Septic-derived Nutrient Contamination of Shallow Groundwater in Lake Tarawera:

Extent, Fate, and Ecological Consequences

A thesis

submitted in partial fulfilment

of the requirements for the degree

of

Master of Science (Research) in Environmental Sciences

at

The University of Waikato

by

Sarah Tetzlaff

THE UNIVERSITY OF
WAIKATO
Te Whare Wānanga o Waikato

2026

Abstract

Onsite wastewater treatment systems (OWTS) are important point sources of nitrogen and phosphorus pollution to lakes, with nutrients transported via groundwater to the nearshore zone where they can stimulate algal growth and degrade littoral ecosystems. At Lake Tarawera (Bay of Plenty, New Zealand), OWTS are estimated to contribute 3–5 % to the annual nutrient load, prompting implementation of a sewage reticulation scheme of the residential area along the lake's western margin. However, the spatial distribution, subsurface transport, and ecological consequences of septic-derived nutrients remain poorly quantified. This study assessed shallow groundwater contamination within the urbanised western margin, examined groundwater–littoral transport pathways, and evaluated potential effects on benthic primary producers.

Groundwater was sampled monthly from 21 piezometers along upslope and downslope transects between April 2023 and March 2025 and analysed for nitrate-N, nitrite-N, ammoniacal-N, and dissolved reactive phosphorus (DRP). Groundwater connectivity to the lake was assessed by relating antecedent rainfall to groundwater flux into the littoral zone, measured using 24-hour benthic chamber deployments. Stable isotope ratios of nitrate ($\delta^{15}N$ and $\delta^{18}O$) in groundwater and pore water, alongside $\delta^{15}N$ and $\delta^{13}C$ in macrophyte, benthic algal, and suspended particulate organic matter (SPOM) tissue, were used to trace septic-derived nitrogen transport. Benthic gross primary production (GPP) was estimated seasonally at six nearshore sites using 1.5-hour benthic chamber incubations to assess potential changes in littoral metabolism.

Results revealed localised areas of elevated nutrient concentrations within the urban zone, with high maximum values observed for ammoniacal-N (61 mg L^{-1}), nitrate-N (7.4 mg L^{-1}), DRP (8.7 mg L^{-1}), and nitrite-N (0.91 mg L^{-1}), consistent with previously reported septic contamination ranges and higher than those measured in a prior study at the lake. No consistent seasonal or annual patterns were observed. Enriched $\delta^{15}N$ values were observed in both upslope groundwater (5.50 ± 3.18 ‰) and littoral pore water (6.51 ± 3.43 ‰), indicating wastewater-derived nitrogen inputs to the adjacent littoral zone. However, low $\delta^{15}N$ values in SPOM, macrophytes, and benthic algae (1.48 ± 0.83

‰), suggested limited assimilation by primary producers. Sites exhibiting dense epiphytic algal growths during spring and summer displayed the highest benthic GPP rates (maximum = 296 mg O_2 m^2 h^{-1}). Although vegetation biomass explained the most variation in GPP (19 %), multiple lines of evidence suggest that upslope groundwater nutrient enrichment may have promoted short-term epiphytic algal growth at certain sites under favourable growing conditions.

This study provides integrated evidence linking septic-derived nutrient transport through groundwater to altered benthic community structure in Lake Tarawera. Although measurable impacts on primary producer assimilation were limited, substantial nutrient loading to shallow groundwater and the nearshore zone was evident. Periphyton responses appear to be a sensitive indicator of early nutrient pressure preceding broader metabolic changes. Continued inputs may increase the risk of shifts towards greater algal dominance, underscoring the need to address OWTS sources proactively. Lake Tarawera serves as a case study of a widespread and often underestimated issue of nutrient contamination from OWTS across New Zealand, emphasising the importance of strengthened monitoring and management to help combat further freshwater degradation.

Acknowledgements

This thesis was supported by an endless list of people, far more than can be captured within these two pages. Firstly, I would like to extend my sincere gratitude to my supervisors, Deniz Özkundakci and Grant Tempero. The yin and yang of mentorship, I have greatly benefited from your mix of enthusiasm, realism, bluntness, kindness, and excitement for freshwater science. You have shaped me into a well-rounded, robust researcher who has finally learned to write in shorter sentences. There are few supervisors who would take on a distance student who is also a full-time athlete, and the logistical complexities that come with that; thank you for your trust in me to do a good job while remaining understanding of my other commitments in life, and allowing me to extend my deadline. This has been the best foundational experience of my science career that I could have hoped for. Thank you so much.

To Simon Stewart, thank you for your critical advice and assistance on the stable isotopes component of this research. It has been a privilege to receive help from one of the best in the business. To Mary de Winton and Deborah Hofstra at Earth Sciences NZ, thank you for supplying me with monitoring equipment and providing thoughtful advice on macrophyte monitoring. I would like to especially thank Kim Pritchard for your meticulous proofreading, advice, and encouragement in the final moments.

This research was primarily supported by a joint grant funded by Bay of Plenty Regional Council and the Tarawera Rate Payers' Association. Thank you to the homeowners for allowing us to work on your properties, I hope I have generated valuable knowledge that can help protect this beautiful lake. Additional funding from the Tess Embling Memorial Scholarship and the Hillary Jolly Memorial Scholarship further supported this work; I am privileged to have followed in the footsteps of these incredible women and hope that I have carried on their legacy well. Prime Minister Scholarships from High Performance Sport NZ also supported my dual career, and the Sparling Travel Award for Environmental Studies funded my travel to the New Zealand/Australia Joint Freshwater Sciences Conference in 2025, which proved to be a pivotal experience in my career.

This work was not possible without help from lab and field technicians. Ari Brandenburg helped with equipment induction, the creation of laboratory procedures, and answered every one of my questions and emails thoughtfully and promptly. Holly Ferguson and Bonnie Lewis conducted dive surveys, assisted on all field sampling trips, and brought great banter. Kat Rowe and Joe Butterworth additionally helped with dive surveys, field sampling, and morale. Annette Rodgers provided advice and assistance with particle size analysis, and Chloe Kayll-Irvine provided lab assistance in Hamilton. To all my fieldwork helpers, you made this happen, especially those who came swimming in the middle of winter. Thank you all, I greatly appreciate your reliability and readiness to help.

To my Coastal Marine Field Station peers, thank you for accepting me into your circle, despite my area of research lacking salt water. I would especially like to thank my PhD peers Ashton Budd, Maike Kaffenberger, and Fiona Chabbey for all their help with data analysis, life advice, candour, and banter. I believe every person needs blunt European friends in their lives. To my Lakes Research Group peers in Hamilton, thank you for making me feel at home whenever I was on campus. It has been more inspiring than you could know to meet and discuss ideas with so many passionate freshwater scientists. To Whitney Woelmer and Rose Gregersen especially, thank you for your encouragement and life advice, and showing me what a woman in science can look like. When I grow up, I want to be just like you.

To my family. Dad, your demonstration of hard work in your own life and our conversations about the bigger picture have unwittingly shaped me into someone who strives to be better and dream big. Mum, thank you for sparking my love for nature with our bush walks and beach trips growing up, for inspiring me with your own early science career, and for the chats about life. You have both grounded me throughout this process and supported me no matter what. Will, I am so proud to have you as my sibling and find constant inspiration in what you have achieved. I am incredibly lucky to be loved by you all, and am so excited to share this achievement with you.

Finally, to JJ. You are my rock. Thank you for your sacrifices, your unwavering love, and for always showing me what really matters in life.

Table of Contents

List of Figures

List of Tables

Chapter 1

General introduction

Wastewater degradation of freshwater ecosystems

Despite unprecedented technological and scientific advances of the last century, the millennia-old issue of adequate human waste disposal remains unresolved and continues to degrade freshwater systems globally. Each day, an estimated two million tons of wastewater from sewage, industrial, and agricultural practices is discharged into aquatic environments worldwide, and approximately 44 % of domestic wastewater is inadequately treated before disposal (du Plessis, 2022). Freshwater is a finite and vital resource that supports all life, food production, and economic development that occurs on the planet, of which only 0.007 % of the Earth's total water supply is accessible for consumption (Afolalu et al., 2022). Yet pressures on freshwater quality are intensifying, with urban expansion, increased agricultural and industrial activity, and increased climate variability expected to exacerbate wastewater-driven degradation (Afolalu et al., 2022).

Some of the earliest documented cases of anthropogenic water quality degradation occurred within lakes, where wastewater discharge and agricultural runoff caused excessive algae proliferation and hypoxia in bottom waters (Schindler & Vallentyne, 2008). Among freshwater environments, lakes are particularly susceptible to nutrient enrichment because internal nutrient recycling can amplify and sustain eutrophication (Søndergaard et al., 2003). Increased nutrient concentrations can stimulate phytoplankton growth, which increases organic matter deposition to bottom sediments. As this material decomposes, bottom-waters can become anoxic, promoting the release of sediment-bound phosphorus and making it available for further phytoplankton uptake. This internal loading reinforces eutrophication and can maintain poor water quality even if external nutrient inputs are reduced (Søndergaard et al., 2003).

Once these feedback loops have begun, reversal becomes difficult without significant management intervention. Lakes may pass a tipping point into an alternative stable state characterised by phytoplankton dominance and macrophyte loss, after which restoring prior conditions becomes considerably challenging (Hessen et al., 2024). Given the importance of lakes for ecosystem service provision (including drinking water, habitat maintenance, recreation, and cultural values), preventing excessive nutrient inputs before such thresholds are crossed is critical (Reynaud & Lanzanova, 2017).

Onsite wastewater treatment systems

Onsite wastewater treatment systems (OWTS) are widely used in rural, low-density, and poor communities, where connection to centralised sewage reticulation is not feasible (Lusk et al., 2017). Onsite wastewater treatment systems are well recognised and often underestimated point sources of nutrient and contaminant pollution to lake systems (Withers et al., 2014). Globally, more than 500 million OWTS service approximately 2.8 billion people, including 26 % of households in Europe, 25 % in the United States, and 20 % in Australia, making pollution from OWTS a worldwide issue (Rakhimbekova et al., 2021; Gyimah et al., 2024).

A conventional OWTS typically consists of a septic tank and a subsurface soil absorption system, known as a drainfield or leach field (Lusk et al., 2017). Wastewater first undergoes primary treatment within the septic tank, where solids settle to the bottom to form sludge; lighter materials accumulate as floating scum on the surface; and clarified effluent remains in the liquid phase. This effluent then discharges to the drainfield, where it infiltrates natural or artificial soil and undergoes secondary treatment. As effluent passes through the soil, contaminant concentrations, including phosphorus and nitrogen, are reduced via absorption or adsorption to soil particles, and microbial transformations (Lusk et al., 2017). Advanced systems may incorporate an aerobic chamber that further enhances microbial removal (Ministry for the Environment, 2020). The leftover effluent then leaches through the drainfield and into the surrounding sediment, eventually making its way into groundwater (Lusk et al., 2017).

Despite these treatment stages, OWTS do not completely remove all contaminants from effluent, simply due to their design (Gyimah et al., 2024). This can be further exacerbated by system performance; OWTS are prone to failure or poor function due to inadequate location, soil composition, maintenance, technology, local climate conditions, and depth to the water table, which are often suboptimal in rural locations (Lusk et al., 2017). Improperly treated effluent can then percolate into the soil before entering groundwater systems, where it is transported downslope to ultimately discharge into lakes, and other surface water bodies.

Onsite wastewater treatment in New Zealand

In New Zealand, approximately 21% of the population relies on OWTS for domestic wastewater treatment (Ministry for the Environment, 2020). System failure and underperformance are frequently documented in New Zealand, with common causes including lack of regular servicing and management, inappropriate siting in inadequate soil conditions, and aging infrastructure (Ministry for the Environment, 2020). Ray (2007) estimated that 42,000 houses nationwide were experiencing some degree of septic tank failure, impacting more than 100 streams and an unknown number of groundwater systems and lakes. However, national-scale monitoring of OWTS is limited, and the true extent of septic-derived contamination is not known.

Groundwater as a septic contaminant transport pathway

Groundwater can be a dominant pathway for delivering nutrients to lakes (Robinson, 2015). In many lakeshore developments, septic systems are in close proximity to the shoreline, allowing effluent to enter shallow groundwater that flows directly towards the lake (Timoshkin et al., 2018). Subsurface discharge typically occurs within the first 50 m offshore of the lake, meaning any contaminants will be discharged into the biologically active littoral zone (John & Lock, 1977). Hydrogeological conditions strongly influence the delivery rates of septic pollutants to the littoral zone; high water table conditions, permeable soils, and fractures or preferential flow paths can accelerate contaminant transport (Scholes, 2007; Lewandowski et al., 2015). These processes result in heterogeneous groundwater discharge, and subsequently non-uniform contaminant concentrations, along the littoral zone (Naranjo et al., 2019).

Nitrogen species are of particular concern in groundwater, as they are highly mobile and are poorly attenuated in soil (Robinson, 2015). Phosphorus is comparably highly attenuated due to adsorption to soil particles and mineralisation (Robinson, 2015). However, where drainfields become saturated, long-term migration of phosphorus plumes toward lakeshores can occur, with legacy effects persisting for decades even after OWTS removal (Wang et al., 2024). Despite increased recognition of groundwater as a vector for septic-derived contaminants, the magnitude, spatial variability, and ecological consequences in littoral zones remains poorly quantified (Rakhimbekova et al., 2021).

Impact on littoral ecosystems

The littoral zone is a biologically and morphologically complex ecosystem and is a sensitive first responder to nutrient inputs delivered by groundwater (Schindler & Scheuerell, 2002). This habitat supports diverse communities, including epipelic and epilithic algae, macrophytes, sediment-dwelling microbes, invertebrates, and fish. As the interface of both pelagic and terrestrial habitats, the littoral zone plays a critical role in transferring resources and energy across lake food webs (Schindler & Scheuerell, 2002; Vander Zanden et al., 2006). Benthic primary producers can contribute substantially to whole-lake metabolism, and may even exceed pelagic production in shallow or clear-water systems (Vadeboncoeur et al., 2002; 2003).

The structure and balance of the littoral ecosystem are governed by competition for light and nutrients between pelagic and benthic primary producers (Vadeboncoeur et al., 2003). Elevated water-column nutrients can stimulate phytoplankton growth, increase turbidity and shade benthic producers, thereby shifting systems towards pelagic dominance (Vadeboncoeur & Power, 2017). Conversely, nutrient delivery directly to the nearshore zone via groundwater discharge can stimulate periphyton growth before nutrients disperse into the water column (Vadeboncoeur et al., 2003). In recent decades, excessive periphyton and phytoplankton growth has been increasingly observed in otherwise clear-water lakes globally, a phenomenon partly attributed to increasing groundwater nutrient delivery from anthropogenic land uses (Vadeboncoeur et al., 2021).

Methods for tracing sewage contamination

Septic tank contamination can be detected along the entire transport pathway, from the vicinity of the OWTS source to the primary producers that assimilate septic-derived nutrients. A combination of approaches is often required to confidently attribute contamination to septic systems.

Shallow groundwater and pore water

Sampling approaches

Groundwater sampling is one of the most common and cost-effective methods for detecting septic tank contamination. Piezometers or shallow wells are installed downslope of potential contamination sources to intercept shallow groundwater flow paths (Nyenje et al., 2013). These perforated pipes allow groundwater to enter and be extracted for chemical analysis (Wassenaar & Hendry, 1999). Bundles of piezometers extending to varying depths can be used to assess vertical contamination profiles, while transects placed downslope of septic systems can characterise the extent of the horizontal contaminant plume (Harman et al., 1996; Wang et al., 2024).

In littoral environments, sediment pore water beneath the sediment-water interface can also be sampled to assess nutrient delivery from discharging groundwater (Naranjo et al., 2019). Pore water can be extracted using manual suction with a syringe-style device (Hagerthey & Kerfoot, 1998), in-lake shallow piezometers (Naranjo et al., 2019), or centrifugation of lakebed sediment cores (Chen et al., 2025). These measurements provide insights into the nutrient concentrations at the point of discharge into the littoral zone (Lisboa et al., 2024).

Chemical indicators of septic contamination

Groundwater and pore water samples are typically analysed for contaminants that are exclusively products of anthropogenic practices, and which persist in the environment. These typically include nutrients (nitrogen and phosphorus), pathogens (*Escherichia coli* and enterococci), artificial sweeteners (sucralose), major ions (chloride, sulphate), pharmaceuticals and heavy metals (Scholes, 2007; Lusk et al., 2017; Baer et al., 2019; Chen et al., 2025).

Nitrogen species, particularly nitrate-N and ammoniacal-N, are commonly used as septic indicators as they are produced in high quantities in septic leachate that are not normally produced from other sources, and are highly mobile in groundwater, as well as being cost-effective to measure (Lusk et al., 2017). Incorporating multiple septic indicators generally strengthens the attribution of what is in the groundwater directly to a septic source.

Stable isotopes

Stable isotopes of nitrate in groundwater provide an additional tool for tracing septic-derived contamination, as these isotopes do not decay and are conserved throughout the environment (Kendall et al., 2007). Nitrogen and oxygen in nitrate molecules occur as both light and heavy isotopes (e.g. ^{14}N and ^{15}N, ^{16}O and ^{18}O), and the ratio of heavy to light isotopes is expressed in delta (δ) notation relative to an international standard in parts per thousand (‰). A positive delta indicates that the sample is enriched in the heavier isotope relative to a standard. Human and animal waste typically exhibit elevated $\delta^{15}N$ (>4 ‰) due to trophic enrichment and isotopic fractionation during waste processing, where lighter nitrogen isotopes are lost through volatilisation of ammonia after deposition. Therefore, enriched $\delta^{15}N$ values in groundwater may indicate that nitrate in the sample is wastewater-derived (Kendall et al., 2007).

However, source attribution is complicated by overlapping isotopic ranges among nitrate sources (e.g. natural soil nitrogen, atmospheric deposition, sewage, animal waste, fertiliser) and by subsurface microbial processes that further modify isotopic composition via preferential uptake of lighter isotopes (Aravena & Robertson, 1998;

Kendall et al., 2007). For this reason, stable isotopes are best applied with other septic indicators for source tracing.

Isotopic enrichment may also be detected in littoral primary producers. Where septic-contaminated groundwater inputs to the littoral zone are substantial and represent a significant nitrogen source, $\delta^{15}N$ values in producer tissue can become elevated, reflecting integration of septic-derived nutrients into the food web (Steffy & Kilham, 2004; Kohzu et al., 2008; Li et al., 2011). However, as with groundwater signatures, primary producer isotopic signals can be altered due to preferential uptake of lighter nitrogen isotopes, resulting in fractionation that can obscure original sources (Bacchus & Barile, 2005). These signatures are therefore best interpreted alongside additional chemical and ecological indicators.

Benthic primary production as an ecological indicator

Ecological responses provide a complimentary and integrative approach to assessing septic impacts. Benthic gross primary production (GPP) can be measured using in situ benthic chambers, where oxygen or inorganic carbon fluxes are monitored in light and dark conditions to estimate photosynthesis and respiration of the benthic community (Baulch et al., 2005; Davies & Hecky, 2005; Godwin et al., 2014). Because the benthic littoral community is sensitive to both direct groundwater nutrient inputs and indirect effects of increased water column inputs through phytoplankton shading, GPP serves as a useful integrator of ecosystem structure and function in response to environmental conditions (Vadeboncoeur et al., 2003; Davies & Hecky, 2005). Shifts in benthic production and community composition may therefore reflect changes in groundwater nutrient delivery, or other external factors.

Lake Tarawera

Lake Tarawera is a large (41.7 km^2), deep (average depth = 57 m), oligotrophic lake located in the Te Arawa lakes region, Bay of Plenty, New Zealand (Cochrane, 2020). The lake is monomictic and typically stratifies from August to May (Cochrane, 2020). Situated within the Ōkataina Volcanic Centre of the Taupō Volcanic Zone, the present-day lake was formed by the 1886 eruption of Mount Tarawera and remains geothermally active along its southern and eastern margins (Caratori Tontini et al., 2023). Catchment soils are classified as well-drained pumice or recent soils (Wilson, 2022).

Due to its volcanic setting, the Lake Tarawera catchment system is complex. Tarawera receives inflows from precipitation recharged within the immediate catchment, as well as surface water inflows and groundwater seepage from seven upstream Te Arawa lakes (Hamilton et al., 2006; Wilson, 2022; Figure 1.1). Groundwater is a major component of the lake's water budget, comprising approximately 62 % of total inflows, with 30 % of this recharged within the immediate catchment (Wilson, 2022). Geothermal springs located along the southern, southeastern, and northern shorelines contribute nutrient-rich groundwater to the lake (Caratori Tontini et al., 2023).

Currently, the Trophic Level Index (TLI) of Lake Tarawera is 2.9 and has remained relatively stable since 2011 (Land, Air, Water Aotearoa [LAWA], n.d.-a); however, this remains above its management target of 2.6 (Hamilton et al., 2006). There are a wide range of nutrient sources to Lake Tarawera. Within the greater catchment, inputs via groundwater seepage and surface water flow originate from lakes spanning a broad range of trophic conditions (TLI 2.9 – 4.5) with variable catchment land uses (LAWA, n.d.-b; Wilson, 2022; Figure 1.1). These connected lakes contribute an estimated 21.7 % and 23.4 % of total nitrogen (TN) and total phosphorus (TP) loads to Tarawera (Wilson, 2022). Within the inner catchment, land cover comprises 62.5 % native forest, 15.7 % exotic forest, 18.0 % pasture, and 1.0 % urban, contributing nutrients directly via groundwater and overland flow (Hamilton et al., 2006). Anthropogenic land uses within the inner catchment account for an estimated 36.8 % of TN and 35.5 % of TP loads to the lake (Wilson, 2022). Because TN and TP are both strongly retained within Lake Tarawera, achieving the TLI target requires reduction of external nutrient sources rather than in-lake remediation strategies (Hamilton et al., 2006).

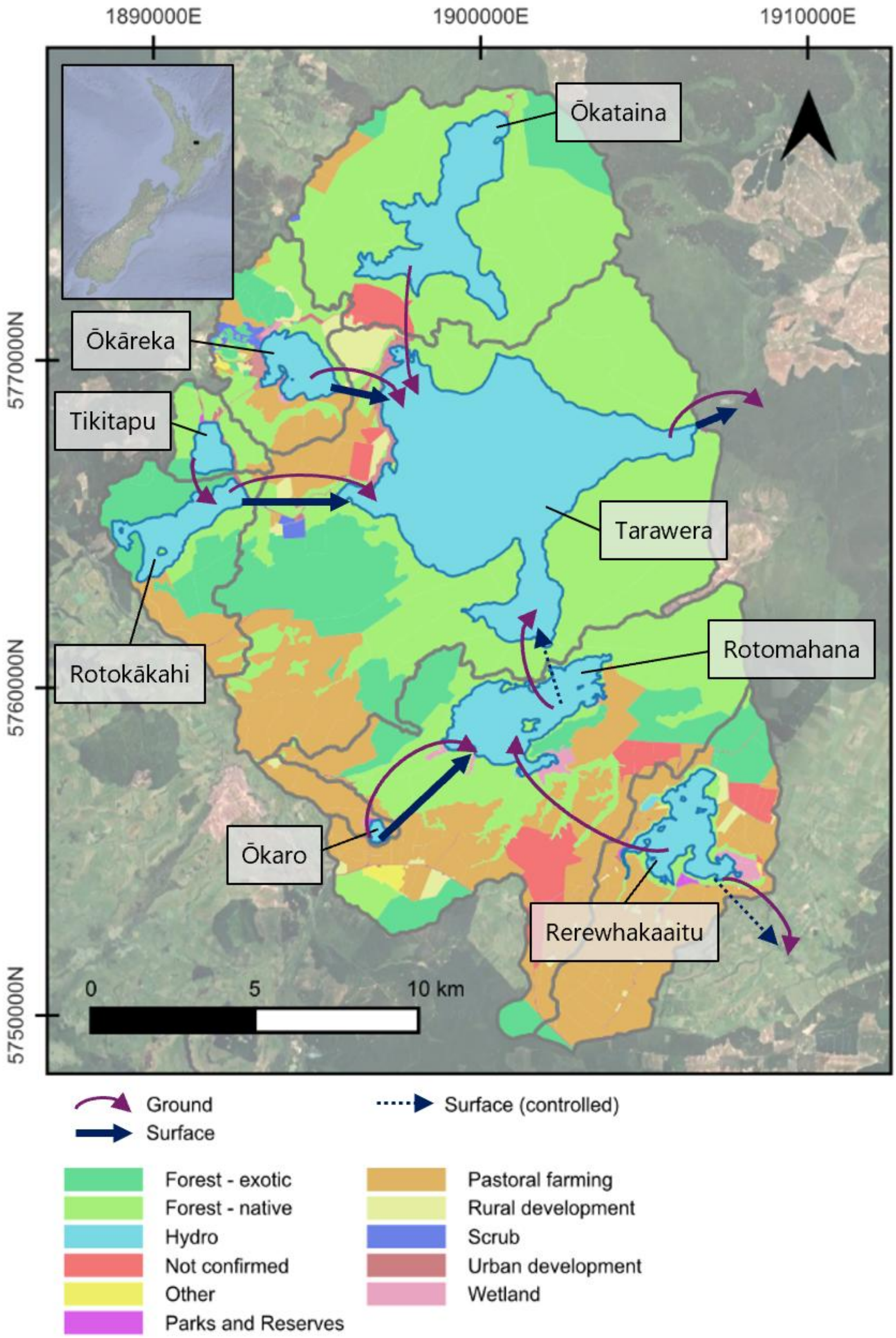

Figure 1.1. The Greater Tarawera Catchment System showing hydrological connectivity (arrows) and catchment land use (coloured areas) of the broader lakes system. Adapted from Prentice et al. (in preparation).

The lake holds significant cultural and social value. Tarawera is a taonga tuku iho (ancestral treasure) of the Te Arawa iwi, who have long used Tarawera and nearby lakes for food provisioning, shelter, and transport, with Mount Tarawera used historically as a burial ground for persons of importance (Bay of Plenty Regional Council, 2015; Kusabs, 2015; Rotorua Lakes Council, 2025a). Today, Tarawera remains a popular recreational lake, particularly within the littoral zone, supporting activities such as fishing, boating, and swimming, and is the primary water supply for a permanent lakeside population of approximately 291 residents (Bay of Plenty Regional Council, 2015). The local community is strongly invested in preserving the lake's clear-water state and associated values.

Ecologically, Tarawera has an extensive and well-developed littoral zone adjacent to the urban shoreline. In some areas, the shoreline is characterised by steep bathymetry supporting tall-growing macrophytes, while other bays with low slope gradients are dominated by low-growing isoetids forming extensive turfs. Both native and invasive species are present, with the most common native vegetation including *Myriophyllum triphyllum*, *M. propinquum*, *Chara globularis* and *Ruppia polycarpa* (Wells et al., 1997). Native species are under significant pressure from highly invasive species introduced post-European settlement, particularly *Ceratophyllum demersum*, *Egeria densa*, *Elodea canadensis* and *Lagarosiphon major*, which form dense beds extending deep into the water column (Wells et al., 1997). Invasive weeds are culled annually at popular boating sites along the western shoreline using the herbicide diquat (Boffa Miskell Limited, 2020). The current Lake Submerged Plant Indicator (LakeSPI) score is moderate (30.8 %), due to the prevalence of invasive species (LAWA, n.d.-a). The lake also supports abundant kōura (freshwater crayfish, *Paranephrops planifrons*), a taonga species for Te Arawa, alongside kākahi (freshwater mussels, *Echyridella menziesii*), inānga (whitebait), and tuna (eels, *Anguilla* spp.), and a renowned rainbow trout (*Oncorhynchus mykiss*) fishery (Abell et al., 2020; Kusabs, 2020; Boffa Miskell Limited, 2020).

Onsite wastewater treatment systems have been identified as 15 % of the manageable nutrient load to Tarawera (Dada et al., 2016). Domestic sewage is estimated to contribute 3.3 % of TN and 2.7 % of TP to the lake's nutrient budget, and reticulation is projected to reduce the total nutrient loading by 3 – 5 % (Dada et al., 2016). This source has been identified as a priority for management, particularly given that a 2001 survey found 65 % of surveyed septic systems to be failing (Scholes, 2007). A reticulation scheme commenced in 2015, and from 2023 existing systems began being removed and replaced with low pressure grinder pumps connecting dwellings to the Rotorua wastewater treatment plant (Rotorua Lakes Council, 2025a). The scheme is scheduled to be fully completed by June 2027 (Rotorua Lakes Council, 2025b).

Rationale and objectives

This thesis provides the "before" component of a Before-After Control-Impact (BACI) study designed to evaluate the effectiveness of the Tarawera Sewage Reticulation Scheme in reducing nutrient loading from septic systems to Lake Tarawera. In New Zealand, sewage reticulation schemes are often implemented without formal, long-term monitoring programs post-implementation to assess ecological outcomes. This research addresses that knowledge gap by establishing an in-depth pre-reticulation baseline.

In addition, this study traces septic-derived nutrients along the entire groundwater–littoral pathway, from shallow groundwater within the urban settlement to primary producers in the nearshore zone. Whereby most studies examine only a single component of this pathway, this research adopts an integrated approach to evaluate both contaminant transport and ecological response.

This study has three primary objectives:

1. Determine the spatial and temporal extent of shallow groundwater nutrient contamination in the western urban area of Lake Tarawera

2. Investigate the shallow groundwater–littoral pathway, including evidence of septic-derived contamination at the sediment–water interface

3. Assess the potential ecological impacts of septic-derived nutrient inputs on littoral primary producers adjacent to the urban settlement.

Thesis outline

This research comprises three field-based studies aligned with the stated objectives. Each research chapter presents its specific methods, results, and recommendations for future work.

Chapter 2 – Shallow groundwater nutrient contamination

This chapter addresses Objective 1 by quantifying shallow groundwater nutrient concentrations within the urban zone using piezometer sampling over a two-year period.

Chapter 3 – Groundwater–littoral zone pathway

This chapter addresses Objective 2 by measuring groundwater inflow rates using seepage chambers and examining relationships with antecedent rainfall. Stable isotope analysis of nitrate in groundwater, littoral pore water, and benthic primary producer tissue is used to detect septic-derived nutrient signals.

Chapter 4 – Benthic primary production

This chapter addresses Objective 3 by measuring benthic gross primary production using in situ chambers deployed seasonally over one year. Drivers of variation in benthic production are examined, and potential links to septic-derived nutrient inputs are evaluated.

Chapter 5 – General discussion and conclusions

This chapter synthesises findings from the three field studies, compares results with previous research, discusses broader implications, advocates for littoral monitoring, identifies priorities for future research, and provides concluding remarks.

Chapter 2

Shallow groundwater contamination from septic systems in the urbanised western margin of Lake Tarawera

Introduction

Groundwater is a major vector for nitrogen (N) and phosphorus (P) transportation to lakes and plays a critical role in the eutrophication and degradation of lake ecosystems worldwide (Rosenberry et al., 2015). Unlike surface runoff, groundwater pathways are difficult to observe and quantify, with discharge rates and nutrient concentrations varying widely across space and time (Lewandowski et al., 2015). Groundwater contamination arises from both diffuse and point sources of anthropogenic nutrient pollution. Diffuse sources are associated with widespread human activities such as agricultural fertiliser use and urban runoff, where nutrients leach through soils into groundwater (Khatri & Tyagi, 2015). In contrast, point sources such as industrial wastewater outlets and septic systems release highly concentrated nutrient loads at discrete locations, creating localised hotspots of groundwater contamination (Khatri & Tyagi, 2015). Point sources often contribute disproportionately to nutrient enrichment of groundwater and are therefore a critical target for mitigating nutrient transport to lakes.

Septic systems are an important point source of nutrient pollution to groundwater in lake catchments (Lewandowski et al., 2015). In rural areas where connection to a wastewater treatment facility is not feasible, septic tanks are used as a way to reduce contaminants in human waste making their way into groundwater (Lusk et al., 2017).

Septic tanks are designed to allow for organic matter to undergo primary and secondary treatment, where organic matter in wastewater undergoes anaerobic digestion both within the septic tank and again in the drainfield below the system (Lusk et al., 2017; Withers et al., 2011). The remaining wastewater then leaches through the unsaturated zone of the sediment (vadose zone), until it reaches the water table (Withers et al., 2011). Once below the water table, nutrients can be transported to the littoral zone of a lake via groundwater flow.

In a correctly working septic system, only significant seepage of nitrate to the saturated zone occurs (Withers et al., 2011). However, septic tanks can often fail or not treat septic effluent effectively. Lack of maintenance, poor site selection, poor design, and old age are often contributors to system failure (Gyimah et al., 2024). Additionally, legacy phosphorus from septic systems can migrate slowly through the sediment due to reversible sorption reactions, creating plumes that extend tens of metres downslope of systems (Robertson et al., 2019; Wang et al., 2024). Localised differences in soil type, climactic conditions, land use and distance to the water table can further impact the magnitude of nutrient loading that can occur from septic tanks to the saturated zone (Robinson, 2015).

At Lake Tarawera, the urbanised western margin of the local lake catchment contains approximately 400 households on septic systems (Hamilton et al., 2006; Bay of Plenty Regional Council, 2015). A report by Dada et al. (2016) estimated that septic tanks contribute approximately 3.3 % of the total nitrogen (TN) load and 2.7 % of the total phosphorus (TP) load to the lake, primarily through groundwater flow. The authors further suggested that wastewater reticulation could reduce external nutrient inputs to the lake by 3 – 5 %, providing the basis for the subsequent development and implementation of the Lake Tarawera sewage reticulation scheme that began in 2023.

However, the true extent of groundwater contamination from septic tanks at Lake Tarawera is unknown. Therefore, this study was undertaken to determine whether shallow groundwater nutrient concentrations are consistent with septic tank leachate along the urbanised western margin of Lake Tarawera prior to the completion of the wastewater reticulation scheme. Nutrient concentrations were measured in groundwater collected from shallow piezometers (<2.8 m deep) installed along the western margin of the lake for a two-year period. This data will provide a baseline for assessing the effectiveness of the wastewater reticulation scheme and help validate previous estimates of nutrient loading from septic tanks to the lake.

Methods

Study design

A Before-After Control-Impact (BACI) study design was initially conceived to determine the effect of sewage reticulation on shallow groundwater nutrient concentrations along the Spencer Road settlement at Lake Tarawera. Paired upslope and downslope piezometer sites were installed along Spencer Road, with upslope sites situated above potential septic tank contamination. Downslope sites were situated lakeside of dwellings, approximately along the same groundwater flow path as the paired above site, with horizontal distances between paired sites ranging from 10 m to 520 m (see Figure 2.1). Site selection was subject to landowner permission, influencing the spatial configuration of upslope and downslope piezometer locations.

Two paired piezometer control sites located in native vegetation sub-catchment at the southern and northern ends of Spencer Road were also installed. An artesian spring in the vicinity of the southern control site was opportunistically included in the sampling as a comparative measure. A third suitable control site was unable to be located.

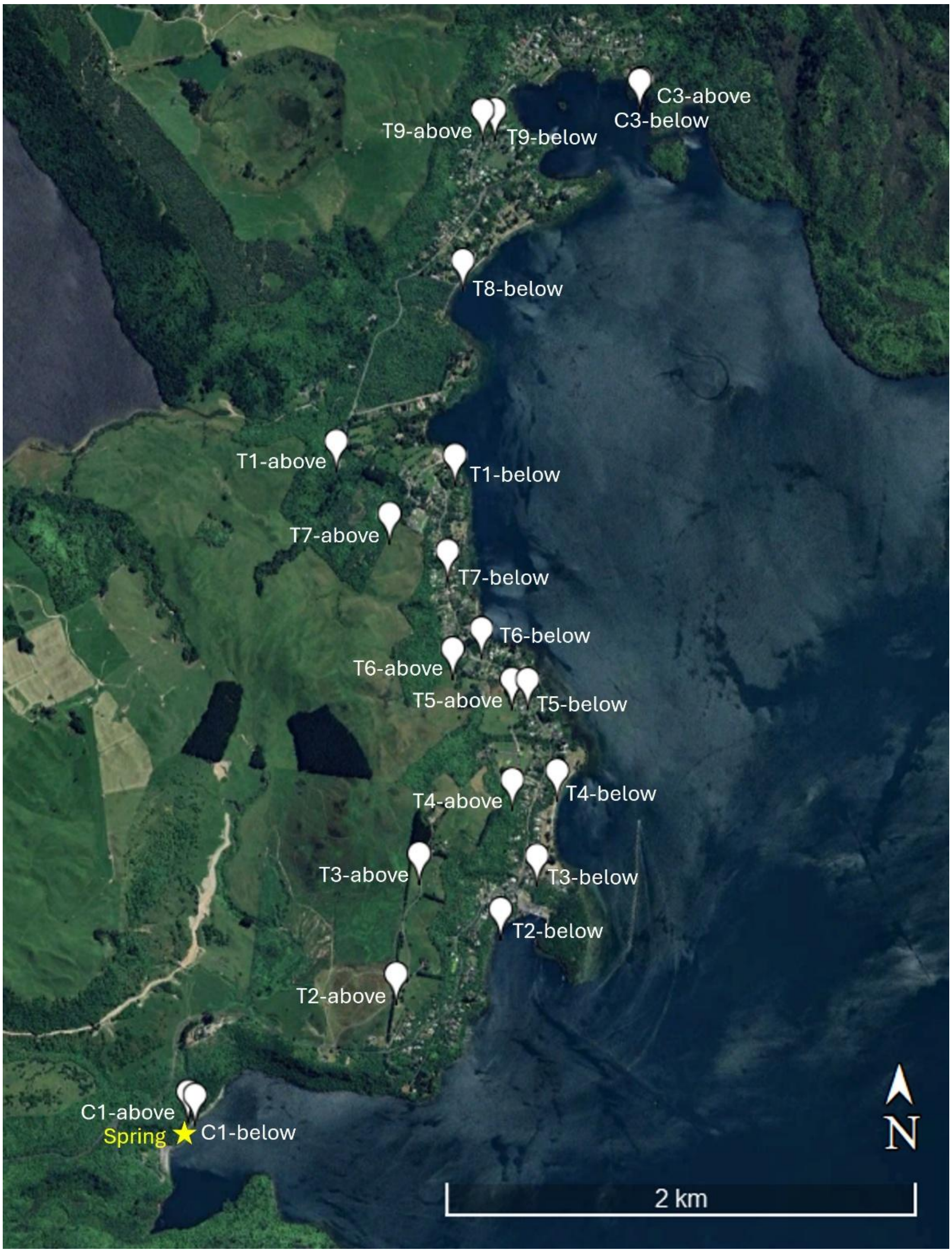

Figure 2.1. Piezometer site locations on the western shoreline of Lake Tarawera, indicated by white placeholders. The yellow star shows the location of the artesian spring.

Piezometer installation

A total of 21 piezometers were installed from January to March 2023, including ten pairs and one unpaired downslope site, where a suitable upslope site could not be located. Piezometers were constructed from 3 m long PVC pipe (ø = 60 mm) with 8 mm holes drilled at 20 mm intervals along the lower 1.5 m of the pipe to allow groundwater infiltration. The sub-soil section of the pipe was covered in a drainage mesh sock to restrict fine soil particles entering the piezometer. A hand hole-borer (ø = 100 mm) was used to dig a hole up to 2.8 m deep at each site, the piezometer was inserted vertically into the hole and the outsides filled with soil to secure it in place. At sites where a depth of 2.8 m could not be achieved due to the presence of fist-sized rocks, the pipe was cut short (see Appendix A - Table A1 for piezometer depths). Piezometers were capped to exclude rainfall and minimise groundwater loss via evaporation. Water samples were retrieved using a 40 mm diameter cup suspended in the piezometer.

Sampling regime

Groundwater from the piezometers was sampled on 24 occasions at monthly intervals from April 2023 to March 2025. Between 20 – 50 mL was filtered through 0.45 µm Minisart filters (Sartorius, Germany) into sample bottles. Samples were then frozen and stored at -18°C until analysis.

Groundwater samples were analysed by Hill Laboratories (Hamilton) for ammoniacal-N, nitrite-N, nitrate-N, and dissolved reactive phosphorus-P (DRP) using a flow injection analyser. Ammoniacal-N concentrations were determined using the phenol/hypochlorite colorimetry method (APHA 4500-NH_3H modified), with a detection limit of 0.010 mg L^{-1}. Nitrite-N concentrations were determined using the automated azo dye colorimetry method (APHA 4500-NO_3^- modified), with a detection limit of 0.002 mg L^{-1}. Nitrate-N was calculated by first using the automated cadmium reduction method (APHA 4500 NO_3^- modified; detection limit of 0.002 mg L^{-1}) to determine nitrate-N + nitrite-N, then the nitrite-N concentration was deducted from this to give nitrate-N concentration, with a detection limit of 0.001 mg L^{-1}. Dissolved reactive phosphorus was determined using the molybdenum blue colorimetry method (APHA 4500-P modified) with a detection limit of 0.004 mg L^{-1}.

Data analysis

Dwelling density calculation

Spencer Road dwelling density was calculated from urban dwelling data obtained from Land Information New Zealand (LINZ, 2025a, 2025b) for both a dwelling count and building polygon layer. As these layers were not complete, points were manually added to unrecorded dwellings in the building polygon layer, with a limit of one point per property boundary as an approximation of septic containment systems. Dwelling density was then calculated by aggregating dwelling points into 0.1 km^2 grid squares using ArcGIS Pro (Esri, 2025).

Statistical analysis

Statistical tests could not be undertaken for group comparisons between native vegetation and urban land use sites, sites located upslope and downslope of dwellings, and temporal trends due to insufficient sample sizes, inconsistent sample availability and large variation in nutrient concentrations. Therefore, only descriptive statistics have been generated, with only general trends comparing groups reported.

Preliminary data analysis identified that sites T5 -above and T7-below exhibited nutrient patterns distinct from all other sites. These sites have therefore been identified separately in subsequent figures to highlight these trends.

Results

Sampling success and hydrological variability

Over the 2-year study period, a total of 180 water samples were collected. The sampling success rate (i.e., the times the water table was high enough to retrieve a sample) appeared to be driven by the trends in lake level (data from Bay of Plenty Regional Council, 2025) with the overall sample rate declining throughout the study until all piezometers were dry in February–March 2025 when the lake level was at its lowest (297.75 m; Figure 2.2). The highest monthly sampling success rate occurred in May 2023 (81 %), following a 0.27 m rise in lake level that occurred over a 12-day period, 19 days prior to sampling. The highest sample rate for both years occurred in May, as well as June in the second year. Site-specific sample collection rates were highest at T9-below (87.5 %), T5-above (75 %) and T3-below (70.8 %), while T6-below and T1-above never contained water (Appendix A - Table A1). Thirteen sites had a sample collection rate of less than 50 %. The artesian spring had steady groundwater flow and consistent nutrient concentrations throughout the entire study.

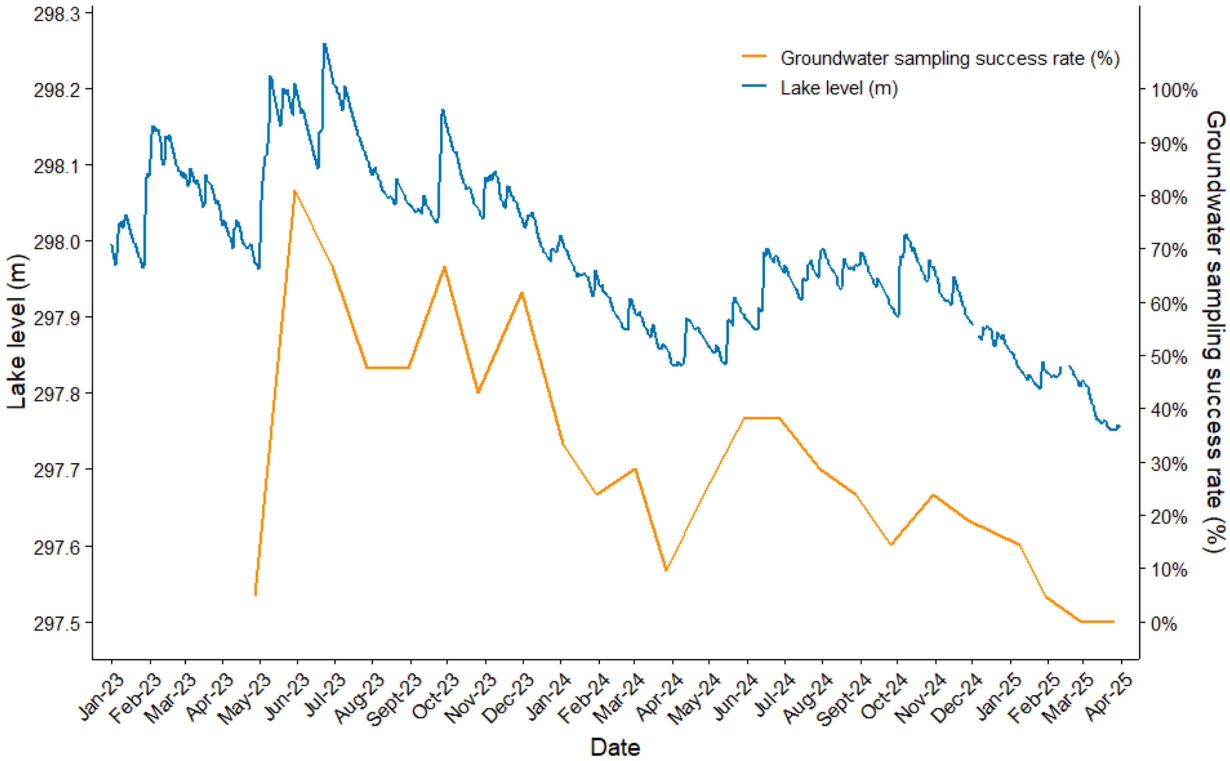

Figure 2.2. Monthly groundwater sampling rate (%) for samples collected between May 2023 and April 2025, and Lake Tarawera lake level data (m) measured at Te Wairoa by the Bay of Plenty Regional Council (2025). Sampling success rate is the total number of samples.

Nutrient trends across all sites

Nitrate-N concentrations varied from <0.002 mg L^{-1} to 7.4 mg L^{-1}, with an overall weighted mean (± SD) across all sites of 0.43 ± 0.96 mg L^{-1} (Figure 2.3; Appendix A - Table A2). The largest mean and maximum nitrate-N concentrations over the entire study period occurred at T4-below (mean = 1.68 ± 2.05 mg L^{-1}, max = 7.40 mg L^{-1}), and seven other sites recorded maximum values above 1.0 mg L^{-1}.

Nitrite-N concentrations were lower, ranging from <0.002 mg L^{-1} to 0.9 mg L^{-1}, with the highest mean and maximum values occurring at T7-below (mean = 0.20 ± 0.40 mg L^{-1}, max = 0.91 mg L^{-1}). The overall weighted mean was 0.01 ± 0.07 mg L^{-1}. Most sites consistently recorded values below the detection limit (<0.002 mg L^{-1} for 54.5 % of all samples), with three sites recording maximum nitrite-N concentrations above 0.1 mg L^{-1}.

Ammoniacal-N concentrations had the largest range from <0.010 mg L^{-1} to 61 mg L^{-1}. Eight sites had a maximum value >1.0 mg L^{-1}, with the highest maximum and mean values occurring at T5-above (mean = 22.3 ± 20.2 mg L^{-1}, max = 61.0 mg L^{-1}) and the second highest maximum at T7-below (mean = 3.43 ± 6.31 mg L^{-1}, max = 14.6 mg L^{-1}). Thirteen sites had mean ammoniacal-N values greater than 0.1 mg L^{-1}, with the overall weighted mean being 0.22 mg L^{-1}.

DRP varied from <0.004 mg L^{-1} to 8.7 mg L^{-1}, with an overall weighted mean of 0.07 mg L^{-1}. Sixteen sites recorded values consistently below the detection limit (<0.004 mg L^{-1} for 57.1 % of all samples), while seven sites recorded maximum values above 0.1 mg L^{-1}. Similar to nitrite-N, the highest maximum and mean value also occurred at T7-below (mean = 3.44 ± 3.38 mg L^{-1}, max = 8.70 mg L^{-1}).

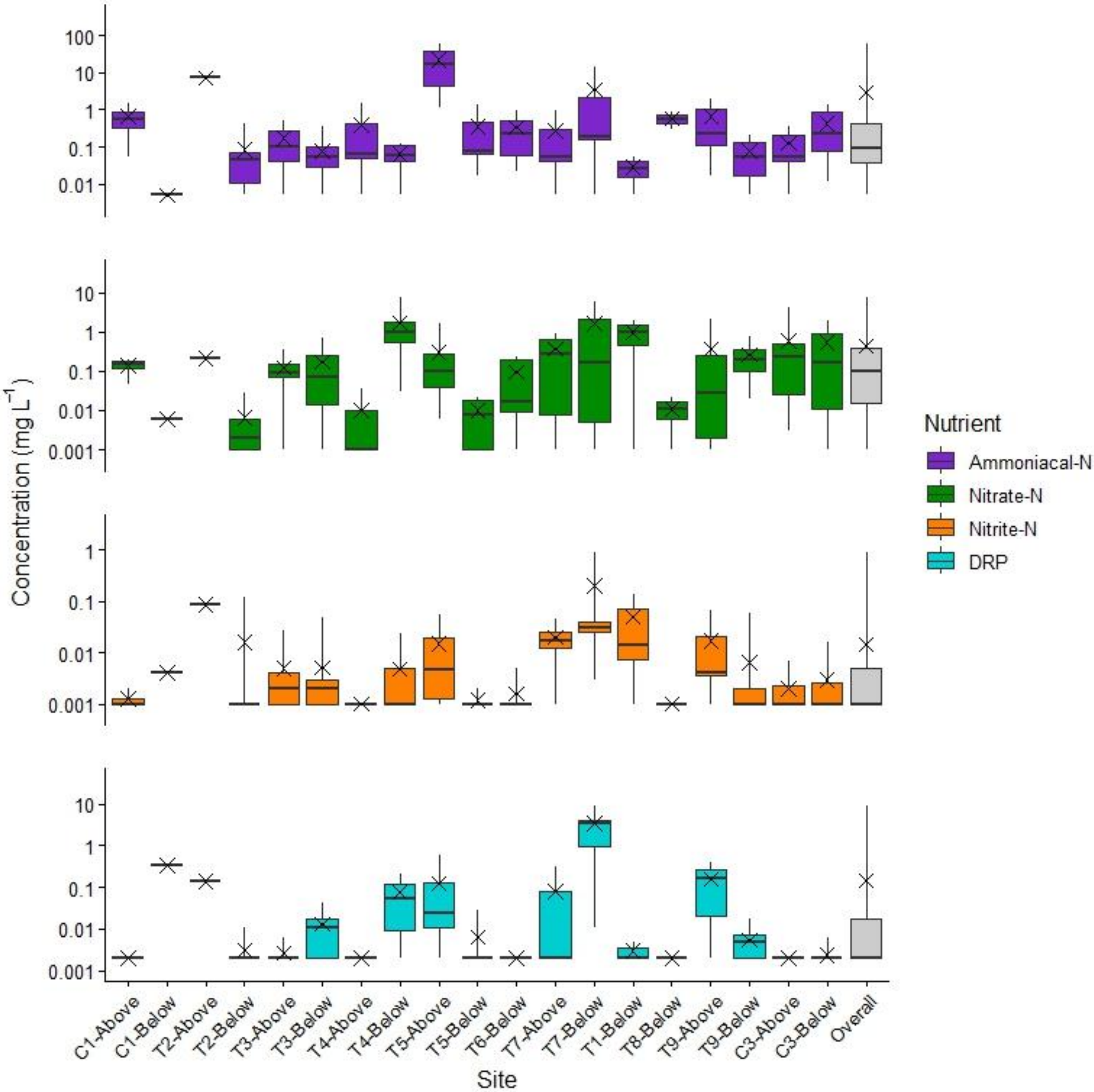

Figure 2.3. Boxplots of groundwater nutrient concentrations at each piezometer site (DRP is dissolved reactive phosphorus). Plots use logarithmic y-axes with differing scales. Boxes indicate the interquartile range, horizontal black lines show median values, crosses are mean values, and whiskers are the minimum and maximum values. Grey boxes represent overall summary statistics for each nutrient. Sample size varies between sites (1 – 21).

Trends at significant sites

T5-above

Ammoniacal-N was consistently the dominant nutrient at T5-above, with concentrations one or two orders of magnitude higher (with concentrations of up to 61 mg L^{-1}) than other measured nutrients (Figure 2.4). Periods of elevated nitrate-N concentrations (up to 1.61 mg L^{-1}) were observed when ammoniacal-N concentrations were relatively low (as low as 3.8 mg L^{-1}) between June and August 2024. In addition to these patterns, the groundwater at T5-above frequently exhibited strong sewage odour and discolouration. To avoid the significant bias this site creates on summary statistics, T5-above was categorised separately from all other sites and excluded from summary statistics.

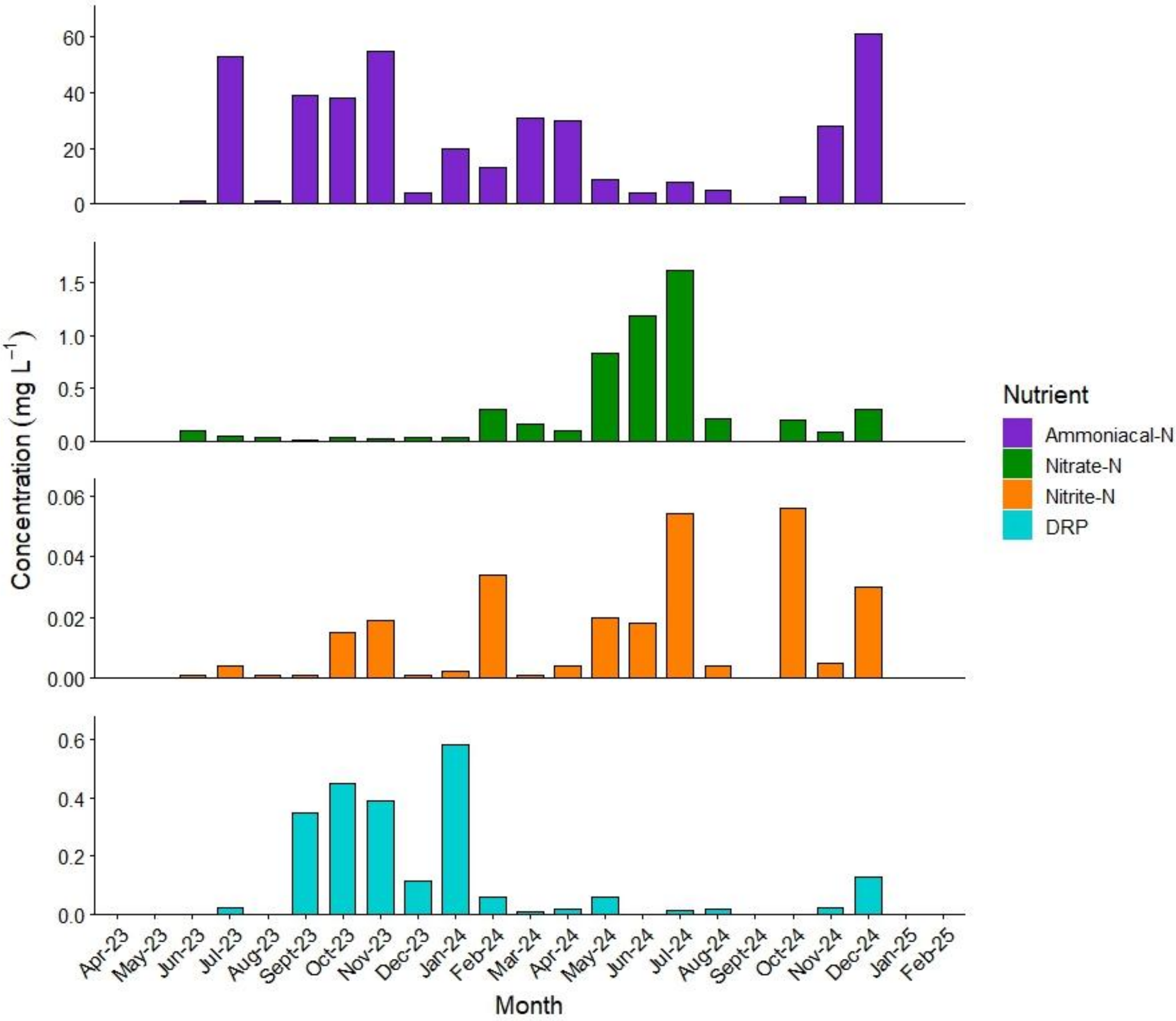

Figure 2.4. Monthly groundwater nutrient concentrations at piezometer site T5-above. Y-axis scales differ among plots. *DRP* is dissolved reactive phosphorus.

T3-below

T3-below experienced a shift from ammoniacal-N dominance in 2023 to nitrate-N dominance in 2024, with ammoniacal-N peaking in November 2023 at 0.36 mg L^{-1}, before declining below 0.1 mg L^{-1} for the remainder of the study (Figure 2.5). From January 2024, nitrate-N concentrations notably increased to above 0.1 mg L^{-1} in 2024, peaking at 0.67 mg L^{-1} in February 2024. This trend of dominant N form switching was also observed at several other sites, including T3-above, T9-below, C3-above and C3-below. Nitrite-N and DRP were consistently low (<0.05 mg L^{-1}).

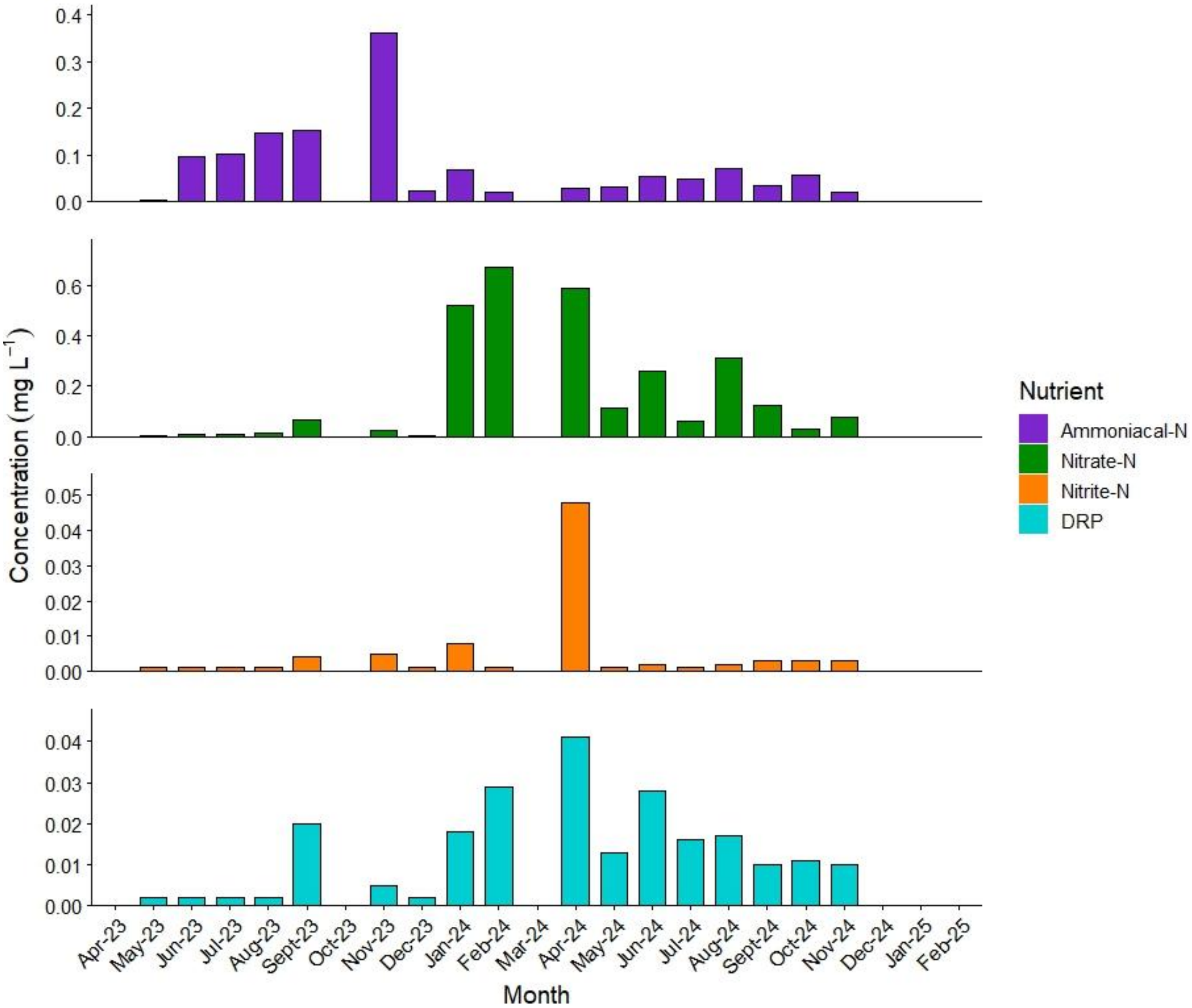

Figure 2.5. Monthly groundwater nutrient concentrations at piezometer site T3-below. Y-axis scales differ among plots. *DRP* is dissolved reactive phosphorus.

T7-below

Excluding T5-above, T7-below recorded the highest values for nitrite-N, ammoniacal-N and DRP across all sites (Figure 2.6). Only five groundwater samples were successfully collected from this site. Initial samples in May – June 2023 were relatively low in nutrient concentrations (ammoniacal-N <0.2 mg L^{-1}, nitrite-N <0.05 mg L^{-1}, nitrate-N <0.05 mg L^{-1}, DRP <1.0 mg L^{-1}), before ammoniacal-N sharply increased to its peak value (14.6 mg L^{-1}) during the next successful sampling in November 2023. In May 2024, maximum values were recorded for nitrate-N (5.9 mg L^{-1}), nitrite-N (0.91 mg L^{-1}), and DRP (8.7 mg

L^{-1}), while ammoniacal-N was also elevated (2.2 mg L^{-1}). Nitrate-N and DRP remained elevated in June 2024, which was the final time this piezometer contained groundwater.

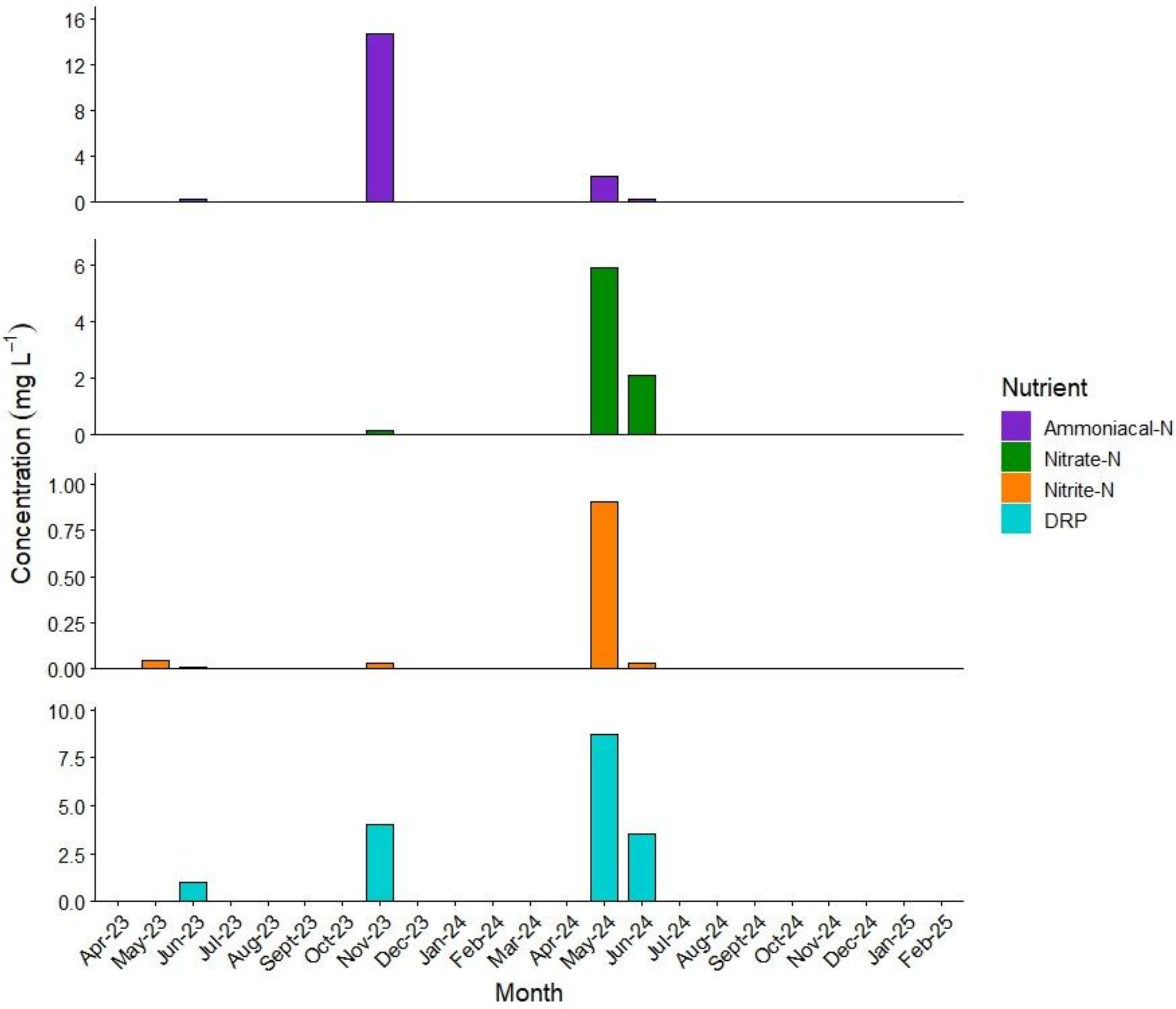

Figure 2.6. Monthly groundwater nutrient concentrations at piezometer site T7-below. Y-axis scales differ among plots. *DRP* is dissolved reactive phosphorus.

Spatial trends

Influence of dwelling density

Native vegetation sites were located in areas with low dwelling density (0 – 1 dwellings per 0.1 km^2), whereas urban sites were located over a range of dwelling density areas (0

– 5 dwellings per 0.1 km^2; Figure 2.7). The highest areas of dwelling density occurred in close proximity to Spencer Road.

Due to a low number of samples, statistical tests could not be undertaken for this section. Areas of high nitrate-N concentrations generally occurred over a range of dwelling densities (1 – 5 dwellings per 0.1 km^2; Figure 2.7). In contrast, ammoniacal-N was highest near the centre of the urban strip, in areas with 5 dwellings per 0.1 km^2 (Figure 2.8). Two small peaks in nitrite-N occurred in the urban area, though these were low (<0.2 mg L^{-1}) and in areas of low to moderate housing density (2 – 5 dwellings per 0.1 km^2; Figure 2.9). Two peaks in DRP concentrations occurred exclusively within the urban area, where dwelling density is 4 – 5 dwellings per 0.1 km^2 (Figure 2.10).

Urban vs. native vegetation sites

Sites located in areas draining urban/agricultural sub-catchment land use tended to experience higher mean nutrient concentrations than native vegetation sites, although the magnitude and spatial distribution varied by nutrient. Nitrate-N and ammoniacal-N showed higher mean concentrations at several urban sites in the centre of the urban strip compared with native vegetation sites, particularly at T7-below and T4-below for nitrate-N, and T5-above and T7-below for ammoniacal-N (Appendix A - Table A3). For most other urban sites, mean nitrate-N and ammoniacal-N concentrations were similar to native vegetation sites. Mean nitrite-N and DRP concentrations at urban sites were similar to or slightly elevated in comparison to native vegetation sites; only T7-below had substantially elevated mean nitrite-N and DRP concentrations relative to all other sites.

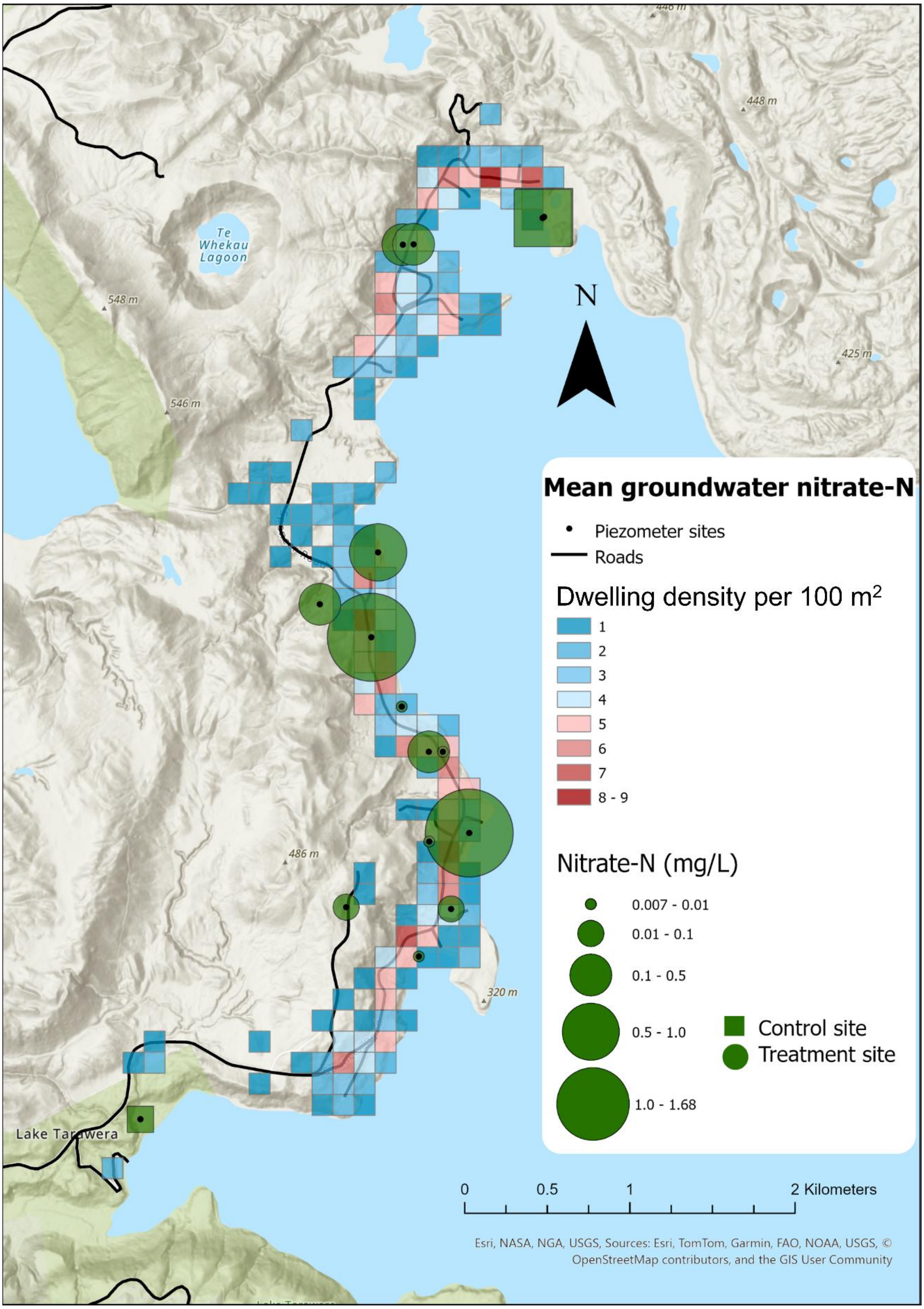

Figure 2.7. Spatial distribution of mean nitrate-N concentrations at control and treatment sites. Control sites drain sub-catchment with native vegetation land use, while treatment sites drain urban/agricultural sub-catchment land use. Symbol size represents nutrient concentration, while shape and colour distinguish site type and nutrient type. Coloured squares represent dwelling density per 100 m^2.

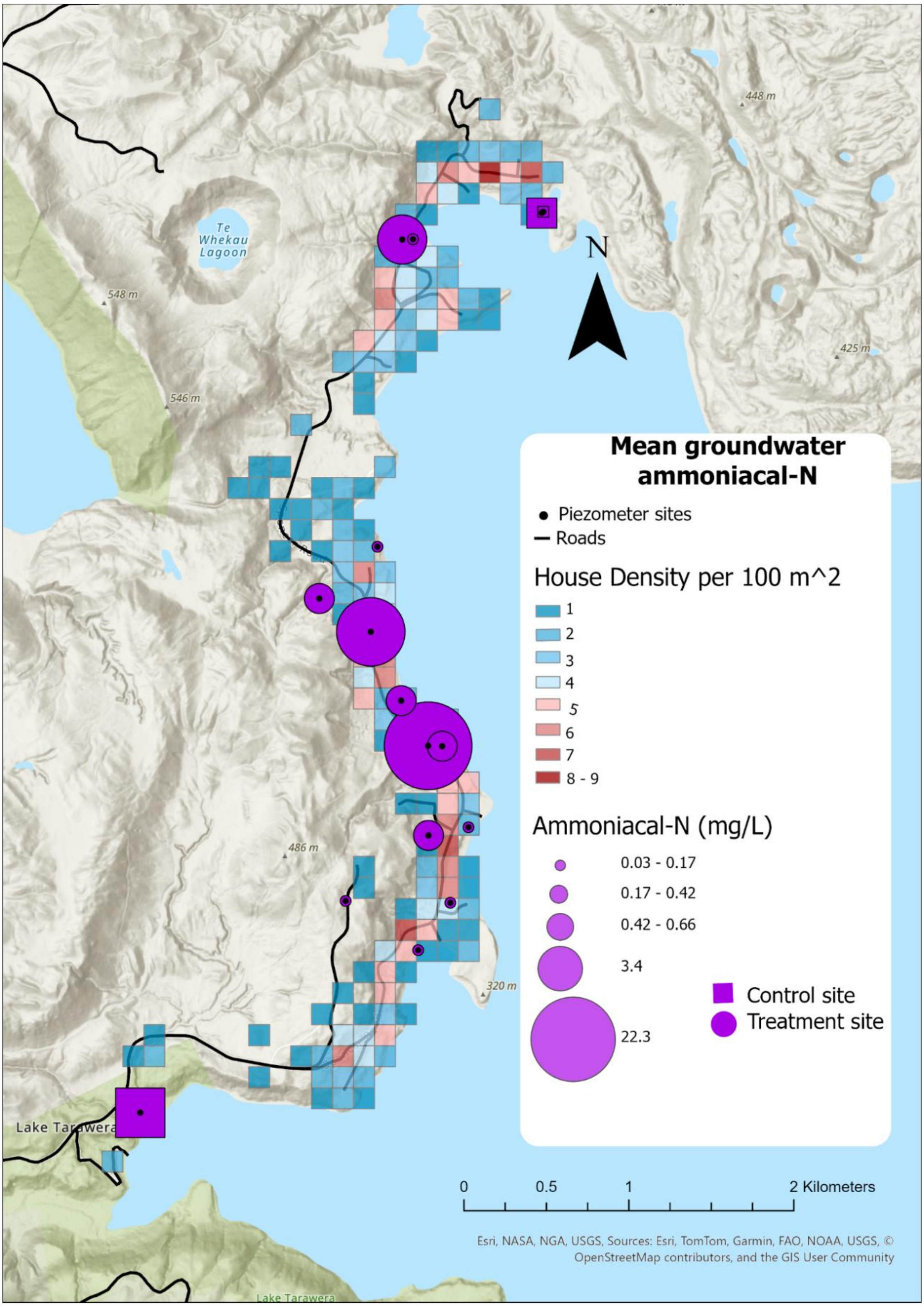

Figure 2.8. Spatial distribution of mean ammoniacal-N concentrations at control and treatment sites. Control sites drain sub-catchment with native vegetation land use, while treatment sites drain urban/agricultural sub-catchment land use. Symbol size represents nutrient concentration, while shape and colour distinguish site type and nutrient type. Coloured squares represent dwelling density per 100 m^2.

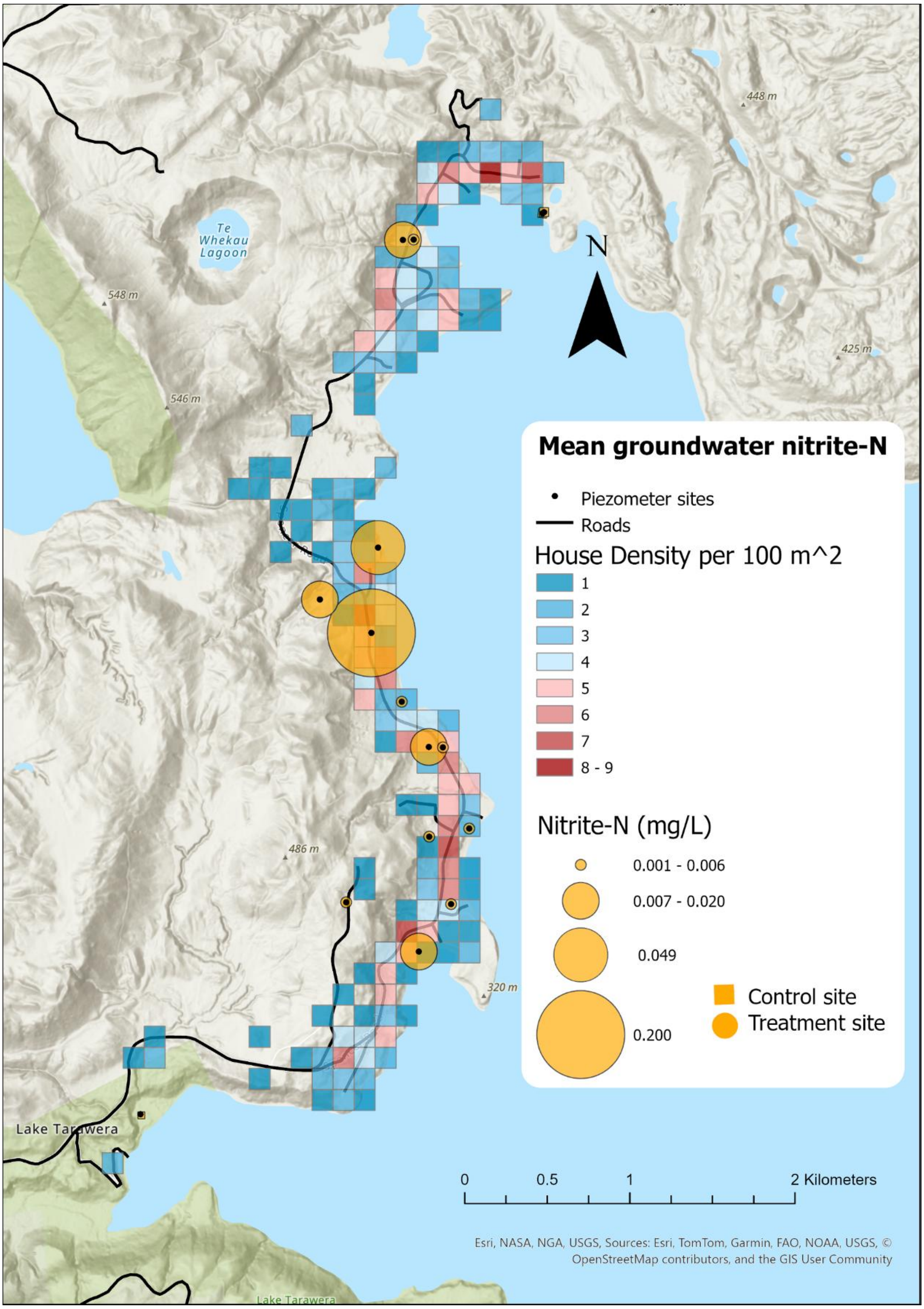

Figure 2.9. Spatial distribution of mean nitrite-N concentrations at control and treatment sites. Control sites drain sub-catchment with native vegetation land use, while treatment sites drain urban/agricultural sub-catchment land use. Symbol size represents nutrient concentration, while shape and colour distinguish site type and nutrient type. Coloured squares represent dwelling density per 100 m^2.

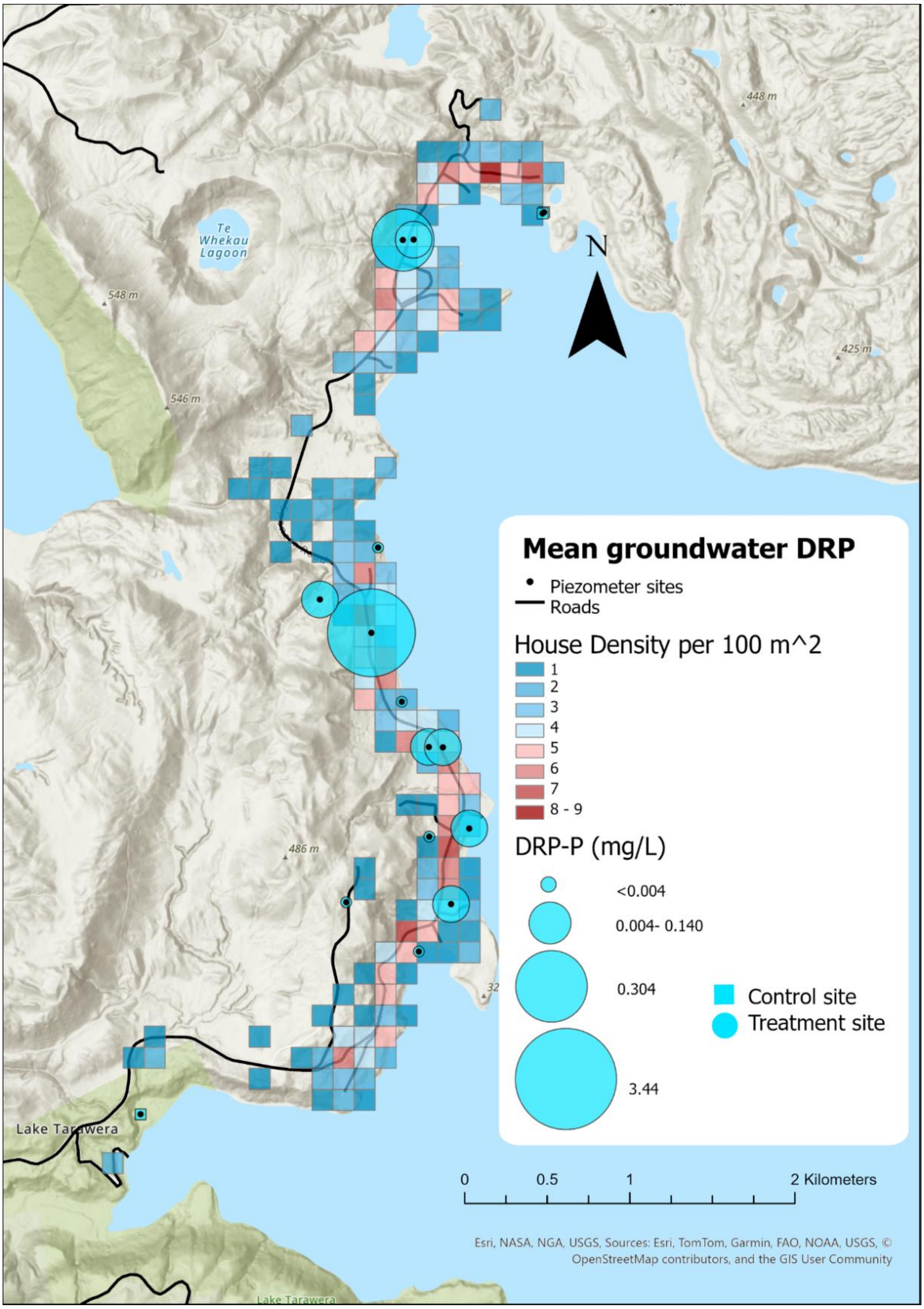

Figure 2.10. Spatial distribution of mean dissolved reactive phosphorus (DRP) concentrations at control and treatment sites. Control sites drain sub-catchment with native vegetation land use, while treatment sites drain urban/agricultural sub-catchment land use. Symbol size represents nutrient concentration, while shape and colour distinguish site type and nutrient type. Coloured squares represent dwelling density per 100 m^2.

Upslope vs. downslope nutrient patterns

Due to inconsistent groundwater availability and the large variation in measured nutrient contaminants, statistical differences in concentrations between upslope and downslope sites with respect to potential septic contamination could not be assessed. Mean DRP was higher at downslope sites (0.20 ± 0.77 mg L^{-1}) compared to upslope sites (0.04 ± 0.07 mg L^{-1}; Table 2.1). Site T5-above had notably higher mean ammoniacal-N (22.33 ± 20.22 mg L^{-1}) than upslope sites (0.47 ± 1.06 mg L^{-1}) and also downslope sites (0.34 ± 0.75 mg L^{-1}), while mean upslope and downslope ammoniacal-N was similar. Mean nitrate-N was similar at T5-above (0.29 ± 0.45 mg L^{-1}) and other upslope sites (0.30 ± 0.21 mg L^{-1}), which were both lower than the mean value for downslope sites (0.52 ± 0.60 mg L^{-1}). Mean nitrite-N was similar across all groups (≤ 0.02 mg L^{-1}).

Table 2.1. Summary statistics for nutrient concentrations at upslope sites, downslope sites, and site T5-above. *DRP* is dissolved reactive phosphorus *stdev* is standard deviation, and *min* and *max* are minimum and maximum values. Mean and standard deviation values are weighted. The dataset includes 44 datapoints from 8 above sites, 94 datapoints from 10 below sites and 18 datapoints from T5-above.

Nutrient	Site group	*n*	Weighted mean	Weighted stdev	Min	Max
Nitrate-N (mg L^{-1})	Upslope	44	0.30	0.21	< 0.002	4.00
	Downslope	94	0.52	0.60	< 0.002	7.40
	T5-Above	18	0.29	0.45	0.006	1.61
Nitrite-N (mg L^{-1})	Upslope	44	0.01	0.01	< 0.002	0.09
	Downslope	94	0.02	0.04	< 0.002	0.91
	T5-Above	18	0.02	0.02	< 0.002	0.06
DRP (mg L^{-1})	Upslope	44	0.04	0.07	< 0.004	0.38
	Downslope	94	0.20	0.77	< 0.004	8.70
	T5-Above	18	0.13	0.18	< 0.004	0.58
Ammoniacal-N (mg L^{-1})	Upslope	44	0.47	1.06	< 0.010	7.30
	Downslope	94	0.34	0.75	< 0.010	14.60
	T5-Above	18	22.33	20.22	1.110	61.00

Temporal trends

Seasonal pattern

No clear seasonal pattern was observed for downslope sites across all mean nutrient concentrations (Figure 2.11; Appendix A - Table A4). Mean ammoniacal-N peaked in Spring 2023 (0.89 mg L^{-1}, SD = 3.00 mg L^{-1}) and reached a minimum in Spring 2024 (0.04 mg L^{-1}, SD = 0.01 mg L^{-1}), with mean nitrite-N also recording minimum values in Spring 2024 and Summer 2023/24 (0.002 mg L^{-1}, SD = 0.001 mg L^{-1}). Minimum mean concentrations for nitrate-N (0.091 mg L^{-1}) and DRP (0.005 mg L^{-1}) occurred in Autumn 2023, before both nutrients, as well as nitrite-N, peaked the following year in Autumn 2024 (1.12, 1.46 and 0.16 mg L^{-1}, respectively). Overall, mean nitrite-N remained consistently low throughout the study, while mean ammoniacal-N was elevated from Autumn 2023 to Autumn 2024, before declining sharply in subsequent seasons. In contrast, mean nitrate-N increased steadily throughout the entire study period, whilst DRP showed elevated concentrations from Autumn 2024 onwards.

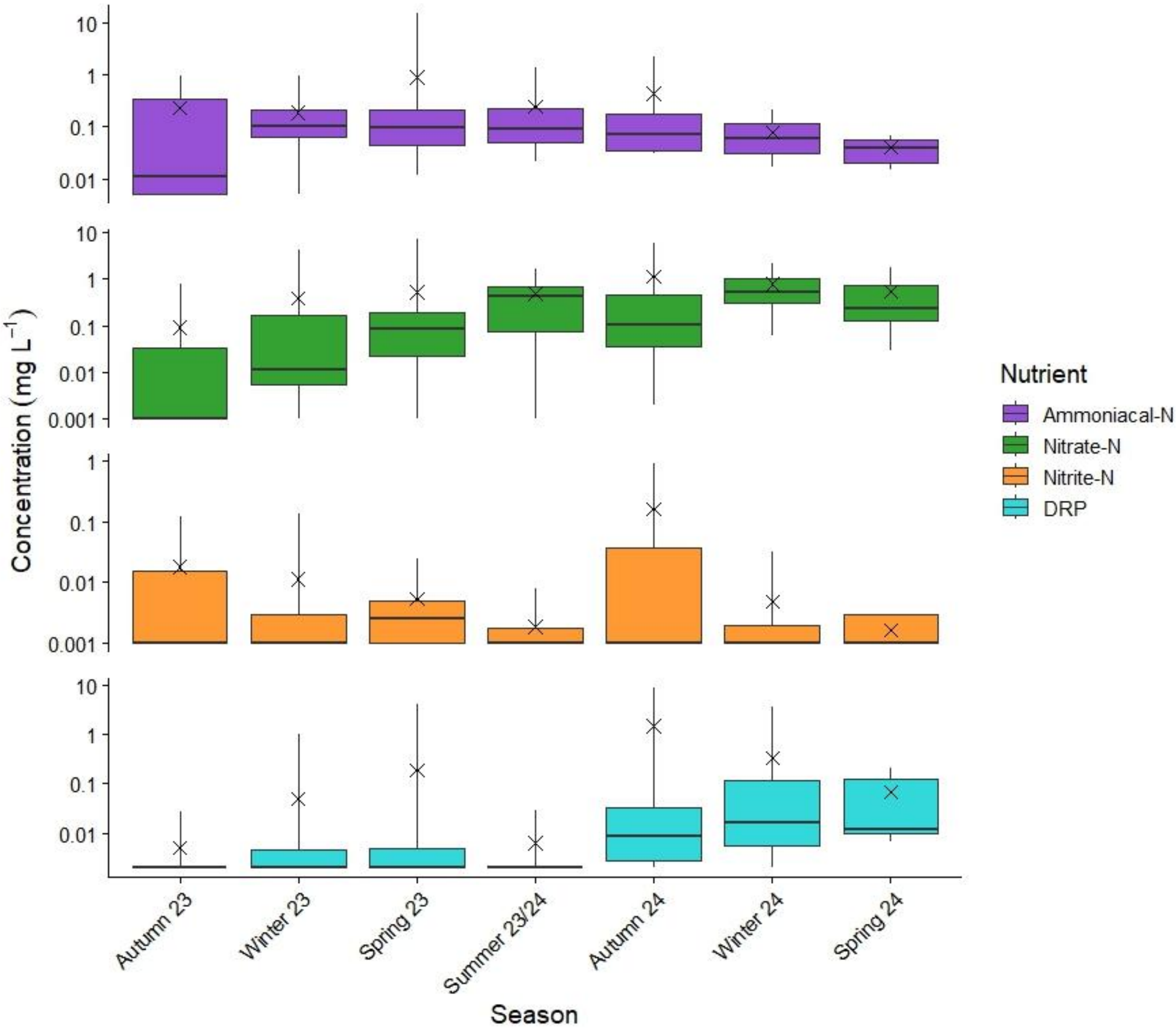

Figure 2.11. Seasonal nutrient concentrations for below sites only. *DRP* is dissolved reactive phosphorus. Panels are displayed on logarithmic y-axes with differing scale ranges. Horizontal black lines are median values, crosses are mean values, boxes represent the interquartile range, and whiskers extend to minimum and maximum values. Summer was defined as December - February, Autumn as March – May, Winter as June – August, and Spring as September – November. Number of samples ranged from 22 in Winter and Spring 2023, to six in Autumn 2024. Number of sites ranged from 10 sites in Autumn and Winter 2023, to three sites in Spring 2024.

Annual trend

Mean nitrate-N at downslope sites appeared to be somewhat higher in the second year of the study (0.80 ± 0.98 mg L^{-1}), compared to the first year (0.39 ± 0.64 mg L^{-1}; Figure 2.12; Appendix A - Table A5). A similar trend was observed for DRP (second year mean = 0.48 ±1.56 mg L^{-1}; first year mean = 0.08 ± 0.35 mg L^{-1}). In contrast, mean ammoniacal-N was higher in the first year (0.43 ± 1.00 mg L^{-1}) than the second year (0.14 ± 0.29 mg L^{-1}). There was no apparent difference in mean nitrite-N concentrations between years. However, the significance of these differences could not be assessed with statistical tests due to uneven sample sizes and number of sites sampled between years (first year n = 66 samples, 10 sites; second year n = 28 samples, 5 sites).

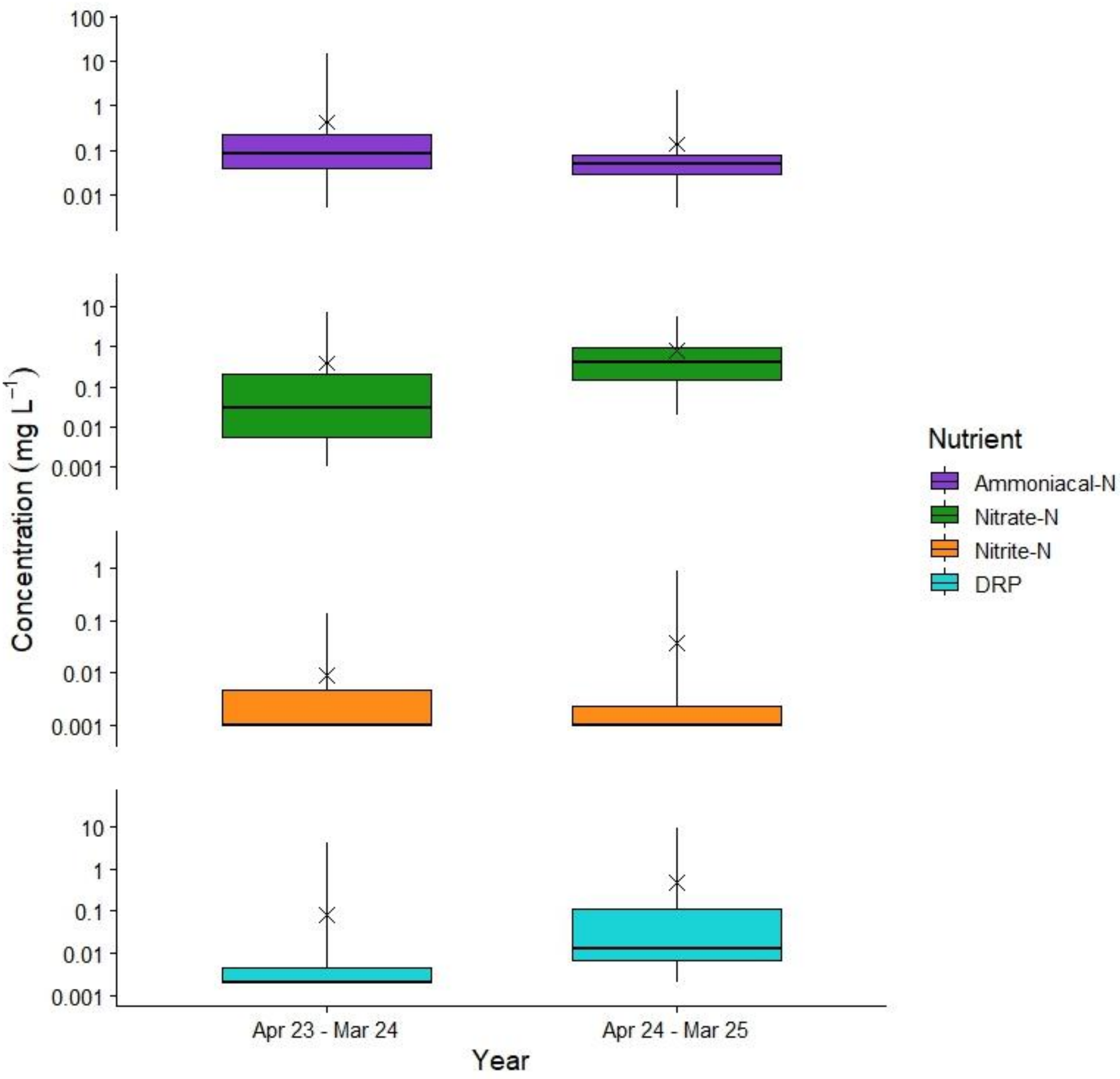

Figure 2.12. Boxplots showing annual piezometer nutrient concentrations for downslope sites. *DRP* is dissolved reactive phosphorus. Each plot is shown on a logarithmic y-axis, with differing scale ranges. Horizontal black lines are median values, crosses are mean values, boxes represent the interquartile range (IQR). Circles represent outliers that are 1.5 x IQR from the quantiles. The dataset includes 66 samples from 10 sites in the April 2023 – March 2024 year, and 28 samples from 5 sites in the April 2024 – March 2025 year.

Discussion

Septic systems are an important point source of nutrient pollution to groundwater in lake catchments and can contribute to overall nutrient enrichment in lake ecosystems (Afolalu et al. 2022). Potential nutrient contamination from septic tank leachate to shallow groundwater along the urbanised western margin along Spencer Road at Lake Tarawera was investigated. Groundwater was sampled monthly from April 2023 to March 2025 using 21 shallow piezometers installed in pairs upslope and downslope of dwellings, including two pairs located within native vegetation. Analyses assessed general nutrient trends, differences between upslope and downslope sites, native vegetation and urban sites, seasonal and annual variation, and the relationship between nutrient concentrations and dwelling density.

Overall nutrient trends

Ammoniacal-N

The weighted mean ammoniacal-N concentration across all sites was 0.22 mg L^{-1}, with eight sites exceeding 1 mg L^{-1}, which are concentrations typically associated with septic plumes (Harman et al., 1996; Katz et al., 2011). Under aerobic conditions, ammoniacal-N is rapidly converted to nitrate-N within approximately 2 m below septic drainfields and usually measures less than 1 mg L^{-1} within 10 – 50 m downslope (Harman et al., 1996; Katz et al., 2011). Ammoniacal-N is the dominant nitrogen species in septic systems and is not commonly derived from natural sources under aerobic conditions, other than animal waste or geothermal activity (Lusk et al., 2017). Therefore, elevated concentrations are likely to indicate a recent or nearby input of septic-derived wastewater that has not undergone complete nitrification at some sites (Baer et al., 2019; Lewandowski et al., 2015).

Groundwater quality has previously been assessed at Lake Tarawera for evidence of septic contamination by Scholes (2007). This study was conducted from March 2003 – February 2005, with groundwater monitored on average bimonthly and at higher frequency during spring. Shallow wells were installed along the western shoreline to detect septic-derived contaminant plumes, including locations downslope of several piezometer sites investigated in the current study. Across the monitoring period, ammoniacal-N concentrations measured 0.002 to 0.811 mg L^{-1}, with elevated values interpreted as evidence of septic contamination (Table 2.2). Comparison of the upslope piezometers from this study with the corresponding downslope shallow wells reported by Scholes (2007) reveals that high ammoniacal-N or nitrate-N concentrations (>0.5 mg L^{-1}) occurred in both datasets at T4-Below – Stoney Point, T5-Below – Tarapatiki Point, and C3-Below – Bay View Road (Table 2.2). This suggests that nutrient-enriched groundwater originating within the urban zone may retain elevated nitrogen species concentrations as it approaches the lakeshore. Where elevated concentrations were not observed in both locations in a site pair, differences may reflect changes in septic loading between study periods, nutrient attenuation during subsurface transport, or the absence of a direct groundwater path linking paired sites. Nevertheless, the present study demonstrates that nutrient enrichment detected in earlier shoreline monitoring likely originates from within the urban settlement.

Table 2.2. Comparison of shallow groundwater ammoniacal-N and nitrate-N concentration ranges measured at monitoring sites located along the same transect from this study (piezometers) and Scholes (2007; shallow wells).

	Ammoniacal-N range (mg L^{-1})		Nitrate-N range (mg L^{-1})	
Paired piezometer–shallow well	Piezometer[1]	Shallow well[2]	Piezometer[1]	Shallow well[2]
T2-below – Boatshed Bay	< 0.01 – 0.41	0.006 – 0.036	< 0.002 – 0.028	0.375 – 0.706
T4-below – Stoney Point	< 0.01 – 0.12	0.001 – 0.811	0.030 – 7.400	0.001 – 0.754
T5-below – Tarapatiki Point	0.02 – 1.32	0.007 – 0.687	< 0.002 – 0.021	0.005 – 0.040
T8-below – Cliff Rd	0.29 – 0.86	0.029 – 0.055	< 0.002 – 0.021	0.001 – 0.020
C3-below – Bay View Rd	0.01 – 1.33	0.049 – 0.091	< 0.002 – 1.910	0.001 – 0.754

[1] This study; [2] Scholes (2007)

Comparisons with other New Zealand and global lakes further support the likelihood of septic contamination at Lake Tarawera. Ammoniacal-N concentrations reported in background groundwater of New Zealand volcanic lakes are typically <0.2 mg L^{-1}, compared with <0.01 – 0.81 mg L^{-1} in lake settlements with OWTS, and 0.1 – 228 mg L^{-1} in heavily septic-impacted areas overseas (

Table 2.3). Shallow groundwater concentrations at Lake Tarawera span this full range, with some sites consistent with background conditions and others overlapping concentrations characteristic of septic plumes, indicating that there is considerable spatial variation in septic leachate reaching Lake Tarawera along Spencer Road.

Table 2.3. Comparison of reported ranges for ammoniacal-N, nitrate-N and dissolved reactive phosphorus (DRP) concentrations in shallow groundwater from various freshwater systems in New

Zealand and internationally. Studies are ordered from least influenced by wastewater to most influenced, with values from this study highlighted in bold.

Degree of wastewater influence	System	Upslope land use	Study duration (sampling regime)	Ammoniacal-N (mg L^{-1})	Nitrate-N (mg L^{-1})	DRP (mg L^{-1})
Background conditions	Lake Taupō[1]	Wetland	2 years (seasonal)	0.02 – 0.18	0.02 – 0.09	< 0.004 – 0.013
	Lake Rotokākahi[2]	Exotic forest/pasture	6 months (monthly)	0.01 – 0.11	0.50 – 1.75	0.005 – 0.04
	Lake Huron (Canada)[3]	Native forested dunes	2 years (monthly in warm season)	0.03 – 0.04	0.03 – 0.04	0.004 – 0.008
	Illwald Forest (France)[4]	Remnant alluvial forest	1.5 years (monthly)	0.06 – 0.07	0.26 – 2.64	0.017 – 0.12
Lakeshore settlement with OWMS	Lake Rotoiti[5]	120 dwellings	2 years (bimonthly)	< 0.01 – 0.27	< 0.01 – 0.05	0.002 – 0.11
	Lake Tarawera[5]	470 dwellings	2 years (bimonthly)	< 0.01 – 0.81	< 0.01 – 0.75	< 0.001 – 0.11
	Lake Tarawera[6]	**470 dwellings**	**2 years (monthly)**	**< 0.01 – 61**	**< 0.01 – 7.40**	**< 0.004 - 8.7**
Township	Ongare Point[5]	51 dwellings	9 years (~3x annually)	< 0.01 – 0.51	0.24 – 7.63	0.001 – 0.07
Known septic plume	Lake Huron (Canada)[7]	550 dwellings	3 days	0.90 – 6.90	0.40 – 6.90	< 0.001 – 0.04
	Ontario (Canada)[8]	School septic tank	10 days	0.27 – 128	0 – 112	< 0.01 – 1.5
	Ontario (Canada)[9]	Individual septic systems	30 years (~4x annually)	0.10 – 228	0.10 – 107	0.02 – 4.6
Dense urban city	Jackson, Florida (USA)[10]	highly-developed Lower St Johns river basin	3 years (seasonally/ biweekly)	1.19	0 – 43.7	0 – 4.0

[1] Eser & Rosen (1999); [2] Noakes (2016); [3] Rakhimbekova et al. (2021); [4] Takatert et al. (1999); [5] Scholes (2006); [6] **this study**; [7] Baer et al. (2019); [8] Harman et al. (1996); [9] Robertson et al. (2019); [10] Ouyang & Zhang (2012).

There is some evidence that other potential sources of ammoniacal-N may occur within the western subcatchment. For example, site T2-above recorded an unusually high ammoniacal-N concentration of 7.2 mg L^{-1} in the only sample collected from this site. As this site was located within farmland with no nearby septic systems, livestock manure runoff is the most likely explanation. Collins et al. (2017) found that groundwater ammoniacal-N concentrations from dairy/cropping farmland in the lower Rangitikei catchment (North Island), ranged from 0.54 – 1.05 mg L^{-1}, although the T2-above value exceeds this.

Additionally, two native vegetation sites (C1-above and C3-below) also recorded ammoniacal-N concentrations exceeding 1 mg L^{-1} despite no known septic systems nearby. At C3-below, elevated ammoniacal-N concentrations may have been accumulated through soil mineralisation during an extended period of higher water table levels (Li et al., 2025), which may be possible given that this site was located in a slight depression and the water level within the piezometer was consistently high. In contrast, this mechanism is unlikely at C1-above, where soils were free-draining. These observations indicate that elevated ammoniacal-N can arise from agricultural runoff or natural processes under specific conditions. However, elevated concentrations at free-draining, urban sites located downslope of dwellings, including T5-above, T5-below, and T7-below, strongly suggest that these sites are instead experiencing septic-derived groundwater contamination.

Nitrate-N

The mean nitrate-N concentration across all sites was 0.44 mg L^{-1}, with values ranging <0.002 to 7.4 mg L^{-1}. This mean lies between the estimated 80th percentile national means for oxic (1.65 mg L^{-1}; 23 minimally impacted sites) and anoxic (0.04 mg L^{-1}; 37 sites of varying land use) groundwaters in New Zealand, suggesting that groundwater within the study area spans a broad range of redox conditions (Daughney et al., 2025). Similar to the ammoniacal-N patterns, nitrate-N concentrations observed in this study also span the ranges reported for a variety of background and septic-influenced systems (see

Table 2.3). Elevated nitrate-N can be an indicator of septic contamination as it is produced though the nitrification of ammonium and readily leaches into groundwater (Lusk et al., 2017). Several sites recorded maximum values that exceeded those reported for moderately impacted systems such as Lake Rotoiti, including T9-above (2.1 mg L^{-1}), C3-above (4.0 mg L^{-1}), T7-below (5.9 mg L^{-1}), and T4-below (7.4 mg L^{-1}). Site T9-above is located above dwellings and approximately 100 m downslope of pastural land use, which may explain this elevated concentration. For the remaining sites, it is likely these maximum values captured septic-derived groundwater inputs.

Nitrite-N

Although nitrite-N is not typically used as an indicator of septic contamination, it is an important intermediate species that forms during nitrification under oxic conditions and denitrification under anoxic conditions (Schullehner et al., 2017). Nitrite-N concentrations were low at most sites with a few localised areas of enrichment occurring sporadically. Elevated nitrite-N at T2-below and T1-below during high water table conditions, and at T7-below during a later low water table period, indicates that redox conditions varied through time and space within the urban zone of Lake Tarawera, confirming the occurrence of both nitrification and denitrification processes in the subsurface (Rogers et al., 2023).

DRP

Phosphorus is generally a poor indicator of septic effluent loading to groundwater as DRP is highly attenuated in sediment via sorption and precipitation reactions, which results in high spatial variability (Robertson et al., 2019). Attenuation rates also depend strongly on sediment type, effluent discharge rate, soil permeability and water table height (Wang et al., 2024). DRP concentrations across all sites ranged from <0.004 – 8.7 mg L^{-1}, with most sites having similar concentrations to those reported in other Te Arawa lakes, as well as concentrations measured in septic-free systems in New Zealand and overseas (

Table 2.3). Septic-derived DRP typically measures <1 mg L^{-1} within 1 – 50 m downslope of septic drainfields, though this can remain elevated if attenuation pathways are saturated or drainfield conditions are poor (Harman et al., 1996; Scholes, 2007; Robertson et al., 2019). Maximum values measured at T7-below (8.7 mg L^{-1}) and T5-above (0.58 mg L^{-1}) exceed typical values for background phosphorus conditions, potentially showing inadequate function of septic drainfields near these sites.

However, examining DRP as a septic indicator may be particularly challenging for Lake Tarawera. The catchment is a complex system of volcanic origin, composed of a mosaic of pyroclastic, rhyolite and ignimbrite deposits, which creates high spatial variability in groundwater flow and nutrient attenuation rates (Wilson, 2022). This was apparent during piezometer installation, where each site showed large differences in soil horizon composition, horizon thickness, and sediment grain sizes. As an in-depth physicochemical analysis of sediment at each site was beyond the scope of this study, it is not possible to determine whether localised high values, such as at T7-below, reflect natural sources or true septic contamination. DRP is therefore interpreted with caution as an indicator of septic contamination.

Site-specific trends

T5-above – a contaminated site

At T5-above, the consistent, exceptionally high ammoniacal-N concentrations (mean 22.3 mg L^{-1} and maximum 61 mg L^{-1}) are similar to reported values of ammoniacal-N concentrations in both septic plumes (see Table 2.3), and septic tank effluent itself, which range from 7 – 165 mg L^{-1} (Harman et al., 1996; Lusk et al., 2017; Robertson et al., 2019). A clogged or failing septic system can cause saturation of the drainfield and create anoxic conditions, leading to high ammoniacal-N accumulation in groundwater (Katz et al., 2011). Combined with visual observations of a consistently saturated soil condition and sewage odour at T5-above, it is highly likely that the T5-above piezometer is located directly in the plume of a failing septic system.

T7-below – an anomalous site

T7-below recorded high maximum values of 14.6 mg L^{-1} ammoniacal-N, 8.7 mg L^{-1} DRP, and 5.9 mg L^{-1} nitrate-N, which exceed ranges reported in comparable New Zealand studies and fall within ranges commonly associated with septic leachate. This site lies approximately 100 m downslope of farmland, making this an unlikely source for high nutrients, particularly ammoniacal-N which would be nitrified along this pathway. Although no nearby septic system was identified during site selection, the magnitude of the nutrient signals strongly indicates a wastewater source.

The maximum DRP value (8.7 mg L^{-1}) is also exceptionally high, exceeding values previously reported in known septic plumes elsewhere (up to 4 mg L^{-1};

Table 2.3). This may be explained by sewage reticulation earthworks that were carried out approximately 5 m upslope of this site in mid-2023. Robertson et al. (2019) hypothesised that excavation of drainfield sediments can redistribute phosphorus-rich soils on the surface, increasing the potential for phosphorus runoff into shallow groundwater. Pronounced DRP spikes occurring in samples collected at this site after these earthworks may support this theory, though a second large nutrient spike one year later did not coincide with any known disturbance. Given the limited dataset for this site (n = 5), the precise cause of these nutrient peaks remains uncertain, though the site may have been influenced both by earthworks disturbance and/or an unidentified faulty septic system, or another unidentified source.

Nitrogen speciation shift

Sites T3-below, T3-above, T9-below, C3-above and C3-below showed a shift from ammoniacal-N dominance in year one to nitrate-N in year two. This likely reflected changes in groundwater redox conditions following several extreme, compounding weather events in January – February 2023. During this period, the Rotorua Airport weather station recorded 336 mm of rainfall in January (+242 % of the January average) and 224 mm (+124 %) in February, raising the lake level by 0.27 m between January and June 2023 (Bay of Plenty Regional Council, 2025; National Institute of Water and Atmosphere Research [NIWA], 2023a, 2023b). The associated rise in the water table was evident from the highest piezometer sampling success rates occurring in May, June and September 2023. Prolonged saturation from a high water table can promote anaerobic sediment conditions that inhibit nitrification and lead to ammoniacal-N accumulation (Lusk et al., 2017).

In year two, lower lake levels resulted in a lower water table, allowing greater oxygen penetration into shallow sediments and creating more aerobic conditions that promoted nitrification of accumulated ammoniacal-N to nitrate-N (Robertson et al., 2019; Roumelis et al., 2025). Similar seasonal patterns were reported by Nyenje et al. (2013) for pit latrines in Kampala, Uganda, where high water tables during the wet season favoured ammonium leaching, shifting to nitrate during the dry season. As discussed, the cause of this pattern at native vegetation sites is likely natural soil mineralisation and subsequent nitrogen transformation; while at urban sites, the same trend may instead reflect the accumulation and transformation of septic effluent, although this cannot be resolved without additional source tracers.

Spatial trends

Influence of dwelling density on groundwater nutrients

Observations of mean nutrient concentrations showed that spikes in nitrate-N and ammoniacal-N generally occurred in areas where dwelling density exceeded 4 dwellings / 0.1 km^2, although no statistical test was performed to assess the significance of this pattern. Similar patterns have been observed by Iverson et al. (2018), who reported that streams draining high-density septic tank areas (18.8 systems / 0.1 km^2), exported more than twice the median amount of phosphate and TN compared with lower density areas (<10 systems / 0.1 km^2) and control catchments (no septic systems). Similarly, Hoghooghi et al. (2016) found a positive linear relationship between septic system density and stream nitrate, although this was only evident when density exceeded 7.5 systems / 0.1 km^2, and no relationship was observed between ammoniacal-N and system density. It is therefore plausible that dwelling density is also an important factor for explaining elevated nitrate-N and ammoniacal-N along the urban areas at Lake Tarawera.

It is worth highlighting that it is generally difficult to detect a strong relationship between dwelling density and groundwater nutrients as septic plumes can disperse both vertically and laterally, and are often diluted within groundwater, which can obscure the true extent of nutrient pollution from septic tanks (Baer et al., 2019). Piezometer site selection was primarily influenced by accessibility rather than housing density, with no sites being located in higher density areas (7 – 9 systems / 0.1km^2). Therefore, only an indicative association can be made between nutrient contaminants from septic systems and dwelling density in this study.

Upslope vs. downslope nutrient patterns

Due to the inconsistent presence of samples in paired upslope and downslope sites and high variation in contaminant concentrations, statistical tests could not be conducted between upslope and downslope sites. It was expected that sites located downslope of septic systems would show some degree of nutrient enrichment. This has been observed by Withers et al. (2011), who monitored ditch water upslope and downslope of five dwellings on septic systems in Loddington, UK. The authors found that the downslope site exhibited higher mean concentrations of ammoniacal-N (0.99 mg L^{-1}), DRP (0.3 mg L^{-1}), nitrate-N (6.7 mg L^{-1}), and nitrite-N (0.16 mg L^{-1}), compared to the upslope site (0.05, 0.078, 3.9, and 0.02 mg L^{-1}, respectively). At Lake Tarawera, mean DRP and nitrate-N concentrations were elevated at T3-below, T4-below, and T7-below relative to their paired upslope sites, with ammoniacal-N also relatively elevated at T7-below. Despite sampling limitations and high variability, the higher nutrient concentrations at select downslope sites is consistent with documented septic system impacts, suggesting that septic effluent may be the main source of localised nutrient enrichment of shallow groundwater within the urban strip.

Temporal trends

Mean nitrate-N and DRP were notably higher in the second year of monitoring (April 2024 – March 2025), but no clear seasonal trend emerged for downslope sites throughout the two years of the study. Some seasonal variability was expected due to the seasonal use of septic systems by holiday homeowners; however, shallow groundwater nutrient concentrations are often more strongly influenced by rainfall patterns and water table fluctuations than distinct seasons (Roumelis et al., 2025). For example, Narancic et al. (2020) reported that small tributaries of Lake St. Charles, Canada, draining areas with dense septic systems, exhibited significantly higher nutrient loading during autumn when rainfall was high. Total nitrogen increased from 134 kg/day in summer to 3438 kg/day in autumn, and TP increased from 10 kg/day to 285 kg/day. In our study, the extreme rainfall events of January – February 2023 likely disrupted typical seasonal processes, with groundwater nutrients still recovering from these disturbances, and may have masked seasonal trends through dilution during wetter periods (Nyenje et al., 2013).

Limitations

While this study provides valuable insights into the extent of nutrient contamination in shallow groundwater from septic tanks at Lake Tarawera, several limitations should be acknowledged. The dataset is constrained by infrequent sampling, particularly during periods of low rainfall, which may have missed transient spikes in nitrate-N under aerobic groundwater conditions. Extreme rainfall events also obscured true temporal trends by causing rapid changes in water table levels, influencing nitrogen speciation and nutrient transport rates. Piezometer placement was primarily determined by site availability, resulting in some piezometer pairs not lying on the same groundwater flow path and some downslope piezometers being located beyond potential septic plume areas. Additionally, the study relied solely on nutrient concentrations to determine sources, without the use of chemical tracers such as chlorine and sucralose, or isotopic analyses, which could more effectively distinguish septic contamination from other inputs (Hoghooghi et al., 2016). Limited knowledge of underlying sediment geology and natural groundwater characteristics further complicated efforts to disentangle septic sources from natural inputs. While these results provide preliminary insights into groundwater nutrient dynamics at Lake Tarawera, the observed trends should be interpreted cautiously given these significant limitations and the absence of statistical analyses.

Conclusion

This study identified several localised areas of shallow groundwater nutrient enrichment within the urban zone of Lake Tarawera, where concentrations were consistent with those observed in other Te Arawa lakes affected by septic contamination and in comparable systems internationally. Most sites, however, exhibited nutrient concentrations within typical background concentrations for New Zealand shallow groundwater, indicating that septic influence is not widespread throughout the urban zone. Temporal patterns were more strongly associated with extreme rainfall and water table fluctuations than with seasonal septic tank usage, while spatial trends were tentatively associated with proximity to dwellings and areas of higher dwelling density.

Future research should target lower-elevation sites or extend piezometers deeper below the surface to increase the likelihood of obtaining successful groundwater samples. For volcanic lakes such as Tarawera, where the water table is typically deep due to free-draining soils, pore water sampling near the lake edge may be more suitable for monitoring shallow groundwater nutrient inputs. Additionally, analysing sediment composition at each site could help distinguish septic system impacts from other anthropogenic and natural sources along the urban strip. Implementing these approaches may improve understanding of the risks posed by anthropogenic impacts to both lake water quality and ecosystem health in Lake Tarawera and similar lakes.

Chapter 3

Investigating the shallow groundwater–littoral nutrient pathway along the urban shoreline of Lake Tarawera

Introduction

Globally, lakes are experiencing increased algal blooms and altered vascular plant productivity in the littoral zone as a result of increasing nutrient inputs from anthropogenic land uses (Page et al., 2022). The nearshore littoral zone is the primary interface between catchment and lake ecosystems and is often where the first response to external nutrient enrichment is observed (Vadeboncoeur et al., 2021). In residential catchments, septic systems are a major potential contributor of nutrients to shallow groundwater (Aravena et al., 1993). Effluent-derived contaminants can be transported rapidly along shallow flow paths and typically discharge to lakes within the first 50 m of the shoreline (Cui et al., 2023; Lisboa et al., 2024). However, the influence of septic inputs on the littoral zone is often spatially heterogeneous due to variations in groundwater flow rates, underlying aquifer structure, nutrient loading rates, and differences in hydraulic connectivity, complicating attribution of nutrient contamination to septic sources (Baer et al., 2019).

Stable isotopes of nitrogen ($\delta^{15}N$) and oxygen ($\delta^{18}O$) present in nitrate molecules have been widely used to distinguish between various nitrate sources in groundwater (Kendall et al., 2007; Rogers et al., 2023). Nitrate derived from atmospheric deposition, soil nitrogen, fertilisers, and wastewater typically exhibits distinct ranges of $\delta^{15}N$ and $\delta^{18}O$, allowing isotopic signatures in groundwater to be compared with known source values (Jung et al., 2020). Septic effluent is naturally enriched in nitrate $\delta^{15}N$ relative to other nitrate sources due to volatilisation of ammonia, which preferentially removes ^{14}N and enriches the remaining nitrogen pool in ^{15}N (Su et al., 2023). As a result, elevated nitrate $\delta^{15}N$ in groundwater is often interpreted as evidence of septic contamination (Aravena et al., 1993).

However, isotopic values of septic-influenced groundwater can be substantially altered during transport because nitrification, denitrification, and other microbial processes preferentially consume lighter isotopes, leading to enrichment of the remaining nitrate pool in ^{15}N and ^{18}O and increases in $\delta^{15}N$ and $\delta^{18}O$ values (Hoghooghi et al., 2016). Additionally, evaporation and dilution from precipitation can further alter groundwater isotopic signatures (Kendall et al., 2007). The extent to which these processes occur in the subsurface can cause groundwater isotope signatures at the point of discharge to differ considerably from the original source (Rogers et al., 2023).

Primary producers in littoral zones assimilate nitrogen and carbon from both benthic and pelagic sources during photosynthesis (Bacchus & Barile, 2005). The signatures of $\delta^{15}N$ and $\delta^{13}C$ in their tissues can therefore provide insight into the isotopic composition and ratio of these sources (Li et al., 2011). However, littoral primary producers also preferentially integrate lighter nitrate isotopes when nitrate is abundant, resulting in fractionation from the original source (Bacchus & Barile, 2005). These processes must therefore be carefully considered when interpreting isotopic data, as both subsurface transformation and biological fractionation can mask the original source signature.

The previous chapter revealed that there is evidence of nutrient contamination in shallow groundwater from septic systems along the western shoreline of Lake Tarawera. However, it is unknown whether a direct groundwater connection exists between the catchment and the nearshore littoral zone, whether septic-derived nutrients are transported along this pathway, and whether these nutrients are assimilated by littoral primary producers. This chapter aimed to address these questions by: (1) comparing previous rainfall with groundwater flow measured using seepage chambers to determine hydraulic connectivity, (2) analysing pore water and groundwater nutrients and nitrate stable isotopes ($\delta^{15}N$ and $\delta^{18}O$) to detect potential septic influence, and (3) measuring $\delta^{15}N$ and $\delta^{13}C$ in suspended particulate matter (SPOM), macrophytes, and periphyton to identify potential septic-derived nitrogen assimilation.

Methods

Hydrological connectivity

Groundwater inflow to littoral zone

Ten sites in the nearshore littoral zone on the urban western shoreline of Lake Tarawera were selected for measuring shallow groundwater inflows to the lake (see Figure 3.1). Sites corresponded to subcatchment piezometer sites in Chapter 1, apart from Bayview Road (BVR) which was substituted due to inaccessibility of the littoral zone at site T9-below. Groundwater inflows were measured monthly for a 1-year period, alternating between half the sites from April 2023 to March 2024. No sampling was conducted during the summer months (December 2023 and January 2024) due to high recreational use of littoral zones by the public, potentially confounding the measurements.

Perspex chambers (V = 16 L, Ø = 400 mm) were used to capture groundwater inflows (Figure 3.2). A modified 1 L saline intravenous (IV) bag (Capes Medical, Tauranga) containing 100 mL of deionised water was first attached to the side of a chamber via a two-way valve. The chamber was then pushed into the lake sediment until firmly secured, at a water depth of 0.4 – 0.8 m. A two-way valve at the top of the chamber was used to release any trapped air bubbles before being closed, after which the side valve to the IV bag was opened. One chamber was deployed at each site and left to collect groundwater inflows for an approximately 24-hour period.

Chambers were collected the following day. The groundwater inflow was calculated as the total water volume in the bag, minus the initial 100 mL. Seepage inflow (L m^{-2} day^{-1}) was then calculated following Sebestyen and Schneider (2001) (1):

$$\mathrm{Inflow}_{\mathrm{L\,m^{-2}day^{-1}}} = \frac{\mathrm{V_f - V_0}}{\Delta\mathrm{t \cdot A}} \times \frac{24}{1000} \tag{1}$$

where V_f and V_0 are the final and initial volumes in the IV bag (mL), Δt is elapsed time of chamber deployment (decimal time), and A is chamber area (0.126 m^2).

Figure 3.1. Littoral sites for groundwater, nutrient inflow and stable isotope sampling along the western shoreline of Lake Tarawera. Note: Bayview Road (BVR) was substituted as the littoral zone was not suitable for chamber deployment at site T9.

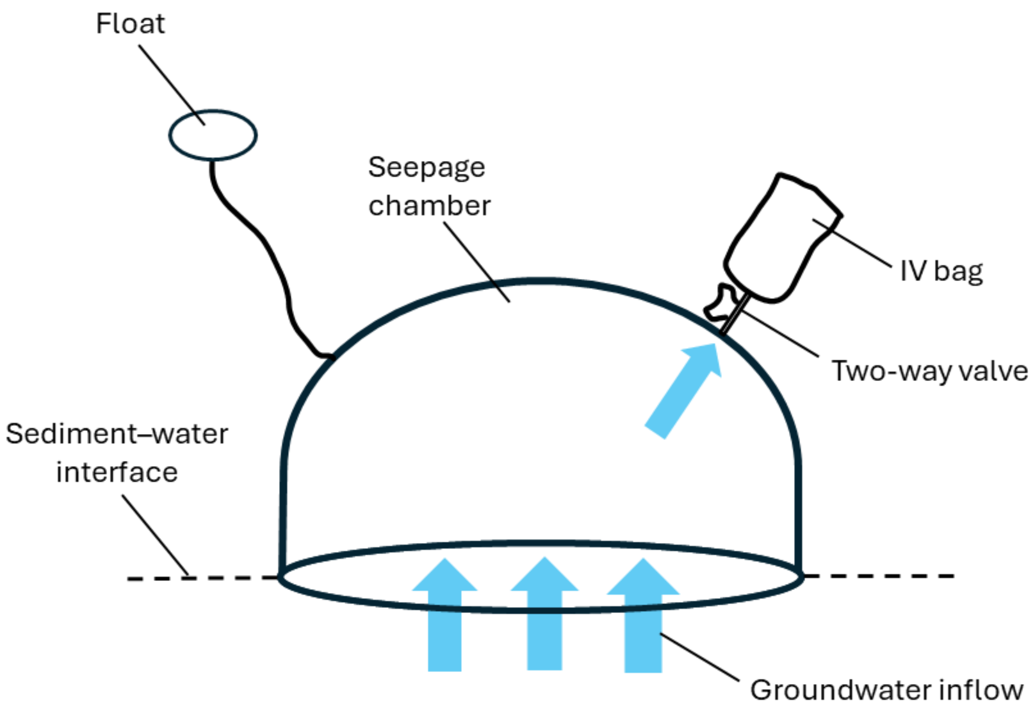

Figure 3.2. Seepage chambers used in the study. Blue arrows indicate groundwater inflow from the sediment into the chamber and the transfer of chamber water into the intravenous (IV) bag for infiltration rate calculations.

Previous rainfall

As no climate station is located at Lake Tarawera, data from the Virtual Climate Station Network (VCSN) maintained by NIWA (Earth Sciences New Zealand from 2025) was used. Data was obtained from Virtual Climate Station 27368, situated on the northern shore of Lake Tarawera. Rainfall estimates are generated using a thin-plate smoothing spline model for spatial interpolation (Earth Sciences New Zealand, 2025). This model incorporates latitude and longitude, as well as a third “pattern” variable (the 1951 – 1980 mean annual rainfall, digitised from an expert-guided contour map) to improve interpolation accuracy.

Previous rainfall was calculated from daily rainfall data as the cumulative one-month (28 day) total prior to each chamber deployment date. The previous period began on the day before chamber deployment, which excluded rainfall that fell during the 24-hour chamber deployment.

Data analysis

Spearman rank correlation analyses were used to assess the relationship between various periods of previous rainfall (24-hour, 48-hour, 72-hour, 5-day, 7-day, 2-week, 3-week and 1-month) and groundwater inflow.

Stable isotopes and nutrients

Sample collection

Macrophytes, periphyton, pore water, and suspended particulate organic matter (SPOM) samples were collected from 10 sites in the littoral zone along the western shore of Lake Tarawera (see Figure 3.1). Macrophytes were collected by random hand sampling along approximately 10 m of the shallow littoral zone (<1 m deep) on 15th October 2024. Algae plates were deployed on the same date for one month at ~1 m above the lakebed to allow sufficient biomass accumulation, after which the plates were removed on 28th November. Plates and macrophytes were stored on ice and transported to the lab, where periphyton was then scraped from the plate surfaces.

Pore water was extracted from the sediment in the littoral zone (<1 m deep) using a syringe filter on 28th November 2024. A 10 cm tube was attached to the syringe nozzle and inserted into the sediment to minimise the inclusion of lake water. Suspended particulate organic matter samples were collected by filtering up to 2 L of lake water (collected from ~20 cm below the surface) through Whatman GF/C glass fibre filter paper. Additionally, groundwater from five piezometers and the artesian spring collected on 30th October were included in the analysis as a comparison for pore water. All samples were stored on ice during transport to the laboratory and processed on the same day.

Processing

Macrophyte, periphyton, and SPOM filter samples were oven dried at 60°C for one week. Dried macrophyte and periphyton tissue was ground to a fine powder using a mortar and pestle, which was rinsed with ethanol between samples. Approximately 10 mg of each ground sample was stored in plastic containers. Suspended particulate organic matter filters were stored in sealed ziplock bags. All organic material samples were kept in a cool, dark room until analysis.

Piezometer and pore water samples were filtered through 0.45 µm Minisart filters (Sartorius, Germany) and split into two subsamples. Approximately 20 mL was stored frozen at -18 °C for nutrient analysis. A small amount of the second subsample was used to rinse a 100 mL bottle three times before the remaining sample was transferred to it. To preserve the isotope composition of the sample and remove nitrite, 1 mL of 10 % HCl plus sulfanilic acid was added to the sample, and the pH was confirmed to be ≤ 3 using pH paper.

Nutrient analysis

Pore water and groundwater samples were analysed by Hill Laboratories (Hamilton) for ammoniacal-N, nitrite-N, nitrate-N, and dissolved reactive phosphorus (DRP) using a flow injection analyser. Ammoniacal-N concentrations were determined using the phenol/hypochlorite colorimetry method (APHA 4500-NH3H modified), with a detection limit of 0.010 mg L^{-1}. Nitrite-N concentrations were determined using the automated azo dye colorimetry method (APHA 4500-NO3- modified), with a detection limit of 0.002 mg L^{-1}. Nitrate-N was calculated by first using the automated cadmium reduction method (APHA 4500 NO3- modified; detection limit of 0.002 mg L^{-1}) to determine nitrate-N + nitrite-N, then the nitrite-N concentration was deducted from this to give nitrate-N concentration, with a detection limit of 0.001 mg L^{-1}. Dissolved reactive phosphorus was determined using the molybdenum blue colorimetry method (APHA 4500-P modified) with a detection limit of 0.004 mg L^{-1}.

Stable isotope analysis

All stable isotope analyses were performed at GNS Science (Lower Hutt). Macrophyte, periphyton, and SPOM samples were analysed by combustion on a Eurovector elemental analyser coupled to an Isoprime isotope ratio mass spectrometer (IRMS). Isotopic values for $\delta^{13}C$ and $\delta^{15}N$ were reported relative to Vienna Pee Dee Belemnite (VPDB) and ambient inhaled air (AIR), respectively, and normalised to GNS's internal Leucine standard (−28.3 ‰ for $\delta^{13}C$, 6.5 ‰ for $\delta^{15}N$). Analytical precision was ±0.3 ‰ for $\delta^{15}N$ and ±0.2 ‰ for $\delta^{13}C$.

For water samples, nitrate was first reduced to nitrite using cadmium and then converted to nitrous oxide using sodium azide in an acetic acid buffer. Nitrous oxide was purged from the water sample, and water and carbon dioxide were removed through a series of chemical traps. The nitrous oxide was cryogenically trapped in liquid nitrogen, cryofocused in a second trap, and then introduced to a gas chromatography (GC) column before entering an Isoprime IRMS for isotopic analysis of nitrogen and oxygen. Results were reported relative to AIR ($\delta^{15}N$) and Vienna Standard Mean Ocean Water (VSMOW; $\delta^{18}O$) and normalised to the international standards USGS-34 (−1.8 ‰ for $\delta^{15}N$, −27.9 ‰ for $\delta^{18}O$), IAEA-NO_3 (4.7 ‰ for $\delta^{15}N$, 25.6 ‰ for $\delta^{18}O$), and the GNS internal standard KNO_3-b (10.7 ‰ for $\delta^{15}N$, 11.7 ‰ for $\delta^{18}O$). Analytical precision was ±0.3 ‰ for $\delta^{15}N$ and $\delta^{18}O$, except for samples below 100 µg -N L^{-1}, which may have had lower precision.

Data analyses

Isotope biplots for $\delta^{15}N$ / $\delta^{18}O$ and $\delta^{15}N/\delta^{13}C$ were generated to visualise the isotope ratios for different samples. Spearman rank correlation (ρ) was used to assess the relationship between isotopes for each plot, in order to evaluate whether isotopic enrichment in one tracer corresponded with enrichment in another. Due to limited replication of stable isotope and nutrient concentration samples at each site, formal statistical testing was not undertaken, and results are therefore presented descriptively.

Results

Hydrological connectivity

Groundwater inflow

Over the 1-year groundwater seepage study period, a total of 50 chamber deployments were undertaken, with 37 successfully measuring groundwater inflows. Failed chamber deployments were primarily due to wave action, either uplifting the chambers from the sediment bed or detachment of the sample bag off the chamber valve. The highest mean (± standard deviation) groundwater inflow rate was 4.65 ± 3.54 L m^{-2} day^{-1} at T6, and the lowest was 1.22 ± 0.76 L m^{-2} day^{-1} at T4 (Figure 3.3; Appendix B - Table B1). The overall mean groundwater inflow rate across the entire study was 2.32 ± 1.77 L m^{-2} day^{-1}. The interquartile ranges (IQRs) for C1, T2 and T6 were notably higher than those of other sites, spanning 2.8 – 6.4 L m^{-2} day^{-1}, while the IQRs of all other sites ranged from 0.5 – 2.8 L m^{-2} day^{-1}.

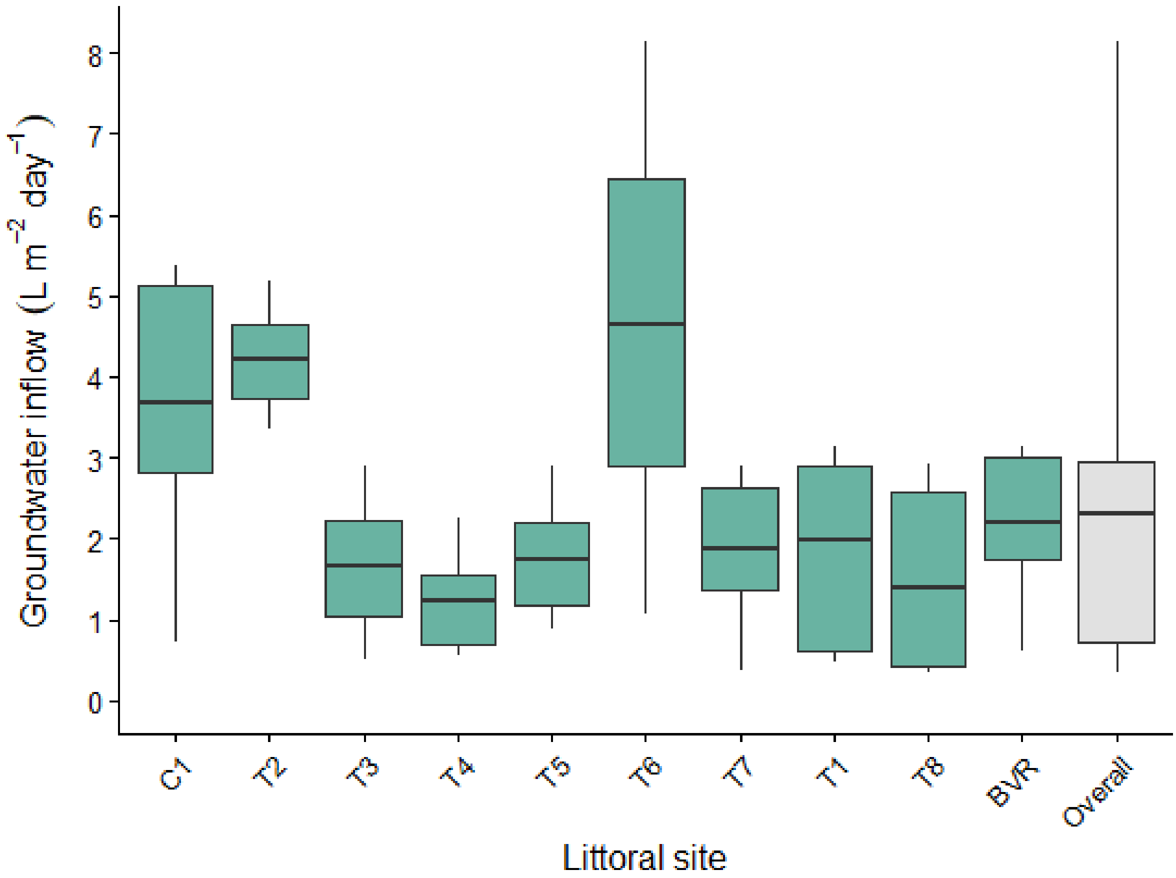

Figure 3.3. Boxplots showing groundwater inflow at each littoral site. Boxes represent the interquartile range, horizontal black lines show mean values, and whiskers are the minimum and maximum values. The grey box indicates nutrients across all sites.

Previous rainfall

Virtual Climate Station 27368 daily rainfall rates ranged from 0.0 to 114.7 mm from 1 January 2023 to 30 March 2024 (Appendix B - Figure B1). The maximum daily rainfall rate occurred on 28th January 2023. The mean rates for previous rainfall periods prior to sampling dates ranged from 5.6 mm for 24-hour previous rainfall to 184.6 mm for 1-month previous rainfall (Appendix B - Table B2).

Rainfall and groundwater inflow correlation

Significant positive correlations were found between groundwater inflow and 3-week previous rainfall ($\rho = 0.43$, $p = 0.007$), as well as 1-month previous rainfall ($\rho = 0.48$, $p = 0.003$; Figure 3.4).

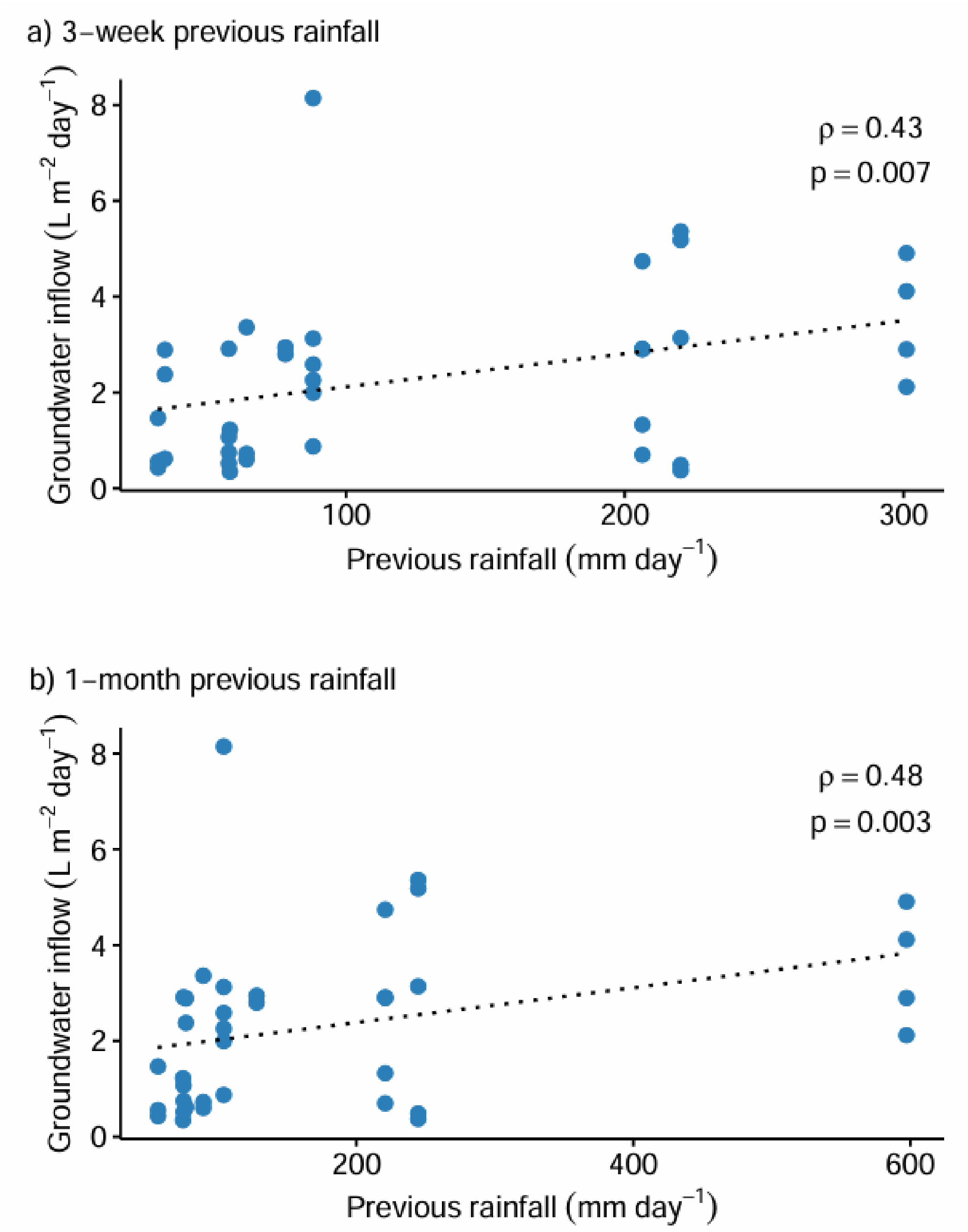

Figure 3.4. Relationships between a) 3-week previous rainfall *and* b) 1-month previous rainfall and groundwater inflow into the littoral zone. Note the different x-axis scales between panels. Spearman rank correlations (ρ) are shown, as well as significant p-values (p <0.05). Dotted black lines represent linear fits for visualisation only. The previous rainfall period begins on the day before seepage chambers were deployed for measuring groundwater inflow, with rainfall data sourced from the virtual climate station 27368, operated by Earth Sciences New Zealand. One month previous rainfall represents a four-week accumulation period.

Subsurface nutrient concentrations

Pore water

Pore water nitrate-N concentrations ranged from <0.002 to 1.84 mg L^{-1} with the highest concentration recorded at site T5 (Figure 3.5; Appendix B - Table B3). Ammoniacal-N concentrations ranged from 0.02 mg L^{-1} at C1 to 0.51 mg L^{-1} at T1. Nitrite-N concentrations were below the detection limit (<0.002 mg L^{-1}) at 50 % of the sites, with all sites measuring <0.005 mg L^{-1}. DRP concentrations ranged from 0.007 to 0.33 mg L^{-1}, with the maximum concentration observed at T1, similar to ammoniacal-N.

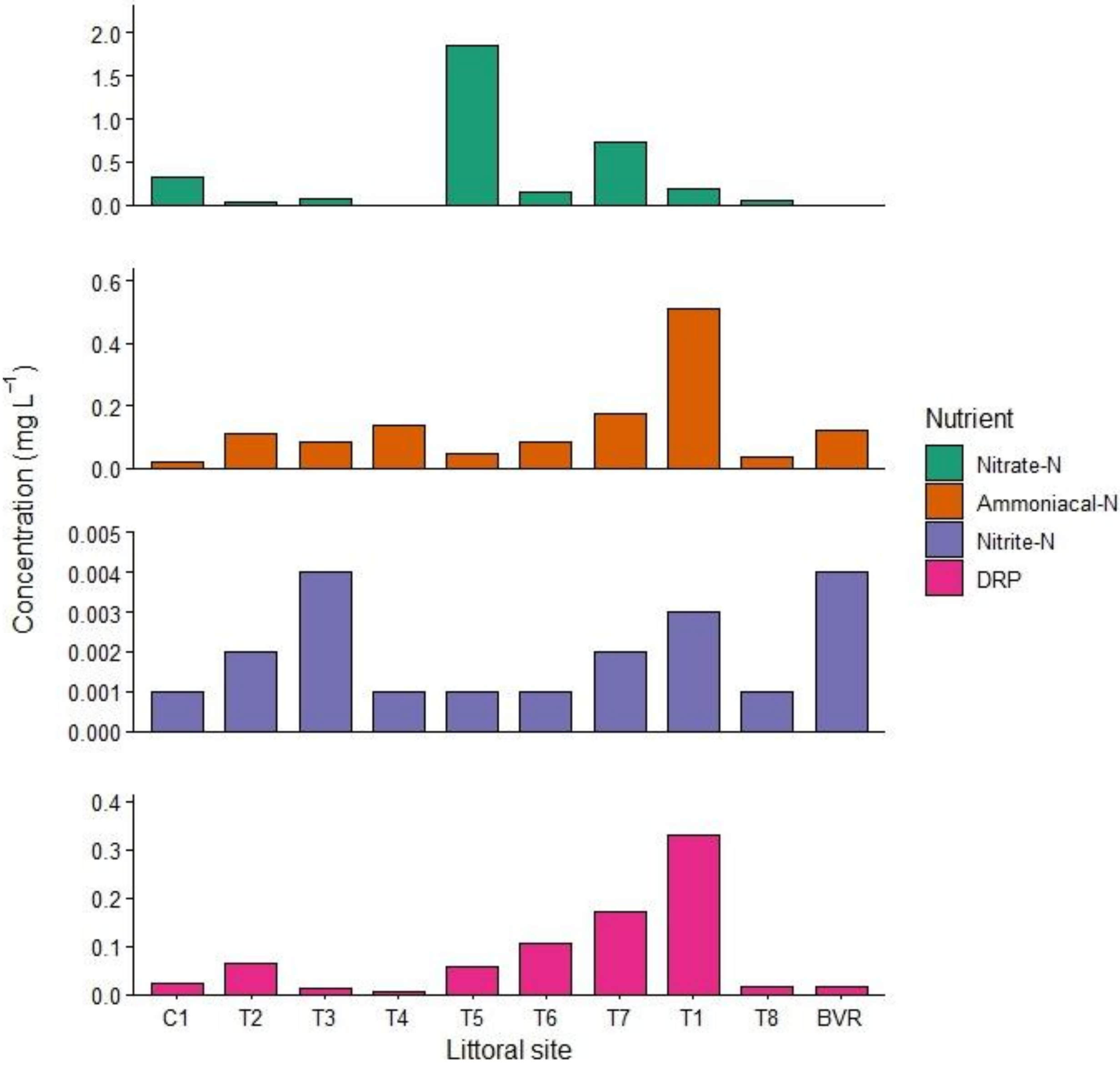

Figure 3.5. Pore water nutrient concentrations at each littoral site sampled on 28 November 2024. *DRP* is dissolved reactive phosphorus. Where concentrations were below the detection limit for nitrate-N and nitrite-N (<0.002 mg L^{-1}), concentrations are presented as half the detection limit (0.001 mg L^{-1}).

Groundwater

Groundwater was present in five piezometers and the artesian spring at the time pore water sampling was conducted. Three piezometers were located upslope of corresponding littoral sites (T3-below, T4-below, and T5-below). Nitrate-N concentrations ranged from 0.076 to 1.48 mg L^{-1}, while ammoniacal-N concentrations ranged from 0.01 to 28 mg L^{-1} (Appendix B - Figure B2; Table B3). Nitrite-N concentrations were ≤ 0.005 mg L^{-1} at all sites, and DRP concentrations ranged from 0.01 to 0.21 mg L^{-1}. The maximum nitrate-N and DRP concentrations were recorded at T4-below, while the maximum ammoniacal-N and nitrite-N concentrations were observed at T5-above.

Stable isotope tracing of nutrient sources

Water samples

In pore water, $\delta^{15}N$ values ranged from 2.48 ‰ at T1 to 11.90 ‰ at T4, with a mean (± standard deviation) of 6.51 ± 3.25 ‰ (Figure 3.6; Appendix B - Table B3). $\delta^{18}O$ values in pore water ranged from -4.07 ‰ at T6 to 9.6 ‰ at T4, with a mean of 1.06 ± 4.37 ‰. Groundwater samples exhibited similar stable isotope ranges, with $\delta^{15}N$ values ranging from 2.03 to 10.88 ‰ and $\delta^{18}O$ ranging from -3.44 to 8.62 ‰. A significant positive correlation was observed between $\delta^{15}N$ and $\delta^{18}O$ values for water samples (both pore water ($\rho = 0.77$, $p < 0.001$).

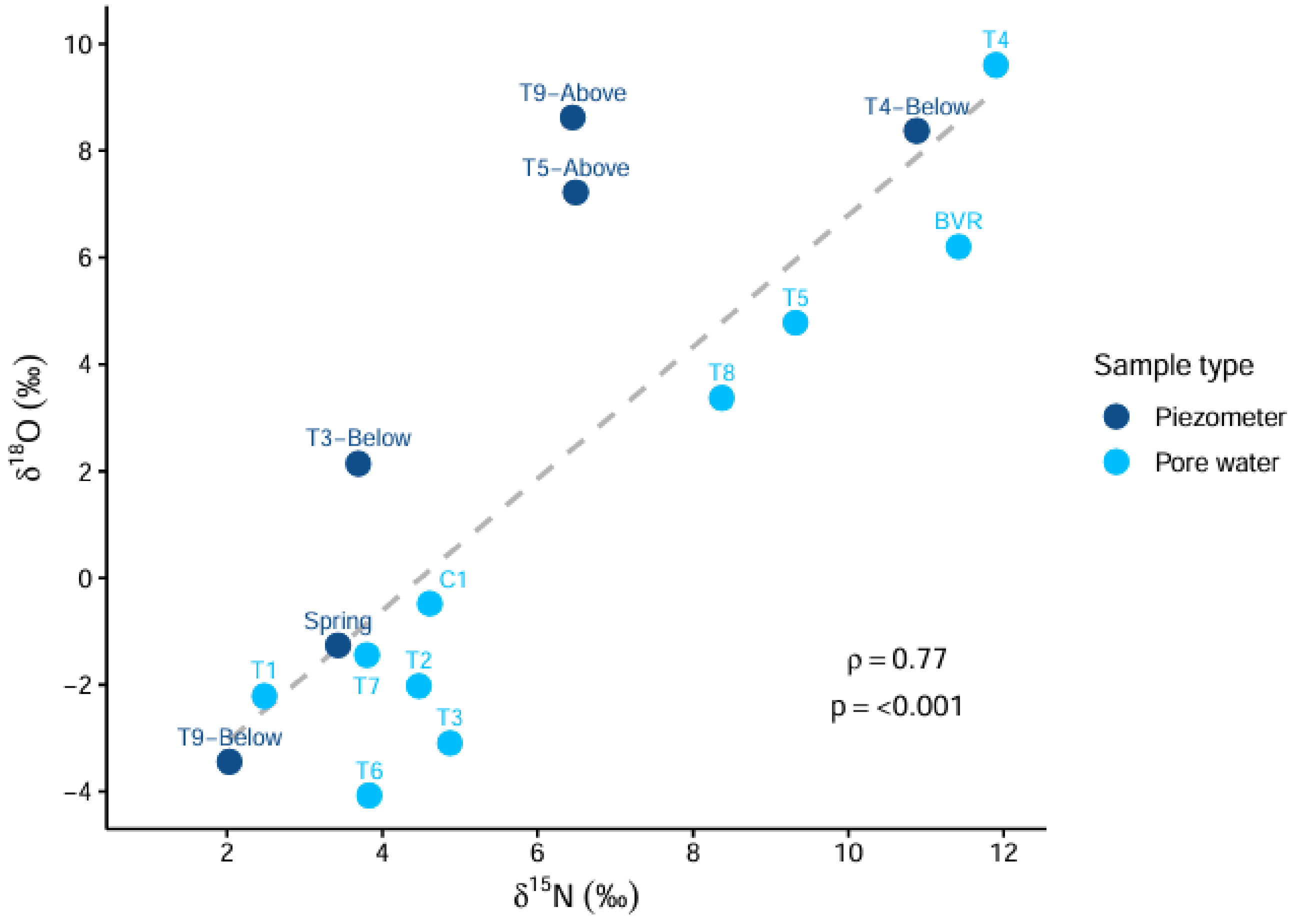

Figure 3.6. $\delta^{15}N$ and $\delta^{18}O$ stable isotope biplot for pore water (light blue) and groundwater (dark blue) samples. Labels above each point indicate the sampling location. A single regression line is fitted across all samples to illustrate the overall relationship, with Spearman's ρ and associated p-value shown on the plot.

Organic material

$\delta^{15}N$ values for SPOM, periphyton, and macrophytes fell within a similar range (Figure 3.7; Appendix B - Table B4). In contrast, $\delta^{13}C$ signatures in SPOM were distinctly depleted compared to macrophytes and periphyton. Suspended particulate organic matter $\delta^{15}N$ values ranged 0.40 – 3.06 ‰ and $\delta^{13}C$ from -30.55 to -19.61 ‰. Periphyton and macrophytes exhibited a wider range of values, with $\delta^{15}N$ values and $\delta^{13}C$ measuring 0.18 – 2.83 ‰ and -28.17 to -12.48 ‰ for macrophytes, while periphyton ranged from 0.34 – 2.54 ‰ for $\delta^{15}N$ and -26.29 to -5.89 ‰ for $\delta^{13}C$.

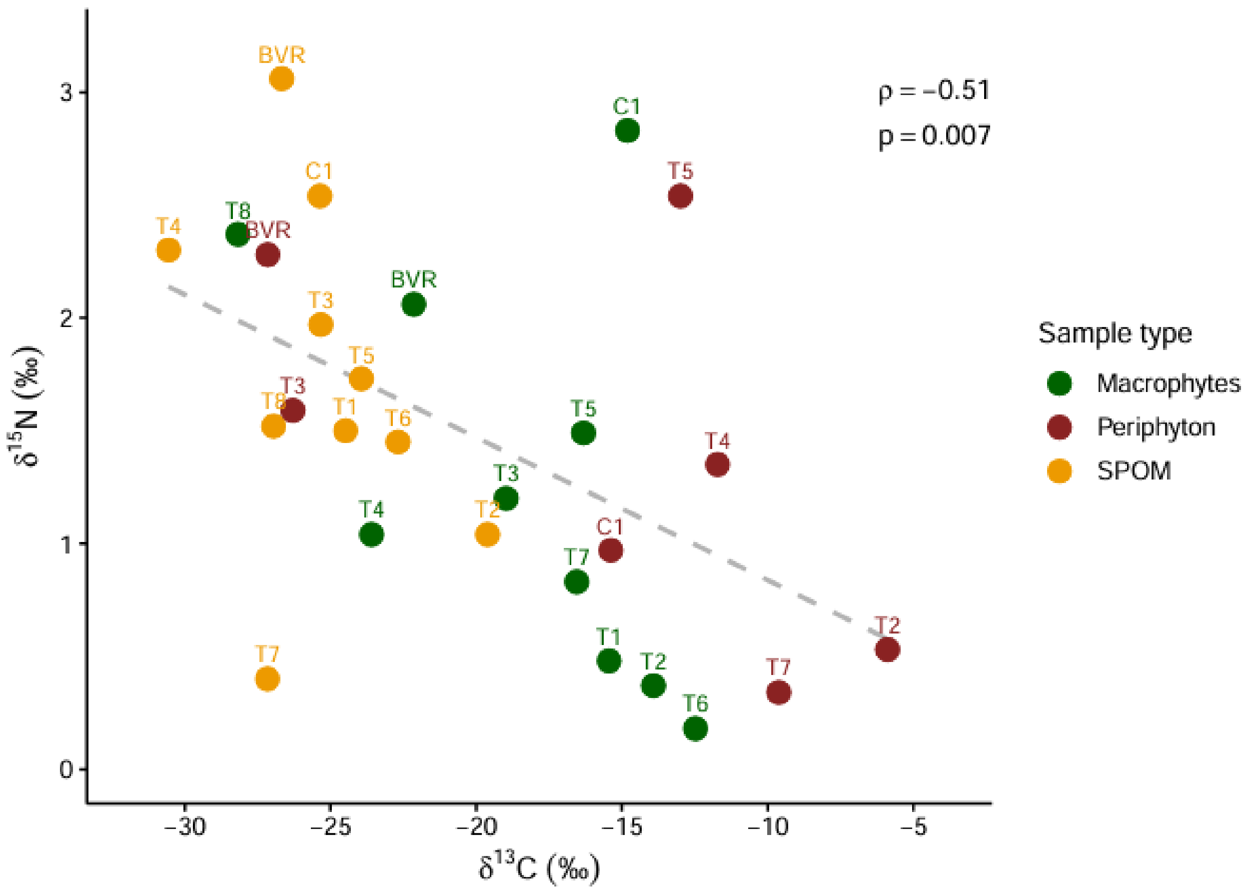

Figure 3.7. $\delta^{15}N$ *and* $\delta^{13}C$ stable isotope biplot for periphyton, macrophyte*,* and suspended particulate organic matter (SPOM) samples. Labels above each point indicate the sampling location. A single regression line is fitted across all samples to illustrate the overall relationship, with Spearman's ρ and associated p-value shown on the plot.

Discussion

Groundwater inflow to the nearshore littoral zone

Groundwater inflow rates ranged from 0.35 to 8.15 L m^{-2} day^{-1}, with a mean of 2.32 L m^{-2} day^{-1} across the 10 littoral sites. The broad range in seepage rates is not unexpected, as groundwater inflow is influenced by seasonal rainfall trends, localised rainfall events and patterns, and geomorphological characteristics of the watershed (Lisboa et al., 2024; Sebestyen & Schneider, 2001). In comparison, John and Lock (1977) deployed seepage chambers in Lake Rotorua and later in Lake Taupō (Lock & John, 1978) at depths of 0.2 – 1.4 m, recording inflows of 2.7 – 127.5 L m^{-2} day^{-1} in Rotorua and 1.4 –

475 L m^{-2} day^{-1} in Taupō. Their results showed that groundwater inflow decreased approximately exponentially with distance offshore, a pattern corroborated by subsequent studies (Attanayake & Waller, 1988; Hector, 2004; Kazmierczak et al., 2016). It is suggested that low groundwater inflow at one site at Lake Rotorua was due to reduced surface permeability of the town foreshore, and the occurrence of either recent Rotomahana mud or more permeable yellow-brown pumice soil affecting inflows at other sites. At Lake Tarawera, similar contrasts in catchment land uses and substrate composition likely influenced surface and subsurface permeability and consequently groundwater inflows.

Comparable studies overseas have also reported a broad range of inflows across diverse lake types, spanning approximately -14.6 to 630 L m^{-2} day^{-1} in sandy lakes in Denmark, glacial and seepage lakes in the USA, and glacial lakes in Canada, including a fractured basin lake in Nova Scotia (Attanayake & Waller, 1988; Sebestyen & Schneider, 2001; Kidmose et al., 2011; Kazmierczak et al., 2016; Lisboa et al., 2024). Some lakes exhibit negative groundwater flux (aquifer recharge) under low precipitation conditions; for instance, Lower Sylvan Pond in the Adirondack Mountains, USA, recorded groundwater inflows ranging from -12 to 9.6 L m^{-2} day^{-1} during a single summer (Sebestyen & Schneider, 2001). Collectively, these studies highlight that groundwater exchange within littoral zones is highly heterogeneous through both space and time, driven by catchment characteristics and climate variability. For Lake Tarawera, this suggests that nutrient delivery via groundwater is also likely to vary spatially along the urban shoreline, resulting in localised contaminant hotspots that may be exacerbated by the fractured and highly porous nature of the underlying volcanic substrate in the catchment.

Rainfall effect

Cumulative 3-week ($\rho = 0.43$, $p = 0.007$) and 1-month ($\rho = 0.48$, $p = 0.003$) previous rainfall were positively correlated with groundwater inflow, suggesting that groundwater in the nearshore littoral zone is recharged by localised precipitation within the inner catchment. Similar relationships between antecedent rainfall and groundwater inflow are commonly reported, although the relevant rainfall periods differ among systems (Downing & Peterka, 1978; Sebestyen & Schneider, 2001). For example, seepage flux in a glacial lake in New York State was most strongly correlated with 7-day previous rainfall (p <0.001), whereas in a North Dakota lake, mean daily rainfall was significantly correlated with groundwater inflow (p <0.001) despite large variations in catchment soils and the rainfall gauge being located 22 km from the lake (Downing & Peterka, 1978; Sebestyen & Schneider, 2001). It is possible that the nearshore littoral zone at Lake Tarawera responds more rapidly to rainfall than our sampling frequency could capture, as other sandy-bottomed lakes have experienced far quicker increases in seepage rates within minutes to hours following rainfall (Rosenberry et al., 2013). The relationship with these longer periods of previous rainfall could reflect recharge originating from higher in the catchment that typically takes longer to travel towards the lake (Lisboa et al., 2024). Nonetheless, these results suggest that precipitation that falls within the catchment will eventually discharge in the nearshore littoral zone. This implies that nutrients discharged from septic tanks in the inner catchment will likely also be transported along this groundwater pathway.

Pore water nutrients

The highest nutrient concentrations measured in pore water (1.84 mg L^{-1} nitrate-N, 0.51 mg L^{-1} ammoniacal-N and 0.33 mg L^{-1} DRP) are comparable to concentrations reported in peri-urban and rural bays of Lake Taupō (Gibbs et al., 2005; Hector, 2004). In particular, nitrate-N and DRP values fall within the range observed in beach sediment 5 m from the lake edge and at the 3 m depth contour, indicating that pore water nutrients in Lake Tarawera are consistent with concentrations measured in nearshore sediments of similar lake environments in New Zealand.

Nutrient concentrations in the current study also fell within the observed range reported for littoral pore water in other oligo-mesotrophic seepage lakes with small residential communities worldwide for both ammoniacal-N (<0.01 – 1.46 mg L^{-1}) and DRP (<0.01 – 2.22 mg L^{-1}), while exceeding the range for nitrate-N (<0.01 – 0.12 mg L^{-1}; Hagerthey & Kerfoot, 1998; Lisboa et al., 2024; Naranjo et al., 2019; Périllon et al., 2017). This exceedance is likely caused by differences in sediment redox conditions altering nitrogen speciation, which has been discussed in depth in Chapter 2 (Lusk et al., 2017).

It appears that nutrient concentrations at the sediment-water interface are elevated due to the urban catchment land use, but the overall contamination extent is not excessive for a small urban community.

Contrary to expectations, the paired piezometer–pore water site T4 did not show enriched nitrate-N in pore water (<0.002 mg L^{-1}) despite a high concentration being measured in groundwater (1.48 mg L^{-1}) approximately 15 m upslope at the time of sampling. Pore water nutrient concentrations may not reflect contamination in the catchment due to a range of biogeochemical and physical processes occurring, such as clogging at the sediment-water interface, microbial transformations, redox reactions, and spatially variable groundwater pathways (Naranjo et al., 2019; Chen et al., 2025). These processes can either attenuate or release nitrate-N at various rates, complicating direct attribution of pore water enrichment to catchment land use. Additionally, nutrient contaminants may be entering the lake at deeper points, which has been observed at Lake Taupō where a 'band' of groundwater seepage was discovered at a depth of 6.5 m (Gibbs et al., 2005). As a result of the sampling limitations, pore water nutrient concentrations need to be interpreted with caution and are not used as a primary line of evidence for septic-derived contamination in this study.

Future research suggestion

Syringe-filtering pore water from the sediment posed several challenges, including difficulty consistently sampling from the same 10 cm depth in compacted sediment, the risk of drawing in overlying lake water, and the inherent limitation that a single pore water sample cannot quantify nutrient flux to the lake. A more appropriate measure of nutrient loading would be to directly capture flux to the littoral zone, which we initially attempted using seepage chambers. However, the large chamber volume (16 L) likely diluted inflowing groundwater nutrients with ambient lake water, resulting in nutrient concentrations below detection limits. Extending deployment times would reduce dilution and provide a more accurate measure of nutrient flux than single-point pore water samples.

Stable isotopes

Water samples

Overall patterns

Pore water and groundwater samples exhibited overlapping ranges of $\delta^{15}N$ and $\delta^{18}O$ values, with $\delta^{15}N$ spanning 2.03 – 11.90 ‰ and $\delta^{18}O$ spanning -4.07 – 9.6 ‰. These values fall within the ranges reported in a nationwide study of nitrate stable isotopes in groundwater throughout New Zealand ($\delta^{15}N$ = -11.6 – 42.5 ‰, mean = 6.8 ‰; $\delta^{18}O$ = -9.8 – 31.8 ‰, mean = 3.6 ‰), where samples represent a wide range of land uses, depths, and biogeochemical settings (Rogers et al., 2023). Although $\delta^{15}N$ values >4 ‰ are generally interpreted as reflecting wastewater influence in the literature, this range overlaps with typical ranges associated with signatures of natural soil nitrogen (4 – 8 ‰; Kendall et al., 2007). A distinct group of several pore water and groundwater sites exhibited $\delta^{15}N$ values >6 ‰, similar to the values reported for septic plumes in shallow groundwater in Ontario, Canada ($\delta^{15}N$ = 6.3 – 13.9 ‰), suggesting that these sites are similarly influenced by septic inputs (Aravena et al., 1993; Baer et al., 2019). These results emphasise the heterogeneous nature of wastewater impact along the urbanised western margin of Lake Tarawera, likely reflecting similarly heterogenous wastewater contamination occurring within the inner catchment.

Caveats in $\delta^{15}N$-based source attribution

The direct attribution of elevated $\delta^{15}N$ values to wastewater remains challenging because isotopic signatures are modified by both subsurface source mixing and microbial transformations, which can substantially fractionate stable isotope ratios between the source and point of sampling (Chen et al., 2025). During nitrification of septic wastewater, microbes preferentially utilise the lighter isotopes of ammonium (^{14}N and ^{16}O), enriching the residual pool with heavier isotopes (Kendall et al., 2007). Ammonia volatilisation can further elevate $\delta^{15}N$ in the remaining ammonium (Hoghooghi et al., 2016). Denitrification also fractionates nitrate, enriching the residual pool in both $\delta^{15}N$ and $\delta^{18}O$ as lighter isotopes are reduced to N_2 (Cui et al., 2023; Jung et al., 2020). Mixing between septic effluent and precipitation-recharged groundwater further obscures nitrate source distinctions (Kendall et al., 2007). For these reasons, it is important to cautiously interpret $\delta^{15}N$ values when tracing nitrate sources to lakes.

Significant positive correlations between $\delta^{15}N$ and $\delta^{18}O$ were observed in both groundwater (p = 0.03) and pore water (p <0.001), which is characteristic of denitrification occurring in the subsurface (Cui et al., 2023). Although denitrification typically enriches nitrate $\delta^{15}N$ and $\delta^{18}O$ in a ratio between 1:1 – 2:1, regression slopes in the present study exceeded 1:1, and $\delta^{18}O$ values at several sites were higher than those expected within septic plumes ($\delta^{18}O$ = 0.1 – 5.4 ‰; Aravena et al., 1993; Jung et al., 2020; Cui et al., 2023). This indicates that denitrification alone cannot explain the observed $\delta^{18}O$ enrichment. During nitrification, two oxygen atoms are incorporated from water and one from atmospheric O_2 (typically $\delta^{18}O$ = +23.5 ‰), producing nitrate with elevated $\delta^{18}O$ (Rogers et al., 2023). As a result, it is more likely that nitrification contributed to the enriched $\delta^{18}O$ values measured in pore water and groundwater. Collectively, this evidence of microbial processes indicates that nitrate undergoes substantial transformation along shallow subsurface flow paths before reaching the littoral zone, weakening our ability to use isotopes for source attribution. Given the concurrent biogeochemical processes occurring and the limitations of the isotope dataset, enriched nitrate signatures at the sediment-water interface cannot be attributed to septic inputs with confidence, although the observed trends suggest that wastewater is likely entering the littoral zone.

Future research suggestions

Confidence in results could be improved by measuring additional geochemical tracers in pore water commonly associated with septic waste. These include major ions (e.g. chloride, sulphate, metals) and artificial sweeteners such as sucralose and acesulfame, which are highly persistent and widely used as indicators of septic influence (Baer et al., 2019; Chen et al., 2025). Incorporating a temporal component into any future study would further improve the ability to capture the true extent of septic contamination along the western shoreline, as groundwater inflow, nutrient concentrations, and stable isotope composition of catchment sources vary seasonally with precipitation patterns and hydrological conditions (Naranjo et al., 2019; Wang et al., 2025).

Periphyton, macrophyte and SPOM samples

Overall patterns

The range of $\delta^{15}N$ and $\delta^{13}C$ values measured for SPOM were similar to those of macrophytes and periphyton (see Table 3.1). These $\delta^{15}N$ ranges were comparable to those reported for oligotrophic Lake Wanaka, where SPOM, native and invasive macrophytes, and periphyton showed similar patterns, although $\delta^{13}C$ values in the present study were notably more variable across all producer groups (Table 3.1; Kelly & Hawes, 2005). $\delta^{13}C$ is generally not a reliable source indicator for aquatic plants because its assimilation and fractionation are controlled more by physiological factors related to photosynthesis, rather than by kinetic effects associated with preferential uptake of lighter isotopes (Kendall et al., 2007). Globally, several studies have shown that freshwater primary producers become enriched in $\delta^{15}N$ when exposed to sewage inputs or intensified land use, with reported values ranging from 0.5 – 14 ‰ for macrophytes, 2.1 – 10.9 ‰ for SPOM, and 0.77 – 13.7 ‰ for periphyton (Cole et al., 2004; Steffy & Kilham, 2004; Bacchus & Barile, 2005; Benson et al., 2008; Kohzu et al., 2008; Li et al., 2011; Meyer et al., 2022b; Page et al., 2022). Although these studies span a diverse range of vascular plant and periphyton species with differing functional traits, resource acquisition strategies, and external nutrient pressures, they remain broadly comparable to the current study, which also included a wide range of species exposed to varying pore water nitrate concentrations (Cole et al., 2004). The alignment of our producer $\delta^{15}N$ values with oligotrophic ranges position them at the lower end of sewage-enriched ranges and indicate that littoral primary producers along Spencer Road are not assimilating large amounts of septic-derived nitrate.

Table 3.1. Comparison of isotopic ranges ($\delta^{15}N$ and $\delta^{13}C$) of primary producers in this study and Lake Wanaka (Kelly & Hawes, 2005). *SPOM* is suspended particulate organic matter.

	Lake Tarawera[1]		Lake Wanaka[2]	
Producer group	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)
SPOM	0.40 to 3.06	–30.55 to –19.61	0.2 to 4.1	–28.8 to –27.5
Macrophytes	0.18 to 2.83	–28.17 to –12.48	0.0 to 3.0	–16.7 to –13.6
Periphyton	0.34 to 2.54	–26.29 to –5.89	0.6 to 2.4	–24.2 to –20.5

[1] Present study.
[2] Kelly & Hawes (2005).

Future research suggestions

To better clarify the impact of septic wastewater on the benthic ecosystem, primary producers should be separated by both species and tissue type. Macrophyte leaves primarily assimilate nitrate from the water column, while roots uptake nitrogen from the sediment, resulting in distinct isotopic signatures in above- and below-ground biomass (de Brabandere et al., 2007). Different macrophyte species also have different resource acquisition strategies and therefore may be more (or less) susceptible to assimilating septic-derived benthic inputs (Bacchus & Barile, 2005). To better detect benthic uptake, future studies should place greater emphasis on analysing the isotopic composition of below-ground biomass.

Conclusion

This study aimed to capture the extent of septic-derived nutrient contamination in the nearshore littoral zone of the urbanised western shoreline of Lake Tarawera, and to assess whether these nutrients are assimilated by littoral primary producers. A hydrological connection between the inner catchment and the littoral zone was supported by the positive correlations observed between groundwater inflow and 3-week/1-month previous rainfall. Stable isotope analyses of nitrate in pore water revealed hotspots of enriched $\delta^{15}N$ at T4, T5, T8 and BVR, with values comparable to those observed in septic plumes elsewhere. However, isotopic analysis of periphyton, macrophytes, and SPOM showed little $\delta^{15}N$ enrichment, with values similar to those measured in oligotrophic Lake Wanaka and at the lower end of wastewater-impacted systems globally. Several factors, including microbial processing, fractionation during uptake, and the overall limited stable isotope dataset constrained our ability to confidently attribute nitrate enrichment to septic inputs.

Overall, this study provides some evidence that wastewater from septic systems is likely entering the nearshore littoral zone of Lake Tarawera, although this is spatially heterogeneous along the urban shoreline. These inputs appear to have minimal impact on the littoral primary producers, suggesting that other nitrate sources are more important for producer assimilation. Future research should incorporate additional geochemical tracers of septic effluent (e.g. sucralose, chloride), a seasonal stable isotope monitoring program, and improved seepage chamber deployments to better quantify nutrient influx and sources to the nearshore littoral zone. Species- and tissue-specific isotopic analyses of primary producers will further refine estimates of benthic nitrogen uptake and clarify the contribution of septic-derived nutrients to littoral biomass.

Chapter 4

Benthic primary production in the nearshore littoral zone of Lake Tarawera and the potential impact of anthropogenic nutrients

Introduction

Benthic primary production is an important component of whole-lake metabolism, particularly in oligotrophic systems (Vadeboncoeur et al., 2003). Globally, many clear-water lakes are experiencing increased algal growth in the littoral zone despite pelagic water quality metrics appearing healthy, potentially reflecting increased nutrient pollution of groundwater from anthropogenic catchment practices (Vadeboncoeur et al., 2021). As benthic production supports primary consumers and the wider food web, alterations to its structure and function can have cascading effects throughout lake ecosystems (Vander Zanden et al., 2006). Despite growing evidence that catchment alteration can substantially modify nutrient dynamics, ecosystem structure, and light availability in littoral zones, the extent to which these changes influence benthic primary production and metabolism remains poorly quantified.

Lake benthic communities typically comprise periphyton (attached algae), microphytobenthos (algae within the sediment), and macrophytes. In oligotrophic systems, these producers can regulate nutrient availability to the water column, suppressing phytoplankton growth and maintaining a clear-water state (Vadeboncoeur et al., 2003). Increased periphyton growth is often an early warning indicator of eutrophication from anthropogenic activities (Naranjo et al., 2019). When nutrient loading to the littoral zone becomes excessive, phytoplankton can outcompete benthic producers for light, leading to shading and the eventual collapse of benthic production (McCormick et al., 2021). Understanding these signs of ecosystem change are important for lake management and conservation; however, the lack of baseline metabolic data often limits our ability to detect early ecological shifts and interpret their significance.

Benthic chambers provide a low-cost, effective means of estimating gross primary production (GPP), a measure of the rate at which benthic producers convert inorganic carbon into organic matter. Typically constructed from clear acrylic plastic, chambers are inserted into littoral sediments to create closed environments over benthic producers (Godwin et al., 2014). Incubations lasting minutes to as long as days allow changes in dissolved oxygen (DO) or dissolved inorganic carbon to be used as proxies for GPP rates (Hall et al., 2007). Chambers are usually deployed in paired light and dark configurations to measure net ecosystem production (NEP) and community respiration (CR), which together allow estimation of GPP. This approach is particularly effective because it captures metabolism under natural conditions and allows repeated measurements across spatial and temporal gradients (Hall et al., 2007).

At Lake Tarawera, the urbanised western margin is likely contributing nutrients to the littoral zone via groundwater inputs from septic systems, as discussed in chapters 2 and 3. However, the potential influence of these inputs on benthic communities is unknown, as benthic production in the lake has not previously been quantified.

I hypothesised that areas influenced by septic contamination would exhibit elevated benthic production. Using benthic chambers, this study aimed to: (1) quantify benthic production along the littoral zone adjacent to the urban margin across space and time; (2) identify the key environmental drivers of benthic production using multivariate analyses; and (3) assess whether spatial and temporal patterns in benthic production correspond to areas likely influenced by septic-derived nutrient inputs. While the study design does not allow a direct causal test of septic-derived nutrient effects, these analyses provide a baseline for understanding littoral metabolism and its potential sensitivity to urban-derived nutrient inputs.

Methods

Experiment design

Six sites in the littoral zone on the urban western shoreline of Lake Tarawera were selected for measuring littoral community primary production (see Figure 4.1). Littoral sites C1, T2, T4, T5, and BVR were the same as those included in the Chapter 3 experiment, with a new control site included (C2), located on the southern shoreline of the lake. Additionally, C1, T2, T4 and T5 also lie downslope of piezometers monitored in the Chapter 1 experiment. Seasonal sampling was conducted on February 21 – 23, June 4 – 6, September 4 – 6, and December 4 – 6 in 2024. Fieldwork was undertaken at two sites per day, with one site visited in the morning and the other in the afternoon.

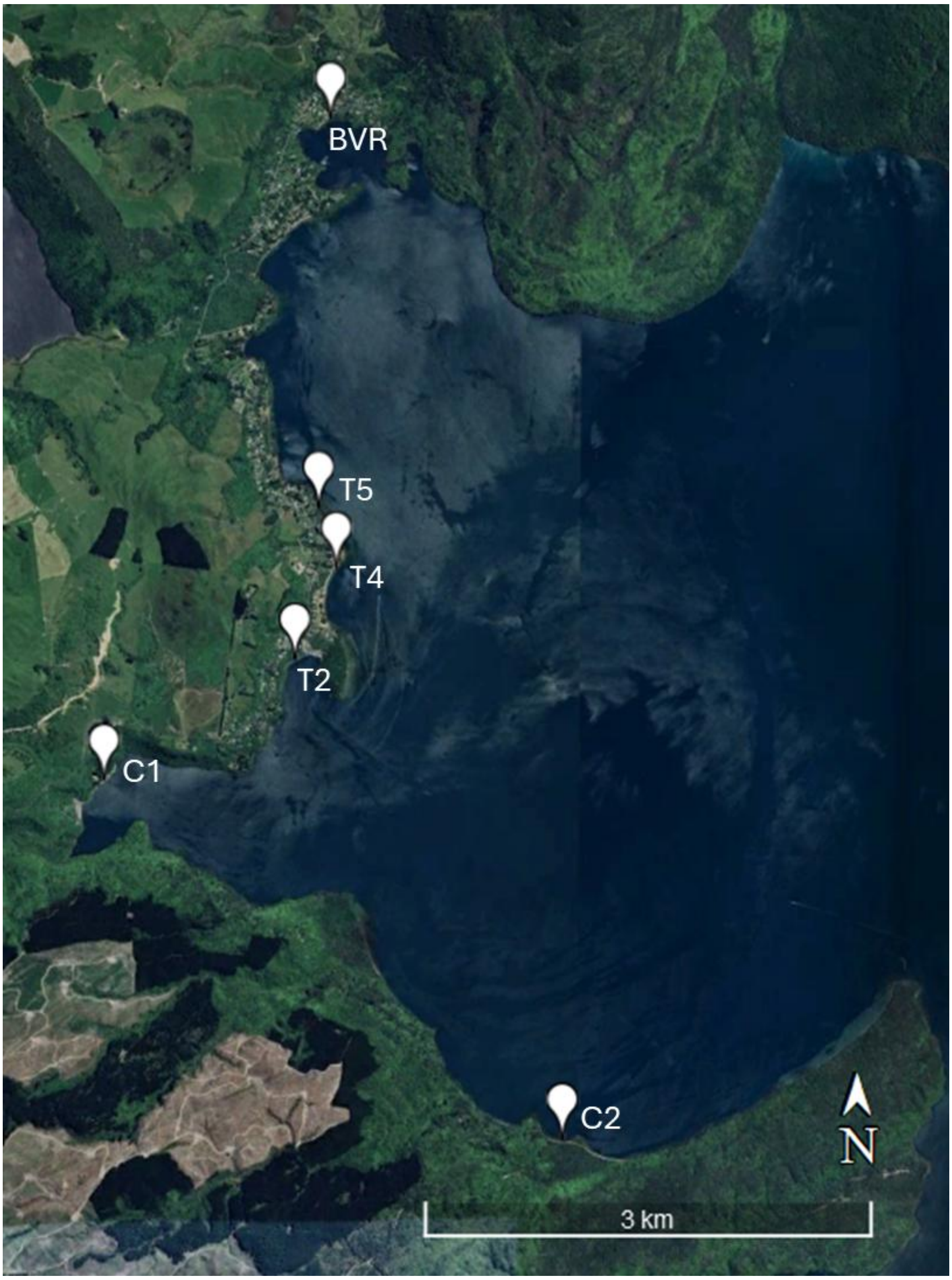

Figure 4.1. Littoral sites for benthic chamber incubations along the western and southern shorelines of Lake Tarawera. Site C2 and C1 are control sites not located near the urban zone.

Benthic chamber design

Benthic chambers were used to quantify gross primary production via dissolved oxygen flux (GPP; Figure 4.2). Chambers captured DO flux from the entire littoral community and overlying water column, including macrophytes, epiphytic and periphytic algae, microphytobenthos and phytoplankton, subsequently referred to as the community. The chambers were the same as those used to measure groundwater seepage in Chapter 2 and had a radius of 0.4 m, area of 0.126 m^2, and volume of 16 L, after accounting for DO logger volume. Each chamber was fitted with two-way valves attached to the side and top of the chamber. Three of the chambers were transparent (light chambers) to measure oxygen fluxes driven by photosynthesis (net ecosystem production; NEP), while three chambers were covered in black duct tape to exclude light (dark chambers) in order to quantify oxygen flux associated with respiration (community respiration; CR). The duct tape successfully reduced ambient light levels to 2.9 – 384 lux in dark chambers (compared to light chamber: 3630 – 50900 lux; see Appendix C - Table C1). In addition, two control chambers were deployed to measure water column metabolism exclusively. These chambers were secured to Perspex base plates, thereby isolating water column from the benthic environment.

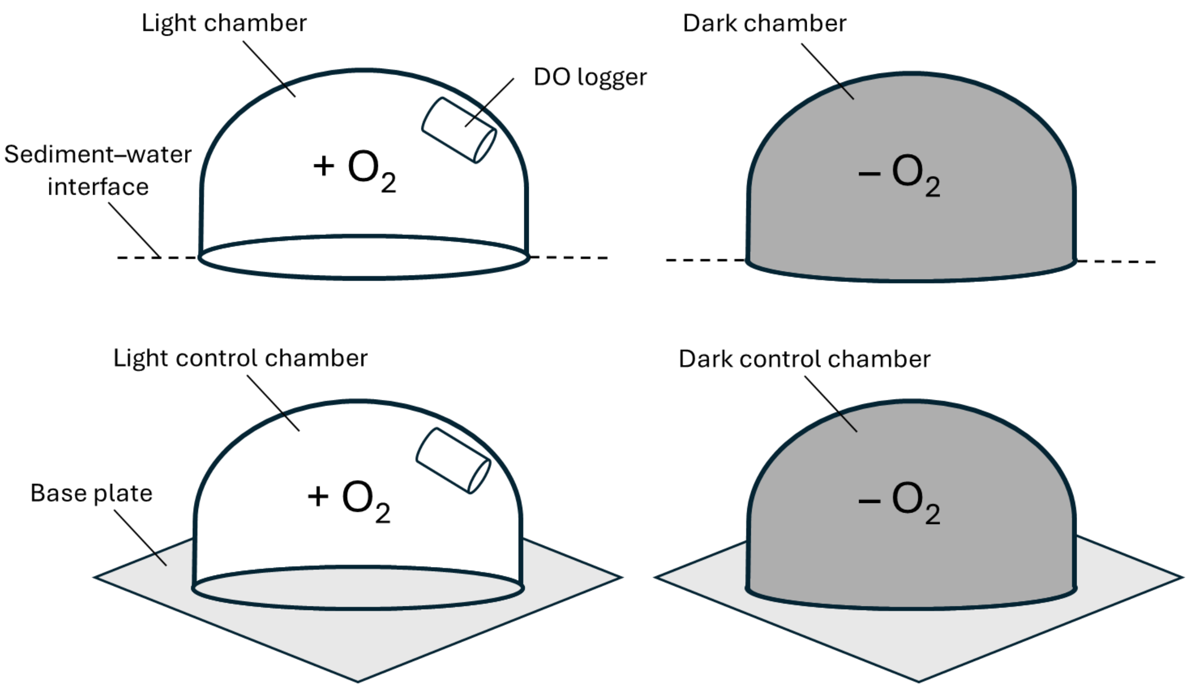

Figure 4.2. Benthic chambers used in the study. Chambers are made of clear acrylic plastic, with dark chambers covered in black duct tape to block incident light. Each chamber is fitted with a MiniDOT logger (PME, California, USA) attached to the inside. The expected increase or decrease in dissolved oxygen (O_2)

for each chamber is included. Control chambers were secured to baseplates and exclusively measured dissolved oxygen flux from the water column.

GPP incubations

Dissolved oxygen, light and temperature measurement

Prior to deployment, a MiniDOT DO and temperature logger (PME, California, USA) was secured to the inside of each chamber and set to log DO at 2-minute intervals. No mixing mechanism was secured inside the chambers due to the short deployment duration. A HOBO Pendant MX Temperature/Light Data Loggers (Onset, Bourne, USA) was placed on the baseplate inside the light control chamber to measure light availability and water temperature at 2-minute intervals.

Chamber deployments and sample collection

Paired chambers (one light and one dark) were deployed at 2 m intervals along a 10 m transect at a depth of 0.5 – 0.8 m running parallel to the lakeshore. The control chambers were placed in a pair at either end of the transect. Open-bottom chambers were pushed approximately 5 cm into the sediment, with valves left in the open position to allow any air to escape. Paired chambers were selectively placed over areas of similar vegetation coverage to ensure NEP and CR remain comparable. Control chambers were filled with ambient water and clamped to baseplates to create a tight seal. Once all chambers were installed, all valves were switched to the closed position, and the start time recorded.

Chambers were deployed for 1.5 hours, after which 50 mL of water was sampled from each control chamber and the surrounding water and filtered through Whatman GF/C filter papers for chlorophyll-*a* (chl-*a*) analysis. Filters were then placed on ice. As each chamber was removed individually from the sediment, a single sediment core, including overlying vegetation, was taken from a random location within the chamber footprint. Sediment cores were taken using a 7 cm internal diameter aluminium tube pushed ~5 – 10 cm into the sediment. Cores were then carefully removed, placed in individual ziplock bags, and stored on ice.

Laboratory processing and analysis

A one teaspoon subsample was taken from each core for sediment chl-*a* analysis and placed in a small ziplock bag. All samples were frozen at -20°C until analysis.

Sediment and water chlorophyll-a

Once frozen, sediment subsamples were freeze-dried for 24 hours. Approximately 0.15 g of dried sediment was then weighed into 15 mL centrifuge tubes and chl-*a* was extracted using 90 % buffered acetone. Water column chl-*a* was determined by maceration of filtered samples followed by extraction in 90 % buffered acetone. Following overnight extraction, all samples were centrifuged for 10 minutes at 3300 rpm, and chl-*a* concentrations were determined by fluorometric acidification analysis.

Vegetation

Sediment samples were thawed for vegetation removal. Each sediment sample was placed in a white tray and vegetation was manually removed by hand and rinsed in a separate water bucket. Vegetation samples were oven-dried for one week at 55 °C. Samples were weighed for dry vegetation biomass (g), which was scaled to g m^{-2} using the sediment core area (39 cm^2).

Organic matter and particle size

Following vegetation removal, each sediment sample was subsampled for organic matter content and particle size analysis. Subsamples (~20 g) were oven-dried at 55 °C for one week, after which each sample was divided into two subsamples. Organic matter content was determined by loss on ignition, whereby organic matter content was determined from the difference in mass before and after combustion in a furnace at 550 °C for four hours. The second subsample was used for particle size analysis. Samples were initially treated with 10 % hydrogen peroxide to remove organic material, followed by particle size determination using laser diffraction with a Malvern Mastersizer 3000 (Malvern Panalytical, Massachusetts, USA).

Data processing for GPP

To estimate GPP, DO time series data was manually trimmed to remove noise associated with chamber acclimation, water intrusion resulting from chamber breaches, and fluxes occurring in the opposite direction of the expected biological flux. As linear DO trends took significantly less time than the allocated 30 minutes to stabilise, only the first four minutes of each deployment were removed, with additional points trimmed on a case-by-case basis where trends took longer to establish (generally <30 minutes). Chamber breaches and water intrusions were identified by abrupt spikes in DO concentration; in these cases, data were removed from the initial point of deviation until DO returned to a consistent linear trend. In rare cases where the DO trend reversed direction towards the end of the incubation, all datapoints were removed following the point of deviation.

Following trimming, linear regressions were fitted to each DO time series, and the slope of the regression was taken as the rate of DO flux (mg L^{-1} min^{-1}). This rate was converted to a daily rate (mg O_2 m^{-2} day^{-1}) using the chamber footprint area (0.126 m^2), chamber volume (16 L), and duration of the trimmed incubation period. Regression R^2 values ranged from 0.63 to 0.99 for open-bottom chambers and 0.12 to 0.98 for control chambers.

Community GPP (mg O_2 m^{-2} hr^{-1}) was calculated as the sum of production and respiration:

$$\mathrm{GPP}_{\mathrm{community}} = \mathrm{NEP}_{\mathrm{community}} + \mathrm{CR}_{\mathrm{community}} \tag{1}$$

Where, $\mathrm{NEP}_{\mathrm{community}}$ is net ecosystem production measured as the net DO flux in open-bottom light chambers, and $\mathrm{CR}_{\mathrm{community}}$ is community respiration treated as a positive DO flux in paired open-bottom dark chambers.

Water column GPP (mg O_2 m^{-2} hr^{-1}) was calculated using the same approach:

$$\mathrm{GPP}_{\mathrm{wc}} = \mathrm{NEP}_{\mathrm{wc}} + \mathrm{CR}_{\mathrm{wc}} \tag{2}$$

Where, $\mathrm{NEP}_{\mathrm{wc}}$ and $\mathrm{CR}_{\mathrm{wc}}$ were measured in paired closed-bottom light and dark control chambers, respectively.

Benthic GPP (mg O_2 m^{-2} hr^{-1}) was then isolated by correcting mean community GPP at each site for the contribution from the overlying water column:

$$\mathrm{GPP}_{\mathrm{benthic}} = \left(\mathrm{NEP}_{\mathrm{community}} - \mathrm{NEP}_{\mathrm{wc}}\right) + \left(\mathrm{CR}_{\mathrm{community}} - \mathrm{CR}_{\mathrm{wc}}\right) \tag{3}$$

Across all calculations, CR fluxes were negative but have been treated as positive values. This approach assumes that water column processes measured in closed-bottom chambers were representative of those occurring within open-bottom chambers, and that respiration rates are similar in light and dark incubations.

Statistical analyses

Spatial and temporal differences in GPP

As water column GPP was measured using only a single paired control chamber, it was not possible to calculate benthic GPP by correcting for water-column processes for spatial and temporal comparisons in this section. Consequently, uncorrected community GPP derived from paired open-bottom chambers was used.

Due to low sample size and unequal variances among groups (i.e., seasons and sites), Welch's ANOVA tests were used to assess differences in mean community GPP among sites and among seasons, A two-way ANOVA including the interaction between site and season was not tested due to limited sample size. When significant effects were detected, Games-Howell post-hoc tests were applied for pairwise comparisons (p <0.05).

To further examine spatial variations within seasons, Welch's ANOVA was conducted separately for each season to test for differences in community GPP among sites. Holm-adjusted p-values were applied across seasonal tests to control for Type I error. Games-Howell post-hoc tests were used for significant pairwise comparisons (p <0.05).

Drivers of GPP

Whole community GPP

Due to uneven sample sizes between GPP and environmental variables, data was aggregated to site-season means, resulting in a dataset of 24 observations. A permutational multivariate analysis of variance (PERMANOVA) was then used to assess the environmental drivers of GPP, as this approach is robust to non-normality and can accommodate unbalanced designs (Anderson, 2001). As replication could not be retained in the aggregated dataset, statistical power was limited. Consequently, a Spearman rank correlation matrix was used to identify highly correlated predictor variables and reduce multicollinearity prior to model fitting. Predictor variables were also standardised to z-scores. PERMANOVA models were run with site and season included as factors, retaining the same set of environmental predictor variables. All models were implemented using the *adonis2* function in R (R Core Team, Vienna) with 999 permutations, and marginal effects were assessed.

Water column and benthic GPP

As water column GPP was not measured in replicate at each site, there was not enough data to support formal hypothesis testing. Therefore, Spearman rank correlation analyses were used to assess relationships between water column GPP and environmental drivers, as well as between benthic GPP and the same set of drivers. These relationships were compared with Spearman correlations for community GPP to evaluate any differences in environmental drivers of benthic and pelagic primary production in the nearshore littoral zone.

Results

Site characterisation

Dominant macrophyte species

Sites C2, C1, and T2 had steeper lakebed gradients (~4–8 %) and were dominated by tall-growing macrophyte species. Control sites C2 and C1 were dominated by the native species *Myriophyllum triphyllum* (water milfoil), while the invasive species

Lagarosiphon major (oxygen weed) was dominant at T2 (Figure 4.3). Sites T4 and T5 were characterised by lower lakebed gradients (<2 %) and a resulting mixed turf environment, with the dominant species being *Eleocharis pusilla* (dwarf spike rush). Site BVR was a large bay of low lakebed gradient (<2 %) and patchy vegetation cover, dominated by both *E. pusilla* and the tall-growing invasive *Ceratophyllum demersum* (hornwort). The macrophyte communities at each site remained relatively stable across all sampling events. During the September sampling, however, a dense epiphytic brown algal growth occurred at T5, while in December a large epiphytic green filamentous algal growth occurred at BVR (pictured in Figure 4.3f).

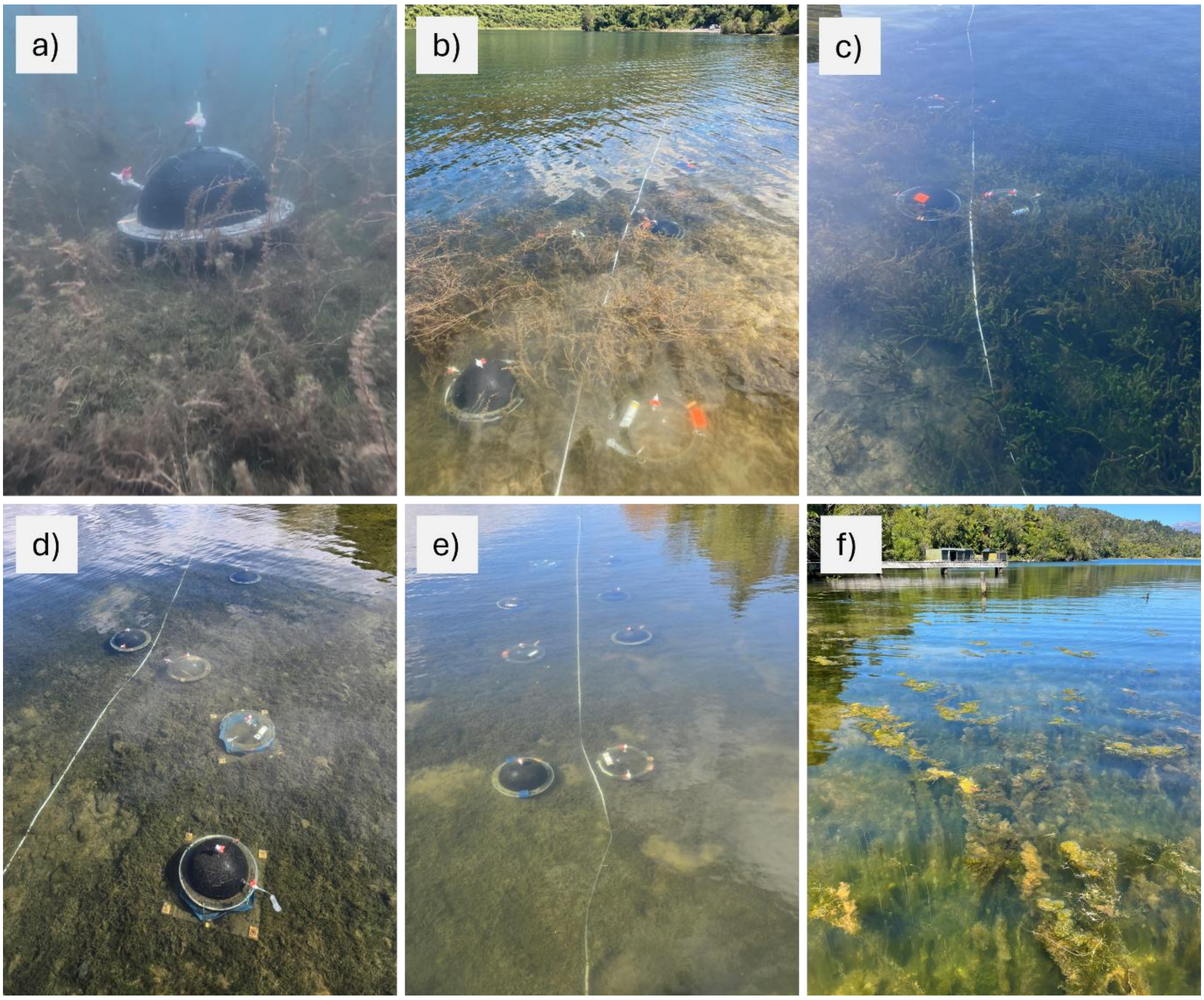

Figure 4.3. Images of each sampling site with dominant macrophyte species present. Sites C2 (a) and C1 (b) were dominated by *Myriophyllum spicatum*, while T2 (c) was dominated by *Lagarosiphon major*. Sites T4 (d) and T5 (e) were dominated by *Eleocharis pusilla*, and BVR (f) was a mixed *E. pusilla/Ceratophyllum*

demersum assemblage, with an epiphytic green filamentous algal bloom present in the December sampling round.

Environmental and ecological parameters

Across all seasons, mean D_{50} sediment particle size was largest at C1 (258 µm) and smallest at C2 (107 µm; Table 4.1). Mean light at the lakebed was variable over sampling days, with C2 experiencing highest mean light (32,541 lux) and T2 experiencing the lowest (12,943 lux). Mean water column chl-*a* was highest at T5 and (17.3 µg L^{-1}) and lowest at C2 (11.9 µg L^{-1}), while water column total nitrogen was also lowest at C2 (0.09 mg L^{-1}) but highest at T4 (0.23 mg L^{-1}). Vegetation showed high variability across sampling occasions, with highest mean (± standard deviation) dry vegetation biomass occurring at BVR (440 ± 720 g m^{-2}) and lowest at T4 (174 ± 92 g m^{-2}).

Several parameters showed a comparably low range of mean values across sites, including water temperature (18.4 – 20.0 °C), total phosphorus (0.010 – 0.012 mg L^{-1}), sediment chl-*a* (2.91 – 5.51 µg L^{-1}), sediment organic matter (2.1 – 4.8 %), and nitrate-N + nitrite-N (0.010 – 0.020 mg L^{-1}).

Table 4.1. Mean values (± standard deviation) for environmental parameters at each littoral site. Number of samples (*n*) differs for some environmental parameters. Light at lakebed values have been rounded to three significant figures.

		Littoral site					
Environmental parameter	*n*	C2	C1	T2	T4	T5	BVR
Water chl-*a* (μg L^{-1})	12	11.9 (± 0.5)	14.5 (± 0.5)	16.2 (± 0.7)	15.9 (± 0.8)	17.3 (± 1.1)	13.0 (± 0.5)
Sediment chl-*a* (μg g^{-1})	24	2.91 (± 3.03)	3.08 (± 2.66)	3.56 (± 1.93)	5.04 (± 2.24)	4.28 (± 2.72)	5.51 (± 3.08)
Total nitrogen (mg L^{-1})	4	0.09 (± 0.04)	0.21 (± 0.02)	0.12 (± 0.04)	0.23 (± 0.09)	0.15 (± 0.03)	0.14 (± 0.02)
Nitrate-N + Nitrite-N (mg L^{-1})	4	0.010 (± 0.008)	0.020 (± 0.016)	0.013 (± 0.012)	0.011 (± 0.005)	0.015 (± 0.007)	0.012 (± 0.009)
Total phosphorus (mg L^{-1})	4	0.010 (± 0.002)	0.011 (± 0.003)	0.010 (± 0.003)	0.012 (± 0.006)	0.011 (± 0.002)	0.012 (± 0.002)
Water temperature (°C)	> 180	20.0 (± 4.9)	19.2 (± 4.6)	18.4 (± 4.2)	18.7 (± 4.1)	18.6 (± 4.6)	19.3 (± 4.9)
Light at lakebed (lux)	> 180	32500 (± 14300)	28600 (± 10800)	12900 (± 5290)	15000 (± 13300)	18000 (± 11900)	16300 (± 9890)
Sediment organic matter (%)	12	4.1 (± 0.4)	2.4 (± 0.3)	4.8 (± 1.0)	3.6 (± 0.5)	3.4 (± 0.8)	2.1 (± 0.2)
Sediment particle size (D_{50}; μm)	12	107 (± 13.3)	258 (± 27.7)	199 (± 37.5)	191 (± 26.0)	211 (± 40.1)	252 (± 17.1)
Dry vegetation biomass (g m^{-2})	24	398 (± 245)	136 (± 199)	198 (±117)	174 (± 92)	245 (± 148)	440 (± 720)

Community and benthic GPP

Community GPP

Whole community GPP measured in open-bottom chambers ranged from 67.8 to 397.0 mg O_2 m^{-2} hr^{-1} across all sites and sampling occasions, with an overall mean of 166.5 ± 73.3 mg O_2 m^{-2} hr^{-1} (Figure 4.4). Site T4 experienced the lowest mean and minimum GPP values, the highest mean GPP rate occurred at C1, and BVR experienced the largest range of GPP rates as well as the maximum rate.

There was no statistically significant difference in mean GPP between sites (p = 0.063, F = 2.36). There was a significant difference in mean GPP between sampling occasions (p = 0.002, F = 5.87). Post-hoc tests revealed that mean GPP was significantly lower in June than in September (p = 0.01) and December (p = 0.007).

Benthic GPP

When corrected for water column GPP, mean site benthic GPP rates ranged from 57 to 296 mg O_2 m^{-2} hr^{-1} (see Appendix C - Table C2). The lowest rate was observed at T5 in February, and the highest rate at BVR in December. The overall mean rate across all sites and seasons was 138.4 ± 64.0 mg O_2 m^{-2} hr^{-1}.

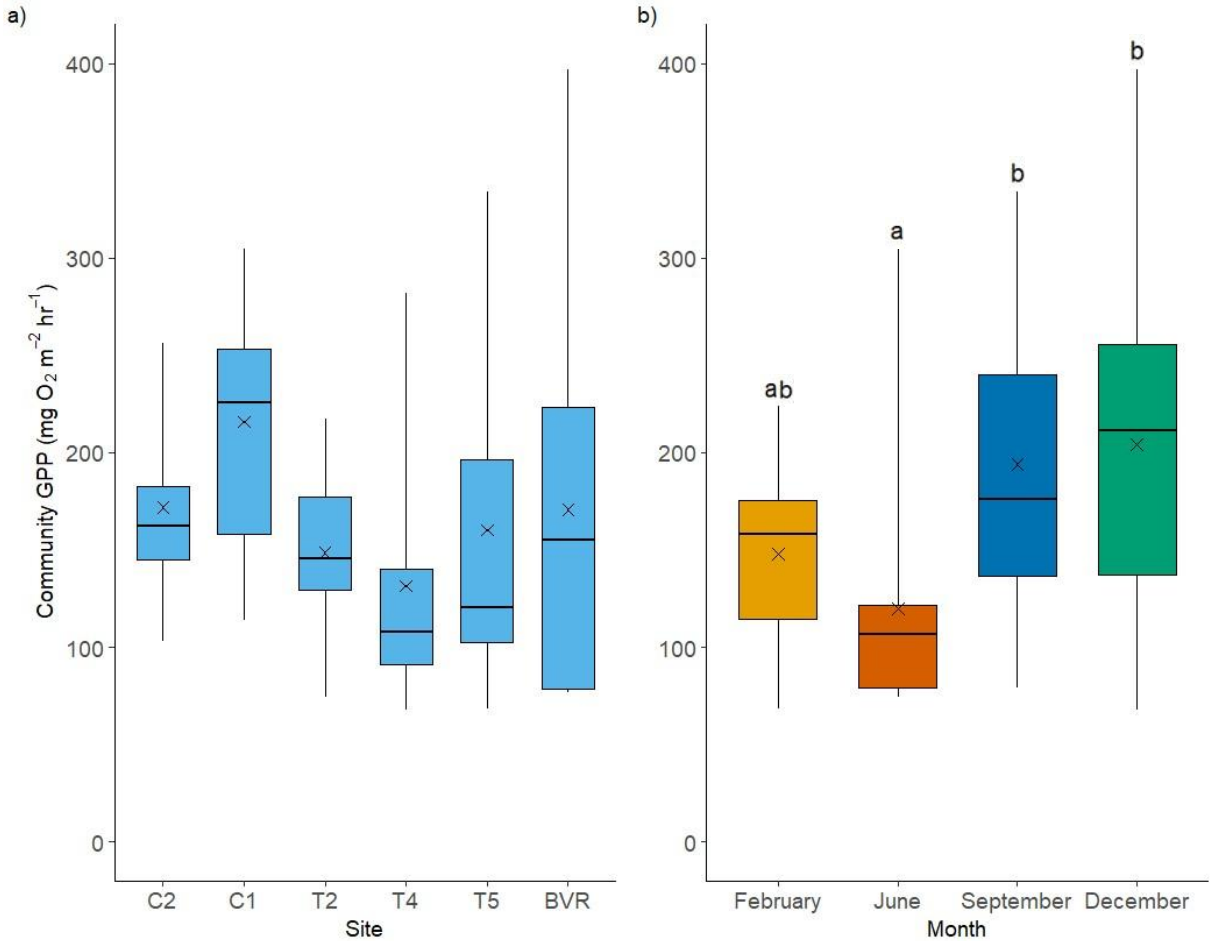

Figure 4.4. Boxplots showing community gross primary production (GPP) rates across a) sites and b) sampling occasions. Boxes represent the interquartile range, horizontal black lines indicate median values, black crosses show mean values, and whiskers denote the minimum and maximum values. Horizontal dashed lines at zero separate positive and negative GPP rates. Where shown, letters above whiskers indicate significant differences identified by Welch's ANOVA followed by Games-Howell post-hoc tests, with shared letters denoting no significant difference between groups.

Within-season patterns

Welch's ANOVA tests identified significant differences in mean site GPP within all seasons (p <0.05; Appendix C - Table C3). During the February sampling round, site BVR had significantly higher mean GPP than sites T4 and T5, with no other significant pair-wise comparisons detected (Figure 4.5). In June, despite the global test being significant (p = 0.039), no significant pair-wise comparisons were detected, likely due to limited replication and variance among sites reducing the power of post-hoc tests. In September, C1 and T5 had significantly higher mean GPP than T2 and BVR, with C1 additionally having a higher mean GPP than C2. In December, mean GPP was only significantly higher at C2 and C1 than T4.

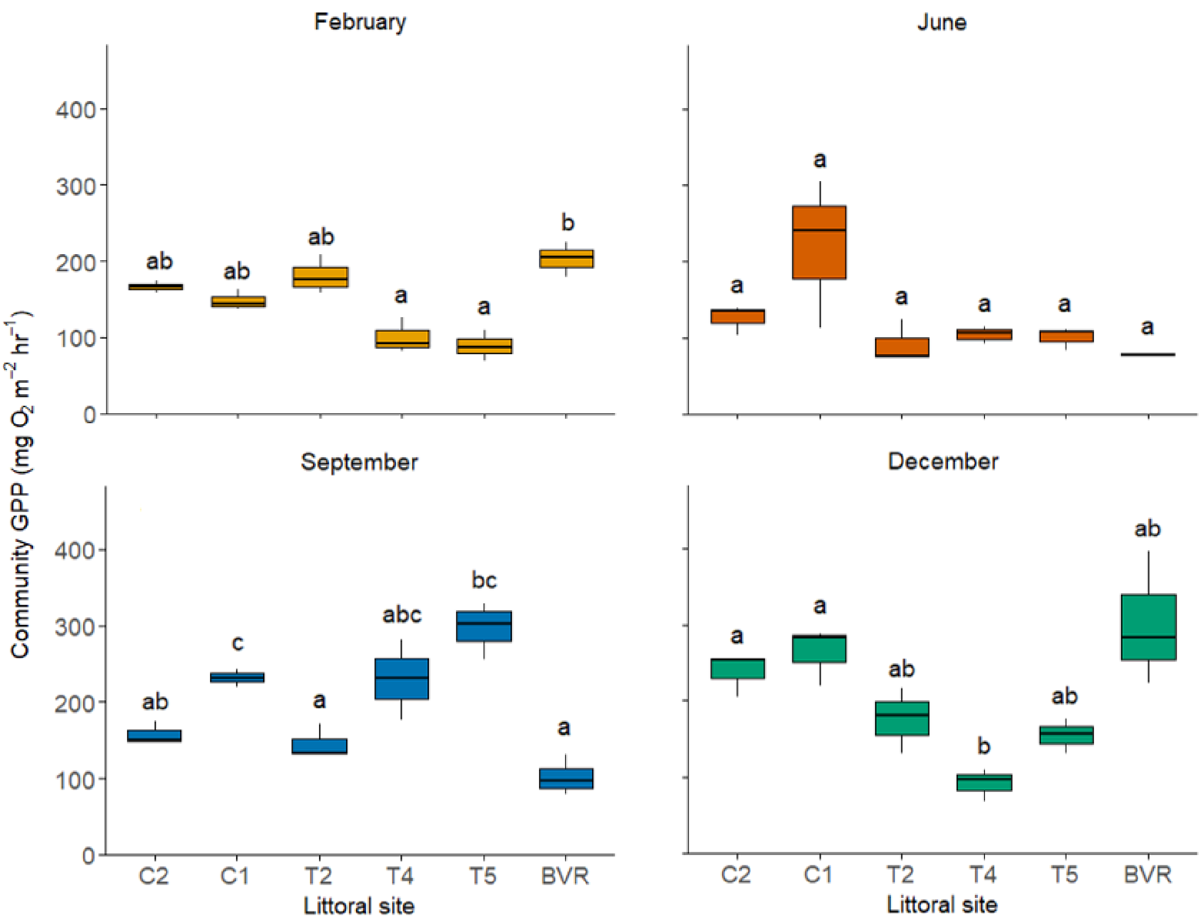

Figure 4.5. Boxplots showing community gross primary production (GPP) at each site within each season (n = 3). Boxes represent the interquartile range, horizontal black lines indicate median values, and whiskers denote the minimum and maximum values. Horizontal dashed lines at zero separate positive and negative GPP rates. Where shown, letters above whiskers indicate significant differences ($p < 0.05$) identified by Welch's ANOVA followed by Games–Howell post-hoc tests, with shared letters denoting no significant difference between groups. Separate Welch's ANOVA tests were conducted for each season with Holm-adjusted p-values interpreted; therefore, letters should not be compared across seasons.

Drivers of GPP

During PERMANOVA model selection, several variables were removed before final model selection due to high correlation with other variables. Soil organic matter and sediment particle size were strongly negative correlated (Spearman's ρ = -0.81), with particle size retained as it provides a more direct measure of benthic habitat suitability for macrophytes. Water column chl-*a* and sediment chl-*a* were moderately correlated (ρ = 0.56), with sediment chl-*a* retained due to its relevance to the benthic community. Water temperature was also strongly correlated with light availability (ρ = 0.73), with light retained as the dominant driver of photosynthesis. Total nitrogen (TN) and total phosphorus (TP) were moderately correlated (ρ = 0.56). Total nitrogen was retained because it is often the more limiting nutrient for pelagic primary productivity in oligotrophic New Zealand lakes (Abell et al., 2010), and periphyton in New Zealand streams can be limited by TN, TP, or co-limited, depending on species composition (Biggs, 1990), making TN the more informative variable to retain.

The final PERMANOVA model included dry vegetation biomass, light availability at the lakebed, TN, and sediment particle size. Sediment chl-*a* was excluded from the final model as it explained less than 0.001 % of variation in community GPP when included.

Dry vegetation biomass was the only variable identified as a significant predictor of GPP across the site and season PERMANOVA models (Table 4.2). Vegetation biomass explained 19 % of the variation in GPP in the global model (p <0.01, R^2 = 0.19), 23 % in the site model (p = 0.01, R^2 = 0.23), and 16 % in the season model (p = 0.03, R^2 = 0.16). Light availability was a marginally significant predictor of GPP in the global model (R^2 = 0.11, p = 0.05). Collectively, predictors in the global model explained 51 % of the variation in GPP, with an improvement to 57 % when site was included as a factor, and 53 % when season was included as a factor. Sediment particle size and TN were not significant predictors of GPP in any PERMANOVA models (p >0.05, R^2 <0.1).

Table 4.2. Results of PERMANOVA models examining the influence of environmental predictors on gross primary production (GPP), run separately by site, by season, and as a global model. All predictors were standardised to z-scores. The proportion of variance explained (R^2), associated *p*-values, F-statistic, sum of squares and degrees of freedom (Df) are reported for each predictor and the model residual. Statistically significant predictors (p <0.05) are identified in bold. Sediment particle size is the median value of the particle size distribution.

Global model					
Predictor	Df	Sum of squares	R^2	*F*-statistic	*p*-value
Dry vegetation biomass	1	18181	0.19	7.21	**< 0.01**
Light	1	11258	0.11	4.46	**0.05**
Total nitrogen	1	2779	0.03	1.10	0.32
Sediment particle size	1	8793	0.09	3.49	0.09
Model residual	19	47935	0.49	-	-
Site model					
Predictor	Df	Sum of squares	R^2	*F*-statistic	*p*-value
Site	5	6084	0.06	0.41	0.83
Dry vegetation biomass	1	22553	0.23	7.54	**0.01**
Light	1	3400	0.03	1.14	0.29
Total nitrogen	1	284	< 0.01	0.09	0.77
Sediment particle size	1	743	< 0.01	0.25	0.67
Model residual	14	41850	0.43	-	-
Season model					
Predictor	Df	Sum of squares	R^2	*F*-statistic	*p*-value
Season	3	2158	0.02	0.25	0.86
Dry vegetation biomass	1	15856	0.16	5.54	**0.03**
Light	1	7292	0.07	2.55	0.13
Total nitrogen	1	3459	0.04	1.21	0.29
Sediment particle size	1	8580	0.09	3.00	0.09
Model residual	16	45776	0.47	-	-

Components of GPP in the littoral zone

Water column GPP

GPP rates measured in closed-bottom control chambers ranged from 5.3 mg O_2 m^{-2} hr^{-1} at BVR to 66.3 mg O_2 m^{-2} hr^{-1} at T5 (Figure 4.6; Appendix C - Table C4). Three sites recorded negative NEP values in December, including BVR, C2, and T5. Mean water column GPP was highest in September and lowest in June, while the most variation among sites occurred in December. The mean contribution of water column GPP to mean community GPP across all data was 19.1 %, with mean contributions ranging from 13.7 % at C1 to 26.3 % at T5.

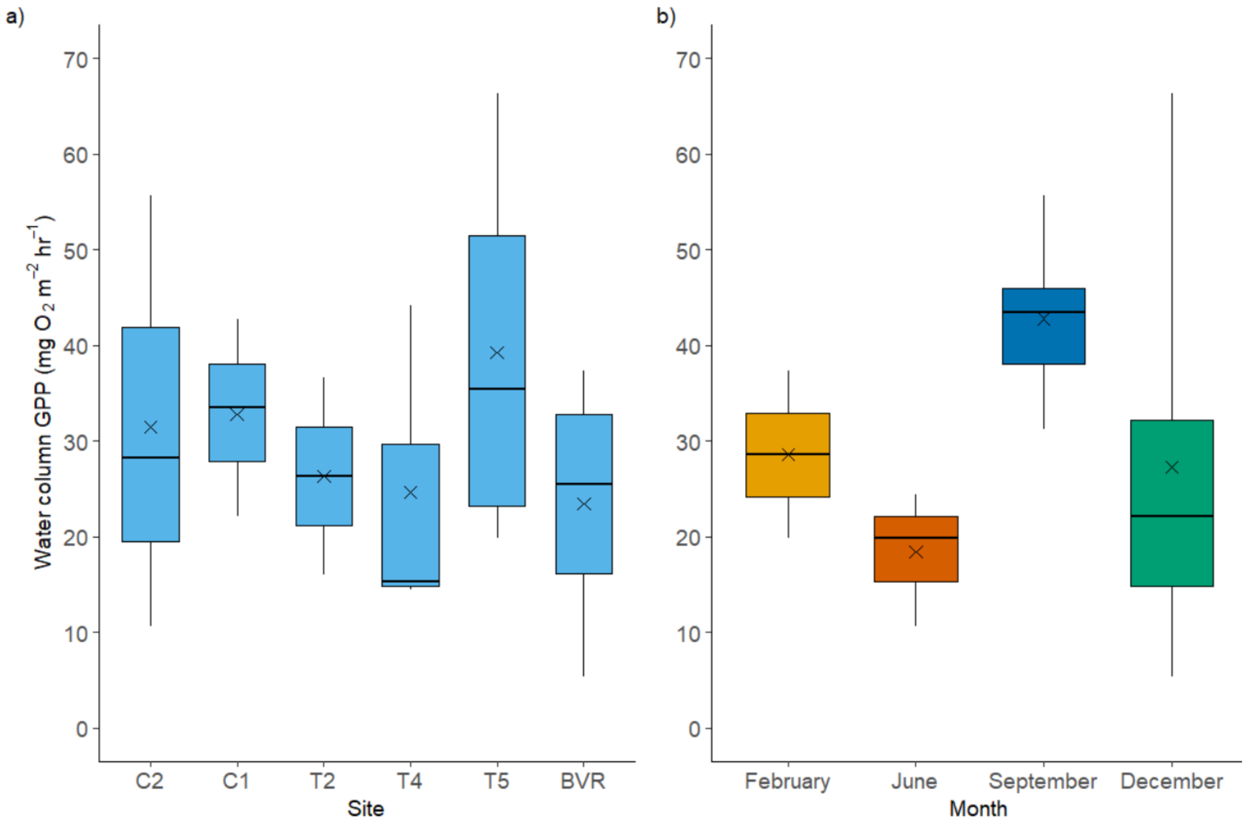

Figure 4.6. Boxplots showing gross primary production (GPP) rates in closed-bottom control chambers across a) sites and b) sampling occasions. Boxes represent the interquartile range, horizontal black lines indicate median values, black crosses show mean values, and whiskers denote the minimum and maximum values. Horizontal dashed lines at zero separate positive and negative GPP rates.

Correlations between environmental variables and GPP components

Community and benthic GPP were both positively correlated (Spearman) with light availability (p = 0.004 and 0.02, respectively; Figure 4.7). No other significant correlations were observed. Although not statistically significant, water column GPP showed positive associations with TN and TP, whereas benthic GPP was negatively associated with these variables.

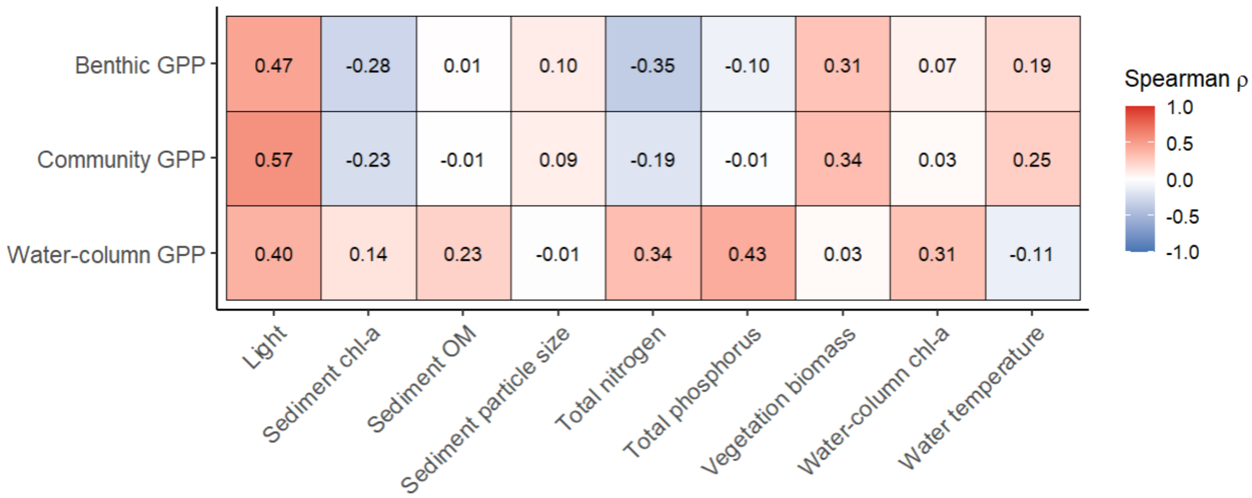

Figure 4.7. Spearman rank correlations between benthic, community, and water column gross primary production (GPP) and environmental drivers. Square colours indicate the direction and strength of correlations. Values within squares represent Spearman's ρ.

Discussion

Overall patterns of benthic community GPP

This study provides the first measurements of whole community GPP in a New Zealand lake using oxygen flux as a proxy for metabolism. Measured rates ranged from 67.8 to 397.0 mg O_2 m^{-2} hr^{-1}, with water column-corrected benthic GPP ranging from 57 to 296 mg O_2 m^{-2} hr^{-1}. Although this field remains underdeveloped in New Zealand, several comparable studies have measured the production of individual benthic producers using the ^{14}C uptake incubation method. This method generally involves the addition of sodium ^{14}C–bicarbonate to chambers, with ^{14}C uptake by producers used as a proxy for primary production (Andersson & Brunberg, 2006).

To facilitate comparison, benthic NEP measured in this study (33 – 271 mg O_2 m^{-2} hr^{-1}; see Appendix C - Table C2) was compared to published production rates in the New Zealand literature, after converting carbon fluxes to oxygen units using a 1:1 molar equivalence and a mass conversion factor of 2.67 (Puts et al., 2022). Where studies reported daily rates, values were divided by 24 to obtain hourly equivalent rates. Global GPP rates discussed in later sections have been converted using the same approach to ensure consistency in comparisons.

New Zealand studies of benthic production

Hawes and Smith (1994) measured periphyton production at depths between 0 and 40 m in the littoral zone of Lake Taupō, reporting comparably high production rates of 646 – 967 mg O_2 m^{-2} hr^{-1}. In a separate study using similar methodology, Kelly and Hawes (2005) measured production of both exotic and native macrophyte species in Lake Wanaka. In their study, invasive *Lagarosiphon major* and *Elodea canadensis* exhibited a mean production rate of 265 mg O_2 m^{-2} hr^{-1}, approximately ten times greater than native *Myriophyllum triphyllum* and *Isoetes alpinus* (24 mg O_2 m^{-2} hr^{-1}). Measurements of epiphyton grown on artificial macrophytes showed that epiphyton production was significantly higher on invasive species (16.5 mg O_2 m^{-2} hr^{-1}) than on native species (11.7 mg O_2 m^{-2} hr^{-1}), due to the more than two times greater surface area available for colonisation on exotic plants (Kelly & Hawes, 2005).

The ^{14}C method measures an unknown value between net and gross production, which limits direct comparison with production estimates measured using the oxygen flux method (Hall et al., 2007). Nevertheless, both benthic NEP and GPP values measured in Lake Tarawera are comparable to macrophyte and epiphyton production in Lake Wanaka, but considerably lower than the reported periphyton rates in Lake Taupō. The similarity between production in Wanaka and Tarawera is unsurprising, as both lakes are microtrophic to oligotrophic, contain similar macrophyte species, and have relatively similar proportions of native vegetation and urban catchment land uses (LAWA, n.d.-a, n.d.-c). In contrast, the unusually high periphyton productivity observed in Lake Taupō is likely driven by nutrient inflow from deeper groundwater sources (Gibbs et al., 2005), which has also been observed elsewhere globally (Hagerthey & Kerfoot, 1998; Périllon et al., 2017). These comparisons indicate that Lake Tarawera falls within the ranges of benthic productivity reported in similar macrophyte-dominated clear-water New Zealand lakes. However, given that groundwater inputs are a significant component of the Tarawera nutrient budget (Wilson, 2022), these findings also highlight the lake's susceptibility to higher periphyton growth in areas with increased groundwater inflow.

Global studies of benthic GPP

Reported benthic community GPP from light-dark chamber studies in lakes across the USA, Sweden, Iceland, and Canada ranges from 0 to 700 mg O_2 m^{-2} hr^{-1} (Table 4.3). Benthic GPP measured in Lake Tarawera falls within this global range; however, most published studies are from systems where macrophytes are sparse or absent. As a result, comparisons are primarily with periphyton-dominated rather than macrophyte-dominated littoral zones. The upper benthic GPP values observed in Lake Tarawera are most comparable to those reported from highly productive systems such as Lakes Erie and Ontario, and Lake Mývatn (Iceland), where benthic production is periodically elevated by significant periphyton blooms of filamentous green algae (*Cladophora glomerata*; Davies & Hecky, 2005; Malkin et al., 2010; McCormick et al., 2021). In the present study, the highest benthic GPP rates occurred at sites T5 (250 mg O_2 m^{-2} hr^{-1}) and BVR (296 mg O_2 m^{-2} hr^{-1}) during periods when extensive epiphytic algal growth was observed on macrophyte surfaces. This suggests that epiphytic algal growth is likely also a major driver of elevated benthic GPP in Lake Tarawera.

Different components of GPP (periphyton, epiphyton, microphytobenthos) are more commonly measured separately in the literature. Reported periphyton GPP ranges typically span 1.1 to 1200 mg O_2 m^{-2} hr^{-1}, epiphyton has been measured as 28.7 mg O_2 m^{-2} hr^{-1} in Lawrence Lake (United States of America), and microphytobenthos has ranged from 0.08 to 384 mg O_2 m^{-2} hr^{-1} in Lake Eckarfjärden (Sweden; see Table 4.3). No limnological studies have specifically incubated macrophytes to quantify their contribution to benthic GPP. Although this highlights a knowledge gap in understanding how perennial plant producers contribute to benthic GPP, the broader literature consistently shows that algal growth is the primary driver of large variability in benthic GPP and is a sensitive indicator of benthic ecosystem change (Vadeboncoeur et al., 2003). In macrophyte-dominated Lake Tarawera, periods of epiphytic algal growth corresponded with the highest benthic GPP rates, whereas variability observed throughout the remainder of the study is less readily explained. This indicates that benthic GPP measured in this study reflected the combined signals of macrophytes and epiphytic algae, rather than macrophytes alone. Distinguishing between these components is therefore important for interpreting seasonal and spatial patterns of benthic production, especially in macrophyte-dominated systems.

Table 4.3. Literature values for gross primary production (GPP) measured for different components of benthic communities in lakes, using paired light and dark incubations. All GPP rates have been standardised to mg O_2 m^{-2} hr^{-1} for comparisons. The method describes what was measured as a proxy for photosynthesis for each study, including dissolved oxygen (DO), ^{14}C bicarbonate uptake (^{14}C), and dissolved inorganic carbon (DIC). Studies that incubated sediment cores (sed core) instead of placing chambers over the benthic component are noted.

Primary producer	Lake	GPP (mg O_2 m^{-2} hr^{-1})	Method	Location	Reference
Benthic community	Lake Tarawera	57 – 296	DO	New Zealand	This study
	Emerald lake	7 – 53	DO	USA	Sadro et al. 2011
	27 Swedish lakes	0 – 37	DIC (sed core)	Sweden	Puts et al. 2022
	Lake Erie	0 – 375	DO	Canada	Davies & Hecky 2005
	10 USA lakes	1 – 32	DO	USA	Craig et al. 2015
	5 USA lakes	14 – 47	DO (sed core)	USA	Devlin et al. 2016
	Lake Mývatn	0 – 700	DO (sed core)	Iceland	McCormick et al. 2020
Epiphyton	Lawrence Lake	28.7	^{14}C	USA	Allen. 1971
	Lake Okeechobee	< 1 to > 3	DO	USA	Havens et al. 1999
Microphytobenthos	Lake Eckarfjärden	0.1 – 384	^{14}C (sed core)	Sweden	Andersson & Brunberg 2006
Periphyton	Lake Lacawac	15 – 150	DO	USA	Fairchild & Everett 1988
	11 Greenland lakes	56	^{14}C (sed core)	Greenland	Vadeboncoeur et al. 2003
	Lake Okeechobee	50 – 250	DO	USA	Havens et al. 1999
	4 USA lakes	160	^{14}C (sed core)	USA	Vadeboncoeur et al. 2003
	12 Danish lakes	80	^{14}C (sed core)	Denmark	Vadeboncoeur et al. 2003
	Lake 239	14 – 26	DIC	Canada	Baulch et al. 2005
	Castle Lake	1 – 65	DO (sed core)	USA	Vander Zanden et al. 2006
	Lake Tanganyika	1.5 – 5	DO	Tanzania	McIntyre et al. 2006
	Bear Lake	400 – 1200	DO	USA	Page et al. 2022
	Lake Biwa	6 – 990	DO	Japan	Nozaki 2001

Spatial and temporal patterns of benthic GPP

Temporal patterns

Benthic GPP in Lake Tarawera was significantly higher in September and December (spring/summer) than in June (winter). Seasonal peaks during spring and summer have been observed in other studies and are often attributed to new algal growth (Nozaki, 2001; Andersson & Brunberg, 2006; Malkin et al., 2010; Périllon et al., 2017). However, seasonal patterns are not consistent across systems, and some studies report little seasonal variation in algal production (Hagerthey & Kerfoot, 1998; Page et al., 2022), winter peaks linked to nutrient delivery via higher groundwater discharge (Naranjo et al., 2019), or summer declines due to algal sloughing and self-shading (Nozaki, 2001; Malkin et al., 2010; Page et al., 2022). Localised processes, such as upwelling events caused by wind in Lake Tanganyika can also drive spikes in benthic GPP independent of environmental conditions (McIntyre et al., 2006). These studies indicate that temporal patterns in benthic GPP are primarily governed by benthic algae growth dynamics and localised nutrient supply, rather than season alone. In Lake Tarawera, elevated spring GPP at T5 and summer GPP at BVR suggests that both seasonal conditions and localised nutrient supply influenced production patterns.

Spatial patterns

Although spatial differences in benthic GPP were expected due to variation in benthic community composition, wave exposure, lakebed slope, and catchment land use, no significant difference between sites was detected ($p = 0.06$). Spatial variation in benthic GPP has been observed elsewhere and is often driven by vegetation structure, macrophyte species, and substrate type. For example, in Lake Wanaka, higher surface area on exotic macrophytes supported greater epiphyte production than native vegetation (Kelly & Hawes, 2005), and epiphyton productivity was higher on emergent macrophytes compared to submerged macrophytes in a Michigan lake (Allen, 1971). In Lake Erie, heterogeneous rocky substrate and macrophyte cover caused significant differences in GPP over 10 m scales (Davies & Hecky, 2005). Sediment can also enhance production through increased groundwater flow (Vadeboncoeur et al., 2003). In Lake Tarawera, substantial fine-scale variation in community assemblage, macrophyte coverage, periphyton growth, and substrate likely contributed to the large within-site and within-season variability in benthic GPP, obscuring broader spatial trends given the limited replication of measurements.

Drivers of benthic GPP

PERMANOVA models identified dry vegetation biomass as the strongest predictor of benthic GPP, with light availability only marginally significant. Biomass is rarely reported as a major predictor because it is seldom included in models. In contrast, previous studies have often identified light as the dominant driver of benthic GPP (Vadeboncoeur et al., 2001, 2003; Puts et al., 2022), while temperature is a key predictor in lakes that experience winter ice cover (Baulch et al., 2005; Andersson & Brunberg, 2006; Sadro et al., 2011; Puts et al., 2022). These studies of northern hemisphere systems are strongly controlled by seasonal temperature gradients and ice cover, conditions that differ from temperate New Zealand lakes such as Lake Tarawera and are therefore unlikely to be influential drivers of benthic GPP (Godwin et al., 2014).

The strong relationship between biomass and benthic GPP in this study likely reflects differences in macrophyte growth form rather than biomass per se. Sites with greater biomass were typically dominated by tall-growing macrophytes or periods of abundant epiphytes, meaning that biomass effectively acted as a proxy for species composition. Consequently, species-level differences in macrophyte community likely obscured ecological responses to external drivers such as light and nutrients.

Water column GPP

In control chambers, nearshore water column GPP ranged from 5.3 to 66.3 mg O_2 m^{-2} hr^{-1}. Few studies have measured nearshore water column metabolism, with reported GPP ranging from 0.02 mg O_2 m^{-2} hr^{-1} in an oligotrophic, high-elevation lake in California to 534 mg O_2 m^{-2} hr^{-1} in eutrophic Lake Michigan (Havens et al., 1999; Nozaki, 2001; Sadro et al., 2011; Althouse et al., 2014). Nearshore phytoplankton GPP in Lake Tarawera was comparable to that measured in mesotrophic Lake Biwa, Japan (3.6 to 58.7 mg O_2 m^{-2} hr

$^{-1}$), where dominance alternated between *Spirogyra* benthic mats and phytoplankton through time (Nozaki, 2001). In that study, phytoplankton contributed a relatively constant 25 % of total littoral production, similar to the mean contribution of water column GPP to total littoral GPP in Lake Tarawera (20.2 %).

These comparisons suggest that nearshore phytoplankton in Lake Tarawera contribute a larger fraction of littoral GPP than might be expected in a strictly oligotrophic system. Given that Lake Tarawera is on the cusp of mesotrophic classification (Trophic Level Index = 2.9; LAWA, n.d.-a), this elevated nearshore phytoplankton contribution reflects its current trophic position. Nevertheless, benthic producers clearly dominate littoral production at the spatial scale of this study, reinforcing their importance to whole-lake metabolism.

Autotrophic structure

Although water column GPP was not the primary focus of this study, it provides insight into the ecosystem balance within Lake Tarawera. Trophic status influences the autotrophic structure of the lake, defined as the relative contributions of benthic and pelagic production to whole-lake metabolism (Althouse et al., 2014). Benthic and pelagic GPP are often inversely related due to competition for light and nutrients between planktonic and benthic producers (Vadeboncoeur et al., 2003). In shallow, well-lit lakes, benthic productivity can dominate, exceeding 90 % of total lake GPP (Nozaki, 2001; McCormick et al., 2021), whereas high water column nutrient concentrations can shift dominance towards phytoplankton, reducing benthic GPP to less than 1 % during bloom events (Vadeboncoeur et al., 2003; McCormick et al., 2021). The dominance of benthic GPP over nearshore production in Lake Tarawera indicates that littoral conditions generally favour benthic producers, and that primary production is closely coupled to the sediment-water interface. This shows that the littoral community is currently resilient to groundwater nutrient inputs and can sequester nutrients within this zone, maintaining ecosystem composition and health. However, benthic algal growth during optimal growing conditions implies that tipping points in the autotrophic structure are possibly being reached in some localised areas of the littoral zone. In the future, increased pressure from altered climate conditions and increased nutrient delivery to this zone may increase benthic algae proliferation, which could in turn alter the balance and function of the whole benthic community and potentially increase the likelihood of nearshore planktonic algae blooms occurring.

Evidence of septic contamination

Septic contamination was hypothesised to increase nutrient availability in the littoral zone, enhancing benthic production. Although variability in benthic GPP and environmental conditions prevented formal testing of this relationship, substantial epiphytic algal growth observed at sites T5 and BVR during spring and summer may indicate elevated nutrient inputs at these locations. Similar patterns have been observed in other lakes affected by wastewater. For example, Hawes and Smith (1993) found that sewage effluent discharged into groundwater near Lake Taupō resulted in increased periphyton biomass directly downslope, while Timoshkin et al. (2018) reported that benthic *Spirogyra* blooms in Lake Baikal coincided with seasonal increases in visitor numbers to coastal villages reliant on unlined cesspools, as well as with enriched pore water nutrients in the lake sediment. While Lake Tarawera shows analogous patterns, the present dataset does not allow definitive attribution of epiphytic growth to septic contamination.

Future research suggestions

This study presents the first application of chamber incubations using the DO flux method to estimate benthic GPP in a New Zealand lake and provides a foundation for future work. However, several methodological refinements could strengthen future studies.

Seasonal monitoring captured substantial variation in environmental drivers, but a single sampling event per season was insufficient to fully characterise benthic GPP dynamics. Higher-frequency monitoring (e.g. weekly) during the spring-summer growth period may better capture the growth and decay of epiphytic and periphytic algae through their life cycle.

Aggregating site and seasonal means for statistical analyses reduced the retention of raw variability and limited multivariate inference. Future studies with more frequent monitoring could use raw data, thereby improving statistical power and allowing more robust exploration of environmental drivers. Incorporating morphological and biochemical indicators of benthic producers (e.g. C, N, P, and chl-*a* content) would also improve understanding of how benthic communities respond to environmental variation.

Separating benthic community components during incubations, as demonstrated by Kelly and Hawes (2005), would clarify the contributions of macrophytes, microphytobenthos, periphyton, and epiphyton to total benthic production, and how these contributions vary through time. Finally, concurrent measurements of septic indicators (e.g. pore water nutrients and sucralose) alongside GPP incubations would enable more robust testing of relationships between nutrient inputs and benthic metabolism.

Conclusion

This study provides the first estimates of benthic community production in the littoral zone of Lake Tarawera using the dissolved oxygen method in benthic chambers. Whole community GPP was highest in spring and summer compared to winter, with vegetation biomass identified as the main driver of variation in benthic production. Water column GPP contributed approximately 20 % of total littoral community GPP, indicating that primary production in the littoral zone is strongly dominated by benthic producers.

Substantial growth of epiphytic algae occurred at sites T5 and BVR during spring and summer, respectively. This may indicate localised nutrient enrichment consistent with groundwater inputs from septic sources, although this relationship could not be empirically tested within the constraints of the study design. Despite high variability across sites and seasons, benthic production rates fell within the range expected for an oligotrophic system with extensive macrophyte coverage, and the influence of nutrient enrichment from the urban settlement on ecosystem function appeared to be relatively minor.

These findings establish an important baseline for littoral production in Lake Tarawera and demonstrate the dominant role of benthic producers in the autotrophic structure of the littoral zone in an oligotrophic, temperate lake. This research demonstrates a viable alternative approach for future work in New Zealand lakes seeking to understand both the role of benthic production and how anthropogenic pressures influence littoral ecosystem functioning.

Chapter 5

General discussion and conclusions

Nutrient enrichment is the leading driver of lake ecosystem degradation worldwide, causing eutrophication, anoxic conditions, habitat decline and loss of biodiversity (Schindler & Vallentyne, 2008). Among nutrient sources, on-site wastewater treatment systems (OWTS) are recognised as important contributors to nutrient loading to lakes, especially where groundwater provides a direct transport pathway to the littoral zone (Withers et al., 2014). Despite this recognition, the spatial extent, transport dynamics, and ecological consequences of septic-derived nutrient inputs to lakes remain poorly understood (Ray, 2007; Withers et al. 2014), especially in oligotrophic systems where early impacts of these nutrients may be subtle yet ecologically significant (Vadeboncoeur et al., 2021).

Key findings

The extent, fate, and ecological impact of nutrients from septic systems at Lake Tarawera were determined by studying nutrient concentrations in shallow groundwater and pore water, groundwater fluxes to the littoral zone, stable nitrate isotopes in water and primary producer samples.

Monitoring of shallow groundwater at 21 sites during April 2023 – March 2025 along the Spencer Road settlement found shallow groundwater nutrient concentrations to be generally within a normal range for New Zealand soils; however, localised areas of elevated ammoniacal-N (up to 61 mg L^{-1}) and nitrate-N (up to 7.4 mg L^{-1}) occurred within the urban zone. This is likely indicative of septic-derived contamination, especially as these areas had nutrient concentrations consistent with those observed in other septic-influenced lake catchments globally (Harman et al., 1996; Ouyang & Zhang, 2012; Robertson et al., 2019).

Littoral groundwater seepage chambers were used to measure groundwater inflow and rates were compared with previous rainfall measurements from a virtual climate station. The results confirmed the hydrological connection between the urban catchment and the littoral zone with a positive correlation between 1-month previous rainfall and groundwater inflow ($\rho = 0.48$). Additionally, measurements of groundwater and pore water nutrients and nitrate stable isotopes revealed enriched nitrate $\delta^{15}N$ occurring in both piezometer and pore water samples (>6 ‰), which is typically associated with a sewage source of nitrogen (Kendall et al., 2007). However, it appears that septic-derived nutrient delivery to the lake is highly attenuated in the sediment, with pore water nutrient concentrations lying within the typical range for other urban nearshore sediments in similar New Zealand lakes (Gibbs et al., 2005; Hector, 2004).

Estimates of benthic GPP in Lake Tarawera using paired light and dark chambers to measure dissolved oxygen flux from lakebed communities revealed that highest GPP occurred in spring and summer and were comparable to production in other lakes globally. Benthic producers dominated nearshore primary production rates (80 % of whole community GPP) compared to water column producers, although there was evidence that this balance could tip towards water column dominance during warmer months. A key finding was that sites with extensive epiphytic algal growth produced the highest GPP rates (up to 296 mg O_2 m^{-2} hr^{-1}), with these littoral sites coinciding with either known nutrient hotspots in upslope piezometers, enriched $\delta^{15}N$ values in upslope piezometers and/or pore water, or a combination of both. These correlations show that, potentially, extensive benthic algal growth was caused by localised increases in nutrient delivery due to septic contamination.

Comparative case studies of septic-driven littoral algae dominance

The link between septic-derived nutrient contamination and algal blooms is well-established worldwide, though few studies have undertaken a multidisciplinary approach to assessing the septic source–groundwater–surface water ecological pathway. Lake Baikal is one of a few lakes where such studies have been conducted, due to its high ecological significance and the rapid development of nearshore algal blooms a decade ago (Timoshkin et al., 2018). Extensive benthic *Spirogyra* mats in Lake Baikal have been found to occur along shoreline areas with urban settlements and are absent in unsettled areas (Timoshkin et al., 2018; Meyer et al., 2022a). Local groundwater beneath the blooms is enriched with nutrients, with *Enterococci* and *Escherichia coli* also abundant (>100 CFU mL^{-1}), likely originating from unlined sewage cesspool leachate. Additionally, *Spirogyra* has a seasonal growth pattern that coincides with the seasonal influx of visitors during the warmer months (Timoshkin et al., 2018).

Comparable processes, localised to areas of high nutrient loading to groundwater, are likely occurring in Lake Tarawera, as indicated by downslope enriched $\delta^{15}N$ values in pore water and nearshore epiphytic algal growth in warmer months. However, the magnitude of the ecological impact in Tarawera is significantly less than that observed in Lake Baikal, likely due to the overall nutrient loading from septic systems at Lake Tarawera to be significantly less. Settlements along the western shoreline of Lake Baikal are, on average, much larger than Lake Tarawera's settlement (population = ~291 permanent residents), with populations ranging between <300 and >10,000 people (Timoshkin et al., 2018; Meyeret al., 2022a). Many of these settlements have expanded rapidly due to increased tourism as well as illegal building developments along the shoreline relying solely on unlined cesspools (Timoshkin et al., 2018). While Lake Tarawera is unlikely to experience the rapid development of extensive algal mats to the scale seen in Lake Baikal in the immediate future due to less nutrient loading pressure from its settlement and controlled development of the urban zone, continued nutrient loading to groundwater from other sources may increase the frequency and extent of nearshore algal growth over time.

Although Tarawera's impacts are far less pronounced than those observed in Lake Baikal, the elevated groundwater nutrient concentrations, isotopic evidence of septic-derived nitrogen reaching the littoral zone, and the occurrence of extensive epiphytic algal growth during warmer months at impacted sites, indicate that Lake Tarawera is exhibiting measurable ecological changes likely caused by septic leachate. Lake

Tarawera likely represents an early stage of ecological change, preceding the tipping point towards algal dominance observed in more heavily impacted systems elsewhere.

Implications

A nationwide need for wastewater reticulation schemes

While septic systems are widely treated in New Zealand as an “out of sight, out of mind” way of treating wastewater and minimising the impact on the environment, the findings at Lake Tarawera demonstrate that even a small community can generate measurable nutrient loading to shallow groundwater, especially when systems are ageing or performing poorly. A 2001 survey of the Tarawera lakeside community found that 65 % of inspected septic systems failed to meet effluent treatment standards (Scholes, 2007). Nationally, an estimated 400,000 people are reliant on OWTS, with approximately 42,000 households experiencing system failure, yet these figures could be higher due to a lack of consistent regional or national monitoring of system performance (Ray, 2007; Cocks et al., 2024). The situation at Lake Tarawera therefore likely reflects a more systemic and underestimated issue in New Zealand.

Wastewater reticulation is increasingly being implemented as a management response to reduce this risk to freshwater ecosystems. The Spencer Road settlement at Lake Tarawera is one of the last lakes in the Te Arawa region to be reticulated, highlighting how slowly such schemes are implemented, often only after environmental degradation has already been detected. Evidence from Lake Ōkāreka and Lake Tikitapu suggests that reticulation may contribute to improved or stabilised lake trophic state, although reticulation at Ōkāreka was implemented concurrently with wetland establishment, land use change, and alum dosing, making it difficult to isolate the effect of reticulation on water quality (Taylor et al., 2008; LAWA, n.d.-d). The absence of before-after control-impact (BACI) studies of groundwater nutrients elsewhere in New Zealand, limits the ability to quantify the specific reduction of nutrients from reticulation. Only one study by Gibbs (1991) has undertaken a BACI-style study for sewage reticulation in New Zealand. During a 15-year post-reticulation study of the Lake Taupō township reticulation scheme, the author observed sudden drops in groundwater dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) concentrations in the years post-reticulation, by as much as 60 % for DIN (Gibbs, 1991). Elsewhere, Robertson and Harman (1999) monitored groundwater contaminant concentrations in a calcareous sand aquifer downgradient of a septic system for 4-years post-decommissioning, finding that nitrate and other major ions indicative of sewage had decreased significantly to background levels near the septic system and had migrated 60 – 100 m downgradient within the first year. However, phosphate concentrations remained largely unchanged, which was further reinforced in a later study in a similar groundwater setting at Lake Huron, where DRP concentrations remained persistently high more than 30 years after septic system decommissioning (Roy et al., 2017). Legacy phosphorus loading is less likely to be a substantial risk to Lake Tarawera as volcanic ash-derived soils in the catchment exhibit medium to very high phosphorus retention capacities due to the presence of allophane (Saunders, 1965), which limits phosphorus mobility. Regardless, having empirical evidence of the efficacy of these schemes in New Zealand in reducing all sewage contaminants would strengthen the case for implementing them and managing the lasting effects of septic systems.

Internationally, New Zealand has adopted a less consistent and generally later approach to the regulation and monitoring of OWTS than many other developed countries. For example, in the USA, water quality issues associated with wastewater pollution were recognised decades earlier, resulting in regulatory frameworks and requirements for system design, placement, inspection, and maintenance being put in place in the early 2000s (U.S. Environmental Protection Agency, 2002). By comparison, only within the last decade have some New Zealand regional councils introduced compliance monitoring or minimum function standards for OWTS (Auckland Council, 2022; Bay of Plenty Regional Council, n.d.). Recently, new *Water Services (Wastewater Environmental Performance Standards) Regulations 2025* took effect, which aim to ease consenting costs for upgrading wastewater infrastructure and apply stricter controls on wastewater contaminant treatment and their discharge to sensitive water bodies, including lakes. However, these standards do not apply to privately owned septic systems, continuing to leave responsibility for their wastewater treatment to local governing bodies.

The results from the current study support recent calls by Cocks et al. (2024) to increase awareness of the environmental and public health risks posed by poorly performing OWTS. Septic impacts are notoriously difficult to detect, but when identified, they are substantial and likely more significant in extent and magnitude than currently recognised. For rural and lakeside communities exceeding certain density thresholds, reticulation represents a lower-risk wastewater management option than on-site systems.

Littoral monitoring in New Zealand

The increasing frequency and magnitude of filamentous algal blooms in clear water lakes globally (Vadeboncoeur et al., 2021), and evidence of this phenomenon occurring at Lake Tarawera, suggests that the current approaches used to assess littoral ecosystem health in New Zealand lakes may need to be redesigned to capture early ecosystem change. The ecological condition of the littoral zone is assessed using the Lake Submerged Plant Indicator (LakeSPI) index, developed by NIWA (Earth Sciences NZ from 2025; Clayton & Edwards, 2006). LakeSPI relies on scuba observations of the relative presence, abundance, and depth distribution of native and invasive macrophytes along a littoral transect, with macrophyte depths used as a proxy for water clarity and thereby water column health (Clayton & Edwards, 2006). While macrophytes are valuable integrators of long-term ecological condition, this approach assumes that changes in water quality and environmental pressures will be evident in changes to macrophyte community structure. The results from Lake Tarawera suggest that LakeSPI is not sufficiently sensitive to detect early ecosystem changes.

Substantial increases in epiphytic algal growth were observed at two sites during spring and summer, despite no obvious change in water quality. A similar observation has occurred in an alpine lake in New Zealand (Māori Lake), where the macrophyte community collapsed and was replaced by filamentous algal mats, despite relatively low water column TP (Stewart et al., 2021). Pore water DRP concentrations of >10 mg L^{-1} caused by high producing grassland in the catchment, alongside reduced grazer pressure was theorised to have caused state transition (Stewart et al., 2021; Vadeboncoeur et al., 2021). This trend of increasing filamentous algal blooms has been brought to attention by Vadeboncoeur et al. (2021) who document the rise in benthic filamentous algal blooms globally, including Lake Tahoe (USA), Lake Baikal, Laurentian Great Lakes (Erie, Ontario, Michigan), and Lake Tianchi (China). With the current use of LakeSPI in New Zealand, early shifts in autotrophic structure from a macrophyte-dominated to algae-dominated littoral community may go undetected until macrophytes themselves begin to decline, at which point an ecological tipping point is likely to already be passed. By missing the first signs of ecological change, LakeSPI does not allow for proactive management, only reactive.

Incorporating periphyton metrics into lake monitoring has the potential to improve sensitivity and detect early-stage ecosystem changes as they occur. Periphyton is increasingly recognised as a more sensitive and rapid responder to nutrient enrichment (Vadeboncoeur et al., 2021) than macrophytes and may be the critical missing link in littoral monitoring (Timoshkin et al., 2018). In some US programs, periphyton is already used as an indicator of nearshore nutrient pressure in lakes (Stevenson & Bahls, 1999; Tahoe Envrironmental Research Center, n.d.), while in New Zealand, well-established and standardised methods already exist for monitoring periphyton in streams (Biggs, 2000). A targeted monitoring approach could therefore prioritise littoral areas where groundwater contamination is suspected, incorporating periphyton biomass and GPP measurements alongside existing metrics to detect localised nutrient enrichment.

Impact on the wider food web

Increased nutrient delivery to littoral zones can alter the algal community composition present in favour of more opportunistic filamentous taxa. For example, shifts towards filamentous algae in Lake Baikal and Māori Lake resulted in the altering of grazer diets and behaviour when more palatable taxa were replaced (Timoshkin et al., 2018; Stewart et al., 2021). Invertebrates have also been observed to switch towards filter feeding on phytoplankton when nutritional periphyton are replaced with filamentous taxa (Vadeboncoeur et al., 2003; Meyeret al., 2022a). These changes at the benthic primary producer level can propagate through the wider lake food web. In many lakes, fish diets can be subsidised substantially by benthic production, in some cases by over 50 %

(Vander Zanden et al., 2006). If nutrient loading to the littoral zone exceeds a critical threshold, benthic primary production can decline, potentially driving a transition to a phytoplankton-dominated, turbid state and the loss of macrophytes (Vadeboncoeur & Power, 2017). Changes to basal resources in Lake Tarawera may fundamentally alter the wider food web, with effects propagating up to the primary and secondary consumer level (e.g. trout). These potential effects on the food web further reinforce the need for more sensitive monitoring of littoral zone changes, especially before a tipping point in community structure is reached.

Limitations

Several aspects of the study design and dataset constrained the strength of inference and interpretation. Shallow groundwater sampling success was spatially and temporally heterogeneous, influenced by prolonged subsurface saturation following Cyclone Gabrielle in February 2023 and the likely presence of preferential subsurface flow paths that did not always align with piezometer placement. As a result, formal testing of spatial and temporal patterns was not possible. This limited the ability to quantify temporal contaminant loading and to robustly compare between different sites.

The absence of additional septic-specific tracers (e.g. *E. coli*, sucralose, major ions) in groundwater were unable to be included for logistical reasons, which further limits definitive attribution of elevated nutrient concentrations to septic system leachate. While nutrient enrichment and isotopic signatures provide strong potential evidence, direct source confirmation would have strengthened these interpretations.

Stable isotope analyses and pore water measurements were conducted at a single point in time, capturing only a snapshot of potential contamination. Although these data offered critical lines of evidence for septic-derived contamination reaching the sediment–water interface, repeated sampling would have improved understanding of nutrient loading rates along the extent of the urban zone through time.

Benthic GPP measurements were undertaken only once per season at each site, meaning metabolic rates reflected short-term conditions rather than seasonal means. Differences in algal growth stage at the time of incubation may have influenced the dominance of net respiration or production, particularly where periphyton was a major component of the littoral community. In addition, whole community GPP was measured without partitioning contributions of macrophytes, periphyton, and microphytobenthos, which further limited interpretation of which functional groups were driving productivity patterns. Consequently, while these measurements provide insight into seasonal metabolism at each site, they do not capture how different autotrophic components respond to nutrient enrichment and changing environmental conditions.

Future research directions

Further investigations of how increases in benthic algal production influence primary consumers in the littoral zone, especially taonga species such as kākahi (freshwater mussels, *Echyridella menziesii*) and kōura (freshwater crayfish, *Paranephrops planifrons*; Kusabs, 2015), would assist in determining the impacts of benthic production changes on food webs. Understanding whether shifts from periphyton to filamentous algal dominance alter grazer diets, behaviour, condition, abundance, and habitat quality would clarify how nutrient enrichment propagates through the littoral food web.

Expanding benthic chamber monitoring to a regional scale would also be valuable. A broader trophic gradient would provide insight into how benthic primary production, algal community composition, and autotrophic structure vary with increasing nutrient availability. Characterising the dominant benthic algal forms as well as their spatial and temporal extent across lakes could help identify which systems have crossed, or are approaching, critical ecological thresholds towards phytoplankton or filamentous benthic algae dominance. In addition, this would contribute to the body of data on benthic primary production in southern hemisphere lakes, which is currently scarce.

Concluding perspectives

This thesis examined the spatial and temporal extent, transport, and ecological impact of septic-derived nutrient inputs to Lake Tarawera by tracing nutrient contaminants from shallow groundwater in the urban settlement to primary producer responses in the adjacent littoral zone. Multiple lines of evidence revealed that Lake Tarawera is likely in the early stages of ecological change along its urbanised western margin, driven by nutrient inputs from OWTS. Although the system does not appear close to a critical tipping point, the occurrence of substantial algal growth during warm months signals shifts in the littoral community structure that could become more frequent and consequential in the future if anthropogenic nutrient inputs are not controlled. The timing of the Tarawera reticulation scheme is therefore significant: although not quantified, there are indications that nutrient contaminants from septic systems are reaching the lake and early impacts such as alterations to the littoral community may be occurring.

However, reticulation addresses only one pressure within a system threatened by multiple other nutrient sources across both inner and outer catchment systems. Lessons from nearby Te Arawa lakes, including Lakes Ōkaro, Rotoehu and Rotoiti, demonstrate the ecological and economic cost of delayed intervention once lake state has shifted. By recognising and responding to early signs of ecological change, proactive and preventative management offers the opportunity to protect the current ecological, cultural, economic, and intrinsic values of Lake Tarawera, for now and for future generations.

References

Abell, J. M., McBride, C. G., & Baisden, W. T. (2020). *Assessing effects of changes to nutrient loads on Lake Tarawera water quality: Model simulations for 2010 to 2020* (ERI Report 144). Environmental Research Institute, University of Waikato. https://hdl.handle.net/10289/16727

Abell, J. M., Özkundakci, D., & Hamilton, D. P. (2010). Nitrogen and phosphorus limitation of phytoplankton growth in New Zealand lakes: Implications for eutrophication control. *Ecosystems*, *13*(7), 966-977. https://doi.org/10.1007/s10021-010-9367-9

Afolalu, S. A., Ikumapayi, O. M., Ogedengbe, T. S., Kazeem, R. A., & Ogundipe, A. T. (2022). Waste pollution, wastewater and effluent treatment methods – An overview. *Materials Today: Proceedings*, *62*(6), 3282-3288. https://doi.org/10.1016/j.matpr.2022.04.231

Allen, H. L. (1971). Primary productivity, chemo-organotrophy, and nutritional interactions of epiphytic algae and bacteria on macrophytes in the littoral of a lake. *Ecological Monographs*, *41*(2), 97-127. https://doi.org/10.2307/1942387

Althouse, B., Higgins, S., & Vander Zanden, M. J. (2014). Benthic and planktonic primary production along a nutrient gradient in Green Bay, Lake Michigan, USA. *Freshwater Science*, *33*(2), 487-498. https://doi.org/10.1086/676314

Anderson, M. J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology, 26*(1), 32-46. https://doi.org/10.1111/j.1442-9993.2001.01070.pp.x

Andersson, E., & Brunberg, A.-K. (2006). Net autotrophy in an oligotrophic lake rich in dissolved organic carbon and with high benthic primary production. *Aquatic Microbial Ecology*, *43*(1), 1-10. https://doi.org/10.3354/ame

Aravena, R., Evans, M., & Cherry, J. (1993). Stable isotopes of oxygen and nitrogen in source identification of nitrate from septic systems. *Groundwater*, *31*(2), 180-186. https://doi.org/10.1111/j.1745-6584.1993.tb01809.x

Aravena, R., & Robertson, W. D. (1998). Use of multiple isotope tracers to evaluate denitrification in ground water: Study of nitrate from a large-flux septic system plume. *Groundwater*, *36*(6), 975-982. https://doi.org/10.1111/j.1745-6584.1998.tb02104.x

Attanayake, M. P., & Waller, D. H. (1988). Use of seepage meters in a groundwater – lake interaction study in a fractured rock basin — A case study. *Canadian Journal of Civil Engineering*, *15*(6), 984-989. https://doi.org/10.1139/l88-130

Auckland Council. (2022, May 25). *Get your septic system checked - compliance is coming in June 2022*. https://ourauckland.aucklandcouncil.govt.nz/news/2022/05/get-your-septic-system-checked-compliance-is-coming-in-june-2022/

Bacchus, S., & Barile, P. (2005). Discriminating sources and flowpaths of anthropogenic nitrogen discharges to Florida springs, streams and lakes. *Environmental & Engineering Geoscience*, *11*(4), 347-369. https://doi.org/10.2113/11.4.347

Baer, S., Robertson, W., Spoelstra, J., & Schiff, S. (2019). Phosphorus and nitrogen loading to Lake Huron from septic systems at Grand Bend, ON. *Journal of Great Lakes Research*, *45*(3), 642-650. https://doi.org/10.1016/j.jglr.2019.03.003

Baulch, H., Schindler, D., Turner, M., Findlay, D., Paterson, M., & Vinebrooke, R. (2005). Effects of warming on benthic communities in a boreal lake: Implications of climate change. *Limnology and Oceanography*, *50*(5), 1377-1392. https://doi.org/10.4319/lo.2005.50.5.1377

Bay of Plenty Regional Council. (2015). *Tarawera Lakes Restoration Plan* (Environmental Report 2014/12). https://www.rotorualakes.co.nz/vdb/document/1049

Bay of Plenty Regional Council. (2025). *Lake Level.Moturiki@FJ567596* [Data set]. Environment Data Portal. Retrieved October 7, 2025, from https://envdata.boprc.govt.nz/Data/DataSet/Chart/Location/FJ567596/DataSet/Lake%20Level/Moturiki/Interval/Custom/2022/10/01/2025/04/01

Bay of Plenty Regional Council. (n.d.). *Managing your wastewater: Building, buying and living in a property with an on-site wastewater treatment system*. https://atlas.boprc.govt.nz/api/v1/edms/document/A3483812/content

Benson, E. R., O'Neil, J. M., & Dennison, W. C. (2008). Using the aquatic macrophyte *Vallisneria americana* (wild celery) as a nutrient bioindicator. *Hydrobiologia*, *596*(1), 187-196. https://doi.org/10.1007/s10750-007-9095-0

Biggs, B. J. (1990). Periphyton communities and their environments in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research*, *24*(3), 367-386. https://doi.org/10.1080/00288330.1990.9516431

Biggs, B. J. (2000). *New Zealand periphyton guideline: Detecting, monitoring and managing enrichment of streams*. Ministry for the Environment. https://www.environment.govt.nz/assets/Publications/Files/nz-periphyton-guide-jun00.pdf

Boffa Miskell Limited. (2020). *Rotorua Te Arawa lakes: Aquatic pest management plan*. https://lakeswaterquality.co.nz/wp-content/uploads/documents/Rotorua-Lakes-Aquatic-Plant-Management-Plan-May-2020.pdf

Caratori Tontini, F. C., de Ronde, C., Black, J., Stucker, V., & Walker, S. (2023). The geology and geophysics of Lake Tarawera, New Zealand: Implications for sublacustrine geothermal activity. *Journal of Volcanology and Geothermal Research*, *433*, 107731. https://doi.org/10.1016/j.jvolgeores.2022.107731

Chen, Y., Zhao, Z., Wang, F., Ye, C., Zhang, M., Li, C., & Wang, Y. (2025). Deciphering anthropogenic nutrient impacts on aquatic eutrophication through coupled nitrate isotope tracers and hydrogeochemical fingerprints. *Science of the Total Environment*, *1001*, 180473. https://doi.org/10.1016/j.scitotenv.2025.180473

Clayton, J., & Edwards, T. (2006). *LakeSPI: A method for monitoring eological condition in New Zealand lakes* (User manual, Version 2; CRBV062). NIWA. https://niwa.co.nz/sites/default/files/import/attachments/lakespi_manual.pdf

Cochrane, K.-L. M. N. (2020). *The presence, speciation, and movement of arsenic in Lake Tarawera* [Master's thesis, University of Waikato]. University of Waikato Research Commons. https://hdl.handle.net/10289/14049

Cocks, J., Dakers, A., & Lowe, H. (2024). *On-site wastewater management systems risk and mitigation*. Water New Zealand. https://www.waternz.org.nz/Attachment?Action=Download&Attachment_id=6426

Cole, M. L., Valiela, I., Kroeger, K. D., Tomasky, G. L., Cebrian, J., Wigand, C., McKinney, R. A., Grady, S. P., & Carvalho da Silva, M. H. (2004). Assessment of a $\delta^{15}N$ isotopic method to indicate anthropogenic eutrophication in aquatic ecosystems. *Journal of Environmental Quality*, *33*(1), 124-132. https://doi.org/10.2134/jeq2004.1240

Collins, S., Singh, R., Rivas, A., Palmer, A., Horne, D., Manderson, A., Roygard, J., & Matthews, A. (2017). Transport and potential attenuation of nitrogen in shallow groundwaters in the lower Rangitikei catchment, New Zealand. *Journal of Contaminant Hydrology*, *206*, 55-66. https://doi.org/10.1016/j.jconhyd.2017.10.002

Craig, N., Jones, S. E., Weidel, B. C., & Solomon, C. T. (2015). Habitat, not resource availability, limits consumer production in lake ecosystems. *Limnology and Oceanography*, *60*(6), 2079-2089. https://doi.org/10.1002/lno.10153

Cui, R., Zhang, D., Hu, W., Zhao, X., Yan, H., Liu, G., & Chen, A. (2023). Nitrogen in soil, manure and sewage has become a major challenge in controlling nitrate pollution in groundwater around plateau lakes, Southwest China. *Journal of Hydrology*, *620*(Part B), 129541. https://doi.org/10.1016/j.jhydrol.2023.129541

Dada, A. C., McBride, C. G., Verburg, P., & Hamilton, D. (2016). *Modelling the impact of sewage reticulation on water quality of Lake Tarawera* (ERI Report 85). Environmental Research Institute, University of Waikato. https://hdl.handle.net/10289/12466

Daughney, C. J., Morgenstern, U., Moreau, M., & McDowell, R. W. (2025). Reference conditions and threshold values for nitrate-nitrogen in New Zealand groundwaters. *Journal of the Royal Society of New Zealand*, *55*(1), 1-31. https://doi.org/10.1080/03036758.2023.2221034

Davies, J.-M., & Hecky, R. E. (2005). Initial measurements of benthic photosynthesis and respiration in Lake Erie. *Journal of Great Lakes Research*, *31*(Suppl. 2), 195-207. https://doi.org/10.1016/S0380-1330(05)70314-2

de Brabandere, L., Frazer, T. K., & Montoya, J. P. (2007). Stable nitrogen isotope ratios of macrophytes and associated periphyton along a nitrate gradient in two subtropical, spring-fed streams. *Freshwater Biology*, *52*(8), 1564-1575. https://doi.org/10.1111/j.1365-2427.2007.01788.x

Devlin, S. P., Vander Zanden, M. J., & Vadeboncoeur, Y. (2016). Littoral-benthic primary production estimates: Sensitivity to simplifications with respect to periphyton productivity and basin morphometry. *Limnology and Oceanography: Methods*, *14*(2), 138-149. https://doi.org/10.1002/lom3.10080

Downing, J. A., & Peterka, J. J. (1978). Relationship of rainfall and lake groundwater seepage. *Limnology and Oceanography*, *23*(4), 821-825. https://doi.org/10.4319/lo.1978.23.4.0821

du Plessis, A. (2022). Persistent degradation: Global water quality challenges and required actions. *One Earth*, *5*(2), 129-131. https://doi.org/10.1016/j.oneear.2022.01.005

Earth Sciences New Zealand. (2025). *VCSN timeseries data, Agentno: 27368, location: [-38.175,176.4225]* [Data set]. Retrieved October 10, 2025, from https://data.niwa.co.nz/products/vcsn-timeseries/files/675b79fb47ec2f9a228253b0

Eser, P., & Rosen, M. (1999). The influence of groundwater hydrology and stratigraphy on the hydrochemistry of Stump Bay, South Taupo Wetland, New Zealand. *Journal of Hydrology*, *220*(1-2), 27-47. https://doi.org/10.1016/S0022-1694(99)00062-1

Esri. (2025). *ArcGIS Pro* (Version 3.5) [Computer software]. https://www.esri.com/en-us/arcgis/products/arcgis-pro/

Fairchild, G.W., & Everett, A.C. (1988). Effects of nutrient (N, P, C) enrichment upon periphyton standing crop, species composition and primary production in an oligotrophic softwater lake. *Freshwater Biology, 19*(1), 57-70. https://doi.org/10.1111/j.1365-2427.1988.tb00327.x

Gibbs, M. (1991). Nutrient concentration changes in the ground water beneath Taupo township following sewage reticulation. *New Zealand Journal of Marine and Freshwater Research*, *25*(2), 153-161. https://doi.org/10.1080/00288330.1991.9516465

Gibbs, M. M., Clayton, J. S., & Wells, R. (2005). *Further investigation of direct groundwater seepage to Lake Taupo* (TR 2005/34). Environment Waikato. https://www.waikatoregion.govt.nz/services/publications/tr200534/

Godwin, S. C., Jones, S. E., Weidel, B. C., & Solomon, C. T. (2014). Dissolved organic carbon concentration controls benthic primary production: Results from in situ chambers in north-temperate lakes. *Limnology and Oceanography*, *59*(6), 2112-2120. https://doi.org/10.4319/lo.2014.59.6.2112

Gyimah, R., Lebu, S., Owusu-Frimpong, I., Semiyaga, S., Salzberg, A., & Manga, M. (2024). Effluents from septic systems and impact on groundwater contamination: A systematic review. *Environmental Science and Pollution Research*, *31*(54), 62655-62675. https://doi.org/10.1007/s11356-024-35385-1

Hagerthey, S. E., & Kerfoot, W. C. (1998). Groundwater flow influences the biomass and nutrient ratios of epibenthic algae in a north temperate seepage lake. *Limnology and Oceanography*, *43*(6), 1227-1242. https://doi.org/10.4319/lo.1998.43.6.1227

Hall, R. O., Thomas, S., & Gaiser, E. E. (2007). Measuring freshwater primary production and respiration. In A. K. Knapp & T. J. Fahey (Eds.), *Principles and standards for measuring primary production* (pp. 175-203). Oxford University Press.

Hamilton, D. P., Hamilton, M., & McBride, C. G. (2006). *Nutrient water budget for Lake Tarawera* (CBER Contract Report 46). Environmental Research Institute, University of Waikato. https://hdl.handle.net/10289/16735

Harman, J., Robertson, W., Cherry, J., & Zanini, L. (1996). Impacts on a sand aquifer from an old septic system: Nitrate and phosphate. *Groundwater*, *34*(6), 1105-1114. https://doi.org/10.1111/j.1745-6584.1996.tb02177.x

Havens, K. E., East, T. L., Hwang, S. J., Rodusky, A. J., Sharfstein, B., & Steinman, A. D. (1999). Algal responses to experimental nutrient addition in the littoral community of a subtropical lake. *Freshwater Biology*, *42*(2), 329-344. https://doi.org/10.1046/j.1365-2427.1999.444479.x

Hawes, I., & Smith, R. (1993). Effect of localised nutrient enrichment on the shallow epilithic periphyton of oligotrophic Lake Taupo, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *27*(3), 365-372. https://doi.org/10.1080/00288330.1993.9516576

Hawes, I., & Smith, R. (1994). Seasonal dynamics of epilithic periphyton in oligotrophic Lake Taupo, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *28*(1), 1-12. https://doi.org/10.1080/00288330.1994.9516592

Hector, R. P. (2004). *Investigation of direct groundwater and nutrient seepage into Lake Taupo* [Unpublished master's thesis]. University of Waikato.

Hessen, D. O., Andersen, T., Armstrong McKay, D., Kosten, S., Meerhoff, M., Pickard, A., & Spears, B. (2024). Lake ecosystem tipping points and climate feedbacks. *Earth System Dynamics, 15*(3), 653-669. https://doi.org/10.5194/esd-15-653-2024

Hoghooghi, N., Radcliffe, D. E., Habteselassie, M. Y., & Clarke, J. S. (2016). Confirmation of the impact of onsite wastewater treatment systems on stream base-flow nitrogen concentrations in urban watersheds of metropolitan Atlanta, GA. *Journal of Environmental Quality*, *45*(5), 1740-1748. https://doi.org/10.2134/jeq2016.04.0139

Iverson, G., Humphrey Jr, C., O'Driscoll, M., Sanderford, C., Jernigan, J., & Serozi, B. (2018). Nutrient exports from watersheds with varying septic system densities in the North Carolina Piedmont. *Journal of Environmental Management*, *211*, 206-217. https://doi.org/10.1016/j.jenvman.2018.01.063

John, P. H., & Lock, M. A. (1977). The spacial distribution of groundwater discharge into the littoral zone of a New Zealand lake. *Journal of Hydrology*, *33*(3-4), 391-395. https://doi.org/10.1016/0022-1694(77)90049-X

Jung, H., Koh, D.-C., Kim, Y. S., Jeen, S.-W., & Lee, J. (2020). Stable isotopes of water and nitrate for the identification of groundwater flowpaths: A review. *Water*, *12*(1), 138. https://doi.org/10.3390/w12010138

Katz, B. G., Eberts, S. M., & Kauffman, L. J. (2011). Using Cl/Br ratios and other indicators to assess potential impacts on groundwater quality from septic systems: A review and examples from principal aquifers in the United States. *Journal of Hydrology*, *397*(3-4), 151-166. https://doi.org/10.1016/j.jhydrol.2010.11.017

Kazmierczak, J., Müller, S., Nilsson, B., Postma, D., Czekaj, J., Sebok, E., Jessen, S., Karan, S., Stenvig Jensen, C., & Edelvang, K. (2016). Groundwater flow and heterogeneous discharge into a seepage lake: Combined use of physical methods and hydrochemical tracers. *Water Resources Research*, *52*(11), 9109-9130. https://doi.org/10.1002/2016WR019326

Kelly, D. J., & Hawes, I. (2005). Effects of invasive macrophytes on littoral-zone productivity and foodweb dynamics in a New Zealand high-country lake. *Journal of the North American Benthological Society*, *24*(2), 300-320. https://doi.org/10.1899/03-097.1

Kendall, C., Elliott, E. M., & Wankel, S. D. (2007). Tracing anthropogenic inputs of nitrogen to ecosystems. In R. Michener & K. Lajtha (Eds.), *Stable isotopes in ecology and environmental science* (2nd ed., pp. 375-449). Blackwell Publishing. https://doi.org/10.1002/9780470691854

Khatri, N., & Tyagi, S. (2015). Influences of natural and anthropogenic factors on surface and groundwater quality in rural and urban areas. *Frontiers in Life Science*, *8*(1), 23-39. https://doi.org/10.1080/21553769.2014.933716

Kidmose, J., Engesgaard, P., Nilsson, B., Laier, T., & Looms, M. C. (2011). Spatial distribution of seepage at a flow-through lake: Lake Hampen, Western Denmark. *Vadose Zone Journal*, *10*(1), 110-124. https://doi.org/10.2136/vzj2010.0017

Kohzu, A., Miyajima, T., Tayasu, I., Yoshimizu, C., Hyodo, F., Matsui, K., Nakano, T., Wada, E., Fujita, N., & Nagata, T. (2008). Use of stable nitrogen isotope signatures of riparian macrophytes as an indicator of anthropogenic N inputs to river ecosystems. *Environmental Science & Technology*, *42*(21), 7837-7841. https://doi.org/10.1021/es801113k

Kusabs, I. A. (2015). *Kōura (Paranephrops planifrons) populations in the Te Arawa lakes: An ecological assessment using the traditional Māori tau kōura harvesting method and recommendations for sustainable management* [Doctoral thesis, University of Waikato]. University of Waikato Research Commons. https://hdl.handle.net/10289/9346

Kusabs, I. A. (2020). *Lakes Tarawera and Ōkāreka: Rotorua Te Arawa lakes kōura monitoring programme 2020* (Report number 4). Ian Kusabs & Associates Ltd. https://www.rotorualakes.co.nz/vdb/document/1839

Land, Air, Water Aotearoa. (n.d.-a). *Lake Tarawera*. Retrieved February 9, 2026, from https://www.lawa.org.nz/explore-data/bay-of-plenty-region/lakes/lake-tarawera

Land, Air, Water Aotearoa. (n.d.-b). *Bay of Plenty region: Lakes*. Retrieved February 9, 2026, from https://www.lawa.org.nz/explore-data/bay-of-plenty-region/lakes

Land, Air, Water Aotearoa. (n.d.-c). *Lake Wanaka*. Retrieved January 18, 2026, from https://www.lawa.org.nz/explore-data/otago-region/lakes/lake-wanaka

Land, Air, Water Aotearoa. (n.d.-d). *Lake Okareka*. Retrieved February 9, 2026, from https://www.lawa.org.nz/explore-data/bay-of-plenty-region/lakes/lake-okareka

Lewandowski, J., Meinikmann, K., Nützmann, G., & Rosenberry, D. O. (2015). Groundwater – the disregarded component in lake water and nutrient budgets. Part 2: effects of groundwater on nutrients. *Hydrological Processes*, *29*(13), 2922-2955. https://doi.org/10.1002/hyp.10384

Li, Q., Xiong, Y., Deng, Y., Gan, Y., & Du, Y. (2025). Contrasting sources and fate of ammonium in groundwater systems of two large alluvial-lacustrine basins. *Applied Geochemistry*, *180*, 106297. https://doi.org/10.1016/j.apgeochem.2025.106297

Li, Z., Zhang, M., Cao, T., Zhang, M., Ni, L., Xie, P., & Xu, J. (2011). Variation in stable isotope signatures of the submersed macrophyte *Vallisneria natans* collected from several shallow lakes in China. *Journal of Freshwater Ecology*, *26*(3), 429-433. https://doi.org/10.1080/02705060.2011.562000

Land Information New Zealand. (2025a). *NZ building points (topo, 1:50k)* [Point layer]. LINZ Data Service. Retrieved August 20, 2025, from https://data.linz.govt.nz/layer/50245-nz-building-points-topo-150k/

Land Information New Zealand. (2025b). *NZ building outlines* [Polygon layer]. LINZ Data Service. Retrieved August 20, 2025, from https://data.linz.govt.nz/layer/101290-nz-building-outlines/

Land Information New Zealand. (2025c). *New Zealand LiDAR 1m DEM* [Grid layer]. LINZ Data Service. Retrieved August 20, 2025, from https://data.linz.govt.nz/layer/121859-new-zealand-lidar-1m-dem/

Lisboa, M. S., Schneider, R. L., Rudstam, L. G., & Walter, M. T. (2024). Groundwater inputs could be a significant but often overlooked source of phosphorus in lake ecosystems. *Scientific Reports*, *14*(1), 16269. https://doi.org/10.1038/s41598-024-66985-z

Lock, M. A., & John, P. H. (1978). The measurement of groundwater discharge into a lake by a direct method. *Internationale Revue der gesamten Hydrobiologie und Hydrographie*, *63*(2), 271-275. https://doi.org/10.1002/iroh.19780630212

Lusk, M. G., Toor, G. S., Yang, Y.-Y., Mechtensimer, S., De, M., & Obreza, T. A. (2017). A review of the fate and transport of nitrogen, phosphorus, pathogens, and trace organic chemicals in septic systems. *Critical Reviews in Environmental Science and Technology*, *47*(7), 455-541. https://doi.org/10.1080/10643389.2017.1327787

Malkin, S. Y., Bocaniov, S. A., Smith, R. E., Guildford, S. J., & Hecky, R. E. (2010). *In situ* measurements confirm the seasonal dominance of benthic algae over phytoplankton in nearshore primary production of a large lake. *Freshwater Biology*, *55*(12), 2468-2483. https://doi.org/10.1111/j.1365-2427.2010.02477.x

McCormick, A. R., Phillips, J. S., Botsch, J. C., & Ives, A. R. (2021). Shifts in the partitioning of benthic and pelagic primary production within and across summers in Lake Mývatn, Iceland. *Inland Waters*, *11*(1), 13-28. https://doi.org/10.1080/20442041.2020.1859868

McIntyre, P. B., Michel, E., & Olsgard, M. (2006). Top-down and bottom-up controls on periphyton biomass and productivity in Lake Tanganyika. *Limnology and Oceanography*, *51*(3), 1514-1523. https://doi.org/10.4319/lo.2006.51.3.1514

Meyer, M. F., Ozersky, T., Woo, K. H., Shchapov, K., Galloway, A. W., Schram, J. B., Rosi, E. J., Snow, D. D., Timofeyev, M. A., & Karnaukhov, D. Y. (2022a). Effects of spatially heterogeneous lakeside development on nearshore biotic communities in a large, deep, oligotrophic lake. *Limnology and Oceanography*, *67*(12), 2649-2664. https://doi.org/10.1002/lno.12228

Meyer, M. F., Ozersky, T., Woo, K. H., Shchapov, K., Galloway, A. W., Schram, J. B., Snow, D. D., Timofeyev, M. A., Karnaukhov, D. Y., & Brousil, M. R. (2022b). A unified dataset of colocated sewage pollution, periphyton, and benthic macroinvertebrate community and food web structure from Lake Baikal (Siberia). *Limnology and Oceanography Letters*, *7*(1), 62-79. https://doi.org/10.1002/lol2.10219

Ministry for the Environment. (2020). *The New Zealand wastewater sector* (CR 452). https://environment.govt.nz/assets/Publications/Files/wastewater-sector-report.pdf

Narancic, B., Laurion, I., Wolfe, B. B., Behmel, S., & Rousseau, A. N. (2020). Seasonal contributions of water and pollutants to Lake St. Charles, a drinking water reservoir. *Canadian Water Resources Journal/Revue canadienne des ressources hydriques*, *45*(2), 125-143. https://doi.org/10.1080/07011784.2019.1706641

Naranjo, R. C., Niswonger, R. G., Smith, D., Rosenberry, D., & Chandra, S. (2019). Linkages between hydrology and seasonal variations of nutrients and periphyton in a large oligotrophic subalpine lake. *Journal of Hydrology*, *568*, 877-890. https://doi.org/10.1016/j.jhydrol.2018.11.033

National Institute of Water and Atmosphere Research. (2023a). *Climate database statistics report: January 2023*. https://niwa.co.nz/sites/default/files/Climate%20Statistics%20-%20January%202023.pdf

National Institute of Water and Atmosphere Research. (2023b). *Climate database statistics report: February 2023*. https://niwa.co.nz/sites/default/files/Climate_Statistics_February_2023.pdf

Noakes, K. (2016). *Monitoring and hydrodynamic modelling of groundwater inflows into Lake Rotokakahi, New Zealand* [Master's thesis, University of Waikato]. University of Waikato Research Commons. https://hdl.handle.net/10289/11239

Nozaki, K. (2001). Abrupt change in primary productivity in a littoral zone of Lake Biwa with the development of a filamentous green-algal community. *Freshwater Biology*, *46*(5), 587-602. https://doi.org/10.1046/j.1365-2427.2001.00696.x

Nyenje, P., Foppen, J., Kulabako, R., Muwanga, A., & Uhlenbrook, S. (2013). Nutrient pollution in shallow aquifers underlying pit latrines and domestic solid waste dumps in urban slums. *Journal of Environmental Management*, *122*, 15-24. https://doi.org/10.1016/j.jenvman.2013.02.040

Ouyang, Y., & Zhang, J.-E. (2012). Quantification of shallow groundwater nutrient dynamics in septic areas. *Water, Air, & Soil Pollution*, *223*(6), 3181-3193. https://doi.org/10.1007/s11270-012-1100-1

Page, M., Goldhammer, T., Hilt, S., Tolentino, S., & Brothers, S. (2022). Filamentous algae blooms in a large, clear-water lake: Potential drivers and reduced benthic primary production. *Water*, *14*(13), 2136. https://doi.org/10.3390/w14132136

Périllon, C., Pöschke, F., Lewandowski, J., Hupfer, M., & Hilt, S. (2017). Stimulation of epiphyton growth by lacustrine groundwater discharge to an oligo-mesotrophic hard-water lake. *Freshwater Science*, *36*(3), 555-570. https://doi.org/10.1086/692832

Prentice, M. J., Abell, J. M., McBride, C. G., Beckett, T., & Özkundakci, D. (in preparation). *Nutrient pathways in hydrologically connected lakes: Groundwater as the primary driver of nutrient loading in a large, deep, monomictic lake* [Manuscript in preparation]. School of Science, University of Waikato.

Puts, I. C., Bergström, A. K., Verheijen, H. A., Norman, S., & Ask, J. (2022). An ecological and methodological assessment of benthic gross primary production in northern lakes. *Ecosphere*, *13*(3), e3973. https://doi.org/10.1002/ecs2.3973

Rakhimbekova, S., O'Carroll, D. M., Oldfield, L. E., Ptacek, C. J., & Robinson, C. E. (2021). Spatiotemporal controls on septic system derived nutrients in a nearshore aquifer and their discharge to a large lake. *Science of the Total Environment*, *752*, 141262. https://doi.org/10.1016/j.scitotenv.2020.141262

Ray, D. E. (2007). *Septic tank failure in NZ: How serious is the problem?* Environmental Management Services Ltd. https://www.waternz.org.nz/Attachment?Action=Download&Attachment_id=1352

Reynaud, A., & Lanzanova, D. (2017). A global meta-analysis of the value of ecosystem services provided by lakes. *Ecological Economics*, *137*, 184-194. https://doi.org/10.1016/j.ecolecon.2017.03.001

Robertson, W., & Harman, J. (1999). Phosphate plume persistence at two decommissioned septic system sites. *Groundwater*, *37*(2), 228-236. https://doi.org/10.1111/j.1745-6584.1999.tb00978.x

Robertson, W. D., Van Stempvoort, D. R., & Schiff, S. L. (2019). Review of phosphorus attenuation in groundwater plumes from 24 septic systems. *Science of the Total Environment*, *692*, 640-652. https://doi.org/10.1016/j.scitotenv.2019.07.198

Robinson, C. (2015). Review on groundwater as a source of nutrients to the Great Lakes and their tributaries. *Journal of Great Lakes Research*, *41*(4), 941-950. https://doi.org/10.1016/j.jglr.2015.08.001

Rogers, K. M., van der Raaij, R., Phillips, A., & Stewart, M. (2023). A national isotope survey to define the sources of nitrate contamination in New Zealand freshwaters. *Journal of Hydrology*, *617*(Part C), 129131. https://doi.org/10.1016/j.jhydrol.2023.129131

Rosenberry, D. O., Lewandowski, J., Meinikmann, K., & Nützmann, G. (2015). Groundwater - the disregarded component in lake water and nutrient budgets. Part 1: effects of groundwater on hydrology. *Hydrological Processes*, *29*(13), 2895-2921. https://doi.org/10.1002/hyp.10403

Rosenberry, D. O., Sheibley, R. W., Cox, S. E., Simonds, F. W., & Naftz, D. L. (2013). Temporal variability of exchange between groundwater and surface water based on high-frequency direct measurements of seepage at the sediment-water interface. *Water Resources Research*, *49*(5), 2975-2986. https://doi.org/10.1002/wrcr.20198

Rotorua Lakes Council. (2025a). *Tarawera sewerage scheme*. Retrieved February 9, 2026, from https://www.rotorualakescouncil.nz/our-council/projects/tarawera-sewerage-scheme#toc-link-6

Rotorua Lakes Council. (2025b). *Tarawera sewerage scheme: Property owner factsheet* (Version 8). https://r3nk7o.digitalcityplatform.online/RedPublishPROD/api/docs/21442202?f=inline

Roumelis, C., Willert, F., Scaccia, M., Welch, S., Gabor, R., Carrera, J., Folch, A., Salgot, M., & Sawyer, A. H. (2025). Water table fluctuations control nitrate and ammonium fate in coastal aquifers. *Water Resources Research*, *61*(1), e2024WR038087. https://doi.org/10.1029/2024WR038087

Roy, J. W., Spoelstra, J., Robertson, W. D., Klemt, W., & Schiff, S. L. (2017). Contribution of phosphorus to Georgian Bay from groundwater of a coastal beach town with decommissioned septic systems. *Journal of Great Lakes Research*, *43*(6), 1016-1029. https://doi.org/10.1016/j.jglr.2017.09.001

Sadro, S., Melack, J. M., & MacIntyre, S. (2011). Spatial and temporal variability in the ecosystem metabolism of a high-elevation lake: Integrating benthic and pelagic habitats. *Ecosystems*, *14*(7), 1123-1140. https://doi.org/10.1007/s10021-011-9471-5

Saunders, W. M. H. (1965). Phosphate retention by New Zealand soils and its relationship to free sesquioxides, organic matter, and other soil properties. *New Zealand Journal of Agricultural Research*, *8*(1), 30-57. https://doi.org/10.1080/00288233.1965.10420021

Schindler, D. E., & Scheuerell, M. D. (2002). Habitat coupling in lake ecosystems. *Oikos*, *98*(2), 177-189. https://doi.org/10.1034/j.1600-0706.2002.980201.x

Schindler, D. W., & Vallentyne, J. R. (2008). *The algal bowl: Overfertilization of the world's freshwaters and estuaries*. University of Alberta Press.

Scholes, P. (2007). *Onsite effluent disposal in the Bay of Plenty 2006* [Environmental Publication 2007/01]. Environment Bay of Plenty. https://atlas.boprc.govt.nz/api/v1/edms/document/A3888669/content

Schullehner, J., Stayner, L., & Hansen, B. (2017). Nitrate, nitrite, and ammonium variability in drinking water distribution systems. *International Journal of Environmental Research and Public Health*, *14*(3), 276. https://doi.org/10.3390/ijerph14030276

Sebestyen, S. D., & Schneider, R. L. (2001). Dynamic temporal patterns of nearshore seepage flux in a headwater Adirondack lake. *Journal of Hydrology*, *247*(3-4), 137-150. https://doi.org/10.1016/S0022-1694(01)00377-8

Søndergaard, M., Jensen, J. P., & Jeppesen, E. (2003). Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia*, *506*(1), 135-145. https://doi.org/10.1023/B:HYDR.0000008611.12704.dd

Steffy, L. Y., & Kilham, S. S. (2004). Elevated $\delta^{15}N$ in stream biota in areas with septic tank systems in an urban watershed. *Ecological Applications*, *14*(3), 637-641. https://doi.org/10.1890/03-5148

Stevenson, R. J., & Bahls, L. L. (1999). Chapter 6: Periphyton protocols. In M. T. Barbour, J. Gerritsen, B. D. Snyder, & J. B. Stribling (Eds.), *Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish* (2nd ed., EPA 841-B-99-002, pp. 6-1–6-22). United States Environmental Protection Agency, Office of Water. https://nepis.epa.gov/Exe/ZyPDF.cgi/20004OQK.PDF?Dockey=20004OQK.PDF

Stewart, S. D., Kelly, D., Biessy, L., Laroche, O., & Wood, S. A. (2021). Individual diet specialization drives population trophic niche responses to environmental change in a predator fish population. *Food Webs*, *27*, e00193. https://doi.org/10.1016/j.fooweb.2021.e00193

Su, C., Jiang, J., Xie, X., Han, Z., Wang, M., Li, J., & Shi, H. (2023). Sources and cycling processes of nitrogen revealed by stable isotopes and hydrochemistry in a typical agricultural lake basin. *Applied Geochemistry*, *156*, 105662. https://doi.org/10.1016/j.apgeochem.2023.105662

Tahoe Environmental Research Center. (n.d.). *Periphyton monitoring*. UC Davis. https://tahoe.ucdavis.edu/periphyton-0#:~:text=The%20amount%20of%20periphyton%20biomass,shore%20(right)%20of%20Lake%20Tahoe

Takatert, N., Sanchez-Pérez, J. M., & Trémolières, M. (1999). Spatial and temporal variations of nutrient concentration in the groundwater of a floodplain: Effect of hydrology, vegetation and substrate. *Hydrological Processes*, *13*(10), 1511-1526. https://doi.org/10.1002/(SICI)1099-1085(199907)13:10<1511::AID-HYP828>3.0.CO;2-F

Taylor, A., Singleton, P., & Özkundakci, D. (2008). Land use impacts on lake water quality. In D. J. Lowe (Ed.), *Guidebook for pre-conference North Island field trip A1 "Ashes and Issues", 28-30 November 2008, Australian and New Zealand 4th Joint Soils Conference, Massey University, Palmerston North* (pp. 107-111). New Zealand Society of Soil Science. https://www.researchgate.net/profile/David-Lowe-2/publication/240355556_Guidebook_for_Pre-conference_North_Island_Field_Trip_A1_'Ashes_and_Issues'/links/62a075ff55273755ebdc19ab/Guidebook-for-Pre-conference-North-Island-Field-Trip-A1-Ashes-and-Issues.pdf

Timoshkin, O., Moore, M., Kulikova, N., Tomberg, I., Malnik, V., Shimaraev, M., Troitskaya, E., Shirokaya, A., Sinyukovich, V., & Zaitseva, E. (2018). Groundwater contamination by sewage causes benthic algal outbreaks in the littoral zone of Lake Baikal (East Siberia). *Journal of Great Lakes Research*, *44*(2), 230-244. https://doi.org/10.1016/j.jglr.2018.01.008

U.S. Environmental Protection Agency. (2002). *Onsite wastewater treatment systems manual* (EPA/625/R-00/008). https://www.epa.gov/sites/default/files/2015-06/documents/2004_07_07_septics_septic_2002_osdm_all.pdf

Vadeboncoeur, Y., & Power, M. E. (2017). Attached algae: The cryptic base of inverted trophic pyramids in freshwaters. *Annual Review of Ecology, Evolution, and Systematics*, *48*(1), 255-279. https://doi.org/10.1146/annurev-ecolsys-121415-032340

Vadeboncoeur, Y., Jeppesen, E., Zanden, M. J. V., Schierup, H. H., Christoffersen, K., & Lodge, D. M. (2003). From Greenland to green lakes: Cultural eutrophication and the loss of benthic pathways in lakes. *Limnology and Oceanography*, *48*(4), 1408-1418. https://doi.org/10.4319/lo.2003.48.4.1408

Vadeboncoeur, Y., Lodge, D. M., & Carpenter, S. R. (2001). Whole-lake fertilization effects on distribution of primary production between benthic and pelagic habitats. *Ecology*, *82*(4), 1065-1077. https://doi.org/10.1890/0012-9658(2001)082[1065:WLFEOD]2.0.CO;2

Vadeboncoeur, Y., Moore, M. V., Stewart, S. D., Chandra, S., Atkins, K. S., Baron, J. S., Bouma-Gregson, K., Brothers, S., Francoeur, S. N., & Genzoli, L. (2021). Blue waters, green bottoms: Benthic filamentous algal blooms are an emerging threat to clear lakes worldwide. *BioScience*, *71*(10), 1011-1027. https://doi.org/10.1093/biosci/biab049

Vadeboncoeur, Y., Vander Zanden, M. J., & Lodge, D. M. (2002). Putting the lake back together: Reintegrating benthic pathways into lake food web models. *BioScience*, *52*(1), 44-54. https://doi.org/10.1641/0006-3568(2002)052[0044:PTLBTR]2.0.CO;2

Vander Zanden, M. J., Chandra, S., Park, S.-K., Vadeboncoeur, Y., & Goldman, C. R. (2006). Efficiencies of benthic and pelagic trophic pathways in a subalpine lake. *Canadian Journal of Fisheries and Aquatic Sciences*, *63*(12), 2608-2620. https://doi.org/10.1139/f06-148

Wang, S., Robinson, C. E., Spoelstra, J., Schneidewind, U., & Roy, J. W. (2024). Phosphorus loading to nearshore waters from legacy septic system groundwater plumes in a Great Lakes coastal community. *Journal of Great Lakes Research*, *50*(6), 102453. https://doi.org/10.1016/j.jglr.2024.102453

Wang, X., Liu, Z., Xu, Y. J., Mao, B., Jia, S., Wang, C., Ji, X., & Lv, Q. (2025). Revealing nitrate sources seasonal difference between groundwater and surface water in China's largest fresh water lake (Poyang Lake): Insights from sources proportion, dynamic evolution and driving forces. *Science of the Total Environment*, *958*, 178134. https://doi.org/10.1016/j.scitotenv.2024.178134

Wassenaar, L. I., & Hendry, M. J. (1999). Improved piezometer construction and sampling techniques to determine pore water chemistry in aquitards.

Groundwater, *37*(4), 564-571. https://doi.org/10.1111/j.1745-6584.1999.tb01143.x

Water Services (Wastewater Environmental Performance Standards) Regulations 2025. https://www.legislation.govt.nz/regulation/public/2025/0258/latest/whole.html

Wells, R. D., de Winton, M. D., & Clayton, J. S. (1997). Successive macrophyte invasions within the submerged flora of Lake Tarawera, central North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *31*(4), 449-459. https://doi.org/10.1080/00288330.1997.9516778

Wilson, N. J. (2022). *Flows of water and nutrients to Lake Tarawera and connected lakes* [Master's thesis, University of Waikato]. University of Waikato Research Commons. https://hdl.handle.net/10289/15294

Withers, P., Jarvie, H., & Stoate, C. (2011). Quantifying the impact of septic tank systems on eutrophication risk in rural headwaters. *Environment International*, *37*(3), 644-653. https://doi.org/10.1016/j.envint.2011.01.002

Withers, P. J., Jordan, P., May, L., Jarvie, H. P., & Deal, N. E. (2014). Do septic tank systems pose a hidden threat to water quality? *Frontiers in Ecology and the Environment*, *12*(2), 123-130. https://doi.org/10.1890/130131

Appendices

Appendix A: Chapter 2 supplementary materials

Table A1. General statistics for piezometer sites and the artesian spring. Piezometer depth is below ground level. Ground elevation for each site and lake level (296.5m above sea level) was sourced from LiDAR survey data from the LINZ data service, current as of 19th September 2025 (LINZ, 2025c). Sampling success rate is the number of times a piezometer yielded groundwater, out of a total of 24 sampling occasions.

Piezometer site	Installation date	Piezometer depth (m)	Ground elevation above lake level (m)	*n*	Sampling success rate (%)
C1 - Above	18/01/2023	2.70	5.8	4	16.7%
C1 - Below	18/01/2023	2.65	4.7	1	4.2%
T2 - Above	9/03/2023	2.80	109.4	1	4.2%
T2 - Below	18/01/2023	2.80	4.6	8	33.3%
T3 - Above	9/03/2023	2.80	122.1	11	50.0%
T3 - Below	18/01/2023	2.80	5.8	17	70.8%
T4 - Above	10/02/2023	2.80	27.8	4	16.7%
T4 - Below	10/02/2023	2.50	4.2	13	54.2%
T5 - Above	18/01/2023	2.70	18.2	18	75.0%
T5 - Below	18/01/2023	2.80	5.6	6	25.0%
T6 - Above	11/02/2023	2.70	21.7	7	0.0%
T6 - Below	10/02/2023	2.80	3.4	0	29.2%
T7 - Above	9/03/2023	1.30	50.0	4	16.7%
T7 - Below	10/02/2023	2.65	24.4	5	20.8%
T1 - Above	10/02/2023	2.80	44.1	0	0.0%
T1 - Below	10/02/2023	2.80	4.2	3	12.5%
T8 - Below	19/01/2023	1.70	3.9	2	8.3%
T9 - Above	19/01/2023	2.80	16.5	7	29.2%
T9 - Below	22/02/2023	2.90	3.5	21	87.5%
C3- Above	11/02/2023	1.50	3.1	12	50.0%
C3 - Below	11/02/2023	2.80	3.7	12	50.0%
Artesian spring	-	-	5.0	24	100.0%

Table A2. Summary statistics for nutrient concentrations at each individual piezometer site. *Stdev* is standard deviation, *DRP* is dissolved reactive phosphorus, and *min* and *max* are minimum and maximum values.

Nutrient	Piezometer site	Mean	Stdev	Min	Max
Ammoniacal-N (mg L^{-1})	C1 - Above	0.660	0.61	0.05	1.48
	C1 - Below	0.005	-	< 0.01	< 0.01
	T2 - Above	7.300	-	7.30	7.30
	T2 - Below	0.085	0.13	< 0.01	0.41
	T3 - Above	0.170	0.18	< 0.01	0.52
	T3 - Below	0.080	0.08	< 0.01	0.36
	T4 - Above	0.390	0.70	< 0.01	1.44
	T4 - Below	0.066	0.04	< 0.01	0.12
	T5 - Above	22.300	20.20	1.11	61.00
	T5 - Below	0.360	0.52	0.02	1.32
	T6 - Below	0.340	0.34	0.02	0.94
	T7 - Above	0.280	0.48	< 0.01	0.99
	T7 - Below	3.430	6.31	< 0.01	14.60
	T1 - Below	0.029	0.03	< 0.01	0.06
	T8 - Below	0.580	0.40	0.29	0.86
	T9 - Above	0.650	0.85	0.02	2.00
	T9 - Below	0.076	0.07	< 0.01	0.21
	C3- Above	0.130	0.13	< 0.01	0.35
	C3 - Below	0.420	0.46	0.01	1.33
DRP (mg L^{-1})	C1 - Above	0.002	0.000	< 0.004	< 0.004
	C1 - Below	0.330	-	0.33	0.330
	T2 - Above	0.140	-	0.14	0.137
	T2 - Below	0.003	0.003	< 0.004	0.011
	T3 - Above	0.003	0.001	< 0.004	0.006
	T3 - Below	0.013	0.011	< 0.004	0.041
	T4 - Above	0.002	0.000	< 0.004	< 0.004
	T4 - Below	0.078	0.076	< 0.004	0.210
	T5 - Above	0.130	0.018	< 0.004	0.580
	T5 - Below	0.006	0.011	< 0.004	0.028
	T6 - Below	0.002	0.000	< 0.004	< 0.004
	T7 - Above	0.079	0.150	< 0.004	0.310
	T7 - Below	3.440	3.380	0.01	8.700
	T1 - Below	0.003	0.002	< 0.004	0.005
	T8 - Below	0.002	0.000	< 0.004	< 0.004
	T9 - Above	0.160	0.150	< 0.004	0.380
	T9 - Below	0.005	0.004	< 0.004	0.018
	C3- Above	0.002	0.000	< 0.004	< 0.004
	C3 - Below	0.002	0.001	< 0.004	0.006

Nutrient	Piezometer site	Mean	Stdev	Min	Max
Nitrate-N (mg L^{-1})	C1 - Above	0.140	0.07	0.045	0.194
	C1 - Below	0.006	-	0.006	0.006
	T2 - Above	0.210	-	0.210	0.210
	T2 - Below	0.007	0.01	< 0.002	0.028
	T3 - Above	0.120	0.10	< 0.002	0.360
	T3 - Below	0.170	0.22	< 0.002	0.670
	T4 - Above	0.010	0.02	< 0.002	0.037
	T4 - Below	1.680	2.05	0.030	7.400
	T5 - Above	0.290	0.45	0.006	1.610
	T5 - Below	0.010	0.01	< 0.002	0.021
	T6 - Below	0.094	0.11	< 0.002	0.230
	T7 - Above	0.370	0.44	< 0.002	0.910
	T7 - Below	1.630	2.54	< 0.002	5.900
	T1 - Below	0.980	1.00	< 0.002	2.000
	T8 - Below	0.011	0.01	< 0.002	0.021
	T9 - Above	0.380	0.78	< 0.002	2.100
	T9 - Below	0.260	0.21	0.019	0.760
	C3- Above	0.580	1.11	0.003	4.000
	C3 - Below	0.540	0.67	< 0.002	1.910
Nitrite-N (mg L^{-1})	C1 - Above	0.001	0.001	< 0.002	0.002
	C1 - Below	0.004	-	0.004	0.004
	T2 - Above	0.087	-	0.087	0.087
	T2 - Below	0.016	0.042	< 0.002	0.120
	T3 - Above	0.005	0.008	< 0.002	0.027
	T3 - Below	0.005	0.011	< 0.002	0.048
	T4 - Above	0.001	0.000	< 0.002	0.001
	T4 - Below	0.005	0.007	< 0.002	0.024
	T5 - Above	0.015	0.018	< 0.002	0.056
	T5 - Below	0.001	0.000	< 0.002	0.002
	T6 - Below	0.002	0.002	< 0.002	0.005
	T7 - Above	0.020	0.018	< 0.002	0.045
	T7 - Below	0.020	0.400	0.003	0.910
	T1 - Below	0.049	0.072	< 0.002	0.132
	T8 - Below	0.001	0.000	< 0.002	0.001
	T9 - Above	0.017	0.025	< 0.002	0.066
	T9 - Below	0.006	0.014	< 0.002	0.058
	C3- Above	0.002	0.002	< 0.002	0.007
	C3 - Below	0.003	0.004	< 0.002	0.016

Table A3. Summary statistics for groundwater nutrient concentrations at piezometer sites located in native vegetation sub-catchment land use and within the urban zone of Lake Tarawera. *Stdev* is standard deviation, *DRP* is dissolved reactive phosphorus, and *min* and *max* are minimum and maximum values.

Nutrient	Sub-catchment land use	Mean	Stdev	Min	Max
Nitrate-N (mg L^{-1})	Native vegetation	0.450	0.130	0.001	4.00
	Urban	0.440	0.550	0.001	7.40
Nitrite-N (mg L^{-1})	Native vegetation	0.002	0.001	0.001	0.02
	Urban	0.018	0.042	0.001	0.91
Ammoniacal-N (mg L^{-1})	Native vegetation	0.180	0.210	0.005	1.48
	Urban	0.400	0.960	0.005	14.60
DRP (mg L^{-1})	Native vegetation	0.027	0.047	0.002	0.33
	Urban	0.190	0.710	0.002	8.70

Table A4. Summary statistics for seasonal nutrient concentrations for below sites only. Mean and standard deviation values are weighted. Sample counts ranged from 22 samples from 10 sites in Winter 2023, to two samples from one site in Summer 2024/25. *Stdev* is standard deviation, *DRP* is dissolved reactive phosphorus, and *min* and *max* are minimum and maximum values.

Nutrient	Season	Weighted mean	Weighted stdev	Min	Max
Nitrate-N (mg L^{-1})	Autumn 23	0.091	0.215	< 0.002	0.760
	Winter 23	0.393	0.738	< 0.002	4.200
	Spring 23	0.524	0.926	< 0.002	7.400
	Summer 23/24	0.488	0.389	< 0.002	1.590
	Autumn 24	1.119	2.143	0.002	5.900
	Winter 24	0.783	0.587	0.060	2.100
	Spring 24	0.541	0.563	0.029	1.770
Nitrite-N (mg L^{-1})	Autumn 23	0.018	0.034	< 0.002	0.120
	Winter 23	0.011	0.028	< 0.002	0.132
	Spring 23	0.005	0.006	< 0.002	0.025
	Summer 23/24	0.002	0.001	< 0.002	0.008
	Autumn 24	0.160	0.335	< 0.002	0.910
	Winter 24	0.005	0.008	< 0.002	0.031
	Spring 24	0.002	0.001	< 0.002	0.003
Ammoniacal-N (mg L^{-1})	Autumn 23	0.228	0.357	< 0.010	0.940
	Winter 23	0.184	0.143	< 0.010	0.950
	Spring 23	0.885	3.000	0.012	14.600
	Summer 23/24	0.246	0.221	0.021	1.330
	Autumn 24	0.434	0.791	0.030	2.200
	Winter 24	0.077	0.041	0.017	0.210
	Spring 24	0.040	0.011	0.015	0.065
DRP (mg L^{-1})	Autumn 23	0.005	0.008	< 0.004	0.028
	Winter 23	0.050	0.203	< 0.004	0.980
	Spring 23	0.187	0.832	< 0.004	4.000
	Summer 23/24	0.006	0.007	< 0.004	0.029
	Autumn 24	1.461	3.238	< 0.004	8.700
	Winter 24	0.327	0.958	< 0.004	3.500
	Spring 24	0.068	0.081	0.007	0.210

Table A5. Summary statistics for shallow groundwater nutrient concentrations collected from piezometers along the western margin of Lake Tarawera during the first and second years of the study. *Stdev* is standard deviation, *DRP* is dissolved reactive phosphorus, and *min* and *max* are minimum and maximum values.

Nutrient	Year	Weighted mean	Weighted stdev	Min	Max
Nitrate-N (mg L^{-1})	1st Year	0.39	0.64	< 0.002	7.40
	2nd Year	0.80	0.98	0.019	5.90
Nitrite-N (mg L^{-1})	1st Year	0.01	0.01	< 0.002	0.13
	2nd Year	0.04	0.12	< 0.002	0.91
Ammoniacal-N (mg L^{-1})	1st Year	0.43	1.00	< 0.010	14.60
	2nd Year	0.14	0.29	< 0.010	2.20
DRP (mg L^{-1})	1st Year	0.08	0.35	< 0.004	4.00
	2nd Year	0.48	1.56	< 0.004	8.70

Appendix B: Chapter 3 supplementary materials

Table B1. Summary statistics for groundwater inflow during each sampling occasion (once per month) and at each site. *Stdev* is standard deviation, *DRP* is dissolved reactive phosphorus, and *min* and *max* are minimum and maximum values.

			Groundwater inflow (L m^{-2} day^{-1})			
Category	Label	*n*	Mean	Standard deviation	Minimum	Maximum
Sampling period	April-23	6	3.16	2.55	0.88	8.15
	May-23	4	3.51	1.24	2.12	4.91
	June-23	5	2.52	1.58	0.70	4.74
	July-23	3	1.56	1.56	0.61	3.36
	August-23	4	1.31	1.09	0.52	2.91
	September-23	5	2.91	2.43	0.37	5.36
	October-23	3	0.82	0.57	0.43	1.47
	November-23	2	2.87	0.10	2.80	2.94
	February-24	2	0.78	0.62	0.35	1.22
	March-24	3	1.96	1.19	0.62	2.89
Site	BVR	4	2.21	1.15	0.62	3.14
	C1	3	3.66	2.56	0.72	5.36
	T1	5	1.98	1.32	0.49	3.13
	T2	3	4.22	0.92	3.36	5.18
	T3	4	1.66	1.02	0.52	2.90
	T4	4	1.22	0.76	0.56	2.26
	T5	3	1.75	1.05	0.88	2.91
	T6	3	4.65	3.54	1.07	8.15
	T7	3	1.88	1.33	0.37	2.90
	T8	5	1.40	1.25	0.35	2.91
Overall	All samples	37	2.32	1.77	0.35	8.15

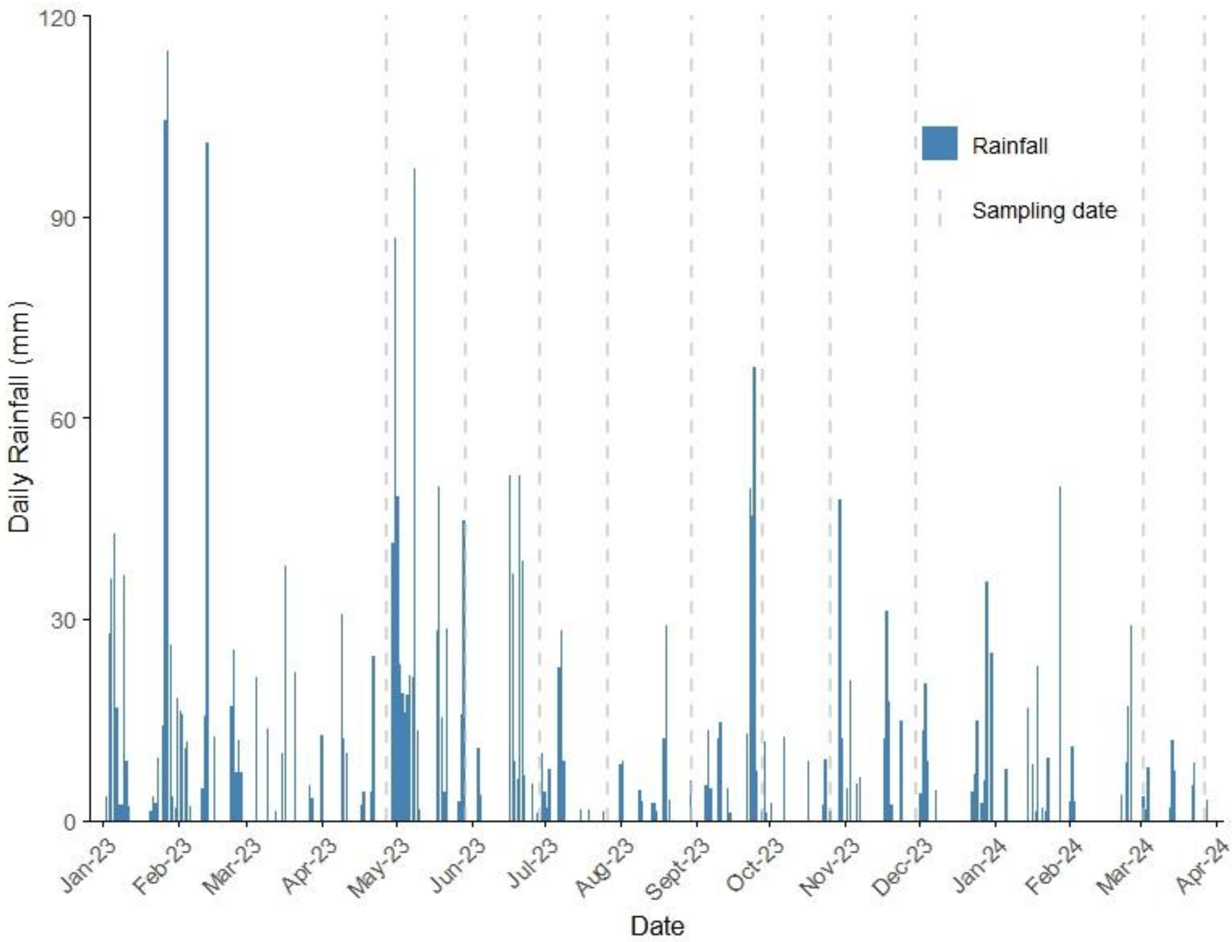

Figure B1. Daily rainfall data from virtual climate station (VCS) 27368, located on the northern shore of Lake Tarawera, for the period 1st January 2023 – 30th March 2024. Data is sourced from Earth Sciences New Zealand (2025). Blue bars are daily rainfall, and dashed grey lines indicate when groundwater inflow sampling occurred.

Table B2. Summary statistics for previous rainfall periods prior to benthic chamber deployment dates (10 deployment dates, 37 measurements of groundwater inflow total). Each period is the accumulation of rainfall during that time, starting from the day before deployment of benthic chambers. One month previous rainfall is four weeks duration.

Previous period of rainfall (mm)	Mean	Standard deviation	Minimum	Maximum
24-hour	5.0	14.0	0.0	44.6
48-hour	8.5	18.6	0.0	60.4
72-hour	18.9	29.5	0.0	74.9
5-day	38.4	55.8	0.0	169.8
7-day	49.1	57.6	0.0	182.8
2-week	96.7	79.5	4.3	206.3
3-week	128.8	91.6	32.5	301.1
1-month	181.4	161.1	56.6	597.3

Table B3. Raw $\delta^{15}N$ and $\delta^{18}O$ stable isotope and nutrient *concentration* data for pore water and piezometer sites. *DRP* is dissolved reactive phosphorus.

Sample type	Site	$\delta^{15}N$	$\delta^{18}O$	Nitrate-N ($mg\ L^{-1}$)	Ammoniacal-N ($mg\ L^{-1}$)	Nitrite-N ($mg\ L^{-1}$)	DRP ($mg\ L^{-1}$)
Pore water	C1	4.61	-0.48	0.320	0.02	< 0.002	0.021
	T2	4.47	-2.02	0.033	0.11	0.002	0.063
	T3	4.87	-3.09	0.067	0.08	0.004	0.012
	T4	11.90	9.60	< 0.002	0.14	< 0.002	0.007
	T5	9.32	4.78	1.840	0.05	< 0.002	0.058
	T6	3.83	-4.07	0.156	0.09	< 0.002	0.106
	T7	3.80	-1.44	0.740	0.17	0.002	0.170
	T1	2.48	-2.21	0.196	0.51	0.003	0.330
	T8	8.37	3.37	0.062	0.04	< 0.002	0.016
	BVR	11.42	6.20	< 0.002	0.12	0.004	0.015
Groundwater	Spring	3.43	-1.26	0.210	0.01	< 0.002	0.028
	T3-below	3.69	2.14	0.076	0.02	0.003	0.010
	T4-below	10.88	8.37	1.480	0.06	< 0.002	0.210
	T5-above	6.49	7.22	0.087	28.00	0.005	0.026
	T9-below	6.45	8.62	0.260	0.02	< 0.002	0.013
	T9-above	2.03	-3.44	-	-	-	-

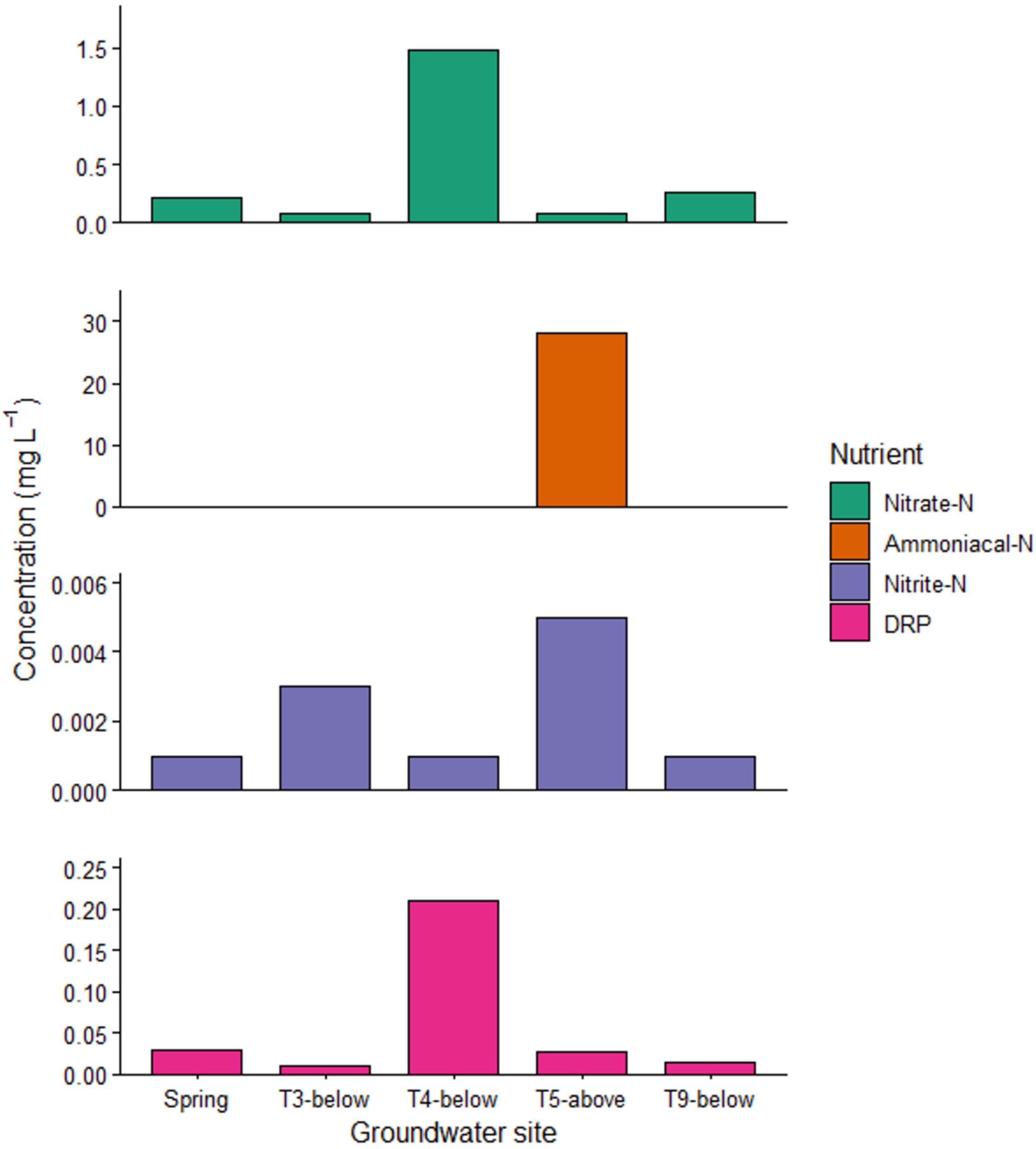

Figure B2. Groundwater nutrient concentrations at piezometer sites. The artesian spring ("Spring") is opportunistically included here as a reference point. *DRP* is dissolved reactive phosphorus.

Table B4. Raw $\delta^{13}C$ and $\delta^{15}N$ stable isotope and carbon (C) and nitrogen (N) content data for primary producer samples in the littoral zone. *SPOM* is suspended particulate organic matter.

Sample type	Site	$\delta^{13}C$	$\delta^{15}N$	N (mg)	C (mg)	N (mg L^{-1})	% C	% N
SPOM	C1	-25.36	2.54	0.028	0.25	0.024	-	-
	T2	-19.61	1.04	0.040	0.34	0.038	-	-
	T3	-25.33	1.97	0.029	0.24	0.014	-	-
	T4	-30.55	2.30	0.016	0.15	0.008	-	-
	T5	-23.94	1.73	0.024	0.23	0.024	-	-
	T6	-22.68	1.45	0.033	0.34	0.031	-	-
	T7	-27.16	0.40	0.018	0.23	0.013	-	-
	T1	-24.48	1.50	0.023	0.18	0.012	-	-
	T8	-26.95	1.52	0.026	0.28	0.015	-	-
	BVR	-26.67	3.06	0.030	0.25	0.020	-	-
Macrophytes	C1	-14.81	2.83	-	-	-	13.8	0.95
	T2	-13.92	0.37	-	-	-	17.0	1.56
	T3	-18.97	1.20	-	-	-	22.3	1.69
	T4	-23.59	1.04	-	-	-	21.8	1.71
	T5	-16.32	1.49	-	-	-	22.0	1.66
	T6	-12.48	0.18	-	-	-	21.3	1.67
	T7	-16.55	0.83	-	-	-	17.4	1.46
	T1	-15.44	0.48	-	-	-	19.7	1.51
	T8	-28.17	2.37	-	-	-	18.9	1.74
	BVR	-22.14	2.06	-	-	-	22.2	1.63
Periphyton	C1	-15.38	0.97	-	-	-	10.4	0.89
	T2	-5.89	0.53	-	-	-	11.7	1.49
	T3	-26.29	1.59	-	-	-	33.8	0.87
	T4	-11.72	1.35	-	-	-	5.6	0.64
	T5	-12.99	2.54	-	-	-	17.5	1.92
	T7	-9.62	0.34	-	-	-	15.5	1.85
	BVR	-27.15	2.28	-	-	-	48.6	0.89

Appendix C: Chapter 4 supplementary materials

Table C1. Mean lux and standard deviations (stdev) for each site in each season. Lux was measured by HOBOloggers placed inside a light chamber, onshore and inside a dark chamber. Missing values indicate where loggers failed. Light chamber and onshore values have been rounded to three significant figures.

		Light chamber (lux)		Onshore (lux)		Dark chamber (lux)	
Season	Site	Mean	Stdev	Mean	Stdev	Mean	Stdev
February	C2	50900	21000	83800	27200	30.9	10.3
	C1	20200	7480	79700	3320	22.7	2.5
	T2	21400	8890	55800	16900	9.1	2.2
	T4	37900	14500	69400	23000	19.6	6.9
	T5	12600	3590	32200	9150	9.7	3.0
	BVR	22100	14400	52000	30100	2.7	1.8
June	C2	11700	2400	60000	21300	199.6	47.5
	C1	15700	5790	45500	5780	106.8	23.9
	T2	7050	2210	46600	6720	117.7	27.4
	T4	8140	2780	36900	3500	132.2	27.1
	T5	4080	873	19100	3590	33.7	6.5
	BVR	3630	717	13400	2450	55.3	10.7
September	C2	29200	7560	-	-	334.4	26.7
	C1	18100	-	54000	2560	-	-
	T2	12500	7570	54300	19400	152.7	58.9
	T4	5090	2340	-	-	-	-
	T5	19000	7580	60700	18700	196.0	58.8
	BVR	10500	3250	-	-	-	-
December	C2	38300	11000	82900	18900	324.2	73.3
	C1	40300	8500	78300	12500	369.4	75.6
	T2	10800	4130	25000	9530	102.5	41.6
	T4	8750	4420	21600	11300	72.9	36.0
	T5	36400	4890	93800	12600	384.4	24.9
	BVR	29100	9300	59000	17700	200.3	52.6

Table C2. Mean gross primary production (GPP) and net ecosystem production (NEP) rates at each site, for each season. Benthic GPP is calculated as mean community GPP minus water column (WC) GPP. Some WC measurements in February were missing due to logger failure; WC GPP was estimated at these sites as the mean of all other sites for that season. Benthic NEP was similarly calculated as the mean community NEP in light chambers minus WC NEP at each site.

		GPP (mg O_2 m^{-2} hr^{-1})			NEP (mg O_2 m^{-2} hr^{-1})		
Season	Site	Benthic	Community	WC	Benthic	Community	WC
February	C2	137.5	166.1	28.6	86.0	106.9	21.0
	C1	119.8	148.3	28.6	78.1	99.0	21.0
	T2	152.0	180.6	28.6	130.9	151.9	21.0
	T4	71.8	100.3	28.6	39.0	59.9	21.0
	T5	68.8	88.6	19.8	50.5	63.5	13.0
	BVR	165.6	202.9	37.3	127.7	156.7	29.0
June	C2	114.8	125.5	10.7	97.0	103.9	6.9
	C1	197.8	219.9	22.1	161.3	181.1	19.8
	T2	101.3	91.4	18.4	57.2	71.1	13.9
	T4	88.9	104.1	15.2	63.2	76.2	13.0
	T5	75.9	100.3	24.4	42.4	59.2	16.8
	BVR	57.4	77.2	19.8	36.3	49.3	13.0
September	C2	102.1	157.7	55.6	115.3	130.5	15.2
	C1	189.0	231.6	42.7	123.4	161.5	38.1
	T2	109.5	146.0	36.6	101.3	120.4	19.1
	T4	185.9	230.1	44.2	138.9	176.3	37.3
	T5	250.9	297.4	46.5	258.3	277.3	19.1
	BVR	70.9	102.1	31.2	63.5	78.7	15.2
December	C2	210.3	238.5	28.2	185.9	176.8	-9.1
	C1	230.1	263.6	33.5	201.6	208.5	6.9
	T2	160.3	176.3	16.0	109.5	117.8	8.4
	T4	76.9	91.4	14.5	32.8	35.0	2.3
	T5	88.4	154.7	66.3	93.5	89.7	-3.8
	BVR	295.9	301.2	5.3	271.2	263.6	-7.6

Table C3. Results of Welch's ANOVA ("global") testing for differences in mean gross primary production (GPP) across sites within each season. A separate Welch's ANOVA was used for each season. All p-values are Holm-corrected. *Comparison* identifies the paired sites used in Games-Howell post-hoc tests. Only significant pairwise results (p <0.05) are shown.

Test	Month	Comparison	p -value
Welch's ANOVA	February	-	0.024
	June	-	0.039
	September	-	0.012
	December	-	0.021
Games-Howell post-hoc	February	T4–BVR	0.033
		T5–BVR	0.017
	September	C1–C2	0.019
		C1–T2	0.048
		C1–BVR	0.025
		T5–T2	0.041
		T5–BVR	0.017
	December	T4–C2	0.015
		T4–C1	0.028

Table C4. Summary statistics for water column gross primary production (GPP) at each littoral site. The percentage of community GPP represents the proportion of mean water column GPP relative to mean community GPP measured using regular chambers.

		Water column GPP (mg O_2 m^{-2} hr^{-1})				
Site	*n*	Mean	Standard deviation	Minimum	Maximum	% of community GPP
C2	3	31.5	22.7	10.7	55.6	18.5
C1	3	32.8	10.3	22.1	42.7	13.7
T2	2	55.2	11.1	16.0	36.6	17.1
T4	3	24.6	16.9	14.5	44.2	16.6
T5	4	39.2	21.5	19.8	66.3	26.3
BVR	4	23.4	14.1	5.3	37.3	19.1