Nutrient and Sediment Dynamics in Two Coastal Plain Rivers in the Bay of Plenty

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Abstract

A large amount of literature concerns eutrophication development in lakes, reservoirs, or estuaries. In contrast, literature concerning to the development of eutrophication in flowing waters is more scarce and, in New Zealand, what there is mostly concerns smaller wadable, cobble-bedded streams and not large soft-bottomed lowland rivers. Lowland rivers are at the distal end of the catchment and suffer from accumulated multiple stressors pertaining to the surrounding land use. In New Zealand coastal plain catchments, land is generally high producing pasture, urban development, or other arable practices

The aim of this study was to develop a better understanding of how two major land-use related contaminants, nitrate, and suspended sediments, vary in two Bay of Plenty rivers, the Kaituna and the Rangitaiki, as they cross the coastal plain. Key goals were to determine the extent to which contaminant concentrations are attenuated or enhanced during this passage. These are two significant North Island New Zealand river systems, with intensive agricultural activities in the lowland part of the catchment. Longitudinal sampling of the lowland reaches took place between November 2020 to June 2021; additional spring samples were collected in November 2021 and early December 2021. The study included measures of nitrate, suspended sediment, turbidity chlorophyll-a, pH, and dissolved oxygen, and nitrate samples from tributary flows and a limited collection of aquatic macrophytes, to assess other aspects of water quality.

The Rangitaiki and Kaituna Rivers both tended to show increasing nitrate concentration as they flowed across the coastal plain, reaching median concentrations of 0.31 and 0.54 mg/NO_3-N/L respectively. Effects of both groundwater and tributaries were evident. Suspended sediment was at relatively low concentrations within both rivers, and generally showed no substantive increase across the lowland reaches of both rivers. Chlorophyll-a and nitrate showed seasonality, and this was largely due to processes in upstream lakes or reservoirs. Little attenuation or development of phytoplankton in the river was evidenced by changes in chlorophyll concentrations, likely due to the very short residence time of water in this part of the river (<24 h). Furthermore, nitrate and suspended solids behaved differently, and there was no correlation in either river $p>0.05$ nor between nitrate and chlorophyll-a, $p>0.05$.

Downstream changes in measured variables in the Rangitaiki and Kaituna rivers could ultimately be related to land use within the lower catchment. The lower Rangitaiki River nitrate

concentration is largely dictated by seasonality within Lake Matahina and activities within the upper catchment, likely supplemented by groundwater influx and any minor drains discharging to the lowland river. The Kaituna River nitrate concentration is predominantly a product of the intensive agricultural catchment, with four significant tributary inflows along the lowland reach, all of which are nitrate rich. Nitrogen enrichment was sufficient to support non-native macrophyte growth, but other adverse effects associated with eutrophication were not realised in either river. This is likely due to a combination of the short residence times within both rivers, the abrasive flow preventing macrophytes accumulating sufficiently to block the channel and the turbulent, shallow water allowing ready reaeration. These results do emphasise the need for further study on lowland rivers to assess the effects of nutrient and sediment enrichment, and the extent to which existing guidelines on river are well suited to lowland reaches.

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Chapter 1

Introduction

Globally, freshwater management is often characterized by the need to balance environmental protection with the need to protect the interests of industry stakeholders (Knight, 2019). The protection of industry has been prominent, with global governments often seeking to “co-opt and depoliticize environmental ideals” (Blue, 2018) metamorphosing the goals of protecting the environment into something that still enables economic growth. This can be cloaked by attempts to make the natural environment measurable, setting limits for contaminants and system degradation which once in place, which can preclude the possibility for change, due to bureaucratic inertia. There has always been a tendency for government to overcome that inertia after serious damage has been done, because the political risk of dealing with environmental damage after it has happened is lower than an adoption of proactive approaches that constrain industry and yet offer no guarantee of success (Knight, 2019). The net result of these policies has been significant degradation of freshwater resources until empirical evidence is significantly compelling to catalyze positive policy change (Blue, 2018) by which time substantial intervention is necessary. Many countries, New Zealand (NZ) included, have struggled with this reactive approach to water management, particularly in terms of allocation and the control of point and diffuse sources of pollutants that are known to affect water quality (Rouse & Norton, 2017).

Contemporary water management in NZ is under the authority of regional councils which is ultimately stipulated by the *Resource Management Act 1991* (RMA) (Cullen *et al.,* 2006). The RMA gave power to regional councils to concomitantly manage land and water, and to develop a hierarchy of plans (national, regional, and local) intended to reflect social expectations of acceptable water quality (Knight, 2019; Rouse & Norton, 2017). However, the RMA has come under criticism due to its inability to manage cumulative effects of resource use and the discharge of pollutants. Historically, a parochial view of environmental effects was taken when granting resource consents for activities that result in discharges to water, which were on a case-by-case basis. This disregarded the synergistic effects of multiple discharges (Larned *et al.,* 2016; Davies-Colley, 2013; Howard-Williams *et al.,* 2010; Larned *et al.,* 2004; Niyogi *et al.,* 2007) and studies now recognize the need to better manage the effects of multiple agents on aquatic environments and aid the success of remediation efforts (Rouse & Norton, 2017).

Eutrophication of freshwater environments is often the result of multiple diffuse agents. These are typically anthropogenic in origin (e.g., fertilizers) and usually occur at levels that the environment does not experience naturally. The resulting effects are usually non-linear and challenging to predict. ‘Tipping points’ are thresholds beyond which results in the rapid degradation of ecosystem services (Gluckman, 2017), tipping points are commonly discussed but still controversial. What is evident is the inextricably interconnected nature of, biological, physical, and social processes, which requires a ‘macroscopic’ view, in the words of pioneering ecologist H.T. Odum (as cited by Nixon, 2009). Managing multiple stressors requires integrating knowledge across catchments and linking in ecosystem responses to ensure that individual stressors are given appropriate attention relative to their impact.

1.1 What is eutrophication?

The term eutrophication was coined by a German botanist, Weber in 1907, to describe the nutrient conditions and stages of biological productivity and plant succession in peat bogs (cited in Sawyer, 1966; Harper, 1992; Schindler, 2006). Swedish botanist Naumann in 1919 (cited in Harper, 1992) adopted the term to categorize lake trophic status (i.e., oligotrophic, mesotrophic, and eutrophic) based on a visual measure of water clarity. This was how the early researchers managed resources, as they were unable to measure nutrient concentrations directly, and categorize them upon the subsequent result (Harper, 1992). The word eutrophication, originated from the Greek word *eutrophe*, meaning ‘well fed’ and was thus originally used in limnology to describe the gradual accumulation of nutrients, sediment, and organic matter in a waterbody from inflows, atmospheric inputs, catchment topography, groundwater, and overland run-off. Such addition of nutrient and sediment eventually encourages prolific macrophyte growth, infilling, and the conversion of lakes into wetlands. Early perceptions that this process is ultimately the fate of all lakes are tempered by an understanding that the time scales that would be required to infill many large, deep lakes are inordinately long, and natural eutrophication remains a process that takes place mostly in shallow lake systems (Callisto *et al.,* 2014; Leng, 2009).

1.2 Cultural eutrophication

Contemporary use of the term eutrophication is no longer associated with the ‘gradual’ addition of nutrients and infilling of sediment. It is now associated with the undesirable effects of anthropogenic delivery of excess nutrients to waterbodies via industry, agriculture, and urbanisation. Such anthropogenic influences also known as ‘cultural eutrophication’ (hereon broadly termed, *eutrophication*), occurs much faster than natural processes and are associated with nutrient pollution and are largely independent of infilling (Leng, 2009; Smith & Schindler 2009; Khan & Mohammad, 2014). The undesirable effects of eutrophication are becoming more and more prevalent in waterbodies around the world (Withers *et al.,* 2014). Symptoms like prolific nuisance algal blooms interfere with the use of waterbodies for fisheries, recreation, and municipal drinking supply (Carpenter *et al.,* 1998). Furthermore, algal blooms can limit light penetration within the littoral zone, shading macrophyte assemblages, and limiting the success of piscivorous predators who rely on vision to hunt (Chislock *et al.,* 2013). Additionally, as algal blooms complete their life cycle and begin to senesce and settle through the water column, the aerobic decomposition process begins (Gantzer *et al.,* 2009), biological oxygen demand from the decomposition process reduces oxygen concentration, particularly in stratified waters causing fish kills (Carpenter *et al.,* 1998) and profound changes to the biogeochemistry of lakes.

Certain algal blooms, pose an extra threat due to toxin production. Certain species of cyanobacteria endogenously synthesize anatoxins and microcystins, which are known to be harmful to humans and animals alike (Chislock *et al.,* 2013). Health effects range from skin irritation, up to digestive issues, neurological impairment, and in some cases, death (Paerl & Huisman, 2009). Other species of cyanobacteria are benign but produce pungent odors, which cause drinking water supplies to be unpalatable (Paerl & Huisman, 2009). Climate change has been cited as a "potent catalyst for further expansion of these blooms" (Paerl & Huisman, 2008) as cyanobacteria are expected to have a competitive advantage due to their better ability to withstand warmer and more strongly stratified conditions that are anticipated to accompany climate warming (Wagner & Adrian, 2009).

Eutrophication in aquatic environments has an economic cost as well as an ecological cost. An analysis from Dodds *et al.,* (2009), puts the annual monetary losses in the United States from eutrophication at $2.2 billion (USD). This is attributed to the increased drinking water treatment needs, losses in amenity value, losses in tourism, and decreased real estate valuation. Another analysis from Pretty *et al.,* (2002), deemed the annual cost of social damage using similar costings, from eutrophication in the United Kingdom to be between £75.0-114.3 million (GBP). Furthermore, Pretty *et al.,* (2002), calculated the cost of managing nutrient load, as well as the cost of developing and implementing strategies combatting eutrophication at £54.8 million GBP annually. These figures adjusted for present day inflation would sit at $2.72 billion USD, and £109.4 - £166.2 million GBP annually, with annual compliance and future proofing strategies at £79.9 million GBP.

1.3 Eutrophication of riverine systems – New Zealand and Abroad

The directional movement of water defines river ecosystems, and unidirectional flow influences the "morphology, sedimentation patterns, water chemistry, and biology of organisms" within the system (Wetzel, 2001). In the foreword of *The Ecology of River Systems,* the authors cite that limnologists' do not entirely understand the ecology of large rivers (Davies & Walker, 2013). Researchers seek to conceptualise how river systems work, e.g., the River Continuum Concept, (see Vannote *et al.,* 1980), but such frameworks do not always fit neatly to each individual riverine system and concepts developed in some locations do not transfer readily to others (Davies & Walker, 2013). For example, paradigms relating to pulsed organic material developed in temperate northern regions dominated by deciduous vegetation do not always apply in NZ where most native vegetation is evergreen. This further means that there are not comparable invertebrate assemblages within the river reaches meaning that ecosystems behave differently in New Zealand than, for example, in Europe. Furthermore, humanity has often imposed severe pressures on the aquatic environment through nutrient enrichment (Smith *et al.,* 1999), before river ecologists had any understanding of how impactful such pollution can be (Davies & Walker, 2013). A consequence of the lack of universal paradigms for river ecosystem function is that the the process of eutrophication is better understood for lakes, reservoirs, and estuaries, then in rivers. Consequently, there has been a tendency to borrow concepts for use in rivers that come from lakes (Hilton *et al.,* 2006).

1.4 Eutrophication development between systems

Concepts central to eutrophication that are well developed in lakes are thus now applied to rivers where analogous concerns exist. However, the biological response to elevated nutrient levels within rivers can be very different to a lake or reservoir because of the way in which nutrients are processed in an open, longitudinal system rather than a nearly closed basin (Wetzel 2001). The nutrient cycling paradigm in lakes is weakly applicable in rivers and streams and has been modified to give rise to the concept of nutrient spiralling (Clarke, 2002), which adds longitudinal flow to the biological transformation of nutrients as waters flow from source to sink. This process is influenced by residence time (retention time). Residence time is how long water (or substance) remains in the hydrological system and is critical to system response (Feng *et al.*, 2018; Harding, 2004). Residence time varies between river basins due to differences in topography, geomorphology, and inflow rates (Feng *et al.*, 2018). Specific zones within a stream or river, such as the benthic boundary layer and hyporheic zone have an extended retention time known as transient storage, which is the period that the water spends within the zone (Harding, 2004). The dynamics of these zones are closely related to discharge, stream geomorphology, and the lengths of transient storage times have implications for entire stream reaches due to their effects on the ability of microbial activity within the zones to turnover nutrients and carbon and return them to the water flow (Drummond *et al.*, 2016).

The river continuum concept (RCC) is linked to nutrient spiralling and proposes a variety of processes that can alter nutrient cycling (i.e., delivery, uptake, transformation, retention, dilution) and can change with relative importance with travel downstream. As discussed above, the RCC is a neat framework that describes the chemical, biological, and physical changes riverine systems go through from source to sink (Doretto *et al.*, 2020) despite its inability to transfer concepts verbatim to Southern Hemisphere rivers (Winterbourn *et al.*, 1981). The RCC splits the entire reach into sections (following Strahler order; Strahler, 1957), headwaters i.e., order 1-3; middle reaches, order 4-5; lower reaches, order >6; each section has fundamental differences which determines the structure of the physical and biological environment dictating aquatic community assemblage.

The headwaters are an intrinsic part of the stream environment, playing an important role in the biogeochemical connectivity and transport of materials downstream which link terrestrial and freshwater environments (Jarvie *et al.,* 2018). Headwaters have high gradients with fast flowing turbulent water, with narrow channels enclosed by riparian vegetation, creating lower light penetration and lower stream temperature thus restricting algae growth. Differences between terrestrial stream additions and in-stream autotrophy reveal the broad scale changes as the river transitions through the middle reaches. The middle reach is where the gradient begins to flatten, residence time increases, and the stream begins to widen, additionally, gross primary production (GPP) begins to exceed community respiration, this happens as rivers transition into "light-rich" catchments (Cummins & Klug 1979). Macrophytes become a prominent resource in this section, the riparian zone no longer stretches across the channel creating diel fluctuations in the physicochemical makeup of the river due to the paucity of shade.

The lower catchment is an area of the catchment where tidal influence is a significant control on community structure, marine and freshwater processes begin to interact creating a unique set of instream conditions. The channel is widest and deepest, often with expansive floodplain areas and a low gradient creating the highest residence times across the river continua. Additionally, the lowland catchment generally has an increased availability of nutrients through land use discharges to the stream as well as large daily fluctuations with high maximum values of physicochemical parameters (Wilcock *et al.,* 1998). These values can also be exacerbated by the displacement of native macrophyte species by invasives (i.e., *Lagarosiphon major, Ceratophyllum demersum*) in turn creating a harsh set of instream conditions.

1.4.1 Estuaries

The ecology of receiving environments i.e., estuaries is dictated by both upper catchment and coastal processes (Drake *et al.,* 2011; Plew *et al.,* 2018) but the trophic state is predominantly a product of the nutrient and sediment load from upstream. Estuarine ecosystems are at risk of eutrophication, owing the position at the lower limits of catchment. Levin *et al.,* (2001) terms receiving estuarine environments “critical transition zones” (CTZs) recognizing them as zones where terrestrial, marine, and freshwater processes interact. Stressors (e.g., excess nutrient supply or sediment supply), can synergistically interact and amplify the deleterious effects on receiving environments (Thrush *et al.,* 2008). This is a major issue as under pristine condition in estuarine systems nutrients are typically limited, meaning primary productivity is constrained by the amount of available nutrients (Harding, 2004). Additionally, estuarine ecosystems are intricately balanced with internal feedback loops within the system, interacting with biotic and abiotic processes that can be easily disrupted (Levin *et al.,* 2001). If eutrophic conditions do not resolve, the ultimate consequence for estuaries is ‘the uncoupling of production and use of organic matter in aquatic systems’ (Maier *et al.,* 2009).

1.5 Sources of nutrient enrichment

Nitrogen (N) and Phosphorus (P) are both crucial elements in rivers, streams, and estuaries; nitrogen is a major component for amino acids, which are the building blocks of proteins. P is essential in the process of making adenosine triphosphate (ATP) and nicotinamide adenine dinucleotide phosphate (NADP) which are required for energy management that drives cell processes and for assembly of DNA and RNA, which control those processes (Wetzel, 2001). Although, N and P are crucial elements within aquatic systems and one or both is usually the most limiting nutrient for growth, constraining primary productivity in those systems.

The delivery of these nutrients largely reflects the dominant land use practice(s) in the catchment (Scarsbrook & Halliday, 1999; Wilcock *et al.,* 2013). To understand eutrophication as a product of dominant land use, it is useful to split enrichment into two categories, point source (PS), and non-point source (NPS). Historically, PS pollutants such as industry outflow, sewerage outflows, and municipal wastewater discharges have received a considerable amount of attention due to the damage on the receiving aquatic environment(s), and usually PS pollution is easier to identify (Smith *et al.*, 1999). As PS pollutants are increasingly well managed, NPS pollutants such as, overland run-off, sewerage overflows and leaks, rubbish dumps, spills, forestry activity, mine tailings and run-off, atmospheric deposition, and groundwater leachate are receiving far more attention as NPS pollutants are much more of a challenge to manage (Duda, 1993; Howard-Williams *et al.*, 2010). Diffuse sources of N and P account for 82-84% of nutrient delivery to waterways in the US (Wurtsbaugh et al 2019).

Primary industries are now identified as significant contributors to NPS pollution through fertilizing crops and pasture (Khan & Mohammad, 2014). Surplus nutrients enter waterbodies through run-off and groundwater, creating harsh changes to the stream environment that have wide ranging effects on the biology and ecosystem services (Niyogi *et al.*, 2007). Agricultural practices are reported to contribute 53 and 48% of the diffuse N and P flux to waterways in the United States (US) (Wurtsbaugh et al 2019). Nitrogen and Phosphorus can also enter through erosion of unstable sediments, and livestock entering the stream (Howard-Williams *et al.*, 2010). Additionally, nutrients can also enter rivers through leaves, woody debris, and other organic matter through remineralization via heterotrophic metabolism.

1.6 Adverse effects of eutrophication in rivers and streams

Historically, rivers were thought of as insensitive to nutrient enrichment due to biotic and abiotic factors that regulated algal growth within rivers (Smith *et al.*, 1999). However, experimental studies showed that elevated levels of Nitrogen (N) and Phosphorus (P) led to proliferation of primary producers (Carpenter *et al.*, 1998; Khan & Mohammad, 2014; Schindler, 2012; Withers *et al.*, 2014) and impacts on community structure at differing levels of biological organisation (Canning *et al.*, 2021). In the latest Environment Aotearoa report (2019) default guideline values

(DGVs) were used as estimates of concentrations of N and P that would occur under natural catchment conditions, that is with no change in land use. For rivers within current pastoral catchments, 86% of the total river length exceeded their DGVs for N as well as 29% total river length in residual native catchments. Furthermore, 90% of total river length in pastoral catchments exceeded P DGVs, and 26% of total river length in native catchment exceeded P DGVs (Ministry for the Environment & Stats NZ, 2019).

Nutrient enrichment results in a variety of water quality issues in rivers (Table. 1.1). The Murray-Darling river in south-eastern Australian is a large slow-moving river which suffered “the largest river bloom of blue-green algae recorded anywhere in the world” (Murray-Darling Basin Ministerial Council, 1994). This led to livestock deaths and additional concerns for human health. Aesthetic impairment of rivers is another symptom of over enrichment and excessive algal growth, which is usually filamentous in rivers in smaller, hard-bottomed rivers (Dodds & Welch, 2000). Such alterations cause the degradation of water resources and loss of amenity value (Smith, 2003), furthermore, it decreases property valuation (e.g., Dodds, 2009). Additionally, prolific algal growth can create poor conditions instream for biota, with large diurnal fluctuations in pH from photosynthetic processes, and high rates of respiration stripping O_2 from the water column at night creating a sag with the potential to suffocate fish and macroinvertebrates. This illustrates how nutrient enrichment can adversely affect stream animal communities, altering them away from their natural state (Dodds & Welch, 2000).

Table 1.1 A list of undesirable effects of excessive additions of N and P to freshwater systems, from (Smith & Schindler 2009; Carpenter *et al.*, 1998).

- Increased primary production – greater biomass of phytoplankton and macrophytes
- Increased consumer species
- Shift to cyanobacterial communities – potential for toxic *spp* proliferation
- Changes in species composition of macrophyte assemblages
- Decreased species diversity
- Increased turbidity
- Potential water treatment issues i.e., taste/odour
- Oxygen depletion and potential increased incidences of fish kills
- Lower amenity value of lakes/rivers

1.7 Measuring trophic state in rivers and streams

The measure of eutrophication in lakes is historically based on the extent to which phytoplankton biomass stimulated by excess additions of N and P, usually measured as *chl-a* concentration. In large, slow flowing rivers, such as the Murray-Darling described above, this can still be appropriate, but in other lotic environments proliferations are often benthic. This particularly applies to upper reaches within the river continuum, where low order streams are often small, fast flowing a gravel or cobble bedded. Naturally, an initial attempt to translate lake derived concepts of trophic state to rivers, using biomass involved transfer of criteria to benthic algal biomass (Dodds & Smith, 2016, Table. 1.2).

As discussed above, due to the fundamental differences between lotic and lentic systems, the transfer of concepts from lakes to rivers is rarely simple, and a new view of quantifying trophic state is required. *In situ* nutrient diffusing experiments showed that N and P, both on their own and together influenced growth rates of algae in cobble-bedded streams and rivers, but weak relationships between in-water nutrient concentration and benthic biomass were observed (see Francoeur, 2001). Unlike most lake phytoplankton populations, biomass accruing to the benthos in streams is vulnerable to other variables, and unlike the nutrient/chlorophyll phytoplankton relationships are therefore much weaker to lakes (Dodds & Smith, 2016). Periphyton (benthic algae) accrues over time in streams and can be a good indicator of trophic state at lower flows, but stochastic flood flows abrade periphyton thereby restarting successional processes. Meaning biomass is determined by accrual rate (which can be influenced by nutrient supply) but also time since last flood. The fact that instream nutrient enrichment and expected increases in algal biomass can be limited by dynamic factors other than N and P, makes biomass a less reliable metric for assessing trophic state of rivers. Additionally, primary production can also be limited by light availability (Gregory 1980), turbidity (Smith, 2003), and hydrology, (Smith *et al.*, 1999). Most other metrics for assessing eutrophication incorporate physicochemical (i.e., turbidity, clarity, dissolved nutrients, dissolved oxygen) and biological indices (e.g., macroinvertebrates, macroalgae) which provide a basis for management decisions (Ferreira *et al.,* 2011) for which knowing the historical condition of the river system is useful.

Table 1.2 Suggested nutrient parameters for classing trophic status in rivers and streams based on data collected within the United States (US). Originally cited in Dodds *et al.,* (1998), taken from Dodds & Smith (2016).

Variable	Oligotrophic	Mesotrophic	Eutrophic
➢ Mean benthic chlorophyll (mg m^{-2})	<20	20-70	>70
➢ Maximum Benthic chlorophyll (mg	<60	60-200	>200
➢ Suspended chlorophyll (µg L^{-1})	<10	10-30	>30
➢ Total N (µg L^{-1})	<700	700-1500	>1500
➢ Total P (µg L^{-1})	<25	25-75	>75

To address this, Dodds (2006) proposed that trophic state be inclusive of heterotrophic and autotrophic facets, to wholly consider stream processes. Thus, community-based responses are important when assessing the trophic state of a stream (Dodds & Smith, 2016), and Dodds (2006) suggests that eutrophication in rivers and streams be defined as “an increase in a nutritive factor or factors that leads to greater whole-system heterotrophic or autotrophic metabolism”.

Therefore, it would be negligent to only consider autotrophic response when accounting for nutrient enrichment in river systems, as emerging research on heterotrophic, and autotrophic state classifications are gaining more focus from river ecologists (Dodds & Smith, 2016; Wurtsbaugh *et al.,* 2019). Accordingly, integrating biological (i.e., *chl*-a) and physicochemical parameters (i.e., dissolved nutrients) into studies allows quantification of autotrophic and heterotrophic states of production, as N and P are proven regulators of freshwater primary production stimulating both heterotrophic and autotrophic production (Dunck *et al.,* 2015; Ferreira *et al.,* 2015)

1.8 The current issue of nitrate in New Zealand

Nitrogen is a fundamental macronutrient of aquatic ecosystems, which enables nucleic acid formation, as well as being a major component of amino acids; nitrogenous compounds are also found in various other organic materials. It is now estimated that global nitrogen fixation, that is the conversion of dinitrogen gas to mineral form, is ~413 Tg (Tg = 10^{12} g) and over half of that is anthropogenic (Fowler *et al.,* 2013). Overwhelmingly, the largest source of anthropogenic nitrogen is synthetic fertilizers and animal excreta, while not directly anthropogenic in its origin but when applied perpetuates N pollution (Ward *et al.,* 2005). Humans have made significant alterations to the global N cycle through anthropogenic additions of biologically reactive N through energy production, fertilizer addition, and cultivation of crops (Wetzel, 2001; Vituosek *et al.,* 1997). Furthermore, nitrate is water soluble and highly mobile in soils, what is not assimilated by plants and other organisms will eventually percolate through soil and contaminate surface and groundwaters (Ward *et al.,* 2005). Nitrate, primarily of agricultural origin, has been singled out as one of the biggest challenges to manage for the NZ environment (Parliamentary Commissioner for the Environment, 2015).

The effect of nitrate on the environment is well documented. Eutrophication impacts have been discussed above, but nitrate is known to have toxic effects on aquatic animals. Toxicity effects are species-specific and dependent on concentration and exposure time (Camargo *et al.*, 2005). Toxic effects of nitrate exposure range from growth inhibition to mortality; Camargo *et al.*, (2005) stated that concentrations of 10 mg NO_3^-N/l will have chronic effects on species such as Rainbow trout (*Oncorhynchus mykiss*) and Chinook salmon (*Oncorhynchus tshawytscha*) which are both present in NZ freshwater. Furthermore, decreases in pH from nitrate addition can cause further changes in water chemistry and the functioning of the ecosystem. A report detailed that in freshwater lakes which underwent a decrease in pH increased in aluminium (Al^{3+}) concentrations (Nelson & Campbell, 1991) which can disrupt P cycling within freshwaters thus causing ecosystem perturbations (Bibi *et al.*, 2016). Lastly it is worth noting that nitrate has been shown to have adverse impacts on human health (see Fewtrell, 2004) however, will not be addressed directly as it is outside the scope of this study.

The characteristics of New Zealand rivers and streams are constrained by its size, topography, and climate; notably, NZ has an absence of large river systems. New Zealand is a long, thin, and mountainous island nation spanning 13 degrees of latitude on the boundary of two major tectonic plates (Gluckman *et al.*, 2017). This makes NZ's climate spatially variable with conditions ranging from sub-tropical in the north to cooler temperate in the south, with certain areas reaching alpine conditions on the mountain ranges (NIWA, n.d.). Due to the wide environmental variability of NZ, its river catchments display heterogenous biological, ecological, and physical characteristics (Gluckman, 2017; Snelder & Biggs, 2002). However, nowhere in NZ is more than 130km from the coastline (Statistics New Zealand, 2006) and downstream changes in river character occur on short time and distance scales. Freshwater in NZ has been and continues to be an integral asset to New Zealand's economic development. Freshwater plays a vital role in primary sector industries, and freshwater is also vital to tourism, recreation, power generation and cultural identity (OECD, 2017; Davies-Colley, 2013).

However, as discussed above, many lakes and rivers in NZ are degraded by unnaturally high levels of N and P and public concern is growing over the quality and state of freshwater resources in NZ (Hughey *et al.*, 2019). The latest freshwater report in the Ministry for the Environments' environmental reporting series surmised that 95% of all river and stream reaches in pastoral land cover exceeded one or more guideline values for pollution. Nitrate-N was between 11-18 times higher in pastoral catchments, DRP was also threefold higher in pastoral catchments when compared to native forest catchments (Ministry for the Environment & Stats NZ. 2020).

In NZ, intensive agricultural practices are a significant contributing factor to the surface water quality declines (Howard-Williams *et al.*, 2010; Larned *et al.*, 2004) and the agricultural sector has drawn heavy criticism. Despite the important socioeconomic role that agriculture has in NZ society is becoming less tolerant of the wider off farm impacts of nutrient leaching, the impacts on lakes, and the impact on rivers, and estuaries (Clark *et al.*, 2007). The agriculture industry indeed has been and continues to be a key economic driver for NZ, which contributes billions of dollars to NZ's tradeable economy. For instance, the dairy industry alone contributed $19.7 billion in export revenue in 2020, whilst sustaining 50,000 jobs, which is approximately 1.7% of the NZ workforce (Dairy NZ, 2020). Moreover, agriculture accounts for 4.7% of employment in NZ (Ministry for the Environment & Stats NZ, 2019).

Dairy is, however, often singled out as the main contributor to environmental degradation in NZ due to its meteoric expansion and intensification over the past two decades (Foote *et al.*, 2015; Ma *et al.*, 2018). Since 2002, the land area used for dairying almost doubled from 1.2 million ha to 2.2 million ha in 2019 (Stats NZ, 2021a), with dairy cattle growing from 5.1 million to 6.2

million cattle over the same period (Stats NZ, 2021b). Over the same 1990-2020 period the average herd size increased more than threefold from 140 to 440 animals (DairyNZ, 2020). The use of supplemented feeds has increased, which highlights the move to more intensive production for dairying in NZ. Within pastoral agriculture land uses, dairy farming also has the largest diffuse nutrient footprint meaning it has a disproportionate impact on the receiving environment compared to the land area occupied (Gluckman, 2017). Catchment-scale modelling showed dairy also has a disproportional contribution to the N flux to the coastal ocean in NZ. Approximately 37% of N flux to the coastal ocean originated from dairy farms, which occupy only 6.8% of the land area (Elliot *et al.,* 2005).

As farming has intensified the requirement for greater feed to support the larger herds, has led to an increase in the use of N fertilizer. From 1991 to 2019 New Zealand increased its nitrogenous fertilizer application to land from 6.2 x 10^4 tonnes to 4.52 x 10^5 tonnes in 2019 (Figure. 1.1), which is a 629% increase (Stats NZ, 2021). Research shows that economically efficient use of nitrogenous fertilizer is low, on average 80% of nitrogen applied for agricultural purposes is diffusely lost to the environment (Sutton *et al.,* 2013). Furthermore, the latest OECD environmental performance review of NZ determined that New Zealand has the worst nitrogen balance of all member countries. Nitrogen balance is effectively input i.e., fertilizers vs the output i.e., uptake of nitrogen for crop/pasture growth (OECD, 2017). NZ' shows a 2% annual increase in nitrogen balance between 1998 – 2000 to 2007 – 2009.

Phosphorus application adds to agricultural productivity but also compounds the degradation of surface water; between 25-75% of P applied to land is lost to the environment (Sutton *et al.*, 2013). In NZ the primary source of P fertilizer is superphosphate mainly used on sheep and beef farms; nationally the sheep and beef land use continue to decline steadily however, the decrease has been matched verbatim by the increase in dairy land use expansion (Parliamentary

Commissioner for the Environment, 2015). Although between 2006 and 2019 NZ has seen an 18.5% decrease in Phosphorus fertilizer application from 1.89×10^5 tonnes to 1.54×10^5 tonnes (Figure. 1.2; Stats NZ, 2021). Likewise, trend analysis from 2004-2013 in catchments dominated by intensive grazing showed 57% of monitored sites had decreasing levels of dissolved reactive Phosphorus (DRP), 15% showed increases, and 29% showed no change (McDowall *et al.,* 2019). Which overall could be attributed to the decline of sheep and beef farming, the increase in P price, or successful implementation of nutrient management strategies.

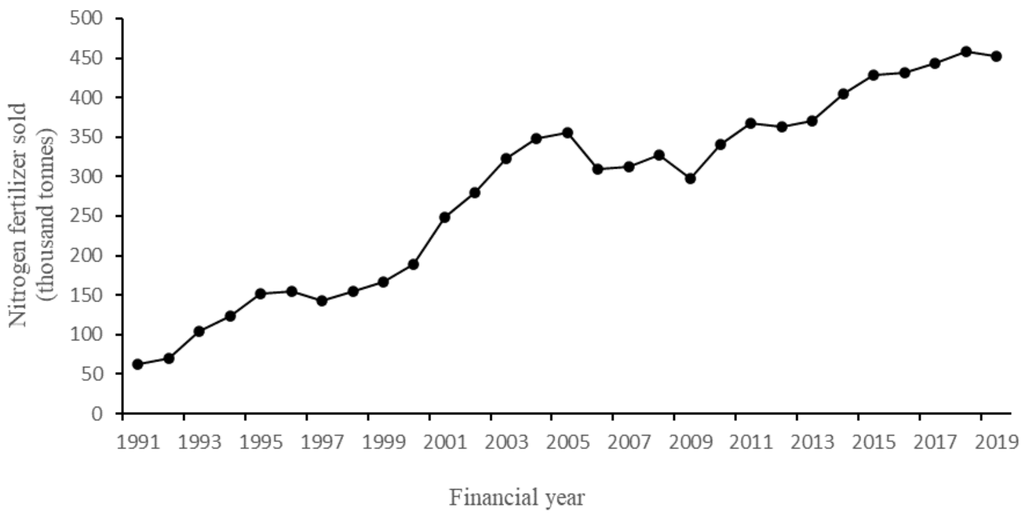

Figure 1.1 Trend of nitrogen fertilizer sold by weight (thousands of tonnes) between 1990-2019 (Data courtesy of Fertiliser Association, n.d.).

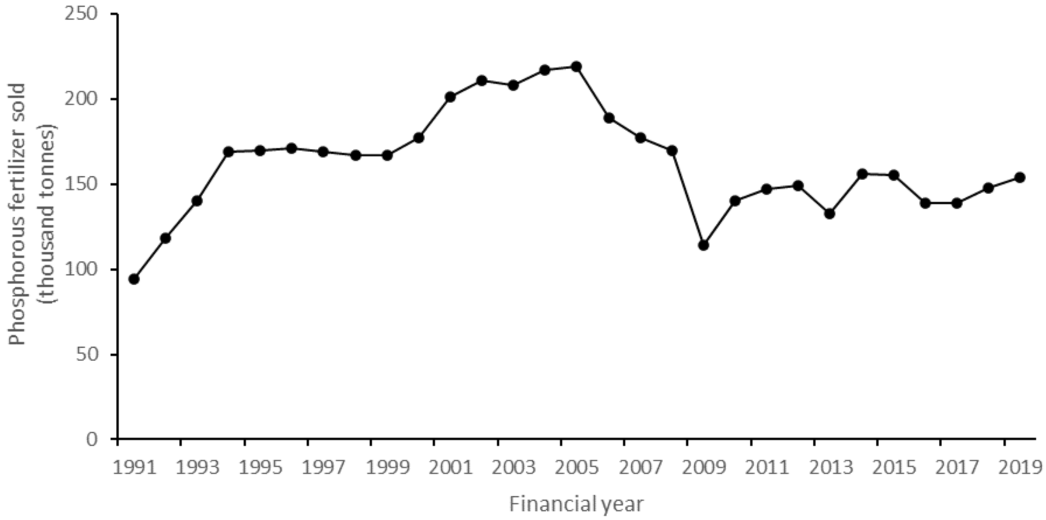

Figure 1.2 Trend of Phosphorus fertilizer sold by weight (thousands of tonnes) between 1990-2019 (Data courtesy of Fertiliser Association, n.d.).

1.9 The challenge of managing intensive agriculture in New Zealand

Given the unseen nature of the issue of diffuse nutrient pollution policy makers worldwide struggle to find effective regulatory standards especially for N (Doole *et al.*, 2013). Between 60-99% of nutrients assimilated by cattle will be returned to pasture, overwhelmingly, urine is the largest source of N returned to pasture with approximately 70% being urea (Haynes & Williams, 1993). Furthermore, what is not assimilated into plant or other organic matter will remain in the soil profile and high rates of N leaching can occur (Di & Cameron, 2002) especially following a high rainfall event (Parliamentary Commissioner for the Environment, 2015).

Nitrogen is one of the biggest water quality challenges to manage in New Zealand, being described as the 'elusive' nutrient due to its soluble and reactive nature (Parliamentary Commissioner for the Environment, 2015). Recent research which has examined the potential to reduce N leaching through improved dairy farming system changes; small-scale farm trials were run in three major dairy farming regions in NZ (Waikato, Canterbury, Otago) (see Beukes *et al.*, 2019). Current farming systems were used as a control alongside the improved systems. Trials used reduced stocking rates but higher producing cattle, reduced N fertilizer application, use of high energy/low N feeds, reduced herd replacement rates, as well as utilizing off-paddock facilities. What was found over all was ‘that the lower input farms (less N fertilizer and low N feed) successfully reduced N leaching 24-30%, but profitability of these systems were impacted, although this was found to be regionally specific. Due to this there may be a reluctance to take on advancements in infrastructure for waste management due to the cost or otherwise, this also highlights the need for innovative changes to exceed the performance of current systems for profitability as well as meeting environmental targets. Where the move away from strict rule based legislative approaches may aid the innovation of agricultural practices in NZ. Pastoral agriculture is a critical part of the economy in NZ and will continue to be into the foreseeable future notwithstanding the persistent social and political pressure to reduce its environmental footprint. New Zealand is world renown for the ‘clean green’ image but if the current application of N fertilizers, and current dairying practices continue the issue of N leaching will be greatly exacerbated perpetuating the issue of surface and ground water pollution NZ is faced with.

New Zealand’s environment is now suffering from diffuse pollution from multiple agents that make cause and effect relationships challenging to quantify due to the lack of data. Multiple studies recognise the compounding negative effects of multiple land uses which is generally reflected by the water quality of lowland rivers. This is owed to the lack of consistent water quality monitoring over long periods of time. The lowland reaches of rivers are where tidal influences and rapid deceleration of water is common, thus pollutants can become hydrodynamically trapped, or redirected into off-channel habitat interacting with dissimilar processes to the main river channel. Often such locations are non-wadable, but numerical and theoretical modelling works well examining and predicting relationships between differing land uses and pollution, although, large lowland rivers are heterogenous and require empirical quantification to assess the trophic state in the river channel (Julian *et al.*, 2017).

NIWA operates The National River Water Quality Network (NRWQN) which commenced in 1989. The NRWQN consists of 77 sites on 35 rivers across the North and South Island monitoring physicochemical variables (i.e., dissolved oxygen), optical variables (i.e., turbidity), nutrient levels (i.e., total N & P) as well as E-coli (since 2005). However, what is currently monitored is hard bottomed, mostly wadable, and easily accessible with a notable absence of data from larger, lowland soft-bottomed rivers and other non-wadable sites in NZ. Since NIWA began operating the NRWQN, there have been only two national-scale assessments of lowland river water quality (see Larned *et al.*, 2004; Larned *et al.*, 2016). The national policy statement for freshwater (NPS-FW) stipulates attributes requiring limits on resource use for rivers, lakes, and streams in NZ. The limits to key drivers of ecological change are underdeveloped and overlooked for lowland rivers in NZ, creating ambiguity around adequate management where the management practices are intended to reflect the social expectations of adequate water quality for contact recreation. Addressing these issues is a critical step toward preserving natural character of the stream environment, human health and ecosystem services which is already under threat by climate change. This study will seek to elucidate in stream processes in two Bay of Plenty rivers, the Rangitaiki and Kaituna. As discussed above, the absence of consistent monitoring data from similar locations to this study, and the underdeveloped limits to the drivers of environmental change inhibit positive policy change only and only prolong the issue. This study will provide a basis for decisions to be made regarding the management of the Kaituna and Rangitaiki rivers.

1.10 Aims and hypotheses

The concept of managing downstream reaches within rivers requires an intimate knowledge of what activities affect the concentration of contaminants, and how they change across whole river reaches. Such data can be used to improve our current understanding of nutrient and sediment dynamics in lowland rivers, where we currently studies have been relatively sparse. In this study, sampling will provide empirical evidence on locations within catchments where contaminant concentrations change. We will use Eulerian sampling methods to investigate the NO_3^- and suspended sediment dynamics of two major Bay of Plenty rivers, the Rangitaiki and Kaituna, and at the same time collected data on other water quality variables, pH, dissolved oxygen, planktonic chlorophyll-a, and turbidity. The aim is examining the relative importance of controlling processes, seeing if they result in discontinuous downstream levels of contaminants through attenuation or enrichment. We hypothesize: 1) that catchment-related processes will influence TSS, Chl-a, and NO_3^- levels along the river reaches displaying discontinuous changes downstream in both rivers coincident with confluences or changes in land use; 2) that NO_3^- and chl-a concentrations will have a positive correlation, with stronger relationships observed in spring and summer months 3) that TSS will not be closely correlated to NO_3^-; due to the differing contaminant pathways.

Chapter 2
Methodology

2.1 Site description

Both the Kaituna and Rangitaiki are designated by the Bay of Plenty Regional Council (BOPRC) as major rivers within the region. The Kaituna and Rangitaiki are regulated rivers meaning they are controlled upstream for human purposes (i.e., hydropower, flood control), the Rangitaiki has three major hydro-electric schemes along its reach, and the Kaituna has flow control gates at Okere Falls.

The Rangitaiki River runs from the Kaimanawa Ranges and discharges at Thornton; the Rangitaiki is also the longest river in the BoP (115km), as well as having the largest catchment (2947 km^2) (Brown, 2018). The river has three main tributaries that all come to a confluence in the upper reaches of the catchment. A significant portion of the Rangitaiki catchment is in forestry (80%) (Figure. 2.1) which is predominantly exotic plantation (Brown, 2018), the remainder of the catchment use is mainly pastoral, with dairying prominent, native forest, and horticulture are the remaining uses as well as a major dairy processing plant in the Edgecumbe township (LAWA, n.d.).

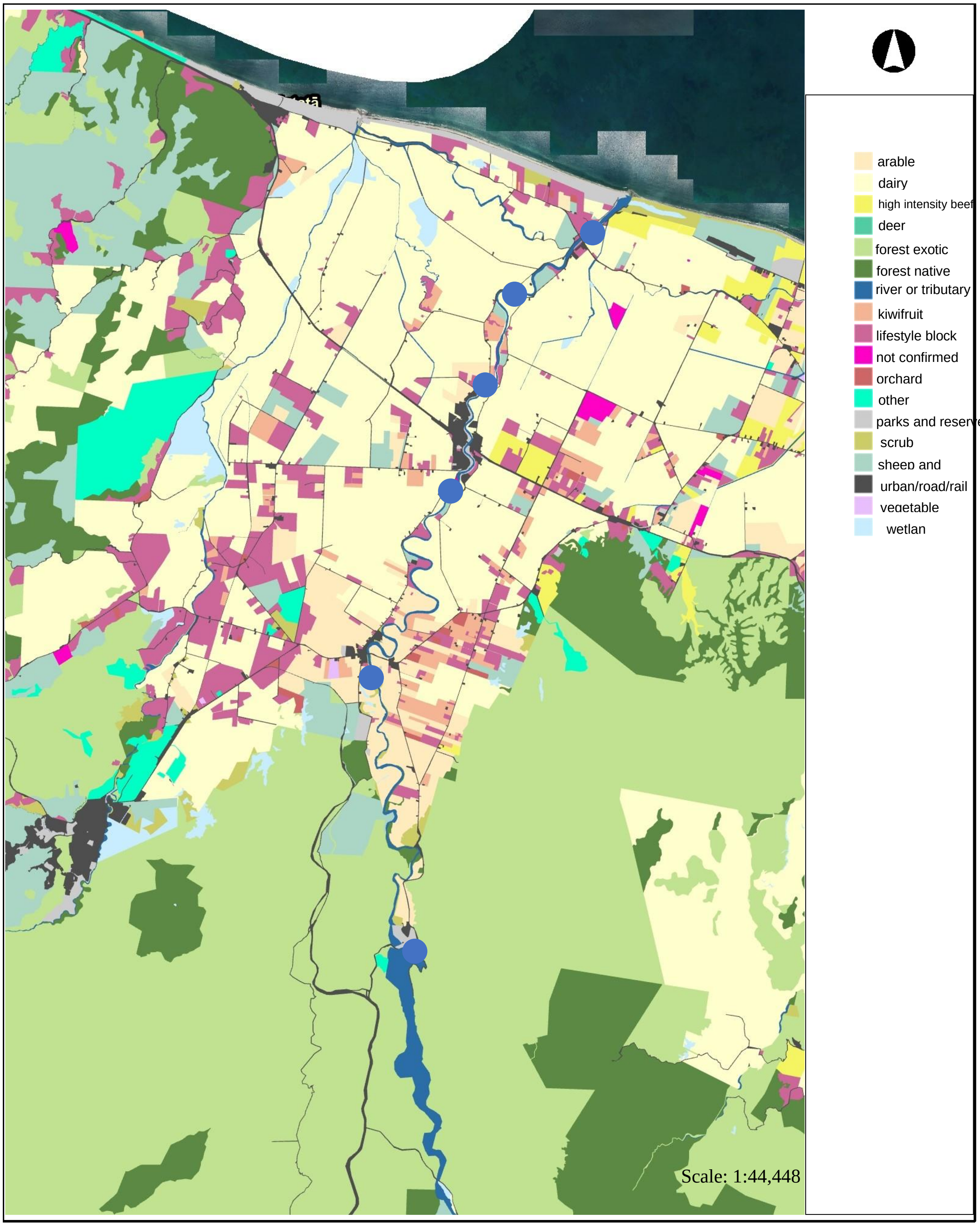

Figure 2.1 Dominant land use for the lowland area of the Rangitaiki river catchment, data courtesy of the Bay of Plenty Regional Council. Blue markers indicate sampling locations in the main channel.

The Kaituna River is approximately 50km in length with a total catchment area of 1218 km^2, the river drains Lake Rotoiti from Okere Falls and discharges to the sea at Te Tumu (White *et al.*, 1978). The Kaituna also drains Lake Rotorua via the Ohau Channel due to a project commissioned by the Bay of Plenty Regional Council diverting nutrient rich water directly down the Kaituna in a move to remediate the water quality of Lake Rotoiti (Hamilton *et al.*, 2009). The upper section of lower catchment is primarily pastoral and forestry land uses, the mid sections mainly used for horticulture (predominantly kiwifruit) and the lower sections productive grassland where the dominant land use is dairy farming (Park, 2007, Figures. 2.2 and 2.3). Additionally, within the lower catchment there is a meat processing works which has consent to discharge wastewater to the river (LAWA, n.d). In the lower section of the Kaituna there are two drainage discharges in between sites four and five, the lower drain (LD) which is fed from the Waikoura stream and the upper drain (UD) which is fed from the Raparapahoe stream; at site four the Waiari river (lower tributary (LT)) also joins the main channel. In the upper section the Mangorewa river (upper tributary(UT)) joins the main channel below site one.

Figure 2.2 Cattle in the Kaituna River, photo captured by Nicholas Wilson, March 2021.

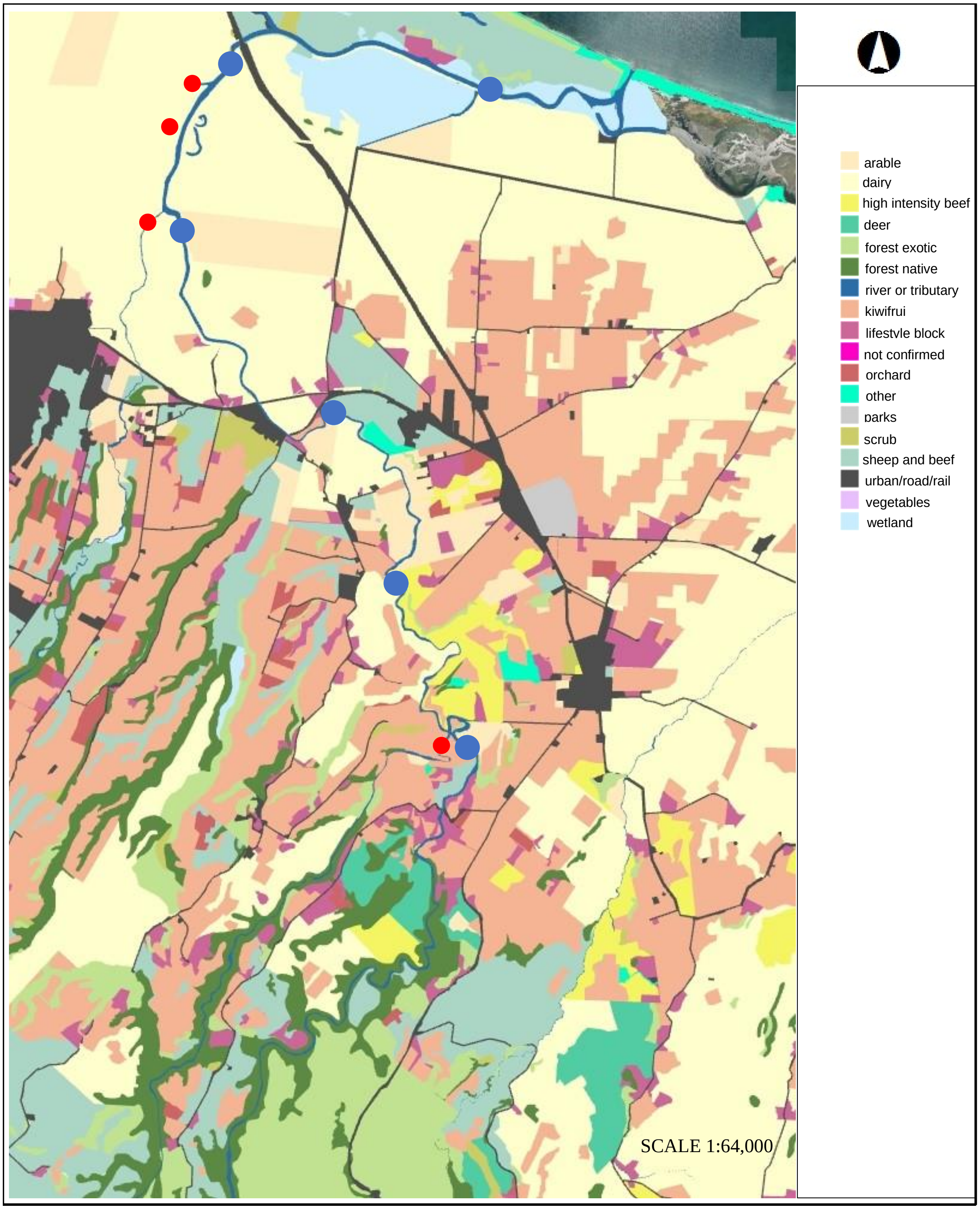

Figure 2.3 Dominant land use for the lowland area of the Kaituna River catchment, data courtesy of the Bay of Plenty Regional Council. Red markers indicate the locations of the drains and or tributaries sampled through this project, blue markers indicate the main channel sampling sites.

2.2 Field sampling methods

2.2.1 Eulerian sample collection

Samples were collected on both rivers between November 2020 and December 2021. Each river had six sites systematically selected along their reaches, water samples and physical measurements were collected and repeated at each site. The sites were a minimum of 1 km apart to cover as much of the lowland area as possible (Tables 2.1 and 2.2). Each site was selected as a point of interest i.e., above or below a township, industrial plant, above and or below a tributary. Water samples were collected, and physical measurements taken at six different points (Figures. 2.4 & 2.5) along each river, each process specified below was carried out at each location. An additional four nitrate samples were taken on each sampling occasion on the Kaituna River excluding the 2-Dec. These samples were taken from the lower drain (LD), the upper drain (UD), the lower tributary (LT), and the upper tributary (UT) There are no notable tributaries present on the Rangitaiki River below the Matahina dam therefore such measurements are omitted for the Rangitaiki. Water samples for total suspended solids (TSS) were collected approximately 5-10 cm beneath the surface of each site in 2 litre (2 L) bottles respectively. TSS bottles were rinsed three times prior to each sample being collected to guarantee accuracy of results. Each bottle was labelled with the respective site location and stored in a cooler until the samples could be processed. For the chlorophyll-a (*chl-a*) samples, a syringe was used to filter 300 mL of water through a 0.2 µm Whatman GF/C glass microfiber filter. Each filter paper was folded in half, each placed in an individual plastic container, labelled, and placed in the cooler out of direct sunlight until the samples could be stored in the laboratory freezer. In addition to this, the first 50 mL of each *chl-a* sample was filtered into a falcon tube for nitrate analysis, the tubes were labelled, and placed in a cooler until the samples could be stored in the laboratory freezer.

Dissolved oxygen, temperature, conductivity, pH, and turbidity (NTU) were measured in situ with an AquaTROLL 600 Multiparameter Sonde.

Table 2.1 Rangitaiki River site GPS coordinates and the cumulative distance from the first sampling location to the lowest site on the river reach.

Site	GPS coordinates	Distance from site one (km)
1	38° 6'54.16"S 176°49'11.73"E	0
2	38° 2'9.53"S 176°47'55.96"E	12.51
3	37°59'8.29"S 176°49'40.14"E	23.83
4	37°57'43.36"S 176°50'16.99"E	27.46
5	37°55'42.00"S 176°51'15.79"E	32.40
6	37°55'10.87"S 176°52'2.74"E	34.20

Table 2.2 Kaituna River site GPS coordinates and the cumulative distance from the first sampling location to the lowest site on the river reach.

Site	GPS coordinates	Distance from site one (km)
1	38° 6'54.16"S 176°49'11.73"E	0
2	38° 2'9.53"S 176°47'55.96"E	4.66
3	37°59'8.29"S 176°49'40.14"E	6.78
4	37°57'43.36"S 176°50'16.99"E	12.15
5	37°55'42.00"S 176°51'15.79"E	14.15
6	37°55'10.87"S 176°52'2.74"E	18.38

To support interpretation of the impact of nitrate on river ecology, samples of aquatic macrophytes were collected along the river. At each site, on each river, (Rangitaiki 2/12/21; Kaituna 24/11/21) samples of all submerge macrophyte taxa present were collected, identified, and preserved for the analysis of nitrogen content. The nutrient content of aquatic plants is well known to integrate medium term nutrient supply, and the intent was to measure internal N content determine whether the conditions in the study rivers were driving N limitation of growth.

2.2.2 Historical data

Over the twelve-year period (2004-2016) the BOPRC compiled running data sets on water quality parameters, including nitrate and turbidity which are within the current study. Therefore, we thought it useful to include historic nitrate and turbidity data within this study to illuminate any trends that may not have been captured by the small snapshot from this study. BOPRC sampling locations on the Rangitaiki were at sites one (Matahina dam) and two (Te Teko township). On the Kaituna none of the BOPRC sampled sites fell within the current study reach but were higher in the catchment but are as follows; Maungarangi Road, Te Matai Road, the Rotoiti outlet, Okere Falls.

Figure 2.4 Sampling site location along the reaches of the Kaituna River.

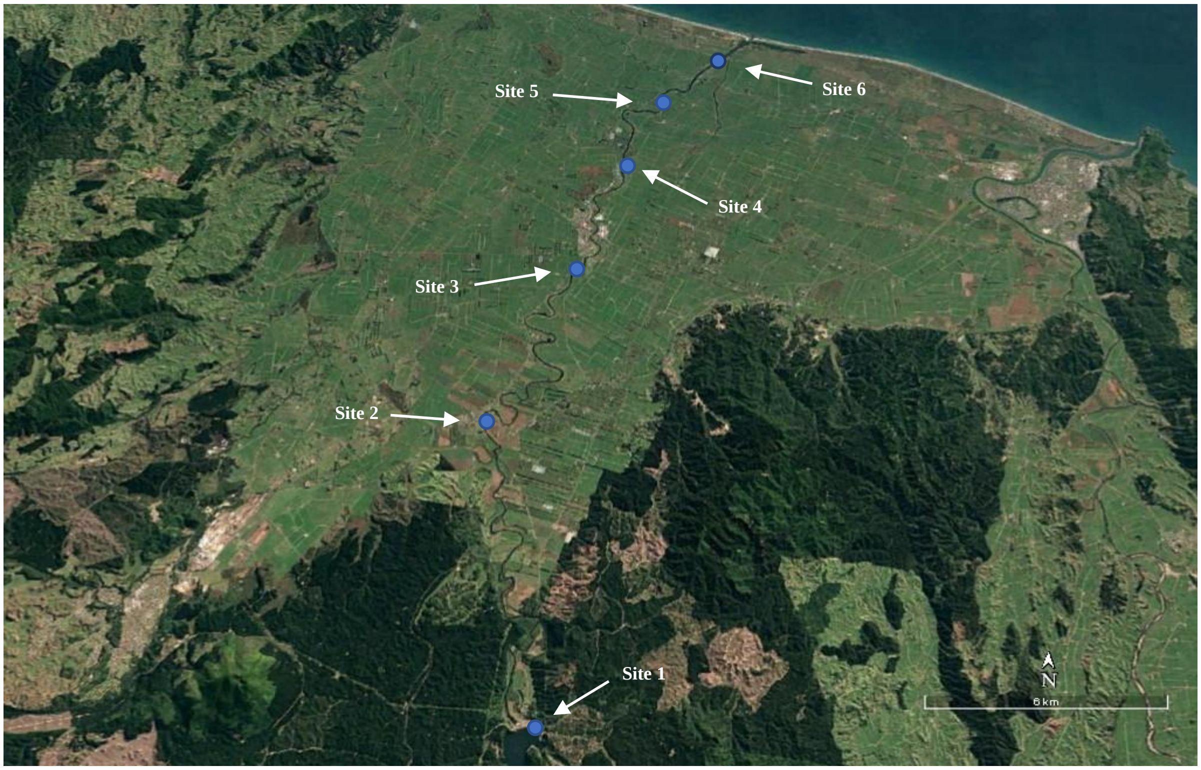

Figure 2.5 Sampling site locations along the reaches of the Rangitaiki River.

2.2.3 Lagrangian sample collection

The plan for this sampling mode was to deploy a neutrally buoyant drifter in the upper sections of the low coastal plains of the Kaituna and Rangitaiki Rivers (Sites 1 through 6) at intervals of approximately one kilometre. The development of a drifter was assigned as a final year project for an engineering student at The University of Waikato. Unfortunately, the drifter proved to lack functionality in the field. To account for the missing Lagrangian samples, extra effort was placed into the Eulerian (fixed point) sampling.

2.3 Laboratory analysis

2.3.1 Total suspended solids

TSS samples were immediately processed on return from the field with a MultiVac 610-MS-T Multi-Branch Filtration system; each 2 L TSS sample was filtered through a 1.2 µm, 47mm Whatman GF/C glass microfiber filter. Post filtering each paper was placed in tinfoil, labelled, and placed into the oven at ~50 ºC for 72 hours until dry. Once dry, the filter papers were weighed on a balance and the results recorded (N.B: each filter paper was individually weighed prior to filtering).

2.3.2 Chlorophyll-*a* extraction and analysis

A high-performance liquid chromatographic (HPLC) method was used to extract and analyse *chl-a*, based on Zapata *et al.,* (2000). Extraction and analysis were carried out over three consecutive days in the laboratory. Each filter paper was cut into small pieces and placed into its own respective falcon tube with 5mL of 95% acetone solution. The tubes were placed in the freezer (-20 ºC) for 4 hours. After the samples were steeped, they were shaken, then centrifuged at 3000 G for 15 minutes. A syringe was then used to extract 2 mL of solution from each respective tube and inoculated into vials ready for HPLC analysis. Results were elucidated based on calibration curves derived from known chlorophyll-a concentrations.

2.3.3 Nitrate analysis for collected water samples

Nitrate samples were stored in the freezer (-20 ºC) prior to analysis. Nitrate-N concentrations were quantified using vanadium(III) chloride (VCl_3) reduction based on the method stipulated in Schnetger & Lehner (2014). This method is a manual spectrophotometric procedure for the elucidation of nitrate + nitrite within a sample of which under oxidising conditions the overwhelming majority tends to be nitrate. For simplicity, hereafter nitrate + nitrite is referred to as nitrate.

2.3.4 Reagent preparation

I. 0.8 g of VCl_3 was dissolved in 20 mL of Milli-Q water, 8.4 mL HCl (37 wt.%) was added and diluted to 100 mL with Milli-Q water. Using this method was the best way to obtain a particle free solution; if there was remaining particles the solution was filtered with a syringe filter (2 µm).

II. 100 mg N-1-naphthylethylenediamine dihydrochloride (NEDD) was dissolved in 100 mL Milli-Q water.

III. 2 g of sulphanilamide was dissolved in 100 mL 10% HCl.

IV. To create the NOx reagent the Griess (NEDD + sulphanilamide) and the reduction reagent (VCl_3) are added together at the following ratio: five parts I + one part II +one part III (75mL VCl_3 + 15mL NEDD + 15mL sulphanilamide).

The ratio of reagent and sample were 5 mL sample and 3 mL reagent combined in 15 mL falcon tubes and placed in the oven at approximately 50 ºC for 45 minutes to allow colour development. Once the colour was developed, samples were transferred to cuvettes and readings were taken with a UV–2600 IV-VIS spectrophotometer. Nitrate-N concentration was calculated based on a five-point standard curve of known concentrations of KNO_3 solution.

2.3.5 Nitrogen contents of aquatic plants:

As many species of submerged aquatic plants as possible were collected in the immediate vicinity of the sample site using a rake. Plants were bagged and labelled then placed into a cooler. Upon return to the lab plants were separated into species and placed into an oven at ~50ºC until dry. Once dry the plants were ground into a fine powder using a hammer mill, the grinding apparatus being cleaned between uses to ensure no cross contamination of samples. For each digestion, 10 mg of ground plant material was placed into a 50-mL falcon tube prior to the addition of reagents, followed by addition of 30 mL of a digestion reagent. Digestion followed the alkaline-persulphate method described by Patton and Kryskalla (2003). Digests were analysed in triplicate using a microplate method.

For microplate analyses, 60 μL of digest was added in triplicate to wells in the plate, then diluted with 60 μL of Milli-Q water, after which 150 μL of mixed reagent was added to all wells. The mixed reagent for this procedure is the same as described for nitrate analysis above. Plates were then covered and incubated at 50°C in an oven for 60 minutes. After cooling, plates were read on microplate analyser at 540 nm. Each plate had blanks and between three and five calibration standards. Calibration standards were made up in digestion reagent and autoclaved as per the samples themselves.

2.3.6 Statistical analysis

Simple linear regression analysis was used to elucidate whether higher concentration of nitrate lead to an increasing concentration of chlorophyll-a and TSS in the Kaituna and Rangitaiki Rivers. R version 4.0.0 was used to assess the assumptions of normality and visualize the data.

Chapter 3

Results

3.1 Environmental variables

Site selection was primarily based on locations of interest (i.e., drainage discharges, tributary inflows) along the Kaituna and Rangitaiki rivers. Effort was made to cover above and below all drainage discharges as well as tributaries along both river reaches to ensure the results reflected an accurate representation from source to sea, allowing valid conclusions to be drawn from this research. Measurements of environmental variables were taken at a minimum interval of 1-km apart on both the Kaituna and Rangitaiki rivers. Both rivers showed seasonal and spatial patterns, patterns were generally evident. Predictably temperature showed strong seasonal variation, but no real spatial patterns being highest in January and February, c. 21 in the Kaituna (top) and c. 22 in the Rangitaiki (bottom) (Figure. 3.1). However, due to access issues across private land we were unable to take a reading from the main channel thus site 1 on the Rangitaiki was measured in the littoral zone of Lake Matahina. Being a lentic system there was strong stratification in the summer months which meant that temperatures were up to c. 3 degrees higher than downstream sample sites.

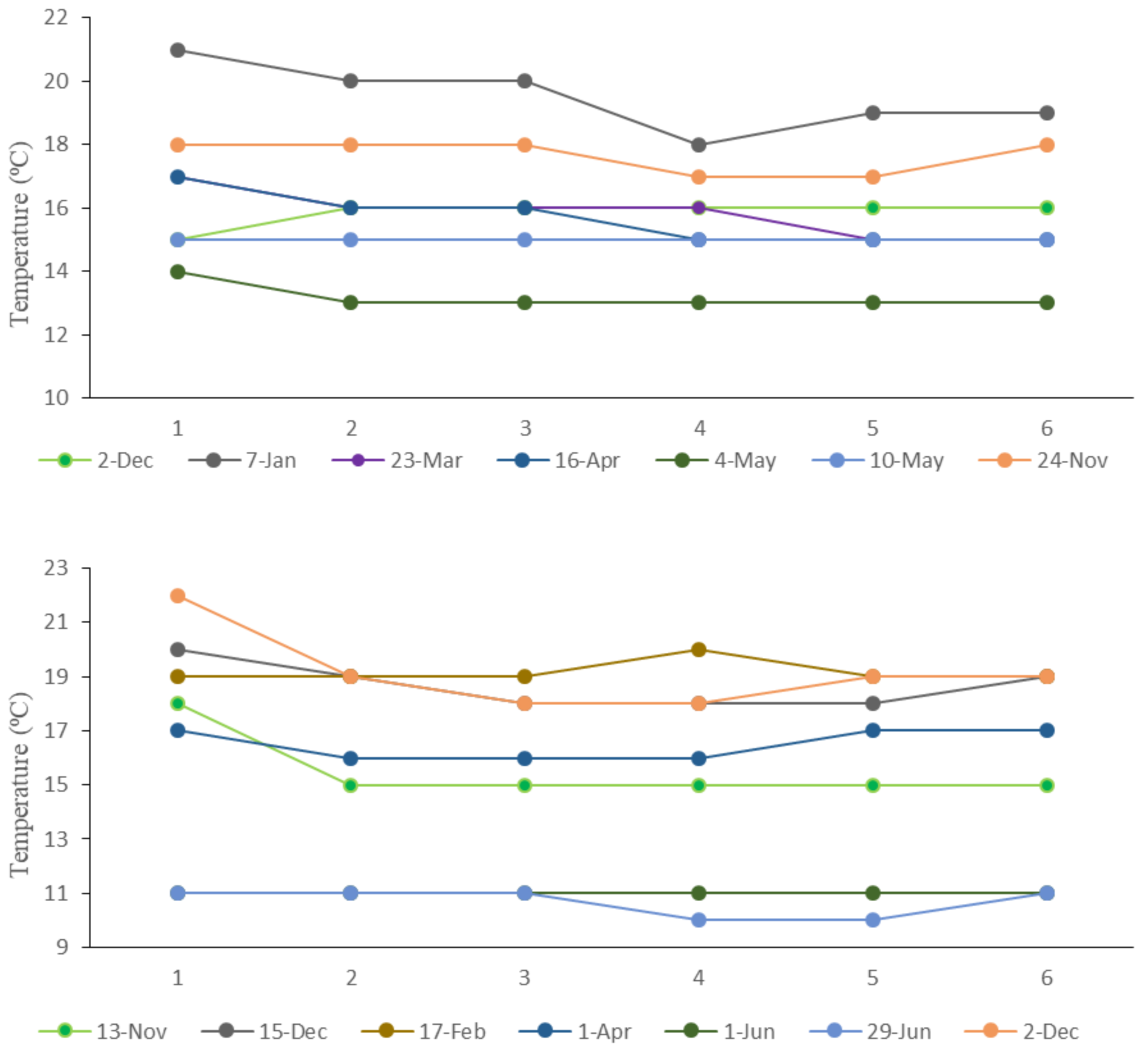

Figure 3.1: Reach scale water temperatures (ºC) for the Kaituna (top) and the Rangitaiki (bottom).

Specific conductivity displayed relatively uniform readings from sites 1 through 4 in both rivers, with winter (May/June) and autumn (March/April) showing the highest readings. Occasionally there was a slight increase in conductivity between sites 1 and 2 on the Rangitaiki, and slight decreases downstream in the Kaituna. Between sites 5 and 6 the Rangitaiki (Figure. 3.2) reached a maximum reading of 5877 µS/cm but showed consistently lower readings between sites 1 and 4 with a mean reading of 100.06 µS/cm. In contrast the Kaituna (Figure. 3.3) reached a maximum of 604.65 µS/cm but had an average reading of 142.43 µS/cm between sites 1 and 4. The lowest sites were within the tidal wedge of both rivers and high conductivities here are to be expected due to up flowing salt water. Elevated readings were dependent on whether it was in ebb or flood tide as some sampling occasions at sites 5 and 6 showed readings consistent of freshwater ranging between 77 – 141 µS/cm.

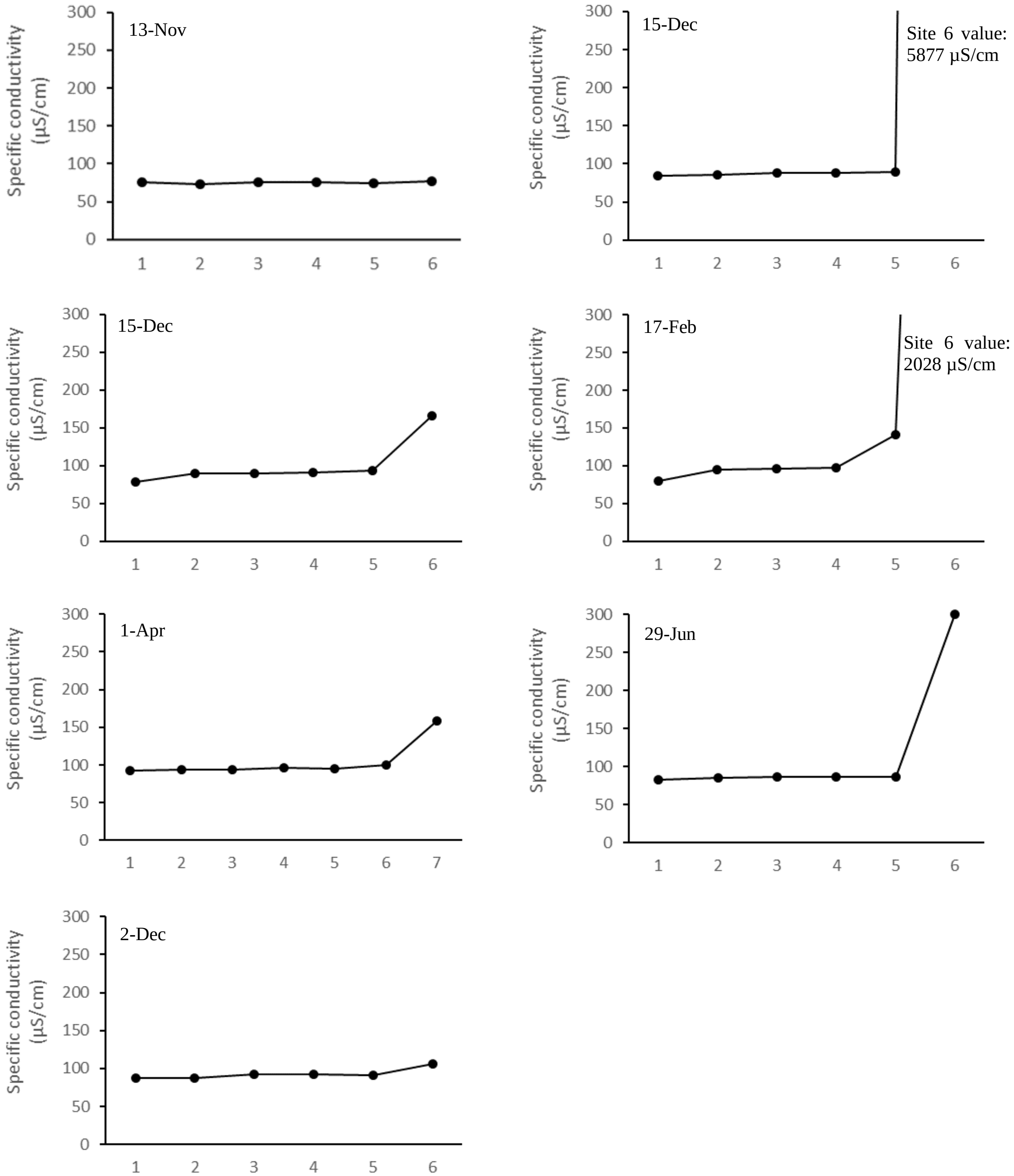

Figure 3.2: Reach scale plots of specific conductivity (µS/cm) in the Rangitaiki River. Sampling days are in order from 13-Nov to 2-Dec. Note that as the river approaches the discharge point, freshwater and marine processes begin to interact thus we observe spikes in conductivity. To keep scale accurate, it is noted where the value exceeds 300 µS/cm.

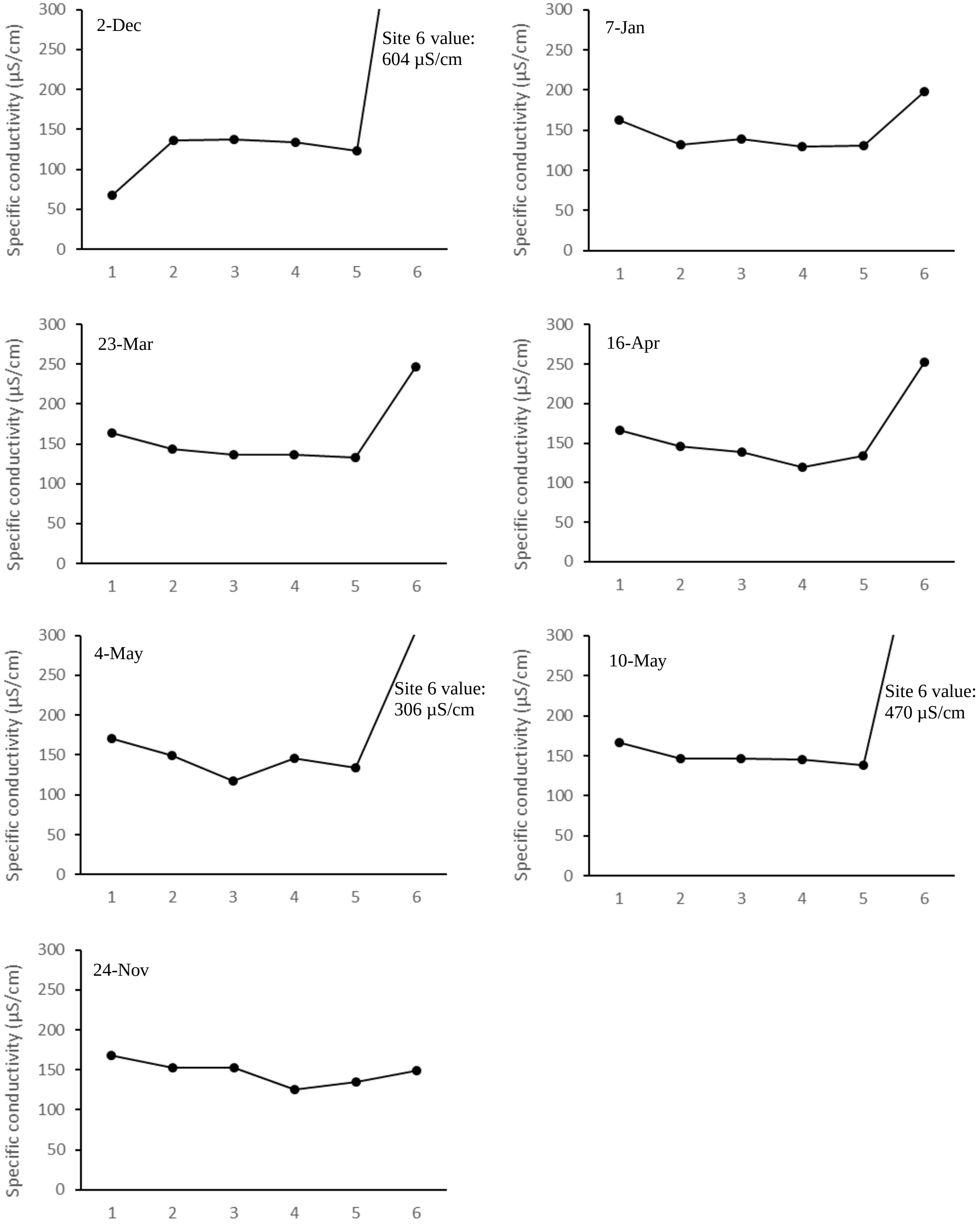

Figure 3.3 Reach scale plots of specific conductivity (µS/cm) in the Kaituna River. Sampling occasions are in order, from left to right 2-Dec to 24-Nov. Note that as the river approaches the discharge point, freshwater and marine processes begin to interact thus we see spikes in conductivity, values are noted in text.

The relationship between pH and oxygen saturation is ultimately independent but linked by the change in carbon dioxide (CO_2) in the river. Respiration and photosynthesis remove and produce O_2 respectively, which gives an insight to ecosystem metabolism. At the same time these processes produce and consume CO_2, with more CO_2 tending to drive pH lower. During this project all samples were taken between 0800 – 1400 hrs when plants were photosynthesising, potentially missing any O_2 sags caused by night-time respiration. Most running waters in New Zealand are typified by a pH range of 7-8 due to buffering from bicarbonate creating a better resistance to pH change from excessive (i.e., prolific algae growth) primary productivity.

Over the course of sampling the pH of the Rangitaiki River ranged between 6.6 and 8.6; at the same time the dissolved oxygen (DO) ranged between 83 and 143% (Figure. 3.4). We did not observe any seasonality in terms of pH and DO, although spatially we observed the highest values consistently at sites one and two. On half of all sampling occasions, we observed pH starting just below or just above 8 then sharply dropping to around 7 (~ ± 0.3) at site two, remaining steady around that value through to site six. Dissolved oxygen generally showed a similar trajectory to pH, starting with the highest values at site one which were usually > 105% occasionally DO would remain high through the first three sites but more often dropped steadily to 85-90%.

The Kaituna displayed lower readings of pH, ranging between 5.99 and 7.13, with DO ranging between 82 and 141% (Figure. 3.5). As a general pattern, pH remained around 6.5 and rarely exceeded 7, showing a linear pattern with minor fluctuations apparent. What is also apparent in all, disregarding 24-November, is a trend of increasing in pH from sites four through six as freshwater begins to mix with seawater. Dissolved oxygen also stays parallel to pH on all apart from two occasions, displaying a gradual drop from higher levels at site one to lower levels at sites five and six.

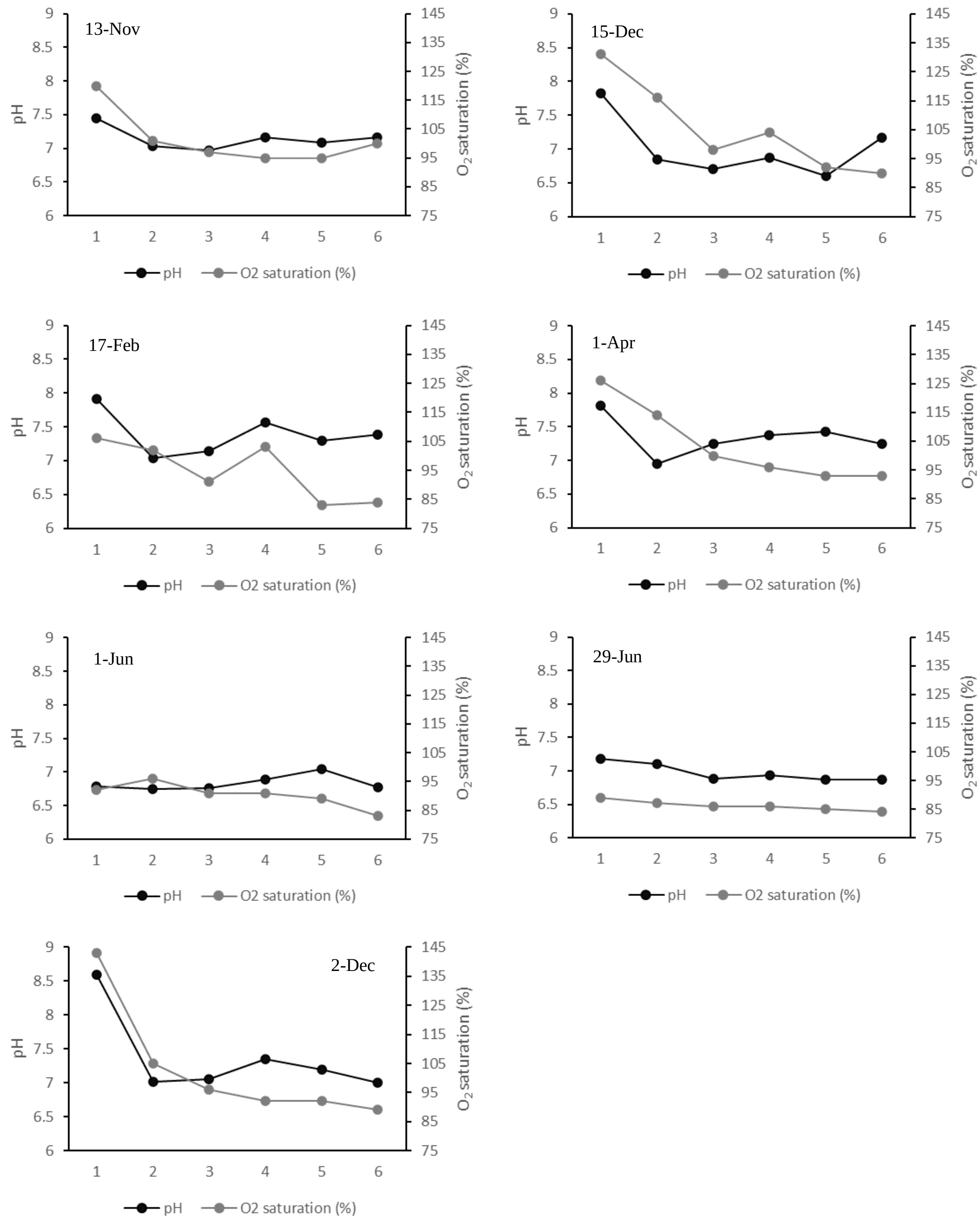

Figure 3.4 Reach scale plots of dissolved oxygen saturation (%) and pH from the Rangitaiki River. Sampling occasions are in order left to right from 13-Nov to 2-Dec.

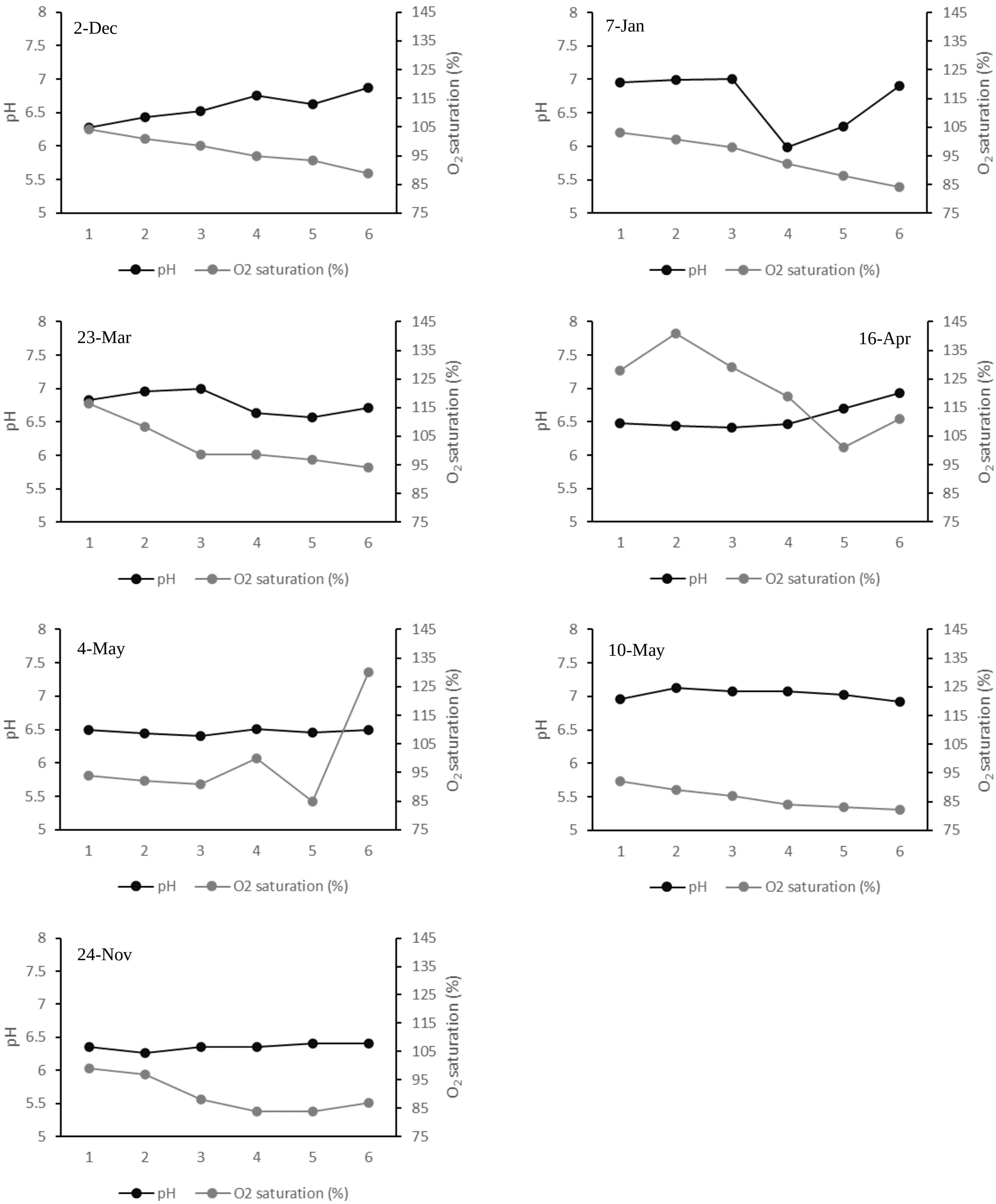

Figure 3.5 Reach scale dissolved oxygen saturation (%) and pH from the Kaituna River. Sampling occasions are in order from left to right from 2-Dec to 24-Nov.

The Kaituna displayed values that were consistent with base-flow conditions of close to 17 $m^3 s^{-1}$, reaching a maximum single day flow of 23.2 $m^3 s^{-1}$. The Rangitaiki had larger overall variability of flow values reaching a maximum of 117.88 m^3s^{-1} (Tables. 3.1 and 3.2). The Rangitaiki, river does not drop below 35 m^3s^{-1} due to a minimum flow being set for Trustpower as part of their resource consent to operate the Matahina Dam. The significantly elevated flows on the sampling days have come from a short-term dam release or high rainfall as the three day mean from Feb – Jun is slightly above base flow.

Table 3.1 Mean daily flow and three day mean flow (m^3s^{-1}) on each sampling occasion for the Kaituna River, ± one standard deviation.

Date collected	Mean daily flow (m^3s^{-1})	3-day mean (m^3s^{-1})
02/12/20	23.2 ± 0.05	23.34 ± 0.36
07/01/21	18.24 ± 0.02	18.87 ± 0.81
23/03/21	15.94 ± 0.02	16.06 ± 0.07
16/04/21	16.68 ± 0.03	16.89 ± 0.21
04/05/21	16.64 ± 0.03	16.69 ± 0.05
10/05/21	16.82 ± 0.02	16.62 ± 0.1
24/11/21	19.91±0.38	20.07±0.22

Table 3.2 Mean daily flow and three day mean flow (m^3s^{-1}) on each sampling occasion for the Rangitaiki River, ± one standard deviation.

Date collected	Mean daily flow (m^3s^{-1})	3-day mean (m^3s^{-1})
13/11/20	117.88 ± 18.89	91.40 ± 16.50
15/12/20	53.13 ± 4.79	44.87 ± 2.97
17/02/21	41.21 ± 2.10	37.07 ± 0.49
1/04/21	40.27 ± 3.52	33.93 ± 1.87
1/06/21	50.54 ± 6.45	41.54 ± 5.62
29/06/21	98.34 ± 25.38	48.35 ± 5.43
02/12/21	55.09 ± 5.24	61.63 ± 5.01

The Rangitaiki had well correlated patterns between turbidity (NTU) and the total suspended solids (TSS) (Figure. 3.6). We found that values increased from site one to site two and stagnated through the middle reaches; or remained linear from site one to six with no major changes downstream. The Rangitaiki had an average reach scale reading of 1.75 NTU and TSS readings of 3.56 mg/L. There was an exception to this on 17-Febuary which presented both maximum values for NTU (5.5 NTU) and TSS (10.1 mg/L^{-1}) at site one, although values dropped considerably at site two to site six, NTU and TSS remained high through the entire reach. Additionally, sites four through six showed spikes likely where freshwater and marine processes began interacting and creating discrepancies in the readings. Additionally, with the absence of tributaries and drainage discharges the Rangitaiki along the reaches we did not observe localized increases in NTU or TSS in the middle reaches highlighting the absence of any major contaminant pathways below the dam.

The relationship between NTU and TSS that the Kaituna displayed was similar to that of the Rangitaiki, with most sampling occasions showed consistent patterns, and near-uniform values from site one through to site six (Figure. 3.7). The Kaituna had a low reach average of 1.04 NTU reaching a maximum value of 1.78 NTU; the average reach scale TSS reading was 1.81 mg/L^{-1} reaching a maximum value of 5.4 mg/L^{-1}. Although there were uniform concentrations across most sites, there were consistently visible increases in NTU and TSS at sites five and six, this could be freshwater and marine processes beginning to interact or the three significant inflows; the Waikoura stream, the Raparapahoe stream, and the Waiari River, all within two kilometers of one another. This was consistent in all samples except two. The 2^{-} Dec sampling showed a spike in NTU and TSS values from site one to four, then dropping slightly at sites four through six; this sampling happened to occur at the highest flow levels throughout the sampling (see Table. 3.2). On the 7-Jan the NTU sensor on the sonde failed thus we were unable to record the turbidity on that day.

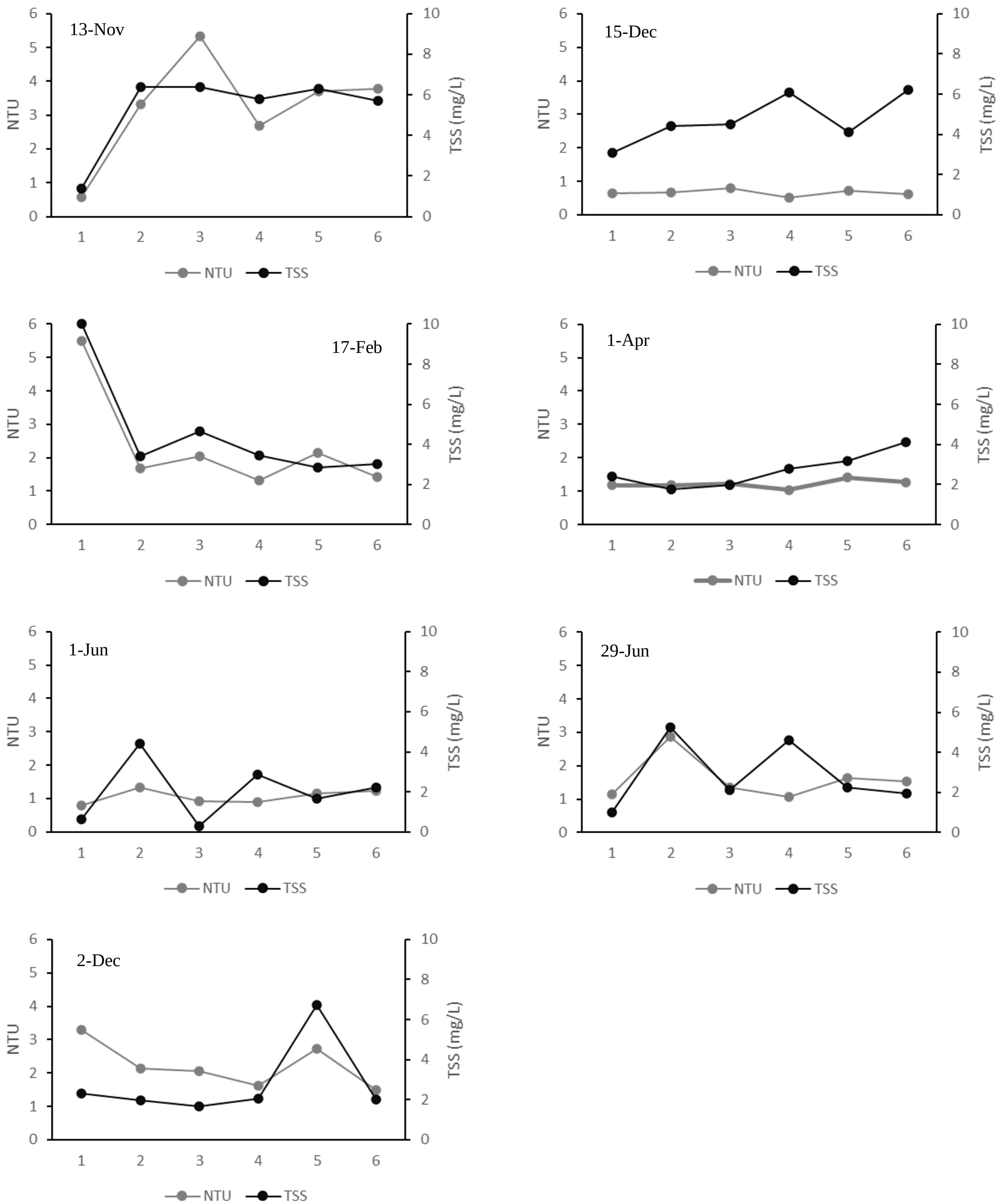

Figure 3.6 Reach scale plots of total suspended solids (mg/L) and turbidity (NTU) in the Rangitaiki River. Sampling occasions are in order left to right, 13-Nov to 2-Dec.

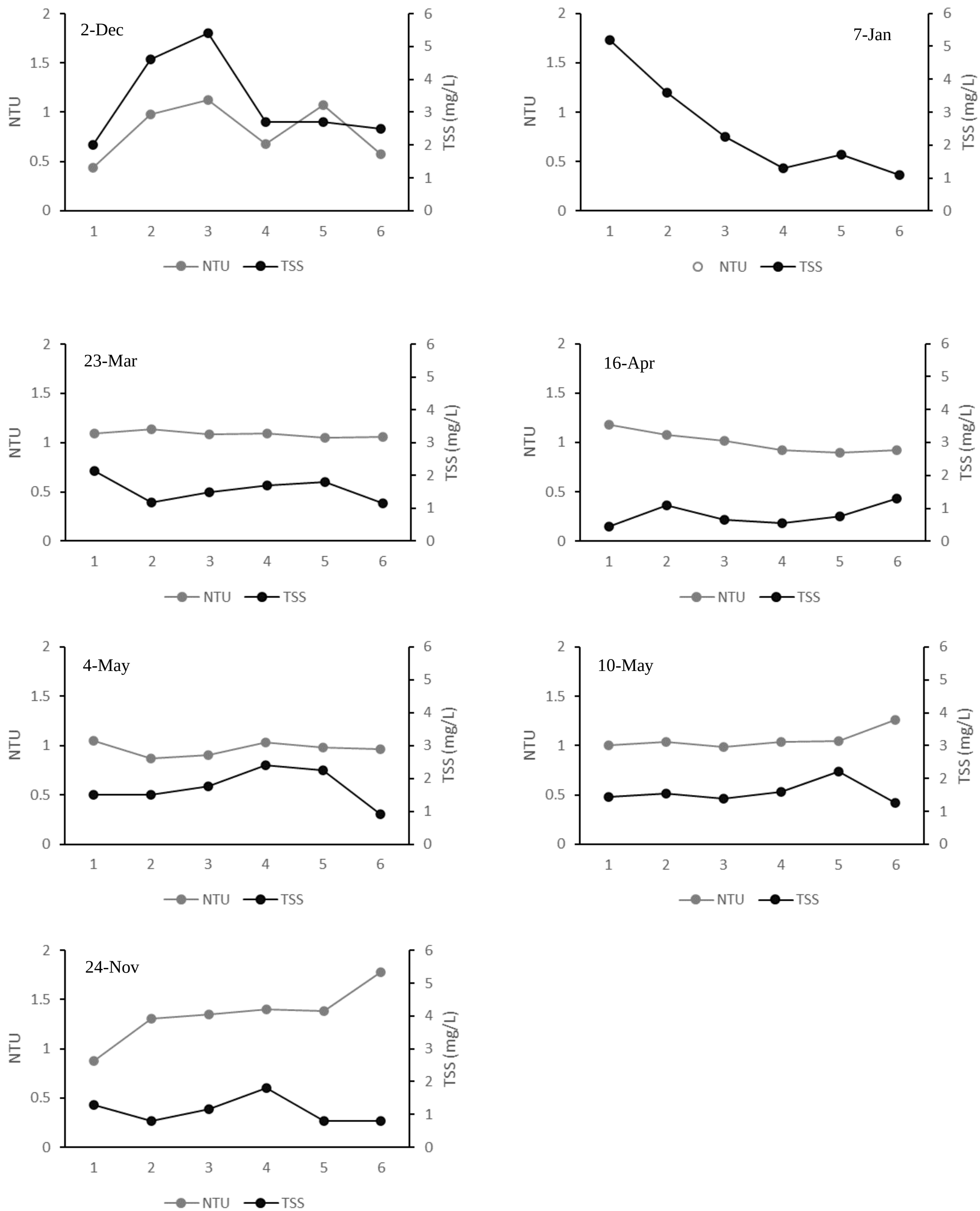

Figure 3.7 Comparative reach scale plots of turbidity (NTU) and total suspended solids (TSS) on the Kaituna River. Each sampling occasion is labelled chronologically from left to right. Note that the 7th of Jan has NTU omitted due to equipment failure in the field.

Chlorophyll-*a* (Chl-a) displayed discernable spatial and seasonal patterns in the Kaituna and Rangitaiki rivers (Figure. 3.8). Through winter and summer months the Kaituna showed a uniform pattern across the reaches with a relatively low mean concentration of 1.24 µg/L. Generally, sites five and six showed a slight upward trend in winter and summer, except for 7-January. An Autumn and spring pattern emerged in the Kaituna where chl-*a* concentrations started high and rose from site one through two, except 16-April. Autumn samples also tended to show decreases in concentration over sites three and four in the middle reaches, peaking upwards over sites five and six except for 4-May which showed no change. Overall, autumn samples on the Kaituna showed the highest concentrations of chl-a with an average of 2.77 µg/L over the reach.

The Rangitaiki consistently displayed higher concentrations of chl-*a* at site one in all seasons with 15-December and 13-November being exempt; high chl-*a* levels quickly dissipate over sites two through four to low levels. Interestingly from sites four, five and six chl-a concentrations trended upwards, especially 13-November which spiked from 0.17 µg/L at site five to 7.43 µg/L. Similarly, Autumn presented the highest values for the Rangitaiki c. 14.54 µg/L with a reach scale average of 5.77 µg/L. These values were closely followed by early summer (2-Dec) but was not consistent with the results from 15-December or 17-Febuary.

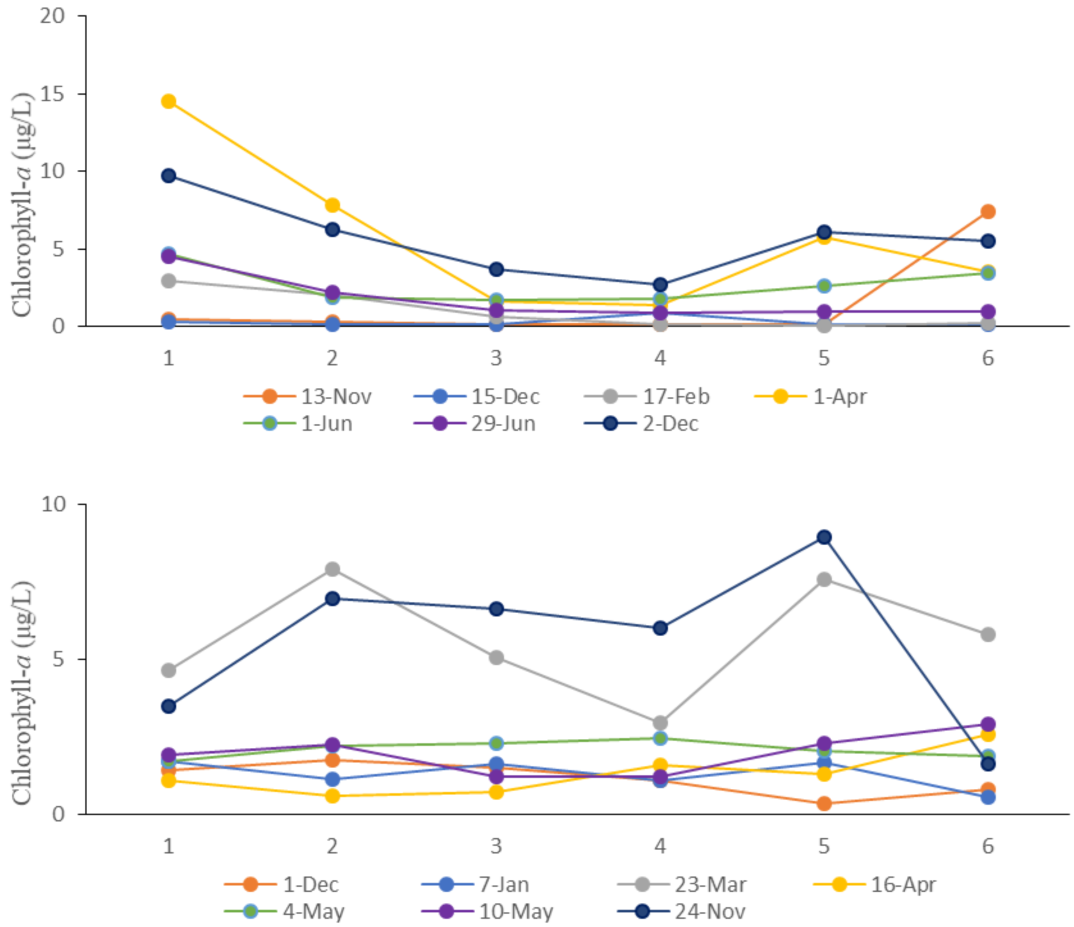

Figure 3.8 Reach scale chlorophyll-a concentrations of the Rangitaiki (top) and Kaituna (bottom) rivers.

3.2 Main channel nitrate concentrations

Over the course of one-year (Nov 2020 – Nov 2021) nitrate samples from the drains and tributaries were collected and analysed. The paradigm of increasing distance concentration with distance from source to sink is evident in both rivers but more obvious in the Kaituna but also dependent on season. The Rangitaiki River nitrate-N concentrations ranged from 0.038 to 1.11 mg/L^{-1} whilst showing a strong spring/summer and loose autumn/winter pattern (Figure. 3.9). Nitrate-N concentrations peaked strongly in late spring and early summer (Nov & Dec), mainly at sites two, three, and four. In late summer and mid-autumn (Feb & Mar) we observe very low mean concentrations of nitrate-N (0.13 mg/L^{-1}) which also transfers into winter, although 29-

June does show slightly elevated readings than samples around the same time. Overall, what we observe is relatively uniform nitrate-N concentrations but a strong seasonal distinction. The Rangitaiki River has no major tributaries or confluences below the Matahina dam, therefore, what exits the dam will generally dictate the downstream concentrations. Although at sites five and six we do tend to observe slight fluctuations in concentration where we know marine and freshwater processes are interacting.

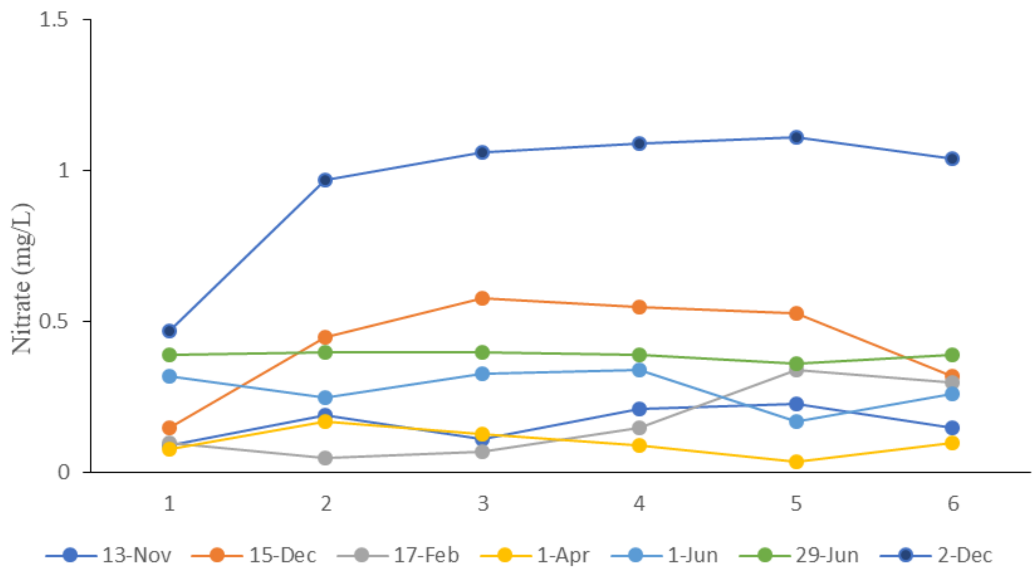

Figure 3.9 Reach scale nitrate-N concentrations on the Rangitaiki River, note that site five returned below detectable limits (BDL ≤0.038 mg/L) on April 1st sampling.

The Kaituna River did show a stronger trend of increasing concentration of nitrate the further downstream the samples were taken; nitrate-N concentrations were recorded at their lowest level in March ranging from 0.02 up to 0.65 mg/L^{-1} and their highest in November ranging from 0.66 to 1.52 mg/L^{-1}. (Figure. 3.10). From site one nitrate-N concentration consistently increased toward site two in all seasons, disregarding 2-Dec and 23-Mar samplings. Sites two to four most often showed stagnant concentrations after the spike from site one and no significant increases in concentration were observed in any season. From sites four through six, concentrations consistently increased (excluding 2-December) as these lower reach sites are often coinciding with tributary junctions. Notably between four and five, the lower drain (LD) which is fed from the Waikoura stream, and the upper drain (UD) is fed from the Raparapahoe stream; immediately below site four the Waiari river comes to a confluence. What is also observed is a downward trend in concentration at site six, some changes are more subtle than others, but the change is consistent over all sampling occasions barring 2-December which shows no change.

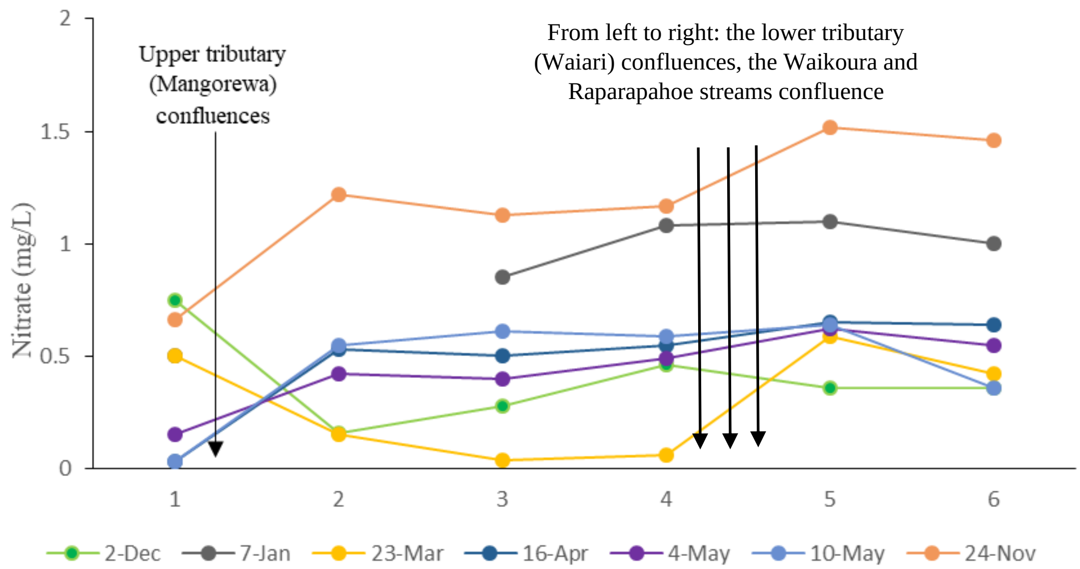

Figure 3.10 Reach scale nitrate-N concentrations on the Kaituna River. Note the missing value at site two on the 7th of January sampling, the sample was lost in transit back to the laboratory.

3.2.1 Kaituna drain and tributary nitrate concentrations

In the lower section of the Kaituna there are two drainage discharges and one major tributary. In the upper section there is another major tributary, the Mangorewa river which joins the main channel immediately below site one. These inflows were sampled to assess their contribution to the nitrate load increases in the main channel (Table. 3.3). What was found was there were substantially elevated concentrations of nitrate-N coming from these inflows. In the upper and lower tributaries nitrate-N concentrations ranged from 1.06 to 3.27 mg/L^{-1}, with the highest concentrations coming consistently from the Mangorewa (UT) river. The highest concentrations of nitrate-N in the upper and lower tributaries occurred in January and November respectively, reaching the lowest levels in May. The lower and upper drains both consistently displayed lower concentrations of nitrate-N, but also a wider range of concentrations ranging from 0.02 to 1.83 mg/L^{-1}. Although the drains showed a similar seasonal pattern to the tributaries with peaks in nitrate-N concentration in January and November. Given that the tributaries were larger rivers which confluence with the main channel it is not surprising that they carried greater load of nitrate than more stagnant drains.

Table 3.3 Nitrate-N concentrations (mg/L) in the tributaries and irrigation drains of the Kaituna River. Each location confluences with the main river channel along the sampled reach. Samples for 2-Dec were not collected hence not applicable.

	Mangorewa (UT)	Waiari (LT)	Waikoura (UD)	Raparapahoe (LD)
2/12	n/a	n/a	n/a	n/a
7/1	1.97	1.61	1.35	1.12
23/3	1.32	1.13	0.68	BDL
16/4	1.31	1.12	1.83	0.31
4/5	1.33	1.06	0.99	0.62
10/5	1.48	1.29	1.12	0.78
24/11	3.27	2.74	2.11	0.97

3.3 Chlorophyll and nitrate

A simple linear regression was used to predict chlorophyll-a concentration based on nitrate-N concentration in both the Rangitaiki and Kaituna Rivers (Figures. 3.11 and 3.12). Visually, in both rivers, results indicate there is some evidence to suggest that chlorophyll-a will increase with greater concentration of nitrate-N. However, a significant regression equation was not found for the Rangitaiki River ($F(1, 40) = 1.72$, $p>0.05$ with an R^2 of 0.041, showing poor predictability of the model. Similarly, the fitted model for the Kaituna showed poor predictability with an R^2 of 0.046, neither was a significant regression equation found ($F(1, 40) = 1.959$, $p>0.05$.

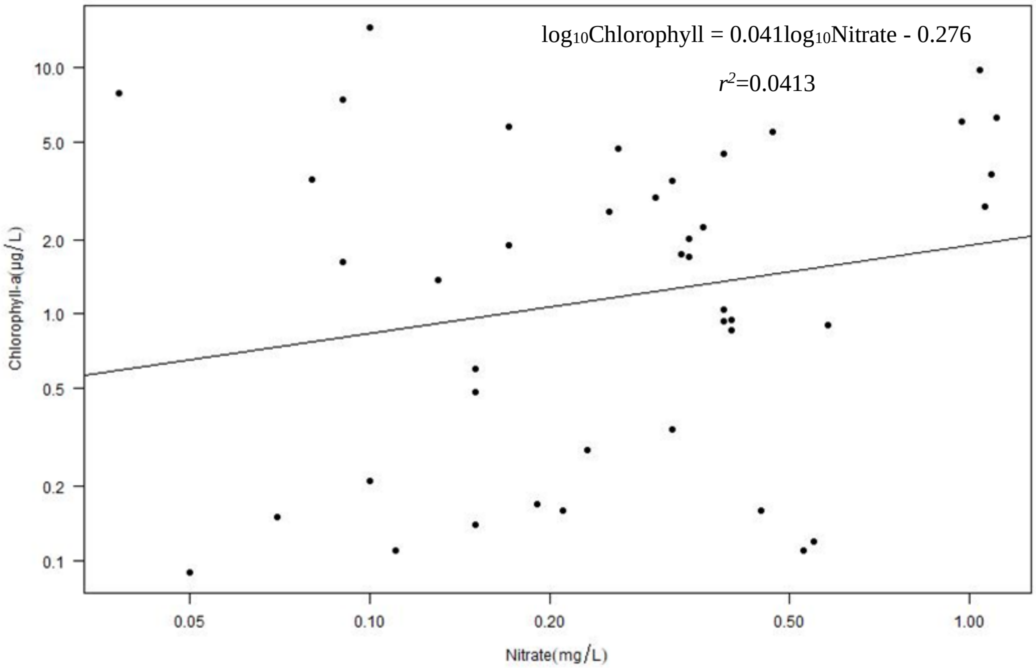

Figure 3.11 Regression analysis displaying the relationship between chlorophyll-a (µg/L) and nitrate-N (mg/L) in the Rangitaiki River.

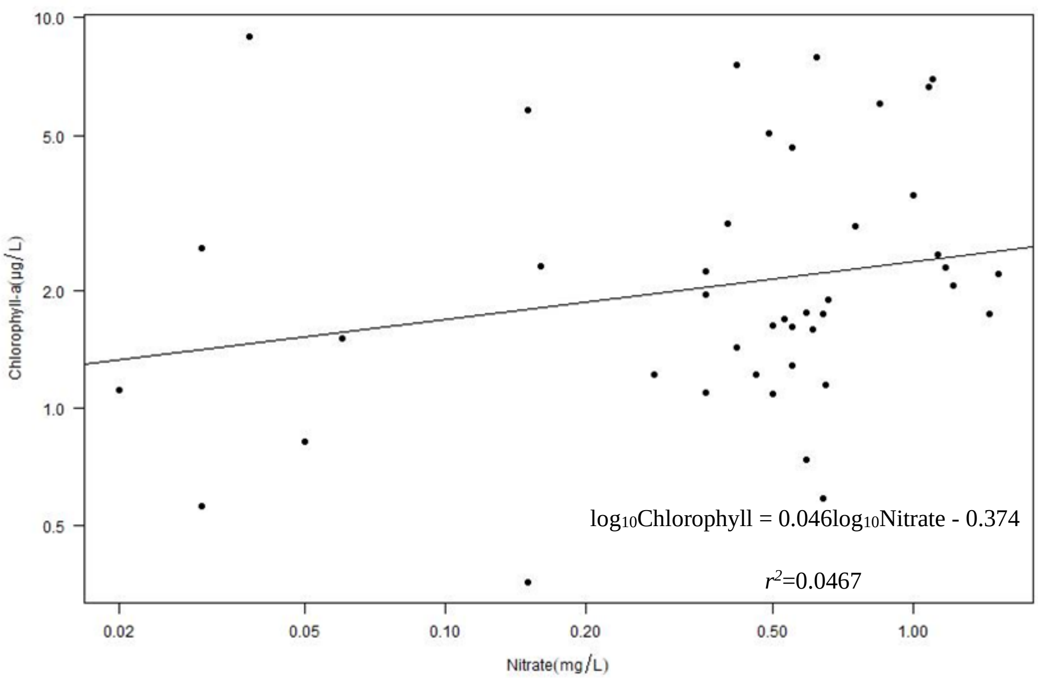

Figure 3.12 Regression analysis plotting chlorophyll-a (µg/L) and nitrate-N (mg/L) in the Kaituna River.

3.4 Total suspended solids and nitrate

Total suspended solids (TSS) are a measure of particles that are entrained in the water column. No discernible relationship was found between TSS and nitrate in either river (Figures. 3.13 and 3.14). The Rangitaiki model showed a non-significant regression equation ($F (1, 40) = 0.68$, $p>0.05$ with an R^2 of <0.01, likewise the Kaituna ($F (1, 40) = 0.0091$, $p>0.05$ with an R^2 of <0.01).

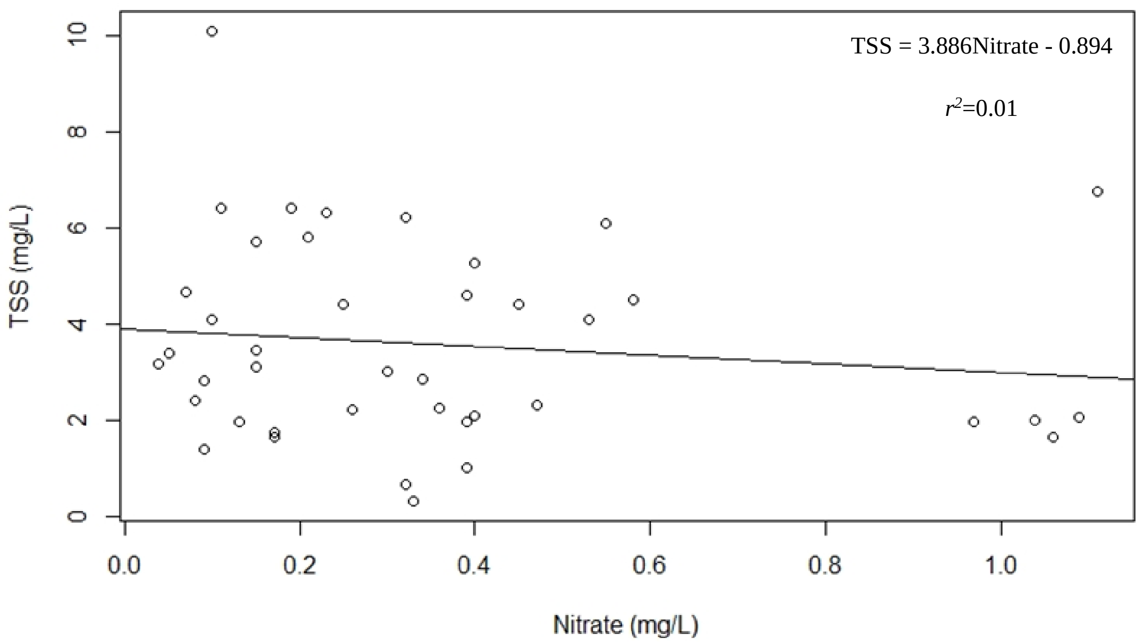

Figure 3.13 Regression analysis plotting total suspended solids (TSS) (mg/L) against nitrate-N concentration, in the Rangitaiki River.

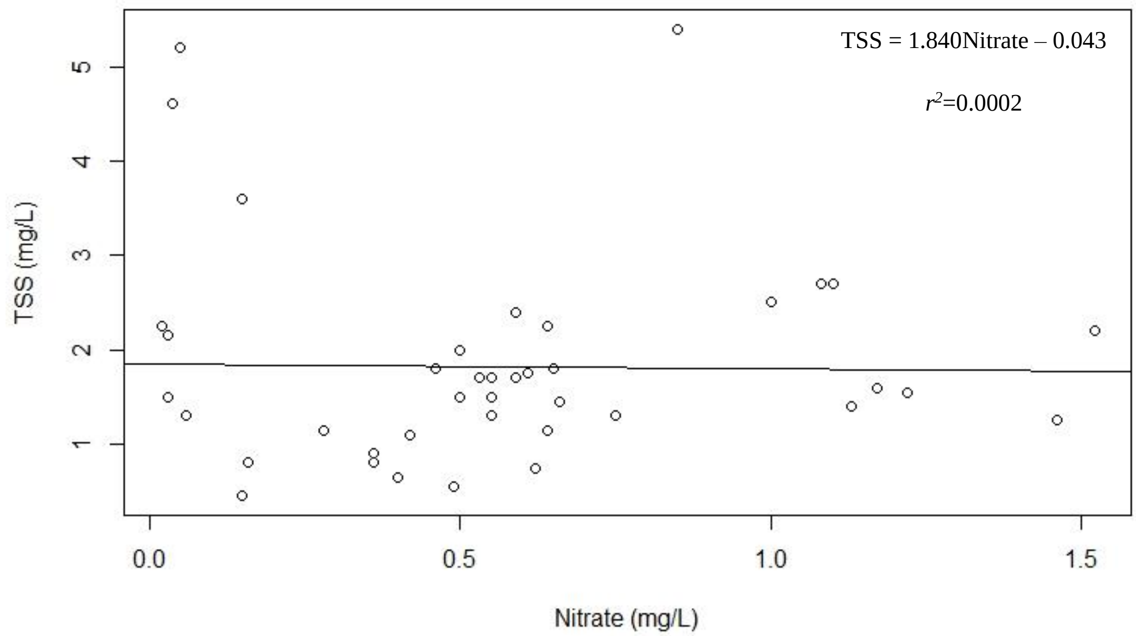

Figure 3.14 Regression analysis displaying the relationship between total suspended solids (mg/L) and nitrate-N (mg/L) in the Kaituna River.

3.5 Historical data – Bay of Plenty Regional Council

This data (Table. 3.4) gives the current study context, showing the historical condition of selected sites that fall either within the study, or give context to the upstream condition of the Rangitaiki and Kaituna Rivers.

Table 3.4 Historical nitrate-N concentration data for sites on the two rivers. Values are all in mg/L, and are the quartile values, the median and the maximum from BOPRC records between 2008-2016.

River	Site	25%	Median	75%	Max
Rangitaiki	SH5	1.01	1.22	1.43	2.36
	Lake Matahina	0.25	0.36	0.53	0.70
	Te Teko	0.29	0.43	0.50	0.80
Kaituna	Lake Rotoiti	0.01	0.03	0.08	1.47
	Maungarangi Road	0.22	0.27	0.31	0.43
	Te Matai	0.47	0.55	0.66	1.38

3.6 Plant species and digestion

Aquatic macrophytes were found at all sites on the Kaituna and all but one site on the Rangitaiki River (Table. 3.5). Typically, these took the form of monospecific beds pushed flat by the river flow, and sediment mounds built up around these clumps. Only one native species was found - *Myriophylluim triphyllum*, at site four on the Rangitaiki River, while six non-native taxa were found along the two rivers. Plant tissue N content exceeded the criteria established by Demars & Edwards (2007) above which nitrogen is unlikely to limit growth rate in all samples except one.

Table 3.5 Plant digestion results for macrophyte species found at each site in the Kaituna and Rangitaiki Rivers, results display the percent weight of N in plant tissue. Bold type indicates the exceedance of the critical nutrient concentration for 95% maximal growth rate in aquatic angiosperms (Demars & Edwards, 2007)

River	Site	Plant *spp*	Weight %N
Kaituna	1	*Lagarosiphon major*	**2.14**
Kaituna	2	*Ceratophyllum demersum*	**2.98**
Kaituna	2	*Ceratophyllum demersum*	**1.61**
Kaituna	2	*Egeria densa*	**2.18**
Kaituna	2	*Elodea canadensis*	**1.84**
Kaituna	2	*Potamogeton crispus*	**1.49**
Kaituna	3	*Ceratophyllum demersum*	**2.12**
Kaituna	3	*Egeria densa*	**2.33**
Kaituna	4	*Ceratophyllum demersum*	**1.77**
Kaituna	4	*Elodea canadensis*	**2.23**
Kaituna	5	*Ceratophyllum demersum*	**1.55**
Kaituna	5	*Egeria densa*	**1.77**
Kaituna	5	*Elodea canadensis*	**2.14**
Kaituna	5	*Lagarosiphon major*	**1.51**
Kaituna	6	*Ceratophyllum demersum*	**1.78**
Kaituna	6	*Egeria densa*	**1.88**
Kaituna	6	*Elodea canadensis*	**2.54**
Rangitaiki	2	*Ceratophyllum demersum*	**1.53**
Rangitaiki	2	*Potamogeton crispus*	**1.49**
Rangitaiki	3	*Ceratophyllum demersum*	0.95
Rangitaiki	4	*Myriophyllum triphyllum*	**2.45**
Rangitaiki	5	*Ceratophyllum demersum*	**1.24**
Rangitaiki	6	*Egeria densa*	**2.25**
Rangitaiki	6	*Lagarosiphon major*	**1.92**

Chapter 4
Discussion

4.1 Catchment scale processes influence water quality parameters in lowland catchments

This study concerns the lowland reaches of two rivers, the Kaituna and the Rangitaiki, which both flow across highly developed flood plains within the Bay of Plenty. In both rivers, the lowland reaches are the end of complex hydrological systems, which drain varied catchments. Both rivers are fed via lakes, the Rangitaiki from Lake Matahina, and the Kaituna from Lake Rotoiti. These rivers were selected for this study as they are representative of lowland rivers which have predominantly intensive agricultural catchment. Both rivers have historical data from multiple sampling sites, providing context to the current study. Lowland river water quality is generally agreed to be largely determined by the wider catchment characteristics, particularly surrounding land use (Larned *et al.*, 2016). The goal of the study was to ascertain how two major land-use related contaminants nitrate, and suspended sediments vary as the rivers cross the plain, primarily to determine the extent to which contaminant concentrations are attenuated or enhanced during this passage. Additionally, the study included measures of chlorophyll-a, pH, and dissolved oxygen. Monitoring also included nitrate samples from tributary flows and a limited collection of aquatic macrophytes, to assess other aspects of water quality. Over the course of one-year, distinct patterns in chlorophyll-a, nitrate, and suspended solids in both the Rangitaiki and Kaituna Rivers were observed.

In the Rangitaiki River, variability was seen in the nitrate data from source to sink. The changes in nitrate-N concentration coincided with the discharge of the river from the Matahina dam on to the coastal plain. We suggest that the upper catchment, in-lake, and coastal plain processes of the Rangitaiki catchment are responsible for the changes observed between sites one and six.

BOPRC historical data shows median nitrate-N concentration at SH5 in the upper Rangitaiki catchment as 1.22 mg/L (Table. 3.4), the upper catchment contains significant areas of pastoral agriculture (LAWA, n.d), and this may explain the high nitrate-N concentrations. As the river flows through Galatea, it is diluted by the Whirinaki River confluence, within which nitrate-N concentration has a median 0.09 mg/L. This middle section of the catchment is predominantly exotic forest, (as is the Whirinaki catchment) with scattered areas of native bush (see Figure. 2.1). It is widely accepted that water quality generally improves through such areas (Harding, 2004; chapter 33), consistent with the reduced nitrate loads in the Rangitaiki as it enters Lake Matahina. Assessing the historical condition of Lake Matahina as of the BOPRC data, there are generally low levels of nitrate-N leaving the lake at a median of 0.36 mg NO_3-N/L.

Nitrate within water leaving Lake Matahina showed no trend in the previous eight years of existing BOPRC records (2008-2016), displaying similar levels to the current study. Duncan *et al.*, (2017) describes a cyclic pattern for nitrate in forested catchments, which is lower in summer and higher into winter and Lake Matahina seems to display a pattern similar to this. Vant (2013) states this pattern is likely due to higher biological uptake in summer than winter, when plant growth slows and/or senescence releases nutrients. Furthermore, increased precipitation over winter creates more run-off collecting in the lake basin and this seasonality crosses over to the river environment.

The Kaituna River is also lake fed, with the discharge from Lake Rotoiti having a low median nitrate-N concentration of 0.03 mg NO_3-N/L. Unlike the Rangitaiki, the lake discharge is not onto the coastal plain, rather it flows through the Kaituna Gorge prior to this, which is an area dominated by native riparian vegetation cover. The BOPRC site at Maungarangi Road (Paengaroa; Figure. 4.1) is approximately 32 km downstream from Lake Rotoiti, which is effectively the beginning of the coastal plain, which is ~1.5 km above site one in this study. This site has a long-term nitrate-N median of 0.27 mg/L, and Park (2007) found the nitrate-N concentration at the same site to be 0.19 mg/L (Table 4.1). As could be anticipated, passage through the rapids of the Kaituna Gorge has resulted in an increase in the suspended solids and turbidity in the river (Table. 4.1). An increase in nitrate is less expected, but, the Kaituna has a significant inflow from a cold-water spring in the upper catchment which carries naturally higher levels of N and P from volcanic rock (Environment Bay of Plenty, 2009; White *et al.*, 1978). This could potentially explain the somewhat elevated specific conductivity readings on the Kaituna (mean 170 µS/cm) relative to the Rangitaiki.

Table 4.1 Relevant water quality parameters to the current study for the Kaituna River, displaying historical insight into downstream trends. Adapted from Park (2007). N.B. AFFCO is an abbreviation for Auckland Farmers Freezing Company.

	Lake outlet	Paengaroa	Above AFFCO	Te Matai	Te Tumu
Distance from outlet (km)	0	32	39	41	53
SS (mg/L)	3.3	10.5	9.9	9.7	11.2
NTU	1.8	3.0	2.9	3.0	4.5
Nitrate (mg/L)	0.04	0.19	0.36	0.38	0.41

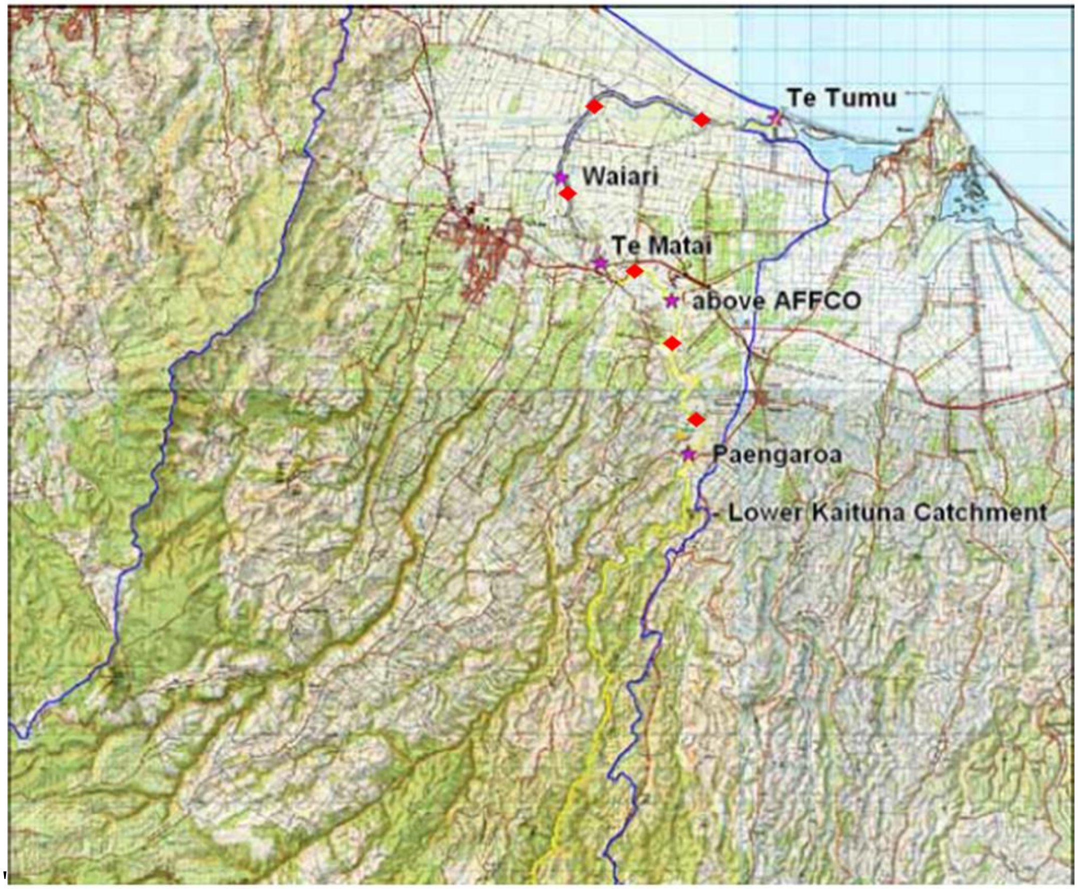

Figure 4.1 Map adapted from Park (2007) displaying the BOPRC sampling locations on the Kaituna River. Red diamonds indicate the sampling sites in the current study for reference.

As the rivers discharge onto the plains, both rivers flow through a series of modified reaches, with many banks armoured to minimise meandering and erosion, and both can be essentially described as a channelised, soft-bottomed lowland river. The entire Kaituna, from lake to sea is ~53 km long, has a residence time of 24 h (Park, 2007), while the Rangitaiki from the Matahina dam has a length of ~34 km and can, by comparison to the Kaituna be expected to have a residence time of <24 h. These short residence times can be expected to constrain the extent to which biological processes will have time to act on biologically active contaminants such as nitrate, while addition of water from confluences and exchanges with groundwater can be expected to have site-specific impacts. To this end it is instructive to compare the dynamics of nitrate in the two rivers, one of which has few large overland tributaries (Rangitaiki) and the other (Kaituna) that has several, most of which at times were found to carry high concentrations of nitrate.

As the Rangitaiki flows onto the coastal plain area of the catchment, the most common pattern seen is a rise in nitrate-N concentration at site two (Te Teko township), evident on 4 of 7 river profiles. By the time the river has reached Te Teko the land use has changed to intensive agricultural practises dominated by dairy but also kiwifruit and other arable land (Figure. 2.1). In this section of the catchment there are no major overland drains nor stream confluences into the main channel and nitrate influx is likely due to groundwater incursion or minor drains. Increase in nitrate between site one and site two is most pronounced in the two December samplings, though these were unremarkable in other measured variables. In terms of discharge, neither of these events corresponded with a high flow in the river, rather both occurred at near to baseflow conditions.

There was also a general tendency for a gradual increase in nitrate-N concentration in the Rangitaiki from sites two through six. This too is consistent with the surrounding land use increasingly dominated by intensive dairy as well as urban with the Te Teko and Edgecumbe townships. Three major highways also cross through the sampled area, SH. 30, SH. 2, and the Pacific Coast Highway (SH 5 also crosses but above Matahina dam). The major issues in urban areas are impervious surfaces (i.e., roofs, roads) which prevent rain soaking in which is immediately turned into surface run-off, collecting nutrients and sediments (Harding, 2004; chapter 35). Additionally, urbanisation typically removes riparian vegetation and turns areas into lawn which decreases interception (Fahey & Rowe, 1992, as cited in Harding, 2004, chapter 34). Urbanisation also creates extensive stormwater networks, delivering run-off directly to the stream or directly to the coast. Te Teko and Edgecumbe are small rural towns likely to have minimal effect on nitrate-N concentrations although we do find that nitrate-N concentrations reach the highest level between sites two and four which covers both townships. However, no obvious patterns of nitrate emerge after passing the townships, although additional sampling may prove useful to assess this in greater detail.

Similarly, there is also a tendency for an increase in nitrate-N concentration between sites one and two on the lowland part of the Kaituna River. In this case, immediately below site one, the Mangorewa River (UT) confluences with the main channel. The UT has shown of the course of sampling consistently high levels of nitrate (see Table. 3.3) which are typically well above those in the main channel of the river. This tributary drains a sub-catchment of the Kaituna, it also happens to be the largest, draining an area of 189 km^2 (32% total catchment area) (Park, 2007). Dairy is conveniently used as a scapegoat for water quality issues in lowland catchments but in this case, we see a significant amount of kiwifruit land cover in this sub-catchment with a small presence of dairying. After speaking with an orchard owner, he noted that this area of the BoP has particularly porous well-draining soil (R. Mayston, personal communication, Dec 4, 2021). Considering the general increasing trend of N fertilizer use for all types of agricultural land, it is likely that kiwifruit cropping is substantive contributor to the elevated nitrate-N concentration in this part of the catchment.

On two occasions, there was a large decrease in nitrate-N concentration between sites one and two on the Kaituna River. For one of these (2/12/20) no tributary sample was available, but for the other (23/3/21), the tributary was rich in nitrate and could be expected to increase concentration. Flows of tributaries were not measured, and this exception may have been due to prolonged dry weather that reduced tributary flow. Indeed, this sampling occasion saw the lowest nitrate-N concentrations between sites two and four, only increasing abruptly after site four where a further three nitrate-rich tributaries, the Waiari River, Waikoura, and Raparapahoe, join the Kaituna main channel.

The Waiari (LT) is the second largest sub-catchment on the Kaituna covering 72 km^2 (12% total catchment area). This sub-catchment drains the Te Puke township, with patches of native bush and wetland higher up in the catchment but the majority of land use is dairy. Additionally, the Te Puke wastewater treatment plant discharges treated sewage to the Waiari River (Bay of Plenty Regional Council, 2009). The Waikoura (UD) and Raparapahoe (LD) Drains, and the Waiari River all join the Katiuna within a 2 km stretch of each other between sites four and five. Both the UD and LD exclusively drain dairy land (Figure. 2.3). Across all sampling occasions there is no substantive change in nitrate-N concentration between sites two and four, and it is only at these tributary confluences that regular change in nitrate-N concentration is evident. Not excluding the gradual seepage of nitrate from groundwater, as inferred by Rangitaiki data, but confirms that abrupt changes are typically a consequence of nitrate rich confluences.

At the same time as an increasing nitrate-N concentration there is a tendency for chlorophyll-a (chl-a) concentration to decline in both rivers from site one and on down through the remaining sites. This is a consistent pattern over the sampling year but displays expected seasonal changes. Site one (Lake Matahina) chl-a concentrations were lowest in winter on the Rangitaiki. It is likely that during winter Lake Matahina is in holomixis, and stratified for the rest of the year, and all but winter sampling showed site one temperatures consistently higher than temperature downstream, by up to c. 3ºC on some sampling occasions. High DO saturation at site one in summer also reflects the level of photosynthesis in the lake, reaching levels up to c. 140% at site one dropping to an expected range (95-105%) at sites two through six. Such high %DO was not seen at site one in the Kaituna River no doubt due to the turbulent flow through the Kaituna Gorge resetting atmospheric saturation.

The Kaituna did not show the same seasonality of chl-a patterns in lowland waters as the Rangitaiki. This reflects the lack of a summer maxima for chl-a in Lake Rotoiti – where rather a weak winter maxima are observed. Such winter maxima in large NZ lakes are not uncommon which reflects increased nutrient supply during winter mixing. As described by Vincent (1983) Lake Taupo experienced its highest algal biomass in winter during mixing at the lowest annual temperatures, which is offset by increased N and P supply, supporting phytoplankton growth. For the majority, we see that chlorophyll concentration change down river is a slow and relatively linear decline, although there are trivial discontinuities observed across the lowland reach. The significant drops in chl-a concentration down-river are likely caused by dilution or grazing by filter feeding benthic invertebrates. No increase in chl-a concentration was observed, despite what would appear to be good growing conditions, with open canopies, high water clarity, and ample nutrients. This most likely reflects the short residence time, discussed above. With <24 hours from site one in each river to the ocean in a highly channelised, short, fast-flowing river there is likely insufficient time for phytoplankton cells to divide faster than they are being washed out, lost to sedimentation or grazing. If the time an algal cell takes to duplicate is not quicker than the residence time of the river a growth will not be realised (Hilton *et al.*, 2006; NIWA, 2015). At the lower sites in both rivers, five and six, we begin to see fluctuations in chl-a concentration, we attribute this to the mixing of marine and freshwater processes, which is supported by the spikes in conductivity due to the residual chloride ions in the water.

4.2 Eutrophication and environmental degradation

Within Aotearoa New Zealand, The Ministry for the Environment (MFE) Default Guideline Value (DGV) for nitrate in lowland rivers, which follows the ANZG guidelines, is 0.44 mg/L nitrate-N (Appendix 1). Exceedance of this value indicates a potential environmental problem requiring management action (ANZECC, 2000; McDowall *et al.*, 2013), but the median values leaving both lakes, given above, are well below this value. The NPS-FM does not currently set limits of nitrate for lakes and rivers, other than for toxicity (currently set at 2.4 mg NO_3-N/L) and to provide advice to manage periphyton growth in cobble-bedded rivers. Soft-bottomed rivers have no specific guidelines on concentrations above which ecological issues, such as nuisance macrophyte blooms can occur within the current NPS-FW.

Total Nitrogen (TN) rather than nitrate is used to gauge trophic levels in lakes and the NPS-FW gives an ‘A’ band reading for trophic status as <0.160 mg/L TN. Under this threshold “lake ecological communities are healthy and resilient, similar to natural reference conditions” (Ministry for the Environment, 2020). The historic median for Lake Rotoiti sits at 0.164 mg N/L, close to this near-reference classification. The historic median TN value for Lake Matahina is 0.540 mg/L (BOPRC data) this places lake Matahina in the NPS ‘C’ band, which states “lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrient levels that are elevated well above natural reference conditions”. Hence, we can see that the levels of nitrate and TN leaving each lake are elevated to some degree above pristine but neither lake exceeds national bottom lines trophic state, ANZECC guidance, or for nitrate toxicity. Although additions of nitrate from the agricultural landscape increase nitrate-N concentrations towards and in some cases in excess of ANZECC guidelines, they do not exceed those for toxicity.

Catchment modelling acknowledges the disproportionately larger footprint dairying has on N flux to the coastal ocean (see Elliot *et al.,* 2005) and the significant impacts on water quality parameters in lowland streams (Wilcock *et al.,* 2006). Pastoral and arable catchments also have higher run-off and storm flows (Dons, 1987), stock tramping compacts the soil and reduces the capacity of infiltration exacerbating overland run-off. Trustpower is obligated under the RMA to maintain a minimum flow of 35 $m^3\ s^{-1}$ under normal inflow conditions (Trustpower, 2017). While studies have shown how increases in flow can also present elevated readings of nitrate and sediment (e.g., Masotti *et al.,* 2018), although this was not evident in this study.

Considering the lack of tributaries on the Rangitaiki, the heavily channelised bank in many areas, and significant groundwater network in a porous geology (Brown, 2018), the nitrate entering the river may reflect nutrients that infiltrate to the water table and eventually end up discharged to the main channel (Jarvie *et al.,* 2013). Ecologists have emphasised the slow delivery of groundwater nutrients in some NZ catchments, which may take decades to emerge from ground to surface water and warned of 'the load to come' from this process (Gluckman, 2017) which is especially true on porous well drained soils (Julian *et al.*, 2017). The geologic characteristics of the Bay of Plenty coastal plain make it susceptible to this process. The Rangitaiki catchment is part of the Taupo pumice country sitting on the ignimbritic Kaingaroa plateau (Phillips & Nelson, 1981) which is relatively porous. Given the inference of groundwater loading of nitrate to maintain and enhance already elevated nitrate-N concentrations in the Rangitaiki and Kaituna lower reaches, managers may need to be conscious of the potentially long recovery time should any remedial action be taken to reduce nitrate load.

Elevated nitrate-N concentrations may create conditions that favour macrophyte growth. Nitrogen and Phosphorushorus are the nutrients that typically limit aquatic macrophyte growth (Barko *et al.,* 1991) and evidence of relationships between nutrients and biomass of macrophytes in stream have been found on many occasions. For example, Mebane *et al.,* (2014) found that increases in biomass were related to sediment and water column nitrogen and sediment loosely held Phosphorus via sorption, and O'Hare *et al.,* (2010) found that increased soluble reactive Phosphorus (SRP) in stream water increases yields in biomass, and that the relationship was strengthened when coupled with an increased bicarbonate availability. However, a relationship between nutrients and biomass is not always clear. Studies that have investigated abiotic influences on macrophyte biomass have also found no apparent relationship between the two, instead sometimes an effect of flow velocity (Giorgi *et al.,* 2005). When studying the effects of eutrophication, it has different effects in different environments (Artigas *et al.,* 2013; O'Hare *et al.,* 2018). Removal and presence of productive species can lead to excessive biomass production, termed "nuisance growth" due to its adverse effect on the ecosystem and its services (Matheson *et al.,* 2012), a phenomenon especially found in nutrient rich lowland streams (Haslam, 1978).

Over the sampled sites we observed dense beds of non-native macrophytes which potentially impact on the nutrient dynamics of the river (Table. 3.5). Species such as *Lagarosiphon major* tend to create dense monospecific beds (present in the Rangitaiki and Kaituna) which are known to control physicochemical conditions within rivers and streams (Esteves & Suzuki, 2010; Wilcock *et al.,* 1999). We saw that the majority of plant *spp* found on the Kaituna contain N at concentrations that is considered to be growth rate saturating i.e., ≥1.14 %N by weight (Demars & Edwards, 2007). Thus, while these plants may have stripped nitrate from the water column to reach these high contents, their ability to do so is likely saturated and in-river concentrations of N (and likely P) are clearly sufficient to support high rates of growth of non-native taxa. In a report prepared for DairyNZ, NIWA (2015) surmised that increased mobilisation and senescence of *Ceratophyllum demersum* (hornwort) within Lake Karapiro, (also present in the Rangitaiki and Kaituna) likely contributed to downstream increases in nutrient concentration. This could be consistent with findings in this study in terms of contribution to increased downstream nitrate-N concentrations. However, it was unlikely that this process had a significant influence on nitrate concentrations within the river during this study due to the near base flow conditions across the sampling year.

4.2.1 Total suspended solids

Suspended solids can show higher levels with elevated flow and changing land use, similar to nitrate but the process is nuanced due to the differing contaminant pathways. The NPS-FW sets limits for SS within rivers, however, unlike in this study the units are expressed as visual clarity (m). Under the RMA, regional councils have an obligation to monitor SS levels but choose to measure turbidity (NTU) and convert the measure to visual clarity (m) (Ministry for the Environment, 2020). Elevated levels of SS are a significant indicator of physical and aesthetic degradation of a river or stream and a good indicator of other contaminants, especially Phosphorushorus, but can also include heavy metals (Affandi & Ishak, 2019; Packman *et al.*, 1999).

The measures of SS and turbidity in the Rangitaiki and Kaituna Rivers more often than not mirror each other, as would be generally expected. The majority of recorded values in the current study are not high in regard to TSS or NTU in the Kaituna, reaching a maximum of 5 mg TSS/L and 2 NTU, on the sampling occasion with maximum river discharge (02/12/20). In Lake Matahina (site one) a single sample reached 10 mg TSS/L and 5.5 NTU, though this was anomalous. Studies suggest that SS concentration of 8 mg/L and between 7-10 NTU can cause increased drift in benthic invertebrates (NIWA, 2018; Rosenburg & Wiens, 1978). These higher levels were only found on a singular sampling occurrence within Lake Matahina, but never in the main channel where a drift response could have been elicited.

On occasions SS and turbidity show a level of separation, with SS being higher than NTU, and the ratio of NTU to SS was different for the two rivers. Packman *et al.,* (1999) suggests that higher SS for a given turbidity may be explained by an increased level of fine particulate organic matter (FPOM). Although, Gilvear & Petts, (1985) find the opposite relationship concluding that the differences were in optical properties of particulate matter thus we cannot be certain that it is specific fines or FPOM creating discontinuity in SS levels or NTU in this study. It is well cited that nitrate and suspended solids are both well correlated with and respond well to increased flow (Lewis *et al.,* 2007; Neal *et al.,* 2006; Sloto *et al.,* 2012). Mentioned above, both are good indicators of water quality degradation but neither variables when analysed together show any relationship, due to the differing contaminant pathways. Regression analysis indicated that there is no relationship between nitrate and suspended sediment in either river (Figure. 3.13 and 3.14). Perhaps an oversight but building a model that factors flow may go some way to explain the relationship between nitrate and SS seen in both rivers.

4.2.2 Rangitaiki and Kaituna Rivers from a management perspective

A picture thus emerges of two rivers enriched in nitrate, likely by combinations of direct nutrient delivery i.e., surface run-off, tributary confluences, and indirect via contaminated groundwater, but within which the adverse impacts of eutrophication are not causing serious degradation. Oxygen concentrations are close to 100% most of the time, pH is circum-neutral, TSS is low, and clarity high, and while exotic macrophytes proliferate they rarely impede flow. Rapid, turbulent flow may play a part in prevention of large accumulations of macrophytes or periphyton through scouring. Likewise, turbulent flow in relatively shallow channels prevents excessive fluctuation of oxygen saturation or pH, and short residence time prevents accumulation of phytoplankton. During sampling trips trout were frequently seen, as were mullet, and trout are often associated with high quality aquatic habitats.

To a large extent we suggest that the lowland sections of these two rivers act as rapid conduits of nutrient-enriched water, but carrying relatively low concentrations of suspended sediments, and showing few direct symptoms of eutrophication, to the receiving estuaries and coastal waters. Overall, the Rangitaiki and Kaituna currently satisfy the limits set out by the NPS-FW for elements of their overall ecological status i.e., DO, nitrate toxicity, but regularly exceed ANZECC trigger value guidelines for nitrate (Appendix. 1). What we have identified in this study is the nuanced cause-and-effect relationships for two major land-use related contaminants which are not directly addressed for lowland rivers in current policy. Meaning management for lowland rivers remains ambiguous due to limits on key drivers of environmental change remaining underdeveloped. For example, the levels of nitrate above which blooms are facilitated and dense monospecific beds of non-native macrophytes, much like what is already seen in the Waikoura, and Raparapahoe Drains are not part of the thinking within the NPS-FM framework. Likewise, they are providing little advice relevant to this type of water body on suspended solid concentration with its propensity to smother the benthic macroinvertebrate community as well as to deliver Phosphorus. Park (2007) states that the BOPRC are seeking to obtain better water quality data from all tributary rivers to help build models to predict changes in water quality parameters over time.

Similar studies note that lowland water quality trends and are highly variable across the country and can be site specific, although the consensus from such studies surmises that the general state of lowland rivers is poor (Ballantine & Davies-Colley, 2014; Wilcock *et al.*, 1999). However, some improving trends in lowland streams were found for certain indicators of water quality (see Appendix. 2). Ballantine *et al.*, (2010) found stable trends in NH_4, DRP, and TP (1998-2007), Larned *et al.*, (2016) reported improving 10-year trends in visual clarity and TP at natural and pastoral sites, outnumbering degrading trends; but reported degrading trends of NH_4 at natural sites and NO_3-N at pastoral sites (2004-2013). Ballantine *et al.*, (2010) also reported that 5-year trends tended to be stronger i.e., greater rates of change than the 10-year trends showed, giving the example that the NO_3-N trend in the Waikato River was stronger between 2003-2007 than 1998-2007 potentially indicating an accelerated rate of degradation in water quality. Larned *et al.*, (2004, 2016) also stated in both studies that NO_3-N concentrations at pastoral and urban sites, exceeded the ANZECC trigger values, likewise found in the current study.

A 21-year analysis (1989-2009) of the NRWQN dataset found that the national trend was increasing, especially for TN and NO_3-N, additionally with a declining trend of DO and an increase in conductivity, implying and overall decrease in water quality (Ballantine & Davies-Colley, 2014). However, over the same period Ballantine & Davies-Colley, (2014) found improving trends of visual clarity and high concentrations of SS nor NTU, that would support high clarity as evident in this current study. Larned *et al.,* (2004) reported measures of optical variables in pastoral catchments were still only 30-60% of what was seen in native catchments. Overwhelmingly, the most significant impacts on lowland rivers in Aotearoa New Zealand come from high-producing pasture that requires synthetic fertilizer addition to support large herd sizes to maximise productivity (Julian *et al.*, 2017). Not only for dairying, but also high intensity beef, which have equivalent nutrient requirements (McDowall *et al.*, 2008 as cited in Julian *et al.*, 2017). As these studies have highlighted the negative influence of high intensity agriculture on low elevation streams with the overall declines in water quality as mentioned above. However, the proliferation of deleterious effects of eutrophication are likely river-specific relating to the land use, catchment geology, and the hydrology of the system.

4.2.3 Ecological effects on receiving environments

We suggest future studies should look toward the need for integrated catchment management due to the significant effects of upstream land use on the downstream environment. This is especially true in areas that have intensive pastoral agriculture and large-scale urban developments within the catchment much like the Rangitaiki and the Kaituna. Estuaries, due to being at the distal end of catchments, receive per unit area, a greater amount of nutrients than any other ecosystems (Howath, 1993, as cited in Smith, 2003) and with the increasing concern regarding the effects of deteriorating N trends, receiving environments are highly susceptible to the effects of eutrophication (see Table. 1.1). Although the effects of excess nutrient delivery have not been realised in the Maketu estuary (which sits at the bottom of the Kaituna catchment) from river sources, as the river was channelised in the mid-1950s directly out to sea, by-passing the estuary for flood control purposes (Everitt & De Monchy, 2013) and only recently diverted back to the estuary (Bay of Plenty Regional Council, 2022). With our findings of nutrient enriched tributaries and enhancing nitrate-N concentration downstream, this, coupled with the re-diverted flow into the estuary there may be a risk of increasing undesirable effects of excess nutrient delivery, echoed by Everitt & De Monchy, (2013).

The Rangitaiki does not have the same extensive estuarine habitat at the bottom of the catchment but a small lagoon (Okorero/Thornton Lagoon) which is the only remaining wetland area in the lower Rangitaiki. Due to the conversion of the lowland reaches to pasture as well as the channelisation of the river itself the lagoon has already suffered from environmental degradation (Bay of Plenty Regional Council, 2017). Although the BOPRC have installed culverts to reconnect the lagoon to the main channel in hopes of remediating its water quality the effects of nutrient enrichment. It is unclear whether the degradation of the Okorero Lagoon is owed to the exchange with the current channel or a combination of groundwater influx and surface run-off but likely a combination of multiple stressors.

4.3 Study limitations

There were practical drawbacks of the study, one of the largest constraints and most challenging to navigate was the COVID-19 (CV-19) pandemic. The lockdowns that followed with the pandemic limited sampling time and forced less ideal, out of season samplings. Additionally, the limited amount of time permitted for an MSc degree further constrained the number of samples taken on each river in turn limiting our understanding of broad scale patterns within the two sampled rivers. As discussed above, there have only been two national scale studies into lowland river water quality leaving limits for key ecological drivers in the NPS-FW are ambiguous. Firstly, further research is required to understand the dynamics of nutrients (N and P) and sediments in such systems; secondly to alleviate the disconnect between policy and management practices creating robust scientifically backed policy. Another constraint was the omission of lagrangian sampling from this study, deemed impractical due to various river hazards, access issues, and personal safety concerns. Lagrangian sampling would have tracked packets of water, assessing the properties as it moves through a system; this method facilitates a broad understanding of the spatial extent of processes that affect water quality parameters (Kraus *et al.*, 2017). In contrast, the eulerian methods place a lesser weighting on spatial variability and the interactions between position and time in the river reaches. A broad spatial extent of both river reaches would have allowed the study to make a more definitive assessment to each river system thus creating a greater wealth of knowledge of lowland river catchments in the Bay of Plenty. Not only that but also being able to directly compare results from each method, allowing advice to be given on whether lagrangian sampling would be worth rolling out for wide scale use.

As we move forward in a post pandemic world, if the study were to be repeated the sample collection would have been better planned and seasonally less sporadic. Most samples were taken between 0800-1400 so temperatures would vary somewhat for the early afternoon samples opposed to morning. Although, seeing winter and autumn readings as well as morning against afternoon samples are useful metrics to consider daily fluctuations as well as for long term river management. Although interesting it may have introduced artefacts to the data sets, throwing out averages from simply sampling at certain periods over the day, month, or year.

4.4 Conclusions

The concept of integrated catchment management requires an intimate knowledge of the entire hydrological system, what activities affect contaminants, and how they transform across the river continuum. The Rangitaiki and Kaituna Rivers both showed changes in all measured variables as they flowed through their lower catchments that were ultimately relatable to land use. Findings suggest that there are no clear relationships between nitrate and suspended sediment concentrations within both rivers, with seasonality, river confluences, and changing land-uses within the catchment affecting one or both variable. We surmise the lower Rangitaiki River nitrate-N concentration is largely dictated by seasonality within Lake Matahina and activities within the upper catchment, likely supplemented by groundwater influx and minor drains discharging to the river across the coastal plain. Whereas the Kaituna is predominantly a product of the intensive agricultural catchment, with four significant tributary inflows along the lowland reach, all of which are nitrate rich. We see no predictability between nitrate and suspended solids due to the differing contaminant pathways concluding suspended solid concentrations are likely dictated by the flow regime. Additionally, we find the inverse of what was hypothesised with chlorophyll-nitrate relationships, due to winter maxima within Lake Matahina and Lake Rotoiti, the absence of increasing concentration downriver likely reflecting the short residence times. Attenuation of neither nitrate nor sediment was evident across the coastal plain. Overall, we found analogous studies, supplemented by historical records, pointed to similar results as this study, progressive deterioration of water quality within lowland rivers. However, despite high nutrient loadings, other than abundant non-native macrophyte growth, the Rangitaiki and Kaituna lacked any other significant deleterious effects of eutrophication within the river or on the receiving environment. We feel that the omission of key drivers of environmental change (i.e., nitrate) illustrates a significant disconnect between science and policy, with a paucity of study in this area, requiring attention from policy makers and resource managers moving forward.

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Appendices

Appendix 1: ANZG trigger value levels, in exceedance (disregarding DO) requires management action to be taken to remediate water quality (McDowall *et al.*, 2013).

Indicator	Upland river	Lowland river
FRP (µg L^{-1})	9	10
TP (µg L^{-1})	26	33
NO_x (µg L^{-1})	167	444
NH_4 (µg L^{-1})	10	21
TN (µg L^{-1})	295	614
pH upper limit	8.0	7.8
DO (% saturation) lower limit	99	98

Appendix 2: Physical and chemical indicators of water quality degradation (McDowall *et al.*, 2013).

Indicator type	Indicator name	Description	Units
Physical	Clarity	Black disc visibility	m^{-1}
	Conductivity	Electrical conductivity	µS cm^{-1}
	SS	Suspended solids	mg L^{-1}
	pH	Hydrogen ion concentration	
	DO	Dissolved oxygen	%
	Turbidity	Turbidity	NTU
	Temperature	Water temperature	^{o}C
Nutrients	NH_4-N	Ammoniacal nitrogen	µg L^{-1}
	NO_3-N	Nitrate	µg L^{-1}
	TN	Total nitrogen	µg L^{-1}
	FRP	Filterable reactive phosphorus	µg L^{-1}
	TP	Total phosphorus	µg L^{-1}
Faecal indicator bacteria count	*E. coli*	*Escherichia coli*	MPN 100 mL^{-1}