Restoring freshwater quality: an integrated environmental, economic and ecosystem services assessment of reducing nutrient loads to Lake Rotorua

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Abstract

The management of freshwater resources is a global concern as anthropogenic pressures on these resources have intensified. A decline in the quantity and quality of freshwater urgently requires concerted restoration actions to reverse the decline. Restoration of lakes is impacted by complex relationships of ecosystem responses to management actions, economic constraints, societal pressures and policy contexts. Research in this thesis was aimed at an integrated assessment covering these aspects to inform future lake restoration action. The overarching objective of this thesis is to evaluate ecological and economic aspects of lake restoration, assessing the economic importance of ecosystems through exploring the costs of restoring and managing them, and assessing the ecological effectiveness of restoration actions. A focus was the evaluation of ecosystem services provided by a lake and its catchment, and uses quantitative analysis to study lake management dynamics to analyse what impedes successful lake restoration.

This research uses a case study of Lake Rotorua, central North Island, New Zealand. Like many lake ecosystems, this lake has been subject to pressures from human activity for decades, driving eutrophication and decline in ecosystem health. Land use has played a major role in nutrient enrichment of this lake. Restoration actions have been targeted at limiting nutrient loss from land, and reducing internal loading of the lake through engineering solutions. The aim of the research here was to analyse historical management responses and current management options alongside economic values of the lake and the catchment. The interdisciplinary assessment is aimed at connecting the lake to the catchment, as well as ecological to economic and policy assessments; such an integrated approach provides insight not only on the restoration of Lake Rotorua, but can also contribute to the management of lakes throughout New Zealand.

A main aim was to offer an integrated environmental, economic and policy assessment of the process of restoring water quality by reducing nutrient loads from the catchment alongside engineering options designed to reduce nutrients within the lake. The objectives for this thesis were to (1) explore a number of underlying drivers of lake management and the mechanisms for implementation of restoration initiatives, historically and in present times, (2) analyse ecosystem services and associated economic values of the lake ecosystem that are currently not valued by markets, and (3) evaluate cost-effectiveness of options to manage nutrient loads to the lake and internal loads through in-lake mitigation options, synthesising information on possible environmental outcomes and economic benefits for the lake and its catchment.

Several different methods were used to address these research objectives. To analyse the drivers of ecosystem degradation and resulting management initiatives, water quality data and research publications on Lake Rotorua were collected from 1922 to 2012. These data were categorised using the Drivers-Pressures-State-Impact-Response (DPSIR) model. Important management and environmental changes were used as independent variables in the year of their occurrence. Results of a Generalised Maximum Entropy (GME) regression, a specialised multinomial regression that is suitable for small datasets with a diverse range of variables (binary and continuous), showed that management was reactive, and policy responses (followed by regulatory interventions) often took effect only when ecosystem decline was already well advanced. There was also a disconnect between land use intensification and its role in driving water quality change, and long social lag times between the recognition of the environmental issue and a policy response delayed implementation of restoration actions.

To complement the historical study of lake restoration, the economic value of the lake and costs associated with degradation were evaluated to inform future management. Based on standard ecosystem services classification systems such as the Millennium Ecosystem Assessment in 2005, five different ecosystem services provided by the lake (habitat provision, food, nutrient processing, aesthetics, and recreation) were assessed. A range of direct and indirect valuation approaches, including existence value, hedonic pricing, and replacement cost, were used to assess the annual economic value of the lake ecosystem. A range of direct and indirect valuation concepts, including existence value, hedonic pricing and replacement cost, were used to assess the annual economic value of the lake ecosystem. A potential damage cost of the impacts of continued eutrophication was estimated as part of this economic assessment based on the current value of the lake and estimated reduction factors in ecosystem service provision and associated values.

To assess the effectiveness of options to reduce nutrient loads to the lake, a range of nutrient load scenarios was analysed for their effect on lake water quality using a hydrodynamic-ecological lake model (DYRESM-CAEDYM). Options needed to meet nutrient reductions as key component of a lake water quality target set by policy included mitigation on land, mitigation in the lake, and land use change. Different costs were attributed to each option, including lake and land mitigation costs. Values of different land use types and ecosystem services values of the catchment were included in the analysis. Costs and changes in values were determined for each scenario, to integrate an analysis of environmental effectiveness and costs of management options. Costs associated with these options were determined to illustrate the environmental effectiveness and potential costs of management options.

Main findings of this thesis are that freshwater management is subject to response lags between the recognition of environmental decline and the restorative action undertaken to address this decline. These lags relate to the visibility of environmental problems in the public sphere, with action often only undertaken once effects such as lake weeds, low water clarity and possibly toxic algal blooms impact the public. Social lag times also exist as part of the general bureaucratic management process, and within scientific research to inform management. Lags mean that ecosystem degradation can be allowed to proceed until impacts, at times, become irreversible. This research found that a better integration of science and policy could inform management decision making by providing a holistic framework integrating ecological knowledge, pursuits of economic growth and societal constraints.

The ecosystem services value of Lake Rotorua based on the ecosystem services of food, habitat provision, nutrient processing, aesthetics and recreation was calculated to be $100-145 million per year. Ecosystem services provided by the catchment based on the current land use types were $176 million annually. A policy target for water quality was set for the lake. The ecological model was used to evaluate whether reduction scenarios meet such a water quality standard before assessing the management costs. The most cost-effective option to achieve the nutrient load reduction to the lake to meet this policy target was a combination of mitigation practices and land use change in the catchment. The current practice of alum dosing of two tributaries to the lake was instrumental in meeting water quality targets under current land use and resulting nutrient loads from the catchment. Best water quality outcomes were achieved by a conversion of intensive land use types

to exotic or indigenous forest. This option also showed the best economic outcomes, when non-market values including ecosystem services values were considered. Results here show the importance of integrating ecological and economic assessments for best management outcomes. They also show the economic significance of preserving and restoring ecosystems.

This thesis has revealed how environmental and social lag times can hinder the success of lake restoration actions, which at times can lead to irreversible degradation. It has shown the importance of making environmental decline visible in the public sphere, and the integration of science and policy to achieve effective management action. It has revealed how lake ecosystems are a major economic asset as well as being highly important for the provisioning of ecosystem services. The research has illustrated the economic potential of ecosystem services in the catchment. Response lags and ensuing degradation, as well as the economic importance of the lake ecosystem and its surrounding catchment mean that it is crucial to assess lake and land management in an integrated fashion. To restore the lake and maintain a resilient ecosystem, long term reduction of nutrient losses from intensive agriculture is an important component complementing in-lake restoration actions. This can be achieved through a combination of mitigation options reducing nutrient losses from intensive land use types, as well as land use change away from these types. This research analysed the management of both land and lake elements of a lake ecosystem. It has integrated analysis of ecological processes with an economic assessment of lake and catchment ecosystem services, and placed this within a policy and management context. The integration of lake and catchment, and ecology and economics in an applied context contributes to shifting research and management focus towards more integrated assessments, taking ecological, economic, social and cultural values into consideration.

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There are many people that have contributed to the research and writing of this thesis in many ways. I would like to thank my supervisors David Hamilton and Graeme Doole for their support, mentoring and advice throughout the preparation of this thesis. Thank you for ongoing discussions of ideas, constructive feedback and not least of all patience while I was working through the many challenges of an interdisciplinary PhD research project.

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Preface

This thesis consists of a general introduction discussing the background and context of this research. The main body of the thesis is made up of three research chapters. Research chapters have been published or submitted for publication. For this reason, some repetition and minor differences in style can be found among chapters. The work and research presented here is based on my own ideas, and initiative. Contributions by the authors are outlined where applicable. The first research chapter (Chapter 2) was published as “Response lags and environmental dynamics of restoration efforts for Lake Rotorua, New Zealand” by H. Mueller, D. Hamilton and G. Doole in *Environmental Research Letters* in July 2015. The General Algebraic Modelling System (GAMS) code for the maximum entropy model applied in this paper was developed by Prof. Graeme Doole. The second research chapter (Chapter 3) was published as “A framework to evaluate services and damage costs of degradation of a major lake ecosystem” by H. Mueller, D. Hamilton and G. Doole in *Ecosystem Services* in March 2016. The third research chapter (Chapter 3) has been prepared for publication in *Global Environmental Change* as “Costs and effectiveness of land use change and mitigation options to reduce nutrient loads to a eutrophic lake” by H. Mueller, D. Hamilton, G. Doole, J. Abell and C. McBride. The set up and calibration of the lake model applied in this paper was provided by Dr Jonathan Abell. The main body is concluded by a general discussion of research findings, management recommendations and areas of future research.

1 General introduction

The intercept of environmental, economic, social and policy factors influencing the process, including failures, of lake restoration is the focus of this PhD thesis. Resulting is an interdisciplinary study of ecological processes in the lake and the catchment; economic implications of land use, lake management and regulation; and implications of historical policy as well as future options for policy and planning. The interdisciplinary approach taken here is not commonly attempted, and brings with it the challenge of incorporating concepts, methods, and ideas of a range of scientific fields. However, the approach offers valuable insights into the management of natural resources by addressing the trade-offs between resource exploitation, economic growth, and biodiversity conservation (Section 1.2). The combination of in-depth knowledge of ecological processes and complexities, constraints due to the pursuit of economic growth and resource management processes therefore promises to offer solutions to common management questions.

Freshwater ecosystems are facing a range of pressures driven by human activity. Land use change, invasive species and climate change are some of the major pressures threatening these systems (Foley et al. 2005; Paerl and Paul 2012; Cline et al. 2014). These pressures can also be cumulative: for example, the effects of nutrient enrichment of lakes through land use are expected to have an even larger impact on lakes with a change in climate (such as increased temperatures and alterations in rainfall) (Paerl and Paul 2012). Interactions between these human drivers and ecosystem responses are complex, and subject to thresholds beyond which ecosystems may change into a substantially degraded state that can be driven even by small environmental changes (Groffman et al. 2006). At times these anthropogenic impacts can lead to irreversible change in the state of the ecosystem (Carpenter et al. 1999). Management of freshwater resources, including lakes, thus needs to account for the complexities of ecosystem responses, the diverse impacts of human activities on ecosystems, and their cumulative effects. At the same time, the socio-economic context of human activities and management from a policy and regulatory perspective need to be given consideration for effective freshwater management (Section 1.2.4). This thesis is placed at the nexus of ecological, economic and policy contexts of managing freshwater and restoring lakes. The interdisciplinary approach is taken to analyse processes of lake restoration and land use management, to account for socio-economic factors, and to provide insights into options for improved management of both the catchment and the lake.

1.1 Restoration of lake water quality

The impact of processes and activities in the surrounding catchment of a lake has been increasingly recognised, studied and taken into consideration in lake restoration measures (Smith 2003) (Section 1.2). Research has shown the importance of nutrient enrichment from anthropogenic sources accelerating the natural eutrophication process (enrichment of a lake with nutrients and associated increase in lake biomass productivity). In this process, it is important to consider which nutrients are limiting productivity within the system, and therefore need to be controlled artificially with chemical or mechanical interventions, or nutrient management within the catchment (Brock and Carpenter 2003; Schindler and Vallentyne 2008; Schindler 2012). All or several nutrients can play a limiting role in the productivity of a system and therefore need to be addressed in lake restoration actions (Harpole et al. 2011). In enriched states, nutrients (especially phosphorus and nitrogen) also accumulate in the bottom sediments of lakes. During stratified periods in nutrient enriched lakes, anoxic conditions can persist in the hypolimnion, which are influenced by increased productivity in the epilimnion. An increase in nutrient burden allows for increased algae growth, leading to higher oxygen consumption when these organisms die and sink to the lake bed. This anoxia can lead to the release of phosphorus from the sediment (Bade 2009), and toxic sulphur compounds can also develop (Harper 1992; Carpenter et al. 1999; Schindler and Vallentyne 2008; Larned et al. 2011).

A further feedback loop exists in lake ecosystems where high turbidity supports the recycling of anoxic phosphorus from the sediment. This leads to more algal growth, which in turn increases turbidity. In contrast, low turbidity prevents the recycling of anoxic phosphorus from sediments and hence is a limitation to algal growth (Schindler and Vallentyne 2008). Additional negative effects of algae proliferation are fluctuations in pH of the water and the physical occupation of habitat that excludes other organisms. In combination with decreases in oxygen within the water column, these effects mean a decrease in habitat quality and potentially, increased mortality of organisms existing within the ecosystem (Larned et al. 2011).

With an increase in internal nutrient load, oxygen depletion can be more severe and more prevalent, leading to even more nutrient release from the sediment (Ingall and Jahnke 1997). This characteristic of biochemical changes and feedback loops is important to consider in the management of any lake system, as at times the mere reduction of nutrients from the surrounding catchment is not sufficient to halt anthropogenic drivers of eutrophication and restore a lake, especially where continuous nutrient inputs from the catchment have led to a significant amount of stored nutrients in the sediment, or where internal loading of nutrients is naturally high.

Based on these feedback mechanisms, ecosystem regime shifts can take place if influencing factors are sufficiently strong. For example, lakes can suddenly change from one state to another, such as tipping from oligotrophic to eutrophic levels (Scheffer et al. 2001; Scheffer and Carpenter 2003). Sudden shifts often occur without prior warning, meaning that a change in a lake can be hard to detect until a regime shift takes place. For deeper lakes, these changes tend to occur more smoothly than those in shallow lakes (Scheffer et al. 2001).

As with all ecosystems, lakes exhibit a certain amount of resilience towards external pressures. Resilience here refers to the ability of the system to remain in a stable state despite the presence of a number of major pressures (Carpenter and Cottingham 2002; Carpenter and Folke 2006). For lakes, these include external nutrient loading, invasive fish and plant species, climate change, and a range of other anthropogenic factors (Zhao et al. 2006; Allan et al. 2012; Özkundakci et al. 2014). Worldwide, human impacts have led to a number of changes to ecosystems, which have threatened the ability of diverse ecosystems to remain resilient to these impacts. Large scale abstractions, engineering of flows, construction of dams, distribution of invasive species, overfishing, and pollution with heavy metals, waste disposal, and nutrient inputs from diffuse and point sources have all put significant stress on rivers, lakes, wetland and estuaries (Jackson et al. 2001; Carpenter et al. 2009; UNEP 2012). In New Zealand, similar pressures are being exerted on lake ecosystems, with nutrient run-off from surrounding intensive land use being a major driver of water quality decline and threats to the resilience of many lakes around the country (Verburg et al. 2010; PCE 2015).

1.2 Freshwater management, land use and land use change

Water is the most important resource for humans, for ecosystems and for economies worldwide (PCE 2012). Freshwater is a global concern as these ecosystems face a range of anthropogenic pressures threatening their integrity. The resulting declines in ecosystem integrity lead to the degradation of one of themost important natural resource on which human societies, economics, and ultimately survival, depend (Jansson and Nohrstedt 2001; Gleick and Cooley 2009). A steady decline in freshwater quantity and quality is recognised as one of the most pressing environmental problems globally, and has affected lakes, rivers and wetlands worldwide (UNEP 2012). While ecosystems and economies alike depend on water (Wilson and Carpenter 1999; Foley et al. 2005), this resource is increasingly degraded by human activity (Cech 2005; Schindler and Vallentyne 2008). The amount of freshwater available in readily usable form is only a small fraction of all water on the planet, making up less than 2% of the total water available (Schindler and Vallentyne 2008). Yet freshwater covers 4.6 million km^2 of the continental land surface, and these ecosystems have important functions for processes of the biosphere, including the global carbon cycle (Downing et al. 2006; Tranvik et al. 2009).

Despite the high dependency on lakes, rivers and groundwater for a range of services from drinking water and fishing, sustaining livelihoods to hydropower and recreational uses, governments all over the world have often failed to protect and restore freshwater resources (Palmer and Richardson 2009; Moss 2010). Anthropogenic impacts on freshwater ecosystems vary across the globe from pollution to over-allocation of resources (Vörösmarty et al. 2010), and the management of these impacts is complex, costly and subjects to many uncertainties (Scheffer et al. 2000; Jeppesen et al. 2003). A major driver of changes in the biodiversity within freshwater ecosystems is intensive land use, leading to adverse effects such as habitat loss, water pollution and predisposing systems to the establishment of invasive species. These impacts have a significant impact on the health, resilience, and integrity of freshwater ecosystems (Stendera et al. 2012). Human factors also drive eutrophication of freshwater systems through increasing nutrient inputs lost from surrounding land use: for example, fertiliser application to support intensive farming systems. Widespread effects of eutrophication lead to increased phytoplankton growth, algal blooms, depletion of oxygen in bottom waters and can reduce values from recreational, cultural and aesthetic perspectives (Smith 2003).

A major human activity that impacts freshwater ecosystems worldwide is that of land use intensification pursued for agriculture (Foley et al. 2005; Erol and Randhir 2013). Intensive agricultural land use impacts on freshwater quality, but also produces food, fibre and other agricultural products. Furthermore, income from agricultural production can constitute a significant part of national wealth, complicating the trade-off facing regulators. Throughout the developmental stages of societies, agricultural land use has increased while indigenous vegetation has decreased. Land use patterns observed in many countries are now dominated by intensive agriculture and urban areas (Foley et al. 2005). Mitigating nutrient losses from intensive land use is valuable to maintain or restore ecosystem function, but this also often impairs agricultural production and profit (Doole and Kingwell 2015); thus, there appears an apparent trade-off between meeting the needs of human societies, and their capability to meet societal needs into the future (Foley et al. 2005). It is also a trade-off between economic growth, and the ability to sustain such growth in the long-term (Farber et al. 2006). While win-win situations here may exist, frequently resources are depleted rapidly with little regard for future economic, let alone social or environmental, sustainability (Farber et al. 2002). Within this context of environmental exploitation and subsequent degradation of natural resources, management approaches globally tend to be reactive – attempting to mitigate, minimise and restore – rather than proactively protecting these resources from over allocation to avoid costly consequences in the long term (Carpenter et al. 1998). A major objective of this research is to discover reasons for continued exploitation of natural resources without regard for long-term impacts, and what alternative approaches might be possible.

1.2.1 Land use and land use change in New Zealand

In New Zealand, industries such as farming, tourism, recreation and hydropower are highly dependent on the availability of water, while at the same time exerting a range of pressures on both the quality and quantity of this resource. With European colonisation beginning in the 19th century, land use change started to have major impacts on freshwater ecosystems. Agricultural expansion resulted in a major extent of lowland wetlands being drained to enable pasture growth in these fertile areas. Drainage was in fact encouraged and subsidised by the government (Brown et al. 2015). European settlement significantly altered the composition and integrity of both freshwater and terrestrial ecosystems through the conversion of indigenous vegetation to land sustaining agriculture. Agricultural land use now makes up more than 40% of the New Zealand land mass, with most catchments impacted by its activities through a reduction in biodiversity, increases in soil erosion and point and non-point source pollution of waterways (Paert et al. 2012).

Agriculture is now a significant source of diffuse pollution, affecting waterways through the production of excess nutrients, sediment and faecal bacteria (Parkyn et al. 2002; MacLeod and Moller 2006; Jay 2007; Baskaran et al. 2009; Abell et al. 2011). Intensification of land in agricultural use is continuing, and in fact accelerating. Between 2008 and 2012, close to 160,000 ha of dairy land have been added nationwide, mainly through conversion of sheep and beef farms, driven by profitability and commodity price relativities. Most of this change occurred in the Waikato, Canterbury, Otago and Southland regions. A focus on increasing productivity from these intensive land uses has led to a number of environmental effects, including increased pressures on waterways from excess nutrients and soil erosion (PCE 2015). At the same time, agriculture is a critical part of New Zealand’s economic growth agenda, and continues to be a major contributor to the country’s economy (see section 1.2.4 for a more detailed discussion).

Land use intensification and a focus on increased productivity are linked to dramatic changes in stocking rates over the last decade. While total stock numbers of sheep and beef have decreased where sheep and beef land has been converted to dairy, dairy cow numbers have increased by 1.6 million between 2004 and 2014 (Statistics NZ 2014). More intensive farming, especially within the dairy industry, has seen an increase in fertiliser applied to land, especially during the years 2000-2010 (Figure 1). There was also an increase in the amount of supplementary feed imported from overseas, such as palm kernel, which was first imported in 1997, and by now is a major feed supplement used in the industry (Figure 1). A waste product of palm oil production, this supplementary feed also has global environmental implications, as palm oil is a leading cause of deforestation in tropical areas, and a major driver of biodiversity loss (Fitzherbert et al. 2008; Koh and Wilcove 2008).

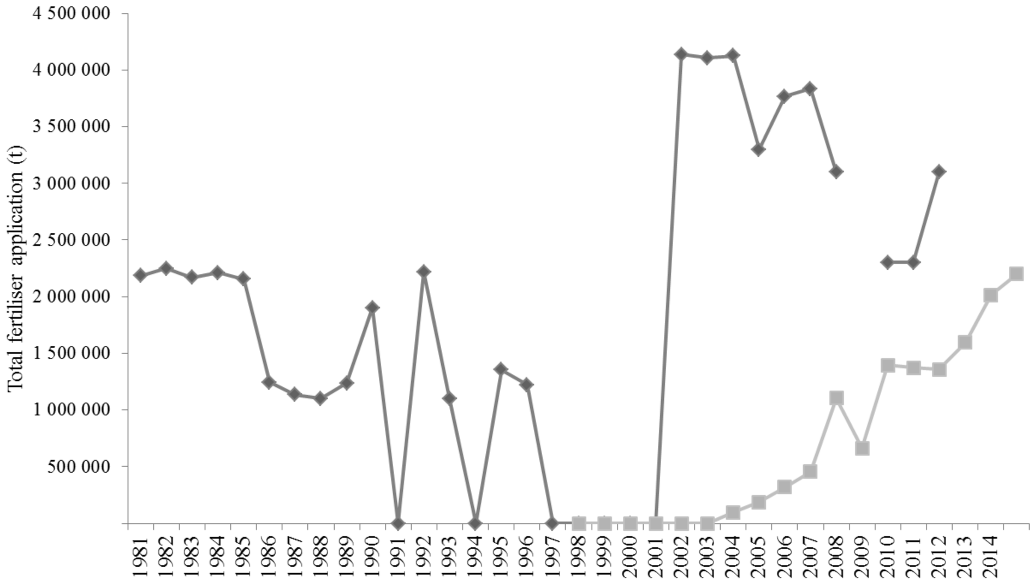

Figure 1 Total annual fertiliser use and palm kernel expeller (PKE) import in New Zealand, 1981-2014. Blank cells shows years where no figures were available. No PKE was imported before 1997. Data source: Statistics NZ[1].

1.2.2 Freshwater management in New Zealand

Similar to many countries with intensive agricultural sectors, water quality in New Zealand has been subject to steady decline. Lakes in the country are, in particular, affected by pollution from diffuse sources; for example, 44% of lakes were found to be eutrophic or worse, with 32% found to be of poor or very poor water quality in one study (Verburg et al. 2010). Almost half of the freshwater beaches in New Zealand were graded as poor or very poor water quality for recreational use in 2011, meaning that water quality is sufficiently poor to make swimming unsafe from a public health perspective (MfE 2012). Water quality and quantity are in decline in many regions in New Zealand, with lowlands particularly affected by the impacts of the surrounding land use (Cullen et al. 2006). Freshwater ecosystems in New Zealand are facing cumulative pressures from land use intensification (section 1.2.1), overharvesting of vulnerable indigenous species, and invasion of exotic species. Freshwater ecosystems are declining faster than terrestrial and marine ecosystems, and 74% of native freshwater fish species are threatened with

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[1] Palm Kernel Expeller: Statistics New Zealand (2015). Infoshare: Imports and exports – Harmonised Trade – Import. Retrieved from Statistics New Zealand Infoshare website (www.stats.govt.nz/infoshare). Fertiliser use: Amount of fertiliser applied in New Zealand, 1981 to most recent. Statistics compiled by Ministry for Agriculture and Forestry (now Ministry for Primary Industries), 2012, based on data provided by Statistics New Zealand, 2012.

extinction (Brown et al. 2015). Nonetheless, many of these species such as long-fin eel (*Anguilla dieffenbachia*), koaro (*Galaxias brevipinnis*) and giant kōkopu (*Galaxias argenteus*) are still commercially and recreationally harvested.

National management of freshwater resources has only recently made noticeable progress towards offering a quantified management framework and water quality standards. The establishment of a collaborative process (the Land and Water Forum) between many key stakeholder groups for New Zealand freshwater in 2009 motivated rapid progress towards the development of water quality standards, monitoring procedures and management approaches based on scientific evidence. The forum is made up of representatives of industry, non-governmental organisations (NGOs), iwi, scientists and other stakeholders of freshwater management. The Land and Water Forum (LWF) has since released a number of reports offering recommendations towards policy changes. The last one was issued in November 2015 (LWF 2015), and addresses water quality limit setting as well as the economic implications of improved freshwater management.

Informed by the LWF, and aligned with a wider reform of the Resource Management Act (1991), a National Policy Statement – Freshwater (NPS FM) was released in 2011, followed by a revised version in 2014 (MfE 2014). The bottom line in the NPS FM requires regional councils having to 'maintain or improve' the overall water quality within the region. . As part of the NPS FM, a National Objectives Framework (NOF) was developed to provide national water quality limits as a baseline for standardised monitoring and management of freshwater bodies. The NOF sets national bottom lines below which no ecosystem may fall, and offers classifications and maximum limits for parameters such as nutrients, dissolved oxygen, sediment and *E. coli*. While the NPS FM came into effect in 2014, it encompasses a lengthy time period during which councils have to implement the guiding principles of water quality maintenance of improvement in their regional plans and consenting processes: implementation of the NPS FM is not required until 2025, with a possible extension until 2030 where it is deemed otherwise unfeasible by a council (MfE 2014). Furthermore, some questions remain around the level of water quality limits set under the NOF with regards to their ability to maintain ecosystem integrity, as well as account for community values such as being able to swim in and take food from the managed freshwater resources (NZFSS 2014; LWF 2015).

1.2.3 Management of Lake Rotorua

Formed 230,000 years ago, Lake Rotorua (Table 1) is situated amongst a group of volcanic lakes (Figure 2) named the Te Arawa lakes. These lakes are named after the Te Arawa waka (canoe) that arrived at Whangapāraoa, North Island during the 1300s, eventually ensuing settlement in the Western Bay of Plenty region (Stafford 1986). Lake Rotorua and the surrounding lakes are of significant cultural value to local Te Arawa iwi (tribe). Rotorua in particular was seen as an important location for fish harvesting, including kōura (freshwater crayfish, *Paranephrops planifrons*) and kākahi (freshwater mussel, *Echirydella menziesii*) (Kusabs 2015). The original land cover in Rotorua's surrounding catchment, dominated by temperate rainforest, was cleared soon after European settlement to create pastoral land cover for farming (Clarkson et al. 1991). In the 1920s, the Rotorua lakes were owned by the Crown and intensive agriculture in its catchments was promoted (Stafford 1988), leading to continuous water quality decline in many of the lakes (Fish 1963; Fish 1964; Chapman and Brown 1966). The Crown returned legal ownership of the lake bed to Te Arawa iwi in 2006 as part of the Te Arawa Lakes Settlement Act.

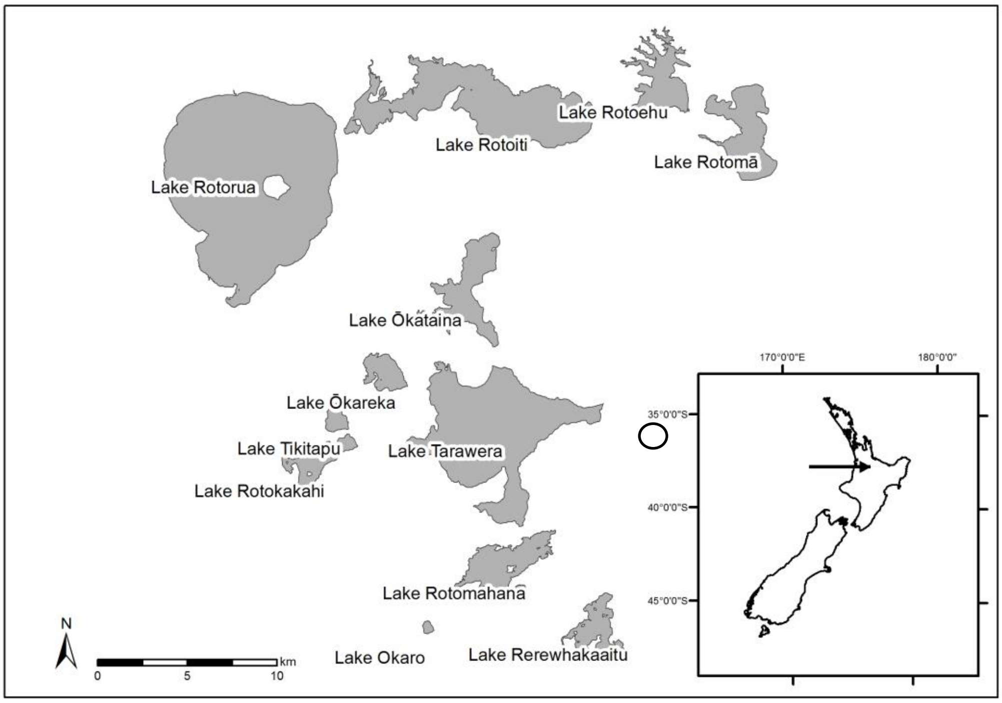

Figure 2 Map of location of Lake Rotorua, and the Te Arawa Rotorua lakes.

While water quality problems were recognised in the 1920s (Phillips and Grigg 1922; Stafford 1988), significant research on water quality however only commenced in the 1960s (Hellaby 1960; Annett 1961; Fish 1963), at which point water quality had been severely compromised in several lakes. Catchments were predominated by increasingly intensified land use. In Lake Rotorua, degradation in water quality led to noticeable occurrences of weed growth in the 1960s (Annett 1961). Aerial spraying of weeds with Diquat (diquat dibromide) commenced in 1966, but action taken was insufficient in solving weed issues in the long-term, and did not address the underlying issues of water quality decline (Stafford 1988).

Table 1 Basic lake characteristics, Lake Rotorua.

Lake area	8000 ha
Catchment area	45000 ha
Mean depth	10 m
Maximum depth	45 m
Surface elevation	280 m above sea level
Mixing regime	Polymictic
Trophic status	Eutrophic

In 1980, a project was launched to investigate environmental issues concerning the Rotorua lakes. A report from Future Options for the Rotorua Lake District (FORLD) (Planning Consultants 1981) called for an institutional framework in order to better manage the lakes (Stafford 1988; Miller 2003). At national level, the Resource Management Act (RMA) of 1991 provided an incentive to better manage natural resources. However, this did not immediately transpire into improved water quality in the lakes. In 1998, the Lakeweed Control Society became the Lakes Water Quality Society (LWQS), which is still active today. In the same year, a Lakes Management Working group made up of the Te Arawa Māori Trust Board, Environment Bay of Plenty (now Bay of Plenty Regional Council) and Rotorua District Council came together to facilitate cooperation between various interest groups involved in managing the lakes, as well as developing solutions manage the lakes more effectively (HRC 2009). Community involvement with the restoration process exerted an amount of pressure on regulatory and managing authorities (Bay of Plenty Regional Council and Rotorua District Council), which played an important role in the actions taken towards improving Lake Rotorua's water quality (McLean 2014).

The first action to this effect was taken by regulating diffuse pollution as part of the Regional Water and Land Plan in 2008, known as Rule 11. Rule 11 requires the setting of nutrient benchmarks and specifically targets declining water quality in some of the Rotorua lakes. It came into operation in 2008, more than seven decades after water quality issues were first noted in newspapers and research literature, and more than four decades after the public debate around water quality problems in the lakes started. The role of lag times, referring to long periods of time passing between the recognition of environmental degradation and management responses implemented to counter this decline, is an important contribution to the failure to halt ecosystem decline and restore lake ecosystems. These lag times are quantified and discussed in Chapter 2.

The regulatory approach described as part of Rule 11, sets a limit on nutrients lost from the catchment at 435 tonnes (t) of nitrogen and 37 t of phosphorus per year (Regional Water and Land Plan 2008). Current losses are estimated at approximately 750 t N and 47 t P annually (EBOP 2008; EBOP 2009). Nutrient limits were set to achieve water quality levels that are expected to have been present in the 1960s, quantified through using the water-quality indicator Trophic Level Index (TLI) (Burns et al. 1999). The TLI, an aggregate indicator using concentrations of nitrogen, phosphorus, and chlorophyll *a*, and Secchi depth as a measurement of clarity, is commonly used in New Zealand to represent the trophic state of a lake. The policy target for Lake Rotorua was set at a TLI of 4.2, which is within the eutrophic range, but is still below the current level observed (TLI of 4.4. for 2014/2015, BOPRC 2015) (Table 2). A graph showing TLI changes over the last decades is shown in Figure 4. A recently proposed catchment management plan (Integrated Management Plan 2015) attempts to achieve this nutrient load reduction through partial implementation of nutrient loss reduction through best management practices on intensive agricultural land use, land use change supported by an incentives scheme, and engineering solutions (Omundsen 2013; Rotorua Lakes 2015).

Table 2 Trophic Level Index and trophic status classes, example lakes in New Zealand. Adapted from Burns et al. (1999).

Trophic Level Index	Trophic State	Example lake in New Zealand
<2	microtrophic	Lake Pukaki, Canterbury
2-3	oligotrophic	Lake Rotoma, Bay of Plenty
3-4	mesotrophic	Lake Rerewhakaaitu, Bay of Plenty
4-5	eutrophic	Lake Rotorua, Bay of Plenty
>5	hypertrophic	Lake Ellesmere/Te Waihora, Canterbury

Alongside regulatory actions taken to manage nutrients lost from intensive land uses within the catchment, restoration measures were also implemented in the lake. In 2006, Bay of Plenty Regional Council commenced the application of aluminium sulphate (alum) to one of Rotorua's surface inflows (the Utuhina stream). Inflows in a second stream (the Puarenga stream) have been dosed since 2009. Alum is added to the inflows to reduce dissolved phosphorus concentrations as it distributed in currents throughout the lake. Alum binds to phosphorus and flocculates it, thereby removing it from the water column. In Lake Rotorua, alum has also been found to have an effect on the phosphorus concentrations in the sediment, effectively locking an amount of it in the sediment and preventing its release (McIntosh 2012). However, depending on several factors including oxygen concentrations in the bottom waters and pH levels, the sorption of phosphorus to alum in the sediment is not permanent and may be released in significant amounts back into the water column (Hamilton et al. 2015). There are also concerns around long term toxicity effects of continuous alum application on the ecosystem (Tempero 2015). The significance of alum for the restoration of Lake Rotorua to previous water quality levels and a number of management implications are discussed in detail in Chapter 4.

1.2.4 Socio-economic context of freshwater management and lake restoration in New Zealand

Restoration of lakes and the management of freshwater resources are set within a context of socio-economic constraints caused by the pursuit of economic growth, in New Zealand and globally. Natural resources are subject to a trade-off between exploitation for economic growth, and sustainable management of ecosystems to ensure resources are abundant for future generations (Farber et al. 2002; Foley et al. 2005; Farber et al. 2006) (Section 1.2). Agriculture, and dairy farming in particular, in New Zealand is seen as the 'backbone' of the country's economy, providing the country and the world with much needed food, and an integral part of many aspects of society (Mueller 2011; Doole and Kingwell 2015). While currently much governmental effort is directed at increasing productivity (for example through subsidisation of irrigation schemes), public awareness of the environmental effects of the intensification of farming is increasing (PCE 2004; PCE 2015). Yet, the trade-off between economic benefits and environmental costs remains one, if not the main, barrier to improved water quality and restoration of freshwater ecosystems (Edgar 2008).

At present, farming operations are not directly accountable for the environmental effects they might have on downstream freshwater ecosystems. Policy failures effectively grant a 'right' to pollute, without any accountability for water pollution caused. Water as a resource is taken freely for economic gain, but no payment is required for the damages done. This situation can also be described in terms of the 'Tragedy of the Commons' dilemma where a shared, freely-available resource is degraded through the non-cooperation of individuals, possibly causing its collapse (Hardin 1968). In this context, environmental pollution is an externality: clean-up costs and damages are not accounted for in markets (Tait et al. 2011). Instead, these costs are mostly borne by the environment and, if the externality is not adequately dealt with, then this may have implications for the rate-paying community and tax payers, where rates and taxes are used for restoration projects and land use mitigation. Key examples are central government funding of restoration projects, such as the Lake Taupō scheme and the Rotorua lakes action programmes (Paragahawewa 2006; EBOP 2009).

The environmental degradation that accompanies the sustained profitability of intensive agriculture has repeatedly been questioned, and is an underlying motivation of this research project. The contribution of dairy farming to New Zealand's GDP in 2013 was 2.8% or NZ$ 6.6 billion (MPI 2013). In comparison, tourism in the same year contributed 4.9% or $ 11.8 billion to GDP (Statistics New Zealand 2015). While dairy farming and other agricultural sectors make an important contribution to the New Zealand economy, tourism also contributes significantly.

1.3 Valuation of natural resources

The underlying motivation to include a form of valuation of the ecosystem studied here is driven by the desire to understand the trade-offs that are made with regards to cost and benefits of different types of land use, and to explore the impact human activities have on ecosystems in terms of changes in values due to changes in ecosystem state.

The valuation of nature and natural resources has historically caused some controversy, chiefly from perspectives opposed to bringing an economic context to something intrinsically valuable as nature and ecosystems (Foster 2002; Turner et al. 2003). However, ecosystem health is a prerequisite for sustainable economic growth (Foley et al. 2005). Economic dependence on the environment is described in Section 1.2. The exercise of valuing natural resources in market terms can be seen as a necessity to encourage smarter resource use (Turner et al. 2003). Economically, nature can be invisible, meaning that globally vast amounts of natural resources are lost as economic growth is pursued with little regard for the wider values of these resources. The failure to recognise the economic value of natural resources leads to major losses in these assets. Natural values need to be recognised, demonstrated and captured in markets in order to enable mechanisms that reward conservation on some level (de Groot et al. 2010).

Within freshwater resources, non-point source pollution is causing substantial economic losses (Carpenter et al. 1999). Economic impacts of eutrophication include decreases in water quality, reduced fish stocks, health risks to both humans and livestock, costly programmes to clean up and restore lakes, increased spending on water treatment, loss in recreational use such as swimming and boating, and reduced aesthetic value (Schindler and Vallentyne 2008). Natural resources are being exploited as they are valuable, but are being lost at significant scales where their degradation is not adequately valued.

The concept of ecosystem services can be used as a tool to capture, and a language to communicate, the value of natural resources. Ecosystem services are those services that are provided by ecosystems that are valued by humans and include regulatory services such as flood control, supporting services such as primary production, provisioning services such as food and cultural services such as recreational values (MEA 2005) (Table 3). Ecosystem services as a concept provide a classification to which monetary valuation methods can be applied. Indeed, a major study of the economic value of ecosystems, using a range of different valuation methods, estimated the value of all the world's ecosystems between US$ 16 and 54 trillion a year, based on 1997 prices (Costanza et al. 1997). This figure was updated to US$ 125 trillion per year in 2007 prices in a more recent study (Costanza et al. 2014). While there are concerns around the valuation techniques utilised for this study and the adequacy of the results (Boeraeve et al. 2015), this study is nonetheless an important contribution to fruthering understanding of the valuation of ecosystem services globally. While the numbers need to be considered carefully, this study shows the significance of ecosystem services in economic terms. Issues that were present in the 1997 study (Costanza et al. 1997) such as double counting have also been rectified in the updated study (Costanza et al. 2014).

Within freshwater research, existing valuation studies attempt to analyse the value of water quality by focusing on ecosystem services and assigning a monetary value to each of the relevant services, thereby determining an overall value of all combined ecosystem services in relation to the water body. Pretty et al. (2003) develop a cost-category framework in order to price the environmental costs of freshwater degradation in Wales and England. This approach divides environmental costs into social damage, ecological damage and policy-response costs. Pretty et al. (2003) use various methods for their valuation study, including proxies such as charges for licenses to estimate the value of water abstractions; days of lake closure to assess the effect of eutrophication on recreational use; reductions in the sales prices of waterside dwellings; costs of removing algal toxins as part of drinking water treatment costs; routine maintenance and clean-up costs of waterways; and health costs to humans, pets and livestock. The estimated total damage costs is £105-160 million per annum, with total policy response costs estimated at £77 million per annum (Pretty et al. 2003).

Table 3 List of ecosystem services provided by a lake ecosystem, including type and descriptive example. Table adapted from Schallenberg et al. (2013).

Type	Ecosystem services	Example
Provisioning	Water	Drinking water supply, stock water, irrigation
	Fisheries	Commercial fishing Spawning habitat
	Food	Wild food provision
	Waterfowl	Habitat
	Biodiversity	Species, habitat, ecological functions
Regulating	Nutrient processing	Removal of nutrients (e.g. denitrification); water filtering (e.g. filter feeders)
	Sediment processing	Water filtering (e.g. kakahi)
	Hydrological regulation	Flood control, landscape water retention
	Climate change mitigation	Carbon cycling/sequestration
Cultural	Recreation	Contact recreation Sport fishing Hunting Tourism
	Aesthetics	Amenity values
	Cognitive information	Science, research, environmental education

A further study focuses on the potential economic damage of eutrophication to freshwater in the United States. Dodds et al. (2009) determine annual value losses associated with eutrophication degrading freshwater ecosystems. Factors taken into consideration include losses in recreational water usage, waterfront property values, recovery of endangered species and drinking water. In this study, biochemical characteristics of water bodies such as current and reference nutrient concentrations of total nitrogen and total phosphorus, Secchi depth and richness of macroinvertebrates, fish and primary producers were used as a basis for economic value estimations. The total monetary losses estimated were $2.2 billion per year, with the biggest impacts evident for real estate values and losses associated with the recreational use of waterbodies (Dodds et al. 2009).

Critics of these studies have questioned the usefulness of ecosystem services and its paradigms. The concept is inherently anthropocentric as it is focused on the benefits of ecosystems to humans, and is often used as a tool to integrate ecosystems and conservation with human activities. Concerns have been expressed about the contribution of ecosystem services valuation to further conservation aims (Redford and Adams 2009), and underlying concepts used for valuation studies (Boeraeve et al. 2015). While employing economic logic to nature and conservation has apparent risk, the concept has nonetheless been used widely to describe the complexity and importance of maintaing ecosystem integrity (Carpenter et al. 2009; de Groot et al. 2010). Other criticism identifies the concept as being misleading, misrepresenting and liable to offer a justification for continued natural resource exploitation rather than genuine conservation (Sharman 2010). Given the increasing amount of literature and research in relation to ecosystem services (Kinzig 2009; Atkinson et al. 2012; Allan et al. 2012), in this research the concept was nonetheless valuable to illustrate the importance of ecosystems to a broad audience within science and policy making. It was also viewed as an important tool to explore values associated with ecosystems that few other concepts currently provide.

Valuation studies have also been viewed as problematic because ‘double counting’ is an issue. Double counting refers to counting values more than once in different parts of the valuation process (Fu et al. 2011). Double counting can be avoided by having clearly defined spatio-temporal scales for ecosystem services, using a consistent classification, selecting appropriate valuation methods and valuing final benefits of ecosystem services (ibid).

1.4 Models and methodology

As an interdisciplinary research project, a number of models were drawn on as part of the research methodology to enable analysis of the ecological, economic and policy context of lake restoration and catchment management. A brief introduction of the models and methodologies used is included here to complement the methodology descriptions in the individual research chapters.

1.4.1 DPSIR model

The underlying theoretical framework used in Chapter 2 is the Drivers-Pressures-State-Impact-Response model (DPSIR), which is founded in environmental frameworks such as the Pressure-State-Response model (PSR) (OECD 1993). PSR is a reporting tool to describe the pressure on the environment caused by human activities that change the state of natural resources, and lead to responses in the way these are managed. Derived from this tool, DPSIR is a model used to analyse resource management processes, to understand dynamics of causes and effects at the intersection of society and environment and to inform decision-making in the policy process (Atkins, Burdon, et al. 2011; Pinto et al. 2013). The framework has been widely applied in research to manage diverse aquatic and terrestrial ecosystems, ranging from local to global scales (Tscherning et al. 2012).

Potential shortcomings of the DPSIR model include the exclusion of non-human factors exerting pressure on the environment and the assumption that causes and effects of pressures and responses follow a linear relationship (Svarstad et al. 2008). To adequately represent dynamics of natural resources, both managed and unmanaged factors should be considered, thereby placing studied systems in a wider context of social and environmental dynamics (Atkins, Gregory, et al. 2011). These aspects and limitations are further discussed in Chapter 2 (2.5.4).

The five categories of the DPSIR framework represent steps in the process within which human activities influence ecosystems. This can be seen as a linear, and possibly cyclical, process similar to the model explored in Carpenter et al. (1998). The process starts with *drivers*, which refers to human activities that are responsible for changes in ecosystems. These exert *pressures*, meaning any causes and processes for environmental change. The following step describes the *state* of the ecosystem, and changes associated with exerted pressures. Any change in the state of the ecosystem then has an *impact* in terms of how this affects the human population. Lastly, a *response* to the occurring changes takes place in how human activities are managed (Table 4)

Table 4 Five different categories and brief description within the DPSIR model.

Category	Description
Drivers	Human activities responsible for ecosystem changes
Pressures	Causes and underlying processes of environmental change
State	Changes in the background state of the ecosystem
Impact	Impacts that affect human population
Response	Responses to changes in human activities and management

1.4.2 General Maximum Entropy regression

Generalised Maximum Entropy (GME) is used in Chapter 2 to analyse a small set of data with qualitative characteristics: it is applied to a set of research publications categorised under the DPSIR model. The model is used to identify the statistical significance of various events and changes in the environment of a lake ecosystem within the framework of the DPSIR model. GME is an economic-statistical model for the estimation of unordered multinomial discrete choice problems (Golan, Judge, and Miller 1996). Advantages of this method lie with the avoidance of parametric assumptions and applicability to small sample sizes, even when correlations are present (Golan, Judge, and Perloff 1996). Using GME, it is possible to recover information about systems with incomplete or multinomial response data (Golan, Judge, and Miller 1996). GME models as described by Golan et al. (1996) are based on the concept of entropy developed by Shannon (1948) and Jaynes (1957). Here, entropy is related to measures of uncertainty in a variable Jaynes (1957) developed this concept to use as a basis to estimate pure inverse problems that cannot be solved using traditional techniques.

GME uses a linear regression described as:
(1):

$$y_t = \sum_{k=1}^{K} \beta_k X_{k,t} + e_t \ \ \forall t$$

where y_t is the dependent variable, β_k for $k = 1, 2, \ldots, K$ are unknown coefficients, $X_{k,t}$ are data for each parameter $k = 1, 2, \ldots, K$ over $t = 1, 2, \ldots N$ observations, ($\forall$ means ‘for all’ and e_t is the error term. Using GME, each regression coefficient β_k

and error term e_t is transformed to bounded discrete random variables on a compact support interval. The coefficient β_k is estimated through:

(2)

$$\beta_k = \sum_{c=1}^{C} P_{k,c}\, z_{k,c}\ \forall k$$

where probabilities $P_{k,c}$ are decision variables that can be calculated through nonlinear optimisation, $z_{k,c}$ are fixed supports that are chosen, and c = [1, 2, …, C] is the index of support points. Probabilities $P_{k,c}$ are subject to the constraints $P_{k,c}$ ∈ [0, 1] and $\sum_{c=1}^{C} P_{k,c}\, z_{k,c} = 1\ \forall k.$

Estimates of the error term (disturbance) are calculated through:

(3)

$$e_t = \sum_{d=1}^{D} W_{t,d}\, v_{t,d}\ \forall t$$

where probabilities $W_{t,d}$ are decision variables computed through nonlinear optimisation, $v_{t,d}$ are fixed supports that are chosen, and d = [1, 2, …, D] is the index of support points. Probabilities $W_{t,d}$ are subject to the constraints $W_{t,d}$ ∈ [0, 1] and $\sum_{d=1}^{D} W_{t,d} = 1$.

Equations 2 and 3 substituted into equation 1 results in the data equation for the GME regression as follows:

(4)

$$y_t = \sum_{k=1}^{K} \beta_k\, X_{k,t} + e_t = \sum_{k=1}^{K}\sum_{c=1}^{C} P_{k,c} z_{k,c}\, X_{k,t} + \sum_{d=1}^{D} W_{t,d}\, v_{t,d}\ \forall t$$

The objective function used in GME is the maximisation of the entropy criterion:

(5)

$$\max J\ = -\sum_{k=1}^{K}\sum_{c=1}^{C} P_{k,c}\, ln\left(P_{k,c}\right) - \sum_{t=1}^{T}\sum_{d=1}^{D} W_{t,d}\, ln\left(W_{t,d}\right)$$

Equation (5) is maximised subject to $\sum_{c=1}^{C} P_{k,c}\, z_{k,c} = 1;\ \sum_{d=1}^{D} W_{t,d} = 1$; $P_{k,c} \geq 0$; and $W_{t,d} \geq 0$ and generates a solution involving uniform probabilities (i.e. $P_{k,c}$ = 1/C and $W_{t,d}$ = 1/D). The probability distribution with the highest entropy is the uniform distribution since the equal allocation of probability between finite supports provides the least amount of information. The maximisation of the entropy equation

(5) subject to the constraints above as well as *N* data constraints specifying the regression model (4) identifies the probabilities that could have been generated by the data in the most number of ways.

1.4.3 DYRESM-CAEDYM lake model

The DYRESM-CAEDYM lake model used for water quality simulations in Chapter 4 is a process-based dynamic model consisting of two individual constituents. The Dynamic Reservoir Simulation Model (DYRESM) is a one-dimensional hydrodynamic model. It simulates physical processes within vertically distributed layers of the water column, including parameters such as temperature, salinity and density (Trolle et al. 2008). DYRESM represents mixing processes, inflows and outflows of a lake or reservoir (Hamilton and Schladow 1997). The second component, Computational Aquatic Ecosystem Dynamics Model (CAEDYM), is an ecological model simulating the aquatic ecosystem. CAEDYM includes biochemical and physical processes such as phytoplankton and zooplankton biomass, nutrient cycling and dissolved oxygen content (Hipsey et al. 2006).

Spatially, the model is resolved vertically into a maximum of 100 Lagrangian layers, each with a dynamic width that corresponds to changes in the inputs of inflows and outflows as well as meteorology. Initial layers are configured using measured data representing vertical patterns in temperature, salinity, and density (Hamilton and Schladow 1997). Inputs required for the coupled model are resolved at a daily time step. Meteorological input data comprises air temperature, rainfall, wind speed, vapour pressure, and short and long wave radiation. Input parameters for the inflows include nutrient concentrations, volume and temperature (Trolle et al. 2008). Output of the model includes vertically resolved physical parameters of temperature, salinity, and density; concentrations of particles; and biochemical outputs such as nutrient, dissolved oxygen and chlorophyll *a* concentrations (Hamilton and Schladow 1997).

DYRESM-CAEDYM has been successfully applied to simulate processes and water quality in a range of studies for lakes and reservoirs globally (Romero et al. 2004; Bruce et al. 2006; Trolle et al. 2011; Özkundakci et al. 2014). The model can be seen as a complex one-dimensional representation of physical and biochemical lake processes. While a multi-dimensional representation of the lake would be more representative of actual lake processes, the simplicity of a one-dimensional approach can account for the complexity of the natural system without increasing uncertainty in outcomes (Hamilton and Schladow 1997). Further shortcomings of the model include overall statistical model performance in comparison to measured results. Representation of observed data by the model has been found to be overall satisfactory, yet this has been noted to decrease with an increase in trophic level of the lake studied (Trolle et al. 2008). Performance of the model is limited by the availability of measured data used as input for parameters that represent key lake processes; for example a vertical resolution of temperature, density, and nutrient parameters (Tanentzap et al. 2007), or data on fish stock dynamics (Trolle et al. 2008).

Representation of physical aspects, such as temperature, are often more closely aligned with measured data than biogeochemical parameters such as nutrient concentrations (Hamilton et al. 2015). Further limitations, such as the ability of the model to represent the application of alum to Lake Rotorua, are discussed in Chapter 4. In this study, as in other existing applications, the model has been useful to simulate water quality outcomes for a range of scenarios such as the analysis of policy targets for nutrients (Trolle et al. 2008), restoration measures and climate change (Hamilton et al. 2012), the impact of internal and external nutrient loads on lake biomass (Burger et al. 2008), and lake nutrient cycling (Bruce et al. 2006). Water quality effects of different nutrient loads from the catchment are simulated in Chapter 4. For this study, the model has been highly relevant in linking catchment land use changes and associated changes in external nutrient loading to water quality outcomes within the lake.

1.5 Research objectives

To explore where lake ecosystem restoration has been impeded and how the process can be improved, this research project aims to analyse economic and policy constraints to lake restoration, catchment management to reduce nutrient losses, and the ecological enhancement of freshwater resources. It aims to contribute to answering questions around trade-offs between environmental and economic interests; as well as around future options for land management that move away from high intensity land use and agricultural systems as the dominant means to support profits and economic growth. This thesis used a case study of Lake Rotorua (Central North Island, New Zealand) and its catchment as a context of these objectives. The research is situated in a timely context where public concern for conservation is gaining momentum on a national and global scale. In this wider context, this research presents an innovative, interdisciplinary view on the management of freshwater and lakes in particular; exploring new ways of valuing these resources, and ways of connecting ecological knowledge and economic principles.

This research attempts to overcome some of the barriers and socio-economic constraints highlighted in Section 1.2.4. The objectives of this thesis were to offer an integrated analysis of lake restoration and lake ecosystem services by (1) exploring mechanisms and drivers of the freshwater management process, historically and in present times, in the context of Lake Rotorua, (2) developing a systematic approach to assess ecosystem services and associated economic values of the lake ecosystem that are currently not valued in market terms, and (3) evaluating the cost-effectiveness of options to manage nutrient loads to Lake Rotorua with a focus on both environmental outcomes and economic benefit for the catchment and the lake.

The objective is to illustrate that there is a value associated with a lake ecosystem that can be described in economic terms, and that degradation of the ecosystem can lead to a loss in this value which also has economic implications for the region that associates with this lake ecosystem. The intention is not to assign a definite monetary figure to the ecosystem, but rather to explore what values are in fact associated with it, and how these values can be damaged through environmental exploitation. This was intended to inform the ongoing debate around trade-offs between options for natural resource utilisation, which often leads to the depletion of these resources and the destruction of ecosystems.

There are a number of research gaps that this study addresses. As an interdisciplinary study, it aims to integrate an ecological study of lake restoration with resource management, as well as with aspects of policy and economic analysis. Few existing studies have successfully integrated these aspects (Carpenter et al. 1998; Wilson and Carpenter 1999; Farber et al. 2006; de Groot et al. 2010). In order to offer such an integrative approach, this study uses novel ways of applying quantitative methods such as general maximum entropy regression to a qualitative dataset representing various events and changes in the environment of a lake ecosystem, while using a qualitative theoretical model as a framework (Chapter 2).

It also combines ecological methods, such as lake modelling, with economic analysis considering both the lake and the catchment scale; an integration that is not frequently achieved (Le et al. 2010; Kragt et al. 2011) (Chapter 4). A further gap which this research contributes to filling is the representation of lake ecosystems in the ecosystem services and ecosystem services valuation literature. While some global studies incorporate lakes in their valuation (de Groot et al. 2010; Costanza et al. 1997), studies of individual lake ecosystems are underrepresented in the fast growing body of ecosystem services valuation literature (e.g., Peterson et al. 2003; Dolinar et al. 2010; Schallenberg et al. 2013; Liu 2014). Chapter 3 offers a contribution to existing literature of economic studies of eutrophication and lake restoration of Lake Rotorua (Bell and Butcher 2003; Bell et al. 2004; Kerr et al. 2007; Parsons et al. 2015) by developing a systematic approach to value ecosystem services provided by a lake, as well as potential losses in values through environmental degradation and eutrophication. This approach is useful for future studies of lake ecosystem services and losses associated with degradation in New Zealand and internationally.

1.6 Thesis overview

This thesis consists of three separate research chapters that have been published or submitted for publication in peer-reviewed journals.

In Chapter 2, main factors of failure to successful lake restoration and management are evaluated by analysing research, water quality changes and important management responses towards restoration of Lake Rotorua over the last century using the DPSIR model and GME regression.

In Chapter 3, a systematic approach is described to value the ecosystem services of a lake, as well as potential losses in value caused by environmental degradation. This is then applied to the ecosystem services provided by Lake Rotorua, including a damage cost scenario associated with water quality decline.

In Chapter 4, a lake model is applied to simulate water quality changes associated with different nutrient loads from the Rotorua catchment. Scenarios of land use change and mitigation options to meet these nutrient loads are then evaluated for their cost, and compared to their effectiveness in improving water quality decline and meeting a set policy target.

The last chapter summarises main research findings and limitations of the study. It also draws on wider implications of the research for freshwater management and the restoration of Lake Rotorua, including recommendations for management and opportunities for future work.

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2 Response lags and environmental dynamics of restoration efforts for Lake Rotorua, New Zealand

2.1 Introduction

There is growing concern globally about degradation in the health of freshwater ecosystems (Dudgeon *et al* 2006; Vörösmarty *et al* 2010). Despite a high dependency on freshwater for the provision of a range of ecosystem services, governments globally have often failed to protect, and struggled to restore, freshwater resources, including rivers, streams, lakes and wetlands (Palmer and Richardson 2009; Moss 2010). In freshwater systems in particular, pressures from human activity have led to profound changes in ecosystem integrity and resilience (Folke 2006; Dudgeon *et al* 2006; Strayer and Dudgeon 2010). Management responses have often been insufficient or slow to mitigate these effects and at times ecological changes pass a threshold where they are difficult to reverse (Carpenter and Brock 1999; Scheffer and Carpenter 2003). In many cases, freshwater management is reactive rather than proactive (Carpenter *et al* 1998; Smith 2003), and mitigation of anthropogenic impacts on ecosystems is inherently complex, expensive, and subject to many uncertainties (Jeppesen *et al* 2003; Schindler 2012).

Inability to fully integrate ecological processes with management actions has been identified as contributing to flawed decision-making processes for freshwater management (Moss 1999). Whilst some studies of human impacts on ecosystems have considered an integrated ecological management approach and socio-economic aspects (Carpenter *et al* 1999; Atkins *et al* 2011; Pinto *et al* 2013), socio-economics and management actions have not necessarily been measured and quantified. Statistical analysis of qualitative datasets is not commonly incorporated into ecosystem management studies (e.g. Ellison 1996; McCann *et al* 2006). Ability to quantify outcomes from integrated management pathways based on ecological principles could address this constraint. Combining qualitative with quantitative research could lead to demonstrable impacts on ecological, socio-economic and management systems.

The Drivers-Pressures-State-Impact-Response (DPSIR) model is used to analyse resource management processes and inform decision-making in the policy process (de Groot *et al* 2010; Atkins *et al* 2011; Pinto *et al* 2013). This framework is widely applied in ecological research to manage diverse aquatic and terrestrial ecosystems, from local to global scales (Tscherning *et al* 2012). Within DPSIR, the dynamics of change can be viewed as linear or cyclical processes in the context of water management, as described by Carpenter *et al* (1998). The process starts with *Drivers* referring to human activities responsible for changes in ecosystems. These exert *Pressures*, which are processes leading to environmental change. The middle step describes the *State* of the ecosystem. A change of the state then has an *Impact* in terms of effects on ecological processes and the human population. Lastly, *Response* refers to how the human activities are managed to prevent adverse impacts.

The primary objective of this paper is to ascertain which factors were driving research focus with regards to the five categories of the DPSIR framework, using a case study of an iconic lake in New Zealand. Historical data for Lake Rotorua (North Island, New Zealand) from 1922–2013 have been used to provide a quantitative analysis of management responses. Publications are here used as a proxy to represent environmental changes and management dynamics. Other options that could have been used include newspaper articles, interviews, data on research funding or regulatory documents. Publications were chosen as they were easily quantifiable, could be categorised into the DPSIR framework, and were the only dataset available for most years of the study period.

Water quality problems in Lake Rotorua have been linked to changes in land use and are characterised by long lag times (Fish 1969; Howard-Williams *et al* 1986; Abell *et al* 2011). We test the hypotheses that management responses to ecosystem degradation are linked to the visibility of ecosystem degradation in the public sphere; that social lag times between recognition of environmental issues and regulatory action can slow down such management responses; and that management strategies are reactive rather than proactively preventing ecosystem decline, and prone to fail to address the underlying causes for environmental degradation. We examine the intersecting elements of social lag times and poor visibility of environmental change that have affected the management of the lake historically, and how contemporary restoration approaches will need to avoid these elements. My approach was to use general maximum entropy (GME) regression to analyse collected data. This multinomial method is suitable for the analysis of small datasets of continuous and binary variables, even when correlations between explanatory variables are present (Golan *et al* 1996).

2.2 Methods

To identify mechanisms causing failures to prevent lake ecosystem degradation, environmental changes, and management-response barriers to the restoration of Lake Rotorua were analysed. Data on historical changes of ecological health and management of the study lake were collected. Research publications were used as the best available proxy representing knowledge of environmental change and management responses. The Drivers-Pressures-State-Impact-Response (DPSIR) framework (Atkins *et al* 2011) was applied to the case study. Multinomial logistic regression analysis based on the general maximum entropy model (Golan *et al* 1996) was used to study the effect that water quality and important management changes have had in stimulating studies in each variable category. Binary explanatory variables were included to explore the significance of relevant regulatory and institutional events. The objective of this study was to examine the significance of events and changes over time within the context of the DPSIR categories. We identified significant trends and evaluated the usefulness of the DPSIR framework.

2.2.1 Study site

The Te Arawa lakes in the Central Volcanic Plateau, North Island, New Zealand, comprise 12 lakes of volcanic origin with varying characteristics and ranging in trophic state from oligotrophic to hypertrophic (Burns *et al* 2005; Scholes 2011). The largest lake, Lake Rotorua (Figure 3), is the subject of this study. Its mean depth is 10 m and maximum depth is 45 m. It is presently classified as eutrophic. The original dominant land cover for this lakes region was temperate rainforest (Clarkson *et al* 1991). Soon after European settlement in the 1880s, bush and forest were cleared and farming became widely established. Ownership of the lake was taken from the Te Arawa tribe by the Crown in 1922 and returned in 2006.

The change in land use led to water quality problems that were first recognised in the 1920s through the occurrence of weed growth in the lake (Phillips and Grigg 1922; Stafford and Rotorua District 1988). More intensive water quality research commenced in the 1960s (Hellaby 1960; Annett 1961; Fish 1963), at which point water quality had been severely compromised in the lake. The catchment was subject to increasing pastoral conversion and agricultural intensification during this time and in subsequent years. Proliferations of exotic submerged weeds were noted at the beginning of the 1960s (Annett 1961). Aerial spraying of weeds with Diquat® commenced in 1966, but did little to address underlying causes of water quality decline (Stafford and Rotorua District 1988). Discharge of treated wastewater from the adjacent city of Rotorua (population c. 50,000) into Lake Rotorua ceased in 1991, when a tertiary-treatment land application commenced. This reduced nutrient loads, but after some improvement in water quality in the 1990s (Rutherford *et al* 1996), the quality continued to decline (Burger *et al* 2008). For an overview of changes in Lake Rotorua, selected water quality parameters measured over the last decades are given in Appendix I.

The first step towards regulating nutrient inputs to restore the eutrophic lake to its pre-1960s levels was taken in 2002 by proposing a nutrient loss limit for the catchment (Rule 11; Table 5). Alum dosing of the Utuhina Stream inflow to Lake Rotorua commenced in 2006 and was extended to include the Puarenga Stream inflow in 2009. The objective was to flocculate phosphorus (P) in the streams, reduce P loads to the lake and limit algal growth in the lake. Currently, a regulated cap on the catchment nitrogen load is intended to reduce nitrogen loads from the catchment by around 40% by 2030. Following alum dosing, water quality has recently improved and in 2012 met the target consistent with 1960s water quality levels for the first time (BOPRC 2013).

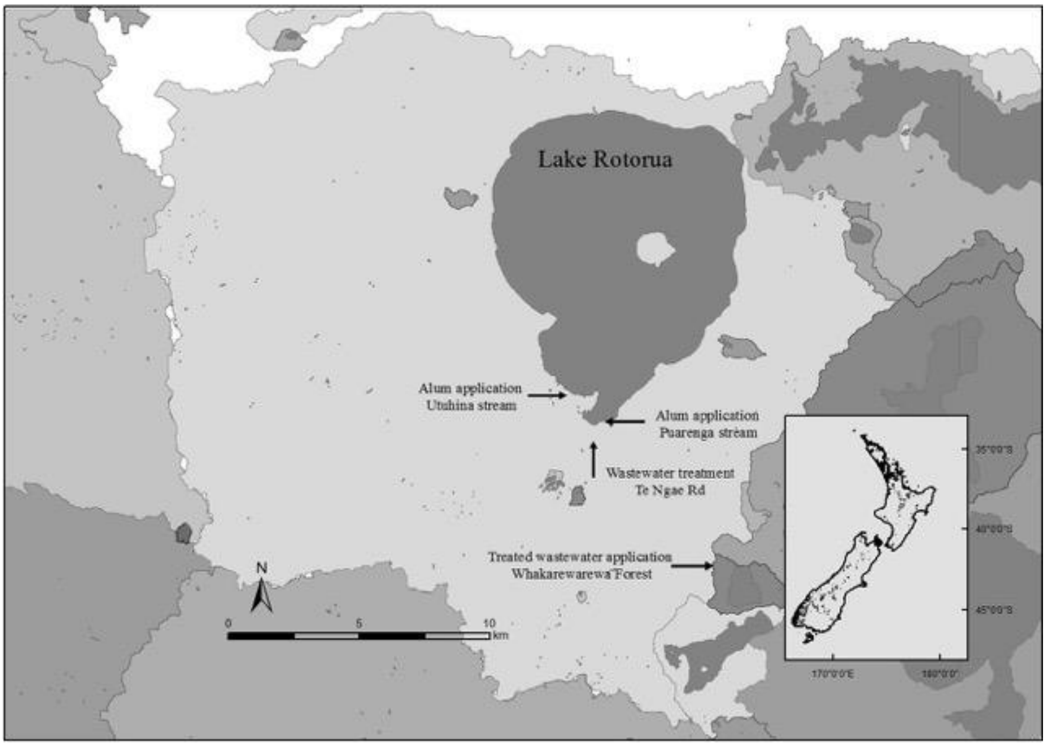

Figure 3 Catchment of Lake Rotorua, North Island, New Zealand. Arrows indicate streams for alum application and land-based treated wastewater application.

2.2.2 Data collection

We collected published data on water quality and ecosystem health of Lake Rotorua using database searches. A primary data source was a comprehensive bibliography covering years 1922–2002 (Miller 2003). A keyword search of Google Scholar, ISI Web of Science and NZ Science was conducted to expand and complement this dataset. Keywords used were “eutrophication”, “water quality”, “nutrient*” and “Lake Rotorua”. Data collection yielded a list of published documents addressing water quality issues for the lake from 1922 to 2013 (Appendix II). Each publication was assigned to one of the five categories of the DPSIR framework, according to the primary focus of the paper. The focus was determined through scanning of the text where possible; where the text was not accessible, abstracts and titles were used to choose the category the document primarily addresses (see category description in Introduction). Publications that covered more than one category were counted in the category that corresponded with their predominant focus. A total of 351 publications was collected and categorised.

2.2.3 Regression analysis

The dependent variable of the regression analysis consisted of a multinomial response, denoting whether a study focused on *Drivers, Pressures, State, Impact* or *Response*. A range of explanatory variables was incorporated. The regression analysis was applied to the 351 publications to investigate which factors (Table 5) had the greatest impact on the probability that a publication belonged to one of the five different DPSIR categories of the dependent variable.

One explanatory variable in the regression reflected water quality as indicated by the Trophic Level Index (TLI) observed at time of publication. TLI, similar to the Trophic State Index (Carlson and Simpson 1996), gives an assessment of the trophic state of a lake and is widely used in New Zealand as an integrative proxy for water quality (e.g. Verburg *et al* 2010). The index includes measurements of total phosphorus (TP), total nitrogen (TN) and chlorophyll *a* concentrations and Secchi depth (Burns *et al* 1999). TLI levels range from 0.0 (ultra-microtrophic) to 7.0 (hypertrophic). A eutrophic condition is denoted by a TLI level of 4.0 – 5.0.

TLI is calculated using equations relating to the four variables:

Chlorophyll *a* $\mathrm{TLc} = 2.22 + 2.54 \log(\mathrm{Chl}\,a)$ (1)
Secchi depth $\mathrm{TLs} = 5.10 + 2.60 \log(1/\mathrm{SD} - 1/40)$ (2)
Total phosphorus $\mathrm{TLp} = 0.218 + 2.92 \log(\mathrm{TP})$ (3)
Total nitrogen $\mathrm{TLn} = -3.61 + 3.01 \log(\mathrm{TN})$ (4)
Integrated value $\mathrm{TLI} = 1/4\,(\mathrm{TLc} + \mathrm{TLs} + \mathrm{TLp} + \mathrm{TLn})$ (5)

TLI was calculated using measured data provided by Bay of Plenty Regional Council and the National Institute for Water and Atmospheric Research (NIWA), based on samples taken routinely in the central region of the lake. Prior to commencement of measurement, TLI levels between 1922 and 1966 were interpolated under the assumption of a linear increase from a modelled TLI of 4.04 for the 1920s (Hamilton *et al* 2012). While earlier TLI levels are likely to have fluctuated, as measurements of later years show, this linear increase is expected to at least represent the trend of a continuous decline in water quality over those years.

Eleven binary explanatory variables were used to analyse changes in water quality research, management and environmental state. One variable indicated whether the publication was peer reviewed. This variable was included as it was deemed important to test whether or not the nature of the publication would have a statistically significant impact. Ten further variables were chosen as indicators of regulatory developments, environmental changes, or changes in institutions and science. Explanatory variables are listed in Table 5; a statistical overview showing the range of values for each variable is given in Table 6. Equations used for the regression analysis are given in Table 7. Entropy refers to measures of uncertainty in a variable, making it possible to recover information about systems with incomplete response data. GME is based on a linear regression problem where probabilities are calculated though nonlinear optimisation. The maximisation of the entropy equation (Table 7) identifies the probabilities that could have been generated by the data in the most number of ways. The regression problem is solved using nonlinear optimisation code, more specifically the General Algebraic Modelling System (GAMS) (Brooke *et al* 2014).

Results from the regression analysis yield an odds ratio, as well as a measure of statistical significance (p value) for each of the explanatory variables. The odds ratio shows how strongly one property or outcome, corresponding to the categories of the dependent variable, is associated with the presence of another property in a dataset. A coefficient for each explanatory variable shows the probability that publications focused on one of the categories of the dependent variable, compared to a baseline category. The *state* category from the DPSIR framework was chosen as the baseline category for the calculation of odds ratios. This was chosen for simplicity, because it is intermediate in the DPSIR series, and because this classification contained the highest number of observations.

Table 5 Overview of explanatory variables and description.

Variable name	Year	Description
TLI	1922-2013	Representation of water quality by Trophic Level Index which includes measurements of total phosphorus, total nitrogen, and chlorophyll *a* concentrations and Secchi depth
Peer review	1922-2013	Indication of whether publication was subject to peer review
Lake Weed Society	1961	Formation of society working to improve water quality, founded after occurrence of major lake weed problems at Lake Rotorua
Kaituna catchment	1975	Upper Kaituna catchment control scheme to promote soil conservation and further to control lake levels including flood protection stopbanks and planting of riparian margins
FORLD	1980	Future Options for the Rotorua Lake District, project formed to deal with land use and water quality issues, developing more sustainable resource use and alternative management options for the Rotorua lakes, including Lake Rotorua
RMA	1991	Resource Management Act; national level legislation promoting the sustainable management of natural resources, including water
Sewage	1991	Rotorua sewage treatment plant upgrade and diversion of treated waste water away from lake to land
Fonterra	2001	Formation of Fonterra Co-operative Group; large dairy marketing and processing co-operative, including more than 12,000 dairy farmers
EBOP Chair	2002	Bay of Plenty Regional Council Chair in Lake Restoration, based at the University of Waikato, to promote better management and monitoring of water quality within the Rotorua lakes
Cyano blooms	2003	Major blooms of cyanobacteria (e.g. *Anabaena planktonica*) in Lake Rotorua
Lake Settlement	2006	After being seized by the Crown in 1922, ownership of Lake Rotorua (lakebed) was returned to Te Arawa through signature of a deed of settlement
Rule 11	2008	Regional Water and Land Plan. First regional legislation affecting land management of the catchment of selected Rotorua lakes, aimed at controlling land use intensification, in particular nutrient (nitrogen) run-off from farms

Table 6 Summary of statistical information of explanatory variables.

Variable	Type	Mean	Std dev	Min	Max
TLI	Continuous	4.55	0.25	4.04	5.06
Lake Weed Society (1961)	Binary	0.98	0.15	0.00	1.00
Kaituna catchment (1975)	Binary	0.81	0.39	0.00	1.00
FORLD (1980)	Binary	0.64	0.48	0.00	1.00
RMA (1991)	Binary	0.46	0.50	0.00	1.00
Sewage (1991)	Binary	0.46	0.50	0.00	1.00
Fonterra (2001)	Binary	0.33	0.47	0.00	1.00
EBOP chair (2002)	Binary	0.30	0.46	0.00	1.00
Cyano blooms (2003)	Binary	0.28	0.45	0.00	1.00
Lake Settlement (2006)	Binary	0.22	0.42	0.00	1.00
Rule 11 (2008)	Binary	0.17	0.38	0.00	1.00
Peer review	Binary	0.33	0.47	0.00	1.00

2.3 Temporal resolution analysis

Time series analysis was used to analyse temporal resolution of the DPSIR framework and to quantify the lags between the DPSIR categories and the explanatory variables of the multinomial regression. For the analysis, a Pearson correlation coefficient r was calculated for each of the binary explanatory variables (except peer review, which is not linked to a particular year) relating to each of the categories of the DPSIR framework, using the equation

$$r = \frac{n(\sum xy) - (\sum x)(\sum y)}{\{n\sum x^2 - (\sum x)^2\}\{n\sum y^2 - (\sum y)^2\}},$$

where n (=351) is the number of pairs, x is the number of x scores (year of publication) and y is the number of y scores (number of publications). Coefficients were calculated for time lag series of 3 years, 5 years and 10 years after the year of each explanatory variable. To test statistical significance, the p value for each correlation coefficient was also calculated.

Table 7 Generalised maximum entropy regression equations and description.

Equation	Formula	Description		Constraints
Coefficient estimator	$\beta_k = \sum_{c=1}^{C} P_{k,c}\, z_{k,c}\ \forall k$	β_k	Regression coefficient	$P_{k,c} \in [0,1]$ and
		$P_{k,c}$	Decision variables	$\sum_{c=1}^{C} P_{k,c}\, z_{k,c} = 1\ \forall k$
		$z_{k,c}$	Fixed supports	
		$c = [1,2,\dots,C]$	Support points index	
Error term	$e_t = \sum_{d=1}^{D} W_{t,d}\, v_{t,d}\ \forall t$	$W_{t,d}$	Decision variables	$W_{t,d} \in [0,1] W_{t,d}$ and
		$v_{t,d}$	Fixed supports	$\sum_{d=1}^{D} W_{t,d} = 1$
		$d = [1,2,\dots,D]$	Support points index	
Data equation	$y_t = \sum_{k=1}^{K} \beta_k\, X_{k,t} + e_t$	$X_{k,t}$	Parameter data	
		$k = 1,2,\dots,K$	over N observations	
	$= \sum_{k=1}^{K}\sum_{c=1}^{C} P_{k,c} z_{k,c}\, X_{k,t} + \sum_{d=1}^{D} W_{t,d}\, v_{t,d}\ \forall t$			
Entropy equation	$\max J = -\sum_{k=1}^{K}\sum_{c=1}^{C} P_{k,c}\, ln(P_{k,c}) - \sum_{t=1}^{T}\sum_{d=1}^{D} W_{t,d}\, ln(W_{t,d})$	Objective function: Maximisation of the entropy criterion		$W_{t,d} \geq 0$ $P_{k,c} \geq 0$ $\sum_{c=1}^{C} P_{k,c}\, z_{k,c} = 1$ $\sum_{d=1}^{D} W_{t,d} = 1$

2.4 Results

2.4.1 *Water quality research and data*

All documents were categorised according to their primary focus within the DPSIR framework. Forty-seven publications were categorised in the *drivers* category, 75 in the *pressures* category, 116 in the *state* category, 20 in the *impact* category, and 93 in the *response* category (Appendix II). A timeline of environmental change, management responses and other important events represented by the binary explanatory variables is plotted in Figure 4. Results of the data collection, including water quality (TLI) and number of publications, are also given in Figure 4. A decline in water quality (i.e., an increase in TLI) is indicated by a move towards a TLI of 5.0 in the 1970s, with further peaks in TLI levels observed in 1985 (5.06) and 2003 (5.03). Publication numbers reached a first peak at a similar time (early 1970s), dropped off in the 1980s and a second peak occurred in the late 1990s and early 2000s.

There was a substantial increase in research soon after 1960 when water quality problems first started to become obvious to the public, with a peak in 1975 (total of 22). When visibility of eutrophication subsided after initial management responses, such as physical removal and aerial spraying of weeds, the number of publications decreased. Recent times saw consistently higher numbers of publications after 2003, when algal blooms again became prevalent on Lake Rotorua (Burger *et al* 2007).

Times of visible water quality changes led to responses in management to address the changes after some lag time of five to ten years (Figure 4). Lake weed occurrences in the 1960s were followed by the Kaituna catchment scheme in 1975, while cyanobacterial blooms in the 2000s were followed by Rule 11 in 2008. High trophic levels in mid-1980s were also followed by the sewage upgrade scheme in 1991 (Figure 4). Water quality of Lake Rotorua has fluctuated over the years, but a continuous trend of degradation resulted in these management responses. Only since 2006 has water quality continuously improved (Figure 4), suggesting that interventions until then were at best partially successful in addressing water quality issues, allowing degradation to continue.

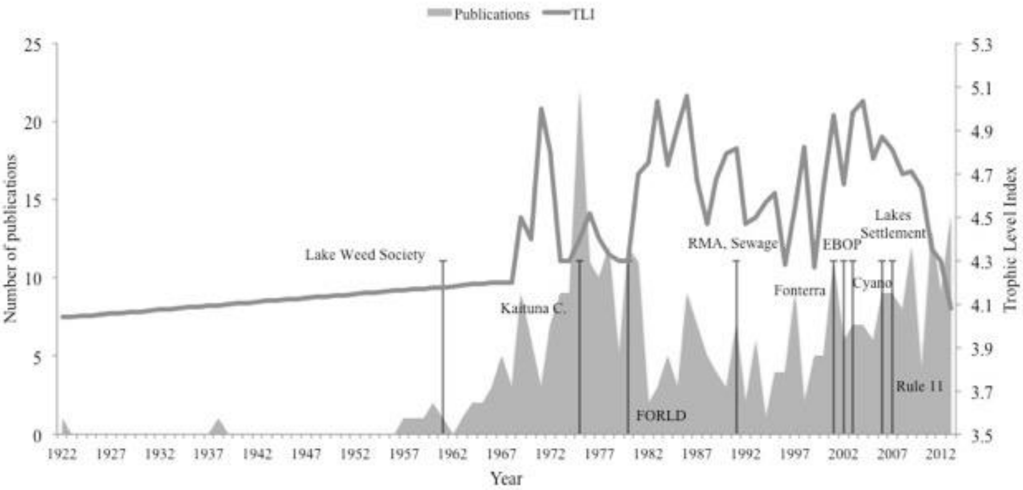

Figure 4 Water quality of Lake Rotorua and related publications for Lake Rotorua 1922-2013. Explanatory variables are marked in year of occurrence.

2.4.2 GME regression analysis

Water quality (TLI) had a significant relationship (p<0.05) with the category of publications (Table 8),with a shift in focus of publications to the *impact* category, relative to the *state* category. Occurrences of cyanobacterial blooms in 2003 also had a statistically-significant impact on the categorical focus of publications (p<0.05), and led to a noticeable shift in focus to the *impact* category, with odds being 73.2 times higher that a publication was focused on *impact*, relative to the *state*. A further statistically-significant effect (p<0.05) was the formation of the Lake Weed Society, which occurred following major lake weed problems in the 1960s. The Society associated with this variable had a significant impact in terms of changing the focus of publications, indicated by odds of 58.8 times higher for *impact* relative to *state*.

The variable with the least statistical significance (p>0.1) was Lake Settlement. The introduction of the Resource Management Act (1991) and the Future Options for the Rotorua Lakes (FORLD) project also had no statistical significance (p>0.05). The formation of the Fonterra dairy cooperative in 2001 and the Kaituna catchment scheme (1975) had no significant impact on the categorical focus of publications, indicated by low odds ratios. Contrary, the appointment of a University of Waikato-based Chair in Lake Restoration (2002) and the implementation of Rule 11 as a regulatory step aimed at managing land use for improvement of water quality in 2008 were of significance (p<0.05). Rule 11 had an impact in changing the focus of the publications to the *drivers* category, indicated by odds of 7.93 times higher relative to *state*. Odds for the response category were 3.27 times higher. The Lakes Chair appointment had an impact in changing the focus to the *drivers* (odds 57.0 times higher) and *pressures* (51.7) categories, relative to the *state* category.

The variable associated with removal of point source pollution through sewage treatment upgrades (1991) had the highest statistical significance ($p<0.01$), with odds ratios that indicated a shift in focus towards the *pressures* category, with odds three times higher relative to the *state*. The peer review variable showed no statistical significance ($p>0.05$) and low odds ratio (<1), indicating that this variable had no impact on the categorical focus of publications. My results show that water quality levels, visible water quality changes (algal blooms) and public campaigns were most significant in determining the focus of water quality research, which is shown by the high statistical significance and odds ratios of the explanatory variables most closely linked to this (water quality indicated by TLI, cyano blooms, Lake Weed Society, EBOP chair and Rule 11) (Table 8).

Table 8 Results of GME regression analysis presenting odds ratios of explanatory variables. Asterisks indicate statistical significance (*=statistically significant at 10% level, **=significant at 5% level, *=significant at 1% level).**

Explanatory variable	Type	p Value	Drivers	Pressures	Impact	Response
Trophic level	Continuous	<0.05**	1.28	1.71	8.98	1.99
Lake Weed Society (1961)	Binary	<0.05**	1.51	1.51	58.83	1.51
Kaituna catchment (1975)	Binary	<0.05**	0.84	0.68	1.83	0.38
FORLD (1980)	Binary	>0.05	1.88	4.36	0.53	5.82
RMA (1991)	Binary	>0.05	0.46	0.04	8.09	0.74
Sewage (1991)	Binary	<0.01***	1.78	3	0.89	0.94
Fonterra (2001)	Binary	<0.05**	0	0	0.95	0.74
EBOP chair (2002)	Binary	<0.05**	56.95	51.71	0	1
Cyano blooms (2003)	Binary	<0.05**	0.76	3.06	73.2	3.34
Lake Settlement (2006)	Binary	>0.1	2.66	0.6	3.39	0.14
Rule 11 (2008)	Binary	<0.05**	7.93	1.39	1.7	3.27
Peer review	Binary	>0.05	0.34	0.74	0.96	0.27
Count R^2	54.4%					

2.4.3 Temporal resolution analysis

Results of the time lag analysis show the temporal resolution of the DPSIR framework and the explanatory variables. A time lag of five years after each year associated with occurrence of the explanatory variables was identified as the most significant time step. A three-year lag showed some significance, while a ten-year lag showed little statistical significance (Appendix III). The most significant explanatory variables after the five-year lag included FORLD, Rule 11, EBOP chair and Lake Weed Society. The *pressures* category showed most statistically significant correlations, whereas the *impact* category showed low significance (Figure 5). RMA and Sewage showed weak correlations after three years only. After five years, correlations were strongest for EBOP chair, Lake Weed Society, Fonterra and FORLD for the *pressures* category. Fonterra and Kaituna catchment show strong correlation with the *state* category. The Lake Settlement variable showed no statistically significant correlations, with the highest (<0.65) correlation coefficients occurring after a five-year lag time. After ten years, the only significant correlations are Cyano blooms related to the *drivers* category, and Lake Weed Society to the *state* category (Appendix III).

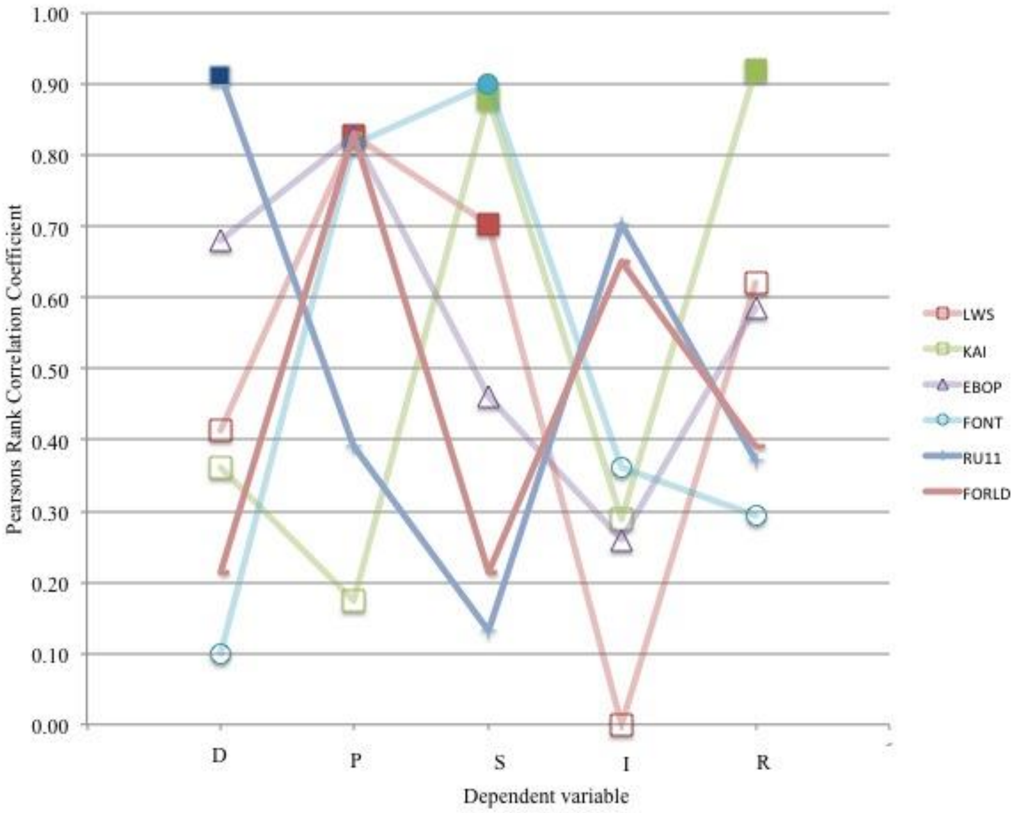

Figure 5 Explanatory variables that exhibited statistically significant relationship; plot shows the correlation coefficient for each dependent variable (D, P, S, I, R) for a five-year time lag.

2.5 Discussion

This study is the first to integrate GME regression with the DPSIR framework. With this integration we have demonstrated a novel means to analyse trends in published research on lake restoration and water quality in the study area. In the following we address how poor visibility of environmental change, social lag times from recognition of degradation to management responses, and a reactive nature of management responses contribute significantly to failures in restorative lake management.

2.5.1 Visibility of environmental degradation

In this case study, immediately visible events such as the occurrence of extensive lake weed problems (‘Lake Weed Society’) and, decades later, algal blooms (‘cyano blooms’), were most significant in influencing publication numbers and type (Table 8). While variables associated with visible environmental change (e.g. water quality, algal blooms) had a significant impact and shifted the publications’ focus towards the *impact* category (associated with a transition to management responses), other variables not so visible to the public had little impact (e.g. RMA, FORLD). TLI, the formation of the Lake Weed Society, and major lake weed growth and algal blooms commencing in 2003 had a major impact on shifting the focus to the *impact* category. Cyano blooms and Lake Weed Society also were the only variables showing significance in the 10-year timeframe in the correlation analysis, indicating that these events perhaps had a lasting impact.

These events had an immediate negative impact on the public, including visual effects, lake closures or health warnings, which can function as a motivation for both research conducted and management responses taken (Table 8). The variable of *impact* appears important to drive change, as publications within this category focus on the direct impact that degradation has on the public. This variable is not strongly populated in the dataset of publications, with only 20 out of 351. The *impact* category was also of little significance in the temporal resolution analysis. Minimal focus on the impact that water quality issues have on the public may also explain a lack of regulation that could improve management.

2.5.2 Response lag times

Response lag times are related to the time between the recognition of the degradation process of an ecosystem and ensuing management responses. Slow management, political will and the research funding might all play a role in causing these lags. In my case study, this refers to time passing between recognition of environmental change in research and public discussion, including the formation of the Lake Weed Society in the 1960s, and the first attempts to address water quality issues through nutrient load management starting in 1975, up to 15 years later (Kaituna catchment scheme), followed by further changes in 1991 (sewage upgrade). Formal regulation was not implemented until the 2008 (Rule 11); almost fifty years after lake water quality problems were first noticed.

My temporal resolution analysis shows lag times between events represented by the explanatory variables and responses within the DPSIR categories are most significant at five years. This might be associated with the nature of the scientific process, which can contribute lag times through peer-review processes and dependence on research funding availability, which is also influenced by political factors. In my study, there was no significance to whether a study was peer-reviewed or not, which suggests that the approach of studying publications is a valid approximation. Peer-reviewed publications often appear to take longer to be published, but results here indicate that a time lag of five years occurred in all publications, and was a significant temporal effect whereas the type of publication was not significant (Table 8, Appendix III).

Lag times result in separation of recognition of an environmental problem in the scientific community from regulation formulated to counter the problem. These lags are particularly evident in my study in the number of publications focused on water quality, lake ecosystem degradation and its causes in the 1970s (total n=94) (Figure 4). Sound knowledge of how water quality decline was linked to land use was established at that time (Fish 1969), but no regulation was implemented to address land use intensification until 2002 (Rule 11). Lags are also visible at a national scale, where freshwater ecosystem decline was in many cases allowed to continue with insufficient regulation (PCE 2013). Up to five decades between recognition of causes of environmental degradation and responses led to further degradation of the ecosystems.

2.5.3 Reactive management responses

As water quality decline was allowed to progress when knowledge of degradation and its causes already existed, responses tended to be reactive. Responses were often only implemented at a stage of severe, visible environmental degradation in the form of weed growth and algal blooms. Reactive management is common internationally, and often regulations take effect only when ecosystem decline has already progressed significantly, potentially leading to regime shifts and widespread algal blooms (Figure 4; Smith 2003; Scheffer and Carpenter 2003). At that point, management to prevent further degradation may be difficult due to established land use activities, nutrients already in the lake, regime shifts that may be difficult to reverse (Scheffer and Carpenter 2003), and additional nutrients already in transit due to long groundwater lags (Morgenstern and Gordon 2006; Burger *et al* 2008).

The regulatory context allowed for continuation of land use intensification, in particular dairying, in the catchments of several Rotorua lakes (Timmins and Savage 1981; Edgar 2008). As dairying is now firmly established as an industry important to the regional economy, the costs of restoring lakes have to date been borne by the entire rate-paying community through regional council funded initiatives to restore water quality, rather than directing some of the cost to those land uses that have caused the decline. Many agricultural stakeholders favour intensification of land use, which is at odds with lake restoration goals (Abell *et al* 2011). The disconnect between land use impacts and lake ecosystem change means ecosystem degradation is unnoticed by the public until a threshold consistent with publicly (as opposed to scientifically) observable deterioration is reached, such as levels of eutrophication at which blooms of algae are widespread. Within the concept of regime shift, this may coincide with the point where ecosystem degradation has progressed far enough to make restoration more difficult and costly (Scheffer and Carpenter 2003).

2.5.4 Applicability of the DPSIR framework

My results also indicate that the DPSIR framework might not be entirely suitable to describe the process of environmental degradation and its management responses. The concept assumes a linear progression from one category to the next, culminating in a management response. However, our analysis shows that such linear progression might not always take place. My correlation coefficient analysis shows that there is no temporal progression through the categories of the DPSIR framework (Figure 5, Appendix III).

Odds ratios and statistical fit of the model derived from the multinomial regression analysis give insight into the relevance of the DPSIR framework as applied in this context. The fit of the model, illustrated by the Count R^2 (54.4%, Table 8) is comparable to that obtained in other cross-sectional studies that employ entropy regression for analysis of datasets of this kind (e.g. Doole *et al* 2014). This result indicated the adequacy of the method and estimates obtained.

Some variables were expected to have a more significant impact, including trophic level and cyanobacterial blooms, when compared to the sewage treatment upgrade, which showed the highest statistical significance, even though odds ratios were comparatively lower. The ‘peer review’ and ‘Fonterra’ variables were expected to be of low significance. Both variables are not at the forefront of public awareness of environmental change, and were also expected to be of little relevance to research conducted. The RMA as a major piece of natural resource legislation in New Zealand was expected to be of at least some significance. Authorities perhaps needed time to become familiar with implementing this legislation within the intended context, leading to additional lag time.

2.5.5 Wider implications of research findings

The approach of using research publications to represent knowledge of environmental changes and management responses is limited; there are many other factors influencing these publications, including research funding, research employment numbers and the general political and socio-economic climate within a region or country. The science process itself has a significant impact on the number, type and focus of publications in this area. Scientific research processes are complex and (often commercially driven) funding available plays a major role in this (Edmeades 2004). However, we found this approach nevertheless provided useful insights into the dynamics studied, and provided the best proxy available for my study.

My study site is ideally suited to studying regulatory failure arising from long lags between recognition and action in the field of natural resource management. Natural resource regulators of Lake Rotorua were slow to recognise scientific insights into water quality issues caused by land use intensification, and changes have taken place only very recently. These hindrances are applicable to a wider spatial scale, when evaluating how resources are managed globally and how ecosystems are exploited to a point where change becomes difficult to reverse. Business-as-usual is the pathway that generally encourages further ecosystem degradation until a threshold is reached where the public is affected significantly, and governments accept the need to drive change.

A focus on economic development means freshwater management is subject to a trade-off between economic benefits and environmental costs, which creates a barrier to improvements in water quality (Edgar 2008; PCE 2013). Focus on economic aspects is often assumed as the underlying reason why ecosystem health is not prioritised (Scheffer *et al* 2000; Marsh 2012). This appears to be only one aspect of the failure to maintain or restore the health of ecosystems. Even when the protection of waterways provides economic gain, management can still be prone to failure as social lag times and the lack of visibility of environmental issues lead to inaction and complacency. Reasons for the failure to protect lakes from degradation from land use change therefore appear to include long social lags, and a lack of visibility of environmental problems, alongside the unquantified aspect that some land use changes have significant economic implications. For more effective ways of managing lakes and land use, and informing regulation under given environmental and socio-economic constraints, an integrated approach needs to be taken that considers these constraints and can help address social lags.

2.6 Conclusion

This paper provides a quantitative evaluation of how management responses are reactive and what drove failures of lake ecosystem protection and restoration. It explores how knowledge of these dynamics can be integrated into more effective management aimed at reducing human impacts such as pollution and nutrient runoff into waterways. The aim of this novel approach is not wide-scale reduction of environmental impacts compromising socio-economic aspects; rather it is supporting a new framework of policy design where existing shortcomings can be resolved by an integrative, interdisciplinary approach. This encompasses ecological knowledge, economic interests and societal constraints. Simulation and models could help visualise environmental problems to the general public, and inform decisions of policy makers. Sustainable resource management could benefit from a combination of sound scientific knowledge, educated communities and collaborative approaches to regulation that account for all stakeholders' interests.

2.7 Acknowledgements

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3 Evaluating services and damage costs of degradation of a major lake ecosystem

3.1 Introduction

Traditional approaches to ecosystem management have often failed to halt degradation and exploitation of natural resources, mostly due to economic interests (Scheffer et al. 2000). It is recognised that losses of ecosystem services can result in significant negative impacts on the economy (Farley 2012), but these are often not accounted for in policy planning across the globe (de Groot, Alkemade, et al. 2010). The impetus for protection and restoration of ecosystems has been linked to the provision of value and derived benefits from ecosystems (e.g., recreational value or the provision of food such as commercial fisheries). The concept of ecosystem services is a tool to that can be used to demonstrate the values an ecosystem provides, and the economic, social and environmental benefits derived from restoration and conservation of degraded systems. It also enables decision-makers to consider a more diverse range of ecosystem values in planning and regulation contexts.

The concept of ecosystem services is inherently anthropocentric, and is often viewed critically for its focus on human benefits and the commodification of nature (see Schröter et al. 2014 for a detailed discussion of criticisms as well as benefits). By definition, ecosystem services are focused on nature’s benefits to humans, and are classified into supporting, regulating, provisioning and cultural services (MEA 2005; Haines-Young and Potschin 2011). Ecosystem services classifications offer a more uniform methodology to compare studies and develop systematic approaches for ecosystem description, analysis and valuation (Fisher and Turner 2008; Haines-Young and Potschin 2011).

Ecosystem services can provide a tool to communicate the importance of ecosystems to sustain human wellbeing, including derived economic benefits (Bateman et al. 2011). The provision of ecosystem services is essential to sustain human wellbeing and economic profit, and the more resilient and healthy an ecosystem, the more benefits it can typically provide (La Notte et al. 2015).

Describing ecosystem services with a monetary value often implies that the derived services can somehow be substituted, and are therefore not essential (Farley 2012). However, economic valuation of ecosystems can illustrate currently undervalued services, offer additional information for decision-makers by highlighting the importance of ecosystem services that cannot necessarily be replaced by human-made services, and allow for a comparison of values of natural and converted ecosystems (Schröter et al. 2014).

A wide range of studies has included economic assessments of ecosystem services. Global natural capital has provided a total value of global ecosystem services, estimated for 1994 was twice the combined global Gross National Product (GNP) (Costanza et al. 1997). Other global analyses which have used this approach include the updated study of Costanza et al. (1997) given in Costanza et al. (2014) and studies by de Groot, Fisher, et al. (2010) and Farber et al. (2002). Valuations of freshwater ecosystems have included estimates costs of eutrophication in the United Kingdom (Bateman et al. 2013; Pretty et al. 2003) and the United States (Wilson and Carpenter 1999; Dodds et al. 2009). In New Zealand, ecosystem services and their valuation have also gained increasing attention (Dymond 2013).

While the biophysical relationships associated with ecosystem services arising from freshwater systems, including lakes, have been well studied over the past decade, economic assessments, including valuation studies, are rare. A small number of studies have addressed lake ecosystem service values (e.g., Zheng et al. 2008; Liu 2014; Bujnovský 2015). Some have also addressed the potential economic benefits of improved water quality (Wang et al. 2013; Van Houtven et al. 2014). Within a resource management and policy context, this means that values of ecosystem services derived from lakes appear to be commonly ignored, and likely underestimated.

In this study, the concept of ‘value’ is used to illustrate the benefits derived from the ecosystem and place it both within an economic and a management and policy context. The hypothesis of this research is that allowing degradation to occur may incur costs that are not currently given consideration in economic assessments, regulation or policy planning. Conservation and restoration may have valid economic justifications, including for ecological or social purposes (Palmer et al. 2006). Thus, the value of an ecosystem, and the potential additional value derived from its restoration, would provide input for cost analyses of investment in restorative activities.

While ecosystem services assessments have become commonplace recently (Haines-Young and Potschin 2011; Braat and de Groot 2012), they are rarely applied to freshwater ecosystems, in particular lakes. The objective of this study was to address this critical gap through the development of a set of steps that allows for the ecosystem services valuation of lakes. The application involves the simulation of a scenario, for which the derived values for various ecosystem services can be used to inform resource management decisions. Values were derived for the current status of the lake, and then adjusted to reflect a degradation scenario. This scenario is focused on a situation where the ecosystem has been degraded by land use impacts to a more eutrophic state. The resulting damage costs as a function of lost value can be estimated from the decrease in ecological services provided, such as nutrient cycling and habitat provision, as well as social and economic costs such as effects on recreation, property values and drinking water treatment (Pretty et al. 2003; Dodds et al. 2009). I apply this to a case study assessment of the ecosystem services provided by a large lake ecosystem, Lake Rotorua, located in the central North Island of New Zealand. This lake is recognised as being iconic at the national scale (Edgar 2008) and is a major cultural asset to Māori (Kusabs et al. 2015). I aim to contribute to the existing ecosystem services valuation studies by examining a lake ecosystem, offering a systematic valuation and damage cost assessment, and ultimately demonstrating the importance of fully functional lake ecosystems to society, as well as the potential economic losses associated with degradation.

3.2 Methodology

Three major methodological steps were developed for this lake ecosystem services valuation, which were applied to the case study site (Lake Rotorua, North Island, New Zealand). A detailed overview of all three stages and relevant input data are given in the flowchart in Figure 6.

3.2.1 Ecosystem services

The first step was the identification of ecosystem services provided by a lake based on commonly-used mapping approaches to classify ecosystem services, including the Millennium Ecosystem Assessment report (Millennium Ecosystem Assessment 2005), the Common International Classification of Ecosystem Services (Haines-Young and Potschin 2011), and other foundation studies of ecosystem services (Costanza et al. 1997; de Groot, Fisher, et al. 2010). For the case study, ecosystem services were selected after systematically reviewing the range of services that is provided by the lake (for example, drinking water is not provided by the lake studied here). The availability of data regarding quantity and valuation details were also a factor of choice (for example, no quantitative data are available to value the service of carbon cycling for this case study). Output of this first assessment stage is a list of relevant ecosystem services, complemented by a range of indicators applicable for the valuation stage (Figure 6).

3.2.2 Valuation

Ecosystem services values have been given as a range consisting of a low and a high estimate. For the value estimate, a suitable pricing method was chosen for each of the ecosystem service indicators based on previous ecosystem services valuations, including valuation studies of Costanza et al. (1997), de Groot et al. (2002) and de Groot, Fisher, et al. (2010), and studies of freshwater systems and the economic impact of eutrophication by Pretty et al. (2003) and Dodds et al. (2009). The output of the second stage was a value range for both the ecosystem services and the lake ecosystem. The function for each ecosystem service valuation was made up of a denominator quantifying the provision of the ecosystem service, such as amount of food consumed, angling days or recreational use days, and a numerator of the value such as market price of food and expenditure of tourists per day.

Valuation methods for ecosystem services are diverse. Stated preference methods involved participant surveys that can give estimates of ecosystem service values by asking participants about their preferences and choices. This method is generally survey-based, involving willingness to pay (WTP) for specific ecosystem services (Van Houtven et al. 2014), or willingness of participants to accept compensation (WTA) where an ecosystem service is lost (Patterson and Cole 2013). Revealed preference methods include WTP illustrated by money spent to benefit from a particular service, for example travel cost revealing how much people are prepared to spend to travel to a particular ecosystem (Wilson and Carpenter 1999).

Types of values that can be studied through stated preference methods include existence value, which can be used to describe the value associated with the existence of a certain species, for example through stated preference methods (Fromm 2000). Similarly, option values derived from future scenarios, and values based on ethical motivation such as bequest value from passing on natural resources to future generations, can be used to estimate ecosystem services values (de Groot, Fisher, et al. 2010). A type of value that can be established using state preference methods is replacement cost, which measures the value where the loss of an ecosystem service necessitates a compensatory replacement service, such as engineered water treatment (de Groot et al. 2002). Hedonic pricing is a further indirect value that can be used to value ecosystem services, such as amenity values or aesthetic services, by studying property prices to reflect a human value (Leggett and Bockstael 2000). Where ecosystem services contribute to incomes, such as those based on fisheries, income can be used as an indirect market valuation (de Groot et al. 2002).

Examples of direct market values for ecosystem services include the pricing of the amount and types of food and raw materials consumed, extractive uses such as drinking water consumption, or non-extractive uses such as energy gained from hydropower generation (Costanza et al. 1997). The use of ecosystems in a recreational capacity can also be priced directly by assessing the expenditure associated with recreational usage (using revealed preference methods) (Dodds et al. 2009).

3.2.3 Damage costs

Damage cost describes the value that can be lost when an ecosystem is allowed to degrade, leading to a reduction in ecosystem services. The exact quantification of the loss of ecosystem services provision associated with degradation is not well studied (Braat and de Groot 2012). For the final stage, I assessed damage costs based on effects of lake eutrophication. Eutrophication is a response to increased levels of nutrients, leading to an increase in chlorophyll *a*, occurrences of algal blooms and weed growth, and a decrease in water clarity and bottom-water dissolved oxygen (Harper 1992). These changes are expected to reduce ecosystem services (Carpenter et al. 1998; Pretty et al. 2003; Dodds et al. 2009) but may not act in isolation because introduced species, for example, can also affect the provision of ecosystem services (Rothlisberger et al. 2012). The function of the damage cost assessment was comprised of the initial ecosystem service value, and a reduction factor that was based on the associated degradation and resulting changes in clarity, algal bloom occurrences, and overall water quality of the lake.

To calculate damage costs for each ecosystem service it is necessary to quantify the relationship between degradation and ecosystem service provision. This is specific to the lake and depends on characteristics of the lake such as size, depth and morphology. It also depends on the resilience of the lake to water quality change, and the vulnerability of the flora and fauna in it. Modelled scenarios, specific value loss studies and technical reports of the study site can be used to assess damage costs (e.g., (Butcher et al. 2000; Marsh and Woodham 2011; Hamilton et al. 2012). The opinion of experts familiar with the study site can also be consulted to overcome gaps in knowledge within published literature (Hamilton and Parparov 2010).

The outcome of the damages cost step at the third stage is a specific percentage reduction in value estimated for each service. The last two steps of the third stage are the application of the reduction factor to the initial value and an estimate of a total damage cost given as a range of value loss. As for the valuation, a low and a high value of each reduction factor are applied to the respective low and high value of each ecosystem service. The value loss range for the entire ecosystem is calculated based on the low and high value for each individual ecosystem service.

I based the damage cost assessment on a modelled scenario of water quality change associated with continued degradation, using expected water quality levels, Secchi depth, and algal bloom occurrences for the year. The degradation scenario was taken from Hamilton et al. (2012) that simulated the effects of business-as-usual land use management on water quality levels in 2032. Climate change was also accounted for (see details in section 3.3.4). Compared to other modelling exercises published in literature, model performance was assessed to lie within an acceptable model error range for the purpose of the predictions made in the modelling study. I expect a satisfactory level of confidence in the predictions of water quality changes used as the basis for the damage cost assessment.

3.2.4 Valuation and damage cost uncertainties

Values of the ecosystem services were based on the best available data and estimates. However, there were unknowns associated with insufficient or outdated data. The applied pricing methods are also subject to uncertainties, including limitations of revealed preference methods in their applicability to actual WTP (Farley 2012). The assessment was systematically checked for potential sources of double counting of ecosystem services values. Errors of double counting in this study were limited by using site-specific data and analysing values for a particular year, as well as the careful and simplified choice of ecosystem services used for this study.

For damage costs there may be insufficient data to accurately quantify the impact of degradation on ecosystem services and ensuing value losses. Therefore, the reduction factor described in section 3.3.4 should be considered as an estimate based on the best available knowledge. Valuations of ecosystem services are commonly subject to complex relationships between ecological processes and their descriptions through economic concepts (de Groot et al. 2012; Johnson et al. 2012). In this study I address the impact of uncertainty using a range of indicators and relevant proxies that can be valued (Pretty et al. 2003; Dodds et al. 2009; de Groot, Fisher, et al. 2010).

3.3 Case study

3.3.1 Study site

I used Lake Rotorua as the case study for this valuation study (Figure 7). It is a shallow polymictic lake of volcanic origin located on the central volcanic plateau of the North Island of New Zealand. The lake formed around 240,000 years before present and is situated in a group of lakes of varying characteristics and trophic states. Lake Rotorua has an area of 80 km^2, with a mean depth of 10 m and a maximum depth of 45 m. The catchment area of the lake is approximately 450 km^2, dominated (> 50%) by pastoral land use including grass-based dairy farming, dry stock farming and forestry (Scholes 2009). The lake is classified as eutrophic and has historically been impacted by both point-source and diffuse nutrient inputs (Burns et al. 2009). While point sources have largely been eliminated, including land-based treatment of municipal wastewater (Rutherford et al. 1996), diffuse pollution is still a major driver of eutrophication of the lake. Invasive aquatic weeds, occurrences of algal blooms and continued high rates of external nutrient loading impact on the health and resilience of this lake ecosystem. As one of New Zealand's major iconic lakes, Lake Rotorua is of significant economic value and supports a major tourism industry (Edgar 2008). The lake is culturally significant and is a tāonga (treasure) to tangata whenua (indigenous Māori people), and has important spiritual values for them (Kusabs et al. 2015).

3.3.2 Ecosystem services

Ecosystem services that were not relevant to my case study were eliminated to arrive at a final list representing each provisioning, regulating and cultural service. Five ecosystem services were included in this valuation (Table 9). The list of identified ecosystem services is comprised of habitat provision, food, nutrient sequestration, amenity and aesthetics, and recreation. Table 9 summarises ecosystem services relevant to freshwater ecosystems, studies that have assessed them and whether they were valued in my case study. Relevant services were those that were in fact provided by the case study lake, and that offered some way of quantification with regards to levels of provision that were suited to the case study (e.g. carbon cycling was disregarded as no concrete data on this service was available).

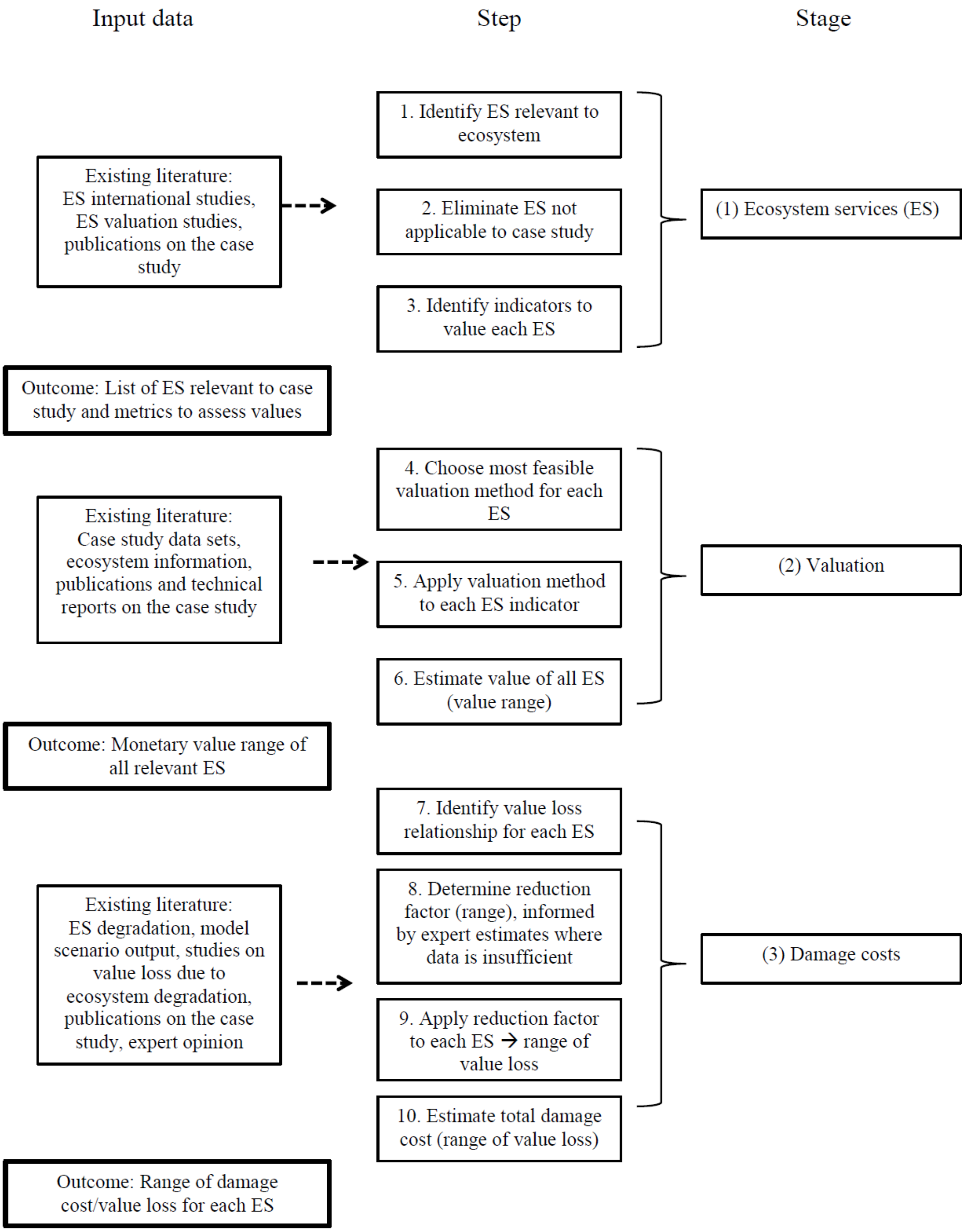

Figure 6 Flow chart outlining the input data for the assessments (left-hand side), the methodology used (centre), and the three key stages of the assessment. ES is ecosystem services.

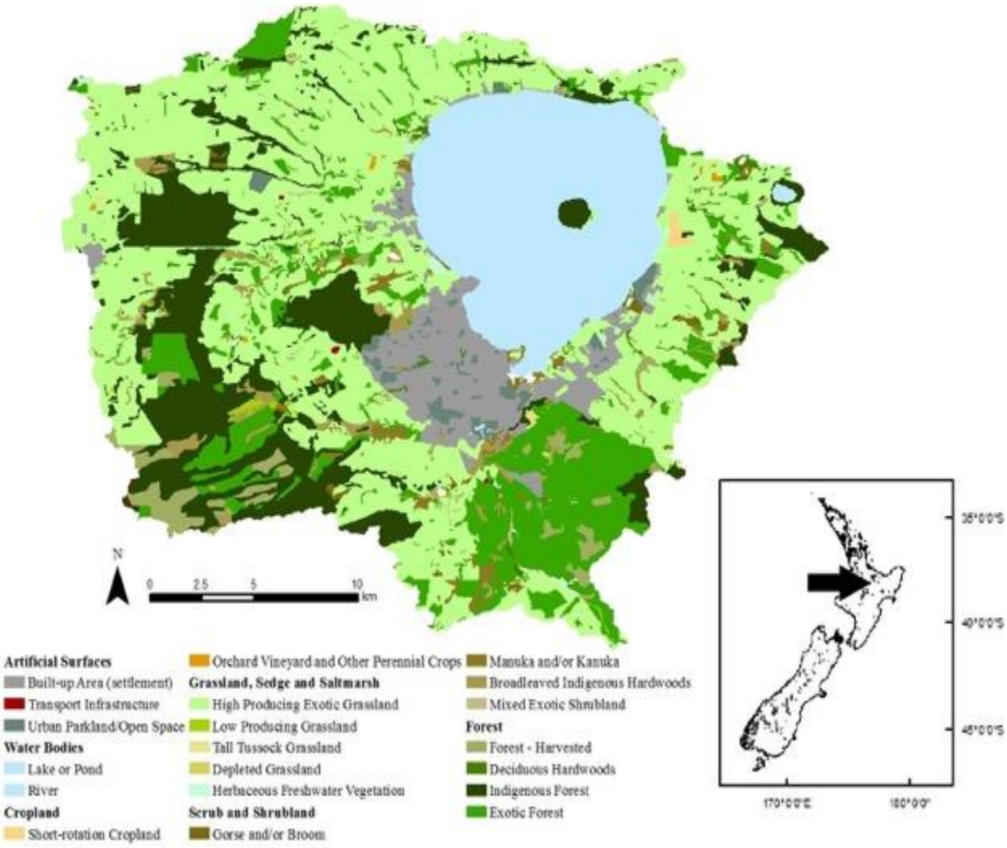

Figure 7 Map of Lake Rotorua, its catchment and location within New Zealand (black arrow).

Table 9 Ecosystem services of freshwater ecosystems at international scale.

Category	Ecosystem Service	Study example	Scale	Inclusion[1]
Provision	Food	Costanza et al. (1997)	Global	Yes
	Water	de Groot et al. (2012)	Global	N/A[2]
	Habitat	Dodds et al. (2009)	U.S.	Yes
	Hydropower	Pretty et al. (2003)	England/ Wales	N/A[2]
Regulation	Carbon sequestration	-		No
	Nutrient sequestration	Pretty et al. (2003)	England/ Wales	Yes
Cultural	Amenity and aesthetics	Dodds et al. (2009)	U.S.	Yes
	Science and education	(de Groot et al. 2002)[3]	Global	No
	Spiritual values	-		No
	Recreation	Dodds et al. (2009)	U.S.	Yes

[1] Included in valuation of the case-study lake

[2] Service not provided by the case-study lake

[3] Not freshwater ecosystem-specific

3.3.3 Valuation

A detailed description of input data and calculations behind the methodological steps for both lake ecosystem service values and potential damage cost estimates is given in Table 10. Any values from years other than 2012 were inflation adjusted to 2012 using the inflation calculator of the Reserve Bank of New Zealand (RBNZ 2015). All values are given in New Zealand dollars, unless otherwise stated. Values were derived using site specific values where possible, in order to provide the most representative and accurate estimate for the study lake. Benefit transfer was used for the habitat provision service, where a nation-wide study was applied to inform values of the lake (Patterson and Cole 1999).

Habitat provision was valued using an existing study of the passive intangible value of New Zealand’s lake ecosystem habitats, which used indirect pricing methods based on existence values and provided value for a lake ecosystem per hectare (Patterson and Cole 1999). Food provision was calculated by using direct market pricing of the amount of wild food consumed out of Lake Rotorua. Median consumption of wild food was studied by Phillips et al. (2011), and market prices were collected for 2012. The number of people consuming wild food in the area was estimated to be 20,000-25,000, or about 30-35 percent of the local population. Nutrient sequestration was valued using the indirect method of replacement cost based on artificial treatment to remove excess nutrients. In Lake Rotorua these treatments include artificial wetlands and dosing of two inflows with Alum. A lower estimate is based on the baseline 2012 spending and a higher estimate based on a amount budgeted for the year for Lake Rotorua (Burns et al. 2009; McIntosh 2012).

Aesthetic value was estimated using the value of houses in the Lake Rotorua region, adjusted to the value of houses in the study year (2012). The value was derived using data from the Real Estate Institute (REINZ 2012) and New Zealand census data for the region (Statistics New Zealand, 2013). Based on a study of waterfront properties in the Rotorua region, which used hedonic prices to identify the value of aesthetics of the lake on property values (Marsh and Woodham 2011), a percentage (7%) was identified to represent the aesthetic value of the lake. The annual capital value of the asset (Lake Rotorua houses) was then calculated using the value of 7% of all houses. This was done using the standard NZ Treasury discount rate of depreciation of housing stock over 50 years (7% depreciation). The result was the annual value of the total aesthetic value captured in the total housing stock across the Lake Rotorua region. Recreational values were calculated by direct pricing of both angling usage and tourism expenditure related to lake use (Butcher et al. 2000; Bell et al. 2004; Unwin 2009).

3.3.4 Damage cost

Damage cost estimates were based on an existing modelled water quality scenario for Lake Rotorua, derived from a one-dimensional hydrodynamic-ecological lake model (DYRESM-CAEDYM) to calculate future water quality for a range of management scenarios (Özkundakci et al 2012). The indicator of Trophic Level Index (TLI; Burns et al. 1999) describing lake trophic state was used to define the resultant water quality for each scenario. The scenario chosen here was a simulation of water quality levels in 2032, assuming a continuation of the current land use and no mitigation measures undertaken. The model does account for the effect of climate change (Hamilton et al. 2012). Water quality results used for the damage cost scenario also included Secchi depth (representing water clarity) and concentrations of cyanobacteria which may strongly impact on lake status and use through formation of surface algal blooms. The resulting water quality results associated with the scenario were then used as basis to calculate potential value losses of ecosystem services.

Table 10 Detailed summary of ecosystem service value estimation: pricing methods, indicators and valuation details.

Ecosystem Service	Pricing	Indicator	Valuation details	Data sources	Applicable methods and data in peer reviewed literature
Biodiversity	Existence value	Passive value of biodiversity	1994 value of lakes per unit area (ha^{-1}) 2012 adjusted value of Lake Rotorua (8,000 ha)	Patterson & Cole 1999	Greenley, Walsh, and Young 1981; Kealy and Turner 1993; McClelland and Schulze 1992; Loomis et al. 2000
Nutrient sequestration	Replacement cost	Nutrient removal spending	2012 fund for nutrient removal costs	BOPRC annual report, 2012/2013	Pretty et al 2003
Amenity & Aesthetics	Hedonic values	Property values	Annual capital value of the Lake Rotorua housing stock, depreciated at 7% over 50 years, with 7% of the total value representing the aesthetic value captured in housing stock	Marsh & Woodham 2011, REINZ 2012, Statistics New Zealand 2013	Dodds et al. 2009
Food	Market price	Median consumption rates per person, estimate of people consuming wild food	Market price of food gathered for 2012, median consumption rates for Rotorua	Phillips et al. 2011, market prices, statistics	De Groot et al. 2012
Recreation	Income/production	Angling usage expenditure Tourism spending based on economic impact survey	Lake Rotorua angling days 2012 based on 2007/2008 data, adjusted to angling day change for the Eastern region; expenditure per angling day (low estimate); 2000 tourism expenditure survey, 50% of spending from lake front survey, 23% of activities related to the lake Adjusted to 2012 values	Bell et al. 2004, Unwin 2009 Butcher, Fairweather, and Simmons 2000	Pretty et al. 2003; Dodds et al. 2009; De Groot et al. 2012

The reduction factor for biodiversity was estimated at 5-15 percent change in value. This was an estimate based on knowledge that many native species are susceptible to changes in water quality from nutrient enrichment and deoxygenation of the bottom waters, such as freshwater crayfish or kōura (*Paranephrops planifrons*) (Kusabs et al. 2014) and freshwater mussels or kākahi (*Echyridella menziesi*). The susceptibility of these key species to water quality changes was used as a proxy for impacts of eutrophication on biodiversity, but a low rate of change was chosen to provide a conservative figure.

The ecosystem service of food provision is linked to similar considerations of sensitivity of native species, including kōura and kākahi. Wild food consumption from lakes in New Zealand has decreased steadily from historic levels, and water quality is assumed to play a role in the presence, consumption safety and availability of the food (Tipa et al. 2010). A reduction factor of 5-15 percent was chosen for effects of water quality on food provision. For nutrient sequestration, the expected response of the lake to additions of nutrients is not expected to be linear, and is therefore difficult to estimate in terms of economic value loss. For this ecosystem service, costs were based on the budget currently directed at removing excess nutrients from the lake. This cost was estimated to remain largely unchanged in forthcoming years (Lamb 2015), so no change in value was assumed for this ecosystem service.

Several studies have found that water quality decline can affect property values. For Lake Rotorua, a 1-m change in Secchi depth (corresponding to the scenario used here) can lead to a loss in value of house prices of seven percent (Marsh and Woodham 2011). Other studies have found that loss of water clarity can incur damage costs to house prices of up to 20 percent (Pretty et al. 2003). Recreational use is also significantly affected by water clarity and growth of weed and algae. With occurrences of algal blooms, tourism use of the lake has been shown to decrease by 23-50 percent (Bell et al. 2004); this percentage change is taken as the corresponding percentage change in tourism value.

3.4 Results

3.4.1 Ecosystem services values

Results of the ecosystem services valuation include a list of five ecosystem services where sufficient data were available to value the services of Lake Rotorua. Other services, such as carbon cycling and cultural values, were also identified as relevant to the lake (Table 9), but excluded from the valuation for reasons elaborated in section 3.5.2. For each ecosystem service, a brief description of the indicator is given Table 11. For the valuation results, a low and a high estimate for each individual ecosystem service for the year 2012 is presented. Lastly, a total value comprising all ecosystem services that were assessed for 2012 was calculated summing all individual values, while specifying low and high estimates to arrive at a value range.

The habitat provision value for Lake Rotorua calculated from the previously established value of lakes in New Zealand was $15.1 million per year. Food provision was calculated as a low and high value at $6.3 and 9.4 million, respectively. Nutrient sequestration using nutrient removal costs was valued at a range of $4.1-13.3 million. The amenity value based the Net Present Value used to derive the value of lake water quality to house prices for Lake Rotorua was estimated at $21.4-34.1 million per annum, and recreational values using angling usages were calculated at $4.0-7.8 million dollars. Tourism expenditure associated with activities on Lake Rotorua was estimated at $47.9-73.5 million and total recreational value comprising recreational use was valued at $52.0-81.3 million per year. The value ranges of the selected individual ecosystem services are presented in Table 11.

The total value of Lake Rotorua for the year 2012 was $99.7-145.0 million. The category of ecosystem service contributing the highest value to the lake value was recreation, followed by amenity value and biodiversity. The lowest values were contributed by nutrient processing services of the lake.

Table 11 Ecosystem services, valued indicators and range estimates of ecosystem service values for Lake Rotorua. Figures in millions of New Zealand dollars.

Ecosystem Service	**Indicator**	**$ Low Estimate**	**$ High Estimate**
Biodiversity	Intangible biodiversity values	15.1	15.1
Food	Wild food consumption	6.3	9.4
Nutrient processing	Nutrient removal cost	4.1	13.3
Amenity and aesthetics	Percentage of lake property sales	21.4	34.1
Recreation	Recreational lake usage	52.0	81.3
Total		**99.7**	**145.0**

3.4.2 Damage costs estimates

A damage cost assessment was calculated for degradation of Lake Rotorua due to eutrophication. The scenario of further degraded water quality was informed by simulations from the DYRESM-CAEDYM model for the year 2032 (Özkundakci et al. 2012), which were then assumed for 2012 to inform the damage cost scenario as a direct comparison to the 2012 value figures (Table 10). For Lake Rotorua, the trophic level in 2012 was indicated by a TLI of 4.1 (Scholes 2013). As an indicator of potential change towards a more degraded state, the model scenario gave a TLI of 4.8[2]. It also predicted approximately 25 days each year (seven percent of all days) of low concentrations (<2 mg L^{-1}) of bottom-water dissolved oxygen, and 139 days each year (38 percent of all days) with elevated levels of cyanobacteria (>20 µg L^{-1} as a cyanobacteria contribution to chlorophyll *a* concentration (Hamilton et al. 2012)).

Reductions were applied to the ecosystem services values to calculate potential value losses of a degraded lake compared to the 2012 status (Table 12). The value loss estimated for biodiversity was $0-2.3 million per year when the lake is in a higher trophic state (TLI 4.8 compared to 4.1). For food provision, the estimated reduction in value was $0.3-1.4 million per year. In the 2032 scenario, no change in value was estimated for nutrient sequestration (see section 3.3.4). The potential damage cost incurred for loss of amenity and aesthetic value was calculated at $1.1-3.8 million. Recreational use is significantly affected by changes

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[2] The scale of TLI levels ranges from 0.0 (ultra-microtrophic) to 7.0 (hypertrophic). A eutrophic condition is indicated by a TLI level between 4.0 and 5.0, and hypertrophic between 5.0 and 6.0. TLI is calculated using equations relating to the four variables of chlorophyll *a*, Secchi depth, total phosphorus and total nitrogen.

in water clarity and weed and algal growth. The scenario yielded predictions of close to 40 percent of days each year that would be affected by elevated levels of cyanobacteria (>20 µg L^{-1} as equivalent cyanobacteria chlorophyll concentrations). The recreational value loss was estimated to be in the range of $12.0-40.6 million for 2032.

The total potential value lost due to continued water quality decline, based on changing the lake trophic state from TLI of 4.1 (just within the eutrophic range) to 4.8 (close to a hypertrophic state) was estimated at $14.5-50.9 million per year.

Table 12 Reductions in ecosystem services of Lake Rotorua for a degradation scenario involving a shift in the Trophic Level Index from 4.1 to 4.8. Low and high estimates of damage costs (low – high) are given in millions of New Zealand dollars.

Ecosystem service	Reduction factor	$ Low estimate	$ High estimate
Biodiversity	0-15%	0	2.3
Food	5-15%	0.3	1.4
Nutrient processing	0	0	0
Amenity and aesthetics	7-20%	1.5	6.8
Recreation	23-50%	12.0	40.6
Total		**14.5**	**50.9**

3.5 Discussion

3.5.1 Case study results and context

Ecosystem services provided by Lake Rotorua in 2012 were valued at approximated $100-145 million per year, with potential further degradation of the lake leading to a loss in value of $14.5-51 million per year (Table 11, 12). The most important value was associated with the cultural ecosystem service of recreation, which illustrates the importance of the lake for local and visitor recreation, and tourism generally. The aesthetic value, a cultural service, was also found to be highly important in this study and reflects Lake Rotorua's value as an iconic destination (Edgar 2008). Lastly, biodiversity was also identified as having high value, reflecting the importance of the lake for iconic threatened species, such as kōura (freshwater crayfish) and kākahi (freshwater mussel) (section 3.3.4).

The use of an ecosystem model also provides insight into the potential damage costs incurred if the lake is allowed to degrade further. At the same time, there is value that could potentially be gained if restoration efforts were successful in reducing the trophic status of the lake and therefore improving its health and biodiversity. Estimates of both value and damage costs are therefore important to help inform future management decisions, as well as directions of policy planning. The value gained by restoration and the resulting water quality improvement can serve as justification for ongoing investment in restorative actions and conservation measures. In contrast, allowing degradation to occur appears to incur costs that are not always accounted for in economic assessments of resource management.

Given both recreational and amenity values were highest ranking in this study and incurred the highest potential value loss, continued degradation is expected to have a major impact on visitors and hence the tourism industry, which is a major contributor to Rotorua's regional economy; and evidence shows degraded lakes are potentially less attractive to visitors who may choose to travel to one of the other Rotorua lakes, for example (Butcher et al. 2000). The results are applicable to other lakes in New Zealand (and globally), which are affected by eutrophication, and perhaps to a wider set of ecosystems which are affected by degradation from unsustainable resource use. On a per-area basis, lakes have been shown to be valuable ecosystems compared with other land-based ecosystems. For example, in a New Zealand study the value of lakes per hectare ($17,159) was significantly greater than for agricultural land use ($1,188) (Patterson and Cole 2013); a comparison which bears significance when considering the agricultural impact on freshwater ecosystem health locally (Abell et al. 2011) as well as globally (Foley et al. 2005). However, there is a vast difference in total area of lakes (0.3 million hectares) and agricultural land use (10.5 million hectares) in New Zealand (Patterson and Cole 2013).

The valuation figures are similar to those of other lake ecosystems across the world. The Costanza et al. (2014) follow-up study of the previous valuation of the world's ecosystem services (Costanza et al. 1997) gave estimates of the value of lake ecosystem services of US$12,512 per hectare for 2011 (Costanza et al. 2014). Adjusted to New Zealand dollars and for the year 2012 yields a value of approximately $134 million for the area (c. 8,000 ha) of Lake Rotorua. The value estimated in my study was $100-145 million for the year 2012 (Table 11).

The alignment of this value with the global study of Costanza et al. (2014), suggests that it is possible to scale ecosystem service valuation results from a regional to a global scale, and vice versa. The value furthermore aligns with findings from a New Zealand study (Patterson and Cole 2013) which estimated the value of an ecosystem of equivalent area to Lake Rotorua at approximately $136 million.

3.5.2 Limitations of the case study

Carbon sequestration as a potential ecosystem service of a lake was not valued in this study for two reasons. Lakes can function as ‘carbon sinks’ when atmospheric CO_2 is taken up by the water column and sequestered in lake sediments in particulate organic form (Ferland et al. 2014). Lakes may also emit carbon to the atmosphere (as methane and carbon dioxide), particularly when a lake is not stratified (Tranvik et al. 2009). In the case of Lake Rotorua, the balance between carbon taken up by the water column and sequestration in the sediment, and the amount emitted at the lake surface is estimated to be similar, meaning that there is no net carbon uptake. Currently unquantified geothermal emissions of carbon from this lake are a complicating factor to such calculations (A. Santoso, *pers. comm.*). Therefore, the economic contribution that a lake makes as a carbon sink remains unquantified and is not commonly valued in studies of ecosystem services of lakes around the world. Including the role of carbon cycling in ecosystem services assessments would add a further dimension to the economic value of a lake, but the primary data collection required both for the biophysical quantification and economic valuation of this service were beyond the scope of this study, which set out to develop a systematic approach to value ecosystem services of lakes first and foremost However, ecosystem services from carbon cycling are expected to make a significant contribution an economic value assessment (Stern 2006).

Spiritual values provided by the lake were not valued in monetary terms in this study. Spiritual values of water bodies such as Lake Rotorua that are of importance to tangata whenua (indigenous people) include the maintenance of the mauri (life giving principle) of the water, the provision of key species for mahinga kai (customary food gathering for sustenance and gifting), and the preservation of the mana (authority, honour, prestige) (Kusabs and Shaw 2008). A valuation of these spiritual values would be highly complex due to the specific cultural context that means each ecosystem has its own individual set of values based on the unique background and history of its people (Tipa and Teirney 2006; Te Aho 2010; Robb 2014). Even for example, the quantification of values such as mauri itself is at times seen as diminishing its significance, and would therefore be culturally insensitive, if not a prohibitively controversial, exercise (Robb 2014). Lastly, any study of these spiritual values should be based on the principles of mātauranga Māori (traditional indigenous knowledge) in order to fully understand and capture the importance of these values. By definition, mātauranga Māori research is conducted by suitably mandated Māori researchers who hold mana whenua status (Henry and Pene 2001). Thus, the quantification and valuation of spiritual values in the context of this study would be inappropriate and culturally insensitive. It is, however, suggested that future research should be conducted in this area guided by the principles of the local Te Arawa Cultural Values Framework, which outlines the importance of the Māori worldview (Te Arawa Lakes Trust 2015). This research direction would provide broader understanding of the cultural and spiritual values of the lake.

3.5.3 Uncertainties and caveats

Studies of ecosystem services and valuations are subject to uncertainties around both the provision of the services and meaningfulness of valuations (Johnson et al. 2012). Freshwater values and damage costs also have inherent uncertainties associated with paucity of data and estimates that may be based on the use of proxies to represent economic values (Pretty et al. 2003; Dodds et al. 2009; de Groot, Fisher, et al. 2010). The complex nature of ecosystem dynamics, especially the common responses and irreversibility of some forms of degradation (Carpenter et al. 1999), discussed in section 3.3.3, lead to difficulties in applying conventional economic concepts that do not generally recognise emergent ecosystem properties such as thresholds and tipping points (Hipsey et al. 2015). Degrading ecosystems do not respond to environmental pressures in a linear manner, and changes in state are subject to such threshold responses. Equally, the restoration process is affected by possible alternative stable states of the ecosystem, and hysteresis meaning that even if environmental pressures are reduced to an earlier state, the ecosystem might still not recover to the original stable state (Scheffer and Carpenter 2003; Schindler 2012). Uncertainties can be addressed by a use of a range of values rather than a single figure, and using the best available data for estimates of ecosystems and their services' contribution in economic terms (e.g., Costanza et al. 1997).

I acknowledge some uncertainties arising from the methods and estimates provided in my study, and these should be viewed within the context and objectives of the study. Hedonic pricing may have a tendency, for example, to focus on a particular sector of society and other studies (e.g. Boeraeve et al. 2015) point out that this method is limited to only a part of society that is able to purchase or sell a house with a desirable view. The valuation of biodiversity based on existence values is also limited in its capacity to represent the inherent complexity and relationships that characterise whole ecosystems (Mace et al. 2012). It is also important to consider that this type of assessment uses a diverse range of valuation methods of both use and non-use values, which limits the explanatory power of the resulting value estimate. The intention of this study was to develop a systematic approach to illustrate the economic value of a lake ecosystem from a perspective not commonly valued and to test the extent of damage costs arising from compromising its ecosystem services through ecological degradation (Boeraeve et al. 2015). The knowledge that lakes provide valuable ecosystem services can inform policy, planning and regulatory decision-making, as well as raising public awareness of the significance of this lake ecosystem in economic, ecological and social contexts.

3.5.4 Complexities of the Lake Rotorua case study

Included in value reductions for Lake Rotorua were loss of biodiversity and food provision value with lake eutrophication. Kōura (freshwater crayfish), for example, may be reasonably resilient, as there is regular occurrence (7 % of the time) of low oxygen levels (<2 mg L^{-1}) in the bottom water habitat in Lake Rotorua. The morphology of the lake consists of a low ratio of bottom-water to total lake volume (i.e., extensive shallow areas) and therefore offers a refuge when anoxic conditions are present in the bottom waters (Kusabs et al. 2015). Furthermore, the early stages of eutrophication can stimulate greater fish production (Carpenter et al. 2001; Kusabs et al. 2014), although Lake Rotorua is already considered to be at a level where further nutrient enrichment (i.e. eutrophication) would have detrimental effects on the fishery in the lake. For nutrient sequestration, the response of the lake to nutrient enrichment can be seen as a threshold, where the system may be relatively resilient up to a certain point, then release large amounts of phosphate from bottom sediments when they become anoxic (e.g., up to 1000 times faster than when oxygen is present; Horne and Goldman 1994). Anoxic conditions in Lake Rotorua can induce high rates of N and P release from the sediment (Burger et al.

2008). The decline of water quality in the modelled scenario for Lake Rotorua, particularly the predicted increased frequency of low oxygen (<2 mg L^{-1}) in the bottom waters, which triggers large nutrient releases, is consistent with other modelling studies of the lake (Burger et al. 2008).

For Lake Rotorua, the effect on the long-term provision of ecosystem services from continuous Alum dosing of two inflows adds complexity to the applicability of the potential value loss scenario. Alum dosing, which commenced in the Utuhina stream in 2006, and the Puarenga stream in 2010, has led to a reduction in TLI over recent years. This response is due to a reduction of phosphorus in the water column as it flocculates, sediments out and becomes buried in the bottom sediments (McIntosh 2012). An improved ecological status of Lake Rotorua, signalled by a TLI <4.0, might offer benefits in terms of ecosystem service provisions and values, rather than further value loss. However, as with any restoration scenario, there are also costs involved in restoring and maintaining the trophic status to the desired level (Abell et al. 2011), as well as potential risks from excessive Alum dosing (Van Hullebusch et al. 2002; Tempero 2015).

3.5.5 Future research options

Publications on freshwater ecosystem services have increased rapidly in the past decade. In 2005 there were only 15 publications while in 2014[3] the number had increased to nearly 100. At the same time, valuation studies have remained relatively rare (Zheng et al. 2008; Liu 2014; Bujnovský 2015). My study has demonstrated the application of a set of valuation and damage cost assessment steps, which can help to address this gap by providing a systematic approach to value lakes and freshwater ecosystems across the world and potentially support restoration costs.

My study has shown that many challenges arise when an integrated valuation system using a variety of methods (biophysical modelling, ecosystem services modelling and various valuation techniques) is applied for the assessment of a specific case study ecosystem, i.e., Lake Rotorua. These challenges include aspects of incompatibility between biological and economic systems (based on threshold responses, nonlinearities and irreversibility), and uncertainties due to lack

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[3] Numbers derived from an ISI Web of Science database query using the keywords “lake AND ecosystem AND service*” (including years 1950-2014).

of data and knowledge of how biological systems can change with respect to ecosystem service provisions. Uncertainties that could not be overcome were mitigated by various commonly used techniques (section 3.5.3). While any valuation of ecosystem services is expected to be affected by uncertainties and will therefore be subject to controversy and criticism, I maintain that such a quantitative assessment is still a meaningful exercise to show the importance of ecosystems in supporting social and economic benefits to society. In this study, I showed that losses of ecosystem services can have a negative impact on the economy, and that investment in restoration therefore has economic and ecological benefits.

Table 9 and10 show the type of information required to conduct a valuation and damage cost study. Further work is required to scale my results from a lake-specific case to include lakes within the local, regional or global context. Economic considerations are at the forefront of many management and policy decisions, and by communicating the economic value of ecosystems, restoration and conservation practices will more likely be implemented.

A range of additional values were not covered in this study due to the scope of the research and the data that was available. While nutrient processing as an ecosystem service was included in the valuation, effectiveness of the treatment cost in terms of nutrients removed from the water column was not addressed. For example, denitrification processes that remove nitrogen from the water column of Lake Rotorua (Bruesewitz et al. 2011) were not quantified as part of this valuation. Science and education were not included in the valuation due to lack of quantitative information and uncertainty of the direction of change; deteriorating water quality could incur additional science expenditure as solutions were sought. Aspects of public health were not directly addressed, albeit these were implied in the context of the occurrence of algal blooms, which have potentially toxic effects detrimental to public health (Falconer 2012; Carmichael 2013). The effects of water quality on the provisioning of water in downstream ecosystems, including other lake, river and estuary ecosystems, was not considered in this study as the scope here was the lake ecosystem itself. Lastly, the quantification of the damage costs did not consider whether any of the ecosystem value losses are potentially irreversible, which could be framed as part of an option value analysis. With additional information, hindcasting analysis linking historical water quality changes with ecosystem services provisions and valuations could be useful. The role of carbon markets could also be explored in order to consider the important role of lakes in carbon cycling as an ecosystem service (Tranvik et al. 2009).

3.6 Conclusion

This study provides a systematic approach to assess the value of a lake ecosystem and estimate potential damage costs associated with degradation of its water quality. The value estimated for Lake Rotorua in 2012 was in the range $100-145 million. Potential damage costs caused by degradation of Lake Rotorua were calculated for 2012 at $14.5-51 million, with a mean of $30.5 million ($3,812 ha^{-1} yr^{-1}). These values are subject to a range of uncertainties. Levels of uncertainty are comparable to existing studies that have employed a variety of techniques to value ecosystem services.

My results show the economic importance of the Lake Rotorua lake ecosystem, as well as the potential damage costs when the ecosystem is allowed to deteriorate. Valuations and damage cost assessments for Lake Rotorua can be applied more widely and provide a useful basis towards better integration of ecosystem values in management decisions and policy planning. The work provides a basis for a cost-benefit analysis, and justifies continued investment in restoration. It can also inform a more integrated assessment of the cost-effectiveness of various mitigation and lake management strategies, including planning options for land use in lake catchments.

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Chapter 4

4 Costs and effectiveness of land use change and mitigation options to reduce nutrient loads to a eutrophic lake

4.1 Introduction

Water quality decline in lakes, as with many other forms of ecosystem degradation, can be difficult to manage. A major challenge is the trade-off between economic growth and land use in a catchment, and the integrity and resilience of the lake ecosystem (Foley et al. 2005). Resource managers are required to balance economic interests with public expectations regarding lake ecosystem health and water quality. Management decisions therefore require consideration of the economic implications of a range of the decisions, as well as their consequences pertaining to the health, integrity and resilience of lake ecosystems. This nexus of economics and ecology is inherently difficult to address (Carpenter et al. 1998; Scheffer et al. 2000; Iwasa et al. 2007), but its integration is required if policy decisions are to result in the implementation of the most cost-effective options that optimise the trade-offs between economic and environmental outcomes for both catchments and lakes, offering solutions that restore lake water quality whilst providing options for economic growth in the catchment.

Excess nutrient input from catchments, and accumulation of these nutrients within lakes, accelerate the natural process of eutrophication, thereby exerting pressure on these ecosystems (Schindler 2012). To improve ecosystem health, lake restoration methods have addressed water quality problems directly within lakes, as well as externally through catchment nutrient load controls (Spears et al. 2014). Different land uses vary in their areal nutrient yields and some (e.g., intensive agricultural production lands) have been widely implicated in eutrophication of lakes (Baron et al. 2002). Many lake restoration efforts have included a combination of management of nutrients within lakes, as well reductions in nutrient loss from catchments by changing land use and implementing best management practices (Baron et al. 2002; Giri et al. 2012).

Lake water quality management requires consideration of both external (surface water and groundwater inflows) and internal (bottom-sediment) nutrient sources. Degradation of water quality is often associated with anoxia which leads to chemical reduction reactions that increase phosphorus releases from metal cations in bottom sediments and result in build-up of ammonium as nitrification is hindered (Søndergaard et al. 2003; Burger et al. 2008). Rates of internal loading can vary between lakes with different characteristics (e.g., trophic state), as well as throughout the year depending on weather conditions and water column stability (Soranno et al. 1997; Burger et al. 2008).

A range of restoration measures has been tested and applied to restore lakes that are subject to anthropogenic eutrophication. In-lake restoration is aimed at removing or inactivating existing nutrient stores, often including large pools of nutrients accumulated as a historical legacy of excess inputs from the catchment (Jarvie et al. 2013). For example, in-lake methods have included removal of lake weed or algae (Hofstra and Clayton 2014), dredging of bottom sediments (Reddy et al. 2007), artificial oxygenation of the hypolimnion (Beutel 2006), capping of the sediment to prevent nutrient release (Ross et al. 2008), or floating wetlands for nutrient removal (Rodríguez-Gallego et al. 2004). Another option that is used for in-lake mitigation of eutrophication is biomanipulation through the introduction of organisms to aid the control of weed growth, for example (Bernes et al. 2015). Alternatively, lake restoration methods also include those that reduce external nutrient loading from the upstream catchment. For example, optimised fertiliser application or reduced stocking rates are options to reduce excess nutrient losses from farmland (Monaghan et al. 2008; McDowell et al. 2011).

Ecosystem management involves reconciling essential actions that maintain or restore ecological integrity with those which promote economic growth and social well-being, in a cost-effective fashion (Carpenter et al. 1999; Withers et al. 2014). Lakes are inherently complex and are characterised by emergent ecosystem properties that are affected by the external input of nutrients (Schindler 2012; Hipsey et al. 2015), groundwater lags (Baron et al. 2002) and long term effects such as climate change (Vörösmarty et al. 2000). Economic feasibility is also an important component of decision making for lake restoration actions (Baron et al. 2002; Søndergaard and Jeppesen 2007; Bateman et al. 2011). Many studies have considered the cost of in-lake solutions to address excess nutrient levels and their effects (e.g., Baron et al. 2002; Jeppesen et al. 2007; Allan et al. 2012; Spears et al. 2014; Hamilton et al. 2013). Other assessments have involved evaluation of the costs associated with changing land use and the implementation of best catchment management practice as a means to reduce nutrient loads to lakes (Monaghan et al. 2008; Ghebremichael et al. 2012; Giri et al. 2012).

An alternative perspective on lake restoration has been offered through ecosystem services studies. This concept refers to services provided by an ecosystem for the benefit of humans, which includes provisioning services such as food and water, regulation services such as climate and flood control, supporting services including nutrient cycling or pollination, and cultural services such as aesthetics and recreation (de Groot et al. 2002). Ecosystem services have been used widely to describe ecosystem values using a range of valuation methods (Costanza et al. 1997; de Groot et al. 2002; Farber et al. 2002; Batabyal et al. 2003; Costanza et al. 2014; Turner et al. 2015). They encompass values of natural resources that are not commonly covered in market valuations (de Groot et al. 2012). The concept has also been applied to lake restoration and land use change (Baron et al. 2002; Allan et al. 2012; Bateman et al. 2013).

In this study an ecological assessment was made using lake modelling, together with an economic analysis, including ecosystem services assessment, considering both the lake and the catchment, and noting that such integration is not frequently undertaken or achieved (Le et al. 2010; Kragt et al. 2011). The costs of lake restoration were considered, including those associated with changes to the value of ecosystem services. Moreover, the direct impact of restoration measures on the lake ecosystem were evaluated by simulating water quality outcomes associated with different management options for nutrient reduction in the lake. This integrated evaluation of restoration costs as well as their ecological effectiveness could contribute to improved restoration strategies by offering insight into different types of restoration procedures, their cost and their effect on lake water quality.

The study site is Lake Rotorua (central North Island, New Zealand) (Figure 8). This lake has been impacted by both point-source and diffuse source pollution over many years (Anderson 1965; Biggs 1980; Mueller et al. 2015). An integrated assessment of the outcomes of phosphorus and nitrogen reduction within the lake and the catchment is provided considering the environmental effects, mitigation costs and changes in land values associated with the nutrient reductions. It was hypothesised that catchment-based options to manage external nutrient loads would be more effective at improving water quality than environmental engineering approaches to reduce internal loads in the lake. Furthermore, inquiry is made of how the economic land value loss (referring here to the loss of profitability from different types of land use, as well as losses in ecosystem services values) for selected land use change options compares to land mitigation costs, especially when values of ecosystem services provided by land use are part of the value change equation.

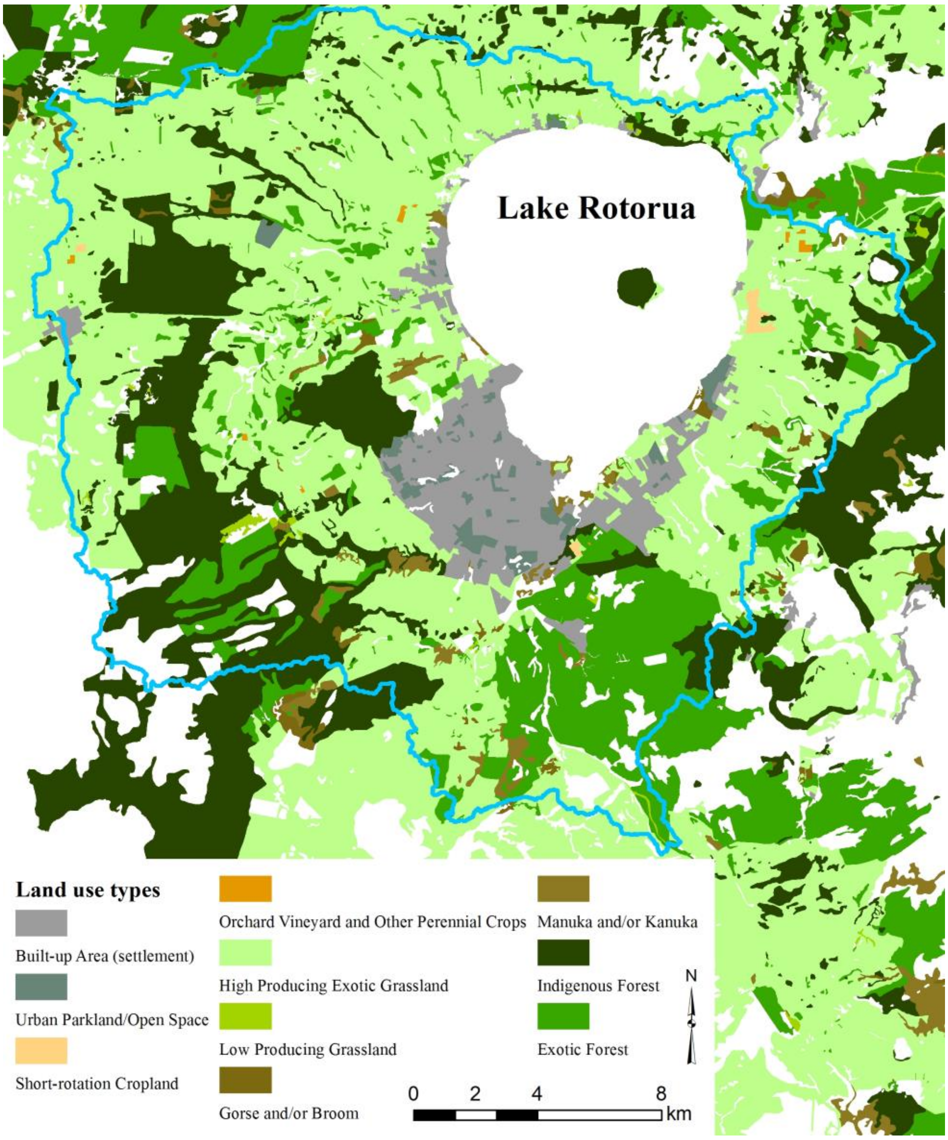

Figure 8 Lake study site and current catchment land use. Land use types derived from Land Cover Database New Zealand Version 4.1 provided by Landcare Research. Catchment boundary is given by outlined by blue line.

4.2 Methodology

4.2.1 Study site

Lake Rotorua is located in the central North Island of New Zealand (Figure 8). It is a shallow polymictic lake of volcanic origin with an area of c. 80 km^2, a mean depth of 10 m and a maximum depth of 45 m. The lake has a mean volume of 762 m^2 x 10^6, which varies depending on lake levels throughout the year (Ellery 2004). The trophic status of the lake is currently classified as eutrophic (BOPRC 2015). The catchment area of Lake Rotorua is c. 460 km^2, 15% of which is dairy, 25% dry stock, 18% forestry and 18% by native forest (BOPRC 2012). Other land use types include urban and geothermal areas. All land use types and areas covered are presented in detail in Appendix V. The eutrophication of the lake has historically been driven by increases in both point and diffuse sources of nutrients (Annett 1961; Burger et al. 2008). Management steps have been taken to address the declining water quality, including the application of a rule to prevent intensification of land uses in the catchment that would result in an increase in nutrient loads to the lake (Rule 11; EBOP 2008). Proposed land use rules have been drafted to implement limits of 435 t of nitrogen and 37 t of phosphorus per annum of loads from the catchment to the lake in order to reach a set water quality target. A combination of land use change and land mitigation is proposed to meet these limits (Rotorua Lakes 2015).

Authorities tasked with management of the catchment and the lake (including Bay of Plenty Regional Council and Rotorua District Council) have taken a number of approaches to attempt to restore Lake Rotorua (Burns et al. 2009). As with the restoration of many lakes, management is subject to the trade-off between economic development in the catchment, economically viable management, and water quality in the lake (Edgar 2008). In 2015, changes were made to the regulatory framework based on a collaboration of three major stakeholders (Bay of Plenty Regional Council, Rotorua District Council and Te Arawa Lakes Trust). Other important stakeholders that have contributed to this management framework include landowners, farm operators, the tourism industry and the general public. The goal of this Integrated Management Plan (2015) is to return the lake to a water quality and ecological status similar to that of the 1960s, when there was general consensus that lake water quality was in a desirable state.

The metric used to measure performance in relation to water quality is the Trophic Level Index (TLI), which gives a numerical value based on Secchi depth measurements and concentrations of total phosphorus, total nitrogen and chlorophyll *a* (Burns et al. 1999). A water quality target set for the lake of a TLI of 4.2) was first met in 2012 (BOPRC 2013), but has been slightly exceeded since (BOPRC 2015). To attain the TLI goal, both land use change to less intensive forms and implementation of best management practice on existing land uses have been considered.

In-lake options for nutrient management have been undertaken though alum dosing (Bruere 2012) and others have also been considered, including dredging of bottom sediments, which, from an economic perspective, was found to be prohibitively expensive (Miller 2007), sediment capping and other P-adsorbent materials, and the diversion of a major inflow, Hamurana spring, to the lake outflow (Hamilton et al. 2012). A floating wetland designed to uptake nutrients was installed near the eastern shore of the lake in 2014. However, the high cost of this restoration action compared with the nutrient load that the wetlands may remove from the lake means that it makes a negligible contribution to attenuating nutrients in its current size (Opus 2010; Tanner et al. 2011).

A number of economic assessments have been made with regard to land use and restoration of Lake Rotorua. The assessments have included the effect of water quality on house prices in Rotorua and settlements around the lake (Marsh and Woodham 2011), economic and policy implications of the implementation of a nitrogen credit trading scheme allowing land owners in the catchment to trade credits following reduction in nitrogen losses from their farm (Kerr and Lock 2009; Cox et al. 2013), cost to land owners of policy options for nitrogen reduction (Anastasiadis et al. 2011), economic effect of land use change (Bell et al. 2004) and nitrogen mitigation options (Park et al. 2014) on land owners, and the effect of nitrogen loss allocations on farm profits and land values (Parsons et al. 2015).

Both phosphorus and nitrogen loads from the catchment need to be controlled (Abell et al. 2015; Smith 2015). Within the catchment, phosphorus loss can be reduced through a wide variety of on-farm management options, including optimising fertiliser applications for pasture production, stream fencing, restricted livestock grazing, sediment traps and constructed wetlands (McDowell and Nash 2012). The baseflow of many of the major inflows to Lake Rotorua is dominated by groundwater. The large groundwater aquifers associated with the subcatchments create a relatively large mean base flow input (11.8 m^3 s^{-1}) as a proportion of the mean total stream inflow volume (13.3 m^3 s^{-1}) (Hoare 1980; Morgenstern et al. 2015). Groundwater inflows from the catchment are naturally low in nitrogen, but have become enriched over time through leaching from more intensive agricultural land use types. By contrast, increasing age of water in the aquifers results in natural enrichment with phosphorus from dissolution of rhyolite ('geological phosphorus'; Morgenstern et al. 2004, 2005). The groundwater lags are challenging for managing nitrogen loads to the lake, with potentially large loads entering the lake for a long time after management actions have been implemented on land. Strategies to reduce nitrogen losses from farms include reduced or no application of nitrogen fertilisers (and in some cases utilising nitrogen-fixing species such as clover to offset nitrogen fertiliser reductions), livestock ‘stand off pads’ to reduce sediment and nutrient losses during periods of high runoff in winter, and reduced stocking rates (Drewry et al. 2006; Monaghan et al. 2008).

An in-lake management action undertaken to achieve nutrient reduction within Lake Rotorua is the dosing of two surface inflows to the lake with aluminium sulphate (alum). Alum has been applied to the Utuhina Stream since 2006, and the Puarenga Stream since 2009. It is applied continuously to each inflow in a volume-weighted proportion in order to adsorb phosphorus and subsequently remove it from the water column by flocculation and settling to the streambed or the lakebed. Rates of alum dosing exceed levels that are theoretically expected to bind all dissolved reactive phosphorus in the stream inflows and it is therefore expected that excess alum also removes dissolved phosphorus from the water column within the lake itself (Hamilton, McBride, and Jones 2015). Alum has reduced phosphorus concentrations in the dosed streams and the lake, and also improved clarity within the lake by removing a proportion of suspended particles from the water column as part of the flocculation process, leading to an overall improvement in water quality and attainment in 2012 of the TLI objective of 4.2 (BOPRC 2015; Hamilton, McBride, and Jones 2015).

An additional issue currently to be addressed is the treatment and disposal of municipal wastewater from Rotorua city. Historically, treated wastewater discharged to the lake contributed 30 t of phosphorus and 150 t of nitrogen annually; nutrients from wastewater were a major driver of water quality decline in the 1980s (Howard-Williams et al. 1986; Rutherford et al. 1989). Treated wastewater is presently applied through a land-based system in a nearby exotic forest, with consented nutrient input limits from this subcatchment to the lake of 30 t of nitrogen and 3 t of phosphorus annually. The expiry of this resource consent means that forest disposal at the current site will cease in 2017 and so alternative tertiary treatment options will be required to meet the resource consent limits.

4.2.2 Research objectives

The objective of this research contains two main aims. The first aim is to study the effectiveness land use change, land management options (mitigation options) and lake treatment options on improving water quality of the lake through varying levels of nutrient load reduction. The second aim is to assess opportunity costs and benefits associated with land use change scenarios analysed, through examining both land profitability of agricultural land use and ecosystem services values.

4.2.3 Methodological approach

Two methodological approaches were used in this study: a quantitative water quality assessment of the effect of different nutrient load options on lake water quality using a lake model and an economic evaluation of land use change and mitigation options to achieve each of these options. By combining the results from these two assessments it is possible evaluate the cost and effectiveness of land use change and mitigation options in achieving a desired level of lake water quality. There were 6 scenarios that formed the basis of this analysis (Figure 9): Option 1 represented the reduction in annual external nutrient loading identified as the policy target outlined in the current Lake Action Plan (EBOP 2009) (to 435 t of nitrogen and 37 t of phosphorus annually), in combination with alum dosing. Option 2 reflected the same nutrient loads without alum dosing. Options 3 and 4 represented the current nutrient load from the catchment (750 t of nitrogen and 47 t phosphorus annually), with option 3 including alum dosing.

Nutrient load options involved adjusting the current nutrient load from the catchment (represented by option 3 and 4) to that of targeted nutrient loads described by policy (option 1 and 2), an additional reduction of phosphorus (option 6) and a scenario based on predicted water quality outcomes from a conversion of agricultural and horticultural land into native forest (option 5). Figure 9 outlines the methods, including a range of nutrient load options, as well as the land use change and mitigation options to meet these nutrient loads. There were six different options for which water quality was simulated and costs were evaluated. The first two options (1 and 2) represent a reduced nutrient load according to the Lake Action Plan; one includes alum dosing (1), and one represents a reduction of external nutrient loads without alum dosing (2) (Table 14).

Options 1 and 2 were split into three subcategories to denominate different ways to achieve this reduction. One subcategory represented land use change (1a and 2a). The scenario of land use change here represents the cessation of all intensive land uses (dairy and dry stock), moving to forestry as a replacement. The second subcategory represented nutrient reduction through land mitigation achieved by optimum (conservative) fertiliser application, reduced stocking rates and riparian plantings (1b and 2b). The third subcategory represented a combination of splitting the nutrient reduction achieved through land use change (50%), and land mitigation (50%) (1c and 2c). Options 3 and 4 were used to represent the nutrient loading from the current land use as described in the Lake Action Plan; option 3 included alum dosing while option 4 did not. Option 5 represents a significant level of nutrient reduction achieved by a simulated conversion of the catchment to native forest. Lastly, option 6 represented an additional reduction in phosphorus lower than that suggested in the Lake Action Plan, with nutrient reductions achieved and assessed in the same manner as options 1 and 2 (land use change, mitigation, and a combination of both (Figure 9).

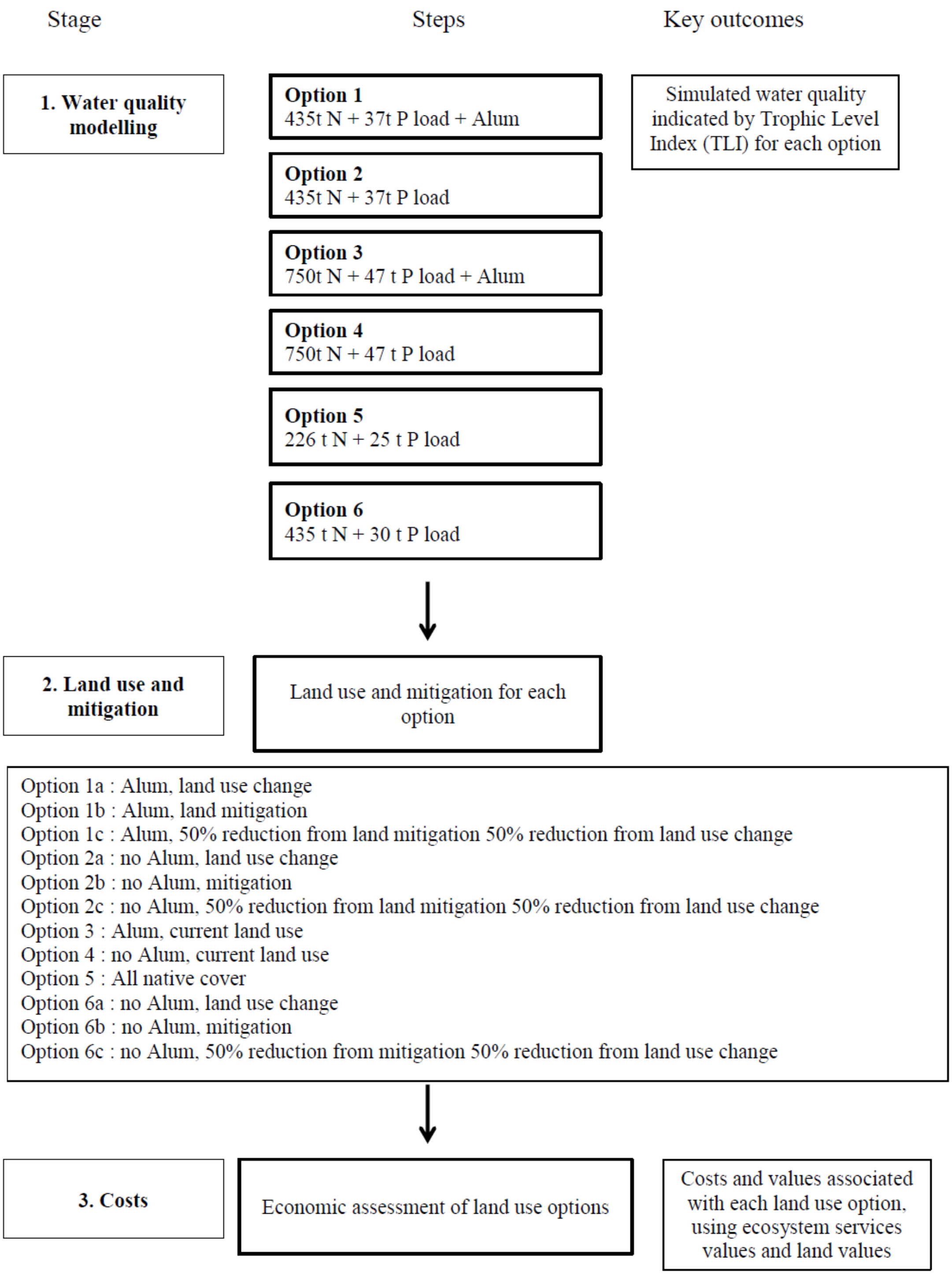

Figure 9 Flow chart outlining methodological steps: three different nutrient load options from the catchment, two additional options include alum dosing as mitigation option. Costs are assessed as a last step of the analysis. Loads are annual totals.

4.2.4 Lake modelling

Water quality outcomes were simulated for each nutrient load option using the one-dimensional coupled ecological-hydrological model DYRESM-CAEDYM

(DYnamic REservoir Simulation Model - Computational Aquatic Ecosystem DYnamics Model) developed by the Centre for Water Research, University of Western Australia. This model simulates the vertical distribution of physical variables such as temperature, and major chemical and biological constituents of the water column that are relevant to lake trophic state. The model was configured to include measured inflow data for the modelling period 2005-2014. The model is not currently set up to account for the application of alum to inflows. To indirectly represent the effects of alum dosing of two inflow streams (options 1 and 3), adjustments were made in the configuration of the model. Concentrations of the inflows were adjusted to reflect the decrease in dissolved reactive phosphorus concentrations expected due to alum dosing. Model parameters for sediment nutrient release rates and particulate matter diameter were also adjusted to account for expected reductions in sediment nutrient release rates and increased organic sediment diameter due to flocculation, respectively, caused by the alum application. The maximum potential release rate of dissolved reactive phosphorus from the lake sediment (under anoxic conditions) was set to reflect suppression of internal loading. Increased flocculation of organic matter due to alum dosing was represented by increasing the simulated particulate organic material diameter (Hamilton, McBride, and Jones 2015). Further detailed discussion of these modifications can be found in Hamilton, McBride, and Jones (2015). A detailed overview of the chosen model parameters specific for this model application is given in Appendix IV.

Data output from the model was used to calculate the Trophic Level Index (TLI) chosen in this study to represent water quality. Similar to the Trophic State Index (Carlson 1977), TLI provides an indication of the trophic state of a lake. The index considers measurements of total phosphorus (TP) and total nitrogen (TN) concentrations, chlorophyll *a* and clarity (Secchi depth) (Burns et al. 1999).

The following equations were used to calculate the TLI:

Chlorophyll *a*	$\mathrm{TLc} = 2.22 + 2.54\log(\mathrm{Chl}\,a)$	(1)
Secchi depth	$\mathrm{TLs} = 5.10 + 2.60\log(1/\mathrm{SD} - 1/40)$	(2)
Total phosphorus	$\mathrm{TLp} = 0.218 + 2.92\log(\mathrm{TP})$	(3)
Total nitrogen	$\mathrm{TLn} = -3.61 + 3.01\log(\mathrm{TN})$	(4)
Integrated value	$\mathrm{TLI} = 1/4(\mathrm{TLc} + \mathrm{TLs} + \mathrm{TLp} + \mathrm{TLn})$	(5)

As the model did not explicitly simulate Secchi depth, this variable was calculated as follows:

$$\text{Secchi depth} = 1.74/k_d \tag{6}$$

where k_d represents the diffuse attenuation coefficient of light, with Secchi depth calculated based on measured values of k_d and a coupling constant derived from a regression of measured values of Secchi depth and k_d for Lake Rotorua (cf. Gallegos 2001). A simulation was completed for each of the nutrient loading options, including with or without alum dosing (Table 14). External nutrient loads were modified for the options that included nutrient reductions from the status quo by applying a reduction factor to the nitrogen and phosphorus loads for each of the inflows. For the option representing the current nutrient load from the catchment, annual load over the simulation period was based on an average annual load of 750 t of N and 47 t of P, informing input of daily nutrient loads for each of the inflows based on daily inputs. Model simulations were run over a 9-year period (2005-2014). Values of *TLI* were calculated using model output and equations 1-6.

Model performance statistics

Performance of the lake model has previously been evaluated by comparing model results with measured data using several statistical metrics, including RMSE (root-mean square error) values (Hamilton et al. 2015). Some variables had high RMSE values (Table 13), including TN and Chl*a*, indicating relatively higher absolute model error for these variables. A previous application using the same model configuration showed reasonable performance for TLI, with simulated results differing ±0.1 TLI units from measurements (Hamilton et al. 2015); the model performance in my study was deemed satisfactory and comparable to or better than previous model applications to Lake Rotorua (Hamilton et al. 2012; Hamilton et al. 2015). Performance of the model was, however, influenced by the dosing of two inflows to the lake with alum as this procedure is not directly accounted for (i.e., by dynamically adjusting the amount of alum dosed on a daily basis). Instead, modifications of the model set up were made to the phosphorus sediment release rates and concentrations in the two dosed inflows (see details in Section 2.2). Alum dosing was also identified by Abell, McBride, and Hamilton (2015) as a factor that decreased the model’s performance in simulating inter-annual TLI values over the same time period as that simulated in my study.

Table 13 DYRESM-CAEDYM model performance statistics based on comparisons with field observations for Lake Rotorua for the calibration (2004-2007) and validation periods (2001-2004). Statistics show root mean square error (RMSE) at a range of lake depths; adapted from Hamilton et al. (2015). Chlorophyll *a* measurements are taken only at the surface (0 m).

	Calibration (2004-2007)			**Validation (2001-2004)**		
	0 m	15 m	19 m	0 m	15 m	19 m
NO_3 (mg N/l)	0.017	0.019	0.020	0.014	0.019	0.014
NH_4 (mg N/l)	0.029	0.053	0.084	0.032	0.053	0.099
TN (mg/l)	0.126	0.118	0.109	0.122	0.118	0.174
PO_4 (mg P/l)	0.006	0.010	0.010	0.009	0.010	0.017
TP (mg/l)	0.011	0.012	0.013	0.011	0.012	0.021
Chl*a* (mg/m^3)	12.55			13.12		

Nutrient load options and associated land use and mitigation options

Areal nutrient loss for dairy and dry stock land uses was based on output from Overseer v5.4.11, a nutrient budget software tool that predicts nutrient export from farms, for each of these pastoral land uses (Wheeler et al. 2011). Losses from native and exotic forest land use were based on a study of nitrogen and phosphorus losses to streams in exotic and native forest catchments by Cooper and Thomsen (1988).

Table 14 Options of catchment nutrient loads, and associated land and lake mitigation used in lake water quality simulation. Details of land use and mitigation are given in Figure 2.

Option	Annual nutrient load	Nutrient mitigation options
1	435 t N + 37 t P + alum	Land use change, land mitigation or 50:50 combination
2	435 t N + 37 t P	Land use change, land mitigation or 50:50 combination
3	750 t N + 47 t P+ alum	Representation of current land use and nutrient loss
4	750 t N + 47 t P	Representation of current land use and nutrient loss
5	200 t N + 23 t P	Nutrient load representing conversion of agricultural, horticultural land to native cover
6	435 t N + 30 t P	Land use change, land mitigation or 50:50 combination

Options 1b and 2b included nutrient mitigation options for both the catchment and the lake. Catchment mitigation was represented by using a simplified approach of reducing N and P losses only from intensive land uses (dairy, dairy support and dry stock), leaving out changes to usage such as horticulture. These land uses represent the largest areal yields (kg ha^{-1} y^{-1}) of nutrients in the Rotorua catchment (EBOP 2009). The combined mitigation and land use change options (1c, 2c, 6c) represents a scenario of 50% of the nutrient load reduction from undertaking land-based mitigation and the remaining 50% obtained through land use change.

For options 1a, 2a, 5 and 6a, land use change was used to achieve the target nutrient loads, focusing on reducing those types of land use with high nutrient losses (dairy, dry stock), while increasing those with low losses (plantation forestry, native forest). For the options where the reduction target was achieved entirely through land use change (options 1a, 2a, 6a), dairy land use was decreased by 9%, dairy support by 4%, and dry stock by 5% of the catchment area. Forestry was increased by 3% and native forest by 15% as replacement for intensive land use types. For options 1c, 2c and 6c, a combination of mitigation and land use change was used. Half of the nutrient reduction was achieved through land mitigation, with the other half through land use change. The area used by dairy and dairy support was reduced by 3% in each case, and dry stock by 7% while forestry was increased by 5% and native forest by 7%. A detailed overview of land use types and areas is presented in Appendix V. Other combinations of land use change and land mitigation are possible, as discussed in section 4.5.

4.2.5 Evaluation of land value changes and mitigation costs

Land values and value change

Land values were calculated using annualised per hectare values added based on land profitability by each land use type from two economic studies (Saunders & Saunders 2012; Wang et al. 2015). Land values (V_{LU}) for each land use type in the catchment were calculated as:

$$V_{LU} = a * v_{lu} \qquad (10)$$

where a is the total area in ha of each land use and v_{lu} is the annualised per hectare land profitability value in $NZ for each land use type (Saunders and Saunders 2012; Wang et al. 2015). Ecosystem services values (V_{ES}) for each land use type in the catchment were calculated as

$$V_{ES} = a * v_{es} \qquad (11)$$

where V_{ES} is the per hectare value in \$NZ per year associated with the land use type. Using benefit transfer, alues of V_{ES} were adjusted from an assessment of the value of land-based ecosystem services in New Zealand (Patterson and Cole 2013). Patterson and Cole (2013) assessed a range of land use types and determined annual ecosystem services values per hectare on a national scale using a total economic value derived from the sum of all use values and passive or non-use values. Double counting for this study was avoided by separating out supporting ecosystem services. No benefit transfer was used for the lake ecosystem services values, where a local study of Lake Rotorua was used instead (Mueller et al. 2016). The individual ecosystem services included in this study for each land use type are listed in Table 15.

For the value of the lake ecosystem services, a site-specific study was employed (Mueller et al. 2016) instead of the national scale assessment (Patterson and Cole 2013), to provide a more specific value. This study used a range of ecosystem services values including existence value, hedonic pricing and replacement cost to value five ecosystem services provided by the lake: biodiversity, food, nutrient processing, aesthetics, and recreation. Background assumptions for ecosystem services values and land profitability values are outlined in (Table 15).

Values of V_{LU} and V_{ES} were calculated for all options. Opportunity costs and benefits of the different land use scenarios were then calculated. For options where nutrient reduction was achieved partially or entirely through land use change (1a, 1c, 2a, 2c, 5, 6a, 6c), value change was calculated by comparing changes in V_{LU} and V_{ES}. Change in value was calculated by subtracting the total value (for each V_{LU} and V_{ES}) of the land use change from the value of current land use in the catchment.

Mitigation costs

For options 1, 2 and 6, catchment mitigation costs were estimated for the load reduction of P and N (i.e., \$NZ per kg) from intensive land use types (dairy, dairy support and dry stock). The costs were derived from mitigation cost studies in New Zealand (Ledgard et al. 2008; McDowell 2010; Vibart et al. 2015). The assessed P mitigation strategies included management through optimising (reducing) soil P levels, stream fencing, restricted grazing, greater effluent pond size, effluent application to land and tile drain amendments (McDowell 2010; Vibart et al. 2015).

The N mitigation strategy costs included reduced fertiliser application, uncovered feed pads, stream fencing, enhanced animal productivity and low-rate effluent application (Ledgard et al. 2008; Vibart et al. 2015). The average for N and P reduction for dairy and sheep and beef farming used here is based on the figures presented in the studies described above, using the practice options of optimum (conservative) fertiliser application, reduced stocking rates and riparian plantings. Sources and cost details are presented in Table 15.

Lake mitigation costs for options featuring inflow treatment with alum were calculated using the annual price in 2015 of $0.6 million for alum application to the two inflows (N. Sumeran, Bay of Plenty Regional Council, *pers. comm.*).

Sensitivity analysis

Sensitivity analysis was conducted to assess the variability and impact of uncertainty in land values, ecosystem services values and land mitigation costs. It was conducted by varying the input for those parameters where uncertainties were assessed to be highest, and comparing the resulting value estimates.

For the estimation of land and ecosystem services values, uncertainties were related to the specific land use values (V_{LU} and V_{ES}, see Section 2.4.1), which were based on New Zealand studies of agricultural land (Saunders & Saunders 2012; Wang et al. 2015). Benefit transfer was required as no catchment-specific studies were found to be applicable to my study (using land profitability values rather than sales prices of land). Ecosystem services values were based on a nation-wide study of land-based ecosystem services (Patterson and Cole 2013). For both sets of land use values, no alternative studies were found to inform a value range for the sensitivity analysis. For this reason, existing values were varied by a percentage to show the effect of uncertainties on the key outputs in this study. To analyse the effects of extreme variability, land values and ecosystem services were varied to show low and high estimates through increasing or decreasing each value by 50%. Literature presented a large range of land values, so a large range of variability was chosen to explore the magnitude of changes in values with a view to land value change outcomes.

The literature exhibited broad variations in the range of potential mitigation costs, in particular for nitrogen mitigation. One study (Park et al. 2014) had a significantly higher estimate of mitigation costs than others. To encompass the range of potential mitigation costs, costs of N reduction were varied from $1.90 to $171 per kg depending on mitigation options used (Park et al. 2014; Vibart et al. 2015) and for P reduction from $30 to $350 per kg (McDowell and Nash 2012).

Table 15 Background assumptions for ecosystem services values, land values and land mitigation cost analysis.

	Horticulture	Agriculture		Scrub	Forest	Lake	Wetland
Ecosystem services valued	Water provisioning, food production, climate regulation, erosion control, pollination	Water provisioning, food production, raw materials, recreation, cultural, gas regulation, waste treatment, biological control, soil formation, nutrient cycling, erosion control, pollination		Cultural, climate regulation, waste treatment, biological control, soil formation, nutrient cycling, erosion control	Raw materials, recreation, cultural, climate regulation, waste treatment, biological control, soil formation, nutrient cycling, erosion control	Nutrient sequestration, habitat provision, food provision, aesthetics, recreation	Water provisioning, recreation, cultural, gas regulation, disturbance regulation, waste treatment, refugia, water storage & retention
Ecosystem services value (\$ ha^{-1} y^{-1})	11937[1]	1188[1]		485[1]	1688[1]	\$14125[4]	30855[1]
Land use value (\$ ha^{-1} y^{-1})	1499[2]	Sheep & Beef	Dairy		1223[3]		
		290[2]	4157[2]				
Land mitigation cost:							
Phosphorus (\$ kg^{-1} y^{-1})		135	135				
Nitrogen (\$ kg^{-1} y^{-1})		15	8				

[1] Patterson & Cole 2013

[2] Saunders & Saunders 2012

[3] Wang et al. 2015

[4] Mueller et al. 2016

4.3 Results

4.3.1 Water quality results

The analysis of water quality for the different nutrient load options was based on comparisons of Trophic Level Index (TLI) values simulated with the lake model (DYRESM-CAEDYM). Values of TLI for each of the nutrient load options are summarised in Figure 10. Values are 9-year averages from the annual average derived from daily simulation output. Error bars in Figure 10 indicate annual variation in model results over the 9-year period. The results simulated here align well with the measurements of water quality over the same time period (Figure 10). Options 3 and 4 of current land use in the catchment gave a TLI of 5.06 without alum dosing and 4.47 with alum dosing; a decrease of 0.59 with alum dosing. The reduced annual nutrient load option of 435 t N y^{-1} and 37 t P y^{-1} (1, 2) had a TLI of 4.44 without alum dosing, and 4.02 with alum dosing, i.e., a reduction of TLI of 0.42 with alum dosing. An additional further reduction of phosphorus represented by option 6 (435 t N y^{-1} and 30 t P y^{-1}) yielded a TLI of 4.21 (9-year average). The 'all native' option (5), with a catchment load of 200 t N y^{-1} and 23 t P y^{-1}, resulted in a 9-year average TLI of 3.63.

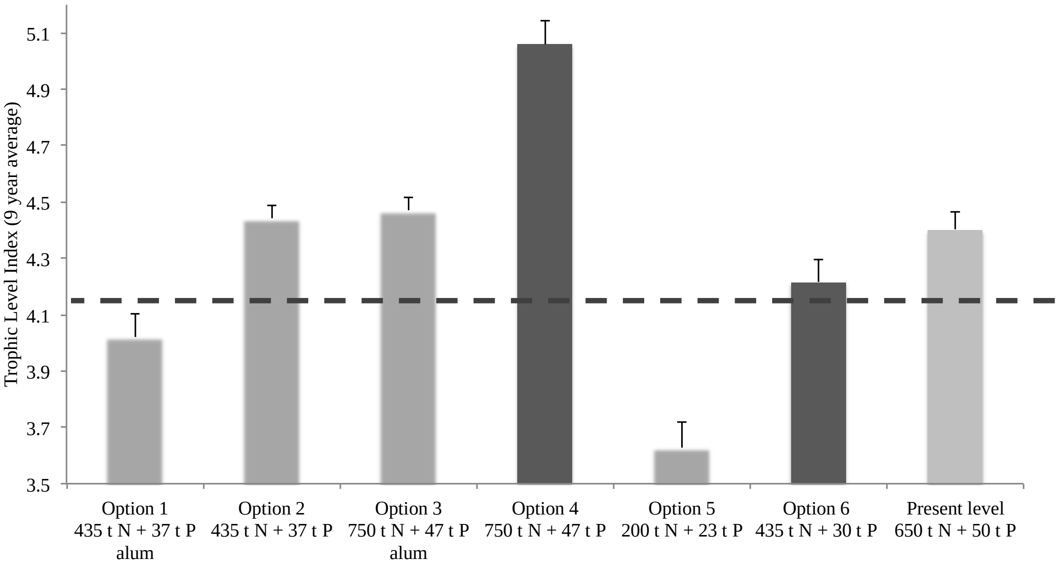

Figure 10 Trophic Level Index values for the different annual nutrient load options from catchment and the present water quality level (measurement). Error bars represent standard deviation of annual variation over the 9-year period. Nutrient loads are annual values. Dashed line indicates TLI target (4.2) in regional water policy.

4.3.2 Catchment land values

For the land value assessment, a total land profitability value was established from the area assigned to each land use in the catchment for each of the options and from the assigned per unit area value of the land type. In addition to total land values, land value changes were estimated by comparing values of the current catchment land use with those simulating land use change (options 1, 2, 5 and 6).

Options 1b, 2b, 3, 4 and 6b had a total land value, represented as value added, of $58.5 million per year. Land values for options 1c and 2c were $52.6 million per year, and for the 'all native' option 5, $49.1 million per year. Land values for option 6a were $45.5 million, and for 6c $48.5 million per year. The lowest total value was associated with nutrient reduction through the land use change-only options 1a and 2a, at $40.6 million per year (Table 16). For change in value (opportunity cost) associated with land use change scenarios, there was a value loss between the current land use (associated with options 1b, 2b, 3, and 4) and the land use change options (option 1a, 2a) of $17.83 million per year. Option 5 was associated with a loss of $9.3 million per year, and options 1c and 2c with a loss of $5.9 million per year. Option 6a yielded a land value loss of $12.98 million, and option 6c a loss of $9.9 million (Table 17).

The ecosystem services value assessment was based on calculation of the ecosystem services value associated with each land use type, which varied with the land use combination for the option considered. As for the land value assessment, ecosystem services value changes were then estimated by comparing values of the present catchment land use (options 1b, 2b, 3, 4 and 6b) with those that included land use change (options 1a, 1c, 2a, 2c, 5, 6a and 6c). The highest value was associated with the 'all native' land use option at $186.6 million per year as forested areas were valued more highly than more intensive land use types. The ecosystem service value for the land use change option (1a and 2a) was $180.8 million, which was comparable to $183.4 million for option 6a. For the combination option, ecosystem services values were $179.4 million per year for options 1c and 2c, and $178.4 million for option 6c. Options 1b, 2b, 3, 4 and 6b, where the current land use in the catchment was represented, had a value of $176 million per year (Table 16).

Opportunity costs and benefits for ecosystem services values were also calculated. Value change of ecosystem services from the current land use to the ‘all native’ option (5) was $10.6 million per year. For the land use change option (1a, 2a), there was an increase in ecosystem services value of $4.8 million per year, and for the combination options (1c and 2c), an increase of $3.5 million per year (Table 17). For option 6a, there was an ecosystem services value increase of $7.4 million per year, and an increase of $2.4 million for option 6c.

Options 1a and 2a had a total land value decrease of $13.0 million, options 1c and 2c a total decrease of $2.4 million, option 6a a total land value decrease of $5.6 million and option 6c a total decrease of $7.5 million per year. Option 5 yielded a total value increase of $1.2 million annually.

Table 16 Total N and total P loads to Lake Rotorua, Trophic Level Index (TLI) based on a 9-year average from simulations with DYRESM-CAEDYM, ecosystem services values, land values and mitigation costs associated with each of the six management options. Options 1a-c, 2a-c and 6a-c show different mitigation options (a, b and c) for the same nutrient load. Refer to Figure 8 for a description of options. Values and costs are given in NZ$ million. The acronym ES is ecosystem services.

Option	TN (t/yr)	TP (t/yr)	TLI	ES value ($M/yr)	Land value ($M/yr)	Land mitigation cost ($M/yr)	Lake mitigation cost ($M/yr)	Total cost ($M/yr)
1a	435	37	4.02	180.8	40.6	-	0.6	0.6
1b	435	37	4.02	176.0	58.5	3.9	0.6	4.4
1c	435	37	4.02	179.4	52.6	1.1	0.6	1.6
2a	435	37	4.44	180.8	40.6	-	no alum	-
2b	435	37	4.44	176.0	58.5	3.9	no alum	3.9
2c	435	37	4.44	179.4	52.6	1.1	no alum	1.1
3	750	47	4.47	176.0	58.5	-	0.6	0.6
4	750	47	5.06	176.0	58.5	-	no alum	-
5	226	25	3.63	186.6	49.2	-	no alum	-
6a	435	30	4.21	183.4	45.5	-	no alum	-
6b	435	30	4.21	176.0	58.5	4.8	no alum	4.8
6c	435	30	4.21	178.4	48.4	2.2	no alum	2.2

Table 17 Changes in land and ecosystem services values in the Lake Rotorua catchment associated with land use change options (opportunity costs and benefits for each option shown in last column). There was no change from current for options 1b, 2b, 3 and 4 as these did not include land use change.

Option	Land value change	ES value change	Total change (opportunity cost/benefit)
1a	-17.8	4.8	-13.0
1b, 2b, 3, 4, 6b	No change	No change	No change
1c	-5.9	3.5	-2.4
2a	-17.8	4.8	-13.0
2c	-5.9	3.5	-2.4
5	-9.3	10.6	1.2
6a	-12.98	7.4	-5.6
6c	-9.9	2.4	-7.5

4.3.3 Mitigation costs in catchment and lake

Mitigation costs for inflow treatment with alum were relevant to Options 1a, 1b, 1c and 3, where alum was used to reduce in-lake nutrient concentrations (Table 16). Total mitigation costs encompassing land mitigation costs within the catchment were $3.86 million per year for option 2b and $1.06 million per year for option 2c (representing a combination of land use change and mitigation, without the application of alum). For options 1b and 1c, the addition of alum dosing to land mitigation costs was $4.43 million annually (1b) and $1.63 million (1c). For option 6b, there was a land mitigation cost of $4.8 million, and $2.2 million for option 6c. All mitigation costs are represented in Table 16.

Figure 11 shows lake and land mitigation costs for each option and also compares the simulated TLI (representing water quality). Appendix V provides details on land use types and their area for each option, total nutrient loads from the catchment, as well as individual catchment values of land and ecosystem services, and mitigation costs for lake and land mitigation.

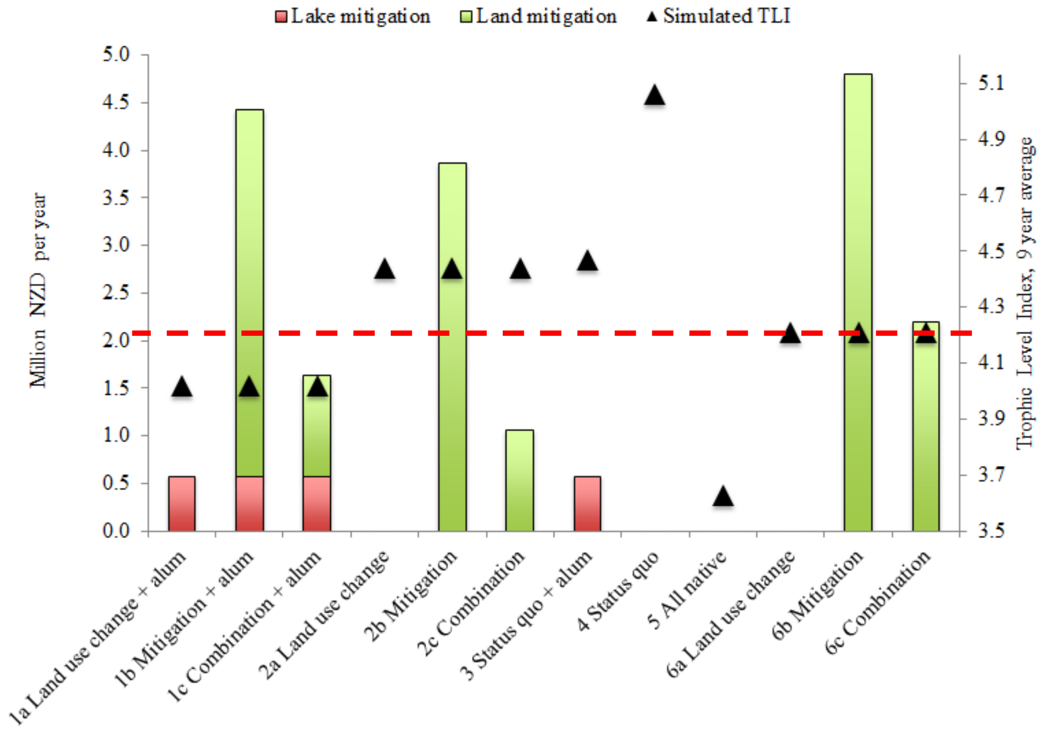

Figure 11 Total lake and land mitigation costs and simulated 9-year average Trophic Level Index for each option. Dashed line indicates TLI target (4.2) in regional water policy plan (EBOP 2009).

4.3.4 Total changes in land value, mitigation costs and water quality outcomes

Adding all mitigation costs and changes in value (i.e., total change represented in Table 17) from both losses and gains in land valuation, the ‘all native’ option (5) yielded the lowest cost and value change, with an opportunity benefit (i.e. a net economic gain) of $1.24 million annually. The options reflecting the current catchment land use with and without alum dosing (3 and 4) had a cost of $0.57 annually for alum (3), and of course no cost for no alum application (4). The combination of land mitigation with alum (1c) yielded a total annual cost and value change of $4.07 million, and $3.50 million without alum dosing (2c). The mitigation option with alum (1b) was associated with a total cost and change in value of 4.43 million per year, and $3.86 million per year without alum (2b). The land use change with alum (2a) gave a total annual cost and value change within the catchment of $13.03 million without alum and $13.60 million with alum (option 1a). The additional reduction of phosphorus represented by option 6 was associated with a total cost and value change of $4.8 million for the mitigation option (6b), $5.6 million for the land use change option (6a) and $9.7 million per year for the combination option (6c) (Figure 12).

Option 5 showed the best environmental outcome with a simulated TLI of 3.63 averaged over the 9-year simulation period. The reduced nutrient load option with alum dosing (1) yielded a TLI of 4.02, meeting the TLI target envisaged by decision makers. Under the reduced nutrient loads simulated for option 2, the water quality target was not reached without the addition of alum dosing (TLI of 4.44). While the current land use (3, 4) was less costly, simulations indicated that these options would not reach the water quality target. This is particularly the case for the option without alum dosing (4; TLI 5.06) (Figure 12).

An additional reduction of phosphorus beyond the target of the Lake Action Plan (Rotorua Lakes 2015) was required to meet the water quality target, represented by option 6, which yielded a simulated TLI of 4.21. The most cost-effective way to achieve the water quality target without alum dosing was offered by the mitigation option, 6b (Figure 12).

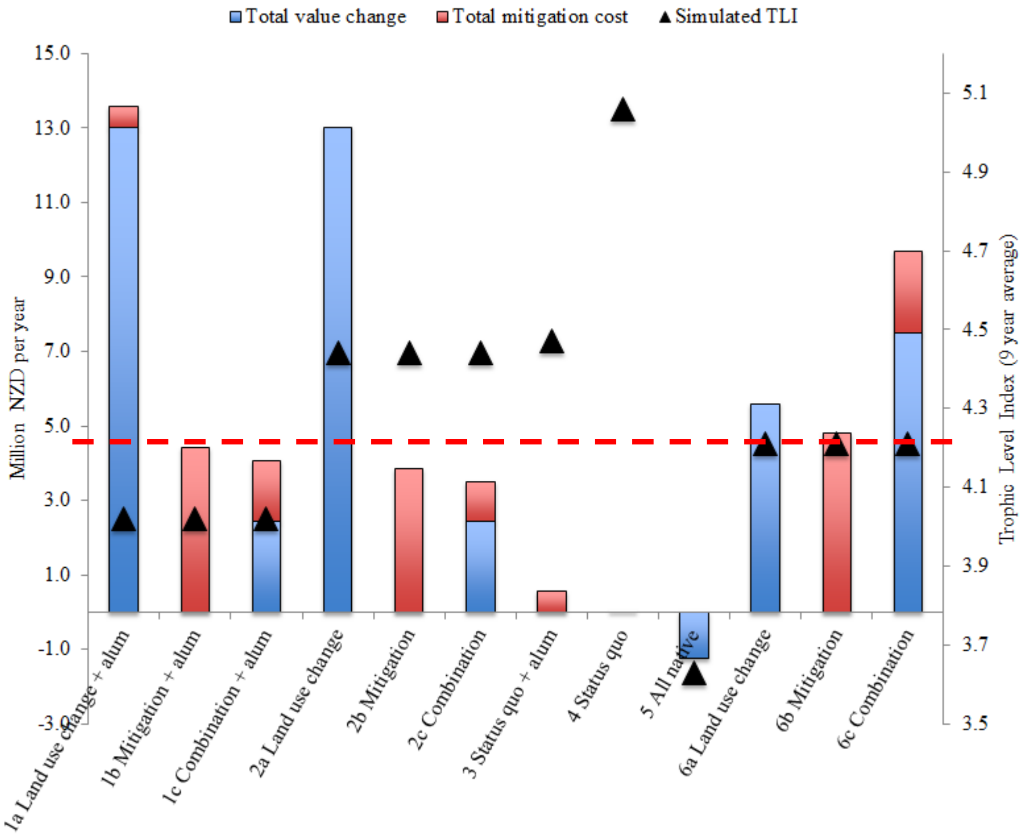

Figure 12 Total land and mitigation costs and catchment value changes, simulated 9-year average TLI for each option. No cost or land use change is involved for option 4. Red line indicates TLI target (4.2) in regional water policy.

4.3.5 Sensitivity analysis

Results of the sensitivity analysis for land mitigation costs, land values and ecosystem services values show the effect of uncertainties associated with mitigation costs and value estimates. A wide range in costs for both nitrogen and phosphorus mitigation was represented in the literature. For nitrogen removal, the range was between \$0.6 and \$53.8 million for options 1b, 2b and 6b, while options 1c, 2c and 6c had a range of cost between \$0.3 and \$22.6 million. For phosphorus removal, analysis showed a range of mitigation cost between \$0.3 and \$3.5 million for options 1b and 2b, \$0.5 to \$5.9 million for option 6b and \$0.3 to \$22.1 million for option 6c. Total mitigation cost ranged between \$0.9 and \$57.3 for options 1b and 2b, between \$0.3 and \$22.6 for options 1c and 2c, \$1.1 to 59.8 million for option 6b and \$0.5 to \$25.1 million per year for option 6c (Table 18).

Variability in total land values and ecosystem services values is represented in Table 18. Low ecosystem services values (decreased by 50%) produced a large amount of variability and ranged between \$88.0 and \$93.3 million. High ecosystem services values (increased >50%) ranged between \$352.0 and \$373.1 million. Land values ranged between \$20.3 and \$29.2 million for the low estimates, and between \$81.3 and \$117.0 million for the high estimates. Total value change for options 1a and 2a ranged between -\$6.5 and -26.1 million, -\$1.2 and -\$-4.9 million for options 1c and 2c, \$0.6 and \$2.5 million for option 5, -\$2.8 and -\$11.2 for option 6a and -\$3.8 and -\$15.1 million for option 6c (Table 18).

Table 18 Results of sensitivity analysis showing low and high estimates of mitigation costs, ecosystem services (ES) values, land values and total value change for each of the management options. All values are in $NZ million per year.

Option	Low mitigation cost	High mitigation cost	Low ES value	High ES value	Low land value	High land value	Total change low	Total change high
1a	-	-	90.4	361.6	20.3	81.3	**-6.5**	**-26.1**
1b	0.9	57.3	88.0	352.0	29.2	117.0	-	-
1c	0.3	22.6	89.7	358.9	26.3	105.2	**-1.2**	**-4.9**
2a	-	-	90.4	361.6	20.3	81.3	**-6.5**	**-26.1**
2b	0.9	57.3	88.0	352.0	29.2	117.0	-	-
2c	0.3	22.6	89.7	358.9	26.3	105.2	**-1.2**	**-4.9**
3	-	-	88.0	352.0	29.2	117.0	-	-
4	-	-	88.0	352.0	29.2	117.0	-	-
5	-	-	93.3	373.1	24.6	98.3	**0.6**	**2.5**
6a	-	-	91.7	366.7	22.8	91.0	**-2.8**	**-11.2**
6b	1.1	59.8	88.0	352.0	29.2	117.0	-	-
6c	0.5	25.1	89.2	356.8	24.3	97.1	**-3.8**	**-15.1**

4.4 Discussion

4.4.1 Effectiveness and cost of management options

This study demonstrates outcomes of lake management options in terms of costs and effectiveness in improving lake water quality. It considers profitability and ecosystem service values of land use, and provides integrated economic and environmental analyses for lake management assessment. Alum dosing is currently required to improve water quality in Lake Rotorua and meet the water quality policy target. The simulation results show that lake water quality would be in the supertrophic range under current nutrient loading from the catchment and no alum application (Burns et al. 1999). Options to meet the water quality target included the conversion of all intensive land use to forest, nutrient reduction from catchment land uses to 435 t of nitrogen and 37 t of phosphorus annually in combination with alum application to the lake, and a scenario that reduced catchment nutrient loads to 435 t of N and 30 t of P annually. The additional reduction of phosphorus load beyond what is proposed by the policy target, to an annual load of 30 t, could be sufficient to meet the water quality target without the requirement of alum dosing (9-year average simulated TLI of 4.21). This result shows the criticality of limiting the inputs of both nitrogen and phosphorus into the lake for sustained water quality improvements.

From an economic perspective, it has been shown that when ecosystem services values are considered, the most effective option to improve water quality with highest economic gain is the conversion of intensive land use types to exotic or indigenous forest. This option yielded a net economic gain (Figure 12).

The next most cost-effective option to meet the water quality target was a combination of land use change, land mitigation and lake mitigation (alum dosing). However, costs of additional phosphorus mitigation from the catchment combined (option 6b) showed a similar cost without the need for alum dosing to achieve the water quality target (Figure 12).

It has been demonstrated that high intensity land use types are not necessarily high value land use types. Using ecosystem services as a tool to describe land ecosystem values, the opportunity benefit added from native forest in the 'all native' option outweighs the potential value lost from more profitable land use types, leading to a net gain in land value. The findings offer an important contribution to the debate about what land use types will have the most desirable economic, social and environmental outcomes at a time when in 2014 a National Policy Statement for Freshwater Management (Ministry for the Environment 2014) has been implemented to provide a limits-based framework for managing diffuse pollution to support water quality

In valuation studies that consider the provision of ecosystem services, the value derived from intensive agricultural practices appears to be lower than in analyses focusing on land values, land price and profitability (de Groot 2006; Nelson et al. 2009; Johnson et al. 2012). A focus on ecosystem services means a valuation applies ecological concepts of functionality and complexity to land use analyses, which therefore can account for a broad range of functions and services (Silvestri et al. 2013). This focus allows to assess annual values as well as opportunity costs and gains of land use that can incorporate the broader impacts of different land use types in ecology, economic and resource management contexts, and provides analysis of trade-offs between different land use and land management options in these contexts.

4.4.2 Context of research findings

The integration of lakes and their catchments, including ecological processes and economic contexts, is important for the successful and sustainable management of freshwater resources (Baron et al. 2002). Water management at the catchment scale has increasingly been incorporated into studies which include an economic analysis, with these studies indicating the challenges of integrating ecological and economic principles in such an analysis (Kragt et al. 2011). Studies of ecosystem services and land use stress the importance of balancing ecosystem services provisions and ecosystem functions, pointing to the importance of ecosystem conservation in ecological and economic terms (Zhao and Tong 2013). When valuing the ecosystem services that flow from different land uses, aspects such as ecosystem functioning and integrity and the need for conservation, are important in assessing the ecological, economic and social viability of the studied area (Turner et al. 2015). An integration of these aspects offers insights into underlying issues of resource management: most importantly, it addresses questions of exploitative resource use, and what ecological and economic implications this has in the long-term. These implications should be addressed in any resource management question that considers conservation and economic growth (Silvestri et al. 2013). A full interdisciplinary integration is inherently difficult (Ekasingh and Letcher 2008) and rarely achieved (Wei et al. 2009).

For Lake Rotorua and its catchment, the findings are placed within a context of economic and ecological assessments that have commonly been completed independently. For example, the economic implications for landowners of policies to reduce both nitrogen and phosphorus have been evaluated (Daigneault et al. 2012). They show the potential reductions in farm land prices when nutrient loss restrictions are implemented. Several studies have been undertaken over a number of years to characterise nutrient limitation of primary productivity in the lake, with a view to informing lake water quality management policy (White 1975; Burger et al. 2007; Abell, Hamilton, and McBride 2015; Smith 2015). The results of nutrient enrichment experiments show that within the last decade or longer, primary productivity has been consistently limited by both nitrogen and phosphorus (Burger et al. 2007). Studies of in-lake N:P ratios also suggest dual nutrient limitation (Abell, Hamilton, and McBride 2015; Peryer-Fursdon et al. 2015; Smith 2015), meaning that policy and management decisions should be aimed at reducing loads of both nutrients to more strongly limit the rate of phytoplankton productivity and reduce biomass. These findings are supported by the water quality simulation results of this study, where water quality level targets were only reached by scenarios with substantial phosphorus input reductions beyond what is proposed by current policy approaches (see Section 4.1). As concluded in previous studies, reductions in both phosphorus and nitrogen loads are required to achieve water quality targets (Burger et al. 2007).

The integrated economic assessment of ways to achieve water quality outcomes presented here offers valuable insights into the policies that could be used to manage nutrient loads to the lake in future. The research integrates changes in the catchment directly and water quality outcomes in the lake. It considers a range of nutrient load options, and assesses the costs and ecological effects of nutrient reduction. It shows that intensive land use types do not offer the most desirable economic outcomes when viewed from an ecosystem services perspective.

4.4.3 Uncertainties in modelling and valuation

A limitation of the model simulations was the inability to directly simulate alum dosing effects on the lake (section 2.2). In practice alum dosing varied with discharge in the two streams, to maintain aluminium ion concentrations within set ranges. In reality, concentrations varied broadly and there were occurrences of no dosing for periods of time. Thus, the impacts of alum dosing were represented by an average condition of in-stream effects and extending into the lake, rather than showing daily variations that occur. This shortcoming of not including the variability of dosing may have had some effect on the accuracy of the simulation results and the TLI values used to indicate water quality in this study. Inclusion of a geochemical model, including aluminium concentrations and their effect on floc formation and bottom sediments nutrient releases, could help to rectify this problem but was beyond the scope of the present study.

The estimates used for land mitigation align well with expectations of mitigation cost, and should offer a fair representation of the average cost per kg of phosphorus and nitrogen reduction (cf. Abell et al. 2011). The sensitivity analysis (Section 4.3.5) has shown that changes in mitigation cost have only a moderate impact on the overall mitigation costs for the catchment. Costs estimated here were deemed representative and appropriate to offer insight into the different costs and benefits of the scenarios.

Land values and ecosystem services values are based on best estimates found in relevant literature, but refinement of parameter inputs based on local information and targeted studies could narrow the range of outputs in this analysis. No such data were available at the time of research, and primary data collection on economic values was beyond this assessment, which had a focus on integrating biophysical modelling with ecosystem services valuation, as well as the economic implications of land use change and land mitigation practices.

Based on the data sources used for the valuation, no distinction was made between native forest and exotic cover; the latter being used for commercial harvesting. No data were available on native forest values for the land value estimates, and the ecosystem services study used as a basis to value ecosystem services provided by land use types also did not distinguish any particular types of forest cover (Patterson and Cole 2013). A study of ecosystem services values conducted in a nearby catchment found that there was little difference in value between these two land use types (Velarde and Yao 2014). Ecosystem services valuation, in particular linked to land use, is subject to high levels of uncertainty due to the relatively early stage of valuation research (Johnson et al. 2012). Further research into valuation of these services would greatly benefit questions of land use trade-offs.

With regards to the land use change options, a simplistic approach of 100% and 50% land use change was used for this analysis, but many options and combinations of land use change and mitigation are conceivable. It is also possible to consider reducing areas of less profitable land use types first; for this analysis, it was found that the conversion of less profitable land uses such as dry stock was not sufficient to achieve the required nutrient load reduction. Targeting those areas within a catchment that are more prone to nutrient loss could make use of knowledge about “critical source areas” where it is possible to reduce nutrient losses more cost-effectively (McDowell and Nash 2012; Parsons et al. 2015). These critical source areas (CSAs) are most likely to contribute to nutrient losses. For example raceways, pugged areas and tilled paddocks are CSAs that can lose about 80% of nutrients from just 20% of the farm area (Sharpley et al. 2003). A spatial analysis of nutrient reduction to account for these differences could add to the options presented here and is specific to management practices rather than land use type. More detailed spatial analysis that considers the configuration and topography of the catchment could also add further value to this study by considering variations in land use and land values throughout the catchment.

4.4.4 Impacts of alum dosing

For current land uses and for potential land use change and mitigation options, in-lake mitigation in the form of alum dosing is required to meet the desired water quality outcome. Without alum dosing, the only management option that met the water quality target was that of converting all intensive land use to forest, which resulted in a simulated TLI of 3.63. Reducing nutrient loads from the catchment to 435 t of N and 37 t of P without alum dosing yielded a simulated TLI of 4.44, falling short of the target. This shows that the current management strategy of the inflow dosing is highly effective at reducing the TLI of the lake. However, the simulation of a lower phosphorus load than suggested by the regional policy plan (reduction of phosphorus input to 30 t annually, option 6) shows that this could suffice to meet the water quality target without alum dosing (see Section 4.1).

Alum has been shown to be a useful restoration tool in various lake restoration projects (van Hullebusch et al. 2002, Hupfer et al. 2015), especially where urgent measures are needed to remedy severe ecosystem degradation evident as potentially toxic cyanobacterial blooms (Hupfer et al. 2015). However, studies of the acute toxic effects of alum on aquatic biota, including fish, aquatic invertebrates, plants, and amphibians, have demonstrated potential for lethal impacts (Gensemer and Playle 1999; Van Hullebusch et al. 2002; Clearwater et al. 2014). By contrast, the long-term effects of continuous low-level dosing of alum are largely unknown and the ultimate fate and reactivity of aluminium ions remain of concern (Tempero 2015). Many studies have demonstrated the cost-effectiveness of alum in terms of cost per kg of phosphorus removed (Huser et al. 2015). On the basis of the unknown long-term effects of continuous alum dosing, mitigation options beyond alum dosing should be considered for long-term management of Lake Rotorua. Implementing additional options to improve water quality will also decrease the reliance on an environmental engineering solution for the desired water quality outcomes. A focus on engineering to improve ecosystems has been questioned with regards to the complexity of processes associated with geo-engineering for lake water quality improvements and uncertainties around long-term effectiveness (Spears et al. 2014). Recommendations have been made that geo-engineering should only be used when other long-term mitigation options, primarily external nutrient reductions, are already in place. In addition, it is possible that Te Arawa iwi will not agree to the renewal of the consent for alum dosing, which may result in its ongoing use being subject to legal contest.

4.4.5 Alternative management strategies

Alternative strategies to manage nutrients in the lake may include sediment dredging and floating wetlands. Sediment capping and the diversion of the largest stream inflow to the lake outlet have been considered in earlier modelling assessments (Hamilton et al. 2012). The cost and limited relative effectiveness of these options currently constrain their applicability. While best economic and water quality outcomes in this study were found to be associated with changing land use of the entire catchment, a combination of catchment measure to reduce nutrient loads and in-lake mitigation appears to be an alternative cost-effective approach to achieving the environmental outcome intended by authorities (a TLI just above the policy target). This aligns with a recently proposed strategy taken to manage nutrient loads from the catchment and inventories in stream inflows and the lake. A policy has recently been proposed to combine incentivised land use change with best management practice on land and continued low-level alum applications in the lake (Rotorua Lakes 2015). However, my assessment shows a very similar cost was associated with an option of reducing phosphorus loading to 30 t annually. This option appears to be sufficient to meet the water quality target without alum dosing, with a similar overall cost associated, in particular where nutrient reduction is achieved through land mitigation (Figure 12).

Lastly, the impact of an upcoming decision on how to manage treated wastewater for the city of Rotorua could affect future management options of the lake. Wastewater was a major driver of water quality decline in the lake in the 1980s and contributed a large proportion of the total nutrient load (Rutherford 1984). Since 1991, treated wastewater has been applied to land in an exotic forest within a subcatchment of Lake Rotorua (Wang et al. 2015), but the resource consent for this scheme is coming up for renewal in 2017 and the forest owners (CNI Forest Management Ltd.) have already negotiated with the owners of the wastewater infrastructure to cease using the current forest disposal site. Depending on the method used for future disposal and its effectiveness in removing nutrients, it is likely that there will be an impact on the level of alum dosing required to maintain water quality levels, with an increase in dosing levels possibly required where saturation of the forest soils leads to increasing phosphorus inputs into the lake from this subcatchment.

Alternative management strategies are not only important for the present case study. Throughout New Zealand, and on a global scale, it is important to find ways to sustainably manage freshwater resources in the long-term. Sustainability here refers to all layers of social, ecological and economic well-being: ensuring that the resources that sustain human populations, economic growth and our ecosystems are managed in a way that they continue to do so into the future. Regulatory decisions are required to integrate aspects of ecological and economic constraints and complexities in policy and planning. Ecological preservation and restoration are not merely a question of biodiversity or conservation; they are ultimately directly linked to economic growth, which is dependent on ecosystem health (Scheffer et al. 2000; Keeler et al. 2012).

4.4.6 Limitations

Baseflow in stream inflows to Lake Rotorua is dominated by groundwater. The mean groundwater transport time of baseflows across the catchment has been estimated to be around 50 years and as long as 145 years in the largest inflow (Morgenstern et al. 2015). These lags have important implications for management of the lake as well as the timeframes within which management targets for nutrient load reductions can be achieved. Due to the time period studied here, groundwater lags have not been included in the analysis. Lake Rotorua is characterised by relatively high natural loads of phosphorus due to dissolution of phosphate ions from rhyolitic pumice and ignimbrite of volcanic origin. By contrast, nitrogen loads are naturally low but increasing progressively as nitrate in groundwater accumulates in response to intensification of land use in the catchment (Morgenstern et al. 2004). Despite relatively high natural P loading, eutrophication of the lake was accelerated by human activities in the catchment (Hamilton 2005, Abell et al. 2011, Tempero et al. 2015). These factors add to the complexity of lake management but the evolving scientific knowledge (Abell et al. 2015, Hamilton et al. 2015, Morgenstern et al. 2015) is being used to define time frames for management actions to be effective.

Management approaches to reduce nutrient loads from land, in particular those employing land use change, have important socio-economic impacts on landowners and farm operators within the catchment. Both mitigation and land use change have impacts on farming operations, profits and land values, and uncertainty for the operators, with potential for regional-scale effects (Parsons et al. 2015). While the conversion of the entire catchment into native cover is unrealistic, including on socio-economic grounds, the evaluation still raises interesting questions around values and profitability of the catchment as a whole. Some land use change studies have shown considerable losses to land owners as a result of land use change (Bell et al. 2004), but when ecosystem services as well as land values are considered, a component of land use change offers benefits as part of the wider long term management of the lake.

4.5 Conclusion

This study has shown that when considering the value of ecosystem services, intensive agricultural land use is not necessarily the most profitable type, and a shift towards alternative land uses within a catchment can lead to both economic benefits and improvements in water quality. In this context, land use change offers an option for water quality improvement that minimises lake and land mitigation costs, while adding value to catchment land use. However, this option may have considerable social and economic limitations. A combination of mitigation options in the catchment to reduce nutrient run off and leaching could be an alternative cost-effective approach to achieving the environmental outcome intended by authorities (a water quality target in this case study), when compared to land use change-only or mitigation-only options. If the phosphorus reduction target is lowered to at least 30 t per year, water quality targets may also be achieved without reliance on alum dosing. The combination option aligns with the current management strategy taken to reduce nutrient loads from the catchment and manage nutrient inventories in the lake, including a proposed policy plan change that combines an amount of incentivised land use change with best management practice on land and continued alum application in the lake.

Findings of this research are important in a national context where new regulatory approaches are being developed to manage freshwater resources. In this context, many initiatives currently fall short or fail, often based on economic arguments. Results of this study indicate that land value losses for land use change options outweigh mitigation costs, especially where value changes in ecosystem services are considered as part of the equation. This study has shown that when taking into account ecosystem services provisions, considerable gains are associated with some degree of land use change to less intensive forms. Improved water quality could add to these gains. It has been demonstrated how an economic and ecological argument for alternative catchment management approaches offers wider economic and water quality benefits for lakes.

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5 Concluding discussion

5.1 Research summary

This research was set in the context of an interdisciplinary study of Lake Rotorua and its catchment, integrating economics of land use, lake management, implications of historical policy and future options for policy and planning. The overall objective of this thesis was to provide an integrated understanding of lake restoration, including the ecological processes within the catchment and the lake, the economic context, as well as barriers for management and policy practices. A specific aim was to consider options to restore water quality of Lake Rotorua, based on management within the lake and the surrounding catchment. Finding answers to questions around barriers leading to failures in freshwater management, the economic context and the efficacy of policies addresses matters of both significance and urgency for Lake Rotorua (Rotorua Lakes 2015), New Zealand (MfE 2014), and the international context (Turner et al. 2000; Foley et al. 2005; Xepapadeas 2011).

As discussed in the introductory chapter (Chapter 1), freshwater resources nationally and globally are facing a range of threats. These resources are of major socio-economic importance and maintaining them is critical to the continued survival of the human species. This thesis offers an important contribution to illustrate options of reconciling economic and environmental interests in lake management. It shows where environmental preservation may serve important social and economic interests; where singular pursuit of economic growth may ultimately harm the economy; and where there are opportunities for enhanced, more cost-effective policy and management approaches.

The objectives of this thesis were to offer an integrated analysis by (1) exploring mechanisms and a number of drivers of the freshwater management process, historically and in present times, in the context of Lake Rotorua, (2) developing a general framework to assess ecosystem services and associated economic values of the lake ecosystem that are currently not valued by markets, and (3) evaluating the cost-effectiveness of options to manage nutrient loads to

Lake Rotorua with a focus on both environmental outcomes and economic benefits for the catchment and the lake.

Environmental changes in Lake Rotorua have been studied extensively, commencing in the 1920s (Phillips and Grigg 1922). Degradation and its causes have been analysed, and management responses have attempted to address water quality problems. To understand the dynamics of restoration efforts and failures to restore the lake, objective (1) was addressed in Chapter 2 to set the context of the thesis, analyse historical changes and make recommendations towards future management. The study showed that management responses were mostly reactive and characterised by failure to halt ecosystem decline. Lag times between the recognition of this decline and actions taken to counter it were a major contributor to restoration failure (Mueller et al. 2015). Additionally, responses were more likely to be taken when environmental degradation was strongly visible in the public sphere. The delay in restoration and conservation actions has major implications for many environmental issues at a global scale; degradation is often allowed to advance substantially without the intervention of timely and appropriate responses. Only when the public is significantly affected and awareness is created, is action taken. This bears many similarities to the debate around climate change (Bulkeley and Newell 2015), amongst many others. My study showed that lake restoration efforts can benefit from a move away from reactive management and an increase in the visibility of environmental problems, supported by an integrated approach. New regulatory approaches taken to manage Lake Rotorua are attempting to provide a level of integration and visibility of the issues, as discussed below in Section 5.2.

To investigate an economic context for restoring lakes, objective (2) of establishing a general framework for assessing ecosystem services was addressed in Chapter 3. Through using ecosystem services as an assessment tool, the study showed that a significant economic value is associated with an ecosystem such as Lake Rotorua but this value is not currently accounted for in the management of either lakes or catchments, especially in New Zealand. This work also offered a general framework to value lake ecosystem services for future studies. It included an assessment of potential damage costs associated with water quality decline. There were some limitations associated with the methods used (Chapter 3), which included the presently limited understanding of integration of economic, social and ecological values within an economic framework, the lack of data available on ecosystem services provision and their economic value, and the difficulty of representing complex ecological processes in an economic valuation context. Comparison of the results with existing New Zealand and overseas case studies showed a close alignment of dollar values. The dollar values given in my study tended to be conservative relative to other valuations, but alignment provided partial validation of my approach. The study showed the significant economic value associated with the Lake Rotorua ecosystem (NZ$100-145 million annually), as well as potential losses associated with its degradation (NZ$14.4-51 million annually for a change in Trophic Level Index from 4.1 to 4.8). Thus there is an economic justification for restoration. Such an assessment would be valuable when applied to study other lakes in New Zealand as part of a justification of the benefits of managing these lakes. This is of particular significance in the context of the new freshwater management reforms (National Policy Statement, MfE 2014), which requires lakes and other water bodies to be managed in a way that their water quality is either maintained or improved. Currently, the implementation of this policy is seen largely as a cost to farmers and the community; however, findings of this study suggest that improvements in water quality can ultimately negate at least some of these losses through economic gains associated with enhanced ecosystem services.

Objective (3) relating to the cost-effectiveness of options to manage nutrient loads was addressed in Chapter 4 through an integrated analysis of lake and catchment nutrient reduction options. The most cost-effective option to manage the lake and catchment, which also met the water quality target set by policy makers, was a combination of land use change, nutrient mitigation on land and mitigation actions within the lake (in the form of alum dosing). A number of concerns are associated with the continuous application of alum to the lake, including possible long-term ecotoxicity and cultural impacts as the addition of this chemical to the lake is regarded as undesirable, or unacceptable, by many representatives of local iwi. Alum is therefore not likely to be a suitable long-term engineering solution for excess nutrients from the catchment (Chapter 4). Additional water quality simulations showed that a reduction of phosphorus to an annual load of 30 t from the catchment could be sufficient to meet the water quality target without the need for continued alum dosing.

Land use change, balanced carefully, is the most sustainable long-term method to achieve the restoration goals for the lake. A balance would need to account for social implications of land use change, including compensation, and allow for appropriate timeframes for change to occur. A recently proposed management plan suggests this should be achieved through a combination of in-lake mitigation, catchment management (gorse removal and improved farm management practices), and land use change supported by a financial incentives scheme (Rotorua Lakes 2015). Water quality simulations conducted in Chapter 4 show that the water quality target can indeed be achieved, but only through coupled N and P mitigation strategies, requiring a phosphorus load reduction to a maximum of 30 t P annually lost from the catchment. A continuation of alum dosing was not required according to the findings of this simulation.

The ecosystem service values of food, biodiversity, nutrient processing, aesthetics and recreation were calculated at $100-145 million per year for Lake Rotorua. Ecosystem services associated with the catchment and based on the current land use types were $176 million annually. These values are currently unaccounted for in markets, but are significant in the wider context of how Lake Rotorua and its catchment will be managed in the future. There is also potential for further economic gain associated with restoration measures of both the lake and catchment. The gains can be assessed from the costs of mitigation as well as their ecological effect, in terms of progress towards desired water quality outcomes. Linking the lake to the catchment is an important exercise for such an integrated assessment, but is not commonly undertaken (e.g., Ticehurst et al. 2008).

In summary, findings from this thesis show that there is a strong economic argument for lake restoration. Ecosystem services of a lake and its catchment have significant economic values that are worthy of preserving and enhancing. In the long term, improved water quality in Lake Rotorua needs to be achieved with strict reductions in both nitrogen and phosphorus loads from the catchment. The recently proposed integrated management plan (Omundsen 2013; Rotorua Lakes 2015) attempts to achieve this load reduction through best management practices on intensive agricultural land use, land use change supported by an incentives scheme, and engineering solutions. Groundwater lags (Morgenstern et al. 2015) mean that it will be some decades before these changes are reflected in an acceptable and stable water quality in the lake.

5.2 Context of research findings

This study provides an integrated assessment of restoring a lake ecosystem that bridges multiple disciplines. Benefits of interdisciplinarity are widely acknowledged (Wilson and Carpenter 1999; Adger 2000; Scheffer et al. 2000; Anton et al. 2010), yet rarely achieved. Assessing aspects of resource management across ecological, economic and policy disciplines is required to improve decision-making and support effective environmental management outcomes in the long-term. In the context of global environmental management questions relating to lakes, this research has shown that the move away from intensive agricultural land use may offer economic benefits coincident with ecological improvements. For New Zealand, this has bearing on the need to find alternative ways for profitable land use and may change job markets towards supporting tourism or recreational activities rather than agriculture, or more closely examine novel options for productive land use such as growing manuka (*Leptospermum scoparium*) for high quality honey production (Funk and Kerr 2007; Edlin and Duncan 2012).

Freshwater management in New Zealand is currently under review, and often subject to criticisms such as the financial burden placed on farmers to reduce their environmental impacts. However, findings of my work show that there is an economic justification for changes in the way New Zealand's land is managed, and that intensification is not the only option for economic growth. Research can inform decision-making based on better integration of all sectors. The analyses can be conducted in a collaborative framework to offer a better understanding of different types of resource management decisions. Global (under legislation such as the European Water Framework Directive) and national level (pertaining here to New Zealand) frameworks to manage freshwater resources can benefit from systematic integration of planning and decision-making under a holistic approach that also includes ecosystem services (Vlachopoulou et al. 2014). In this context, methodology and findings of my study can be used to better align future decision-making and policy plans to cost-effective freshwater management that can achieve the desired water quality and ecosystem health outcomes.

5.3 Implications for Lake Rotorua's management

5.3.1 Management of nutrients in Lake Rotorua

This thesis outlines the important role that alum dosing has played to improve and maintain water quality in Lake Rotorua. Alum is currently applied to two inflows and it reduces the Trophic Level Index (TLI) of the lake by approximately 0.5 TLI units. It has a significant impact on water clarity and therefore visual appeal. However, continuous application of alum may have long-term eco-toxicological effects that are not fully understood or predictable (Tempero 2015). Alum dosing is also a culturally sensitive issue as iwi and hapū are fundamentally opposed to applications of 'foreign' chemicals to freshwaters. Therefore, it is recommended that additional management strategies be considered. A number of in-lake techniques have previously been investigated for improvements in water quality. These include sediment dredging (Miller 2007), installation of a floating wetland (Opus 2010; Tanner et al. 2011) and artificial capping of the bottom sediments in order to reduce the release of nutrients (Ozkundakci and Hamilton 2006). These options offer limited opportunity to support lake restoration based on relatively high costs per unit of nutrients removed (Miller 2007; Opus 2010), as well as potential impacts on aquatic biota (Kusabs and Butterworth 2011; Özkundakci et al. 2011). More research into the effects of alum dosing and cost-effective alternatives for in-lake water quality remediation are required to continuously inform future management whilst alum dosing is used and the lake water quality is vulnerable to decline without this remediation action.

5.3.2 Management within the catchment

This study has shown that continuous efforts to restore Lake Rotorua, and lakes in general, offer economic benefits. Values associated with both the catchment and lake components of the ecosystem are substantial, and should be protected. It is recommended that future management of lakes and restoration actions should take these economic values into consideration – not only for Rotorua – but more generally where lake restorations are contemplated across New Zealand. The valuation suggests that restoration of Lake Rotorua offers economic benefits, in particular to recreational and tourism industries. Tourism in particular is highly significant for the economy of Rotorua and is based around iconic Lake Rotorua and other Te Arawa lakes of the region (Chapter 1.2.4; Butcher et al. 2000; Edgar 2008). In Chapter 3 it was shown that nutrient reductions could increase economic value of the lake due to improved water quality. Moreover, there are economic values associated with land uses such as indigenous vegetation that are commonly ignored. Therefore, the benefits gained from both in-lake restoration and some conversion of intensive agricultural land use to indigenous vegetation could justify, and potentially even pay for, some of the economic losses incurred from reduced intensive agriculture. Tourism and recreational sectors could either directly (through taxes or levies) or indirectly (potentially through economic growth in the region) assist with compensation associated with loss of agricultural land. Alternative land uses besides pastoral farming could also provide economic benefits, including manuka honey production, which shows significant economic potential and could provide other benefits such as carbon sequestration by manuka (Funk and Kerr 2007; Funk et al. 2014). Considerations of alternate land uses provide an opportunity to enhance ecosystem services, including opportunities to reduce nutrients losses to sensitive freshwater ecosystems.

The integrated management plan for Lake Rotorua (Rotorua Lakes 2015) incorporates a nitrogen load reduction of 100 t per year from voluntary land use change and an incentives scheme, and funding of up to $40 million to achieve an overall reduction target of 320 t of N per year. Findings of Chapter 3 show that the water quality target can also be achieved without reliance on alum dosing, by a reduction of phosphorus loads to 30 t annually and at a similar cost to the option of reducing phosphorus to 37 t per year with continued alum dosing (both options are based on simulations of a catchment load of 435 t of nitrogen annually).

A second recommendation arising from Chapter 2 in this study is a closer integration of science with policy making, to help contribute to overcoming the lag times in management responses discussed in Chapter 2. While a strong collaboration exists between scientists and decision-makers in the governing authorities for lakes in the region, there may be opportunities to support more targeted research to answer questions around best management approaches. It could be useful to increase the focus on integration of a wider range of disciplines, including economics, ecology and resource management. It could also be beneficial to increase the presentation of scientific findings and knowledge to decision-makers and the general public, in order to promote a faster and more effective process towards restoring Lake Rotorua, considering the range of stakeholders and interest groups.

Catchment management strategies could first consider those areas of the catchment that are more vulnerable to nutrient leaching (Parsons et al. 2015). Within the catchment, areas have been identified that have higher nutrient losses due to soil characteristics, slope and other features. Commencing mitigation and land use change in these vulnerable zones may offer a cost-effective path towards nutrient loss reductions (Parsons et al. 2015). Vulnerable areas are also found at a scale of individual farms. These zones are often referred to as Critical Source Areas (CSAs), where roughly 80% of the contaminants can be lost from 20% of the catchment area (Sharpley et al. 2003). They are important zones to target initial efforts to mitigate nutrient run off in a cost-effective way.

5.4 Recommendations for future work

5.4.1 Ecosystem health indicators

It is noted that ecological studies of lakes often focus on biogeochemical indicators of water quality, such as use of the TLI as trophic state indicator, rather than broader indices of ecological health such as ecological integrity, a concept that refers to structural and functional components of an ecosystem, indicating its condition (Özkundakci et al. 2014). Ecological integrity is challenging to quantify, but its decline is denoted by measurements such as higher trophic state, higher pH in surface waters, lower light penetration, higher proportion of exotic fish species and lower rotifer diversity (Drake et al. 2011). Internationally, progress has been made to develop practical indicators of biodiversity and ecological integrity and to help those aspects being implemented in policy and planning approaches (Nel et al. 2009).

Besides means to represent ecological integrity via single variables or indices using multiple factors, there are a number of other ecosystem health indicators such as the macroinvertebrate community index (MCI) (Stark 1993), the Rotifer TLI (Duggan et al. 2001), LakeSPI (Submerged Plant Indicators) (Özkundakci et al. 2014), and the Cultural Health Index (CHI) (Tipa and Teirney 2006) which can inform on ecological health of the lake and provide assessments of the effectiveness of long-term management. Use of the CHI as an ecosystem health indicator can also offer insight into past and present state of an ecosystem that is not normally amenable to scientific assessment, such as the loss of species and baseline ecosystem information carried as traditional knowledge through generations (Robb 2014).

Complex interactions between environmental and biological factors within lakes, such as cascading food web interactions and climatic conditions, determine the resilience of a lake ecosystem to anthropogenic pressures in its catchment (Özkundakci et al. 2014). Future research could benefit from considering a wider range of ecosystem health indicators, which could offer a more integrated evaluation of the lake ecosystem, its environmental pressures and ways of assessing and informing restoration measures.

5.4.2 Cultural values of lakes

Lake Rotorua, like many other lakes in New Zealand and globally, has significant cultural values (Stafford 1986; Stafford 1988; Edgar 2008; Kusabs and Shaw 2008; Kusabs 2015). No cultural values were directly accounted for in this research due to limitations of the scope of this study and for reasons of cultural sensitivity. As discussed in Chapter 3, cultural values of water bodies such as Lake Rotorua that are of importance to tangata whenua (indigenous people) include the maintenance of the mauri (life giving principle) of the water, the provision of key species for mahinga kai (customary food gathering for sustenance and gifting), and the preservation of the mana (authority, honour, prestige) (Kusabs and Shaw 2008). Each ecosystem has its own individual set of values based on the unique background and history of its people (Tipa and Teirney 2006; Te Aho 2010; Robb 2014). In addition, the quantification of values such as mauri itself, is at times seen as diminishing its significance, and would therefore be culturally insensitive, if not a prohibitively controversial, exercise (Robb 2014). Any study of these spiritual values should be based on the principles of mātauranga Māori (traditional indigenous knowledge) in order to fully understand and capture the importance of these values. By definition, mātauranga Māori research is conducted by suitably mandated Māori researchers who hold mana whenua status (power associated with possession and occupation of tribal land) (Henry and Pene 2001). Therefore, the inclusion of cultural values of the lake in this study would have been culturally inappropriate. It is suggested that future research should be conducted in this area guided by the principles of the local Te Arawa Cultural Values Framework, which outlines the importance of the Māori worldview (Te Arawa Lakes Trust 2015). This research direction would provide broader understanding of the cultural and spiritual values of the lake. The use of mātauranga Maori can also inform scientific studies by offering tools for a more integrated assessment, including the use of traditional methods for surveying key biota (Kusabs 2015), and the possibility to gather additional information on ecological changes of water bodies based on traditional knowledge (Robb 2014).

It is important to consider cultural values for all management and restoration decisions. Much could be learnt from allowing policy and management plans to be informed by mātauranga Māori. A more integrated perspective, combined with a diverse range of assessment and research techniques, can be a key to successful long-term management of the lake, and freshwater resources in general.

5.4.3 Economic valuation

The economic analysis applied in this study could be expanded to include additional site-specific details as they become available. There is a need to include additional information on ecosystem services of the lake and the catchment, as well as land values and potential changes in values estimated as damage costs in Chapter 3. Ecosystem services valuation is a rapidly expanding field (Atkinson 2012; Boeraeve et al. 2015) and new research reflects imperatives to continuously improve and adjust techniques used to value ecosystem services in order to establish international standards (Haines-Young and Potschin 2011; Crossman et al. 2013). Refinements to methods of valuation, assessment of damage costs, and land values in the catchment would strengthen the basis for decision-making and could also contribute to the reduction in social time lags by supporting more effective development of policy and regulation.

The valuation conducted in this study for Lake Rotorua could also be transferred elsewhere in New Zealand and internationally. Improved valuations offer an opportunity to further validate the systematic set of steps developed here and reduce the uncertainty and range of values that was estimated for Lake Rotorua's ecosystem services in Chapter 3 ($100-145 million annually). Lakes offer a wide range of ecosystem services, and current anthropogenic pressures lead to a decline not only in lake ecosystem health, but also the provision of these services and therefore natural capital (Schallenberg et al. 2013).

5.4.4 Science communication and decision-making

In order to improve the communication of scientific findings to decision-makers and the general public, future research could focus on the creation of tools that aid in the 'observability' of environmental degradation to concerned parties. Such tools could include models, simulation outputs, interactive maps or other innovative means to illustrate and explain the current environmental conditions and the context for lake restoration. Through better communication, environmental problems could be understood and recognised before decline is costly and difficult to reverse, for example where lakes transition into an alternate, more degraded state (Scheffer et al. 2001; Scheffer and Carpenter 2003; Folke et al. 2004). Increased awareness of conservation and restoration needs of natural resources has been reflected in a growing trend of community action to address these needs (Pretty and Ward 2001). In New Zealand, oftentimes resource management agencies are at least partially reliant on the activities of community groups in supporting their restoration outcomes (Peters et al. 2015).

Understanding of environmental issues and their consequences to ecosystems alongside socio-economic well-being dependent on the services ecosystems provide can also be increased in the public sphere through providing opportunities for education, environmental campaigns, and open debate. Offering tools to increase understanding, such as the range of drivers of environmental change within a lake and catchment, as well as options for restoration, could mean that all stakeholders could more easily engage with the decision-making process, thereby decreasing social lag times and increasing public input towards enhanced natural resource management.

5.5 References

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Appendix I Water quality parameters (Chlorophyll a, total phosphorus, total nitrogen and Secchi depth), over the time period 1967-2013.

Year	**Chlorophyll *a* (mg m^{-3})**	**TP (mg m^{-3})**	**TN (mg m^{-3})**	**SD (m)**
1967				2.6
1968				2.6
1969				2.5
1970				2.2
1971				1.6
1972		26.9	279.5	2.3
1973		30.7	232.4	2.7
1974		36.8	210.6	2.3
1975	14.0	35.6	326.5	2.3
1976	11.1	28.5	335.0	2.2
1977	10.7	28.7	297.9	2.4
1978		29.8	275.4	3.4
1979		33.9	315.3	2.8
1980		40.9	431.8	2.2
1981	20.8	37.2	463.0	2.3
1982	29.6	54.1	502.1	2.0
1983	16.4	45.1	375.7	1.8
1984	18.6	53.7	477.9	1.8
1985	18.6	79.7	515.9	1.8
1986	13.0	51.8	399.6	2.4
1987	8.9	41.6	372.6	2.5
1988	10.5	49.8	469.2	2.2
1989	19.8	45.4	461.1	2.4
1990	19.2	44.5	472.3	2.1
1991	13.6	31.6	366.8	2.6
1992	10.9	39.0	389.5	2.6
1993	10.3	49.0	392.6	2.5
1994	6.4	93.4	445.5	
1995	7.9	46.7	393.8	
1996	11.1	26.9	354.2	
1997	18.7	35.9	576.4	
1998	16.8	37.6	383.4	
1999	5.2	27.2	378.6	
2000	13.3	32.2	586.1	
2001	27.2	43.9	406.1	
2002	21.2	46.1	421.8	
2003	25.0	46.5	477.8	
2004	25.3	46.4	555.3	

Year	**Chlorophyll *a* (mg m^{-3})**	**TP (mg m^{-3})**	**TN (mg m^{-3})**	**SD (m)**
2005	16.5	38.8	374.9	
2006	26.9	42.2	511.7	
2007	22.3	31.4	461.5	
2008	15.3	29.3	456.8	
2009	23.4	42.8	468.7	
2010				
2011	13.0	14.0	303.8	3.3
2012	7.3	20.2	324.5	3.6
2013	17.3	22.6	348.1	2.9

All values are measured yearly averages. Data provided by NIWA (1967-1993) and BOPRC (1993-2013).

Appendix II List of Research Publications

ID#	Title	Year	Publication	DPSIR
1	The geochemistry of the thermal lakes, North Island, New Zealand, in relation to problems bearing on the acclimatised Salmonidae	1922	*New Zealand Journal of Science & Technology*	S
2	The Cyanophyceae of the Thermal Regions of Yellowstone National Park, USA, and of Rotorua and Whakarewarewa, New Zealand: With Some Ecological Data	1938	N/A	S
3	Thermal stratification in some New Zealand lakes	1957	*New Zealand Ecological Society Proceedings*	S
4	A preliminary study of some New Zealand lakes	1958	*Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*	S
5	A limnological study of some New Zealand lakes	1959	N/A	S
6	Lake Rotorua weed	1960	Unpublished Report	S
7	Some planktic Staurastra from New Zealand	1960	N/A	S
8	Control of water weed in Lakes Rotorua, Rotoiti	1961	Unpublished Report	R
9	Observations on excessive weed growth in two lakes in New Zealand	1963	*New Zealand Journal of Botany*	R
10	Weed survey: Lake Rotorua	1964	Unpublished Report	S
11	Some aspects of the ecology of Rotorua Lakes	1964	Unpublished Report	D
12	Pollution within the Rotorua basin and eutrophication of Lake Rotorua	1965	Unpublished Report	D
13	Weed in Rotorua Lakes	1965	Unpublished Report	P
14	Fluctuations in the chemical composition of two lakeweeds from New Zealand	1966	N/A	S
15	Lake weed at Rotorua	1966	Unpublished Report	S
16	Report on the lake weed problem	1966	Unpublished Report	P
17	Effects of spraying on phytoplankton in Lake Rotorua, 1966	1967	Book chapter	R
18	Rotorua and Waikato Water Weeds: Problems and the Search for a Solution	1967	Book chapter	P

ID#	Title	Year	Publication	DPSIR
19	*Lagarosiphon major* infestation, Lake Rotorua	1967	Unpublished Report	S
20	Background to and policy of Interdepartmental Committee for Control of Lake-weed	1967	Seminar proceedings	R
21	Further results of spraying lake weeds	1967	Seminar proceedings	R
22	Sewage discharge in Lake Rotorua	1968	Unpublished Report	D
23	Observations of temperatures in some Rotorua district lakes	1968	*New Zealand Journal of Marine and Freshwater Research*	S
24	The comparative limnology of some New Zealand lakes: 1. Physical and chemical	1968	*New Zealand Journal of Marine and Freshwater Research*	S
25	Synoptic surveys of lakes Rotorua and Rotoiti	1969	*New Zealand Journal of Marine and Freshwater Research*	S
26	Aquatic vegetation control on Rotorua, Rotoiti and Waikato hydro lakes	1969	Unpublished Report	R
27	The arsenic content of some water weeds from the Rotorua and Waikato lakes	1969	*New Zealand Limnological Society Newsletter*	S
28	Algal diversity in the North Island Lakes Rotoiti and Rotorua	1969	*New Zealand Limnological Society Newsletter*	S
29	Seasonal variation in phytoplankton from Lake Rotorua and other inland waters, New Zealand, 1966–67	1969	*New Zealand Journal of Marine and Freshwater Research*	S
30	Rotorua and Rotoiti zooplankton	1969	N/A	S
31	Eutrophication in Lake Rotorua	1969	*New Zealand Limnological Society Newsletter*	P
32	Lakes: the value of recent research to measure eutrophication and to indicate possible causes	1969	Government Report	P

ID#	Title	Year	Publication	DPSIR
33	DDT in trout and its possible effect on reproductive potential	1969	*New Zealand Journal of Marine and Freshwater Research*	D
34	Factors affecting the recreational use of Lake Rotorua	1970	Unpublished Report	I
35	A history of the lake-weed infestation of the Rotorua lakes and the lakes of the Waikato hydro-electric system	1970	Government Report	P
36	Eutrophication	1970	Conference Proceedings	P
37	Lake Rotorua - abstract	1970	Unpublished Report	S
38	Submerged vegetation of the Rotorua lakes	1970	Unpublished Report	S
39	Lake weed control, Lakes Rotoiti and Rotorua	1970	Unpublished Report	R
40	Nutrient incomes and water quality of Lake Rotorua	1971	Seminar proceedings	D
41	A nutrient budget for Lake Rotorua	1971	Seminar proceedings	S
42	Eutrophication in Lake Rotorua	1971	Unpublished Report	D
43	Lakes Rotorua and Rotoiti: Environmental impact study	1972	Unpublished Report	D
44	Trophic status and research being done on Lake Rotorua by the Wildlife Service of the Department of Internal Affairs	1972	Unpublished Report	S
45	Lake Rotorua and Rotoiti survey	1972	Unpublished Report	S
46	Research from the Fisheries Research Laboratory, Rotorua	1972	Conference Proceedings	D
47	Weed survey - Lake Rotorua	1972	Unpublished Report	S
48	Eutrophication of Lake Rotorua	1972	Unpublished Report	P
49	Some planktic *Staurastra* from New Zealand	1972	*Svensk Botanisk Tidskrift*	S
50	The relation between primary productivity, nutrients, and the trout environment in some New Zealand lakes	1973	*New Zealand Fisheries Research Bulletin*	P
51	Secchi disc readings, Lake Rotorua	1973	Unpublished Report	S
52	*Calamoecia lucasi* (Copepoda, Calanoida) and other zooplankters in two Rotorua, New Zealand, lakes	1973	*Internationale Revue der gesamten Hydrobiologie und Hydrographie*	S

ID#	Title	Year	Publication	DPSIR
53	Copepod production in some northern lakes	1973	*New Zealand Limnological Society Newsletter*	S
54	Water quality report on Ohau Channel and Okawa Bay	1973	Internal Report	R
55	Weed survey - Lake Rotorua	1973	Unpublished Report	P
56	Eutrophication of Lake Rotorua	1973	Unpublished Report	R
57	Report of the Technical Working Party (1973) of the Officials Committee on Eutrophication	1973	Unpublished Report	D
58	*Actinotaenium*, *Cosmarium*, and *Staurodesmus* in the plankton of Rotorua lakes	1973	*Svensk Botanisk Tidskrift*	S
59	Algal flora of some North Island, New Zealand, lakes, including Rotorua and Rotoiti	1974	*Pacific Science*	S
60	Phytoplankton in Lakes Rotoiti, Rotorua, Rotoma, June 1973, May 1974	1974	Unpublished Report	S
61	Rotorua Lakes	1974	*Nature Heritage*	P
62	Report made on Rotorua sewage disposal	1974	*Soil and Water*	R
63	A Contribution to the Biology of *Nitella hookeri* A. Br. in the Rotorua Lakes, New Zealand	1974	*Hydrobiologia*	P
64	A Contribution to the Biology of *Nitella hookeri* A. Br. in the Rotorua Lakes, New Zealand II. Organic nutrients and physical factors	1974	*Hydrobiologia*	P
65	Rotorua phytoplankton reconsidered (North Island of New Zealand)	1974	*Internationale Revue der gesamten Hydrobiologie und Hydrographie*	S
66	Some planktic Staurastra from New Zealand	1974	*Svensk Botanisk Tidskrift*	S
67	Lake Rotorua and its problems	1974	Unpublished Report	P
68	Lakes Rotorua and Rotoiti, North Island, New Zealand: their trophic status and studies for a nutrient budget	1975	N/A	S
69	Upper Kaituna Catchment control scheme	1975	N/A	R
70	Ecology of macrophytes	1975	Book chapter	S
71	Eutrophication and the trout environment	1975	Book chapter	I
72	Phytoplankton of Lakes Rotorua and Rotoiti (North Island)	1975	Book chapter	S

ID#	Title	Year	Publication	DPSIR
73	Lake Rotorua, notes on recent research	1975	Unpublished Report	P
74	A nutrient budget for Lake Rotorua	1975	Book chapter	D
75	Lake Rotorua - notes on recent research: the monitoring of limnological changes	1975	Unpublished Report	P
76	Lake Rotorua - surface water analyses	1975	Unpublished Report	S
77	The benthic fauna	1975	Book chapter	S
78	Hornwort - Lake Rotorua	1975	Unpublished Report	P
79	Light penetration	1975	Book chapter	S
80	Ecology of Ohau Channel	1975	Unpublished Report	S
81	Thermal conditions	1975	Book chapter	S
82	Report on Lake Rotorua problems	1975	Unpublished Report	D
83	Chemical and biological conditions in lakes of the Volcanic Plateau	1975	Book chapter	S
84	Research concerning the limnology of Lakes Rotorua and Rotoiti	1975	Unpublished Proposal	S
85	Rotorua's salvation lies in lake weed	1975	Unpublished Report	R
86	Water movements in Lake Rotorua	1975	Internal Report	P
87	Land use capability assessment of the Kaituna River catchment, Bay of Plenty region, North Island, New Zealand	1975	Government Report	D
88	An appraisal by the Officials Committee on Eutrophication of the report on Lake Rotorua problems	1975	Unpublished Report	D
89	Bioassay of potential limiting nutrients on Lake Rotorua	1975	Unpublished Report	P
90	Phytoplankton in Lake Rotorua and Lake Okareka, and its interaction with aquatic macrophytes	1976	Thesis	S
91	Microbial parameters and trophic status of ten New Zealand lakes	1976	*New Zealand Journal of Marine and Freshwater Research*	S
92	Heterotrophic potentials and trophic status of ten New Zealand lakes	1976	*New Zealand Journal of Marine and Freshwater Research*	S

ID#	Title	Year	Publication	DPSIR
93	Mercury and other heavy metals in trout of Central North Island, New Zealand	1976	*New Zealand Journal of Marine and Freshwater Research*	I
94	Lake Rotorua - measurements of P-max and bioassay of nutrient additions	1976	Unpublished Report	P
95	Nitrogen and phosphorus content of flood waters in the Lake Rotorua catchment	1976	Internal Report	P
96	Report on Lake Rotorua	1976	Unpublished Report	S
97	Nitrogen and phosphorus content of flood waters in the Lake Rotorua catchment	1976	Unpublished Report	P
98	Use of diquat herbicide in the Rotorua Lakes	1976	Unpublished Report	R
99	Weed survey: Lake Rotorua	1976	Unpublished Report	S
100	Report on the role of Lake Rotorua sediments as nutrient sources and sinks	1976	Unpublished Report	P
101	Eutrophication of Lake Rotorua: a review	1977	N/A	D
102	The comparative limnology of some New Zealand lakes: 2. Plankton∗	1977	*New Zealand Journal of Marine and Freshwater Research*	S
103	The physiology, ecology and succession of lakeweeds with respect to increasing nutrient in New Zealand lakes	1977	Conference Proceedings	S
104	The ecophysiology of *Lagarosiphon* in the Rotorua Lakes	1977	Conference Proceedings	S
105	Some aspects of weed control undertaken by Lands & Survey Department in the Rotorua Lakes	1977	Conference Proceedings	R
106	Assessment of the significance of nutrient concentration differences in stream waters from catchments with differing land uses	1977	Unpublished Report	D
107	The spacial distribution of groundwater discharge into the littoral zone of a New Zealand lake	1977	*Journal of Hydrology*	S
108	Biological studies on the Ohau Channel and its delta, Rotorua	1977	Thesis	S
109	Ultraphytoplankton biomass and production in some New Zealand lakes	1977	*New Zealand Journal of Marine and Freshwater Research*	S

ID#	Title	Year	Publication	DPSIR
110	Preliminary measurements of tritium, deuterium and oxygen-18 in lakes and groundwater of volcanic Rotorua region, New Zealand	1977	Unpublished Report	S
111	Chlorophyll production, in response to nutrient additions, by the algae in Lake Rotorua water	1978	*New Zealand Journal of Marine and Freshwater Research*	P
112	The water quality and ecology of sections of the Puarenga Stream and Sulphur Bay, Lake Rotorua - winter 1978	1978	Unpublished Report	S
113	The water quality and ecology of sections of the Puarenga Stream and Sulphur Bay, Lake Rotorua	1978	Unpublished Report	S
114	Nitrate concentrations in the groundwater around Lake Rotorua	1978	*New Zealand Journal of Marine and Freshwater Research*	P
115	A nutrient budget for Lake Rotorua	1978	Internal Report	D
116	Nutrients in flood flows of the Lake Rotorua catchment	1978	Internal Report	S
117	Particulate matter in the tributaries of Lake Rotorua	1978	Conference Proceedings	P
118	Interlaboratory analysis of Lake Rotorua inflows	1978	Internal Report	S
119	An in-shore sediment study carried out on Lake Rotorua	1978	Internal Report	S
120	Lake Rotorua water quality: synopsis of research conducted to date and an outline of proposed mathematical analysis	1978	Internal Report	D
121	Prediction of nitrogen and phosphorus concentration in Lake Rotorua	1978	N/A	R
122	Sediments of Lake Rotorua as sources and sinks for plant nutrients	1978	*New Zealand Journal of Marine and Freshwater Research*	S
123	Recent stratigraphy of sediments in Lake Rotorua	1979	*New Zealand Journal of Marine and Freshwater Research*	S
124	The water quality and biology of sections of the Puarenga Stream and Sulphur Bay, Lake Rotorua - findings and conclusions	1979	Consultant Report	D

ID#	Title	Year	Publication	DPSIR
125	Current reference file of DSIR Science Information Division: New Zealand freshwaters	1979	N/A	S
126	Chemical analysis of Lake Rotorua sediments	1979	Thesis	S
127	Water quality survey of the Waiohewa Stream, Rotorua, summer 1978-79	1979	Internal Report	D
129	Lake Rotorua: the state of eutrophication	1980	*Soil and Water*	P
130	Primary production in Lakes Rotorua, Rerewhakaaitu, and Rotoiti, North Island. New Zealand. 1973–78	1980	*New Zealand Journal of Marine and Freshwater Research*	I
131	Nitrification in the Waiohewa Stream, Rotorua: a microbiologists viewpoint	1980	Internal Report	P
132	Inflows to Lake Rotorua	1980	*Journal of Hydrology*	P
133	Nitrogen and phosphorus in the base flow of the major tributaries of Lake Rotorua	1980	Internal Report	P
134	Nutrients in flood flows of the Lake Rotorua catchment	1980	Internal Report	P
135	Phosphorus load on Lake Rotorua	1980	Internal Report	P
136	The sensitivity of Lake Rotorua, New Zealand, to additions of phosphorus and nitrogen	1980	Conference Proceedings	P
137	The sensitivity to phosphorus and nitrogen loads, of Lake Rotorua, New Zealand	1980	Internal Report	D
138	Trends in Lake Rotorua water quality	1980	Internal Report	S
139	Water quality studies on the Waiohewa Stream - Part 2	1980	Internal Report	P
140	The macroinvertebrate fauna of the Rotorua Lakes	1981	Book chapter	S
141	Lake Rotorua Project (WL3): Closing Report	1981	Internal Report	P
142	Modelling of phosphorus in New Zealand lakes	1981	Internal Report	R
143	Institutional roles. Future Options for the Rotorua Lakes District: The Implications of Alternative Patterns of Environmental Resource Use and Management for the Rotorua Lakes: Progress Report 3	1981	Consultant Report	R
144	Lake quality assessment. Future Options for the Rotorua Lakes District: The Implications of Alternative Patterns of Environmental Resource Use and Management for the Rotorua Lakes: Progress Report 14	1981	Consultant Report	R

ID#	Title	Year	Publication	DPSIR
145	FORLD: Rotorua Project bibliography. Future Options for the Rotorua Lakes District: The Implications of Alternative Patterns of Environmental Resource Use and Management for the Rotorua Lakes: Bibliography	1981	Consultant Report	R
146	Hydrology of the lakes. Future Options for the Rotorua Lakes District: The Implications of Alternative Patterns of Environmental Resource Use and Management for the Rotorua Lakes: Progress Report 7	1981	Consultant Report	S
147	Water weeds and algae. Future Options for the Rotorua Lakes District: The Implications of Alternative Patterns of Environmental Resource Use and Management for the Rotorua Lakes: Progress Report 11.	1981	Consultant Report	R
148	Lake water quality - land use relationships. Future Options for the Rotorua Lakes District: The Implications of Alternative Patterns of Environmental Resource Use and Management for the Rotorua Lakes: Progress Report 10	1981	Consultant Report	D
149	Rapid physiological assays for nutrient demand by the plankton. I. Nitrogen	1981	*Journal of Plankton Research*	P
150	Rapid physiological assays for nutrient demand by the plankton. II. Phosphorus	1981	*Journal of Plankton Research*	P
151	Nitrogen and phosphorus in the Ngongotaha Stream	1982	*New Zealand Journal of Marine and Freshwater Research*	P
152	Lake Rotorua water quality: summary of results of summer 1981-82 monitoring of chemical and biochemical parameters.	1982	Internal Report	P
153	Lake eutrophication in New Zealand—a comparison with other countries of the Organisation for Economic Co-operation and Development	1983	*New Zealand Journal of Marine and Freshwater Research*	P
154	Penetration of Ohau Channel water into Lake Rotoiti	1983	Government Report	S
155	Nutrient load on Lake Rotorua	1983	Conference Proceedings	P
156	A study of pigment and diatoms in a core from Lake Rotorua, North Island, New Zealand, with emphasis on recent history	1984	*Journal of the Royal Society of New Zealand*	P
157	Nitrogen and phosphorus in Rotorua urban streams	1984	*New Zealand Journal of Marine and Freshwater Research*	D

ID#	Title	Year	Publication	DPSIR
158	Trends in Lake Rotorua water quality	1984	*New Zealand Journal of Marine and Freshwater Research*	D
159	Factors affecting the foods and feeding patterns of lake-dwelling rainbow trout (*Salmo gairdnerii*) in the North Island of New Zealand	1984	*New Zealand Journal of Marine and Freshwater Research*	P
160	Factors affecting clarity of New Zealand lakes	1984	*New Zealand Journal of Marine and Freshwater Research*	S
161	Inferred geothermal inflows to Lake Rotorua	1985	*New Zealand Journal of Marine and Freshwater Research*	S
162	Trends in Lake Rotorua water quality	1985	*New Zealand Journal of Marine and Freshwater Research*	D
163	Nutrient demand and availability among planktonic communities—an attempt to assess nutrient limitation to plant growth in 12 central volcanic plateau lakes	1985	*New Zealand Journal of Marine and Freshwater Research*	S
164	Relative importance of clarity determinants in Lakes Okaro and Rotorua	1986	*New Zealand Journal of Marine and Freshwater Research*	S
165	Nutrient demand and availability related to growth among natural assemblages of phytoplankton	1986	*New Zealand Journal of Marine and Freshwater Research*	S
166	Review of diquat use in New Zealand for submerged weed control	1986	Conference Proceedings	R
167	Rotorua sewage disposal: a statement of the significance of phosphorus and nitrogen in the management of Lake Rotorua	1986	N/A	R
168	Inventory of New Zealand lakes. Part 1: North Island	1986	N/A	S
169	Some properties of the lakes with reference to the Rotorua lakes	1986	Unpublished Report	S

ID#	Title	Year	Publication	DPSIR
170	Macrophyte depth limits in North Island (New Zealand) lakes of differing clarity	1986	*Hydrobiologica*	S
171	Water chemistry of lakes in the Taupo volcanic zone, New Zealand	1986	*New Zealand Journal of Marine and Freshwater Research*	S
172	The impact of the Ohau Channel outflow from Lake Rotorua on Lake Rotoiti	1986	Government Report	P
173	Planktonic cyanobacteria in New Zealand inland waters: distribution and population dynamics	1987	*New Zealand Journal of Marine and Freshwater Research*	P
174	Nitrogen and phosphorus in the catchment of Lake Rotorua	1987	Government Report	P
175	Comments on the potential of artificial wetlands to remove nutrients from small scale sewage plants in the Rotorua lakes area	1987	Seminar proceedings	R
176	Circulation and mixing in Lake Rotongaio and Lake Okaro under conditions of light to moderate winds: preliminary results	1987	N/A	S
177	The effects of changes in both the abundance of nitrogen and phosphorus and their ratio on Lake Okaro phytoplankton, with comment on six other central volcanic plateau lakes	1987	N/A	P
178	The significance of phosphorus and nitrogen in the management of Lake Rotorua	1987	*New Zealand Journal of Marine and Freshwater Research*	R
179	The impact of the outflow from Lake Rotorua on Lake Rotoiti	1987	Conference Proceedings	P
180	Environmental assessment of the proposed Lake Rotorua control structure: impacts on fisheries	1988	Unpublished Report	D
181	The links between exotic forest land use and trout growth	1988	Unpublished Report	D
182	Internal nitrogen and phosphorus loads in Lake Rotorua, New Zealand.	1988	N/A	P
183	Statement on the likely impact on Lakes Rotorua and Rotoiti of the proposed spray irrigation of Rotorua sewage effluent	1988	Consultant Report	D
184	Phosphorus reduction required to control eutrophication at Lake Rotorua, New Zealand.	1988	N/A	P

ID#	Title	Year	Publication	DPSIR
185	Management of phosphorus and nitrogen inputs to Lake Rotorua, New Zealand	1989	*Journal of Water Resources Planning and Management*	R
186	Aquatic weed control in the Rotorua Lakes - a technical evaluation	1989	Government Report	R
187	The aquatic vegetation of 15 Rotorua lakes	1989	Government Report	R
188	An ecophysiological evaluation of the growth and nutrition of three submerged macrophytes in relation to lake eutrophication	1989	Thesis	P
189	Assessment of soil conservation work in the Ngongotaha catchment and the implications to Lake Rotorua	1990	Consultant Report	R
190	Aquatic weed control in the Rotorua lakes - a discussion paper on management issues and options	1990	Government Report	R
191	Particulate phosphorus load on Lake Rotorua	1990	Unpublished Report	P
192	Submerged vegetation and spread of *Egeria densa* Planchon in lake Rotorua, central North Island, New Zealand	1991	*New Zealand Journal of Marine and Freshwater Research*	S
193	Lakes overview report	1991	Technical Report	S
194	Lake Rotorua lake nutrient balance report	1991	Consultant Report	R
195	Temperature and BOD differentials inshore and offshore in Lake Rotorua near Ohau Channel	1991	Unpublished Report	R
196	Report of the status of water net (*Hydrodictyon reticulatum*) in New Zealand and options for its control	1991	Government Report	R
197	Bay of Plenty Regional Council Regional Monitoring Network: bathing beach suitability survey 1991	1991	Technical Report	I
198	Rotorua effluent purification project: souvenir handbook	1991	N/A	R
199	Investigation of septic tank effluent disposal in the Bay of Plenty	1992	Technical Report	D
200	The restoration of Lake Rotorua - comment on progress	1992	Government Report	R
201	Lake RotoruaL status report - 1993	1993	N/A	R
202	Bay of Plenty Regional Council Natural Environment Regional Monitoring Network: freshwater ecology programme, lakes component, 1991/92	1993	Technical Report	S

ID#	Title	Year	Publication	DPSIR
203	Assessment of chemical contaminants in the Lake Rotorua catchment	1993	N/A	I
204	Bay of Plenty Regional Council Regional Monitoring Network: bathing suitability survey, 1993	1993	Environmental Report	I
205	Water Quality Regional Monitoring Network Annual Report 1990/91	1993	Technical Report	S
206	Report on rural land use practices in the Rotorua District	1993	N/A	D
207	Pollution of the aquatic biosphere by arsenic and other elements in the Taupo Volcanic Zone	1994	Thesis	P
208	Sediment investigation of the Rotorua Lakes	1995	Environmental Report	S
209	Water Quality Regional Monitoring Network: Lakes report, 1990-1995	1995	Environmental Report	S
210	Review of Rotorua water quality	1995	Consultant Report	S
211	Natural Environment Regional Monitoring Network: bathing suitability survey (1995)	1995	N/A	I
212	Predictions of phosphorus in Lake Rotorua following load reductions	1996	*New Zealand Journal of Marine and Freshwater Research*	R
213	Watershed riparian management and its benefits to a eutrophic lake	1996	*Journal of Water Resources Planning and Management*	R
214	Rotorua lakes algal monitoring 1991-1995	1996	N/A	S
215	The impacts of weed beds and diquat spraying on the freshwater mussel, *Hyridella menziesi*	1996	Consultant Report	R
216	Comparing past and present trophic states of seven Central Volcanic Plateau lakes, New Zealand	1997	*New Zealand Journal of Marine and Freshwater Research*	S
217	Results of monitoring New Zealand lakes, 1992-1996: volume 3 - data and results	1997	NIWA Client Report	S
218	Enterococcal numbers measured in waters of marine, lake, and river swimming sites of the Bay of Plenty, New Zealand	1997	*New Zealand Journal of Marine and Freshwater Research*	I
219	Rotorua lakes summary report	1997	N/A	S

ID#	Title	Year	Publication	DPSIR
220	Algal pigment stratigraphy in four Rotorua lakes: Okataina, Okareka, Okaro and Rotorua	1997	NIWA Client Report	S
221	Nitrogen and phosphorus in streams draining catchments of different landuse in the Rotorua Lakes region	1997	NIWA Client Report	D
222	Predicting the effects of land use on the water quality of the Ngongotaha Stream	1997	NIWA Client Report	D
223	As assessment of the Rotorua Lakes for aquatic weed control (1996)	1997	NIWA Client Report	R
224	Lakeweed management in the Rotorua Lakes: a working guide	1997	N/A	R
225	Results of monitoring New Zealand lakes, 1992-1996: Volume 1 - General findings	1998	NIWA Client Report	S
226	Results of monitoring New Zealand lakes, 1992-1996: Volume 2 - Commentary on results	1998	NIWA Client Report	S
227	Crustacean zooplankton communities in a New Zealand lake during four decades of trophic change	1999	*New Zealand Journal of Marine and Freshwater Research*	S
228	Lake Rotorua and its inputs in the 1990's.	1999	Consultant Report	D
229	A monitoring and classification system for New Zealand lakes and reservoirs	1999	*Lake and Reservoir Management*	S
230	Mercury bioaccumulation in rainbow trout (*Oncorhynchus mykiss*) and the trout food web in lakes Okareka, Okaro, Tarawera, Rotomahana and Rotorua, New Zealand	1999	*Water, Air and Soil Pollution*	P
231	Aquatic weed control in the Rotorua Lakes (summer 1998/9)	1999	NIWA Client Report	R
232	Lake fisheries summer creel surveying - 1999-2000 summer monitoring report	2000	Monitoring Report	I
233	Septic tanks leachate study for Rotorua Lakes	2000	NIWA Client Report	D
234	Nitrogen removal in natural wetlands below the Rotorua land treatment site	2000	Conference Proceedings	R
235	Aquatic weed control in the Rotorua Lakes (1999/2000)	2000	NIWA Client Report	R
236	Rotorua lakes algae report	2000	Environmental Report	I
237	Trophic level index baselines and trends for 12 Rotorua district lakes, 1990 to 2000	2001	Consultant Report	S

ID#	Title	Year	Publication	DPSIR
238	Trophic level trends in 12 Rotorua District Lakes: 1990 to 2000	2001	Conference Proceedings	S
239	Weed management in the Rotorua Lakes	2001	Conference Proceedings	R
240	Rotorua lakes water quality	2001	N/A	S
241	The impact of potential blue-green algal blooms on catchment management in the Rotorua lakes	2001	NIWA Client Report	R
242	The impact of eutrophication on aquatic food webs as it applies to the Rotorua Lakes	2001	Conference Proceedings	I
243	Shallow groundwater chemistry in the Whakarewarewa Forest and its implications for Lake Rotorua	2001	Conference Proceedings	S
244	Environment B.O.P's lake management plans, present and future	2001	Conference Proceedings	R
245	Exploring the links between Lake Rotorua's trout fishery and algal communities	2001	Conference Proceedings	I
246	Opinion on the paper: "The links between exotic forestry land use and trout growth" by Peter Mylechreest (1988)	2001	N/A	R
247	Rotorua Lakes blue-green algae monitoring	2001	Conference Proceedings	S
248	First order estimation of the nutrient and bacterial input from aquatic birds to twelve Rotorua lakes	2002	Consultant Report	P
249	Rotorua lakes water quality	2002	N/A	S
250	Aquatic plant harvesting in lakes for nutrient renovation	2002	NIWA Client Report	R
251	Okawa Bay water quality study	2002	NIWA Client Report	R
252	Septic tanks leachate study for Rotorua Lakes - Stage 2	2002	N/A	D
253	Rotorua Lakes aquatic weed update to January 2002	2002	NIWA Client Report	S
254	An historical and contemporary review of water quality in the Rotorua Lakes	2003	Conference Proceedings	S
255	Rotorua lakes water quality 2002	2003	Government Report	S
256	Linking catchment land use and lake water quality: A review of the Rotorua lakes experience	2003	Conference Proceedings	D
257	Lake Rotoiti-Ohau Channel: Assessment of Effects of Engineering Options on Water Quality	2003	NIWA Client Report	R

ID#	Title	Year	Publication	DPSIR
258	An economic evaluation of land use change options Section B: Economic impact on Rotorua District and Bay of Plenty Region of water quality induced changes to land use and tourism in Rotorua Lakes catchments	2003	Consultant Report	I
259	Rotorua Lakes: Plants tell the tale	2003	N/A	S
260	A review of short-term management options for Lakes Rotorua and Rotoiti	2003	N/A	R
261	Groundwater age, time trends in water chemistry, and future nutrient load in Lakes Rotorua and Okareka area	2004	N/A	P
262	An Economic Evaluation of Water Quality Induced Changes in Rotorua and Rotoiti Catchments	2004	Consultant Report	I
263	An Economic Evaluation of Water Quality Induced Changes in Rotorua and Rotoiti Catchments: Part A) Land Use Change Scenarios, Part B) Macro-Economic Implications	2004	Consultant Report	I
264	Assessment of the performance of nutrient reduction treatments for the Rotorua Lakes	2004	Government Report	R
265	Nutrient Budget for Lakes Rotoiti and Rotorua. Part I: Internal Nutrient Loads	2004	N/A	P
266	Groundwater in the Lake Rotorua Catchment.	2004	N/A	S
267	Rotorua Lakes: Plants tell the tale	2004	Conference Proceedings	R
268	Land use impacts on nutrient export in the Central Volcanic Plateau, North Island	2005	*New Zealand Journal of Forestry*	D
269	Lake water quality perceptions survey	2005	N/A	I
270	Diatom-based models for reconstructing past water quality and productivity in New Zealand lakes	2005	N/A	S
271	Strategies for managing the lakes of the Rotorua District, New Zealand	2005	*Lake and Reservoir Management*	R
272	Groundwater age and chemistry, and future nutrient loads for selected Rotorua lakes catchments	2005	N/A	P
273	Modelling the Effects of Groundwater Lags on Nitrate Inputs to Lakes Rotorua & Taupo, New Zealand	2005	Conference Proceedings	P

ID#	Title	Year	Publication	DPSIR
274	The environmental status and problems facing the Rotorua lakes	2006	Conference Proceedings	P
275	Foreword–Rotorua Lakes 2006	2006	Conference Proceedings	R
276	Views of primary producers in the Taupo and Rotorua catchments: Implications for water quality policy	2006	Conference Proceedings	R
277	The Government's attitude to the restoration of the Rotorua Lakes	2006	Conference Proceedings	R
278	The Value of Ecosystem Services in the Rotorua Lakes Region	2006	Conference Proceedings	R
279	Restoring the Rotorua Lakes	2006	Conference Proceedings	R
280	The Rotorua Lakes Protection and Restoration Action Programme, Proposals, Costs, Progress	2006	Conference Proceedings	R
281	Prediction of future nitrogen loading to Lake Rotorua	2006	N/A	P
282	Recent studies of sediment capping and flocculation for nutrient stabilisation	2006	N/A	R
283	Nutrient Trading in Lake Rotorua: Goals and Trading Caps	2007	Consultant Report	R
284	Towards Design for a Nutrient Trading Programme to Improve Water Quality in Lake Rotorua	2007	Consultant Report	R
285	Rotorua lakes water quality 2006 report	2007	Government Report	S
286	Remote sensing of water quality in the Rotorua lakes	2007	Consultant Report	S
287	Phytoplankton nutrient limitation in a polymictic eutrophic lake: community versus species-specific responses	2007	*Fundamental and Applied Limnology/Archiv für Hydrobiologie*	P
288	Nutrient Trading in Lake Rotorua: Where Are We Now?	2007	Consultant Report	R
289	Groundwater and surface water hydrology in the Lake Rotorua catchment, New Zealand, and community involvement with lake water quality restoration	2007	Conference Proceedings	R
290	Benthic nutrient fluxes in a eutrophic, polymictic lake	2007	*Hydrobiologica*	P
291	Historical and contemporary perspectives on the sediments of Lake Rotorua	2007	N/A	P
292	Icon Lakes in New Zealand: Managing the Tension Between Land Development and Water Resource Protection	2008	*Society & Natural Resources*	R
293	Nutrient Trading in Lake Rotorua: Social, Cultural, Economic and Environmental Issues around a Nutrient Trading System	2008	N/A	R

ID#	Title	Year	Publication	DPSIR
294	Modelling the relative importance of internal and external nutrient loads on water column nutrient concentrations and phytoplankton biomass in a shallow polymictic lake	2008	*Ecological Modelling*	P
295	Nutrient trading in Lake Rotorua: overview of a prototype system	2008	Consultant Report	R
296	Nutrient Trading in Lake Rotorua: Choosing the Scope of a Nutrient Trading System	2008	Consultant Report	R
297	Nutrient Trading in Lake Rotorua: Determining Net Nutrient Inputs	2008	N/A	R
298	Farmers taking control of their future: Research into minimising nitrogen and phosphorus from pasture land into Rotorua lakes	2008	*Carbon and Nutrient Management in Agriculture*	R
299	Sediment and nutrient accumulation rates in sediments of twelve New Zealand lakes: influence of lake morphology, catchment characteristics and trophic state	2008	*New Zealand Journal of Marine and Freshwater Research*	P
300	Nutrient Trading in Lake Rotorua: Cost Sharing and Allowance Allocation	2009	Consultant Report	R
301	Rotorua Lakes water quality report 2009	2009	Government Report	S
302	Te Arawa - Rotorua Lakes Restoration Programme	2009	N/A	R
303	Lakes Rotorua and Rotoiti Action Plan	2009	Government Report	R
304	Nutrient trading to improve and preserve water quality	2009	*Water and Atmosphere*	R
305	Water flow between Ohau Channel and Lake Rotoiti following implementation of a diversion wall.	2009	Consultant Report	R
306	Assessment of interventions for the Rotorua lakes	2009	Consultant Report	R
307	Sediment and nutrient accumulation rates in sediments of twelve New Zealand lakes: influence of lake morphology, catchment characteristics and trophic state	2009	*New Zealand Journal of Marine and Freshwater Research*	P
308	Denitrification capacity of lake sediments across a gradient of catchment land use in Rotorua, New Zealand	2009	N/A	P
309	Managing the lakes of the Rotorua district, New Zealand	2009	*Lake and Reservoir Management*	R
310	Groundwater catchment boundaries of Lake Rotorua	2009	Government Report	S

ID#	Title	Year	Publication	DPSIR
311	Spread and status of seven submerged pest plants in New Zealand lakes	2009	*New Zealand Journal of Marine and Freshwater Research*	P
312	The tale of two lakes: managing lake degradation, Rotorua lakes, New Zealand	2010	Conference Proceedings	R
313	11. Improving lake water quality through a nutrient trading system: the case of New Zealand's Lake Rotorua	2010	Book chapter	R
314	High frequency monitoring and three-dimensional modelling of temporal variations in water quality of Lake Rotorua, New Zealand	2010	N/A	S
315	Comparative Assessment of Water Quality with the Trophic Level Index and the Delphi Method in Lakes Rotoiti and Rotorua, New Zealand	2010	*Water Quality Research Journal of Canada*	S
316	Reducing the external environmental costs of pastoral farming in New Zealand: experiences from the Te Arawa lakes, Rotorua	2011	*Australasian Journal of Environmental Management*	D
317	Prediction of nitrogen loads to Lake Rotorua using the ROTAN model	2011	Conference Proceedings	P
318	Modelling diffuse nitrogen inputs to lakes–groundwater time lags and attenuation in Rotorua and Taupo	2011	N/A	P
319	Water quality management in Lake Rotorua: A comparison of regulatory approaches using the NManager model	2011	Conference Proceedings	R
320	Regulatory and non regulatory options in achieving reduction in non point source pollution in the Rotorua District	2011	Seminar proceedings	R
321	Does complex hyrdology require complex water quality policy?	2011	N/A	R
322	Does Complex Hydrology Require Complex Water Quality Policy? NManager Simulations for Lake Rotorua	2011	Seminar proceedings	R
323	Denitrification potential in lake sediment increases across a gradient of catchment agriculture	2011	*Ecosystems*	P
324	Landsat remote sensing of chlorophyll a concentrations in central North Island lakes of New Zealand	2011	*International Journal of Remote Sensing*	S
325	The Effects of Climate and Land Use Change on Water Quality of Lake Rotorua, North Island, New Zealand	2011	Seminar proceedings	D

ID#	Title	Year	Publication	DPSIR
326	Valuing trout angling benefits of water quality improvements while accounting for unobserved lake characteristics: An application to the Rotorua Lakes	2011	Conference Proceedings	I
327	Using the ROTAN model to predict nitrogen loads to Lake Rotorua, New Zealand	2011	Conference Proceedings	P
328	Detection, identification and toxigenicity of cyanobacteria in New Zealand lakes using PCR-based methods	2011	*New Zealand Journal of Marine and Freshwater Research*	S
329	Evaluation of the Impact of Different Policy Options for Managing to Water Quality Limits	2012	Consultant Report	R
330	Healing waters - cleaning up the Rotorua lakes	2012	*Water and Atmosphere*	D
331	Predicting the effects of nutrient loads, management regimes and climate change on water quality of Lake Rotorua	2012	Consultant Report	P
332	Alum Dosing Lake Rotorua Streams	2012	Government Report	R
333	Comparing Water Quality Trading Programs: What Lessons Are There To Learn?	2012	*Journal of Regional Analysis and Policy*	R
334	Parameterisation of sediment geochemistry for simulating water quality responses to long-term catchment and climate changes in polymictic, eutrophic Lake Rotorua, New Zealand	2012	*Water Pollution XI*	S
335	Nutrient Trading in Lake Rotorua: A Policy Prototype	2012	N/A	R
336	Catchment land use and trophic state impacts on phytoplankton composition: a case study from the Rotorua lakes' district, New Zealand	2012	*Hydrobiologica*	D
337	Are geothermal streams important sites of nutrient uptake in an agricultural and urbanising landscape (Rotorua, New Zealand)?	2012	*Freshwater Biology*	D
338	Identification and testing of early indicators for N leaching from urine patches	2013	*Journal of Environmental Management*	D
339	Ecosystem Services of Lakes	2013	Book chapter	I
340	Memory of the Lake Rotorua catchment-time lag of the water in the catchment and delayed arrival of contaminants from past land use activities	2013	Conference Proceedings	D
341	Managing Risks and Tradeoffs Using Water Markets	2013	N/A	R

ID#	Title	Year	Publication	DPSIR
342	Movement of phosphorus in soil at the Rotorua land treatment system	2013	N/A	D
343	Costs of harmful blooms of freshwater cyanobacteria	2013	Book chapter	I
344	An integrated model for simulating nitrogen trading in an agricultural catchment with complex hydrogeology	2013	*Journal of Environmental Management*	R
345	The performance of Detainment Bunds (DBs) for attenuating phosphorus and sediment loss from pastoral farmland	2013	Thesis	R
346	Overview of using detainment bunds for mitigating diffuse-source phosphorus and soil losses from pastoral farmland	2013	Conference Proceedings	R
347	Growth of rainbow trout (*Oncorhynchus mykiss*) in warm-temperate lakes: implications for environmental change	2013	Conference Proceedings	D
348	Understanding the Practice of Land Use Modelling	2013	Consultant Report	R
349	Quantifying temporal and spatial variations in sediment, nitrogen and phosphorus transport in stream inflows to a large eutrophic lake	2013	*Environmental Science: Processes & Impacts*	D
350	Bioavailability of phosphorus transported during storm flow to a eutrophic, polymictic lake	2013	*New Zealand Journal of Marine and Freshwater Research*	S
351	Variability in nutrient loading to lake ecosystems and associated impacts on water quality	2013	Thesis	D

Appendix III Pearson correlation coefficients for each of the binary explanatory variables, calculated relating to each of the categories of the DPSIR framework. Coefficients were calculated for time lag series of 3 years, 5 years and 10 years after the occurrence year of each explanatory variable. To test statistical significance, the p value for each correlation coefficient was also calculated.

Variable	**D_3**	**P_3**	**S_3**	**I_3**	**R_3**	**D_5**	**P_5**	**S_5**	**I_5**	**R_5**	**D_{10}**	**P_{10}**	**S_{10}**	**I_{10}**	**R_{10}**
Lake Weed S.	0.77	-	0.77	-	0.45	0.41	0.83**	0.70	-	0.62	0.53	0.52	0.63**	0.40	0.07
Kaituna C.	0.32	0.60	0.60	0.89	0.77	0.36	0.17	0.88**	0.29	0.92***	0.45	0.30	0.83	0.58	0.28
FORLD	0.89	0.84	0.32	0.77	0.26	0.21	0.83**	0.21	0.65	0.39	0.06	0.56	0.14	0.50	0.14
RMA	0.00	0.77	0.45	0.13	0.89	0.41	0.13	0.09	0.26	0.29	0.09	0.07	0.33	0.24	0.12
Sewage	0.00	0.77	0.45	0.13	0.89	0.41	0.13	0.09	0.26	0.29	0.09	0.07	0.33	0.24	0.12
Fonterra	0.00	0.67	0.83	0.13	0.77	0.10	0.82*	0.90***	0.36	0.29	0.09	0.58	0.41	0.48	0.36
EBOP chair	0.26	0.67	0.67	0.63	0.77	0.68	0.83**	0.46	0.26	0.59	0.36	0.29	0.13	0.35	0.41
Cyano blooms	0.45	0.77	0.92	0.63	0.67	0.62	0.71	0.59	0.79	0.71	0.64**	0.04	0.10	0.11	0.36
Lake Settlement	-	0.45	0.45	-	0.18	0.65	0.08	0.62	0.65	0.53	-	-	-	-	-
Rule 11	0.77	0.23	0.77	0.77	0.64	0.91***	0.39	0.13	0.70	0.37	-				

P value indicated by asterisks (*=p=0.05; **=p<0.5; ***=p=0.01); significant correlation coefficients (>0.7) indicated by green shading

Appendix IV Specifications of CAEDYM model parameter adjustments, showing adjusted values used for the configuration of the model's sediment processes. Sediment N release rate, P release rate, Particulate Organic Matter (POM) diameter and sediment oxygen demand were adjusted to reflect the nutrient loading and presence of alum dosing for the individual scenarios associated with each nutrient load option.

Option	**Alum dosing**	**Total N load ($t\ yr^{-1}$)**	**Sediment N release rate ($g\ m^{-2}\ d^{-1}$)**	**Total P load ($t\ yr^{-1}$)**	**PO4 load ($t\ yr^{-1}$)**	**Sediment P release rate ($g\ m^{-2}\ d^{-1}$)**	**Sediment oxygen demand ($g\ m^{-2}\ d^{-1}$)**	**POM diameter (mm)**
1	Yes	435	0.33	37	24.365	0.015	0.67	0.018
2	None	435	0.33	37	25.465	0.030	0.67	0.09
3	Yes	750	0.50	47	29.33	0.040	2.90	0.018
4	None	750	0.50	47	28.13	0.020	2.90	0.09
5	None	226	0.15	25	21.6	0.021	0.09	0.09
6	None	435	0.33	30	24	0.030	0.50	0.09

Appendix V Detailed option analysis including catchment values and mitigation costs. Refer to Figure 9 for details of each option. ES refers to Ecosystem Services. Values are given in $NZ million.

	Land use type	**Area (ha)**	**TN (t/yr)**	**TP (t/yr)**	**Ecosystem Services value ($M/yr)**	**Land value ($M/yr)**	**Land mitigation ($M/yr)**	**Lake mitigation ($M/yr)**
Option 1a **Land use change + alum**	Dairy	220.9	11.8	0.5	0.3	0.9	n/a	n/a
	Dairy support	383.7	10.1	0.5	0.5	1.6	n/a	n/a
	Drystock	10991.2	163.9	15.0	13.1	3.2	n/a	n/a
	Forestry/exotic	11171.7	65.0	2.7	18.9	13.7	n/a	n/a
	Native cover	17404.6	68.7	2.5	29.4	21.3	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.6
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**439.7**	**32.6**	**180.8**	**40.6**	**n/a**	**0.6**
	Land use type	**Area (ha)**	**TN (t/yr)**	**TP (t/yr)**	**Ecosystem Services value ($M/yr)**	**Land value ($M/yr)**	**Land mitigation ($M/yr)**	**Lake mitigation ($M/yr)**
Option 1b **Land mitigation + alum**	Dairy	5014.0	102.0	6.0	6.0	20.8	1.9	n/a
	Dairy support	2514.0	30.0	2.0	3.0	10.5	0.4	n/a
	Drystock	13654.0	90.0	14.0	16.2	4.0	1.5	n/a
	Forestry/exotic	9574.0	55.7	2.4	16.2	11.7	n/a	n/a
	Native cover	9416.0	37.2	1.4	15.9	11.5	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.6
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**435.1**	**37.1**	**176.0**	**58.5**	**3.9**	**0.6**

	Land use type	**Area (ha)**	**TN (t/yr)**	**TP (t/yr)**	**Ecosystem Services value ($M/yr)**	**Land value ($M/yr)**	**Land mitigation ($M/yr)**	**Lake mitigation ($M/yr)**
Option 1c	Dairy	3416.3	102.0	7.0	4.1	14.2	0.6	n/a
Land use change,land mitigation + alum	Dairy support	916.3	20.0	1.1	1.1	3.8	0.0	n/a
	Drystock	9926.0	100.0	13.5	11.8	2.9	0.4	n/a
	Forestry/exotic	12236.9	55.7	2.4	20.7	15.0	n/a	n/a
	Native cover	13676.6	37.2	1.4	23.1	16.7	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.6
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**435.1**	**36.7**	**179.4**	**52.6**	**1.1**	**0.6**

	Land use type	**Area (ha)**	**TN (t/yr)**	**TP (t/yr)**	**Ecosystem Services value ($M/yr)**	**Land value ($M/yr)**	**Land mitigation ($M/yr)**	**Lake mitigation ($M/yr)**
Option 2a	Dairy	220.9	11.8	0.5	0.3	0.9	n/a	n/a
Land use change	Dairy support	383.7	10.1	0.5	0.5	1.6	n/a	n/a
	Drystock	10991.2	163.9	15.0	13.1	3.2	n/a	n/a
	Forestry/exotic	11171.7	65.0	2.7	18.9	13.7	n/a	n/a
	Native cover	17404.6	68.7	2.5	29.4	21.3	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.0
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**439.7**	**32.6**	**180.8**	**40.6**	**n/a**	**0.0**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 2b	Dairy	5014.0	102.0	6.0	6.0	20.8	1.9	n/a
Land mitigation	Dairy support	2514.0	30.0	2.0	3.0	10.5	0.4	n/a
	Drystock	13654.0	90.0	14.0	16.2	4.0	1.5	n/a
	Forestry/exotic	9574.0	55.7	2.4	16.2	11.7	n/a	n/a
	Native cover	9416.0	37.2	1.4	15.9	11.5	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.0
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**435.1**	**37.1**	**176.0**	**58.5**	**3.9**	**0.0**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 2c	Dairy	3416.3	102.0	7.0	4.1	14.2	0.6	n/a
Land use change and land mitigation	Dairy support	916.3	20.0	1.1	1.1	3.8	0.0	n/a
	Drystock	9926.0	100.0	13.5	11.8	2.9	0.4	n/a
	Forestry/exotic	12236.9	55.7	2.4	20.7	15.0	n/a	n/a
	Native cover	13676.6	37.2	1.4	23.1	16.7	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.6
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**435.1**	**36.7**	**179.4**	**52.6**	**1.1**	**0.6**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 3	Dairy	5014.0	267.2	10.3	6.0	20.8	n/a	n/a
Current land use + alum	Dairy support	2514.0	66.1	3.0	3.0	10.5	n/a	n/a
	Drystock	13654.0	203.5	18.6	16.2	4.0	n/a	n/a
	Forestry/exotic	9574.0	55.7	2.4	16.2	11.7	n/a	n/a
	Native cover	9416.0	37.2	1.4	15.9	11.5	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	0.6
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**750.0**	**47.0**	**176.0**	**58.5**	**n/a**	**0.6**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
	Dairy	5014.0	267.2	10.3	6.0	20.8	n/a	n/a
Option 4	Dairy support	2514.0	66.1	3.0	3.0	10.5	n/a	n/a
Current land use	Drystock	13654.0	203.5	18.6	16.2	4.0	n/a	n/a
	Forestry/exotic	9574.0	55.7	2.4	16.2	11.7	n/a	n/a
	Native cover	9416.0	37.2	1.4	15.9	11.5	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	n/a
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**750.0**	**47.0**	**176.0**	**58.5**	**n/a**	**n/a**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 5	Dairy	0.0	0.0	0.0	0.0	0.0	n/a	n/a
All native	Dairy support	0.0	0.0	0.0	0.0	0.0	n/a	n/a
	Drystock	0.0	0.0	0.0	0.0	0.0	n/a	n/a
	Forestry/exotic	0.0	0.0	0.0	0.0	0.0	n/a	n/a
	Native cover	40171.9	158.5	5.8	67.8	49.1	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	n/a
	Geothermal	n/a	36.1	10.4	n/a	n/a	n/a	n/a
	Wastewater	n/a			n/a	n/a	n/a	n/a
	Total	**53257.0**	**246.1**	**23.0**	**186.6**	**49.1**	**n/a**	**n/a**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 6a	Dairy	220.9	11.8	0.5	0.3	0.9	n/a	n/a
Land use change	Dairy support	916.3	24.1	1.1	1.1	3.8	n/a	n/a
	Drystock	8860.9	132.1	12.1	10.5	2.6	n/a	n/a
	Forestry/exotic	11704.3	68.1	2.9	19.8	14.3	n/a	n/a
	Native cover	19534.8	77.1	2.8	33.0	23.9	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	n/a
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**433.4**	**30.7**	**183.4**	**45.5**	**n/a**	**n/a**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 6b	Dairy	5014.0	102.0	3.5	6.0	20.8	2.2	n/a
Land mitigation	Dairy support	2514.0	30.0	1.5	3.0	10.5	0.5	n/a
	Drystock	13654.0	90.0	10.0	16.2	4.0	2.1	n/a
	Forestry/exotic	9574.0	55.7	2.4	16.2	11.7	n/a	n/a
	Native cover	9416.0	37.2	1.4	15.9	11.5	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	n/a
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**435.1**	**30.1**	**176.0**	**58.5**	**4.8**	**n/a**

	Land use type	Area (ha)	TN (t/yr)	TP (t/yr)	Ecosystem Services value ($M/yr)	Land value ($M/yr)	Land mitigation ($M/yr)	Lake mitigation ($M/yr)
Option 6c	Dairy	2351.2	102.0	4.0	2.8	9.8	0.3	n/a
Land use change and land mitigation	Dairy support	1448.9	20.0	1.0	1.7	6.0	0.2	n/a
	Drystock	12588.9	100.0	10.0	15.0	3.7	1.7	n/a
	Forestry/exotic	12236.9	55.7	2.4	20.7	15.0	n/a	n/a
	Native cover	11546.3	37.2	1.4	19.5	14.1	n/a	n/a
	Urban	4677.0	16.2	4.0	n/a	n/a	n/a	n/a
	Lake/pond	8408.0	35.3	2.9	118.8	n/a	n/a	n/a
	Geothermal	n/a	36.1	2.5	n/a	n/a	n/a	n/a
	Wastewater	n/a	32.7	2.1	n/a	n/a	n/a	n/a
	Total	**53257.0**	**435.1**	**30.1**	**178.4**	**48.5**	**2.2**	**n/a**