrogen and Phosphorus Loading Assessment System (NPLAS) model to estimate the TN and TP loads. The future loads from the untreated urbanised sub-catchment were estimated using the proposed 2005 Structure Plan (Storey &

Macaskill, 2007). The average annual areal yield for TN was set to be 8.9 kg ha^{-1} yr^{-1}, which is within the average values given from measurements by Williamson (1993). The specific yield for TP was set at 1.5 kg ha^{-1} yr^{-1} (Williamson 1993). However, it was noted by Diffuse Sources Limited (2010) that this value would be high for the Rotokauri urban sub-catchment and the yield of TP was reduced to an average of 1.2 kg ha^{-1} yr^{-1}. This value does not include the removal of nutrients from any specific treatment and management action.

Scenario II involved the development of urban areas with exposure of soils and sub-soils. Loads from these earthworks sites are difficult to predict (Williamson, 1993). Factors such as rainfall, wind and slope affect loads of particulates carrying nitrogen and phosphorus into the lake (Collins, 2003). Storey & Macaskill (2007) reported that the concentrations of nitrogen and phosphorus in the top-soil are c. 5 mg kg^{-1} and 1 mg kg^{-1}, respectively. On the other hand, sub-soils contain lesser amounts of nutrients (TN ~ 1 mg kg^{-1}, TP= ~ 0.2 mg kg^{-1}) (Diffuse Sources Limited, 2010). The sub-soil concentrations were used to calculate the nutrient loadings for the future urban development as this layer becomes the top-most layer of the soil.

Scenario III involved all rural subdivisions currently being converted to smaller lifestyle blocks with septic tanks installed. Cooke & Parkyn (2005) estimated that each household has three occupants and estimated that loading attributable to septic tanks from the lifestyle blocks of the different rural subdivisions could be 0.15 kg $person^{-1}$ yr^{-1} of phosphorus and 4 kg $person^{-1}yr^{-1}$ of nitrogen. Using this information the future loadings were estimated from septic tanks associated with lifestyle block establishment.

No stormwater reticulation system is currently in place for the urban land around Lake Rotokauri. As HCC has planned best management practices (BMPs) that will be implemented for the Rotokauri sub-catchment, treatment efficiencies were obtained for extended dry detention basins (grass-lined), constructed wetlands, biofiltration swales and floodways based on information from the National (USA) Best Management Practice database (USEPA, 2007). These values were used to generate the TP and TN concentrations for the urban treated water draining into Lake Rotokauri. The TN was predicted to be 2.65 kg ha^{-1} yr^{-1} and TP 0.27 kg ha^{-1} yr^{-1}.

2.6.2 Future Surface Flow

Change in land use affects the evapotranspiration regime as follows:

$$Q_{future} = (1+ ((Total\ E_{present} - Total\ E_{future}) / Total\ E_{future}) \times Q_{present} \quad (17)$$

where Q_{future} = the future surface flow (m^3 d^{-1}), total $E_{present}$= the present total catchment evaporation (m^3 d^{-1}), total E_{future} = total future catchment evaporation (m^3 d^{-1}), and $Q_{present}$ = the present surface flow (m^3 d^{-1}). For the period during the earthworks scenario it was assumed that the surface flow would be halfway between the present surface flow and the future fully developed surface flow.

2.6.3 Future nutrient concentrations

Present and future loads were calculated for each change of land use. The present load was given by:

$$L_{(x)} = V_{(x)} \times C_{(x)} \quad (18)$$

where x is the land use. The total present load was then calculated as:

$$Total\ L_{(present)} = L_{(urban)} + L_{(rural)} + L_{(forest)} \quad (19)$$

The new load was calculated by:

$$L_{(future)} = ((A_{(future\ x)} .Y_{(future\ x)} + A_{(future\ x)} .Y_{(future\ x)}) / (A_{(present\ x)} .Y_{(present\ x)} + A_{(present\ x)} .Y_{(present\ x)})) \times L_{present)} \quad (20)$$

where, L_{future} = the total future loading (kg ha^{-1} yr^{-1}), $A_{future\ x}$ = future area of the land use (ha), $A_{present\ x}$= area of the present land use (ha), $Y_{future\ x}$ = future yield from the land use (kg yr^{-1}), $Y_{present\ x}$= present yield from the land use (kg yr^{-1}) and $L_{present}$ = the total present loadings (kg ha^{-1} yr^{-1}). The new nutrient concentration for each scenario was therefore calculated by:

$$C_{(new)} = L_{(new)}/Q_{(new)} \quad (21)$$

Where, C_{future} = the nutrient concentration, Q_{future} = the future surface flow.

3.0 Results

The results section includes the measured lake results and the range of changes observed in the concentrations throughout the year. This section also includes the comparison of the observed results to the DYRESM-CAEDYM model. The last section shows the models simulation of the future scenarios and compares their results to the base model.

3.1 Measured Lake Results

Temperature in the Lake Rotokauri water column was warmest in February (surface = 24.5 ºC, bottom= 23.5 ºC) and was coldest in July (surface= 9.6 ºC, bottom= 9.7 ºC). Based on values of Lake Number and Wedderburn number, stratification was observed in February, April, September and December. The greatest difference between surface and bottom waters was in April (~3º C).

Concentrations of dissolved oxygen ranged from 2.33 to 11.64 mg L^{-1} over the entire sampling period. The lowest concentrations were at a depth of 3 m in February (2.33 mg L^{-1}) and the highest concentrations were in August (11.64 mg L^{-1}). Dissolved oxygen was nearly uniform through the water column in the months between May and August, but was lower in bottom waters than in surface waters in most of the other months. Total nitrogen (TN) concentrations ranged from 0.64 to 1.7 g m^{-3} and were highest in July, August and September. In July concentrations of TN in bottom waters were maximal (1.7 g m^{-3}). Concentrations of total phosphorus were relatively more variable between surface and bottom waters, ranging between 0.035 and 0.084 g m^{-3}. Higher TP concentrations generally occurred in summer months. The Secchi depth ranged from 0.7 to 2.32 m, with clarity of 2.32 m observed in September and only 0.7 m in March and December. Chlorophyll *a* concentrations ranged from 8.96 µg L^{-1} in September to 44.1 µg L^{-1} in May.

3.2 Model Validity

The one-dimensional assumption of DYRESM-CAEDYM should be applied when variations in the vertical dimension are more important and of greater relevance than those in the horizontal. This would imply that the forces (such as wind stress, surface cooling or inflows) that act to destabilize a water body do not create significant lateral or longitudinal variations in the system (Imberger & Patterson, 1981). These variations can be tested using Lake Number, a non-dimensional number which describes the disturbing influence of the wind relative to the stability of the stratification (Imerito, 2007). For all values of the Lake Number greater than one, the one-dimensional assumption holds (Imberger & Patterson, 1981), and, physically indicates that the stratification is strong and resists the disturbing wind influence (Imerito, 2007).

For calculation of the Lake Number (L_N) for Lake Rotokauri, a computer tool was used, known as "Lake Analyzer" (Imberger & Patterson, 1981). It was noted that Lake Rotokauri had Lake Number above the value of one only for some periods between February 2009 and April 2009. Another criterion to check the validity of the one-dimensional assumption is the Wedderburn Number. The value of the Wedderburn Number should also be greater than unity in order to satisfy the one-dimensional assumption. In the case of Lake Rotokauri, the Wedderburn Number (W) also satisfied the one-dimensional assumption only for periods from January 2009 to April 2009. The results would suggest that DYRESM-CAEDYM can be confidently applied to Lake Rotokauri only for the summer period from February to April of 2009. However, this does not negate the capability to apply the model, in effect, to act as a mass balance tool with which to track the nutrients in inflows and from bottom sediments, and to simulate their ecological effect.

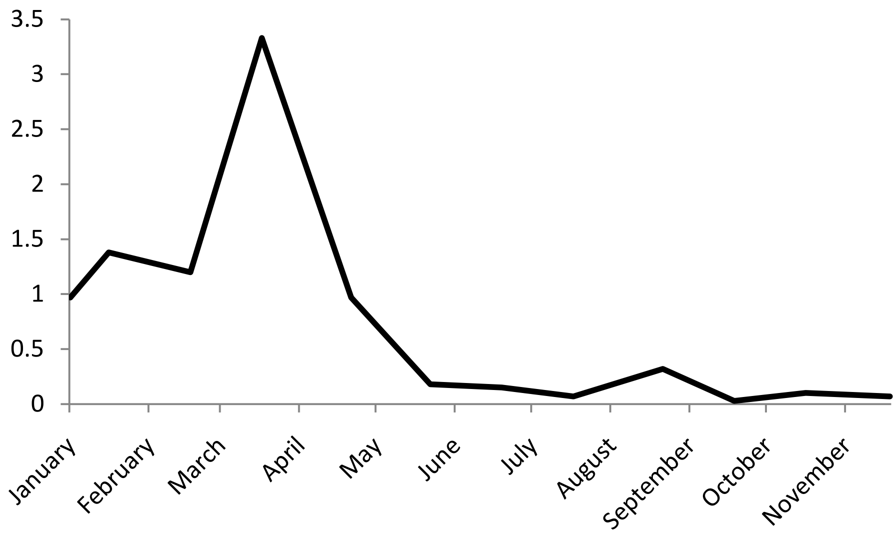


Figure 3.1 Lake Number for Lake Rotokauri for 2009.

3.3 Calibration Results

To check the validity of the calibrated data with the observed data, root-mean-square-error (RMSE) and Pearson correlation coefficient (R) values were calculated for all the observed variables for Lake Rotokauri for the year 2009 (21 March 2009 till 30 December 2009). The simulation was run for a second year with input data looped for the year 2010, to check if the model could reproduce the same trends for the second year. The simulations of temperature in both surface and bottom waters (represented by depth of 3m) of Lake Rotokauri generally showed good agreement with the observed data during the calibration period (Table 3.2). Furthermore, the model was able to capture the stratification that was observed in the months of February, April, September and December (Figure 3.2a, b). The model simulation also generally captured the dynamics of DO in both the surface and bottom waters (3 m), however the R value was quite low for DO. During summer, when the water column stratified, the observed bottom water DO concentrations showed fluctuations and occasionally increased in association with brief periods of mixing. Lower DO concentrations observed in bottom waters (3 m) for the months of September to November (2009) were not reproduced in the model (see Fig. 3.2d).

In terms of phytoplankton biomass the calibrated data showed a gradual increase in diatom biomass throughout the year and a gradual decline in cyanobacteria (Fig 3.2e) but tended to underestimate the observed data. The Pearson correlation coefficient between modelled and observed data for chl *a* concentrations was negative. This can be at least partially explained by the flushing phenomenon when the phytoplankton was being flushed out of the lake during periods of peak flow, before they could grow within the water column. In these circumstances the concentration of chlorophyll *a* and the composition of phytoplankton would be dictated by that in the inflows; in the initial flush period this might be quite high as phytoplankton biomass might be quite high in stagnant drains that are then flushed into the lake with critical levels of rainfall. During the main winter flow period chlorophyll *a* in drain inflows may be considerably lower as water would travel through drains relatively quickly with limited time for phytoplankton to grow.

The Pearson R values for both TP (Fig 3.2l, m) and TN (Fig 3.2f, g) concentrations were positive but lower than desirable, and the calibrated simulations tended to overestimate these concentrations. The under estimation of TN can partially be partially explained by the occurrence of denitrification and release of nitrogen into the atmosphere, which may have been underestimated in the model. The over-estimated value of TP can be partially explained by the same phenomenon of intense mixing. Due to light and convective heat exchanges, total phosphorus is released from the sediment and it moves into the water column. However, due to the high rates of wind exposure, the phosphorus circulates between sediments, water and phytoplankton, resulting in high internal loads of phosphorus in the lake. The model simulations generally showed good agreement with the observed data for both NO_3 (Fig 3.2j, k) and PO_4 (Fig 3.2n, o) (surface and bottom waters) and tended to capture the lower concentrations for both of these nutrients.

Table 3.1 Root-mean-square-error values (RMSE) and Pearson correlation coefficients (R) between modelled and observed data for the surface and bottom waters of Lake Rotokauri.

	Surface Waters		Bottom Waters	
	RMSE	Pearson R	RMSE	Pearson R
Temperature	1.515	0.97	1.602	0.96
DO	1.746	0.37	3.098	0.38
TP	0.024	0.37	0.022	0.38
PO_4	0.004	0.92	0.005	0.90
TN	0.370	0.24	0.417	0.18
NO_3	0.132	0.79	0.121	0.77
NH_4	0.128	0.42	0.135	0.15
Chl *a*	10.742	-0.15		

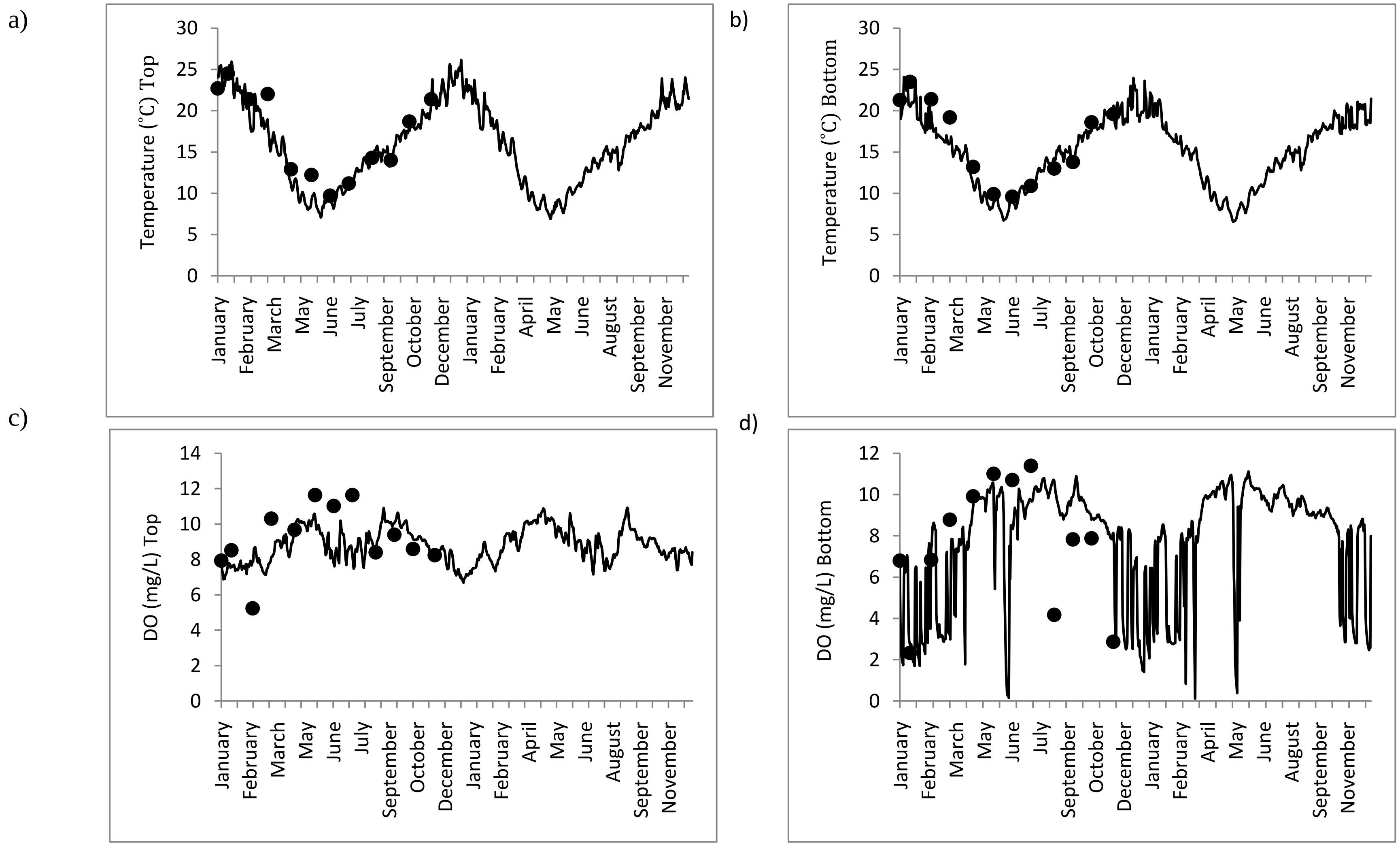


Figure 3.2 Observed temperature (a, b) and dissolved oxygen (c, d) in the surface and bottom waters (dots) and for the calibrated model values (black line).

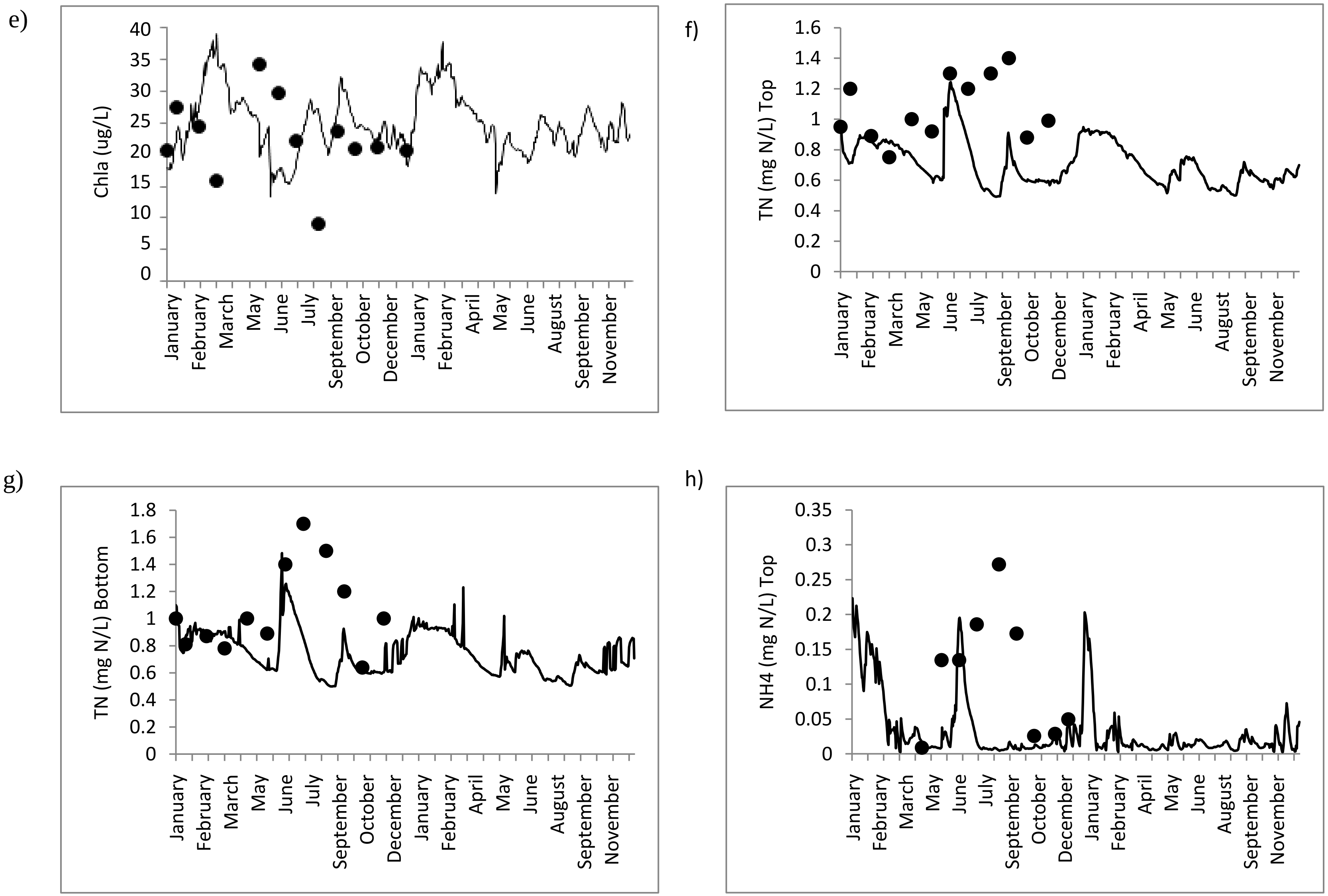


Figure 3.2 Observed chlorophyll *a* (surface only; (e)), TN (f, g) and NH_4 (h) (dots) and calibrated model simulation values (black line) in the surface and bottom waters of Lake Rotokauri for 2009-2010.

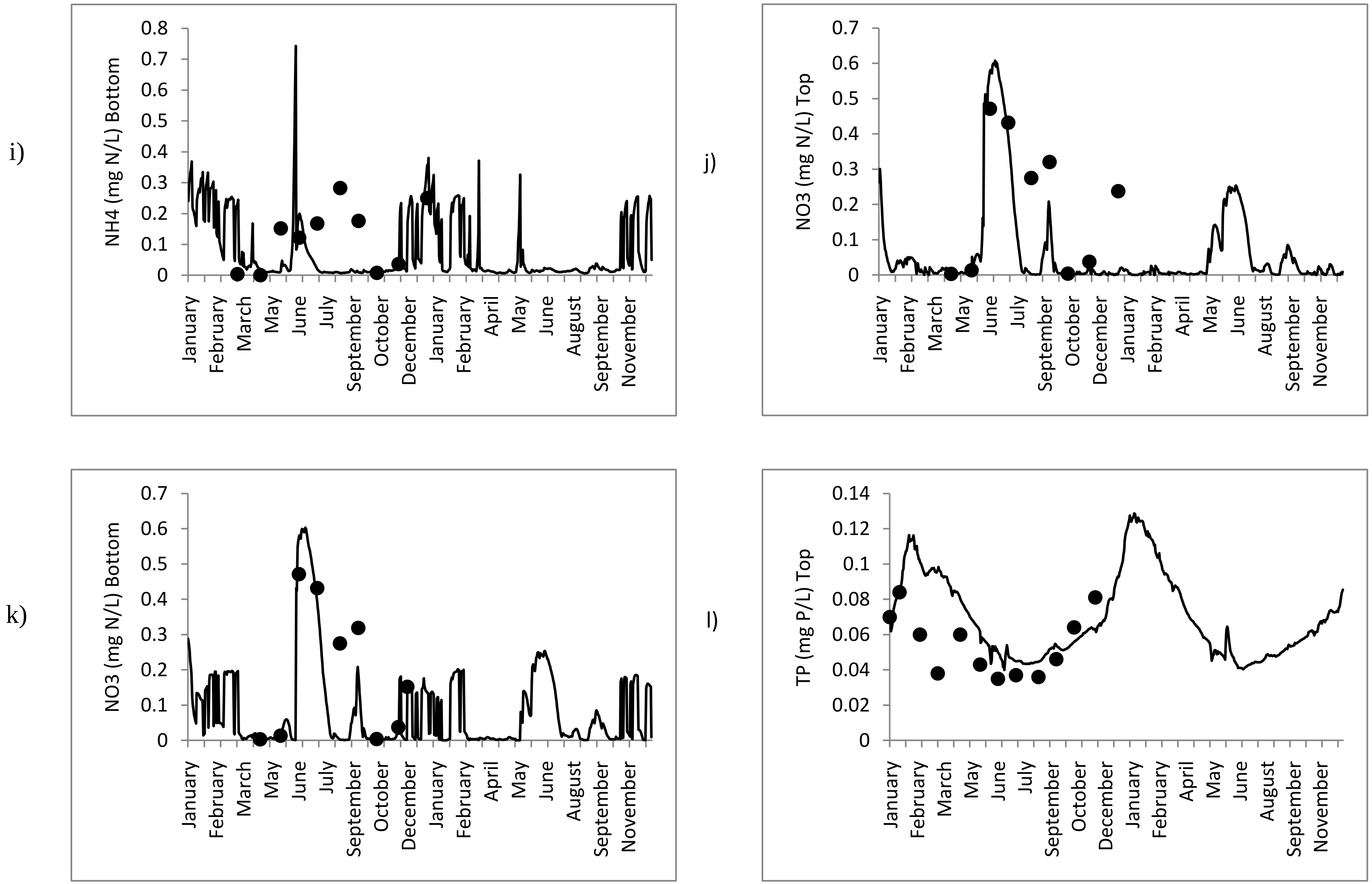
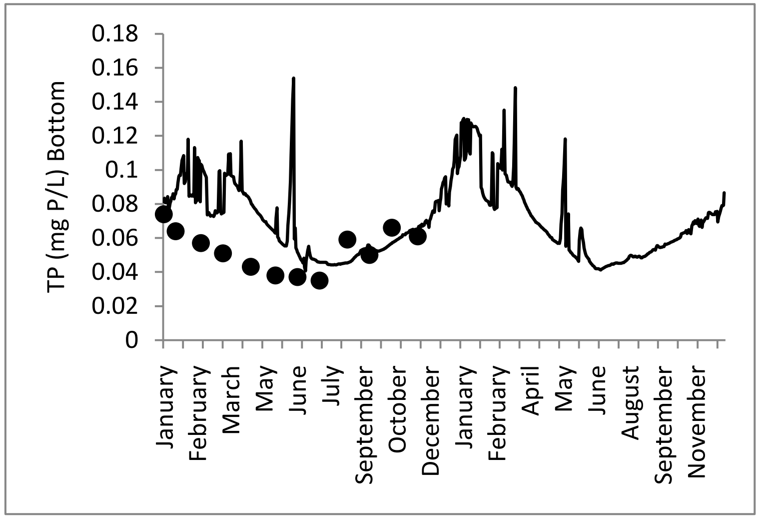
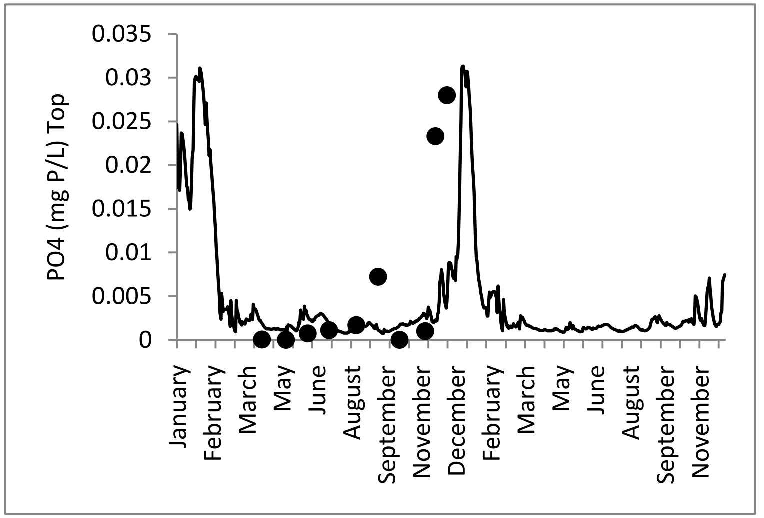


Figure 3.2 (cont.) Observed TP (l), NO_3 (j, k), and NH_4 (i) (dots) and the calibrated model values (black line) in the surface and bottom waters of Lake Rotokauri for 2009-2010.

m)



n)



o)

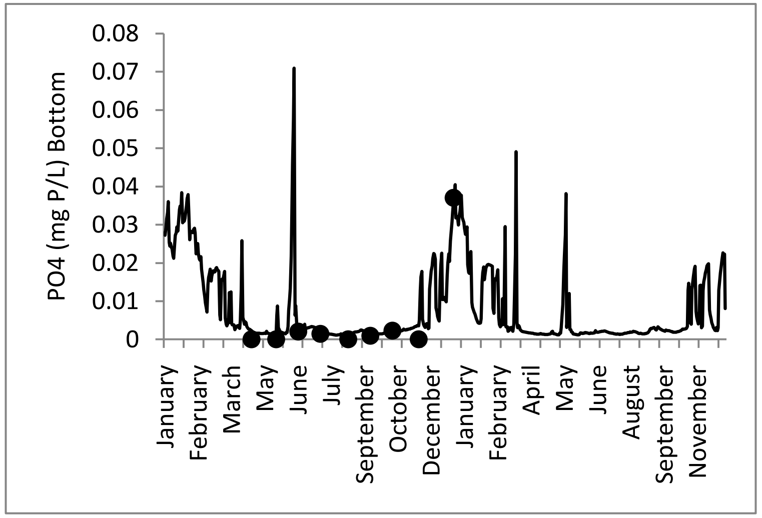


Figure 3.2 (cont.) Observed TP (m) and PO_4 (n, o) (dots) and the calibrated model values (black line) in the surface and bottom waters of Lake Rotokauri for 2009-2010.

3.4 Scenario Results

The data for the three scenarios were compared to the calibrated model using the 2009 data which was looped to provide a total simulation period of two years. The general seasonal pattern of variations in water quality variables for Lake Rotokauri was universal across all scenarios. However, the frequency of the fluctuations between the three scenarios and the base model varied. The untreated and the earthworks scenarios showed relatively little difference. The temperature regime for all the scenarios was similar to the base model.

3.4.1 Scenario I

For the untreated surface flow the DO concentrations at 0 m depth were generally lower in the months of late January-February, early March-June and September through to December (Figure 3.3a, b). The greatest difference in DO concentration was observed on 28 March between the base and scenario I (base-scenario I = 2.69 mg L^{-1}). At depth of 3 m, the DO concentrations showed large fluctuations throughout the year for the calibrated model and the best and worst cases (Figure 3.3b). The base DO concentrations, however, were observed to be greater than scenario I from mid-June to mid-July and from late July to early December (Figure 3.3a). The DO concentrations for the scenario I were observed to be greater than the calibrated case from 22 December to 26 December (Figure 3.4a). The base and scenario I showed deoxygenation on 4 July with concentrations of 0.14 mg $L^{-1}$and 0.12 mg L^{-1}, respectively (see Figure 3.3a, b).

Chlorophyll *a* concentrations of the calibrated model were greater than the best and worst cases of the untreated water for the whole simulation period. Chlorophyll *a* concentrations were greatest in the month of April in all cases. However, the timing of the high concentrations between the calibrated model and the scenarios showed differences of a few days (see Figure 3.3c). In comparison to the chl *a* concentration of the calibrated model, concentrations for both the best and worst cases were low in the months of March and July (Figure 3.3c).

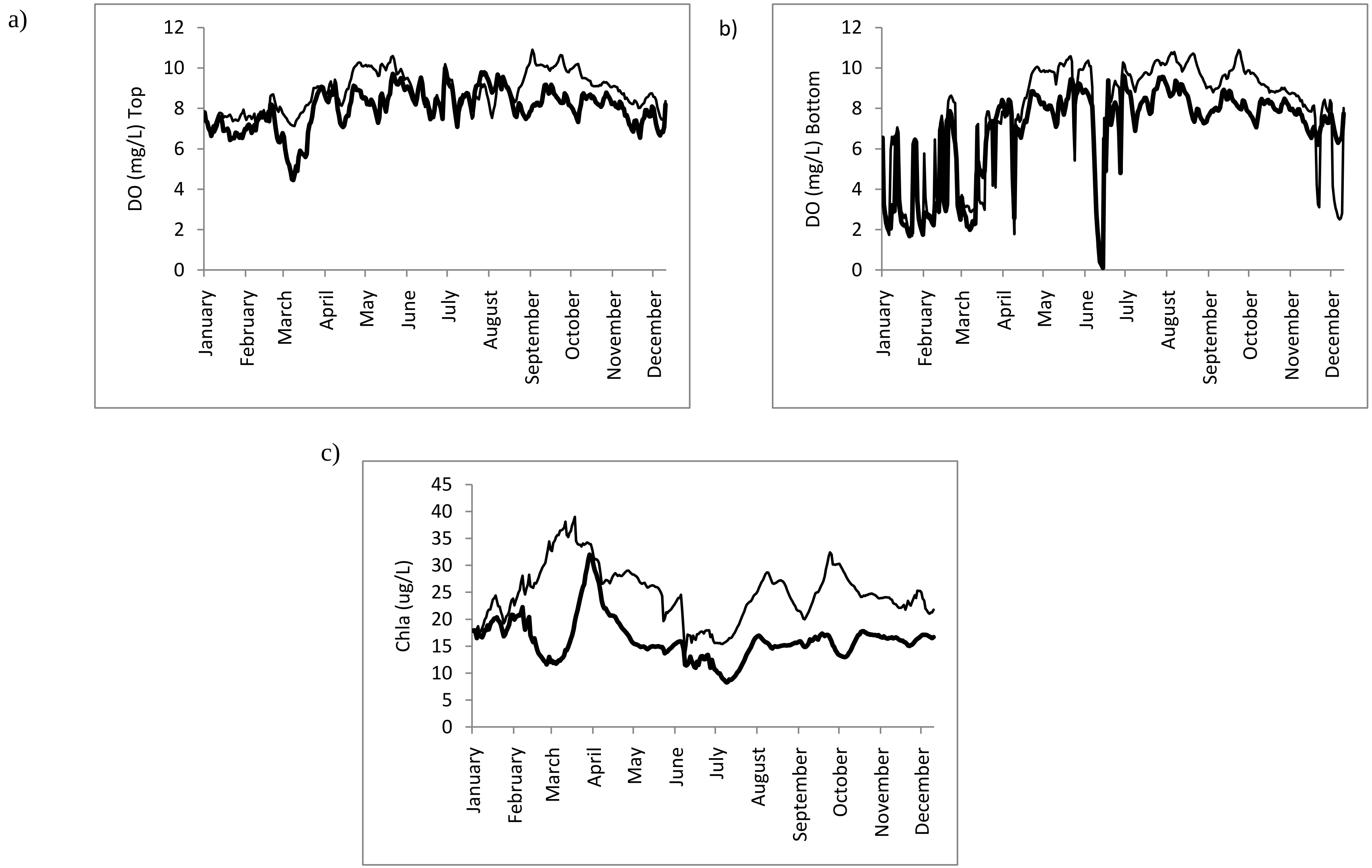
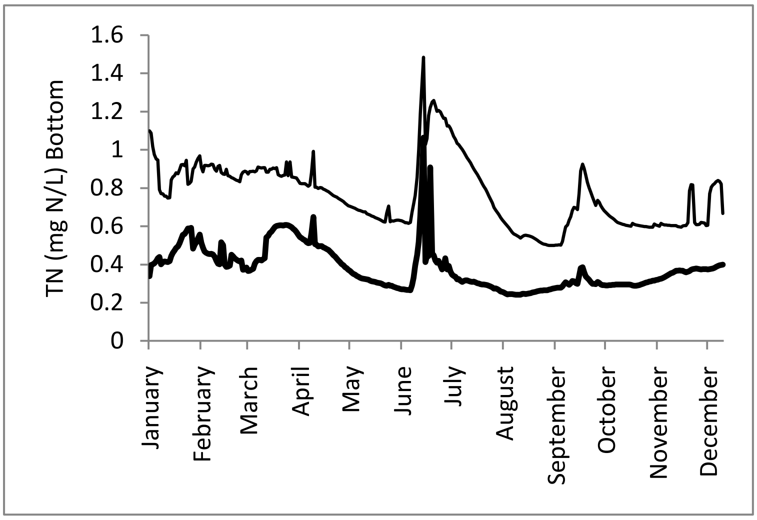


Figure 3.3: Simulations of DO (a, b) and chl *a* (c) for scenario I (bold black line) and the base model (light line).

d)



e)

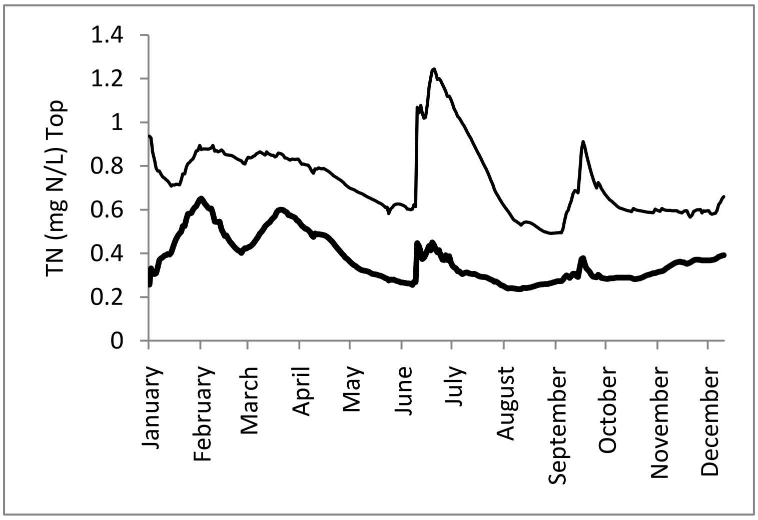


Figure 3.3 (cont.) Simulations for scenario I (dark black line) and base model (light line) for TN (d. e).

The PO_4 concentrations for both the best and worst case peaked (0.036 mg L^{-1}) in April (Figure 3.3h), whereas the highest concentration in the base model was in February at 0 m depth. The greatest difference (0.032 mg L^{-1}) between PO_4 concentrations between the two cases and the base model occurred at depth of 3 m April (Figure 3.3i). The TP (Figure 3.3f, g), TN (Figure 3.3d, e) and NO_3 (Figure 3.3l, m) concentrations of the calibrated model were greater than the concentrations of both the best and worst cases at 0 and 3 m depths over the duration of the simulation. However, NH_4 concentrations at 0 m for the two scenarios were greater than the base model NH_4 concentrations on 31 March (Figure 3.3j). At 3 m depth, the NH_4 concentrations of the base model were greater during early and late summer periods than the two scenarios (Figure 3.3k).

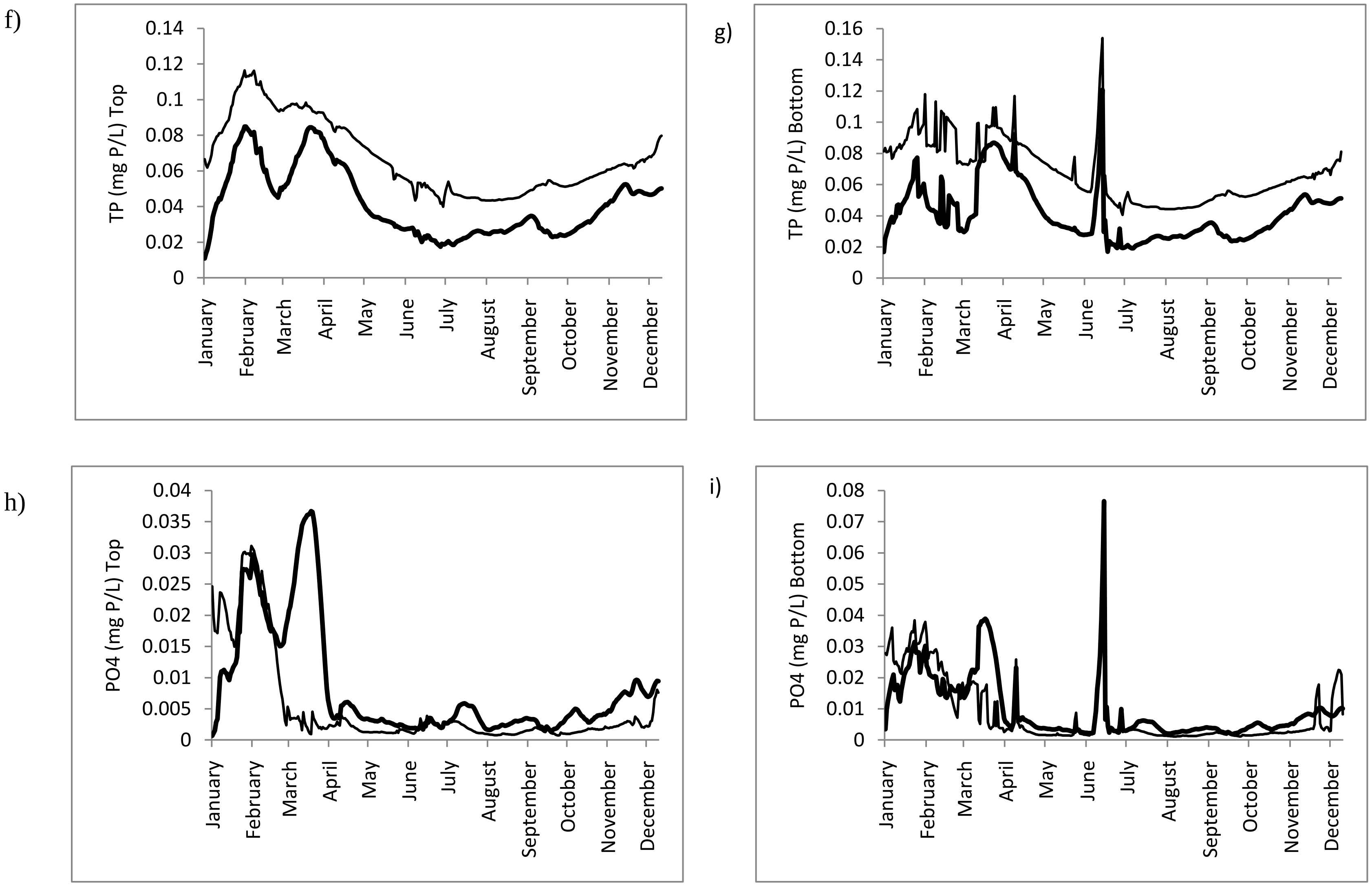


Figure 3.3 Simulations for scenario I (dark black line) and base model (light line) for TP (f, g) and PO_4 (h, i).

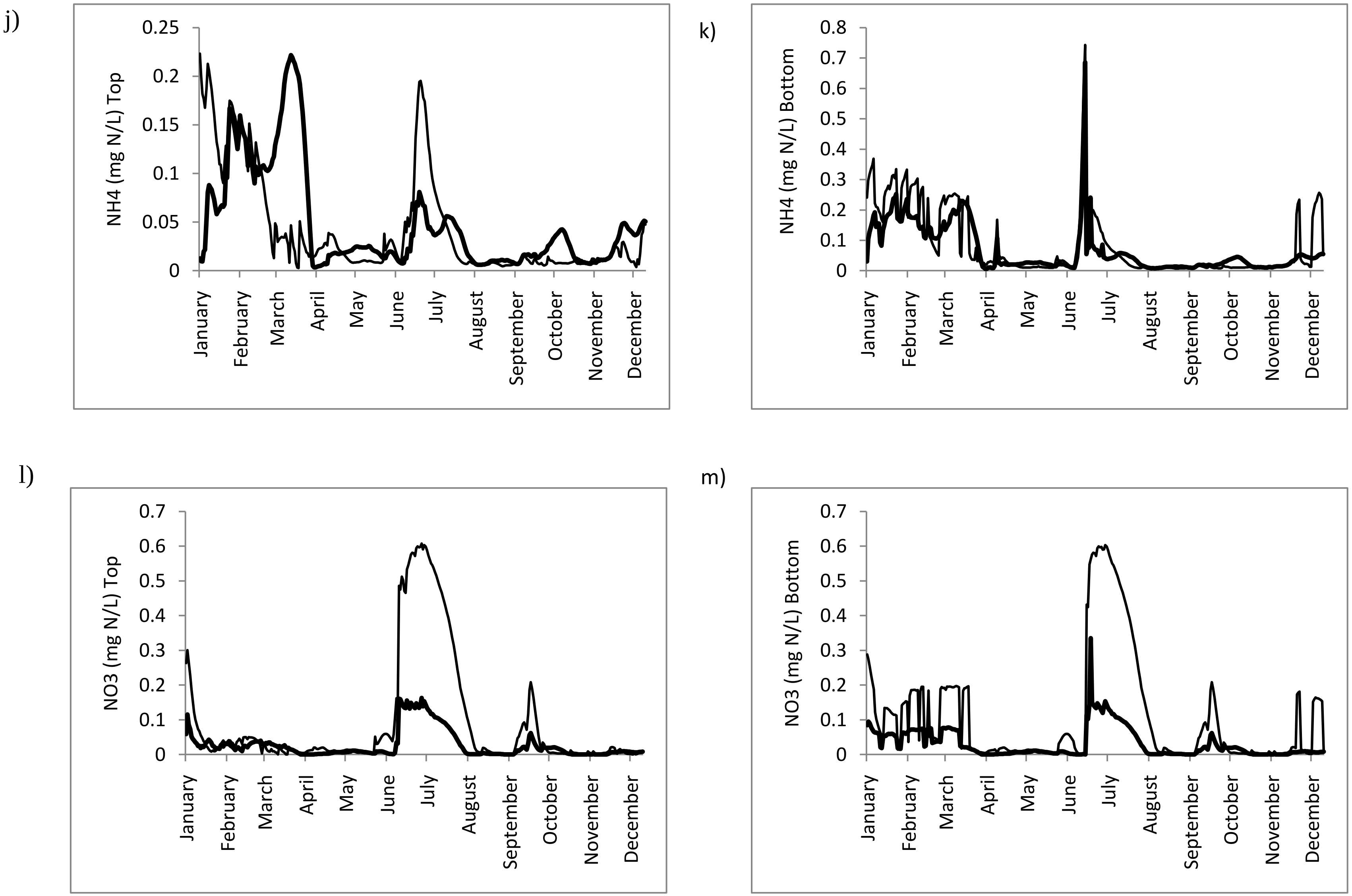
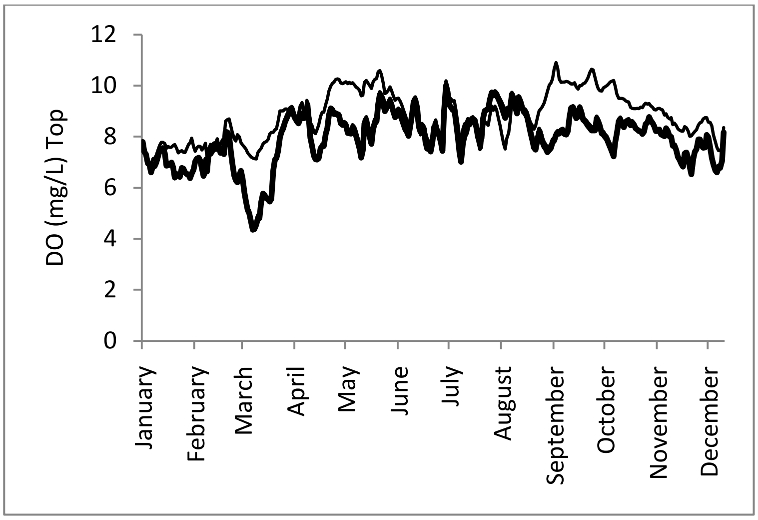


Figure 3.3 (cont.) Simulations for scenario I (dark black line) and base model (light line) for NO_3 (l, m) and NH_4 (j, k).

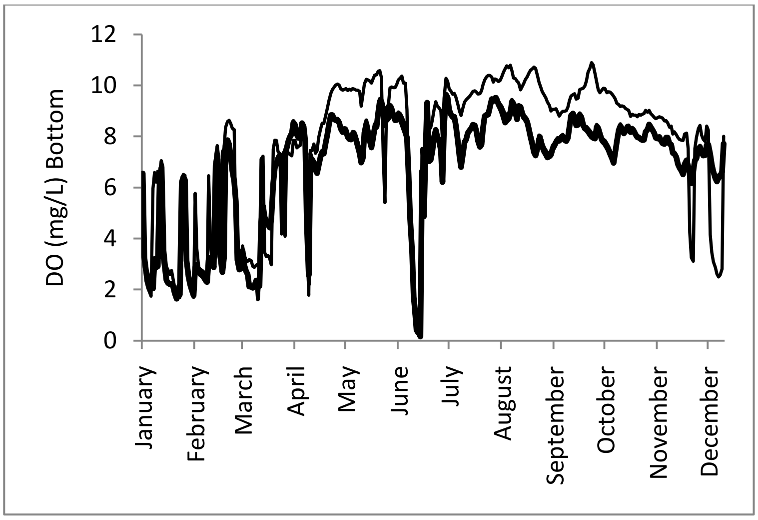
3.4.2 Scenario II: Discharge during construction

The greatest difference between base and scenario II DO concentrations was in the month of March at 0m depth (base DO-scenario II DO= 2.76 mg L^{-1}) (Figure 3.4a). At 3 m depth, Lake Rotokauri was predicted to be anoxic in July (0.18 mg L^{-1}) for scenario II (fig. 3.4b). The chl *a* concentrations for the base case were generally greater than scenario II for the simulation period. The maximum chl *a* concentration for the base case simulation was 38.9 µg L^{-1} while the maximum chl *a* concentration for scenario II was 31.7 µg L^{-1} (Figure 3.4c).

a)



b)



c)

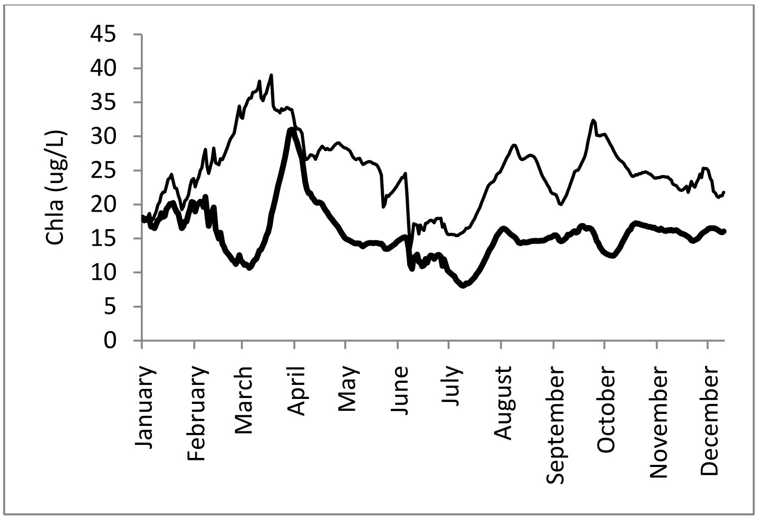
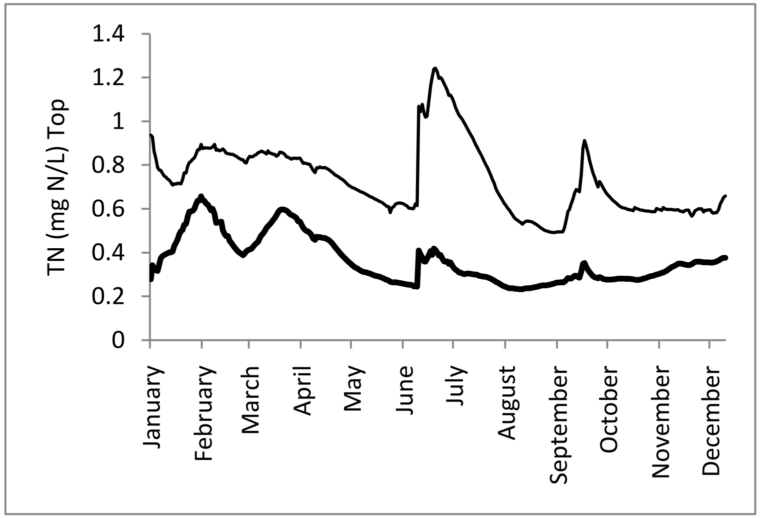


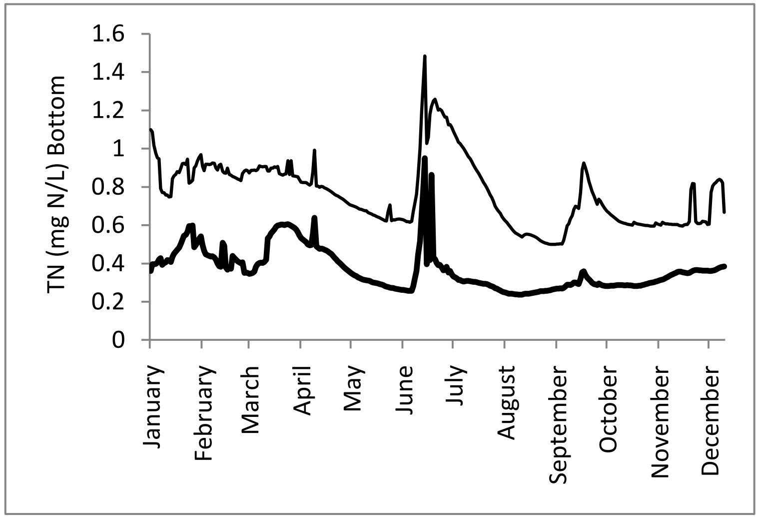
Figure 3.4 Simulations of DO (a, b) and chl *a* (c) for scenario II (bold black line) and the base model (light line).

The TN (Fig. 3.4d, e) and TP (Fig. 3.4f, g) concentrations for scenario II were substantially lower than the concentrations of the calibrated model at both 0 and 3 m depths. Concentrations of PO_4 for scenario II (Figure 3.4h) at 0 m depth were greater than those in the base case. However, at a depth of 3 m PO_4 in the base case reached a peak of 0.071 mg L^{-1} whereas in scenario II the peak was 0.067 mg L^{-1} (Figure 3.4i). At 3 m depth, the NO_3 (Figure 3.4 m) and the NH_4 (Figure 3.4k) concentrations of the base case were greater than scenario II.

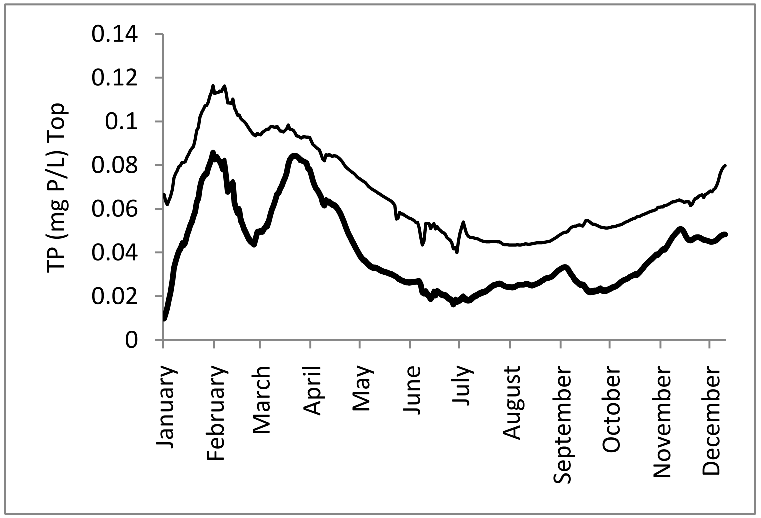
d)



e)



f)



g)

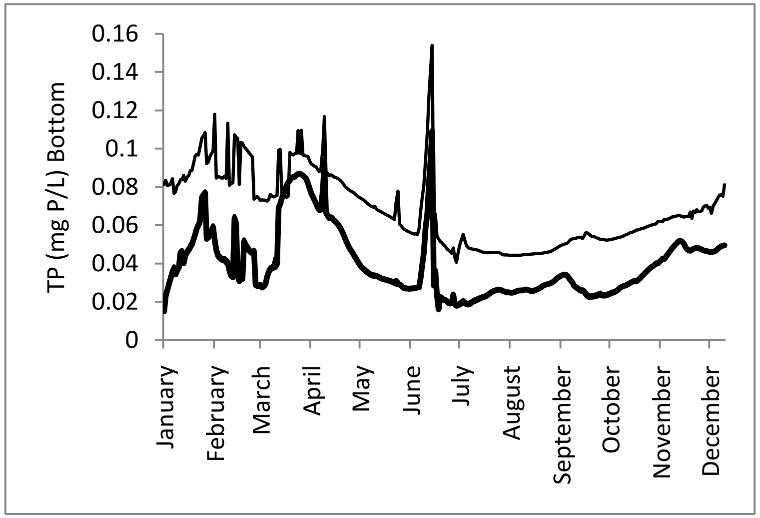


Figure 3.4 (cont.) Simulations for scenario II (dark black line) and base model (light line) for TN (d, e) and TP (f, g).

h)

i)

j)

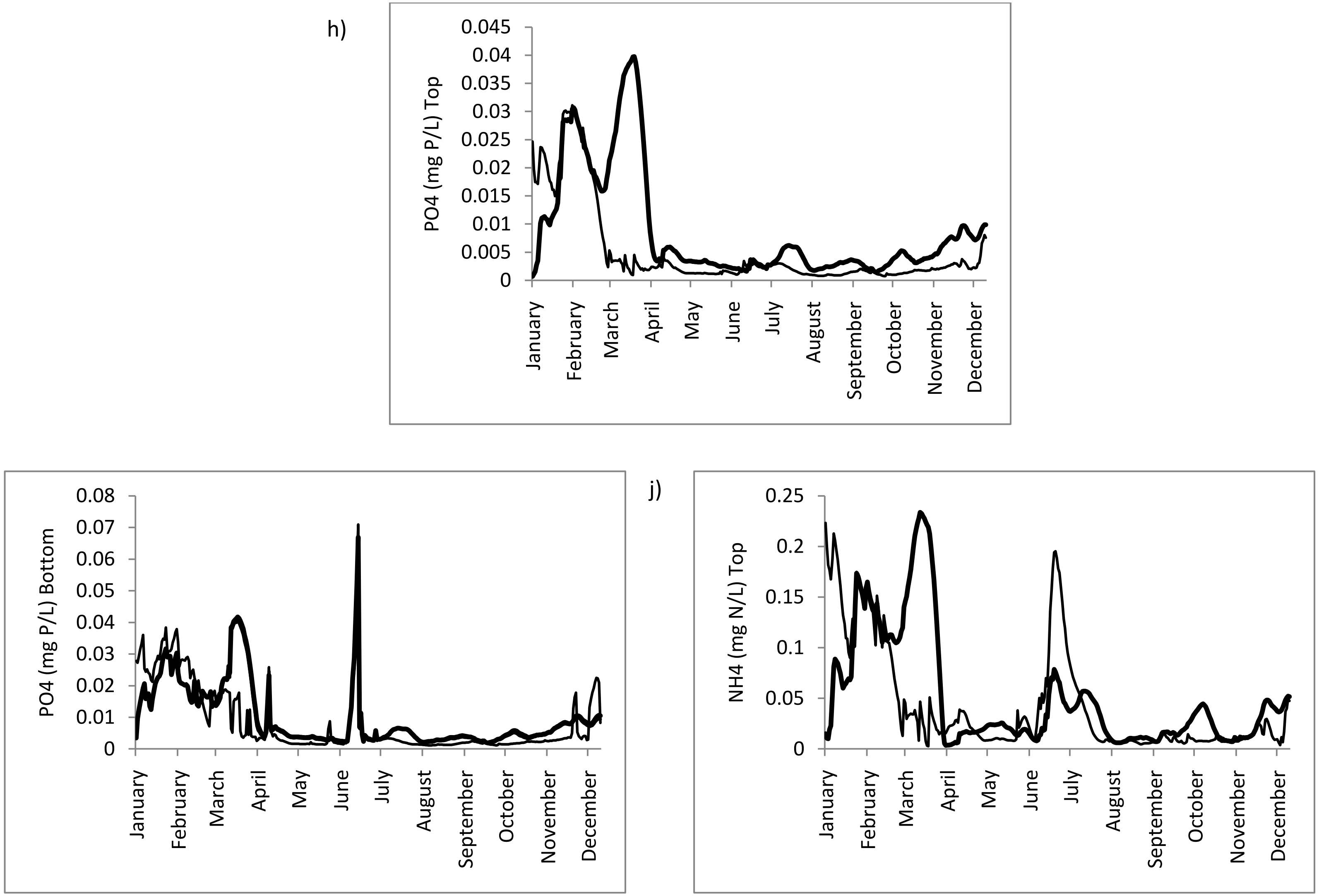
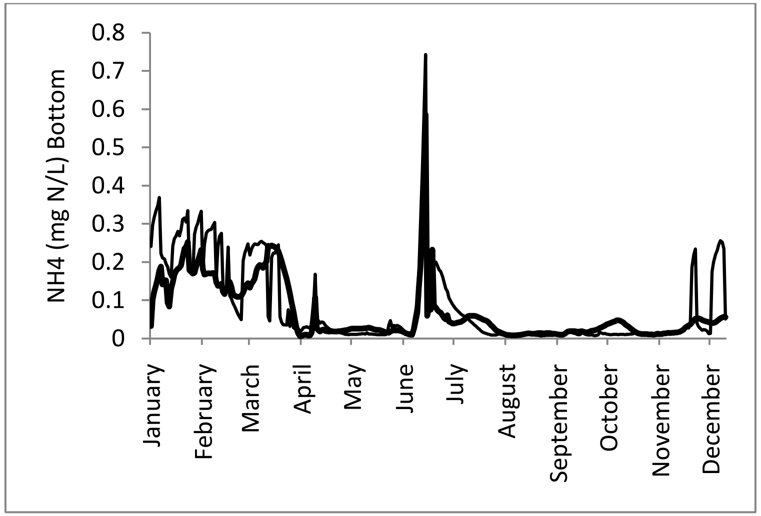
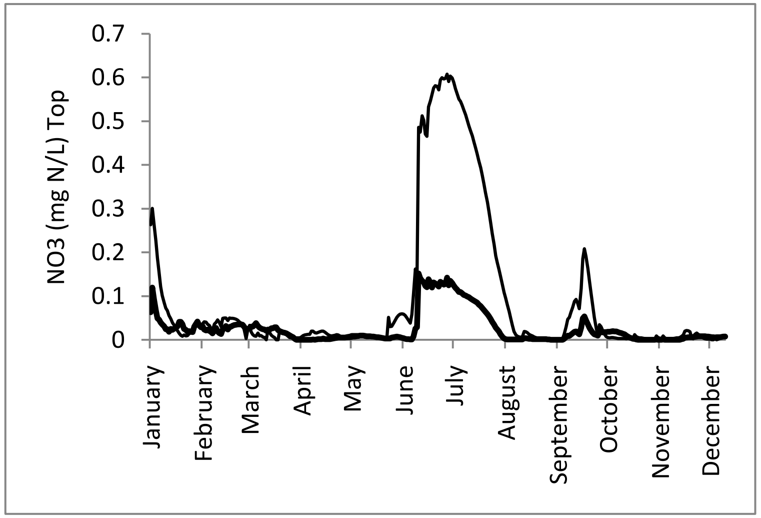


Figure 3.4 (cont.) Simulations for scenario II (dark black line) and base model (light line) for PO_4 (h, i) and NH_4 (j).

k)



l)



m)

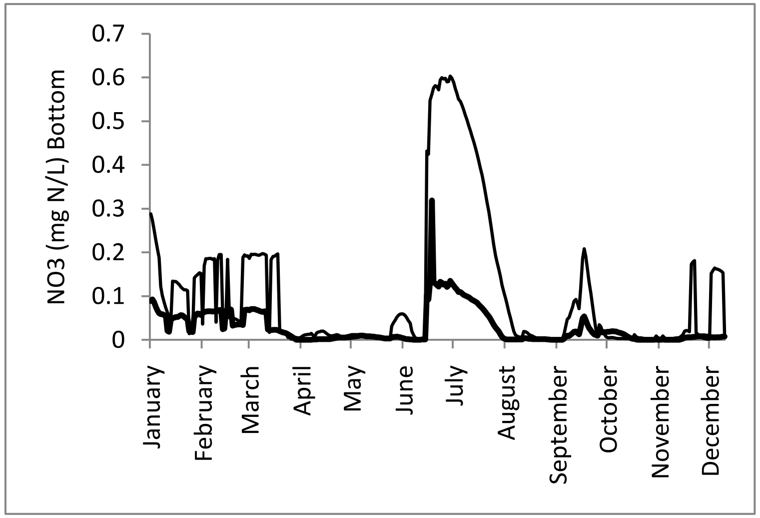
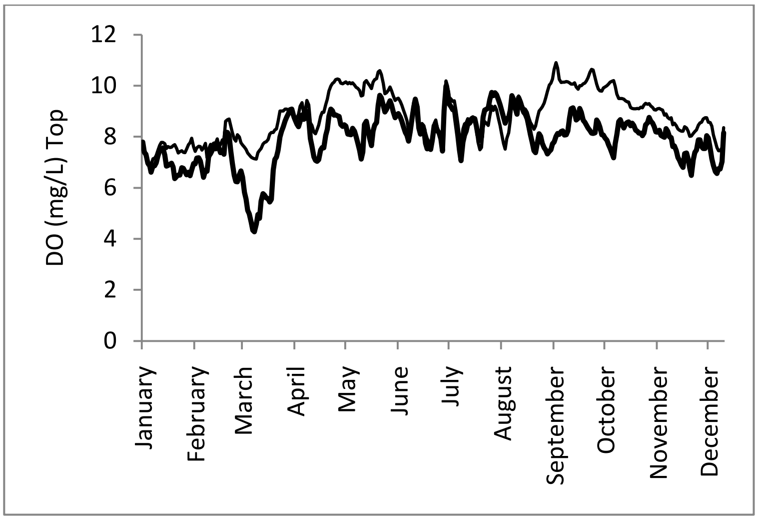


Figure 3.4 (cont.) Simulations for scenario II (dark black line) and base model (light line) for NO_3 (l, m) and NH_4 (k).

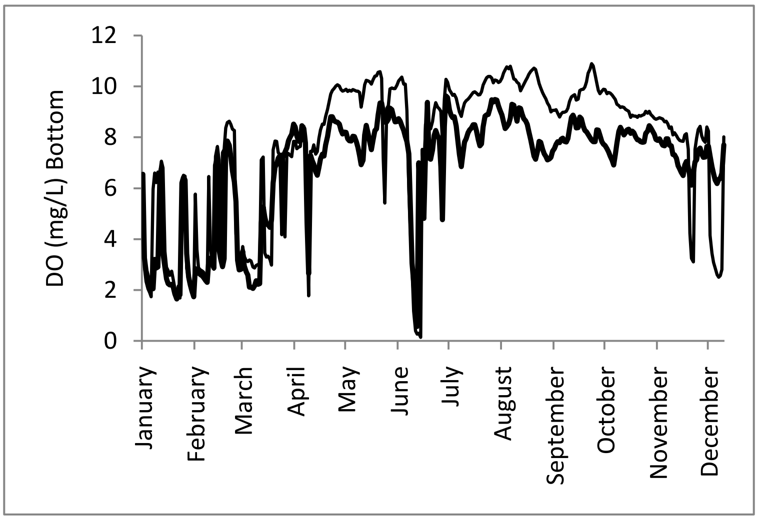
3.4.2 Scenario III: Best Management Practice in Future Fully Developed Sub-catchment

The greatest difference between the DO concentrations of the base model and the treated scenario at 0 m depth was 2.86 mg L^{-1} lower in the month of March (Figure 3.5a). Fluctuations of DO at a depth of 3 m for both the base model and the treated scenario were similar at the beginning of the year (Figure 3.5b), with the base model having greater DO for the months of May to November. Unlike the other two scenarios (scenario I and II), no anoxic environment was detected in the month of July for the treated scenario (Figure 3.5b). The chl *a* concentration in the treated scenario was lower than in the calibrated model. The maximum concentration of chl *a* for the treated scenario peaked at 30.8 µg L^{-1} in March (Figure 3.5c) compared to 38.9 µg L^{-1} observed in the base model.

a)



b)



c)

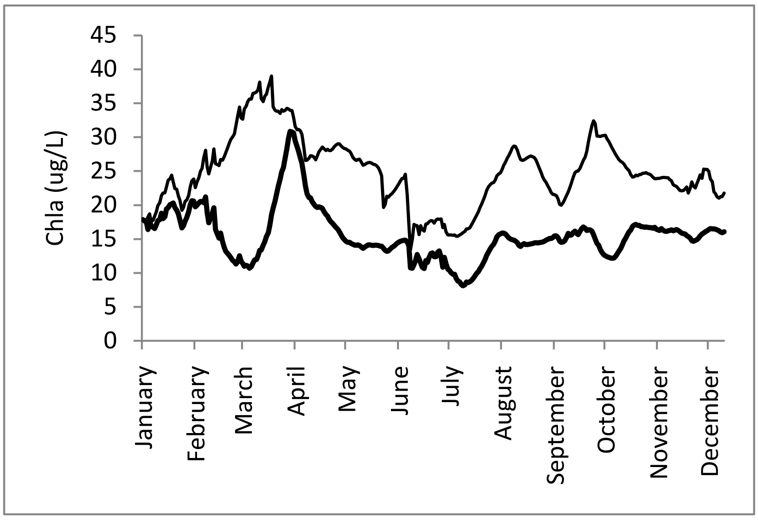


Figure 3.5 Simulations of DO (a, b) and chl *a* (c) for scenario III (bold black line) and the base model (light line).

Concentrations of TP (3.5f, g) and TN (3.5d, e) for the treated scenario were significantly lower than the concentrations in the base model. The PO_4 concentrations of the treated scenario at 0 m depth (Figure 3.5h) were generally greater than the base model, but with peaks of concentration occurring at the beginning of the year for both the treated scenario and the base model. For the treated scenario at 3 m depth PO_4 concentration was low for most of the year except for the month of March (see Figure 3.5i). The NH_4 concentration at 3 m depth was substantially lower in the treated scenario (Fig. 3.5 m) than the base model. However, at 0 m depth variations in NH_4 were greater (fig. 3.5l) than in the base model. Concentrations of NO_3 in the treated scenario (fig. 3.5j, k) were ~25% less than in the base model.

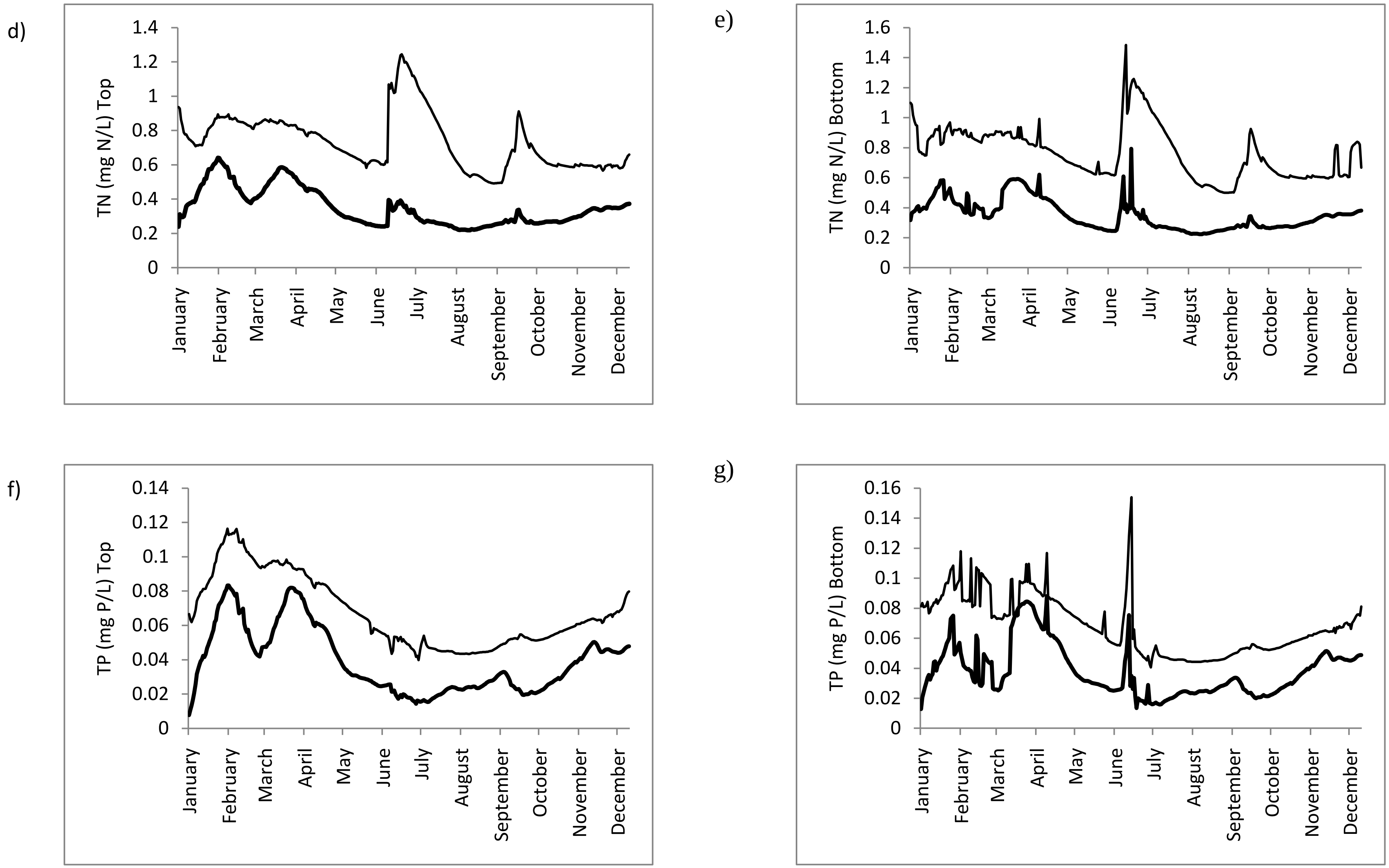
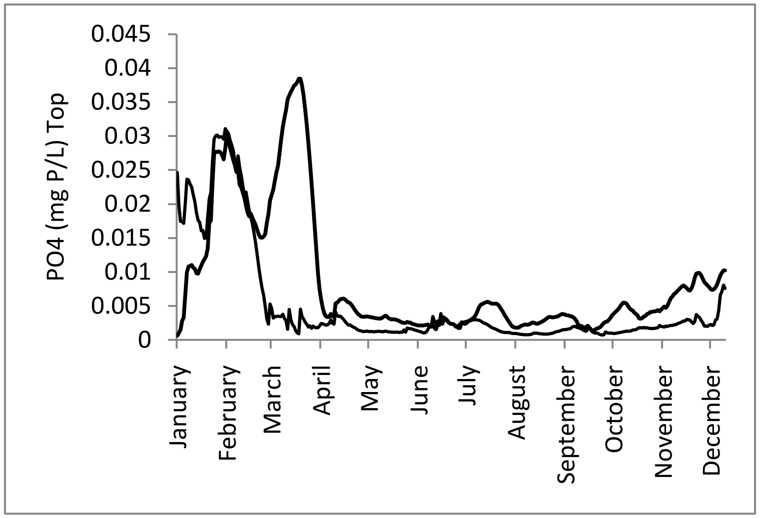
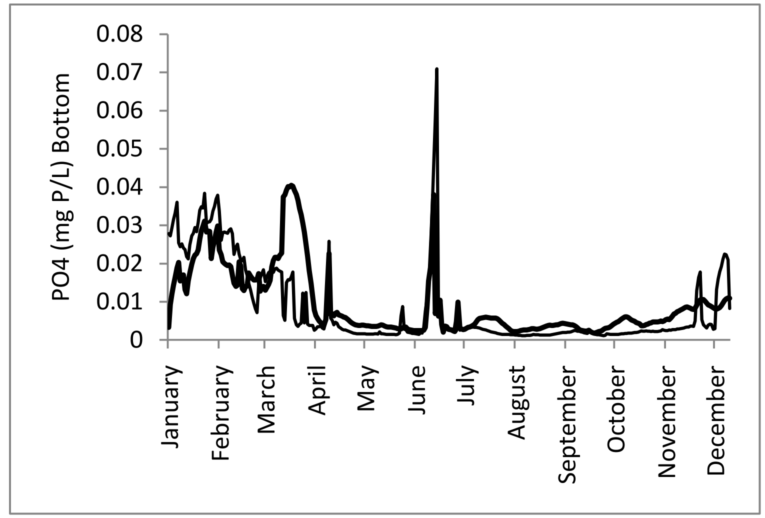


Figure 3.5 (cont.) Simulations for scenario III (dark black line) and base model (light line) for TP (f, g) and TN (d, e).

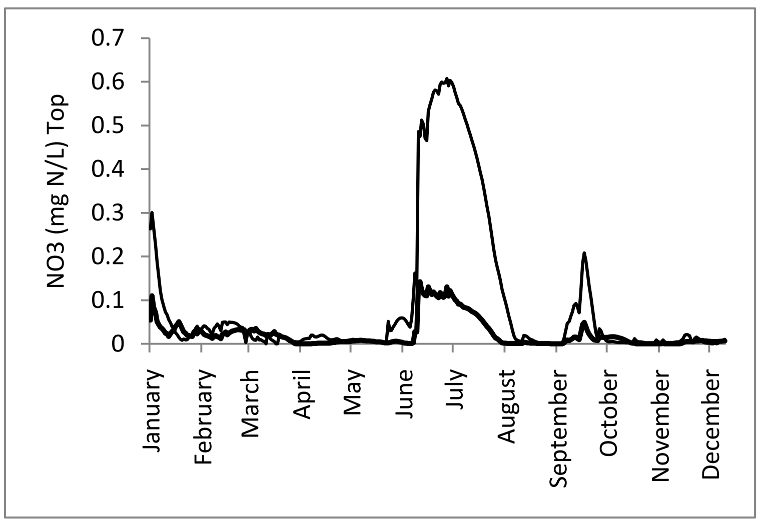
h)



i)



j)



k)

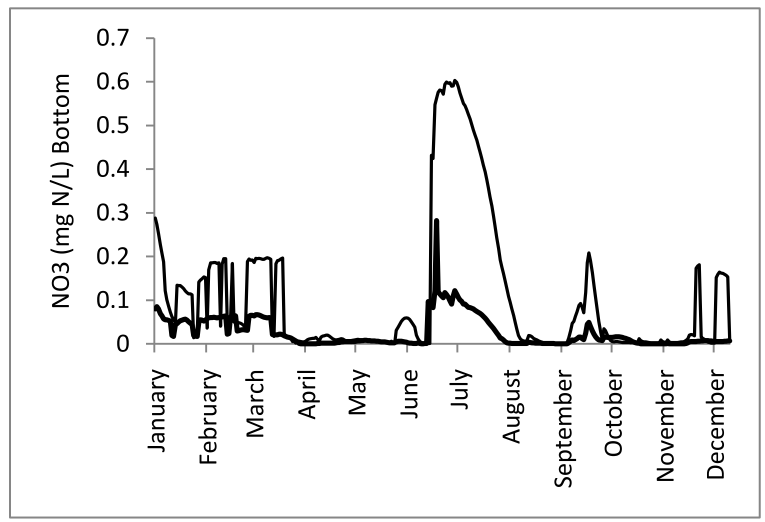
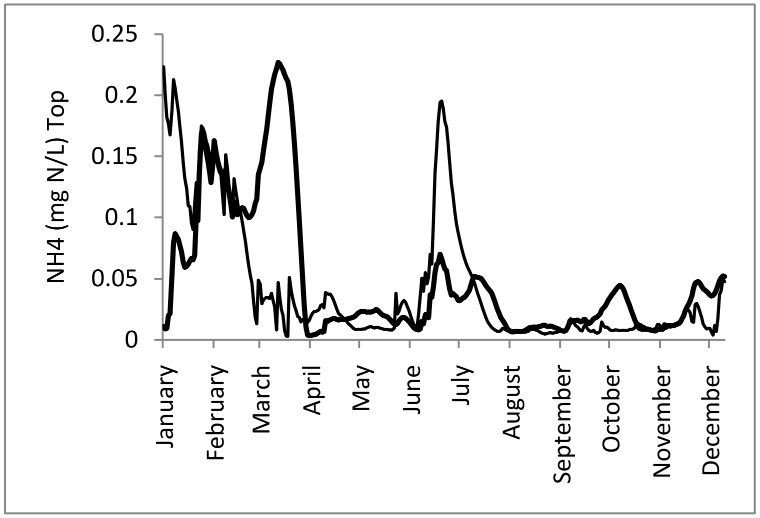


Figure 3.5 (cont.) Simulations for scenario III (dark black line) and base model (light line) for NO_3 (j, k) and PO_4 (h, i).

l)



m)

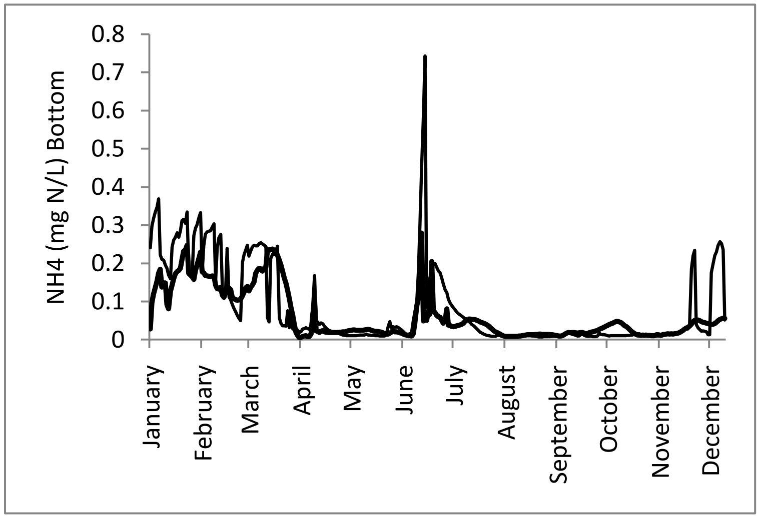


Figure 3.5 (cont.) Simulations for scenario III (dark black line) and base model (light line) for NH_4 top (l) and bottom (m) waters.

4.0 Discussion

The DYRESM-CAEDYM model was used in this study to provide water quality scenarios for Lake Rotokauri under different urban land uses and stormwater management regimes. The model was set up to be aligned with the Hamilton City Council (HCC) vision for the Rotokauri catchment of establishing a sustainable community that is integrated with the environment and the natural heritage of the city and the surrounding adjacent rural areas. The urban development plan seeks to enhance the significant unique and natural features of the area. The primary goal in the HCC urbanisation plan is to maintain and enhance the ecological value of Lake Rotokauri and develop best management practices to reduce trophic state of the lake from its current hypertrophic state.

The model provided reasonable agreements of nutrients and chlorophyll *a* that were observed in the lake in 2009. The model simulations of water column temperatures were slightly cooler than field observations during winter. According to Gal et al. (2003), the DYRESM model is sensitive to the longwave radiation inputs, which in the present study were calculated using daily average cloud cover, which may have contributed to some of the error in water temperature. The model simulations tended to underestimate chlorophyll concentrations. In the model simulations the annual succession of diatoms and cyanobacteria bloom was driven mostly by differences in growth relating to functions for light utilisation, mortality rate and temperature responses. As Lake Rotokauri is a shallow lake (maximum ~ 4m) there is a high water sediment surface area to which the lake water volume is exposed. Shallow lakes are generally subject to high rates of internal nutrient loading (Dokulil et al., 2000) due to resuspension of bottom sediments, including organic material, as well as disturbance of sediment porewaters(Vant, 1987). The model over-estimated the concentration of total phosphorus which may indicate higher simulated internal loading rates or lower simulated rates of uptake of phosphorus by the two phytoplankton groups, As the present model does not contain a dynamic description of sediment dynamics, the sediment phosphorus release rates likely did not capture some of the sediment-water dynamics that may be specific to shallow lakes. The total nitrogen on the other hand was under-estimated. Ammonium concentrations were also under-estimated in the model simulations, so processes specific to its dynamics (i.e. nitrification, ammonification) were likely to be linked to the under-estimation of total nitrogen. Even though ammonium accounts for a relatively small proportion of total nitrogen. There was closer agreement between the model simulations and observed data for concentrations of phosphate and nitrate.

A number of issues arose due to limited data availability for developing the water budget for Lake Rotokauri. It is also important to recognise the implications of assumptions made during the development of the water budget. Outflow from the lake, groundwater levels (to assist with a groundwater flow model) and surface flows should be monitored, at least indirectly (using water level-flow relationships) to refine the water budget for the lake. As noted in other studies (Burger et al., (2007), Gal et. al., (2003)) increased monitoring frequency would assist not only with estimation of the water budget, but also the determination of nutrient loads to the lake. Many drains, including Hamilton Zoo drain, are within the Lake Rotokauri catchment. Under the urban development strategy, however, the only surface drain to the lake will be Rotokauri Stream, and hence this will simplify the future optimal monitoring strategy.

It is important to acknowledge the effect of the necessary conceptual simplifications on the outcome of model scenarios for a system (Harris, 1994). The conceptual model for Lake Rotokauri did not include data for fish, zooplankton, bacteria and submerged macrophytes as no data on these variables were available as input to the model. Any major changes in nutrient loading to lakes can have a major impact on the structure of the phytoplankton community and affect aquatic food webs (Schindler, 2006). For example, increased grazing pressure by zooplankton will decrease phytoplankton populations at a given nutrient loading, and this will render a model less effective in simulating phytoplankton dynamics if it was calibrated without including zooplankton, under conditions of altered grazing pressure (Özkundakci, 2010).

Another limitation in the model application to Lake Rotokauri is the simplification of the phytoplankton taxa into two generic groups (diatoms and cyanobacteria). Different taxa of phytoplankton have an array of complex growth responses to environmental forcings and simplifying the model representation of phytoplankton to two groups is part of the usual conceptualisation process involved in modelling, but greatly simplifies the complex succession responses that occur in natural phytoplankton communities.

Another possible constraint in the conceptualisation was the exclusion of resuspension of suspended solids. Suspended solids inputs to the lake can result from erosion from urban runoff and agricultural land, industrial wastes and bank erosion, whilst internal sources can include bottom-feeding fish (such as carp) and sediment resuspension. High concentrations of suspended solids can reduce water quality by absorbing light and adsorbing or desorbing contaminants such as metals or organic compounds. Suspended sediments increase light attenuation resulting in warmer surface waters and, if there are substantial quantities of organic matter, can reduce levels of dissolved oxygen necessary for aquatic life. In shallow lakes sediment resuspension is often a direct result of surface wave activity (Vlag, 1992), thus, rates of resuspension in shallow waters can fluctuate rapidly according to changes in wind speed and direction (Bengtsson & HellstrÖm, 1992) and at longer time scales depending on coverage by benthic plants and presence of benthic fauna (Madsen et al., 2001).Sediment resuspension is an important contributing factor to the total phosphorus (TP) concentrations in lake water via internal P loading (Kristensen et al., 1992). It tends to decrease the TN: TP in the water column (since the ratio is lower in sediment surface due to denitrification) (Hamilton and Mitchell, 1997). In future studies it will be useful not only to quantify some of the variables that may influence sediment resuspension (e.g. benthic plants, benthivorous fish) but also to carry out periods of high-frequency monitoring that may allow for the frequency and magnitude of sediment resuspenesion events to be quantified.

Validation of a calibrated model against an observed data set is used to evaluate the performance of the model (Robson et al., 2008). The simulation period for Lake Rotokauri was only for duration of one year. To predict the future response to modifications in its catchment it is also important to evaluate past changes in catchment land use and lake water quality. It is known that in the past Rotokauri catchment was subjected to intense farming. This has led to a shift in the state of the lake from being dominated by macrophytes to a relatively turbid, phytoplankton-dominated state. Ideally it would be useful to run model simulations over this period to examine if the transition to a turbid state was able to be captured. In the absence of details of inflow data (flows and nutrient loads) and outflow, this exercise would be hypothetical, however, as nutrient loading is a primary causal factor for the transition to the turbid state.

Besides anthropogenic impacts, it is widely known that climate change will influence lake water temperature, nutrient loading to lakes, rates of organic matter degradation, and lake water chemistry generally (Jackson, 2011). Climatic changes are likely to result in deterioration of lake water quality due to a series of non-linear processes that have internal feedbacks leading to alternative regimes of degraded lake water quality (Scheffer and van Nes, 2007). Increased summer rainfall events and UV-B radiation will significantly decrease dissolved organic carbon concentrations altering biogeochemical cycles (IPCC, 2007).

As the Rotokauri catchment will be subjected to significant urban growth, concerns have been raised as to the effects of urbanisation and changes in nutrient loads in the already-hypertrophic lake. Three scenarios of urban development, land use practices and stormwater management were developed with to compare results with the calibrated model ('base' case) representing water quality of Lake Rotokauri in 2009. The scenarios considered the water quality that could evolve during and after urban development, and with a range of mitigation measures, from relatively modest treatment to best management practices to reduce nutrient loads and attenuate water flows to the lake. The accuracy of the future predictions made by the models is linked to the accuracy of the inflow data and nutrient loadings, so some caution should be applied in assessing the model simulation outcomes.

The nutrient loads predicted for future urban run-off were less than the nutrient yields from the present pastoral run-off. Scenario I related to the fully developed urban area and with no management strategy in place. This scenario produced increased loading, greater fluctuations in dissolved oxygen concentrations and higher nutrient concentrations than was simulated for the other two scenarios. A brief period of anoxia occurred in the bottom waters of Scenario I in the month of July. However, concentrations of total nitrogen and total phosphorus were lower than those of the base model. Concentrations of dissolved nutrients were more variable; PO_4 and NH_4 concentrations for Scenario I were mostly higher than the base case whilst NO_3 concentrations were generally lower. The shallowness of the lake makes the lake more responsive to variations in inputs from runoffs. Scenario II was an intermediate stage towards the fully developed urban catchment represented by Scenario III. Chlorophyll *a* concentrations for scenario 2 were lower than the base case but the bottom waters were still predicted to be anoxic (dissolved oxygen = 0.18 mg L^{-1}) in July for Scenario II.

Scenario III involved simulating water quality for the future land use envisaged by HCC and with best management practices implemented. These practices included dry-extended detention basins (grass-lined), constructed wetlands, biofiltration swales and floodways. At 3 m depth, fluctuations in DO concentration for both the base and scenario 3 were similar at the beginning of the simulated period, but there was greater DO for the months of May to November in the base case. Concentrations of NO_3-N for the treated scenario at 0 m depth were ~25% less than the base model. Hence, with best management practices implemented nutrient loads to the lake were reduced and the potential impacts of other confounding effects (for example, changing climate) can at least be partially mitigated.

The principal issues regarding the future runoff to Lake Rotokauri are the quality and quantity of the discharge, which are affected by complex interactions amongst groundwater levels, peat soils, and the relatively low-lying lake catchment. An effective and sustainable stormwater management plan is required to attenuate discharge into Lake Rotokauri. Hamilton City Council plans to minimise the flood risk by reticulating storm water within the low lying areas. Overland flow swales, wetlands (primarily for removing nitrate) and conventional piped drains will collect storm water and discharge it to the floodways (Rotokauri Structure Plan, 2007). The floodways will be sized for stormwater storage during rainfall events, following by controlled release to Lake Rotokauri. Conventional piped drains will also be used to discharge hill terrain flow to the collecting swales on the flat land. Wetland swales will collect and attenuate storm water flows to enable controlled natural soakage, and will have a shallow grade that will minimise the risk of erosion (Rotokauri Structure Plan, 2007). In New Zealand, stormwater treatment is often undertaken to achieve the 75% reduction in suspended solids (ARC, 2007). However, a 75% reduction in nutrients cannot be assumed for Rotokauri because dissolved nutrients are not as easily reduced as suspended solids, and particulate forms of nitrogen, in organic form, are also unlikely to be reduced to the same extent as suspended solids.

5.0 Conclusion

5.1 Summary

The objective of this study was to determine how urban development will affect the water quality of Lake Rotokauri. The one-dimensional model DYRESM-CAEDYM was used to simulate the physical, biological and -chemical processes within the water column of the lake, based on water budgets and nutrient loads that were varied with land-uses and climate.

The results presented in this thesis provide a quantitative evaluation of the effects of change in land use on the water quality of Lake Rotokauri, and therefore have important implications for its management. The model DYRESM-CAEDYM provided reasonable agreement with the field results and was able to capture the seasonal changes in phytoplankton chlorophyll *a* concentrations. Even with limited data sets and derivation of the water balance with several variables only able to be estimated indirectly, the model highlighted the contribution of nutrients via inflows from different land-uses within the Rotokauri Catchment and indicated the importance of quality of both the residual water in drains during low-flow periods and the high rate of water flushing through the lake in high-flow periods.

To depict the future water quality of Lake Rotokauri when subjected to urbanisation, three scenarios were developed which involved simulations with altered nutrient loads as inputs to DYRESM-CAEDYM. The future scenarios were compared with the calibrated model which represented a 'base' or present case of water quality. The scenarios considered the water quality that could evolve during and after urban development, and with a range of mitigation measures, from relatively modest treatment to best management practices to reduce nutrient loads and attenuate water flows to the lake. The predicted nutrient loads contributed from future urban run-off was less than the nutrient loads currently contributed from pastoral run-off. The model underlined that nutrient loading from the future untreated water (Scenario I) would result in poorest water quality of Lake Rotokauri, closely followed by Scenario

II which examined the water quality during the construction phase (Scenario II). Scenario III (treated water based on best management practices) was most effective in improving the water quality of Lake Rotokauri. Scenario I was set up for the untreated discharges from all the future land uses of the Rotokauri-sub catchment into Lake Rotokauri. Scenario II was the intermediate stage towards scenario III. Rainfall, wind and slope affect the loads of matter carrying nitrogen and phosphorus into the lake, making it difficult to predict nutrient loads with precision for this scenario. Scenario III involved simulating water quality from all future land use changes and best management practices implemented. These practices included dry-extended detention basins (grass-lined), constructed wetlands, biofiltration swales and floodways. Hence, with best management practices in place the loading of nutrients from the sub-catchment could be reduced without any other confounding effects (climatic effects, non-source discharges). However, extensive testing of model performance as well as understanding of diagenetic drivers associated with bottom sediments could logically improve the accuracy of the simulations.

5.2 Recommendations

Calibrations were performed on a very limited input data set, with most data collected monthly from January 2009 to December 2009. The model input data was formulated for daily simulations using calculation of missing variables with mathematical relationships (such as for equations for the water budget equation and evaporation. Regression and interpolation relationships were also used to fill missing values for days without observed data. A second year of simulation, without any alterations in input data from the first year and without comparisons with field data, was used to verify the repeatability and stability of the model simulations. Model simulation results could be improved with more regular and targeted monitoring of key water quality variables that influence the model result, particularly high-frequency inputs of water and nutrient loads. A consistent monitoring plan would ensure that effects resulting from management actions are measurable, and would produce a strong basis for further research and more accurate and valid model results. Restoration plans should also be explored and built into the management plan and these could include the effects of biomanipulation. Management plans should also consider any effects of climate changes and its direct and indirect effects on in-lake processes as well as external processes. Encouraging the re-establishment of collapsed populations of submerged plant communities should be given priority when implementing a management plan as buffers the impact of land-use intensification as well as enhancing in-lake biodiversity.

Beside anthropogenic impacts, it is well known that climate changes also have a direct influence on the distribution of lake temperature, nutrient loading, organic matter degradation, and lake water chemistry (Jackson, 2011). Increased summer rainfall events and higher temperatures will increase the responsiveness of phytoplankton populations to nutrient loading (Bates et al., 2008), placing greater emphasis on best management practices to attenuate nutrient loads to the lake. Water levels could also be expected to be higher with increased precipitation projected with a changing climate, resulting in a change in the balance of inflows and evaporation within a catchment and the lake The development of a comprehensive and consistent monitoring plan will ultimately help to establish any changes in the water quality of Lake Rotokauri before, during and after establishment of urban sub-catchment by Hamilton City Council.

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Appendices

Appendix I

The Lake Rotokauri results of the water quality parameters over the year of 2009

Date	Temperature (ºC) 0m	Temperature (ºC) 3m	DO 0m (mg L^{-1})	DO 3m (mg L^{-1})	Chl*a* (µg L^{-1})	TN 0m (g m^{-3})	TN 3m (g m^{-3})	TP 0m (g m^{-3})	TP 3m (g m^{-3})	Secchi Depth (m)
21 Jan 2009	22.7	21.3	7.93103	6.793	20.7	0.95	1	0.07	0.074	0.75
5 Feb 2009	24.5	23.5	8.51	2.33	27.6	1.2	0.81	0.084	0.064	0.75
9 Mar 2009	21.4	21.4	5.24	6.83	24.38	0.89	0.87	0.06	0.057	0.7
6 Apr 2009	22	19.2	10.3	8.78	15.8	0.75	0.78	0.038	0.051	1.72
11 May 2009	12.9	13.2	9.68	9.91	44.1	1	1	0.06	0.043	1

Date	Temperature (ºC) 0m	Temperature (ºC) 3m	DO 0m (mg L^{-1})	DO 3m (mg L^{-1})	Chl*a* (µg L^{-1})	TN 0m (g m^{-3})	TN 3m (g m^{-3})	TP 0m (g m^{-3})	TP 3m (g m^{-3})	Secchi Depth (m)
11 June 2009	12.22	9.9	11.64	11	34.2	0.92	0.89	0.043	0.038	1.14
9 July 2009	9.7	9.6	11.02	10.7	29.83	1.3	1.4	0.035	0.037	1.1
6 Aug 2009	11.2	10.9	11.64	11.39	22.12	1.2	1.7	0.037	0.035	0.96
10 Sept 2009	14.3	13	8.4	4.17	8.96	1.3	1.5	0.036	0.059	2.32
8 Oct 2009	14	13.8	9.4	7.83	23.64	1.4	1.2	0.046	0.05	1.7
5 Nov 2009	18.7	18.6	8.59	7.88	20.83	0.88	0.64	0.064	0.066	1
8 Dec 2009	21.4	19.6	8.24	2.87	21.1	0.99	1	0.081	0.061	0.7

Appendix II – Lake Rotokauri Water Budget (Raw Data)

Please refer to the CD.