An assessment of critical nutrient (nitrogen and phosphorus) loss pathways and potential mitigation options for improved water quality outcomes in an intensive pastoral farming area in the lower Manawatu River Catchment.

A thesis presented in partial fulfilment of the requirements for the degree of

Master of Environmental Management

at Massey University, Manawatu, New Zealand.

Ben Jonty Woolston

2026

Abstract

New Zealand's rapid agricultural intensification has created significant water quality challenges, particularly in catchments where intensive dairy farming has been established on artificially drained former wetlands. The Moutoa-Whirokino catchment (6,000 hectares) in the lower Manawatu River system exemplifies this transformation, with extensive drainage infrastructure supporting intensive pastoral agriculture on organic-rich soils that discharge to the internationally significant Manawatu Estuary (Ramsar Convention Wetland of International Importance). Preliminary monitoring revealed unusual water quality patterns, namely elevated ammonia-nitrogen and dissolved reactive phosphorus alongside unexpectedly low nitrate-nitrogen which differed from typical agricultural drainage in other catchments which are dominated by nitrate losses.

This research characterised shallow groundwater and drainage nutrient dynamics through eleven months of intensive monitoring (November 2023-October 2024) across ten groundwater piezometers and twenty-three surface water sites. Groundwater analysis revealed predominantly anoxic conditions (74% of 109 samples) controlling nitrogen speciation, with ammonia-nitrogen concentrations measured 6.79-18.30 g/m^3 (median values) at the organic soil groundwater sites, as compared to 1.47-1.54 g/m^3 in the sandy soil sites. Soil type exerted strong control on shallow groundwater and drainage nutrient mobilisation, with organic matter decomposition under oxygen-limited conditions driving coupled ammonia-nitrogen and dissolved reactive phosphorus release. However, surface drain water monitoring showed substantial spatial variability, with ten of twenty-three sites exceeding national and regional ammonia-nitrogen standards, while all of the sites exceeded regional dissolved reactive phosphorus targets.

Evaluation of mitigation strategies demonstrated that controlled drainage successfully manipulated water table elevations by 0.4-0.5 metres, while existing borrow pits in the flood spill way indicated remarkable potential treatment performance with the median ammonia-nitrogen and dissolved reactive phosphorus concentrations in the spill way measured 0.042 g/m^3 and 0.097 g/m^3 respectively, as compared to 0.43 g/m^3 and 0.15 g/m^3 measured in the farm drains. These findings establish that ammonium-dominated drainage from anoxic organic soils requires fundamentally different mitigation approaches than conventional nitrate-focused strategies, providing guidance for managing nutrient losses from New Zealand's extensive drained peatland agricultural systems.

Keywords: organic soil, peat soil, intensive agriculture, ammonia-nitrogen, anoxic groundwater, dissolved reactive phosphorus, nutrient mitigation, artificial drainage, redox conditions, Manawatu Estuary, Manawatu River, controlled drainage, shallow groundwater, water quality.

Acknowledgements

This piece of work would not have been possible without a plethora of people. They say it takes a village to raise a child…I think it takes a big network of supportive people to undertake (and complete!) a piece of work like this.

I begin by acknowledging my supervisor, Professor Ranvir Singh. Upon completion of my undergraduate studies, I told myself (and my wife) I would never go back to study again. However, one of the key catalysts for wanting to undertake this research has been Ranvir, who sparked an interest in catchment hydrology and nutrient dynamics during undergraduate lectures. Ranvir, your wealth of knowledge, pragmatism, and honest advice and guidance have helped me to get through this work successfully. Thank you.

I'd like to thank several others at Massey University who have helped or contributed to this work in various ways:

- The Catchment Solutions Team: thanks for the banter, for putting up with my questions and requests for help, and for lending guidance and practical support where I needed it.
- David Feek: It's astounding how many theses you are acknowledged in David. Without your technical expertise, there would have been no piezometers to sample. Thanks for teaching and helping me in the initial set-up of the research.
- Dr Callum Rees: For the many hours of my standing around your office (either inside or at the door) and answering my questions about anything soil-related!

I have received several generous scholarships over the course of my studies, but would like to particularly thank both the George Mason Charitable Trust and Freemasons NZ for their financial support to get through this study, while supporting a family.

I'd also like to acknowledge the support and drive of the Moutoa-Whirokino Catchment Group who sought to undertake this research in their catchment. Thank you for allowing me to access your land, and for giving me insights into your catchment and its history. Special mention must be made of the McErlean family, Tony Dowman, and Fiona Burke.

In order to be able to complete this thesis and afford to support my family, my employer has been very supportive. They haven't batted an eyelid when I requested to work reduced hours, juggle days worked, or take a day off to work on my thesis. To the Regulatory Team – you guys are awesome, and got me through some crazy times. Thanks to my (several) Team Leaders, and particularly to Greg Bevin, and now Logan Brown for continuing to support me to this point. Thanks to Andrew Steffert for his expertise in providing data and access to historic imagery.

Finally, to my family. Luther, Emmylou, Phoebe and Fleur – thankfully you guys have been mostly too young to see how crazy, sleepless, and stressed Dad has been at times. But you have brought an element of relief and fun to my days when things seemed all a bit hard and serious. I hope I've been a good example to you through this period, and someone whom you aspire to emulate (in the good things) as you grow older. Rosie – I think it will only ever be you and I who truly know the sacrifices you have made for me to complete this thesis. Thank you for the love, the support, the coffees, and for listening to me talk about geeky stuff which you don't really care about.

Praise the Lord, praise the Lord,
let the earth hear His voice!
Praise the Lord, praise the Lord,
let the people rejoice!
O come to the Father through Jesus the Son,
and give Him the glory, great things He has done.

F. Crosby, 1875

Table of Contents

Table of Contents

List of Figures

List of Tables

Chapter 1

1. Introduction

1.1 Background and context

New Zealand has experienced one of the highest rates of agricultural land intensification in the world in recent times, with dairy cattle numbers increasing by 71% nationally between 1990 and 2023 (from 3.4 million to 5.9 million), while nitrogen fertiliser applications increased by 629% between 1991 and 2019 (Ministry for the Environment & Stats NZ, 2023; Stats NZ, 2024; Stats NZ, 2021c). This agricultural transformation, driven primarily by the comparative economic profitability of intensive dairy farming, has fundamentally altered nutrient flows in agricultural catchments. The conversion from extensive sheep and beef farming to intensive dairy farming has resulted in higher inputs of synthetic fertilisers, increased stocking rates, expansion of irrigation systems, and, critically, extensive artificial drainage of poorly-drained soils (Ministry for the Environment & Stats NZ, 2023; Manderson, 2018).

The environmental consequences of this intensification have become increasingly evident in New Zealand's freshwater systems. Current national statistics reveal that 69% of New Zealand's river length has nitrogen concentrations at risk of environmental impairment, while 63% has phosphorus concentrations above natural ranges (Stats NZ, 2022a; Stats NZ, 2022b). Agriculture is considered the main pressure on water quality in New Zealand, with modelling showing that dairy farming contributes a disproportionate load of nutrients relative to land area, for example 37% of total nitrogen load from only 6.8% of land (Elliott et al., 2005). These pressures have been particularly severe in catchments where intensive agriculture coincides with former wetland areas that have been artificially drained, creating efficient pathways for nutrient transport from agricultural land to receiving waters.

1.2 The Manawatu River catchment: A critical case study

The Manawatu River catchment exemplifies New Zealand's struggle to balance intensive agricultural development with environmental protection. Flowing 235 kilometres from the Ruahine Range to the Tasman Sea at Foxton, the Manawatu River drains approximately 5,900 square kilometres of predominantly agricultural land. Research has documented that 98% of nitrogen loads and 80% of phosphorus loads in the upper catchment are derived from diffuse agricultural sources (Roygard et al., 2012; McArthur & Patterson, 2007). Water quality monitoring has revealed severe nutrient enrichment, with the Manawatu River historically recording among the worst readings for dissolved oxygen and primary productivity in comparison to 300 rivers and streams across North America, Europe, Australia and New Zealand (Morgan & Burns, 2009; Clapcott & Young, 2009).

Recent monitoring data demonstrates the persistent nature of these water quality challenges. Between June 2017 and June 2022, only 32% of river monitoring sites in the Horizons Region met Regional Plan targets for Soluble Inorganic Nitrogen (SIN), while just 22% of sites met targets for Dissolved Reactive Phosphorus (DRP) (Horizons Regional Council, 2025a). At Whirokino, near the Manawatu River mouth, the five-year median DRP concentration places the site in the worst 25% of all monitored sites nationally, with DRP showing a very likely degrading trend (LAWA, n.d.-b).

The ecological and cultural significance of the Manawatu River system extends beyond the river itself. The Manawatu Estuary, which receives inputs from the entire catchment, was designated a Wetland of International Importance under the Ramsar Convention in 2005, recognising it as one of the largest estuaries in the North Island that retains a high degree of naturalness and endemic biodiversity. The estuary supports approximately 170 bird species, including nationally important populations of migratory shorebirds, and provides critical habitat for up to 30 threatened or nationally critical fish, plant and bird species (Ramsar Sites Information Service, 2023; Department of Conservation, n.d.; Manawatu Estuary Trust, 2025). However, the estuary faces significant environmental pressures from cumulative nutrient loading, with water quality data showing failure to meet Regional Plan targets for both SIN and DRP between June 2017 and June 2022 (Horizons Regional Council, 2025a).

1.3 The Moutoa-Whirokino catchment: Drained organic soils and unique nutrient challenges

Within the lower Manawatu River catchment, the Moutoa-Whirokino subcatchment represents a landscape that has undergone particularly dramatic transformation. Encompassing approximately 6,000 hectares, this area was historically dominated by extensive wetlands, with the Moutoa and Makerua swamps alone covering many thousands of hectares of flax-dominated wetland systems. These wetlands provided abundant ecological and cultural resources, including mahinga kai (food gathering sites) and materials for traditional Māori industries, while also performing critical hydrological functions such as flood water storage, groundwater recharge, and nutrient filtration (Bataille et al., 2021; Harmsworth, 2002; An & Verhoeven, 2019; Bullock & Acreman, 2003).

Beginning in the 1880s and accelerating through the 20th century, this wetland landscape was fundamentally altered through artificial drainage and flood control infrastructure. The arrival of mechanical excavation equipment in the 1920s enabled large-scale drainage of previously waterlogged areas, while the construction of the Moutoa Floodway and Sluice Gates (completed in 1962) provided comprehensive flood protection for 28,000-32,000 hectares of agricultural land (Horizons Regional Council, n.d.). The catchment now operates under an extensive drainage scheme managing 77.7 kilometres of drains, 42 floodgates, and six pump stations, with a total asset value of approximately $11 million as of June 2024 (Horizons Regional Council, 2025b). Current land use is dominated by high producing exotic grassland (92% of land cover) and short-rotation cropland (4.4%), characteristic of intensive pastoral dairy farming (Landcare Research New Zealand Ltd., 2020).

This transformation has created a landscape where diverse soil types (ranging from well-drained wind-blown sands to poorly drained organic peaty loams formed in historic swamp areas) now support year-round intensive agriculture through artificial drainage. The organic-rich soils, particularly the Makerua and Kairanga peaty silt loams which together account for over 2,100 hectares, contain high organic matter content (up to 35% in surface horizons), creating unique challenges for nutrient management (Cowie, 1978). International research on drained peatlands has demonstrated that drainage-induced aerobic decomposition of organic matter accelerates nutrient mineralisation, releasing both nitrogen and phosphorus that were previously locked in organic forms (Tiemeyer et al., 2007; Van Beek et al., 2007; Becher et al., 2023). However, the specific biogeochemical conditions and nutrient transformation processes occurring in New Zealand's intensively drained organic soils under pastoral dairy systems remain poorly understood.

1.4 Knowledge gaps and research rationale

Preliminary monitoring conducted by the local catchment group (Manawatu River Catchment Collective) beginning in December 2022 revealed an unusual pattern of water quality in the Moutoa-Whirokino drainage system: elevated ammonia-nitrogen and dissolved reactive phosphorus concentrations alongside unexpectedly low nitrate-nitrogen levels in both surface drains and shallow groundwater. This pattern of nitrogen speciation differs markedly from the nitrate-dominated losses reported in international literature on drained peatlands and from typical New Zealand pastoral systems, where nitrate-nitrogen typically comprises more than 80% of dissolved nitrogen exports (Monaghan et al., 2016). The dominance of ammonia-nitrogen over nitrate-nitrogen suggests that site-specific redox conditions, organic matter characteristics, drainage infrastructure design, and soil type variability may be creating unique nutrient transformation pathways within this system.

Despite the extensive body of international research on nutrient dynamics in drained organic and peat soils, particularly from Northern Europe and North America, critical knowledge gaps remain regarding these systems within the New Zealand context. The Manawatu region, with its predominance of artificially drained former wetlands now supporting intensive dairy farming, presents a unique combination of soil types, climatic conditions, and land management practices that may result in nutrient dynamics distinct from those documented internationally. Understanding the origins of elevated ammonium, the hydrological and biogeochemical pathways facilitating nutrient transport from different soil types to surface waters, and the interaction between shallow groundwater and the extensive artificial drainage network is essential for developing effective, practical mitigation strategies tailored to this landscape.

Furthermore, the designation of the Manawatu Estuary as a Wetland of International Importance under the Ramsar Convention creates an urgent imperative to understand and mitigate nutrient losses from these drained organic soil systems. The estuary's international significance for biodiversity conservation, combined with documented water quality degradation and declining populations of key species such as migratory shorebirds (Ramsar Sites Information Service, 2023), underscores the need for science-based management approaches grounded in thorough understanding of local nutrient dynamics.

1.5 Research aim and objectives

This research addresses critical knowledge gaps by investigating nutrient loss pathways and potential mitigation options specific to the Moutoa-Whirokino catchment. The overarching aim is to characterise nitrogen and phosphorus dynamics in an intensively drained organic soil agricultural landscape, and to evaluate practical mitigation approaches for reducing nutrient export to the internationally significant Manawatu Estuary.

Specific research objectives are:

1. To characterise shallow groundwater nutrient concentrations, redox conditions, and hydrochemistry across the catchment's diverse soil types, establishing the biogeochemical framework controlling nutrient mobilisation.
2. To quantify spatial and temporal patterns of nutrient concentrations in the surface drainage network, and to assess relationships between groundwater sources and surface water quality.
3. To identify the dominant pathways and mechanisms facilitating nutrient transport from agricultural land through the drainage system to receiving waters.
4. To evaluate the effectiveness of two complementary mitigation approaches: controlled drainage for source control through water table management, and borrow pit systems for in-stream nutrient treatment.
5. To provide farmers and catchment managers with locally relevant information essential for improving water quality outcomes for the lower Manawatu River and its ecologically important estuary.

1.6 Thesis structure

This thesis is organised into seven chapters. Chapter 2 (Literature Review) synthesises existing knowledge on agricultural intensification and water quality impacts in New Zealand, with particular focus on the Manawatu River catchment. It examines nutrient dynamics in artificially drained agricultural systems, the unique characteristics of organic and peat soils, and edge-of-field mitigation strategies, establishing the theoretical framework and identifying knowledge gaps that this research addresses.

Chapter 3 (Study Area Description) provides comprehensive characterisation of the Moutoa-Whirokino catchment, including its geographic setting, climate, soil types, land use history, and the development of drainage infrastructure that has transformed this historic wetland into intensive agricultural land.

Chapter 4 (Materials and Methods) describes the research design, field sampling protocols for groundwater and surface water monitoring, laboratory analytical methods, and statistical approaches used to analyse spatial, temporal, and hydrochemical patterns.

Chapter 5 (Results and Discussion) presents and interprets findings across four main themes: shallow groundwater characteristics including water table dynamics, redox conditions, and nutrient mobilisation processes; surface water nutrient patterns and their relationships with rainfall and seasonality; and groundwater-surface water connectivity.

Chapter 6 (Nutrient mitigation strategies) evaluates the controlled drainage and borrow pit mitigation strategies.

Chapter 7 (Synthesis and Conclusions) synthesises key findings, discusses their implications for nutrient management in drained organic soil landscapes, acknowledges research limitations, and provides recommendations for catchment management and future research directions.

Together, these chapters provide a comprehensive assessment of nutrient dynamics in an intensively drained New Zealand peatland catchment, offering insights essential for protecting water quality in similar landscapes throughout New Zealand and internationally.

Chapter 2

2. Literature Review

2.1 Intensive agriculture and water quality

2.1.1 Scale and context of agricultural intensification in New Zealand

New Zealand has had one of the highest rates of agricultural land intensification in the world in recent times (Ministry for the Environment & Stats NZ, 2023). Between 1996 and 2018 in New Zealand, almost 60,000 hectares of exotic grassland was converted from low producing to high producing, compared with only 3,500 hectares of exotic grassland converted from high to low producing (Stats NZ, 2021b). Additionally, between 2002 and 2019 the area of land used for dairy farming increased 81%, by 991,000 hectares (Stats NZ, 2021a). This data illustrates the significant trend towards agricultural intensification. This transformation has been particularly seen in pastoral dairy farming, with dairy cattle numbers increasing by 71% nationally between 1990 and 2023, from 3.4 million to 5.9 million (Stats NZ, 2024). In parallel with this, sheep numbers have decreased by 57% in the same time period (57.9 million to 25.1 million) (Stats NZ, 2024).

An analysis of the drivers and barriers to land use change in New Zealand has demonstrated that economic factors have been the primary catalyst for this transformation (Parliamentary Commissioner for the Environment, 2004). The comparative profitability of dairy farming in relation to traditional pastoral systems has created powerful incentives for land use conversion, with dairying showing threefold economic returns over arable farming in Canterbury, and twelvefold returns over sheep farming in Southland during the intensification period (Ministry for Primary Industries, 2017).

Land use change fundamentally alters the nature of nutrient flows in agricultural catchments (Stats NZ, 2024). The conversion from extensive sheep and beef farming to intensive dairy farming equates to more than a mere change in livestock numbers - it illustrates a shift toward higher inputs of synthetic fertilisers, increased stocking rates, an increase in irrigation system construction and use in drier areas, and an increase in artificial drainage of land in poorly-drained soils (Ministry for the Environment & Stats NZ, 2023; Manderson, 2018). For example, it is estimated that between 1991 to 2019 the amount of nitrogen fertiliser applied to land has increased from 62,000 to 452,000 tonnes, or by 629% (Stats NZ, 2021c). There is a strong link between nitrogen loss from intensive agriculture and the number of cows per hectare on that land (Parliamentary Commissioner for the Environment, 2004). When excess nutrients from sources such as fertiliser or livestock urine can't be used by plants, leaching of these nutrients down through the soil profile can occur (Ledgard et al., 1999). Nitrate-nitrogen can leach in this way, where it can then enter groundwater and surface water bodies (Ministry for the Environment & Stats NZ, 2023; Di & Cameron, 2002).

2.1.2 General water quality and ecological impacts

The nutrient enrichment associated with intensive agriculture has implications for both freshwater quality and ecosystems, and their ability to support native biodiversity (McDowell & Wilcock, 2008). Agriculture is considered the main pressure on water quality in New Zealand - whether that be ground or surface water (Parliamentary Commissioner for the Environment, 2004). This has come to the fore in the last several decades with improvement to point source discharges (e.g. wastewater treatment plant discharges to surface water) resulting in water quality pressures coming primarily from diffuse sources (Davies-Colley, 2013; Ballantine & Davies-Colley, 2010). Modelling has shown that dairy farming contributes a disproportionate load of total nitrogen when compared to total land area. Elliott et al. (2005) found that 37% of the total nitrogen load of streams in New Zealand came from only 6.8% of the land.

Current national statistics reveal alarming trends. Models estimate 45% of New Zealand's total river length was unsuitable for recreation between 2016 and 2020, based on faecal contamination risk (Ministry for the Environment & Stats NZ, 2025). There is a growing trend of groundwater being unsafe for drinking, either because of faecal contamination or nitrate-nitrogen exceedance levels. Between 2019 and 2024, around half of the groundwater monitoring sites in New Zealand exceeded the maximum acceptable values for drinking water at least once based upon several groundwater quality measures (Stats NZ, 2025).

Excess nutrient levels are also causing stress and degradation in some river and lake ecosystems in New Zealand agricultural landscapes. In comparison with reference conditions, between 2016 and 2020, 69% of New Zealand's river length had nitrogen concentrations modelled at risk of environmental impairment (Stats NZ, 2022a). Alongside nitrogen, there have been concerning trends in phosphorus contamination found in national water quality data. Between 2013 and 2017, 63% of the country's total river length had phosphorus concentrations modelled above the expected range for natural conditions (Stats NZ, 2022b).

The ecological consequences of nutrient enrichment extend beyond chemical thresholds and can lead to changes in aquatic ecosystems. Excessive nitrogen and phosphorus in freshwater ecosystems drive excessive algal growth and eutrophication, leading to algal blooms that reduce oxygen availability and alter food web dynamics (Davies-Colley, 2013; McDowell & Wilcock, 2008). About 46% of New Zealand's lakes larger than 1 hectare had poor or very poor health in terms of nutrient enrichment between 2016 and 2020, with only 2% being rated good or very good (Ministry for the Environment & Stats NZ, 2023).

A change in long term river water quality across New Zealand from 1990 to 2017 correlated with the proportion of upstream land in pastoral agriculture and how intensive that agriculture was (Snelder et al., 2021). Another example of this has been seen in the Waikato, where between 1992 and 2002, the number of cows increased by 37%, while nitrogen levels in the streams increased by 40% and phosphorus by 25% (National Institute of Water and Atmospheric Research, 2017).

2.2 The Manawatu River catchment and Estuary

2.2.1 Historic context and land use evolution

The Manawatu River catchment and its associated estuary are a critical freshwater system that has become symbolic of New Zealand's struggle to balance intensive agricultural development with environmental protection. Flowing 235 kilometres from the eastern slopes of the Ruahine Range to the Tasman Sea at Foxton, the Manawatu River drains approximately 5,900 square kilometres of predominantly agricultural land, making it one of the most significant waterways in the lower North Island (LAWA, n.d.-a).

The transformation of the Manawatu River catchment from its pre-European state to the present intensive agricultural landscape is an example New Zealand's dramatic anthropogenic environmental and land use changes. Early European records describe a landscape dominated by extensive wetlands, particularly in the lower catchment, with large flax swamps covering thousands of hectares. The Moutoa swamp alone encompassed approximately 1,600 to 2,000 hectares of flax, while the nearby Makerua swamp extended over approximately 5,000 hectares from Foxton upstream to Linton (Kirkland, 1970; Tannock, 1968).

These wetland systems served multiple ecological and cultural functions. For Māori, they provided abundant mahinga kai (food gathering sites), medicinal plants for rongoa (Māori medicines), and resources for traditional industries, with flax harvesting being particularly important for the production of textiles, nets, and other essential items (Bataille et al., 2021; Harmsworth, 2002). The extensive wetlands likely also performed many hydrological functions including storing floodwaters during high rainfall periods, releasing them to recharge groundwater and supplement low flows, and filtering nutrients and sediments from runoff and groundwater flows (An & Verhoeven, 2019; Bullock & Acreman, 2003).

European colonisation begun a transformation of this landscape, driven by agricultural development and the necessitation of flood control and drainage implementation for developing agricultural lands (McWethy et al., 2010). The use of fire as a tool to clear vast swathes of land became widespread, removing much of the native forest cover in preparing the land for pastoral agriculture (Peden, 2009; Knight, 2018). The commercial harvest and use of flax in the region began in the early 1800s and continued through to the 1970s, fundamentally altering the structure and function of the wetland systems (Kirkland, 1970).

2.2.2 Development of flood control and drainage schemes in the Lower Manawatu catchment

The development of flood control and drainage infrastructure in the Lower Manawatu catchment transformed the hydrology and ability to utilise land for agricultural use. Early flood protection efforts began in the 1880s utilising simple hand-dug drains in low-lying swampy areas, with the Moutoa Drainage Board being established in 1908 (Wheeler, 2011; Horizons Regional Council, n.d.). The Wellington and Manawatu Railway Company played a crucial role in early drainage development, undertaking engineering surveys of the Makerua Swamp between 1886 and 1894 that demonstrated the practicability of a drainage scheme (Tannock, 1968). The company constructed three main drains - Makerua Drain, Seifert's Drain, and Tokomaru Main Drain - to increase land values, with surveys revealing considerable fall from the centre of the swamp to the Tokomaru Stream and proving that the land could be drained despite a popular belief that the swamp was a "bottomless morass" (cited in Tannock, 1968, p. 130). The arrival of mechanical excavation equipment in the 1920s enabled large-scale drainage of wetlands and swamps, with various river and drainage boards installing stopbanks and greatly improving drainage and flood protection across the plain (Horizons Regional Council, n.d.).

Following the recurrence of destructive floods in the early 1900’s, a comprehensive flood control scheme became critical to utilising the land for agricultural use. A 1945 engineering report documented how the Lower Manawatu plains had been transformed, "from a wet flax and bush covered swamp" to "very high productivity" agricultural lands. However, the report warned that "further intensive development of the Manawatu valley cannot proceed on account of the ever-increasing risk of flooding" (cited in McNeill-Adams, 1968, p. 27). A devastating flood in 1941 confirmed that effective, coordinated flood protection was badly needed, leading to the design of the Lower Manawatu River flood control scheme in 1946 (Horizons Regional Council, n.d.).

A major test for the scheme came during the 1953 flood, which was contained only with army assistance laying sandbags, and made it clear that the proposed sluice gates were needed (Horizons Regional Council, n.d.). The Moutoa Sluice Gates and floodway were subsequently constructed between 1959 and 1962, becoming operational in 1963. The nine gates can divert up to 2,450 cubic metres of water per second into a 10 kilometre floodway, bypassing 30 kilometres of meandering river channel and protecting around 28,000 to 32,000 hectares of agricultural land in the Lower Manawatu catchment (Horizons Regional Council, n.d.; McNeill-Adams, 1968).

The flood control scheme has enabled the development of intensive agriculture in areas that were previously flood prone. However, the engineering modifications also fundamentally altered natural hydrological processes, reflective of other similar schemes across New Zealand which have coincidentally created efficient pathways for nutrient transport from agricultural areas. Approximately 1.77 million hectares of agricultural land are artificially drained across New Zealand, with the highest concentrations in former wetland areas like those found in the Manawatu region (Manderson, 2018).

The Moutoa-Whirokino catchment exemplifies this transformation. The approximately 6,000 hectare catchment, bisected by the Moutoa Floodway, now operates under an extensive drainage scheme managing 77.7 km of drains, 42 floodgates, and six pump stations providing both pumped and gravity drainage for the catchment of predominantly dairy and cropping land (Wheeler, 2011; Horizons Regional Council, 2025b). The scheme has a total asset value of approximately $11 million as of June 2024, with current land use dominated by high producing exotic grassland (92%) and short-rotation cropland (4.4%), characteristic of the intensive pastoral farming in the area (Horizons Regional Council, 2025b; Landcare Research New Zealand Ltd., 2020).

2.2.3 Contemporary water quality challenges

The Manawatu River's contemporary water quality challenges are a product of decades of intensive agricultural development, point source discharges, and hydrological modifications within the catchment. The river receives contaminants from multiple sources including municipal wastewater treatment plants, industrial discharges, urban runoff, and agricultural runoff. Research has documented that 98% of nitrogen loads and 80% of phosphorus loads in the upper catchment are derived from diffuse agricultural sources, with point source contributions mainly coming from municipal wastewater discharges (Roygard et al., 2012; McArthur & Patterson, 2007).

The river's water quality challenges gained national attention following research that documented severe oxygen depletion and indicated severe nutrient enrichment and eutrophication potential. The Manawatu River had the worst readings of dissolved oxygen and primary productivity in comparison to 300 rivers and streams across North America, Europe, Australia and New Zealand at the time (Morgan & Burns, 2009; Clapcott & Young, 2009).

Nutrient enrichment is the key contemporary water quality challenge facing the Manawatu River system - both in the upper and lower stretches (McArthur, 2009). An analysis of long-term monitoring data revealed complex patterns of nitrogen and phosphorus limitation that vary spatially and temporally across the catchment (McArthur et al., 2010). Recent monitoring data from Horizons Regional Council indicates that nutrient concentrations continue to exceed targets across much of the region. Between June 2017 and June 2022, only 32% of river monitoring sites met the Horizons Regional Council Regional Plan (One Plan) targets for Soluble Inorganic Nitrogen (SIN), while just over 22% of sites met targets for Dissolved Reactive Phosphorus (DRP) (Horizons Regional Council, 2025a). Over a recent twenty-year period approximately 45% of monitored sites across the region show improvements in DRP concentrations (Horizons Regional Council, 2025a). However, recent five-year trends at lower Manawatu River catchment monitoring sites show degrading patterns. These results demonstrate the ongoing nature and complexity of nutrient enrichment issues within the region.

Water quality monitoring at key sites along the Manawatu River demonstrates the persistent nature of these challenges. At Whirokino, near the river mouth, the five-year median DRP concentration of 0.022 mg/L places the site in the worst 25% of all monitored sites nationally, with DRP showing a very likely degrading trend (LAWA, n.d.-b). Similarly, at Ōpiki Bridge in the lower catchment, the five-year median DRP concentration of 0.0205 mg/L also places this site in the worst 25% nationally, with a very likely degrading trend (LAWA, n.d.-c). Total nitrogen concentrations across monitored sites consistently fall within the worst 50% of sites nationally, with five-year medians ranging from 0.58 mg/L at Teachers College in Palmerston North to 0.815 mg/L at Ōpiki Bridge (LAWA, n.d.-c; LAWA, n.d.-d).

Despite these persistent challenges, there are encouraging signs of improvement in some water quality indicators across the region. Long-term trend analysis covering twenty years up to June 2022 shows that ammoniacal nitrogen concentrations are likely or very likely improving at almost all (92%) of monitored river sites within the Horizons Region (Horizons Regional Council, 2025a). However, more recent five-year trends at sites within the Manawatu River catchment reveal concerning patterns, with ammoniacal nitrogen showing very likely degrading trends at multiple Manawatu River monitoring sites, including Whirokino, Upper Gorge, Teachers College, Ōpiki Bridge and Hopelands (LAWA, n.d.-b; LAWA, n.d.-c; LAWA, n.d.-d; LAWA, n.d.-e; LAWA, n.d.-f).

The improvements in longer-term and region-wide datasets have been attributed to changes in industry and territorial authority discharge practices, guided by policy frameworks that have phased out dairy shed discharges directly to water and improved wastewater treatment processes. Additionally, over half of monitored river sites across the region show likely or very likely improvements in SIN concentrations, with sites along the Manawatu River including Upper Gorge, Teachers College, Ōpiki Bridge and Hopelands all showing likely improving trends in SIN and nitrate-nitrogen over five-year periods (LAWA, n.d.-e; LAWA, n.d.-d; LAWA, n.d.-c; LAWA, n.d.-f).

The spatial patterns of water quality degradation reflect the intensity of land use throughout the catchment, with conditions generally deteriorating as the river flows through areas of increasingly intensive agriculture. Smaller streams and watercourses draining intensively farmed areas consistently contribute elevated nutrient loads to the main stem, creating cumulative impacts that are evident in downstream monitoring results (McArthur, 2009). The Manawatu Estuary, which receives inputs from the entire catchment, faces significant environmental pressures from this cumulative nutrient loading, where between June 2017 and June 2022 it failed to meet the One Plan targets for SIN and DRP (Horizons Regional Council, 2025a).

2.2.4 Manawatu Estuary significance and pressures

The Manawatu River estuary is one of the most ecologically important estuary systems in the lower North Island and, arguably in New Zealand given its designation as a Wetland of International Importance under the Ramsar Convention in 2005. The estuary is made up of around 600 hectares of diverse habitats including sandflats, estuary waters, intertidal marshes, and coastal dunes (Ramsar Sites Information Service, 2023). Being designated in this way recognises the site as being one of the largest estuaries in the North Island that retains a high degree of naturalness and endemic biodiversity (Ramsar Sites Information Service, 2023).

Approximately 170 bird species have been identified at and are supported by the estuary, making it an incredibly diverse site for birds in New Zealand (Department of

Conservation, n.d.; Manawatu Estuary Trust, 2025). The estuary is also a nationally important site for migratory shorebirds, and a key breeding location for the regional population of wrybill breeds (Ramsar Sites Information Service, 2023), supporting about 1% of the world's wrybill population (Forest and Bird, n.d.).

Alongside the bird populations, the estuary has aquatic ecological significance. A total of 50 fish species either inhabit or migrate to the site, with many of them classified as threatened or rare, such as Galaxiidae and New Zealand longfin eel (Ramsar Sites Information Service, 2023; Manawatu Estuary Trust, 2025). There are up to 30 threatened or nationally critical fish, plant and bird species which can be found at the site, including fairy terns, wrybill, shore plover, longfin eel, giant kokopu, sea primrose and remuremu (half-star) (Forest and Bird, n.d.; Manawatu Estuary Trust, 2025).

However, the estuary faces significant environmental pressures that threaten its ecological integrity and status of international significance. From 2013 to 2023, the numbers of international migrant shorebird declined, particularly of the red knot (Ramsar Sites Information Service, 2023). Water quality degradation in the inflowing river directly affects estuarine habitats through increased nutrient loading, sedimentation, and contamination with pathogens. Excessive nutrient inputs drive eutrophication processes that can lead to algal blooms, oxygen depletion, and shifts in species composition toward more pollution-tolerant organisms (Dudley et al., 2020).

Having established the regional context of the Moutoa-Whirokino catchment within the broader Manawatu River system, this review now examines the specific biogeochemical and hydrological processes controlling nutrient dynamics in artificially drained agricultural systems. Understanding these mechanisms is essential for interpreting nutrient dynamics and developing effective mitigation strategies.

2.3 Nutrient dynamics in intensive agricultural systems

2.3.1 Hydrological pathways in drained agricultural systems

Water is a key transporter of nutrients from agricultural lands to receiving waters. Several of these key water flow pathways include surface runoff, shallow subsurface flow through soil layers, preferential flow through macropores and soil cracks, and direct transport through artificial drainage channels including mole and tile drains, and constructed drains or ditches (Elsayed et al., 2023; Smith et al., 2015). The importance of these pathways varies between hill country areas dominated by surface runoff, and lowland areas with substantial artificial drainage systems, with drained organic soils representing an extreme case where artificial drainage intercepts and rapidly conveys nutrients that would otherwise undergo substantial biogeochemical transformation.

The timing and magnitude of nutrient transport through the various pathways is heavily dependent on seasonal conditions and storm events. Approximately 75% of nitrogen loss from agricultural catchments can occur during autumn and early winter periods of higher rainfall (Petry et al., 2002). During high soil moisture conditions when tiles are flowing, nutrients are more susceptible to move into the tile system, while on excessively dry soils with visible surface cracks, nutrients have direct pathways into drainage systems.

2.3.1.1 Surface runoff and overland flow

Surface runoff occurs when rainfall or irrigation exceeds the infiltration capacity of the soil, and can lead to the transportation over land of dissolved and particulate nutrients directly to surface water (Petry et al., 2002). In agricultural catchments, this pathway is particularly important during storm events when nutrients, especially phosphate and ammonium, are mobilised from riparian areas where soils have been compacted by livestock or farm machinery (Elsayed et al., 2023; Simmonds et al., 2016; Petry et al., 2002). Hill country landscapes with steeper topography and imperfectly drained soils are especially susceptible to this type of runoff, where erosion and runoff can be the main transport mechanism for the loss of sediment and phosphorus (Burkitt et al., 2017; McDowell & Haygarth, 2024; Quinn & Stroud, 2002). However, in well-drained agricultural lands with extensive artificial drainage infrastructure such as the Moutoa-Whirokino catchment, surface runoff may be less of a factor than subsurface pathways, especially where artificial drainage lowers water tables and increases infiltration capacity (Elsayed et al., 2023).

2.3.1.2 Subsurface flows and groundwater-surface water interactions

Subsurface water movement through deep percolation and groundwater-surface water interactions are a critical pathway for dissolved nutrient transport in agricultural landscapes, particularly in artificially drained systems where natural attenuation processes may be bypassed.

Deep percolation is the primary mechanism by which water and dissolved contaminants reach groundwater systems, and involves the downward movement of water below the root zone (Bethune et al., 2008). This process is influenced by soil hydraulic properties, moisture conditions, rainfall intensity, and the presence of preferential flow pathways. In poorly drained agricultural soils, artificial drainage can change these dynamics by intercepting water before it reaches deeper groundwater layers, effectively creating a short-cut in the pathway between the soil profile and underlying groundwater (Bethune et al., 2008).

Groundwater and surface water are interconnected parts of a single hydrological system, with contaminated aquifers affecting surface water quality (Winter et al., 1998; Sophocleous, 2002). In agricultural catchments, groundwater discharge to streams and rivers can be a pathway for dissolved nutrient delivery, particularly during baseflow conditions. The direction and magnitude of these exchanges is controlled by hydraulic gradients, which vary spatially and temporally in response to precipitation, irrigation, and drainage operations (Winter, 1999).

Subsurface hydrologic connectivity- the degree to which landscape positions are linked through subsurface flow- is a key control on nutrient transport at catchment scales (Jencso et al., 2009). In naturally drained systems, subsurface flow paths can be relatively long, allowing time for biogeochemical processes such as denitrification to attenuate nitrate loads. However, artificial drainage changes these pathways, creating more direct connections between agricultural fields and surface waters and reducing residence time available for natural attenuation processes (Schilling & Helmers, 2008).

Dissolved nitrogen movement through subsurface pathways depends on nitrogen speciation and redox conditions. Nitrate, being negatively charged and highly stable in aerobic conditions, is extremely mobile in groundwater and readily leaches through soil profiles (Di & Cameron, 2002). In contrast, ammonium, being positively charged, readily adsorbs to negatively charged clay minerals and organic matter, substantially reducing its mobility (Robertson & Groffman, 2024). The efficiency of nitrogen transport through subsurface pathways therefore depends not only on hydraulic conductivity but also on the oxidation-reduction status of the groundwater, availability of organic carbon as an electron donor for denitrifying bacteria, and residence time within the subsurface environment (Robertson & Groffman, 2024; Wolters et al., 2022). In well-drained agricultural soils with aerobic groundwater conditions and rapid drainage, nitrate can move through the subsurface with minimal attenuation, making groundwater-surface water discharge a highly efficient pathway for nitrogen delivery to surface waters (Schilling et al., 2020). Conversely, under reducing conditions characteristic of poorly drained or organic soils with elevated water tables and high carbon content, alternative nitrogen species such as ammonium may persist or dominate (Han et al., 2023), creating patterns distinct from the nitrate-dominated losses typical of well-drained mineral agricultural soils.

2.3.1.3 Tile drainage systems

Artificial drainage networks alter natural hydrological processes and create new pathways for nutrient transport from land to water (Carlson et al., 2018; Petry et al., 2002). Approximately 1.77 million hectares of agricultural land are artificially drained across New Zealand, with the highest concentrations in former wetland areas (Manderson, 2018).

Artificial subsurface drainage alters hydrological pathways, creating rapid conduits that bypass natural attenuation processes. Schilling and Helmers (2008) found that tile-drained watersheds exhibit 35%-65% higher baseflow indices compared to naturally drained systems, with extensive tile drainage reducing mean groundwater travel times from approximately 30 years to less than 10 years (Schilling et al., 2020). This means dissolved contaminants move rapidly from agricultural fields to surface waters with minimal opportunity for natural attenuation.

Tile drains can transport up to 82% of nitrate, 77% of total phosphorus, and 67% of dissolved reactive phosphorus loads from agricultural fields to surface water (Van Esbroeck et al., 2016; Williams et al., 2015). In New Zealand pastoral systems, mole-pipe drainage is commonly used in conjunction with tile drainage, creating cracks and fissures in the upper soil profile that drain into buried pipes (Monaghan et al., 2016). Studies of intensively grazed dairy pastures have demonstrated that subsurface drainage is the dominant pathway for nitrogen loss, accounting for approximately 80% of total soluble nitrogen discharged, with mean annual losses ranging from 30-56 kg N ha^{-1}

(Monaghan et al., 2016; Monaghan et al., 2000). Surface runoff proved more important for phosphorus and sediment movement, contributing 68% of dissolved reactive phosphorus losses (Monaghan et al., 2016).

Approximately 75% of nitrogen loss from agricultural catchments occurs during autumn and early winter periods of higher rainfall (Petry et al., 2002), with nitrate concentrations in drainage typically highest in late autumn when drainage commences, reflective of the accumulation of mineral nitrogen from animal urine deposited during summer and autumn (Monaghan et al., 2016). Preferential flow pathways through macropores, root channels, and structural cracks can rapidly transport water and nutrients to drainage systems, with nutrient transport controlled by connectivity between surface soils and drainage networks (Jarvis, 2007). Open surface drains can also accumulate and directly transport diffuse nitrogen pollution, with Smith et al. (2017) documenting significantly higher nitrate-nitrogen concentrations in drains (ranging from <0.25-6.59 mg/L) compared to adjacent streams.

2.3.2 Nutrient dynamics in drained organic or peat soils

Drained organic or peat soils are one of the most challenging agricultural soils for nutrient management due to their unique properties. The conversion from anaerobic carbon sinks to aerobic or fluctuating redox environments through artificial drainage alters nutrient cycling processes fundamentally and leads to accelerated organic matter decomposition and nutrient mobilisation (Tiemeyer et al., 2007; Grønlund et al., 2008). These soils have particularly high organic carbon content, limited phosphorus sorption capacity, and nitrogen transformation pathways that are different from patterns observed in mineral agricultural soils. Understanding these characteristics is important for interpreting nutrient dynamics found in intensively drained pastoral catchments such as Moutoa-Whirokino.

2.3.2.1 Physical and chemical transformations following drainage

Drainage alters organic and peat soil properties through lowering water tables and creating oxidising conditions. The increase in oxygen availability accelerates mineralisation of organic matter and alters soil organic carbon content through both drainage and tillage of soils (Leifeld et al., 2019). These transformations vary spatially, for example Tiemeyer et al. (2007) found soil organic carbon contents ranging from 0.8% to 40.9% and total nitrogen contents from 0.08% to 2.87% within a single degraded peatland catchment under intensive grassland use in North-Eastern Germany. The impact of drainage creates alternating aerobic and anaerobic conditions that influence biogeochemical processes across various depths of the soil profile (Kassim & Yaacob, 2019).

2.3.2.2 Nitrogen dynamics

Increased nitrogen mineralisation is characteristic of drained peat or organic soils due to the aerobic decomposition of organic matter. This process is particularly prevalent in soils used for intensive agriculture, with soils reacting to increased temperatures through more rapid organic matter decomposition (Becher et al., 2023). The microbial decomposition of organic nitrogen in soil organic matter releases plant-available inorganic nitrogen, particularly ammonium, when microbial carbon and energy requirements are met (Hart et al., 1994; Robertson & Groffman, 2024). Mineralisation rates depend on environmental factors such as temperature, soil moisture, pH, and the quantity and quality of organic matter inputs (Robertson & Groffman, 2024).

In drained peat soils, nitrogen transformations are governed by complex microbial processes that control nitrogen availability and fate in soil-water systems (Robertson & Groffman, 2024). Under aerobic conditions created by drainage, nitrification converts the relatively immobile ammonium ion to highly mobile and soluble nitrate, which is susceptible to leaching (Beeckman et al., 2018). This aerobic process is performed by ammonia-oxidising bacteria and archaea, followed by nitrite-oxidising bacteria that complete the conversion to nitrate (Robertson & Groffman, 2024). However, nitrification is an aerobic process and therefore requires dissolved oxygen. In waterlogged or poorly aerated zones characteristic of many drained organic soils, oxygen availability becomes the limiting factor controlling whether nitrification can proceed. The rate of nitrification is mainly controlled by ammonium availability in aerobic environments, and is dependant on oxygen availability, temperature, pH, and moisture conditions (Robertson & Groffman, 2024). In organic soils (both drained and undrained), limited oxygen diffusion can create conditions where mineralisation produces ammonium but nitrification cannot convert it to nitrate, resulting in ammonium accumulation rather than the nitrate-dominated losses typical of well-drained mineral agricultural soils.

Conversely, in anaerobic or oxygen-poor microsites within the soil, denitrification can remove reactive nitrogen from the system through microbial reduction of nitrate to gaseous forms (Robertson & Groffman, 2024). This process begins when oxygen concentrations fall below approximately 2.0 mg/L and requires nitrate availability and carbon as an electron donor (Wolters et al., 2022; Robertson & Groffman, 2024). Under strongly reducing conditions in carbon-rich, nitrate-limited environments, dissimilatory nitrate reduction to ammonium (DNRA) may dominate over denitrification, retaining nitrogen within the soil as ammonium rather than removing it as gas (Silver et al., 2001; Robertson & Groffman, 2024). This pathway can be significant in pastoral organic soils and is examined in detail in the following section.

Nitrogen losses from drained peatlands and organic soils can be substantial, though the speciation of nitrogen losses varies considerably with soil redox status and drainage design. Tiemeyer et al. (2007) documented nitrate-nitrogen concentrations reaching up to 30.1 mg/L in drainage ditches, with annual losses of up to 30 kg N ha^{-1} in degraded peatland under intensive grass land use in an experimental field in North-Eastern Germany. Groundwater nitrate-nitrogen concentrations ranged from 0 to 65.4 mg/L, demonstrating high spatial variability and making single-point measurements insufficient to gain a representative understanding of nutrient patterns (Tiemeyer et al., 2007). Research has shown that drained peat soils are significant diffuse sources of nitrogen to ground and surface water. Vassiljev and Blinova (2012) found nitrogen runoff from drained peat soils in Estonia was 1.5-fold higher than from other agricultural land, producing 10-16 kg/ha per year of nitrogen. Importantly, Van Beek et al. (2007) reported that in intensively managed Dutch dairy grasslands on peat soils, a major portion of nitrogen leaching to surface water originated from deeper, reduced peat layers rather than fertiliser applications, highlighting the complex nutrient dynamics through the soil profile and the importance of mineralisation from organic matter decomposition as a nitrogen source. This finding contrasts with mineral soils where contemporary fertiliser and urine inputs dominate nitrogen exports, and suggests that drainage-enhanced mineralisation in organic soil profiles creates persistent nitrogen losses even under improved nutrient management at the soil surface.

2.3.2.3 Dissimilatory nitrate reduction to ammonium (DNRA)

Dissimilatory nitrate reduction to ammonium (DNRA) is an important yet underappreciated nitrogen transformation pathway in pastoral and agricultural systems, particularly those with high organic carbon content. Unlike denitrification, which removes nitrogen from the system by converting nitrate-N to gaseous N_2 or N_2O, DNRA reduces nitrate-N to ammonia-N, retaining nitrogen within the soil-water system. The difference of nitrogen conservation versus nitrogen loss has implications for understanding nitrogen fate in drained organic soils and interpreting patterns of elevated ammonia-N with unexpectedly low nitrate-N in drainage water.

Recent research has broadened understanding of DNRA's role in pastoral systems. A review by Xin et al. (2019) found DNRA rates observed across current research were highly variable, ranging between 0.00 and 33.34 μg N g^{-1} d^{-1}, or 3% to ~99% of the soil nitrate-N removal production. The highest rate was detected in a dairy grassland soil in Australia at 80% water-filled pore space, with DNRA exceeding denitrification rates across all sites upon rewetting of soils (Friedl et al., 2018). A key finding in Friedl et al. (2018) was that soil redox potential, not carbon to nitrate-N ratio, was the fundamental factor for nitrate-N partitioning between denitrification and DNRA in carbon-rich pasture soils. This challenges traditional assumptions that high C:N ratios alone drive DNRA, demonstrating instead that in high-carbon environments like pastoral soils, redox potential becomes the most important variable associated with whether nitrate-N is reduced to ammonia-N (DNRA) or gaseous products (denitrification).

The dominance of DNRA over denitrification in agricultural systems is not limited to pastoral farming systems. Pandey et al. (2019) found that in long-term low-nitrogen-fertilised rice paddies, up to eight times the amount of soil nitrate-N was ammonified through DNRA versus denitrification rates. Additionally, Pan et al. (2020) documented that in fertilised farmland soils, DNRA contributed 84.44 ± 14.40% of dissimilatory nitrate reduction, which showed a clear dominance over denitrification. A global meta-analysis by Cheng et al. (2022) synthesised 231 observations from 85 studies, revealing global mean DNRA rates of 0.31 ± 0.05 mg N kg^{-1} day^{-1}, with grasslands showing elevated rates of 0.52 ± 0.15 mg N kg^{-1} day^{-1}. In this analysis precipitation was identified as the main global stimulator for soil DNRA, highlighting the importance of wet-dry cycles and oxygen limitation in activating this pathway (Cheng et al., 2022).

In drained pastoral organic soils, an understanding of the effect of DRNA on nitrogen dynamics is important. In systems where surface or shallow subsurface nitrification produces nitrate, this nitrate may leach downward into deeper, more reduced zones with high organic carbon content. Rather than being denitrified to N_2 gas (nitrogen loss), DNRA converts this nitrate back to ammonium. This creates a system of nitrogen retention in drained agricultural soils where nitrogen remains in the system in the form of ammonium rather than nitrate-N. For the Moutoa-Whirokino catchment and similar intensively drained pastoral organic soils, DNRA provides a potential explanation for a pattern of elevated ammonium with low nitrate.

2.3.2.4 Nitrogen speciation in drained organic soils

Integrating the processes of mineralisation, nitrification, denitrification, and DNRA provides a framework for understanding when and where ammonium versus nitrate dominates in drained organic soils. This framework is essential for interpreting the unusual nitrogen speciation patterns observed in catchments such as Moutoa-Whirokino.

In drained organic soils, nitrogen speciation is controlled by spatial and temporal redox conditions associated with microbial transformations in the system. The controlling variables are redox conditions and dissolved oxygen availability, which determine whether conditions permit aerobic nitrification or anaerobic nitrate reduction pathways. Where drainage lowers water tables and oxygen can penetrate, conditions permit nitrification. Ammonia-oxidising bacteria and archaea can convert ammonium (produced via mineralisation) to nitrate-N, creating mobile nitrate that is susceptible to leaching. However, this aerobic zone is typically thin in organic soils even when drained, because rapid oxygen consumption by decomposing organic matter limits oxygen penetration depth (Boonman et al., 2024). Below the aerobic surface, oxygen becomes limiting and redox values decline. Nitrification is inhibited in this zone, so ammonium produced through the process of mineralisation accumulates rather than being converted to nitrate. Additionally, any nitrate leaching from the surface layer or produced in aerobic microsites enters an environment where microbial nitrate reduction through denitrification and DNRA become more prevalent. In deeper peat layers with elevated water tables and high organic carbon content, strongly reducing conditions favour DNRA over denitrification (Friedl et al., 2018; Pandey et al., 2019).

2.3.2.5 Phosphorus mobilisation

Phosphorus dynamics in drained peat or organic soils differ considerably from mineral soils due to severely limited anion storage capacity (ASC), typically below 20% (McDowell & Monaghan, 2015). This results in minimal phosphorus retention and high P leaching susceptibility, with dissolved phosphorus losses ranging from 1.7 kg ha^{-1} from high ASC soils to 87 kg P ha^{-1} from low ASC organic soils (McDowell & Monaghan, 2015), representing a loss of up to 89% of fertiliser phosphorus applied. The low anion storage capacity of organic soils means phosphorus mobilisation and transport is mainly affected by soil storage capacity and transport processes rather than strong soil binding (Gray et al., 2024).

In organic soils, both surface runoff and subsurface drainage serve as important phosphorus transport pathways. Surface runoff transports both dissolved and particulate phosphorus, with losses occurring particularly through attachment with eroded soil particles (Simmonds et al., 2016). The contribution of subsurface drainage as a phosphorus transport pathway is particularly significant in artificially drained organic soil systems. When soil macropores are present in greater proportions, phosphorus losses can increase through preferential flow to subsurface drainage (King et al., 2015). Tile drain systems can transport substantial nutrient loads, both through leaching phosphorus and following effluent applications (Stamm et al., 1998). Due to the hydraulic conductivity and low ASC of organic soils, significant phosphorus loss can occur through drainage as the key pathway, regardless of moisture conditions (Simmonds et al., 2016).

Three primary forms of phosphorus are measured in drainage water: dissolved reactive P (DRP), dissolved unreactive P (DURP), and particulate P (PP). DRP is the most bioavailable form, DURP consists of organic phosphorus forms with relatively high mobility, and particulate P can be both reactive and unreactive and is attached to or part of a solid (Gray et al., 2024). The source of phosphorus affects transport, with farm dairy effluent often producing greater phosphorus losses compared to fertilisers due to its high proportion of phosphorus-rich particles and increased leaching of DURP and PP forms, which have lower adsorption capacity to soil phosphorus sorption sites (Gray et al., 2024).

Documented phosphorus concentrations in groundwater from organic soils demonstrate the potential scale of mobilisation. Sapek et al. (2021) found phosphate-phosphorus concentrations in groundwater from organic soil layers in a Polish meadow complex on peat soils reaching up to 22 mg/L, with approximately 46% of water samples characterised by poor chemical status. In this study, phosphate-phosphorus and ammonia-nitrogen concentrations were significantly positively correlated, indicating that peat mineralisation processes are a common cause for their mobilisation (Sapek et al., 2021). This correlation suggests that the same drainage-enhanced decomposition producing elevated ammonium through mineralisation (as discussed in Section 2.3.2.2) also releases organic phosphorus. Kassim and Yaacob (2019) found that available phosphorus in Malaysian tropical peatlands was at its highest concentration when groundwater fluctuated between 0-40 cm depth, demonstrating the importance of the aerobic-anaerobic interface zone. The variation in phosphorus availability under different groundwater depths shows the complexity of interplay between reduction-oxidation processes and phosphorus storage or release in organic soils. Under anaerobic conditions, reduction of iron and manganese oxyhydroxides releases previously bound soluble phosphorus (Scalenghe et al., 2002).

The timing of phosphorus applications influences its loss potential from organic soils. For up to 60 days following fertiliser application, drainage events can result in substantial phosphorus losses through preferential flow in soil macropores (Gray et al., 2024). Long-term intensive agricultural use can also result in substantial phosphorus accumulation within soil profiles because inputs exceed outputs and sorption capacity of the soil. Vermaat and Hellmann (2010) found Dutch peat soils under agricultural use containing 1400 kg P ha^{-1} in the upper 50 cm compared to 700 kg P ha^{-1} in nature reserves after over 50 years of dairy farming. This phosphorus can become a long-term source even if contemporary applications are reduced, with phosphorus continuing to be released through mineralisation and desorption processes. The combination of low ASC and potential legacy phosphorus accumulation makes drained organic soils among the highest-risk landscapes for phosphorus export to receiving waters.

2.3.2.6 Temporal patterns and other relationships

Nutrient losses from drained peat and organic soils can show distinct seasonal patterns. Van Beek et al. (2007) found that in intensively managed Dutch peat grasslands, approximately 90% of nutrient leaching occurred during winter months (November through March), meaning that nutrient mobilisation and plant demand were not aligned temporally. Other research has identified five primary factors which explain up to 80% of data variance for nutrient levels within groundwater: topography and soil chemical properties; biogeochemical processes; drainage effects; climatic effects; and dilution effects (Tiemeyer et al., 2007).

2.3.3 Groundwater effects on nutrient dynamics

Groundwater systems in agricultural landscapes are critical pathways for nutrient transport, and they influence the timing, magnitude, and speciation of nutrient delivery to surface water systems. In artificially drained agricultural lands, subsurface drainage creates a direct pathway for shallow groundwater-derived nutrients to enter surface waters, bypassing natural attenuation processes that might occur in deeper groundwater layers or during slower subsurface flow through riparian margins (Schilling & Helmers, 2008; Schilling et al., 2020).

2.3.3.1 Groundwater level effects

Groundwater level fluctuations are a primary cause of nutrient mobilisation and transport. Kassim and Yaacob (2019) demonstrated in Malaysian tropical peatlands that different nutrients respond distinctly to water table changes, with both nitrate-nitrogen and ammonia-nitrogen concentrations maximised when water tables were maintained at 40 cm below the soil surface. However, Sapek et al. (2021) found in a Polish meadow complex on peat soils that groundwater level depths in 10 cm intervals largely determined nutrient concentrations, with increasing phosphate-phosphorus and ammonia-nitrogen concentrations as groundwater levels decreased.

Sapek et al. (2007) identified groundwater table fluctuations as the primary factor controlling phosphate and ammonium concentrations in Polish peat soil systems used for agricultural grasslands. Lowering groundwater levels during late summer mobilised phosphate and ammonium in groundwater, with these nutrients then mobilising into deeper groundwater layers and drainage channels. This research established that late summer and early autumn are the primary periods when significant phosphate and ammonium leaching occurs from peat soils.

2.3.3.2 Redox conditions and nutrient transformations

The reduction/oxidation (redox) state of groundwater significantly effects nutrient speciation and mobility. Reducing conditions develop through oxygen consumption during microbial decomposition of organic matter, with denitrification beginning when oxygen concentrations fall below 2.0 mg/L on average (Wolters et al., 2022).

Han et al. (2023) found that nitrogen transformation processes were distinct along redox gradients in high-nitrogen groundwater systems in an intensive agricultural area in northern China. Under oxidising conditions, nitrification converts ammonium to nitrate, while in reducing environments, anaerobic ammonium oxidation (anammox) becomes dominant, converting ammonium to nitrogen gas (Mulder et al., 1995). This research revealed the scale of difference in relation to ammonium production or consumption ratios across redox zones, being 0.7 in oxidising zones, 6.9 in moderately reducing zones, and 51 in strongly reducing zones, indicating that ammonium accumulates under strongly reducing conditions. This shift to ammonium production under strongly reducing conditions explains persistent high ammonium concentrations in anaerobic groundwater, even in systems receiving nitrate inputs from overlying aerobic zones.

In peat and organic soils, the high organic carbon content rapidly depletes dissolved oxygen in saturated layers, creating strongly reducing conditions that alter nutrient cycling processes. The abundant organic matter provides both a source of electron donors through decomposition and an electron-accepting capacity that buffers redox conditions (Guth et al., 2023). Boonman et al. (2024) deployed over 150 redox sensors across Dutch agricultural peatlands, demonstrating that even with drainage, deep peat layers remain oxygen-limited due to restricted oxygen. Therefore, shallow water tables limit oxygen penetration depth through diffusion constraints, meaning even drained peat soils often have strongly reducing conditions below a thin aerobic surface layer. Under these anaerobic conditions, microbial decomposition shifts from aerobic respiration to sequential use of alternative electron acceptors including nitrate, manganese oxides, iron oxides, and sulfate, fundamentally changing the pathways and rates of nutrient transformations. The dominance of reducing conditions in saturated peat layers means that nitrogen cycling is characterized by limited nitrification and enhanced denitrification potential, while phosphorus becomes increasingly mobile through reductive dissolution of iron complexes. In peat and organic soils, the high organic carbon content reduces dissolved oxygen in saturated layers even when water tables are lowered, creating persistently reducing conditions that inhibit nitrification while favouring nitrogen-retention pathways such as DNRA.

Phosphorus dynamics are also sensitive to redox conditions where, under anaerobic conditions, reduction of iron and manganese oxyhydroxides releases previously bound soluble phosphorus (Scalenghe et al., 2002). In drained peat and organic soils with inherently low iron content and limited sorption capacity, this redox-sensitive phosphorus mobilisation is particularly pronounced due to the combination of high organic carbon content promoting reducing conditions and legacy phosphorus accumulation from decades of agricultural use. Herndon et al. (2019) demonstrated that iron oxyhydroxides sequester approximately 46% of inorganic phosphorus in organic soils, representing a substantial pool vulnerable to reductive dissolution during periods of water table rise and oxygen depletion. Van Beek et al. (2004) documented that in Dutch dairy grassland on peat soils, 33-82% of phosphorus loading originated from nutrient-rich deeper peat layers undergoing mineralisation rather than surface fertiliser applications, which shows that internal phosphorus mobilisation through redox-associated processes can be the dominant source in agricultural peat systems. The fluctuating water tables characteristic of artificially drained peatlands create repeated cycles of oxidation and reduction that can deplete iron-bound phosphorus pools over time, with phosphorus release rates in rewetted agricultural peat potentially orders of magnitude higher than in natural peatlands (Zak et al., 2008).

2.4 Edge-of-field drainage nutrient loss mitigations

Edge-of-field mitigation strategies are important tools for reducing nutrient losses from agricultural drainage systems before they reach receiving waters. These approaches vary in complexity, cost, and effectiveness, but all aim to intercept and treat drainage water through biological, physical, or chemical processes. The effectiveness of edge-of-field mitigations depends on the speciation of nutrients in drainage water. Systems designed for nitrate-N removal via denitrification perform differently when treating ammonium-dominated drainage, and phosphorus removal rates vary between particulate and dissolved forms.

2.4.1 Controlled drainage

Controlled drainage involves installing water level control structures within artificial drainage systems to manage water table depths and reduce nutrient loading. These systems utilise adjustable outlets at drainage discharge points, enabling water table management during periods when drainage is not essential for crop growth, while maintaining agricultural productivity through retention of soil moisture (Tanner et al., 2020).

Controlled drainage can reduce nutrient loads compared to conventional free drainage systems. The elevated water table increases residence time for enhanced denitrification in saturated soil conditions while improving plant access to nutrients, leading to a reduction in nutrient discharge from drainage water. However, reductions in nutrient loads are mainly due to a reduction of drain flows rather than substantial nutrient concentration reductions (Tanner et al., 2020).

The practice is most effective on flat terrain where drainage systems can be controlled from a single point. Many existing tile-drained systems in New Zealand are not well-suited for retrofitting controlled drainage infrastructure, however the 'downstream' open drainage channels likely are. The timing of drainage control is important for optimising nutrient reduction and agricultural productivity. These systems must allow sufficient drainage to occur during key crop growth periods while maximising water retention during periods of low crop water demand.

Controlled drainage has been studied in agricultural peatlands internationally, particularly in North America and Northern Europe. Tähtikarhu et al. (2025) analysed controlled drainage effects in Finnish agricultural peatlands with varying peat thickness

(0.4-1.2 m), and found that controlled drainage reduced drain discharge by up to 862 mm over an 18-month period. However, the corresponding increase in groundwater levels was modest (mean difference 0.01-0.21 m depending on peat thickness), indicating that controlled drainage changes flow routes with impacts dependent on peat thickness and hydrogeological connections.

Pham et al. (2023) monitored nutrient leaching from a controlled drainage cultivated peatland in Ruukki, Finland over a three-year period, where they found nitrogen leaching of 25 kg N ha^{-1} $year^{-1}$ and phosphorus loading of 0.30 kg P ha^{-1} (0.20-0.43 kg P ha^{-1}), which was low compared to average cultivated soils in Finland. The study found that nitrogen leaching strongly decreased during periods of thick grass cover, emphasising the importance of maintaining vegetation cover to minimise nitrate leaching, even during winter periods in cultivated peatlands. Significantly, 13% of total nitrogen losses and 50% of total phosphorus losses occurred via surface runoff, highlighting the number of pathways by which nutrients can be lost from these systems.

Research from the Holland Marsh in Ontario, Canada (a drained peatland under intensive horticultural production) demonstrated that controlled drainage systems can reduce nutrient export compared to pump drainage systems. Grenon et al. (2023) found that nitrogen concentrations under controlled drainage were affected by the interaction of drainage management and timing of discharge events, while phosphorus concentrations remained influenced by fertiliser application timing. The effectiveness of controlled drainage as a nutrient mitigation strategy in peat soils is complicated by potential trade-offs, for example enhanced denitrification under raised water tables can reduce nitrate losses but may increase nitrous oxide emissions, and phosphorus mobilisation risks exist under prolonged anaerobic conditions where iron-phosphorus complexes undergo reductive dissolution. These trade-offs require careful management of outlet elevations throughout the year, typically lowering outlets during critical field operation periods while raising them during non-growing seasons to maximise water retention and nutrient attenuation without compromising agricultural use or increasing greenhouse gas or phosphorus release.

2.4.2 Vegetated drainage ditches

Vegetated drainage ditches are similar to conventional agricultural drainage channels, however they also allow water quality treatment. These ditches use vegetation to enhance physical filtration, nutrient uptake, sedimentation, and microbial transformations (Tanner et al., 2020). A global meta-analysis of 115 datasets from 66 studies across a range of agricultural regions showed that ecological ditches achieved Total Nitrogen (TN) and Total Phosphorus (TP) removal rates of 44.1% and 43.1%, respectively (Bai et al., 2025). The presence of vegetation enhances nutrient removal, and research indicates that vegetation promotes higher rates of denitrification and attenuation of various contaminants compared to unvegetated channels (Kumwimba et al., 2018).

Other design considerations include having an appropriate channel width to allow vegetation growth while maintaining drain capacity, planting correct species for climate, regular harvesting of plant biomass, and the strategic placement of flow control structures such as weirs to increase residence time. The optimal hydraulic retention time (HRT) for TN and TP retention was identified as 24-48 hours (Bai et al., 2025), however this must be balanced with flood control and drainage management requirements during storm events (Kumwimba et al., 2018).

2.4.3 Treatment ponds and chemical enhancement

Treatment ponds use sedimentation and biological processes to remove nutrients from agricultural drainage water. Performance varies considerably, with studies showing total nitrogen removal of 36% and total phosphorus removal of 8% in very eutrophic pond systems (Brunet et al., 2021). Nitrate-nitrogen (NO_3-N) removal can be substantially higher (up to 64%) than total nitrogen removal due to the removal of particulate nitrogen as algal biomass and dissolved organic nitrogen (Brunet et al., 2021).

Chemical enhancement using coagulation can significantly improve treatment effectiveness of these ponds. Both aluminium and iron-based coagulation treatments removed over 70% dissolved organic carbon (DOC) from agricultural drainage waters in California's Central Valley, USA (Bachand et al., 2019a).

2.4.4 Constructed wetlands

Constructed wetlands utilise biological, physical, and chemical processes for nutrient removal from agricultural drainage water. Surface-flow constructed wetlands are the most appropriate type for treating agricultural runoff because of their simplicity, ability to withstand highly variable flow conditions, and capacity to deal with sediment loads (Tanner et al., 2020). These systems operate through plant uptake, microbial processing, and sedimentation.

Appropriately sized constructed wetlands are able to reduce sediment, nitrogen, particulate phosphorus and faecal contamination loads (Tanner et al., 2020). Performance of these systems depends primarily on hydraulic residence time, substrate composition, and loading rates. Nitrogen removal occurs through denitrification under anaerobic conditions, while phosphorus removal depends on physical and chemical processes. Coagulation-treated wetlands in California's Central Valley removed 245-349 g/m^2/year DOC, whereas control wetlands produced 51 g/m^2/year (Bachand et al., 2019b).

As a nutrient mitigation tool, the performance of constructed wetlands is influenced by the wetland size relative to catchment area and hydraulic characteristics. Kovacic et al. (2000) monitored three treatment wetlands (0.3 to 0.8 ha in surface area) intercepting subsurface tile drainage from maize and soybean cropland in Illinois over a three-year period. The wetlands received 4,639 kg total nitrogen (96% as NO_3-N) and removed 1,697 kg N (37% of inputs). Phosphorus removal was substantially higher, with wetlands receiving 79 kg total phosphorus and removing 44 kg P (56% of inputs). Lemke et al. (2022) conducted a 12-year study in the midwestern United States on treatment wetlands of varying sizes (being 3%, 6%, and 9% of tile-drained area) intercepting tile drainage from corn and soybean fields on poorly drained soils, and found that small edge-of-field wetlands effectively treated excess NO_3-N and dissolved phosphorus losses from tiles, with annual reductions remaining high throughout the study period.

Phosphorus removal from constructed wetlands is more challenging and uncertain. The NIWA guidelines report total phosphorus removal of 26% (1% wetland area to catchment area), 35% (2%), 40% (3%), and 48% (5%), but warn that these values are not applicable to predominantly dissolved forms of P, but mainly to wetlands that receive both dissolved-P and particulate-P (Tanner et al., 2021). This is an important distinction to make, that wetlands remove particulate P through sedimentation, but dissolved reactive phosphorus (DRP) removal is poor and highly variable, particularly when wetlands are constructed on phosphorus-rich agricultural soils that can become DRP sources rather than sinks. Research has documented that wetlands can transition from net phosphorus sinks to net sources over time as sediment sorption capacity becomes saturated (Land et al., 2016), a process occurring more rapidly when wetlands are constructed on former agricultural land with elevated legacy phosphorus.

Kynkäänniemi et al. (2013) evaluated a newly constructed wetland treating a 26-hectare agricultural catchment with clay soil in Sweden, and found relatively high retention of 36% for total phosphorus in the first year. For wetlands treating drainage from drained organic soils with inherently low phosphorus sorption capacity and elevated legacy phosphorus, DRP export from the wetland itself represents a significant risk unless special substrates are incorporated into the design. Research related to treatment wetlands established on previously drained peatlands in Finland give insight into peat-based constructed wetland performance. Vymazal (2007) found in a global review of constructed wetland performance that removal of total phosphorus varied between 40% and 60% across all types of constructed wetlands, depending on wetland type and inflow loading, noting that phosphorus removal remained low unless special substrates with high sorption capacity are used.

The NIWA constructed wetland guidelines and supporting international research focus primarily on denitrification as the main nitrogen removal mechanism, requiring nitrate as the key nitrogen species. For drainage water dominated by ammonium rather than nitrate (such as preliminary observations from Moutoa-Whirokino) conventional single-stage horizontal-flow wetlands are not ideal treatment options because they cannot provide the sequential aerobic-anaerobic conditions needed for ammonium removal. Effective ammonium treatment requires a two-stage process, where aerobic nitrification ammonia-N to nitrate-N in shallow, well-mixed oxygenated zones, followed by anaerobic denitrification converting nitrate-N to N_2 in deeper, vegetated zones (Toet et al., 2005). Municipal wastewater treatment literature indicates that hydraulic retention times of 4-13 days are required for comprehensive ammonium treatment (4 days minimum for summer conditions, up to 13 days for winter conditions), which is much longer than the 2-3 days required for effective nitrate removal (Toet et al., 2005). This would require much larger wetland areas in order to achieve equivalent total nitrogen removal when treating ammonium versus nitrate-dominated drainage.

2.5 Conclusions

This literature review has discussed the relationships between intensive agriculture and water quality degradation in New Zealand, with the Manawatu River catchment exemplifying these national challenges. New Zealand's rapid agricultural intensification, as found in figures such as the 71% increase in dairy cattle numbers and 629% increase in nitrogen fertiliser use between 1990-2023, has altered nutrient cycling processes and created substantial environmental pressures on freshwater systems.

The transformation of the lower Manawatu River catchment from its pre-European wetland dominated landscape to intensive agriculture shines a light on the broader national pattern of land use change. The drainage of thousands of hectares of flax swamps and the creation of large flood control schemes in Lower Manawatu catchment has not only enabled agricultural development, but at the same time has created pathways for nutrient transport from land to water in Manawatu Estuary (a Wetland of International Importance under the Ramsar Convention). This hydrological modification is reflective of the 1.77 million hectares of artificially drained agricultural land across New Zealand, with the highest concentrations occurring in former wetland areas.

Water quality data has increasingly revealed the scale and severity of nutrient enrichment issues nationally, with 69% of New Zealand's river length having nitrogen concentrations at risk of environmental impairment and 63% having phosphorus concentrations above natural ranges. The Manawatu River system, with 98% of nitrogen loads and 80% of phosphorus loads in the upper catchment derived from diffuse agricultural sources, exemplifies how intensive land use drives water quality degradation. Importantly, recent water quality monitoring reveals contrasting temporal trends. While there has been long-term (20-year) improvements in ammoniacal nitrogen at 92% of the regional sites, the very likely degrading five-year trends at multiple lower Manawatu River sites (including Whirokino) highlights the complexity of nutrient dynamics and suggests that improvements in point-source controls may be offset by ongoing challenges with diffuse agricultural losses.

The analysis of nutrient dynamics in intensive agricultural systems shows the complexity of transport pathways and transformation processes. Artificial drainage networks, particularly tile drain systems are highly efficient pathways for nutrient movement, transporting up to 82% of nitrate, 77% of total phosphorus, and 67% of dissolved reactive phosphorus loads from agricultural areas to surface waters. Drained organic and peat soils are particularly challenging for nutrient management, with limited anion storage capacity meaning minimal phosphorus retention and substantial leaching potential from these soils. Groundwater dynamics are another important component of nutrient transport systems. Water table fluctuations are primary drivers of nutrient mobilisation, while the redox conditions within groundwater systems effect nutrient types and transformation processes.

The examination of edge-of-field mitigation strategies has shown both the potential and the limitations in addressing nutrient losses from intensive agricultural systems. While technologies such as controlled drainage, vegetated ditches, treatment ponds, and constructed wetlands can be viable nutrient removal technologies, their effectiveness depends on design parameters, hydraulic residence times, and site-specific conditions. Critically, constructed wetlands designed for nitrate removal via denitrification require substantially different design for ammonium-dominated drainage. This has important implications for mitigation planning in systems exhibiting elevated ammonium rather than nitrate.

Despite the extensive body of international research on nutrient dynamics in drained organic and peat soils, critical knowledge gaps remain regarding these systems within the New Zealand context, particularly in the lower Manawatu River catchment. While international studies have predominantly focussed on nitrate-nitrogen losses from drained peatlands in Northern Europe and North America, the specific biogeochemical conditions and nutrient transformation processes occurring in intensively drained organic soils under New Zealand's pastoral dairy systems remain poorly understood. The Manawatu region, with a predominance of artificially drained former wetlands now supporting intensive dairy farming, presents a unique combination of soil types, climatic conditions, and land management practices that may result in nutrient dynamics distinct from those documented internationally. Furthermore, having the Manawatu Estuary (designated as a Wetland of International Importance under the Ramsar Convention) as a receiving environment of the Manawatu River creates an urgent need to understand and mitigate nutrient losses from these drained organic soil systems to protect this internationally significant ecological resource.

The Moutoa-Whirokino catchment presents a valuable opportunity to address these knowledge gaps, with preliminary monitoring revealing elevated ammonia-nitrogen and dissolved reactive phosphorus concentrations, alongside unexpectedly low nitrate-nitrogen levels in both surface drains and shallow groundwater. This pattern of nitrogen speciation differs from the nitrate-dominated losses reported in international literature on drained peatlands, suggesting that site-specific redox conditions, organic matter characteristics, drainage infrastructure design, and soil type variability may be creating unique nutrient transformation pathways within this system. Understanding the origins of elevated ammonium, the hydrological and biogeochemical pathways facilitating nutrient transport from different soil types to surface waters, and the interaction between shallow groundwater and the extensive artificial drainage network will be essential for developing effective, practical mitigation strategies. This research therefore addresses these critical gaps by investigating nutrient loss pathways and potential mitigation options specific to the Moutoa-Whirokino catchment, providing farmers and catchment managers with locally relevant information essential for improving water quality outcomes for the lower Manawatu River and its ecologically important estuary.

Chapter 3

3. Study area description

3.1 Geographic location

The Moutoa-Whirokino catchment is situated in the historic floodplain of the Manawatu River, located between the towns of Foxton and Shannon in the lower North Island of New Zealand (Figure 1). The catchment is positioned approximately 25 km southwest of Palmerston North and 10 km from the Tasman Sea coast. The area is divided by the Moutoa Floodway and encompasses approximately 6,000 hectares. The catchment feeds directly into the Manawatu Estuary, which forms where the Manawatu River meets the Tasman Sea at Foxton Beach.

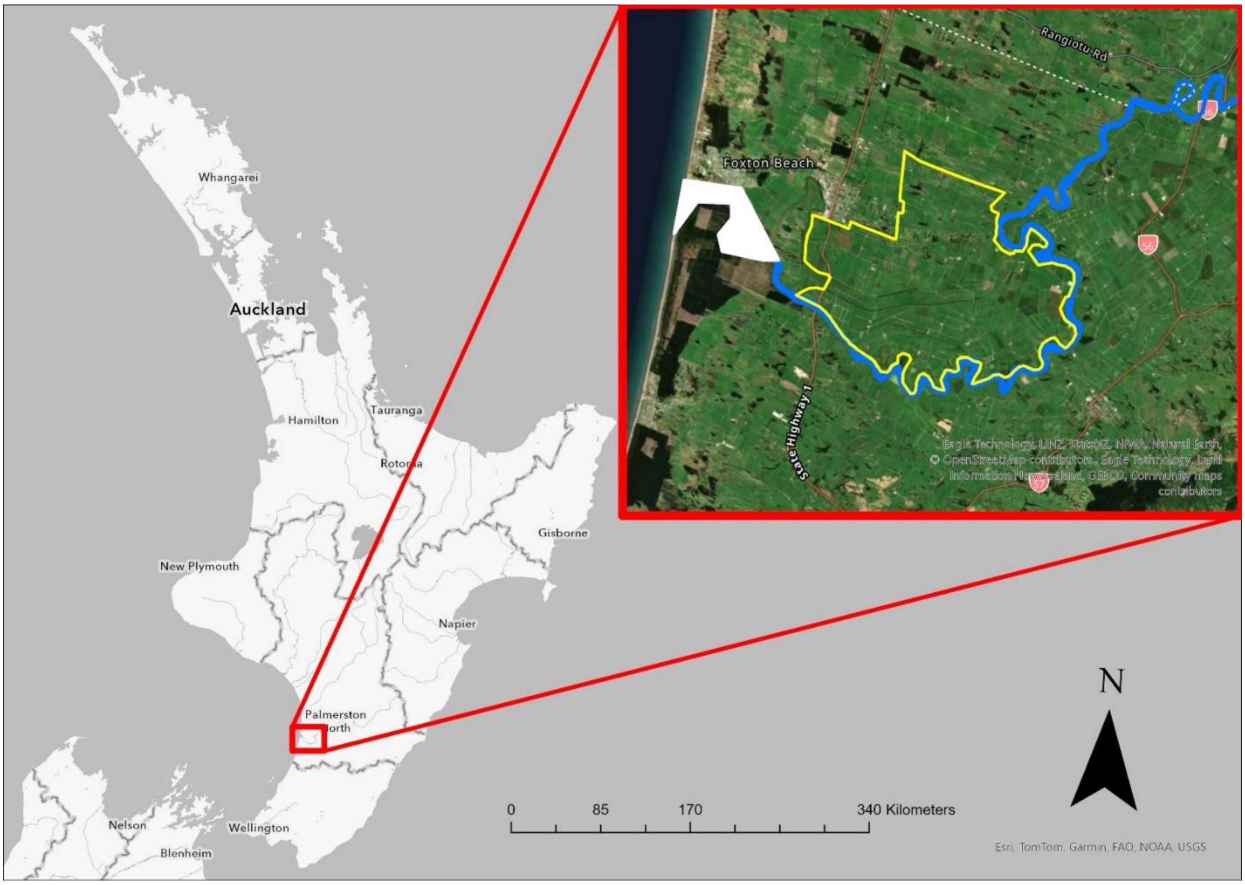

Figure 1: Moutoa catchment context within the North Island of New Zealand. Inset: Moutoa Catchment (yellow outline), Manawatu River (blue line), Manawatu Estuary (white polygon). (F. Burke, personal communication, July 27, 2023; NIWA, 2010).

3.2 Elevation and topography

The Moutoa area is at a low elevation, typical of an actively forming river floodplain. Ground elevations vary considerably across the catchment, with heights ranging from 1.8 metres to 6 metres above mean sea level (Kirkland, 1970). An example of the low-lying nature of this catchment can be seen in that the lowest portions of the Moutoa

Basin lie at only 1.8 metres above sea level, while adjacent areas near the Manawatu River south of Linton sit at approximately 15 metres above sea level (McNeill-Adams, 1968).

3.2.1 Geological context and floodplain development

The Moutoa area sits within a coastal plain developed on marine sediments. The current floodplain topography developed over a period when sea levels were approximately 120 metres lower than present (Clement et al., 2010). As sea levels rose, the incised Manawatu valley was progressively inundated, with estuarine conditions extending as far inland as Shannon and Opiki by approximately 7,500 years ago (Clement et al., 2010). This was followed by an infilling of this area with fluvial and estuarine sediments and created the low-lying coastal plain where the Moutoa catchment is located today (Clement et al., 2010).

As material eroded from the upper catchment has been transported downstream, a pattern of natural levees and inter-levee basins has developed (McNeill-Adams, 1968). Where the Manawatu River and its tributaries have built natural levees through repeated flooding and sediment deposition, the land adjacent to the river channels sits at relatively higher elevations. However, these levees are also an obstruction of drainage away from the river, creating low-lying areas between the levees where water accumulates (Tannock, 1968). The Moutoa Basin is one of the largest of these low-lying areas, with the lowest portions sitting below the level of the adjacent Manawatu River.

3.3 Climate

The Moutoa-Whirokino catchment has a temperate coastal climate which is characteristic of New Zealand's lower North Island west coast. The catchment's proximity to the Tasman Sea coast (approximately 10 km) and its position on the Manawatu floodplain strongly influence local climatic conditions (Chappell, 2015).

3.3.1 Temperature

The Manawatu-Wanganui region has relatively few climatic extremes, with temperatures displaying a relatively small range (Chappell, 2015). Mean temperatures in the low-lying coastal areas around Moutoa range from approximately 8°C in winter (July) to around 17-18°C in summer (January). The area experiences warm summer afternoon temperatures, with coastal locations recording slightly lower afternoon temperatures than sites further inland at similar elevations due to the cooling effect of sea breezes (Chappell, 2015). The daily temperature range in coastal areas like the Moutoa catchment is smaller than that of inland locations due to the modifying effect of the sea (Chappell, 2015). Low-lying areas around the coast have a median annual temperature of around 13.5°C, providing generally moderate conditions suitable for year-round pastoral agriculture (Chappell, 2015).

3.3.2 Rainfall

Rainfall in the Manawatu region increases from approximately 800 mm at the coast to more than 5,000 mm along the inland ranges, with no marked seasonality (Chappell, 2015). The Moutoa area, positioned on the coastal plain, falls within one of the driest parts of the Region, with the coastal and lowland area receiving less than 900 mm of rainfall per year (Chappell, 2015).

The catchment's position relative to the Tararua and Ruahine Ranges influences local rainfall patterns. The Moutoa area is in a partial rain shadow on the western side of these ranges, resulting in moderate rainfall totals. However, the area remains susceptible to heavy rainfall events which can deliver intense rainfall over short periods, contributing to flood risk in the low-lying catchment.

Historical records indicate that the number of rain days (days with measurable precipitation) in the lower Manawatu varies between 150 and 173 days per year (McNeill-Adams, 1968). Rainfall distribution is sufficiently reliable that no season experiences regular drought conditions, making the area well-suited to pastoral agriculture when adequate drainage is provided.

3.3.3 Wind and sunshine

The prevailing wind direction is westerly, with west to northwest winds predominating (Chappell, 2015). The Moutoa area is notably windy, particularly near the coast where onshore west-northwest winds are strong enough to initiate sand transport approximately 33% of the time (Clement et al., 2010). These persistent westerly winds have played a significant role in shaping the broader coastal landscape, driving the formation of extensive dune fields that influence regional land use patterns (Clement et al., 2010). Wind frequency and force decrease gradually from the coast inland, though the exposed nature of the floodplain means wind remains a significant climatic factor throughout the catchment.

Sunshine hours are relatively high in the coastal areas, with the weather often cloudy about the hills but improving toward the west coast where around 2,025 hours are recorded each year (Chappell, 2015). This high sunshine duration, combined with moderate temperatures, provides favourable conditions for plant growth and agricultural activities throughout much of the year.

3.4 Land use

The Moutoa-Whirokino catchment is currently dominated by intensive agricultural land use, with dairy cattle grazing being the main activity, alongside smaller areas of maize and vegetable cropping. Analysis of the Land Cover Database version 5.0 (Landcare Research New Zealand Ltd., 2020) reveals the land cover distribution across the catchment (Table 1).

High producing exotic grassland dominates the landscape, accounting for 92% of land cover, reflecting the intensive pastoral farming that characterises the area. This land use pattern reflects the dramatic transformation from the historical wetland and swamp environment.

Table 1: Land cover distribution proportions in the Moutoa area, as per the Land Cover Database version 5.0 (Landcare Research New Zealand Ltd., 2020).

Land Cover Class	Area (ha)	Percentage
High Producing Exotic Grassland	5,758	92.0%
Short-rotation Cropland	274	4.4%
Exotic Forest	59	0.9%
Lake or Pond	46	0.7%
Low Producing Grassland	37	0.6%
Flaxland	35	0.6%
Herbaceous Freshwater Vegetation	30	0.5%
Other	21	0.3%
Total	6,260	100%

3.4.1 Historical land use context

The Moutoa area has undergone a lot of transformation over the past century. In its pre-human state, the area was predominantly a large wetland with varying degrees of wetness (Tannock, 1968). Prior to European settlement, much of the floodplain was heavily forested, with forests dominated by Kahikatea and Pukatea growing near swamps, while Totara, Matai, and Rimu grew in better drained areas such as river levees (Tannock, 1968). The nearby Makerua Swamp contained extensive stands of Toe Toe and flax, with thick beds of peat beneath and ancient forest logs buried deeper still.

The flax industry drove early land development in the region (Figure 2). The Moutoa Estate, comprising around 2,000 hectares on the north bank of the Manawatu River, provided natural stands of flax (Phormium tenax) that were harvested and processed in nearby Foxton (Kirkland, 1970). By 1938, the Moutoa Estate was the last large area of natural flax remaining in the Manawatu region. As drainage schemes progressed and swamplands were converted to farmland, natural flax areas declined, giving way to the intensive pastoral farming that dominates today.

3.5 Soil types

The Moutoa catchment exhibits considerable soil diversity (Figure 3 and Table 2), reflecting its complex geomorphological history involving river alluvium, wind-blown sand, peat accumulation, and later extensive artificial drainage of soil profiles. Soil types range from well-drained sandy soils derived from coastal sand deposits to historically poorly drained organic peaty loams in historic swamp areas- now modified with artificial drainage- with river alluvium-derived soils throughout (Cowie & Rijkse, 1977).

Figure 2: Photographs associated with flax harvesting activities within the Moutoa area. Top left: Horse and cart loaded with flax from the Moutoa area, adjacent to the Manawatu River; Top right: Workers in the local flax mill named 'Poplar Mill'; Bottom: Men harvesting flax from the Moutoa area. (Foxton Historical Society, n.d.).

3.5.1 Moutoa soil types

The soil survey by Cowie & Rijkse (1977) identifies the soil types in the Moutoa catchment (Table 2).

Table 2: Soil types within the Moutoa catchment, listed from most dominant to least. (Cowie & Rijkse, 1977).

Soil Name	Parent Material	Drainage	Area (ha)
Makerua peaty silt loam (Mr1)	Peat and river alluvium	Very poorly to poorly drained	1,470
Carnarvon brown loamy sand/sandy loam (Cb-F)	Wind-blown sand	Imperfectly and poorly drained	1,386
Parewanui silt loam (Pa3)	River alluvium	Poorly drained	500
Kairanga peaty silt loam (K5)	River alluvium and peat	Very poorly to poorly drained	433
Parewanui fine sandy loam (Pa2)	River alluvium	Imperfectly drained	349
Kairanga silt loam (K2)	River alluvium	Poorly drained	324
Kairanga silt loam on sand (K3)	River alluvium	Poorly drained	316
Awahou loamy sand/sandy loam (A-F)	Wind-blown sand	Somewhat excessively drained	260
Motuiti sand (Hm-Ms)	Wind-blown sand	Somewhat excessively drained	218
Makerua peaty loam (Mr2)	Peat and river alluvium	Very poorly to poorly drained	205
Parewanui heavy silt loam (Pa4)	River alluvium	Poorly drained	196
Manawatu fine sandy loam (M2)	River alluvium	Well drained	141
Rangitikei sandy loam (R2)	River alluvium	Somewhat excessively drained	119
Meeanee-Farndon complex (MFc)	Estuarine alluvium	Poorly drained	96
Manawatu sandy loam M1	River alluvium	Somewhat excessively drained to well drained	75
Foxton black sand (F-O)	Wind-blown sand	Somewhat excessively drained	51
Kairanga heavy silt loam (K4)	River alluvium	Poorly to very poorly drained	36
Pukepuke brown (P-O)	Wind-blown sand	Somewhat excessively/imperfectly/poorly and very poorly drained	30
Rangitikei fine sandy loam (R3)	River alluvium	Well drained	26
Kairanga fine sandy loam (K1)	River alluvium	Imperfectly drained	22

3.5.2 Soil characteristics and agricultural implications

The predominance of Makerua peaty silt loam and Carnarvon soils reflects the catchment's geological history. Makerua soils, classified as organic soils with very poor to poor drainage, occur in low-lying areas that were historically flooded peat swamps (Cowie & Rijkse, 1977). These soils contain high organic matter content, particularly peat, which has important implications for nutrient cycling and potential nitrogen transformations under anaerobic conditions.

Cowie (1978) found approximately 35% organic matter content in the Makerua peaty silt loam soil in surface horizons (20.2% carbon content), creating conditions favourable for nitrogen and phosphorus mineralisation from organic forms. Cowie (1978) also found that carbon to nitrogen ratios were higher in this soil than others nearby, however they were low for a peaty soil and, "...probably a result of drainage and cultivation which have led to oxidation of the organic matter, and also mixing of some alluvial material with the peat" (p. 64). The combination of high organic carbon levels and shallow groundwater levels can create reducing conditions (low redox potential) where nitrification is inhibited and processes such as dissimilatory nitrate reduction to ammonium (DNRA) may contribute nitrogen transformations. Recent research demonstrates that DNRA can dominate over denitrification in pastoral systems with high soil carbon and anaerobic conditions (Friedl et al., 2018; Cheng et al., 2022), with DNRA rates exceeding denitrification by factors of 8 or more in carbon-rich environments similar to Moutoa's organic soils.

Carnarvon brown loamy sand and sandy loam soils, derived from wind-blown coastal sand, occupy the more consolidated sand plains and are slightly better drained in comparison, though they remain imperfectly to poorly drained (Cowie & Rijkse, 1977). In comparison to those soils formed in historic swamp areas, Cowie and Smith (1958) found the organic matter content of Carvarvon brown loamy sand to be approximately 7% (4.1% carbon content).

The Kairanga and Parewanui series, formed predominantly from river alluvium in frequently flooded and low-lying areas (Cowie, 1978), occupy various topographic positions across the floodplain, ranging from river basins to transitional areas near levees. The Kairanga peaty silt loam has formed in similar ways to the Makerua peaty silt loam, particularly occurring on the edges of historic swampland. However, Cowie (1978) found that the once deep and peaty historic upper soil horizon was gradually decomposing with the installation of artificial drainage. In the same way as the Makerua peaty silt loam, this process would create favourable conditions for the mineralisation of organic nitrogen and phosphorus. In comparison to the peaty soils, the Kairanga and Parewanui silt loam soils were found to have relatively low levels of organic matter at 6.3% and 4.6% respectively (Cowie, 1978).

Given the diversity in soil types, parent materials and organic matter content, phosphorus retention capacity is also likely to show variation across the catchment. Organic soils such as the Makerua may have inherently low P sorption capacity due to limited iron (Fe) and aluminium (Al) oxide content, making them vulnerable to P losses under intensive agricultural use. In contrast, alluvial soils such as the Kairanga and Parewanui soils may contain variable Fe and Al oxides that provide moderate P retention through surface sorption reactions. However, in the poorly-drained phases that dominate these series, seasonal reducing conditions may cause reductive dissolution of iron oxides, releasing previously sorbed P. The sandy soils (Carnarvon, Awahou) likely have minimal P retention capacity, functioning essentially as conduits for P transport with little buffering capacity.

The variety of parent materials and drainage classifications across this diverse range of soil types- and within a relatively small catchment- necessitates a complex dynamic of drainage networks and nutrient management in order to utilise this area for agriculture sustainably.

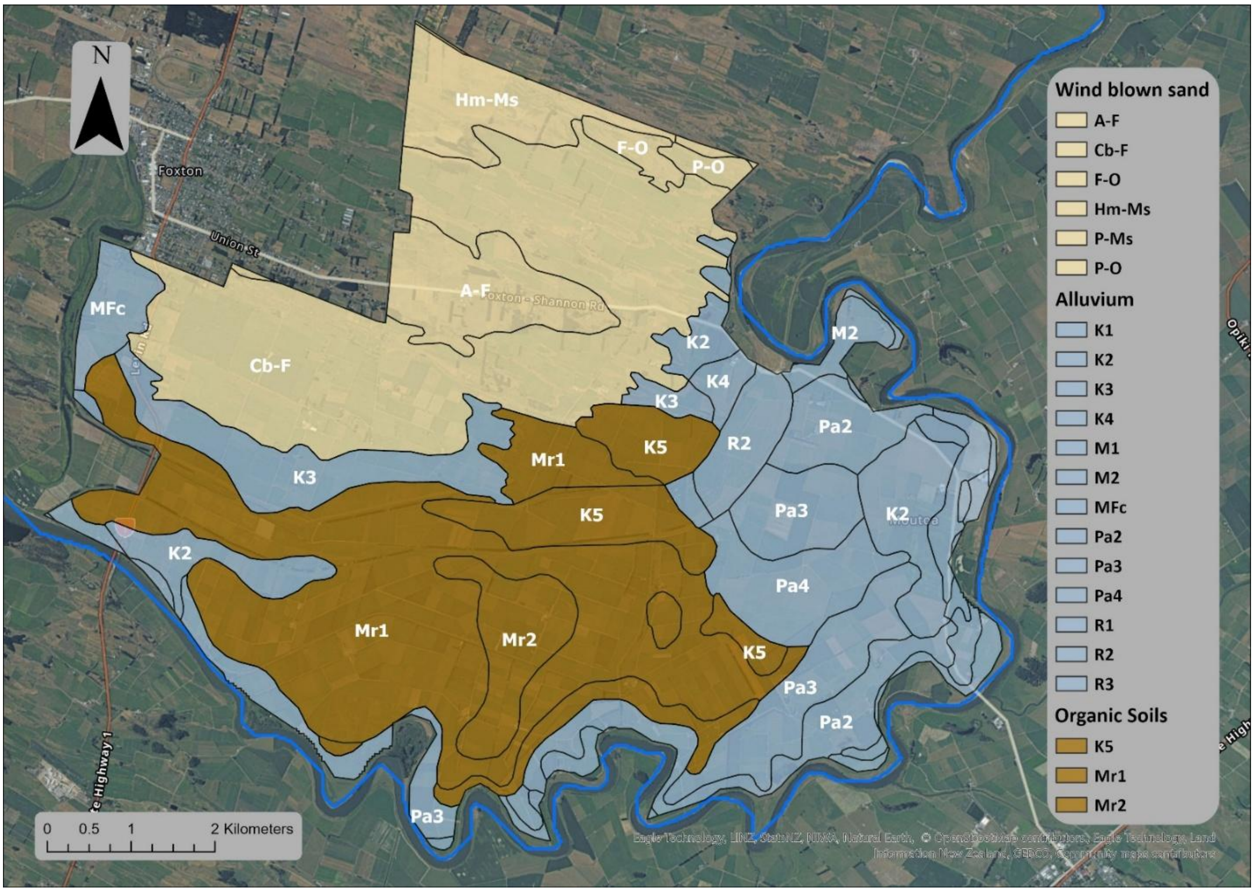

Figure 3: Soil map of the Moutoa catchment, coloured by predominant parent material (see Table 2 above for key to soil types and properties). Manawatu River in blue. (Cowie & Rijkse, 1977; NIWA, 2010).

3.6 Manawatu River, flooding, and the need for a drainage scheme

3.6.1 Topographic controls on flooding and drainage

The low-lying nature of the catchment, combined with its position near sea level, creates significant drainage challenges. During periods of high river flow, water floods from the Manawatu River channel onto the adjacent floodplain. When floodwaters enter the low-lying basins, the topography traps water in these depressions, preventing natural drainage back to the river (McNeill-Adams, 1968). The combination of minimal topographic relief and the obstruction of natural levees means that ponded water can persist for extended periods following flood events. With elevations declining by only a few metres over distances of several kilometres, drainage can be slow. This has required the installation of extensive networks of artificial drains and pumping stations to maintain adequate drainage for agricultural land use.

3.6.2 River characteristics and flood behaviour

The Manawatu River originates in the Ruahine Ranges north of Norsewood and drains an approximately 6,000 km^2 catchment before reaching the Tasman Sea at Foxton (Evans, 1964; Vale et al., 2016). The river follows a distinctive course, beginning on the eastern side of the ranges before cutting through the Manawatu Gorge and continuing westward across the Manawatu Plains (Figure 4). In the lower catchment, the river displays shallow gradients, with a fall of just 18 cm per km over the last 48 km of its journey to the sea (Horizons Regional Council, n.d.).

South of Palmerston North the river adopts a meandering pattern across its floodplain (McNeill-Adams, 1968). The gentle gradient of the lower river means that when there is heavy rainfall from the upper catchment, the lower stretch of river is unable to transport all the water within its normal channel, resulting in overspill onto the floodplain. This flooding challenge has been a defining feature of the area since settlement.

3.6.3 Historical flood impacts

Prior to comprehensive flood control, the Moutoa area underwent frequent and severe flooding. The area was subjected to flooding from the Manawatu River as often as three to four times per year (Kirkland, 1970). These regular inundations meant that much of the area remained as swampland, though the periodic flooding also built up the floodplain with deposits of alluvium, providing fertile, silty soils (McNeill-Adams, 1968).

The flooding posed significant challenges for agricultural development. As D.J. Halley, the first engineer of the Manawatu Catchment Board, stated in 1945, "From earliest times, settlers in the valley have seen their farms periodically flooded, their stock drowned and communications severed" (cited in McNeill-Adams, 1968, p. 27). He further noted that with intensive development of the Makerua and Moutoa areas, each successive flood would cause increasing damage and dislocation, and that further intensive development could not proceed due to the ever-increasing risk of flooding (McNeill-Adams, 1968).

A devastating flood in 1941 confirmed that effective, coordinated flood protection was badly needed (Horizons Regional Council, n.d.). The 1953 flood provided another major test of flood control infrastructure, requiring significant assistance from the army in laying sandbags to contain floodwaters.

3.6.4 Need for drainage

Beyond flood control, the inherently wet nature of the Moutoa area required extensive drainage in order for agricultural development to occur. In its natural state, the area varied considerably in wetness, with extensive areas where water lay permanently, forming large stretches of swamp country (Tannock, 1968). The natural levees built by the river obstructed drainage of these extensive areas, trapping water in low-lying basins.

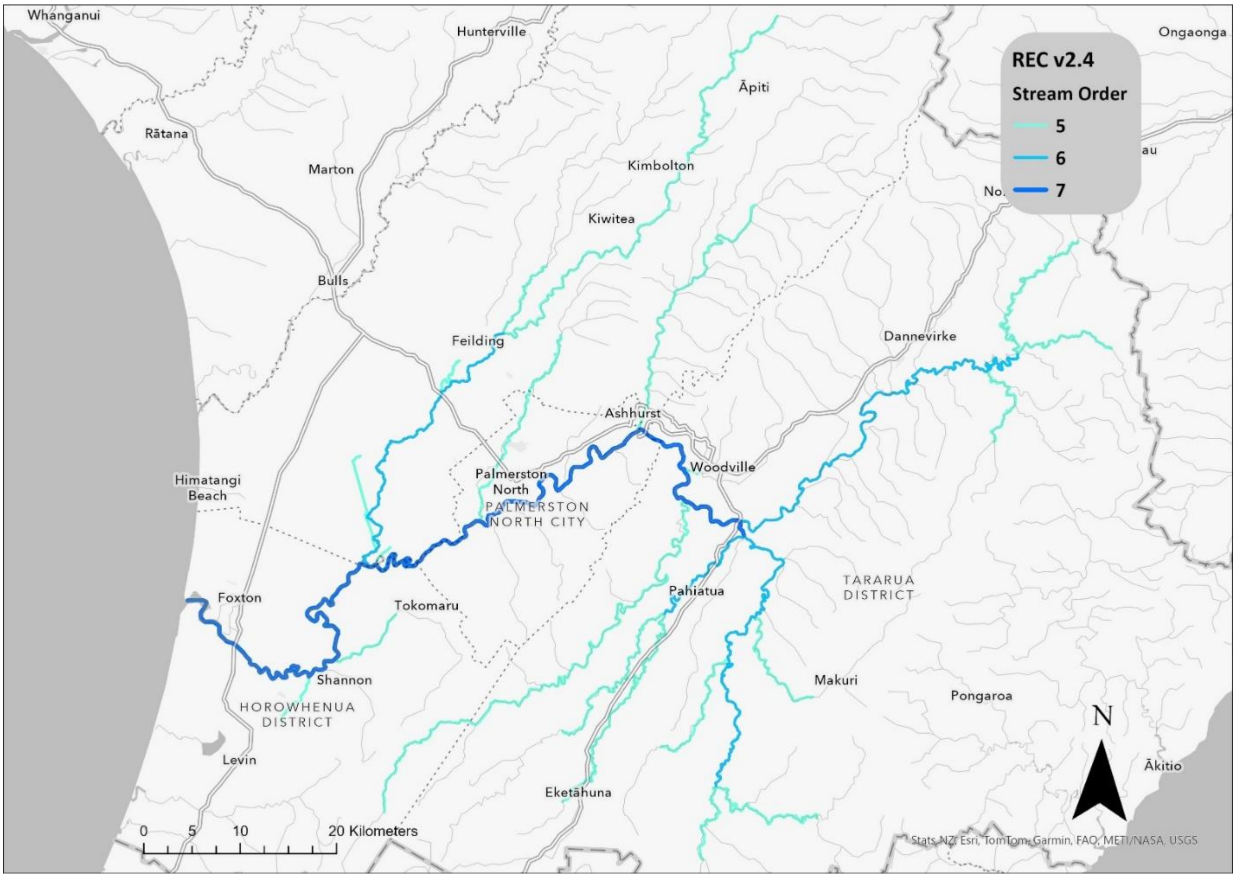

Figure 4: The Manawatu River and its tributaries (shown by blue lines), coloured by River Environment Classification stream orders 5 to 7 (NIWA, 2010).

The combination of frequent flooding and poor natural drainage meant that intensive agricultural land use, particularly dairying and cropping, was only possible with comprehensive drainage and flood control infrastructure. As noted in the study by McNeill-Adams (1968), flood control was deemed essential for successful farming in the Lower Manawatu, and the threat of serious loss from flooding had historically restricted the development of intensive farming on the lower levels of the floodplain.

3.7 History of drainage scheme

3.7.1 Purpose and development of drainage scheme

3.7.1.1 Early drainage efforts

Drainage development in the Moutoa area has evolved over more than a century, beginning with hand-dug drains in the low-lying swampy areas during the 1880s (Horizons Regional Council, n.d.). The flax industry was prospering during this period, and the Moutoa Drainage Board was established in 1908 when there were no stopbanks to prevent regular flooding from the Manawatu River (Wheeler, 2011).

The Wellington and Manawatu Railway Company played a crucial role in early drainage development. Between 1886 and 1894, the company undertook engineering surveys of the Makerua Swamp that demonstrated the practicability of a drainage scheme (Tannock, 1968). The company commenced drainage works to increase land values, constructing three main drains: Makerua Drain, Seifert's Drain, and Tokomaru Main Drain. These surveys revealed considerable fall from the centre of the swamp to the Tokomaru Stream, proving that the land could be drained despite popular rumours that the swamp was a, "...bottomless morass" (cited in Tannock, 1968, p. 130).

3.7.1.2 Development of organised flood control

By 1922, the Manawatu-Oroua River Board had been formed to control flooding, and stopbanks were constructed along the river, though occasional inundation still occurred (Wheeler, 2011). In 1944, the Manawatu Catchment Board was formed, and development of more comprehensive flood protection began, with its first task being to control flooding in the Lower Manawatu (Evans, 1964).

A pivotal point in the history of the drainage scheme came with the arrival of mechanical digging equipment in the 1920s, which enabled substantial areas of previously swampland to be drained and converted for agriculture (Figure 5) (Horizons Regional Council, n.d.). Various river and drainage boards installed stopbanks and greatly improved natural drainage and flood protection on the plain prior to the establishment of the Manawatu Catchment Board.

3.7.1.3 The Lower Manawatu Scheme and Moutoa Floodway

The Lower Manawatu River Flood Control Scheme was designed in 1946 by Paul Evans, Chief Engineer of the Catchment Board, to protect 280 km^2 of pastoral, horticultural, and urban land between Ashhurst and the sea (Horizons Regional Council, n.d.). A key component of the scheme included stopbanking much of the Manawatu River from the Manawatu Gorge to the sea, with the centrepiece being the Moutoa Floodway and Sluice Gates.

Completed in 1962, the Moutoa Sluice Gates and floodway were - and are - considered to be one of New Zealand's outstanding engineering projects of the 20th century (Horizons Regional Council, n.d.). The scheme came into full operation in 1963 and was designed to protect around 28,000-32,000 hectares of land in the Lower Manawatu from flooding (McNeill-Adams, 1968).

The floodway operates by diverting water from the main river through sluice gates into a 10 km channel that bypasses the 30 km of slow-flowing, meandering river channel and rejoins the river at Whirokino (Figure 6) (Horizons Regional Council, n.d.). This system has proven effective, with the gates being opened almost 50 times in their first 40 years to 2002, and on average every 15 months since then.

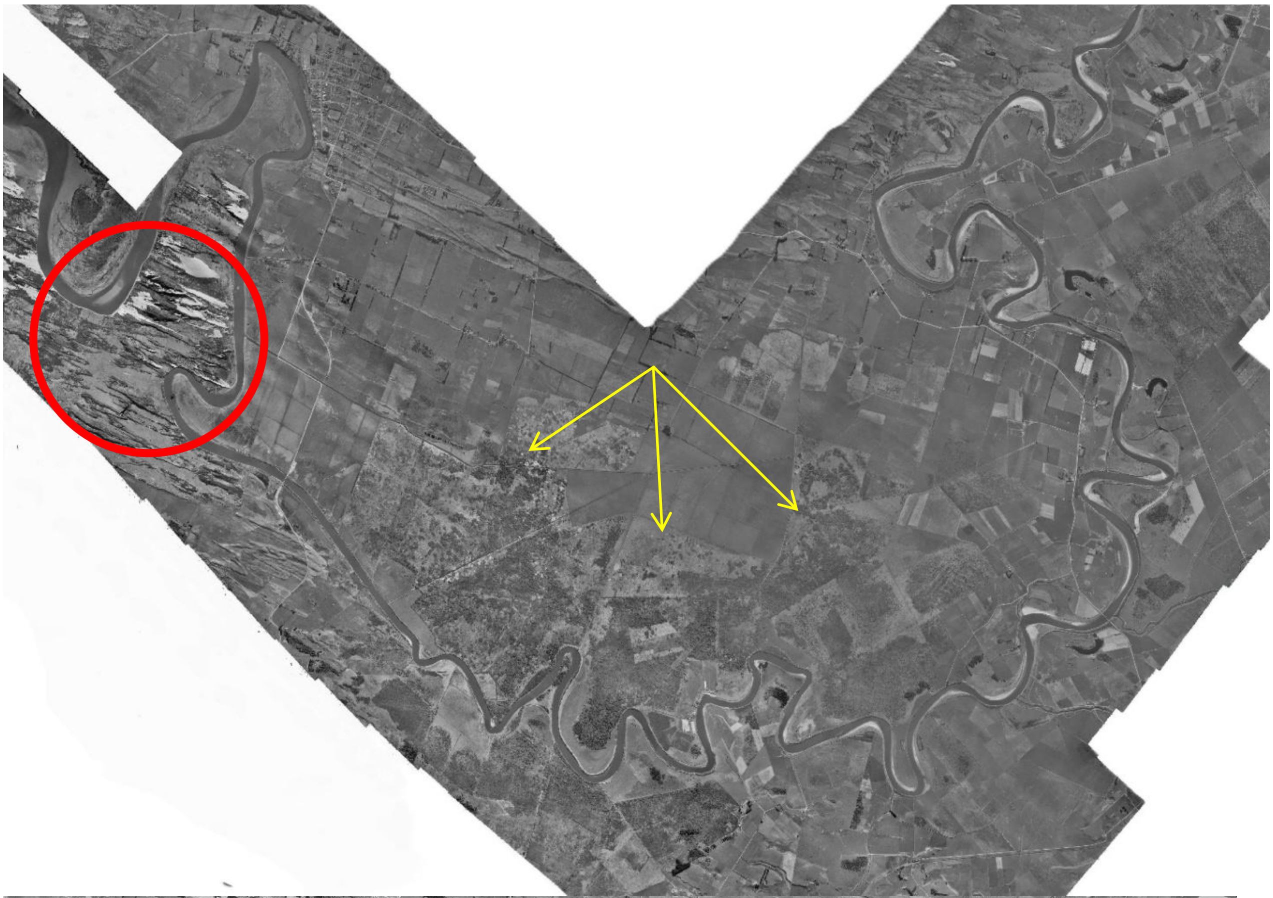

Figure 5: Land use change from 1937 (top) to 1985 (bottom) resulting from the successful implementation and functioning of the flood control and drainage scheme.

Note: the remaining extensive flax and swamp areas in the top image (yellow arrows); the 'Whirokino Cut' which cut off the majority of river flow from Foxton Loop and created a shorter path for the river to the Estuary (red circles in both images); and the floodway (blue arrow). (A. Steffert, personal communication, December 8 2023).

3.7.1.4 Development of pumping infrastructure

Following the successful design and implementation of flood control and with the flax industry decreasing in the area, there was a greater demand for pumping to allow an expansion of intensive land use such as dairying and cropping (Wheeler, 2011). To begin with, private pumps were installed to service small isolated areas. As the pumping network expanded, the relatively large Langleys pump was installed in 1962 by the Department of Agriculture to service the Moutoa Flax Block south of the floodway.

Figure 6: Moutoa sluice gates in operation. Water flowing from the flooded Manawatu River mainstem (mid- and background), into the floodway (foreground) via the sluice gates (Baalbergen, 2024).

In 1966 a large Pleuger pump was installed at the Main Drain outlet, at the end of the floodway (Wheeler, 2011). Langleys pump was replaced by Cooks and Diagonal pumpstations in 1981, which was part of an extensive drainage upgrade by the Department of Lands and Survey in order to enable subdivision of the Moutoa Flax Block into dairy farms. These two pumpstations were handed over to the Drainage Board in 1989.

Further improvements continued with a duty pump installed beside the large Pleuger in 2005, and the refurbished Langley pump installed at Kerekere Road to replace portable pumping (Wheeler, 2011). In 2010, a permanent two-pump station (Bowler pump) was constructed to replace the portable pump at No. 17 drain.

3.7.2 Drainage and flood control assets

The Moutoa-Whirokino Drainage Scheme currently provides pumping and gravity drainage for approximately 5,000 hectares of predominantly dairy and cropping land (Wheeler, 2011). The scheme manages an extensive network of infrastructure with a total asset value of approximately $11 million, as of June 2024 (Figure 7) (Horizons Regional Council, 2025).

3.7.2.1 Drains and floodgates

The Moutoa-Whirokino scheme manages approximately 77.7 km of drains throughout the catchment (Horizons Regional Council, 2025). Maintenance of these drains involves both spraying weeds and mechanical cleaning to remove weed and silt buildup. Smaller drains are generally sprayed once or twice per year in spring or autumn, with mechanical cleaning once every three years, while some large drains may be mechanically cleaned once or twice each year (Wheeler, 2011).

The scheme also manages 42 floodgates, all requiring maintenance. These gates function through gravity flow of water through the system, allowing water to flow downstream when the system is draining, and closing to stop backflow of water through the system in times of increased discharge through the system (Horizons Regional Council, 2025).

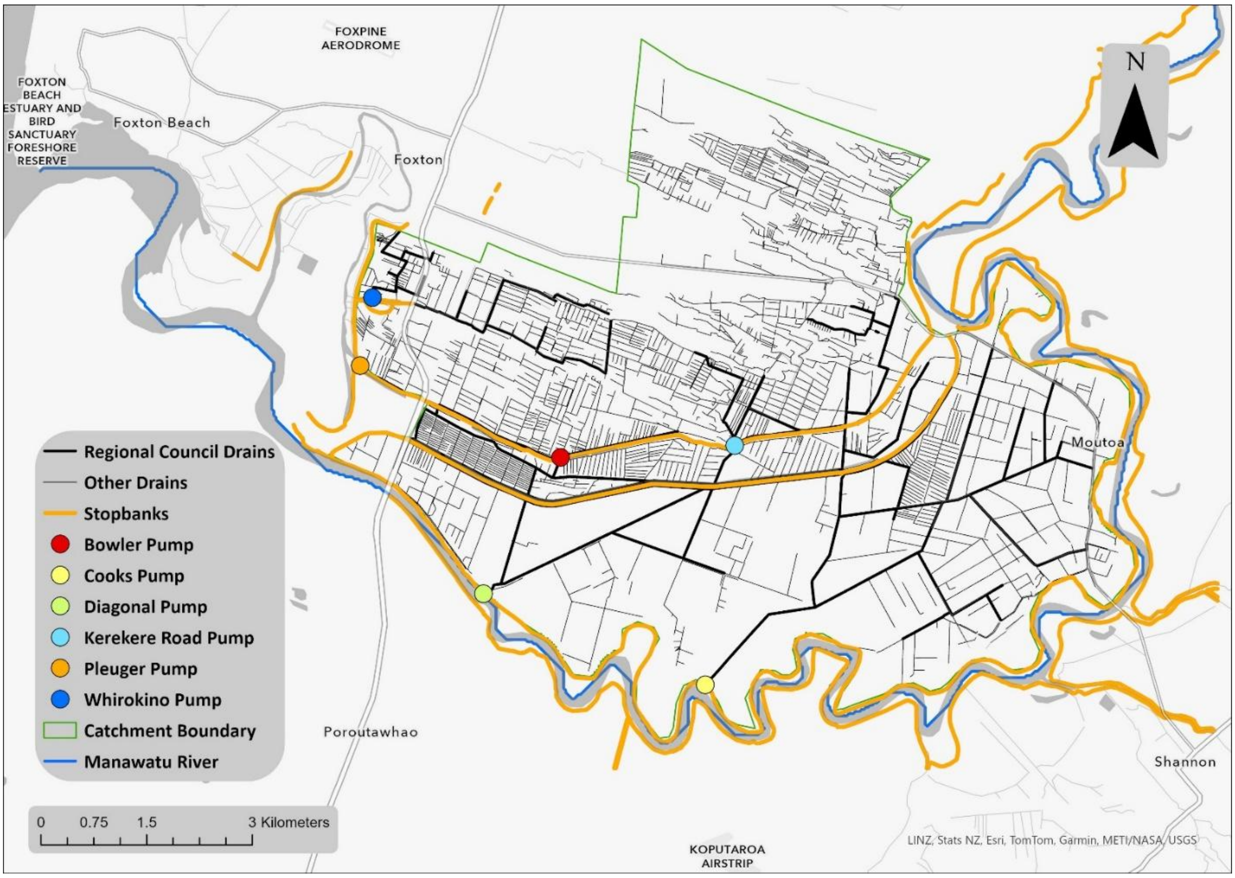

Figure 7: Moutoa catchment drainage network, stopbanks and pump stations. Note the floodway stopbanks bounding the floodway and bisecting the catchment. (A. Steffert, personal communication, December 8 2023).

3.7.2.2 Floodway

The Moutoa Floodway is a 10 km channel, 600 m wide, bounded on both sides by stopbanks 5.5 m high (Horizons Regional Council, n.d.). The floodway is designed to bypass the 30 km of slow-flowing, meandering channel of the lower Manawatu River around Koputaroa and Moutoa, allowing rapid discharge of floodwaters to the sea.

Damage during flood flows is prevented through the spillway through several features including energy dissipation blocks to reduce water speed as it passes under the gates, a curved shape to the structure which enables water spread out across the floodway, and the rising of the floodway away from the gates, which allows water to pool and not erode a channel out (Horizons Regional Council, n.d.).

The floodway is leased to farmers for grazing during normal conditions and operates during large flood events to reduce flood pressure on the lower Manawatu River plains (Figure 8). Nine steel radial gates, each weighing 15 tonnes, control water entry into the floodway. These gates are raised by a pulley system and operated by electric motors, with each gate capable of independent operation and standby power available in case of electricity failure (Horizons Regional Council, n.d.).

Figure 8: Moutoa floodway, borrow pits and farmland under normal conditions (left; note the full borrow pit in the centre of the image), and following an opening of the sluice gates and river discharge through the floodway, prior to being pumped out (right).

3.7.2.3 Borrow pits

The stopbanks surrounding the floodway were constructed using fill taken from large excavated pits called borrow pits (Figure 9) (Evans, 1964). These borrow pits were excavated to depths of 4.9 metres to 5.5 metres and widths of 27 metres to 30 metres using a Rapier W.90 walking dragline with a 33 metre boom (Evans, 1964). The excavation revealed the soil characteristics and layers of the area, with approximately 1.2 metres depth of peat, followed by 1.2 metres to 2.4 metres of clay, underlain by fine sea sand (see schematic at the top of Figure 9). These deep pits have subsequently filled with water, which eventually flows or is pumped out to the Manawatu River downstream.

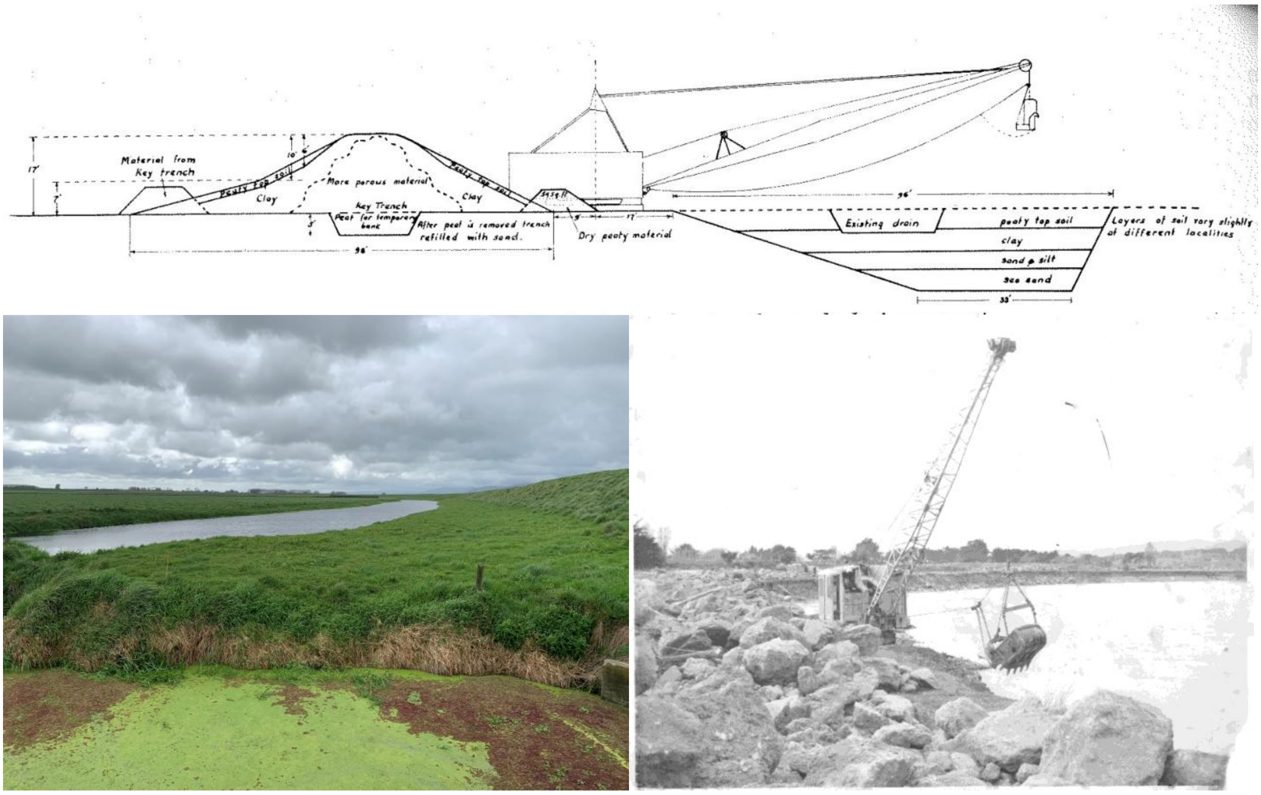

Figure 9: Borrow pits were dug out within the floodway as a source of fill for floodway stopbanks. Top: A schematic of the machine and methodology used to dig the pits and place fill for stopbanks (Evans, 1964); Bottom left: A borrow pit within the floodway full of water, with a stopbank on the right of the photo; Bottom right: The Rapier W.90 dragline in operation during the floodway construction (Evans, 1964).

3.7.2.4 Pump stations

The Moutoa-Whirokino Drainage Scheme operates six pumpstations (Figure 7) that are critical for maintaining drainage in low-lying areas (Wheeler, 2011; Horizons Regional Council, 2025):

1. Diagonal Pumpstation: Provides normal drainage for the extensive low-lying area south of the floodway, largely benefiting the Landcorp Moutoa Block (approximately 1,500 hectares divided into separate dairy units).
2. Cooks Pumpstation: Services slightly higher land east of Kerekere Road, which drains both to the floodway and to this station. During floodway events, water backs up and flows to Cooks Pumpstation.
3. Whirokino Pumps (Whirokino Pumpstation and Pleuger): Located at the Main Drain outlet at the end of the floodway, this is the largest pumping installation with the original Pleuger pump and a duty pump added in 2005.
4. Kerekere Road Pump: Permanent pumpstation installed in 2005 to replace portable pumping.
5. Bowler Pump: Permanent two-pump station constructed in 2010 to replace portable pumping.

The northern area generally drains into the floodway drainage system, with additional pumping capacity provided to maintain drainage when floodway drain levels are excessively high (Wheeler, 2011). Gravity drainage mostly discharges into the old Foxton Loop of the Manawatu River via the Main Drain Floodgate outlet, though this outlet is tidal and requires pumping to maintain drainage levels.

3.7.3 Hydrogeological setting and groundwater-surface water connectivity

The Moutoa-Whirokino catchment's position on the Manawatu floodplain and its low-lying topography result in shallow groundwater systems strongly connected to surface drainage networks. Prior to artificial drainage, the natural water table would have been at or near the surface for much of the year in the low-lying areas, creating the extensive wetland conditions described in historical accounts. The shallow nature of the groundwater system (even with drainage) means residence times would be short, on the order of days to months rather than years. In terms of the hydrology within the catchment this would lead to a strong connectivity between the subsurface and surface water.

3.7.3.1 Effect of artificial drainage on the groundwater system

The extensive mole-tile and open deep drainage systems restructure the natural hydrogeology in several ways:

1. Water table control: Mole and tile drains maintain a lowered water table, creating an artificial "perched" saturated zone above the tile drains during wet periods while rapidly draining excess water.

2. Preferential flow pathways: The mole channels and macropores in the soil, particularly in clay subsoils and peat layers, create rapid transport pathways that bypass the soil matrix. During high-intensity rainfall or irrigation, water and dissolved nutrients can move from the surface to tile drains in hours rather than the weeks or months.

3. Modified redox conditions: Drainage lowers the water table and introduces oxygen to formerly permanently saturated zones. However, in organic-rich subsoils (Makerua, Kairanga peaty series), oxygen may be rapidly consumed by microbial oxidation of organic matter, potentially creating complex redox conditions with oxidised zones near functioning drains and reduced zones in less well-drained areas between drains.

4. Direct connectivity: The pump stations provide direct hydraulic connection from the subsurface drainage network to the Manawatu River and floodway system, with minimal opportunity for natural attenuation of nutrients in riparian buffers or wetlands that would have occurred historically. The combination of shallow groundwater, artificial drainage infrastructure, and pumping creates conditions where nutrients mobilised from the soil zone can be rapidly delivered to surface waters, making this catchment particularly sensitive to nutrient management practices.

3.8 Potential effects of artificial drainage on land use, soils and drainage water quality

The extensive artificial drainage infrastructure described in this chapter has fundamentally transformed the Moutoa-Whirokino catchment, enabling the conversion of historically waterlogged wetlands into productive agricultural land. However, this transformation has created significant implications for land use capability, soil properties, and drainage water quality that are central to understanding the contemporary environmental challenges facing the catchment.

3.8.1 Land use transformation

The drainage scheme, comprising 77.7 km of drains, six pump stations, and extensive subsurface tile drainage, has been the primary enabler of intensive agricultural development in the Moutoa area. Without this infrastructure, the naturally waterlogged conditions would inhibit the high-intensity dairy farming that now dominates 92% of the catchment. The scheme has enabled the conversion of what were once permanently or seasonally inundated wetlands functioning as natural nutrient sinks, into land capable of supporting year-round grazing and cropping activities. This represents a fundamental shift from a landscape that naturally attenuated nutrients through processes such as denitrification and phosphorus sorption in wetland sediments, to one that has become a potential source of nutrients to downstream waters.

3.8.2 Soil alterations

The lowering of water tables through artificial drainage has initiated changes in soil properties and chemistry across the catchment's range of soil types, with direct implications for nitrogen and phosphorus dynamics. The introduction of oxygen to previously anaerobic soil profiles has enhanced aerobic decomposition processes, releasing nutrients that were previously locked in organic forms. This is significant given the high organic matter content characteristic of these soils, which developed over an extensive period in the historic Moutoa and Makerua swamps. In the Makerua and Kairanga peaty silt loams (together accounting for over 2,100 hectares) drainage has created conditions favourable to accelerated mineralisation of soil organic matter and potential accumulation of ammonia-N rather than its oxidation to nitrate, as organic-rich subsoils rapidly consume oxygen through microbial respiration. Under these reducing conditions, any nitrate-N that does form in the aerobic surface layer and leaches downward may be converted back to ammonia-N through dissimilatory nitrate reduction to ammonium (DNRA). The result is that drainage water may carry elevated ammonia-N rather than the nitrate-N typically associated with agricultural drainage.

The historical development of drainage (beginning in the 1880s, with major development in the 1920s-1960s) means that some parts of the catchment have experienced over 100 years of drainage-induced soil changes. Long-term drainage of organic soils results in subsidence and progressive oxidation of organic matter. These modifications have fundamentally altered the soil organic matter quality and nutrient cycling dynamics compared to recently drained areas, creating spatial variation in nutrient transformation and transport processes across the catchment. The catchment's soil diversity, ranging from somewhat excessively drained wind-blown sands to very poorly drained organic soils, creates complex spatial patterns of nutrient mobilisation and transport potential. The artificially drained Carnarvon sandy soils and the river alluvium-derived Parewanui and Kairanga silt loams each respond differently to drainage, influencing nutrient retention and movement across the landscape.

3.8.3 Drainage water quality implications

The artificial drainage network has created efficient pathways for nutrient transport from agricultural land to surface waters. The combination of surface drains, subsurface tile drains, and natural preferential flow paths through soil macropores means that nutrients can reach watercourses rapidly with minimal opportunity for natural attenuation.

In the poorly-drained organic soils occupying low-lying areas of the catchment, reducing conditions may persist in deeper soil layers despite surface drainage, potentially favouring ammonium accumulation rather than its oxidation to nitrate. The limited anion storage capacity characteristic of these organic soils also poses challenges for phosphorus management, with potentially limited retention by soil sorption processes. The pump stations, which operate in response to water levels, further enhance nutrient transport efficiency by rapidly moving water from the catchment to receiving waters.

This efficient drainage system, while essential for the agricultural productivity that characterises the catchment today, has fundamentally altered the hydrological and biogeochemical processes that once provided natural water quality protection. The challenge now facing the Moutoa-Whirokino catchment is to understand nutrient dynamics within this artificially drained landscape and to develop practical management approaches that can reduce nutrient losses while maintaining the drainage capability necessary for continued agricultural land use. This study addresses this challenge by investigating the critical pathways through which nitrogen and dissolved reactive phosphorus are transported from agricultural land through the drainage network to surface waters. Understanding whether ammonia-N dominance reflects reducing conditions in organic subsoils, DNRA processes, or incomplete nitrification in the drainage system is essential for developing targeted, effective mitigation strategies appropriate for this drained wetland environment.

Chapter 4

4. Materials and methods

4.1. Research design

4.1.1 Existing catchment group sampling locations

The local catchment group (Manawatu River Catchment Collective), comprised of local farmers, has been conducting monthly surface drain sampling across seven sites in the study area (Figure 10). This extensive drain monitoring by the local catchment group began in December 2022, establishing a baseline understanding of drain water quality conditions that revealed elevated levels of dissolved reactive phosphorous (DRP) and soluble inorganic nitrogen (SIN), both of which varied across the catchment drains. The main form of SIN was observed as ammoniacal-nitrogen (NH4-N) in some drains. Alongside the elevated ammonium levels, relatively low levels of nitrate-nitrogen (NO3-N) were recorded, prompting the need for more comprehensive research beginning in late 2023.

The existing catchment group sampling locations included (Figure 10):

- Whirokino Pump
- Levin Road
- Diagonal Pump
- Cooks Pump
- Tutoko Dairy
- Top Tributary
- Kerekere Road.

These sites continued to be monitored throughout the research period from 24 November 2023 to 18 October 2024, providing continuity with the initial baseline drain monitoring data and expanding the spatial coverage of the monitoring network.

4.1.2 Selection of study sites (surface and groundwater)

Building on the initial findings of the catchment group's drain monitoring, a more intensive field monitoring program was designed to understand sources and concentrations of nutrients in the drain network, while also identifying potential mitigation strategies. This involved monitoring of shallow groundwaters and drain water flow pathways in the catchment. The expanded drains and shallow groundwater monitoring network was designed to capture the variety of soil types and drainage patterns across the catchment.

4.1.2.1 Study area characteristics

The Moutoa-Whirokino catchment is situated in the historic floodplain of the Manawatu River between Foxton and Shannon, approximately 25km southwest of Palmerston

North and 10km from the coast. The catchment encompasses approximately 6,000 hectares and it was historically a large swamp/wetland area where flax was harvested. However, the predominant current land use is dairy cattle grazing, with a small mixture of maize and vegetable cropping.

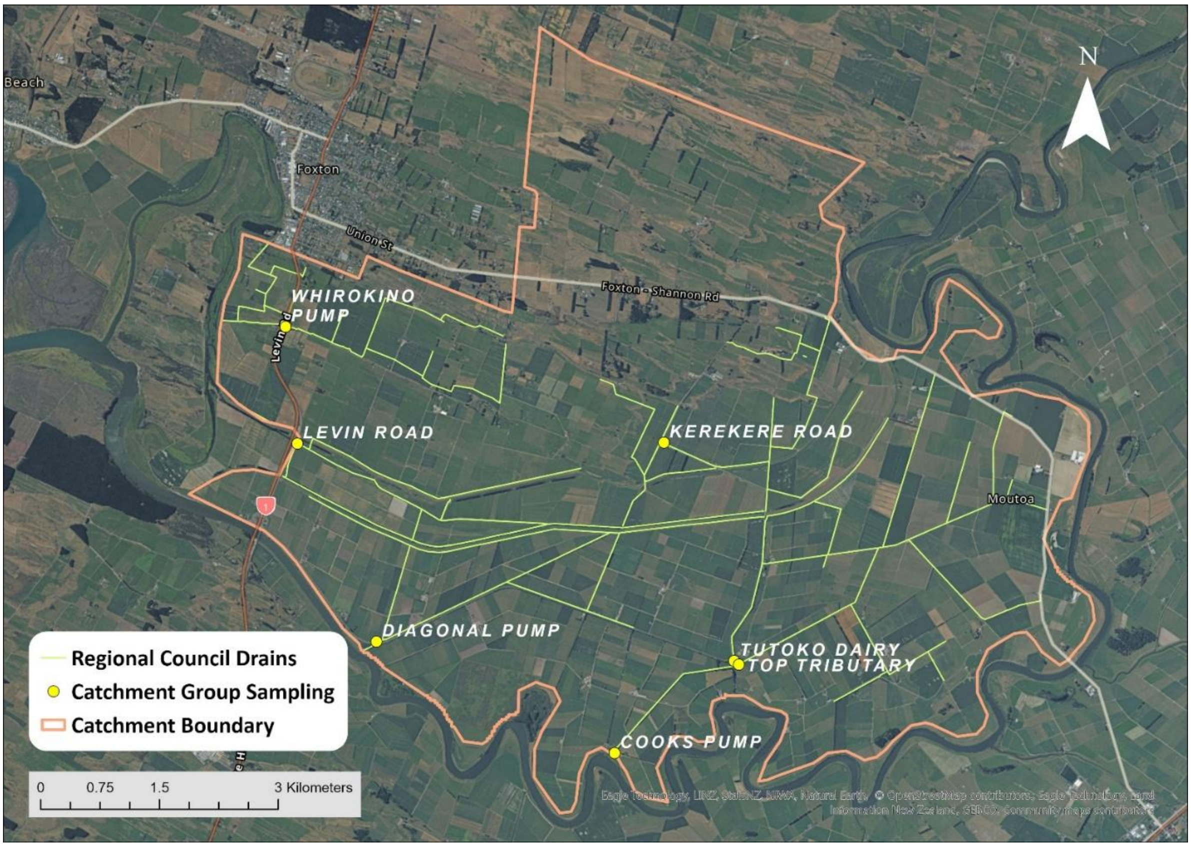

Figure 10: Local catchment group long-term surface water sample points for drain water quality in the Moutoa Catchment, Manawatu, NZ. Catchment outline in orange, main contributing drains to each sampling site in green, sampling points as yellow points.

The catchment's foundation on historic river deposits and former wetland areas has resulted in diverse soil types, including well-drained sand and sandy loams derived from wind-blown coastal sand (Carnarvon brown-Foxton, Cb-F), silt loams from river alluvium (Kairanga Silt Loam, K2; Parewanui silt loam, Pa3), and poorly drained peaty loams in historic swamp areas (Kairanga Peaty Silt Loam, K5; Makerua peaty silt loam, Mr1) (see prior Section Soil types for soil description and map (Figure 3)). These variations in soil type are particularly significant due to their varying levels of organic material and the impact this may have on nutrient cycling and reducing conditions in the shallow groundwater and drainage network (see prior Sections 2.3.2 'Nutrient dynamics in drained organic or peat soils' and 2.3.3 'Groundwater effects on nutrient dynamics').

An extensive drainage system has been established across the catchment to enable intensive agriculture, including large surface water drains, extensive subsurface drainage (mole and tile drains), pump stations, and a constructed floodway with associated floodgates. Borrow pits, created during the floodway construction, have filled with water and partially receive drainage waters from surrounding agricultural areas. See prior Section 3.7 ‘History of drainage scheme’ and Figure 7 for more details on drainage and a drainage network map.

The extensive drainage network was mapped across the catchment using Digital Elevation Models and imagery to aid assessment of water and nutrient flows pathways in the catchment (Figure 7). The drainage network, in combination of soil types and land use maps, was used to identify potential surface drain and shallow groundwater monitoring sites.

4.1.2.2 Shallow groundwater site selection

The monitoring of shallow groundwater was essential to understanding the origins and pathways of nutrients observed in the catchment's surface drains. The extensive artificial drainage network creates hydraulic connections between shallow groundwater and surface drains through subsurface tile and mole drainage systems, potentially facilitating rapid nutrient transport.

The presence of organic-rich peaty soils in historic swamp areas suggested the potential for anaerobic conditions in shallow groundwater. Under reducing conditions, organic matter decomposition can release ammoniacal-nitrogen while inhibiting nitrification processes, potentially explaining the observed pattern of elevated ammonium coupled with low nitrate levels in the catchment drains. Monitoring shallow groundwater across the catchment's soil type variability would enable identification of nutrient sources and inform targeted mitigation strategies.

A total of five shallow groundwater piezometers sampling sites were established across the catchment in late 2023, chosen to represent spatial variability but particularly soil type variability. These sites were identified as follows:

- PS-1 and PS-2: Carnarvon brown-Foxton soil (wind-blown sand);
- PCD-1-S and PCD-1-D: Kairanga Peaty Silt Loam (river alluvium and peat);
- PCR-1 and PCR-2: Kairanga Silt Loam (river alluvium);
- PTS-1 and PTS-2: Parewanui silt loam (river alluvium);
- PR-1 and PR-2: Makerua peaty silt loam (peat and river alluvium).

4.1.2.3 Drains water site selection

An additional sixteen surface drain water sampling sites were established to provide spatial variability, association with the five shallow groundwater piezometer sites, and understanding of potential nutrient flow pathways through the catchment. At least one surface drain water sampling site was positioned near each shallow groundwater sampling site. The network included both researcher-sampled sites (Spres-1, SKK-1, STS-1, STS-2, SR-1, SS-1, SCR-1, SCR-2, SBR-1, SCD-1, SBP-1, SBP-2, SBP-3, SBP-4, SPS-1, SPS-2) and the continuing community group monitoring locations (Figure 10).

The monitoring sites SBP-1 through SBP-4 and SPS-1 and SPS-2 were specifically located in or adjacent to the floodway borrow pits to assess their potential as drain water nutrient mitigation features.

4.2 Field observations and data collection

4.2.1 Shallow groundwater quality

4.2.1.1 Piezometer installation

A set of dual piezometers were installed at each of the five shallow groundwater sampling sites, with one piezometer installed deeper (5.0-5.5m below ground level, bgl) and another approximately 2-3m away at a shallower depth (4.0-4.5m bgl) to sample variation in groundwater parameters with depth (Table 3).

Additionally, at one site (near PCD-1-S and PCD-1-D), a set of seven piezometers (labelled as, JJ1, CD2, CD3, CD4, CD5, CD6, CD7) were installed perpendicular to a surface drain, to only measure shallow groundwater depth variations across the field when the drain flow was controlled.

Piezometers were constructed from 30mm diameter PVC pipe (28mm inner diameter) (Figure 11). The deeper piezometers (5.5m, bgl) were screened with a 0.5m long screen at the bottom (5.0 – 5.5 m, bgl), while the shallower piezometers (4.5m, bgl) had a 3.5m long screen at the bottom (1 – 4.5 m, bgl). The piezometers were screened by drilling 5mm holes approximately 1cm apart around the circumference of the piezometer for the required length, which were then covered using 250μm nylon mesh, with a nylon cap to close the bottom end of the piezometer.

Figure 11: A sample section of piezometer pipe, showing the screen at the bottom, consisting of drilled holes and mesh.

The piezometers installation was conducted as a direct-push using a tractor-mounted fence post driver (rammer) (Figure 12). In this technique, a 1m long steel casing with inner steel rod and drive point was positioned, with the post driver striking a plate attached to the top of the casing to ram it into the earth. The process was repeated, adding additional 1m sections of casing and ramming until the desired depth was reached. Once at the appropriate depth, the inner rod was lifted from within the outer casing using hydraulic lift. The piezometer was then slid into the space evacuated by the inner rod, and the outer casing was removed using hydraulic lift while keeping the piezometer pushed down with the weight of a 1m inner steel rod.

A primary pack consisting of coarse-grained quartz sand was used to pack the area around, and to 30cm above, the piezometer's screen. A secondary pack consisting of finer-grained sand was used to pack the area above the primary pack to 30cm below the ground surface. Approximately 10cm of bentonite was added, before a 20cm layer of cement was poured at the top to seal the casing. Following installation, piezometers were developed by pumping with a peristaltic pump to clear any sediment trapped during installation and to assess their functionality.

Table 3: Names and specifications of piezometers installed across the Moutoa catchment. This includes piezometer depth, screen length, and the associated soil types and parents materials for each piezometer. NOTE: A map of surface water and piezometer locations has not been included, in order to anonymise catchment group data.

Site ID	Depth (m, bgl)	Screen Length (m, bgl)	Soil Type	Soil Parent Material
PS-1	5.5	4.5 – 5.5	Carnarvon brown-Foxton	Wind-blown sand
PS-2	4.5	1.0 - 4.5	Carnarvon brown-Foxton	Wind-blown sand
PCD-1-D	5.5	4.5 – 5.5	Kairanga Peaty Silt Loam	River alluvium and peat
PCD-1-S	4.5	1.0 - 4.5	Kairanga Peaty Silt Loam	River alluvium and peat
PCR-1	5.5	4.5 – 5.5	Kairanga Silt Loam	River alluvium
PCR-2	4.0	1.0-4.0	Kairanga Silt Loam	River alluvium
PTS-1	5.5	4.5 – 5.5	Parewanui silt loam	River alluvium
PTS-2	4.5	1.0 - 4.5	Parewanui silt loam	River alluvium
PR-1	5.0	4.0-5.0	Makerua peaty silt loam	Peat and river alluvium
PR-2	4.0	1.0-4.0	Makerua peaty silt loam	Peat and river alluvium

4.2.1.2 Groundwater sampling (laboratory analysis and in-field)

Groundwater sampling was conducted monthly at all ten piezometers (five paired sites) beginning in late 2023 and continuing until late 2024. Sampling was undertaken in general accordance with the National Protocol for SOE Groundwater Sampling in New Zealand (Daughney, 2006), including pre-sampling, on-site preparation, purging, sample collection, and sample storage, transport and delivery. The water quality meters used in-field were calibrated prior to sampling, sampling equipment was clean, a minimum purge volume was calculated, and stabilisation of field parameters occurred. The collected groundwater samples were collected in 250mL polypropylene sterile containers, kept chilled in the field, and transported to the Central Environmental Laboratories in Palmerston North for analysis.

Figure 12: Installation methodology of piezometers illustrated. Top left: steel casing with driving point used to create space for piezometer; top middle: steel casing being rammed in with post rammer; top right: piezometer being inserted into hole; bottom: piezometers at completion of installation.

4.2.1.3 Groundwater sample analysis - sent to laboratory

Groundwater samples were submitted to the Central Environmental Laboratories, Palmerston North for analysis. The laboratory is accredited by International Accreditation New Zealand, and their tests were performed in accordance with their accreditation conditions. All laboratory analyses followed APHA (American Public Health Association) 24th or Online Edition standard methods where applicable (Table 4).

Table 4: Laboratory analysed parameters for shallow groundwater samples collected across the Moutoa catchment, including both the methods used and detection limits.

Parameter	Units	Analytical Method	Detection Limit
Alkalinity - Total	g/m^3 $CaCO_3$	APHA 24th Ed. 2320 B	1 g/m^3
Calcium - Dissolved	g/m^3	APHA 24th Ed. 3125 B, 0.45 micron filtered	0.1 g/m^3
Soluble Inorganic Nitrogen	g/m^3 N	Calculation from Ammonia, Nitrate and Nitrite	0.005 g/m^3
Chloride	g/m^3	APHA 24th Ed. 4110 B	0.1 g/m^3
Carbon - Dissolved Organic	g/m^3 DOC	APHA Online Ed. 5310 B	g/m^3 DOC
Iron - Dissolved	g/m^3	APHA 24th Ed. 3125 B, 0.45 micron filtered	0.005 g/m^3
Potassium - Dissolved	g/m^3	APHA 24th Ed. 3125 B, 0.45 micron filtered	0.1 g/m^3
Magnesium - Dissolved	g/m^3	APHA 24th Ed. 3125 B, 0.45 micron filtered	0.1 g/m^3
Manganese - Dissolved	g/m^3	APHA 24th Ed. 3125 B, 0.45 micron filtered	0.005 g/m^3
Nitrogen - Ammonia	g/m^3 NH_3-N	APHA 24th Ed. 4500 NH_3-F (Modified)	0.005 g/m^3
Nitrite	g/m^3 NO_2-N	APHA 24th Ed. 4110 B	0.005 g/m^3
Nitrate	g/m^3 NO_3-N	APHA 24th Ed. 4110 B	0.005 g/m^3
Phosphorus - Dissolved Reactive	g/m^3 PO_4-P	APHA 24th Ed. 4500-P E, 0.45 micron filtered	0.005 g/m^3
Sulfate	g/m^3 SO_4	APHA 24th Ed. 4110 B	0.05 g/m^3

4.2.1.4 In-field groundwater quality parameters

The piezometers samples were measured for their in-field water quality parameters using an Aqua TROLL 400 Multiparameter Probe (Table 5) attached to the peristaltic pump and groundwater flow cell. The Aqua TROLL 400 is a probe that measures multiple parameters from a variety of sensors. The probe was factory-calibrated, and field calibrations were conducted according to manufacturer specifications prior to going in the field.

Table 5: In-field parameters measured for shallow groundwater samples collected across the Moutoa catchment , including Aqua TROLL 400 specifications.

Parameter	Units	Accuracy	Resolution	Methodology
Temperature	°C	±0.1°C	0.01°C	Aqua TROLL 400
Dissolved oxygen concentration	mg/L	±0.1 mg/L (0-20 mg/L); ±5% (20-60 mg/L)	0.01 mg/L	Aqua TROLL 400
Dissolved oxygen saturation	%	As above	0.1%	Aqua TROLL 400
Actual conductivity	µS/cm	±0.5% + 1 µS/cm; ±1% max	0.1 µS/cm	Aqua TROLL 400
pH	pH units	±0.1 pH unit	0.01 pH unit	Aqua TROLL 400
Total dissolved solids	ppt	As conductivity	0.001 ppt	Aqua TROLL 400
Water level	m	±1cm	1cm	Dip meter

The probe was connected to a VuSitu Mobile App via Wireless TROLL for real-time data visualization and recording. Groundwater level measurements were recorded monthly as depth below ground level (m, bgl) using a water level dip meter.

4.2.2 Drainage water quality

4.2.2.1 Surface drain sampling

Surface drain water sampling was conducted fortnightly across all 16 locations (plus 7 monthly catchment group samples), focusing mainly on farm drains, over a period of 11 months from 24 November 2023 to 18 October 2024. The drain water samples were collected using a sampling pole to reach mid-channel or active flow areas, avoiding stagnant or standing water zones. The samples were collected in 250mL polypropylene sterile containers, kept chilled in the field, and transported to the Central Environmental Laboratories in Palmerston North for analysis.

4.2.2.2 Surface drain sample analysis (In-field and laboratory)

Laboratory analysis of surface drain samples included nitrogen species (ammonium, nitrite, nitrate, and calculated soluble inorganic nitrogen), total nitrogen, phosphorus species (dissolved reactive phosphorus and total phosphorus), as detailed in Table 6. Nitrogen-ammonia was analysed using APHA 24th Ed. 4500 NH_3-F (Modified), nitrite and nitrate using APHA 24th Ed. 4110 B, total nitrogen using APHA 24th Ed. 4500-P J and 4500-NO_2 B, dissolved reactive phosphorus using APHA 24th Ed. 4500-P E with 0.45 micron filtration, and total phosphorus using APHA 24th Ed. 4500-P J, E. Detection limits for all parameters are provided in Table 6. All laboratory analyses were conducted by the Central Environmental Laboratories, which is accredited by International Accreditation New Zealand (Table 6 and Table 7). The local catchment group samples analysed the same nitrogen and phosphorus parameters as the researcher samples (SIN, ammonia, nitrite, nitrate, DRP) using identical analytical methods and detection limits as listed in the table above. Additional field observations recorded by the catchment group included rainfall, weather conditions, and river flow status.

In-field surface drain water quality measurements were conducted simultaneously with the sample collection using the Aqua TROLL 400 Multiparameter Probe (specifications as detailed in Section 4.2.1.4 'In-field groundwater quality parameters'). The probe measured temperature, dissolved oxygen (concentration and saturation), actual conductivity, pH, and total dissolved solids in real-time.

Table 6: Laboratory analysed parameters for surface drain water samples collected across the Moutoa catchment, including both the methods used and detection limits.

Parameter	Units	Analytical Method	Detection Limit
Soluble Inorganic Nitrogen	g/m^3 N	Calculation from Ammonia, Nitrate and Nitrite	0.005 g/m^3
Nitrogen - Ammonia	g/m^3 NH_3-N	APHA 24th Ed. 4500 NH_3-F (Modified)	0.005 g/m^3
Nitrogen - Total	g/m^3	APHA 24th Ed. 4500-P J and 4500-NO_2 B	0.05 g/m^3
Nitrite	g/m^3 NO_2-N	APHA 24th Ed. 4110 B	0.005 g/m^3
Nitrate	g/m^3 NO_3-N	APHA 24th Ed. 4110 B	0.005 g/m^3
Dissolved Reactive Phosphorus	g/m^3 PO_4-P	APHA 24th Ed. 4500-P E, 0.45 micron filtered	0.005 g/m^3
Phosphorus - Total	g/m^3	APHA 24th Ed. 4500-P J, E	0.01 g/m^3

Table 7: Surface water samples (local catchment group) additional parameters, as analysed by Central Environmental Laboratories.

Parameter	Units	Analytical Method	Detection Limit
Turbidity	NTU	APHA 24th Ed. 2130 B (Modified)	0.01 NTU
Escherichia coli	MPN/100mL	APHA 24th Ed. 9223 B	1 MPN / 100 mL

4.2.3 Soil water quality

4.2.3.1 Suction cup installation

A set of four suction cups were installed at two sites, both adjacent to existing piezometer locations - two adjacent to PCD-1-S and PCD-1-D, and two adjacent to PCR-1 and PCR-2 (Figure 13). These sites were selected to enable comparison between soil water nutrient levels at 60cm soil depth and in shallow groundwater nutrient levels at depth of 1 to 5.5 m, bgl.

Suction cups consisted of porcelain-tipped tubes following the design of Briggs and McCall (1904), which operate by creating a pressure difference that draws water from soil capillary spaces into the porous tube, where it can be collected for analysis.

The suction cups were installed by soil augering to 60cm depth. A small amount of sand silica and 10mL of water were added to the bottom of each augered hole. The suction cups were inserted and the gap around each cup was backfilled with soil before a cap of bentonite was placed around the top to avoid the flow of water from surface water or rainfall. To protect the installations from damage by grazing stock, steel plates and pins were fabricated to cover the suction cups, and 150mm PVC pipe was installed surrounding each suction cup. The installations were positioned so that suction cups sat below the soil surface to prevent livestock damage.

4.2.3.2 Soil water sampling and analysis (parameters)

Soil water sampling was conducted at the same time as the sampling of the adjacent piezometers. A small pump was used to apply suction to the cups, which were left for a period of up to one hour to allow soil water accumulation following rain events. The suction cup samples were then collected in the same manner as groundwater and surface water samples (250mL polypropylene sterile containers, kept chilled, transported to the Central Environmental Laboratories).

Soil water samples were analysed by the Central Environmental Laboratories only for ammonia-nitrogen and nitrate-nitrogen using the same laboratory methods as surface drain and groundwater samples (Table 4 and Table 6).

Figure 13: An example of the suction cup set-up prior to installation (left) and post-installation in the field at Moutoa (right).

Study limitations: Suction cup sampling was implemented as an initial trial near the end of the field-sampling phase of the project. Consequently, only one round of samples was successfully collected from the suction cups. While these samples provided interesting preliminary insights into soil water nutrient concentrations, the limited dataset prevents statistically robust or conclusive findings regarding soil water dynamics. The results are therefore considered indicative rather than definitive in this study.

4.3. Data analysis

Data analysis was conducted using R Studio (Version 2025.09.1+401) and its associated packages. The analytical approach used several statistical techniques to characterise spatial patterns, temporal dynamics, and relationships between measured drain and shallow groundwater quality parameters across the catchment. All statistical tests were interpreted using a significance level of 0.05 unless otherwise stated.

4.3.1. Data preparation

Data from laboratory analyses and field measurements were compiled into structured datasets for analysis using the readxl package (Wickham & Bryan, 2023). Quality control procedures included:

- Verification of data entry accuracy through comparison with laboratory reports;
- Review of outliers in context of field notes and sampling conditions;
- Conversion of units to consistent formats (e.g., mg/L to g/m^3 where 1 mg/L = 1 g/m^3).

Data manipulation and preparation was conducted using dplyr (Wickham et al., 2023a) for operations including filtering, grouping, summarising, and mutating datasets. The tidyr package (Wickham et al., 2023b) was used for data reshaping operations such as converting between wide and long formats. The lubridate package (Grolemund & Wickham, 2011) was used for date-time operations, including calculating rainfall lag periods and temporal groupings. For values reported below detection limits, the detection limit value was used in calculations as a conservative approach.

4.3.2. Spatial analysis between the sites

4.3.2.1 Between-site comparisons

Spatial patterns in drain or shallow groundwater quality were assessed by comparing their fortnightly or monthly measured concentrations across the sampling sites. The measured sites were grouped by soil type association to examine whether soil parent material and organic matter content influenced nutrient concentrations and transformations in shallow groundwater and drain samples. For groundwater sites, comparisons were made between the ten piezometers (five paired locations). For surface drain water sites, comparisons were made across all 23 sampling locations (16 researcher-monitored sites plus 7 catchment group sites).

Descriptive statistics (median, mean, interquartile range, minimum, maximum) were calculated for each parameter at each site using the rstatix package (Kassambara, 2023) to characterise central tendency and variability. Box plots were generated using ggplot2 (Wickham, 2016) to visualise distributions and identify outliers. Sites were ordered by median concentration to identify spatial gradients and hotspots of elevated nutrient concentrations. Multi-panel figures combining multiple plots were created using the patchwork package (Pedersen, 2024).

4.3.2.2 Borrow pit analysis

To evaluate the potential of borrow pits as drain nutrient flow mitigation features, water quality parameters measure in borrow pit sites (SBP-1, SBP-2, SBP-3, SBP-4, and Levin Road) were compared against all other surface water sampling sites. This comparison tested whether borrow pits exhibited significantly different nutrient concentrations and field parameters compared to the surrounding drainage network, which would indicate their potential nutrient attenuation or transformation processes occurring within these features.

4.3.3 Statistical significance testing

4.3.3.1 Selection of statistical tests

Due to the non-normal distribution of most water quality parameters (confirmed through Shapiro-Wilk tests), non-parametric statistical methods were used as the primary analytical approach. For normally distributed data, parametric tests were used. The choice between parametric and non-parametric tests was made on a parameter-by-parameter basis following assessment of normality.

4.3.3.2 Kruskal-Wallis test

The Kruskal-Wallis test was used to determine whether significant differences existed in water quality parameters across groups (e.g., soil types, redox categories). This non-parametric test is appropriate for comparing three or more independent groups when data do not meet parametric assumptions. The test was undertaken using the rstatix package (Kassambara, 2023).

When the Kruskal-Wallis test indicated significant differences ($p < 0.05$), Dunn's post-hoc test with Bonferroni correction was applied using the FSA package (Ogle et al., 2023) to identify which specific pairs of groups differed significantly.

Effect sizes (eta-squared, η^2) were calculated for Kruskal-Wallis tests using the rstatix package (Kassambara, 2023) to calculate the proportion of variance explained by grouping variables. Effect sizes were interpreted as: small ($\eta^2 < 0.06$), moderate ($0.06 \leq \eta^2 < 0.14$), or large ($\eta^2 \geq 0.14$) following Cohen's guidelines adapted for non-parametric tests (Cohen, 1988).

4.3.3.3 Mann-Whitney U test

The Mann-Whitney U test (also known as the Wilcoxon rank-sum test) was used to compare two independent groups when data were non-normally distributed. This test was done using the rstatix package (Kassambara, 2023) and was applied when comparing parameters between two categories, such as anoxic versus mixed redox conditions, or borrow pit sites versus other surface water sites.

4.3.3.4 Paired t-test and Wilcoxon signed-rank test

For comparing paired observations from the same location (e.g., shallow versus deep piezometers at each of the five paired sites), the choice between parametric and non-parametric paired tests depended on data distribution:

- Paired t-test was used when the differences between paired observations were normally distributed;
- Wilcoxon signed-rank test was used when the differences were non-normally distributed.

These tests were done using base R statistical functions and examined whether systematic differences existed with depth at groundwater monitoring sites, providing insight into vertical stratification of water quality parameters.

4.3.3.5 Chi-squared test

The chi-squared test of independence was used to assess whether categorical variables were associated. This test was applied to examine relationships between soil type and redox classification categories, testing whether the proportion of anoxic versus mixed redox conditions differed significantly across soil types.

4.3.4 Correlation and regression analysis

4.3.4.1 Spearman's rank correlation

Spearman's rank correlation coefficient (ρ) was calculated using the rstatix package (Kassambara, 2023) to assess relationships between continuous variables.

Correlation analyses were conducted to:

- Identify relationships between different nutrient species (e.g., ammonia-N vs DRP);
- Examine associations between field parameters (e.g., dissolved oxygen vs pH);
- Assess relationships between nutrients and potential controlling factors (e.g., dissolved organic carbon vs ammonia-N).

Correlation matrices were visualised using the corrplot package (Wei & Simko, 2021) for heatmap displays, with colour scales provided by the viridis package (Garnier et al., 2023).

4.3.5 Temporal analysis

4.3.5.1 Time series visualisation

Time series plots were created using ggplot2 (Wickham, 2016) for all surface water sampling sites to visualise temporal patterns in nutrient concentrations across the monitoring period (November 2023 to October 2024 for researcher sites; December 2022 to September 2025 for catchment group sites). Date handling and formatting were performed using the lubridate package (Grolemund & Wickham, 2011) and scales package (Wickham & Seidel, 2022).

4.3.5.2 Seasonal patterns

For the extended catchment group dataset (December 2022 - September 2025), samples were grouped into temporal periods to assess whether conditions during the intensive study period differed from preceding or subsequent periods:

- Before Study: December 2022 - October 2023
- Study Period: November 2023 - October 2024
- After Study: November 2024 - September 2025.

Median concentrations were calculated for each period at each site using dplyr (Wickham et al., 2023a), and trends were examined to contextualise the intensive monitoring period within longer-term water quality patterns.

4.3.6 Groundwater classification

4.3.6.1 Hydrochemical Cluster and Principal Component Analysis

To identify hydrochemical signatures within the catchment groundwaters based on major ion chemistry, a hydrochemical cluster analysis was performed on major ion percentage data (expressed on a meq/L basis). Major ion concentrations (calcium, magnesium, potassium, chloride, sulfate, and alkalinity as bicarbonate/carbonate) were converted from mg/L to milliequivalents per litre (meq/L) to account for differences in ionic charge and allow meaningful comparison of relative ionic contributions. Sodium was erroneously left out of the laboratory analysis requested for groundwater samples, therefore was not included within any analyses in this research. For each sample, cation percentages and anion percentages were calculated on a meq/L basis, with each group summing to 100%. Cluster analysis was conducted using the cluster package (Maechler et al., 2023), with visualisation and optimal cluster number determination supported by the factoextra package (Kassambara & Mundt, 2020).

Principal Component Analysis (PCA) was performed using functions from the factoextra package to reduce the dimensionality of the major ion dataset and visualise relationships between sampling sites based on their hydrochemical characteristics. The first two principal components were examined to identify the dominant controls on groundwater chemistry, with variance explained by each component calculated.

Piper diagrams for visualising major ion chemistry and hydrochemical facies were created using The Geochemist's Workbench (Aqueous Solutions LLC). Schoeller diagrams showing major ion concentrations on semi-logarithmic plots were created using ggplot2 (Wickham, 2016) with the scales package (Wickham & Seidel, 2022) for axis formatting.

4.3.6.2 Groundwater redox classification

Groundwater samples were classified into redox categories using measured concentrations of dissolved oxygen, nitrate, manganese, iron, and sulfate following the framework of McMahon and Chapelle (2008). This classification is described in detail in

Section 4.4 below. Once classified, samples were grouped by redox category (oxic, mixed, or anoxic) and statistical comparisons were made using Mann-Whitney U tests (rstatix package; Kassambara, 2023) to assess whether redox conditions significantly influenced nutrient concentrations.

4.3.6.3 Soil type influences on groundwater quality

Water quality parameters were compared across the three soil type categories (sandy, silt loam, peaty) using Kruskal-Wallis tests (rstatix package; Kassambara, 2023) followed by Dunn's post-hoc pairwise comparisons (FSA package; Ogle et al., 2023).

Box plots for visualising distributions across soil types were created using ggplot2 (Wickham, 2016), with statistical comparison annotations added using the ggpubr package (Kassambara, 2023).

4.4. Groundwater redox assessment

4.4.1. McMahon and Chapelle (2008) framework

The redox status of groundwater samples was assessed using the framework developed by McMahon and Chapelle (2008), which classifies redox processes based on dissolved concentrations of five commonly measured water quality parameters: dissolved oxygen (O_2), nitrate (NO_3^--N), manganese (Mn^{2+}), iron (Fe^{2+}), and sulfate (SO_4^{2-}).

This classification follows the thermodynamic sequence of microbial respiration, where microorganisms preferentially utilise electron acceptors in order of decreasing energy yield:

$O_2 > NO_3^- > Mn^{4+} > Fe^{3+} > SO_4^{2-} >$ organic carbon.

The framework categorises groundwater into the following redox classes based on concentration thresholds (Table 8):

Table 8: Redox classification based on concentrations of five water quality parameters (adapted from McMahon & Chapelle, 2008).

Redox Category	DO (mg/L)	NO_3^--N (mg/L)	Mn^{2+} (mg/L)	Fe^{2+} (mg/L)	SO_4^{2-} (mg/L)
Oxic	≥ 0.5	Any	Any	Any	Any
Suboxic (Nitrate reduction)	< 0.5	≥ 0.5	< 0.05	< 0.1	Any
Suboxic (Manganese reduction)	< 0.5	< 0.5	≥ 0.05	< 0.1	Any
Anoxic (Iron reduction)	< 0.5	< 0.5	Any	≥ 0.1	> 0.5
Anoxic (Sulfate reduction)	< 0.5	< 0.5	Any	Any	< 0.5
Mixed	Combinations of electron acceptors present				

For samples where dissolved oxygen measurements were occasionally above the 0.5 mg/L threshold while other samples from the same site were below this threshold, a "mixed" classification was assigned to indicate temporal or spatial variation in redox conditions.

4.4.2. Redox classification procedure

Redox categories were assigned to each groundwater sample by evaluating measured concentrations against the threshold values listed above.

For each piezometer site, all samples collected during the monitoring period were classified individually. The dominant redox classification for each site was determined based on the most frequently occurring category across all sampling events. Sites where both oxic and anoxic classifications occurred were designated as "mixed" redox conditions.

The proportion of samples in each redox category was calculated for the entire dataset and for each soil type group. Statistical associations between soil type and redox classification were examined using chi-squared tests of independence.

4.4.3. Redox status association with nutrient dynamics

Once groundwater samples were classified by their redox status, groundwater nutrient concentrations (particularly ammonia-N, nitrate-N, and DRP) were compared across redox categories to assess the association between redox conditions and nutrient speciation and mobilisation. The relationships between groundwater redox conditions and nutrient concentrations were examined through:

- Comparison of median groundwater concentrations across redox categories using Mann-Whitney U tests (rstatix package; Kassambara, 2023);
- Correlation analysis between dissolved oxygen and nutrient species using Spearman's rank correlation;
- Assessment of whether sites with persistent anoxic conditions exhibited systematically different nutrient profiles.

4.5. Potential mitigation options - Borrow Pits

4.5.1. Borrow pit assessment framework

The potential of borrow pits to function as nutrient mitigation features was assessed through comparative analysis of water quality in borrow pit sites versus the surrounding drainage network. This assessment focused on identifying evidence for nutrient attenuation processes rather than quantifying removal rates, which would require influent-effluent monitoring with known hydraulic residence times.

4.5.2. Comparative water quality analysis

Water quality parameters measured in borrow pit sites (SBP-1, SBP-2, SBP-3, SBP-4, and the catchment group site Levin Road) were compared against all other surface water sampling sites using Kruskal-Wallis tests (rstatix package; Kassambara, 2023). Parameters compared included:

- Nutrient concentrations (ammonia-N, nitrate-N, DRP, total nitrogen, total phosphorus);
- Field parameters (dissolved oxygen, pH, temperature, conductivity, TDS).

4.5.3. Identification of potential treatment mechanisms

Based on the comparative water quality data and observed differences in physical characteristics between borrow pits and drainage channels, potential nutrient attenuation mechanisms were identified. This identification of treatment mechanisms was inferred, as the study did not include process-based measurements (e.g., sediment nutrient accumulation rates, nitrification-denitrification measurements, or hydraulic residence time quantification).

4.5.4. Assessment of current passive treatment performance

The passive treatment performance of borrow pits was assessed by calculating percentage reductions in median concentrations between the borrow pits and surface drains, as follows:

Reduction (%) = [(Median drains – Median borrow pits) / Median drains] × 100

This metric provides an indication of potential attenuation occurring between the drainage network and borrow pit waters, recognising that the calculation represents an order-of-magnitude estimate rather than a precise attenuation efficiency. True attenuation efficiency would require paired influent-effluent sampling at individual borrow pits with known residence times and water balance information.

4.5.5. Evaluation of optimisation and expansion potential

The potential for optimising existing borrow pit performance and expanding their catchment-scale impact was evaluated through qualitative assessment of:

- Current limitations (lack of residence time management, no vegetation establishment, unknown sediment accumulation);
- Optimisation opportunities (controlled outlets, vegetation establishment, periodic sediment removal);
- Expansion possibilities (routing high-nutrient drainage from priority areas through borrow pits);
- Infrastructure requirements and constraints.

This evaluation drew on published literature regarding treatment pond design principles and performance expectations, adapted to the specific context of the Moutoa-Whirokino catchment's drainage infrastructure and nutrient source areas identified through the spatial analysis (Section 4.3.2 ‘Spatial analysis between the sites’).

Chapter 5

5. Results and Discussion

This chapter presents and discusses the results of field investigations conducted in the Moutoa-Whirokino drainage catchment between November 2023 and October 2024. Section 5.1 examines shallow groundwater characteristics, including water table dynamics, hydrochemistry, redox conditions, and nutrient mobilisation processes across different soil types. Section 5.2 presents surface water nutrient patterns in the drainage network and their relationships with rainfall, seasonality, and regulatory standards. Section 5.3 evaluates the connectivity between groundwater and surface water nutrient concentrations through correlation analysis. Chapter 6 assesses nutrient mitigation strategies, including controlled drainage for source control and borrow pit systems for in-stream treatment.

5.1 Shallow groundwater analysis: spatial, temporal, chemistry, redox conditions, and nutrients

5.1.1 Shallow groundwater levels, spatial and temporal variations

Shallow groundwater level monitoring across 17 piezometer sites throughout the 11-month study period (24 November 2023 - 4 October 2024) revealed systematic patterns controlled by soil type, seasonal hydrological cycles, and site-specific characteristics. Shallow groundwater depths varied six-fold across the catchment (mean 0.28-1.78 m bgl), with maximum variation exceeding 2.4 m between shallowest (JJ1, PCD-1-S: 0.10 m bgl) and deepest (PTS-2: 2.56 m bgl) measurements (Appendix Table A1).

Soil type and shallow groundwater depth

Sites grouped into three categories based on soil hydraulic properties and topographic position (Table 9). Groundwater levels in peaty soil areas were the shallowest on average (mean 0.46 m bgl across nine sites), groundwater levels in sandy soil areas had intermediate depths (mean 0.94 m bgl), while groundwater levels in silt loam site areas had the deepest average levels (mean 1.27 m bgl). Statistical analysis confirmed significant soil type effects on groundwater levels (Kruskal-Wallis H=31.91, df=2, $p=1.18\times10^{-7}$, $\eta^2=0.282$), with levels in peaty soil areas differing significantly from both sandy (adjusted p=0.002496) and silt loam (adjusted $p=9.65\times10^{-8}$) sites (Appendix Table A2).

Within each soil type category, site-specific patterns emerged. Persistently shallow sites (CD2-CD7, JJ1, PCD-1-D, PCD-1-S) maintained water tables consistently within 0.5 m of ground surface with dampened responses to seasonal drought (ranges 0.45-0.50 m, CV 28-59%). The most stable sites were PCR-1 and PCR-2 (both Kairanga Silt Loam), displaying ranges of only 0.41-0.42 m and the lowest coefficients of variation in the monitoring network (15.38% and 16.96%). In contrast, PR-1, PR-2, PTS-1, and PTS-2 exhibited the most dramatic fluctuations (ranges 1.60-2.00 m, CV 37-62%), with water tables varying between approximately 0.5-2.5 m bgl seasonally.

Table 9: Summary of shallow groundwater level characteristics by associated soil type within the Moutoa catchment.

Soil Type	Sites (n)	Mean depth (m bgl)	Range (m)	Coefficient of Variation (%)
Peaty	11	0.46	0.45-2.00	28-62
Sandy	2	0.94	1.03-1.04	33-34
Silt loam	4	1.27	0.41-1.71	15-39

Seasonal cycles and rainfall response

Time series analysis shows seasonal groundwater level fluctuations superimposed on rainfall data indicating responses groundwater response (Figure 14). Across all monitoring sites within the sampling period (almost a full year), groundwater levels varied with shallow groundwater depths (low values in m bgl) during late spring/early summer (November-December 2023) and late winter/spring (August-October 2024), transitioning to deeper levels (high values in m bgl) peaking during autumn/early winter (March-June 2024). This pattern reflects the seasonal water balance, where higher evapotranspiration rates and typically lower rainfall during summer months drive aquifer drawdown, while winter rainfall recharge elevates water tables.

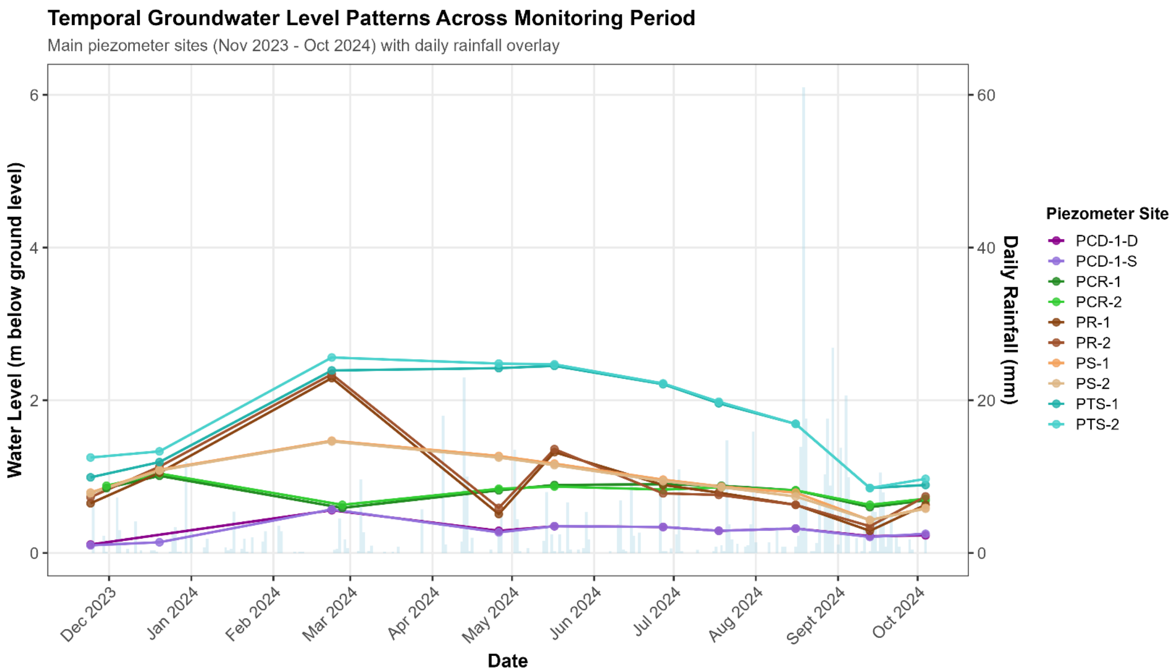

Figure 14: Shallow groundwater level time series plot with rainfall overlay within the Moutoa catchment for the sampling period, by site.

The drier summer period (January-April 2024) demonstrated varying site responses in shallow groundwater depth. Deep, variable sites (PR-1, PR-2, PTS-1, PTS-2) showed substantial groundwater level reductions to approximately 2.4-2.5 m bgl by late

February/early March, while shallow sites (CD cluster, PCD, JJ1, PCR) remained relatively stable. Following April 2024 rainfall, PR sites recovered gradually through May-August, while PTS sites showed minimal response until August, suggesting differing hydrogeological controls such as lower hydraulic conductivity, greater separation from recharge zones, or site-specific drainage network effects.

The substantial rainfall during mid-to-late August 2024 drove rapid water table recovery across all sites. PTS-1 and PTS-2 piezometers exhibited particularly dramatic responses, rising 1.0-1.5 m over approximately 3-4 weeks (from approximately 2.2-2.5 m bgl to 1.0-1.5 m bgl). By October 2024, all sites showed elevated groundwater levels relative to summer conditions, demonstrating the strong coupling between rainfall and groundwater levels in this artificially drained catchment.

Implications for biogeochemical processes

These groundwater level fluctuations create conditions for enhanced nutrient mobilisation, particularly in organic-rich soils. Groundwater level drawdown during summer exposes previously saturated organic horizons to atmospheric oxygen, promoting aerobic decomposition and ammonium production through organic matter mineralisation, with decomposition rates accelerating when increased oxygen availability stimulates aerobic microbial activity (Tiemeyer et al., 2007; Van Beek et al., 2007; Becher et al., 2023). Research by Sapek et al. (2007, 2021) in Polish peat grasslands identified groundwater level fluctuations as the primary factor controlling phosphate and ammonium mobilisation, with lowering water tables during late summer mobilising both nutrients into groundwater. Their findings showed that these nutrients were then transported to deeper groundwater layers and drainage channels when groundwater levels recovered, dynamics reflective of the conditions in the Moutoa-Whirokino catchment.

Sites maintaining shallow, relatively stable groundwater levels (CD sites, PCD sites, JJ1) exist under persistently anaerobic conditions throughout most of the year, favouring continuous ammonium production through mineralisation combined with minimal nitrification. Sites with greater fluctuations (PR, PTS sites) undergo repeated oxidation-reduction cycles potentially releasing accumulated nutrients with each drawdown-recovery event. Kassim and Yaacob (2019) documented in Malaysian tropical peatlands that both nitrate-nitrogen and ammonia-nitrogen concentrations were maximised when water tables were maintained at 40 cm below ground surface. In the Moutoa-Whirokino system, sites maintaining water tables in the 0.3-0.5 m depth range (CD sites, PCD sites, JJ1) are very similar to these conditions for nutrient mobilisation throughout much of the year.

5.1.2 Groundwater chemistry and redox assessment

Groundwater chemistry provides essential context for understanding nutrient behaviour. This section describes field parameters, redox conditions, major ion composition, and hydrochemical facies, establishing the biogeochemical framework controlling nitrogen and phosphorus transformations.

5.1.2.1 Groundwater field parameters

Monthly field measurements at ten piezometers (November 2023-October 2024, n=10-11 samples per site) revealed marked differences in shallow groundwater across their associated soil types for water level, conductivity, pH, TDS, and dissolved oxygen, while temperature showed no significant variation (Table 10, Figure 15). Kruskal-Wallis tests confirmed large effects for conductivity (η^2=0.509, p=7.22×10^{-13}) and TDS (η^2=0.510, p=6.69×10^{-13}), moderate effects for pH and dissolved oxygen (η^2≈0.13, p<0.001), and negligible temperature effects (Appendix Table A3).

Table 10: Shallow groundwater field parameters by associated soil type (median values with interquartile range).

Parameter	Sandy (n sites=2)	Silt Loam (n sites=4)	Peaty (n sites=4)
Water Level (m bgl)	0.91 (0.75-1.15)	0.83-1.83*	0.28-0.75*
Temperature (°C)	15.7 (14.6-16.6)	14.9-16.7*	15.6-15.8*
Conductivity (μS/cm)	498.45 (388.20-726.90)	393.5-686*	1073.7-1128.2*
pH	7.42 (7.06-7.74)	6.58-7.42*	7.35-7.68*
TDS (ppt)	0.32 (0.27-0.47)	0.27-0.45*	0.69-0.73*
DO (mg/L)	0.09 (0.04-0.16)	0.09-4.96*	0.08-0.60*
DO Saturation (%)	0.91 (0.36-1.58)	0.89-49.83*	0.86-6.26*

**Range of median values across sites within soil type (Appendix Table A4 contains complete site-level data)*

Groundwater Conductivity and Total Dissolved Solids

Groundwater electrical conductivity and total dissolved solids showed the strongest differentiation across soil types, with large effect sizes (η^2=0.509 and 0.510, respectively) indicating that soil type groupings explain approximately 51% of the variability in these parameters.

Peaty soil groundwater had elevated conductivity (median 1075-1126 μS/cm) and TDS (median 0.69-0.73 ppt) compared to sandy (498 μS/cm, 0.33 ppt) and most silt loam sites (394-686 μS/cm, 0.27-0.45 ppt). Post-hoc tests confirmed peaty soils differed highly significantly from both sandy (adjusted p=2.52×10^{-8}) and silt loam (adjusted p=6.45×10^{-11}) sites. Following Younger's (2007) classification scheme (freshwater: TDS <1000 mg/L; brackish: 1000-10,000 mg/L), all sites classify as freshwater based on median concentrations (Appendix Table A6). This classification aligns with expectations for a groundwater system approximately 10 km inland from the coast, where marine influence should be minimal in shallow aquifers actively recharged by rainfall and irrigation.

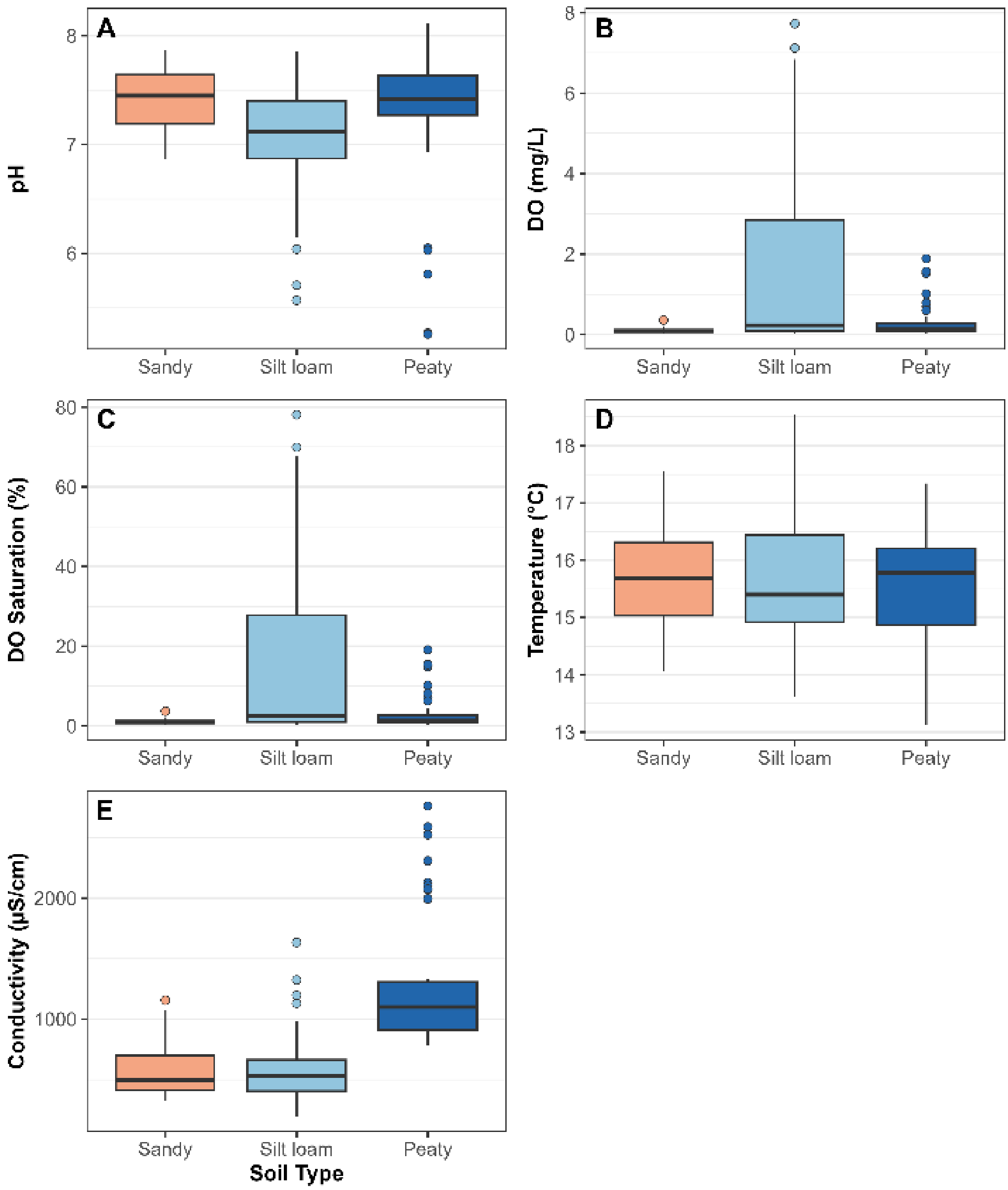

Figure 15: Spatial variability and patterns in groundwater field parameters across soil types (sandy, silt loam and peaty/organic soil).

The elevated TDS in organic soil groundwater reflects enhanced mobilisation of dissolved ions through organic matter decomposition under predominantly reducing conditions, as detailed in subsequent sections. This mineralisation process releases base cations (Ca^{2+}, Mg^{2+}, K^{+}), generates bicarbonate through microbial respiration, and mobilises nutrients (NH_4^{+}, DRP), creating the distinctive hydrochemical signature of drained peatland groundwater.

The near-perfect correlation between conductivity and TDS (Spearman's $\rho=0.993$, $p<0.001$) confirms that conductivity measurements reliably proxy for total dissolved solids in this system, with both parameters fundamentally reflecting ionic content derived from soil-water interactions and organic matter decomposition.

The TDS patterns in Moutoa-Whirokino groundwater align with international observations from intensively managed drained peatlands. Van Beek et al. (2007) reported groundwater TDS in Dutch peat grasslands under dairy farming averaging 600-800 mg/L, which is comparable to the 805-856 mg/L means at Moutoa-Whirokino peaty sites. Tiemeyer et al. (2007) documented even higher variability in degraded German peatlands, with some locations exceeding 1500 mg/L during dry periods, attributed to evaporative concentration combined with intensive fertiliser use.

Groundwater pH

Groundwater pH in showed moderate soil type effects (η^2=0.132, p=0.000329). Groundwater at sandy sites maintained slightly alkaline conditions (median pH 7.28-7.56), at peaty sites exhibited near-neutral to slightly alkaline pH (median 7.35-7.68), while silt loam sites showed greater variability (median 6.58-7.42), with PCR-2 displaying somewhat acidic conditions (median 6.58) contrasting with other silt loam sites (median 7.12-7.42). Post-hoc tests revealed groundwater at silt loam sites differed significantly from both sandy (adjusted p=0.004344) and peaty (adjusted p=0.001258) sites, driven primarily by PCR-2's low pH values.

Dissolved oxygen and redox implications

Dissolved oxygen concentrations in groundwater samples were generally very low across the catchment (median 0.09-0.60 mg/L, saturation 0.91-6.26%), indicating widespread oxygen-limited conditions. Notable exceptions occurred at PCR-2 (median 1.98 mg/L, 21.15% saturation) and PTS-2 (median 4.96 mg/L, 49.83% saturation), suggesting occasional atmospheric oxygen input through rainfall infiltration or preferential flow pathways. However, even these elevated values remained well below full saturation, and detailed redox classification (Section 5.1.2.2) revealed no samples achieved fully oxic status. The consistently low dissolved oxygen across most sites establishes the fundamental condition controlling nitrogen speciation patterns examined in Section 5.1.3.

5.1.2.2 Groundwater redox classification and controls

Redox (reduction-oxidation) conditions effect nitrogen and phosphorus transformations in groundwater systems. This section classifies groundwater redox status using the McMahon and Chapelle (2008) framework, which employs dissolved oxygen, nitrate, manganese, iron, and sulfate concentrations to identify dominant terminal electron-accepting processes occurring during microbial respiration.

McMahon and Chapelle (2008) redox classification

Groundwater redox conditions were classified according to the sequential reduction framework developed by McMahon and Chapelle (2008), which identifies dominant terminal electron-accepting processes based on measured concentrations of dissolved oxygen (O_2), nitrate-nitrogen (NO_3^--N), manganese (Mn^{2+}), iron (Fe^{2+}), and sulfate (SO_4^{2-}). This classification follows the thermodynamic sequence of microbial respiration, where microorganisms preferentially utilise electron acceptors in order of decreasing energy yield: $O_2 > NO_3^- > Mn^{4+} > Fe^{3+} > SO_4^{2-}$ > organic carbon (producing CH_4).

Redox classification results

Classification of 109 groundwater samples revealed predominantly oxygen-limited conditions throughout the shallow groundwater system (Figure 16, Appendix Table A5). No samples met oxic criteria (O_2 >0.5 mg/L combined with low Fe and Mn). The classification revealed that anoxic conditions dominate in shallow groundwater across the catchment, with 7 out of 10 monitoring piezometers sites exhibiting predominantly anoxic redox status (PCD-1-D, PCD-1-S, PCR-1, PR-1, PS-1, PS-2, PTS-1). These sites consistently display dissolved oxygen concentrations below the 0.5 mg/L threshold (mean DO ranging from 0.065 to 0.260 mg/L), indicating oxygen depletion and the prevalence of anaerobic microbial processes. Only 3 sites show mixed (oxic-anoxic) conditions (PCR-2, PR-2, PTS-2), characterised by higher mean dissolved oxygen (0.743 to 4.893 mg/L) and temporal change between oxic and anoxic states.

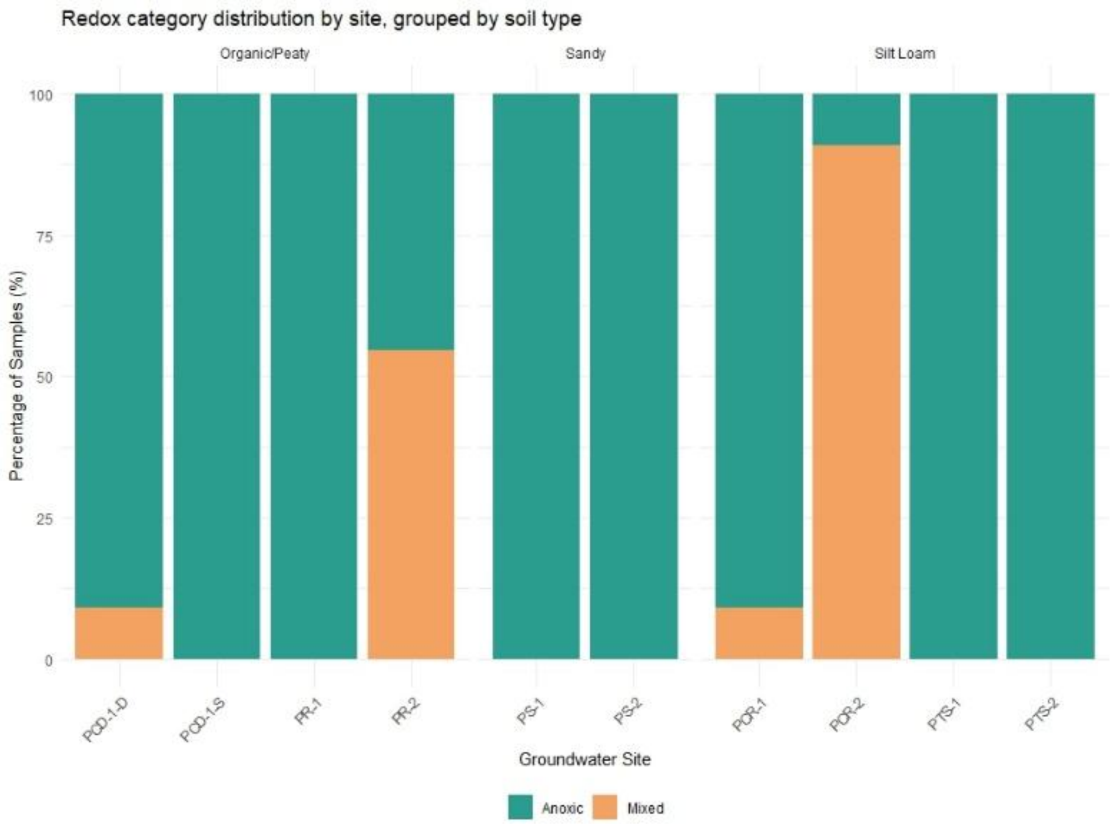

Figure 16: Redox status distribution across groundwater sampling sites, grouped by soil type.

Within the anoxic groundwater sites, manganese(IV) reduction emerges as the most prevalent redox process, occurring at 5 of the 7 anoxic locations (PCD-1-D, PCD-1-S, PCR-1, PS-1, PTS-1). These sites exhibit elevated manganese concentrations (Mn^{2+} means of 0.297 to 0.841 mg/L, all exceeding the 0.05 mg/L threshold) combined with depleted oxygen and nitrate, and iron concentrations below the 0.1 mg/L threshold. Sites PR-1 and PR-2 occasionally exhibited conditions approaching methanogenic status (extremely low sulfate, elevated dissolved organic carbon, deeply reducing), representing the most oxygen-depleted environments in the monitoring network.

The high proportion of anoxic classifications across diverse soil types and water table depths indicates that oxygen consumption through organic matter decomposition exceeds atmospheric oxygen input via rainfall infiltration or diffusion. Even sites with deeper water tables (PTS-1, PTS-2: mean 1.70-1.78 m bgl) maintain predominantly anoxic conditions, demonstrating that vertical separation from the surface is insufficient to promote aerobic conditions when organic matter content provides high oxygen demand.

Implications for nitrogen and phosphorus transformations

The overwhelming dominance of anoxic conditions (74% of samples) establishes the fundamental biogeochemical framework effecting nutrient behaviour in this system. Under oxygen-limited conditions, multiple nitrogen transformation processes combine to suppress nitrate formation and persistence:

- **Nitrification inhibition:** The conversion of ammonium to nitrate requires molecular oxygen as the electron acceptor. Under anoxic conditions (O_2 <0.5 mg/L), nitrifying bacteria cannot oxidise ammonium, allowing ammonium to persist in groundwater (McMahon & Chapelle, 2008).
- **Denitrification:** Any nitrate present or produced during transient oxidation events can be reduced to N_2 gas by denitrifying bacteria using nitrate as a terminal electron acceptor under anoxic conditions, effectively removing it from solution (Rivett et al., 2008).
- **Dissimilatory nitrate reduction to ammonium (DNRA):** Under deeply reducing conditions with high organic carbon availability, some microbial communities reduce nitrate to ammonium rather than N_2 gas (Friedl et al., 2018). This process both attenuates nitrate and regenerates ammonium.

For phosphorus, anoxic conditions promote mobilisation through reductive dissolution of iron oxyhydroxides. Under oxic conditions, iron exists as Fe^{3+} in solid-phase oxyhydroxides that sorb and sequester phosphorus. When iron reduction occurs under anoxic conditions, Fe^{3+} is reduced to soluble Fe^{2+}, releasing previously sorbed phosphorus to solution (Scalenghe et al., 2002). This process is particularly important in organic soils where iron reduction occurs extensively.

The redox framework established here provides mechanistic context for the nutrient patterns presented in Section 5.1.3, where ammonium dominates nitrogen speciation and dissolved reactive phosphorus concentrations are elevated, particularly in organic soil groundwaters maintaining persistently anoxic conditions.

5.1.2.3 Groundwater major cations and anions

The major ion chemistry of shallow groundwater provides critical insights into soil-water interactions, weathering processes, and the influence of organic matter decomposition on groundwater hydrochemistry. Six major dissolved constituents were measured across all groundwater sites: calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), chloride (Cl^-), sulfate (SO_4^{2-}), and alkalinity (reported as mg/L $CaCO_3$, representing the combined contribution of bicarbonate (HCO_3^-) and carbonate (CO_3^{2-})). It should be noted that sodium (Na^+) was not measured in this study, limiting the ability to construct complete cation balances. Nevertheless, the measured ions represent the dominant contributors to groundwater chemistry and provide robust characterisation of hydrochemical patterns across the three soil type groupings.

Major ion concentrations exhibited pronounced differences across soil types, with soil type groupings explaining 47% to 82% of the variance for most ions (Table 11, Appendix Table A7). Kruskal-Wallis tests revealed significant differences ($p<0.05$) across soil types for all major ions, as per Table 11.

Table 11: Kruskal-Wallis test comparing all major ions (meq/L) within groundwater samples across soil types.

Ion	H statistic	Degrees of freedom (df)	p value	Eta squared (η^2)	Effect size magnitude
Ca^{2+}	17.04	2	0.000199	0.14	large
Mg^{2+}	88.52	2	6.00E-20	0.82	large
K^+	84.70	2	4.06E-19	0.78	large
Cl^-	75.53	2	3.98E-17	0.69	large
SO_4^{2-}	7.46	2	0.024	0.05	small
HCO_3^- + CO_3^{2-}	52.38	2	4.22E-12	0.48	large

Magnesium and potassium

Magnesium and potassium concentrations in shallow groundwater demonstrated the strongest soil type associations, with groundwater at peaty soil sites exhibiting markedly elevated levels compared to both sandy and silt loam groundwater environments (Figure 17). The large effect sizes for magnesium (η^2=0.816) and potassium (η^2=0.780) indicate that soil type groupings explain 81.6% and 78.0% of the variance in these cations respectively, representing the most robust soil type control of any measured parameters. Potassium exhibited similar patterns, with groundwater at peaty soil sites showing median potassium concentrations of 21.5-27.7 mg/L (average 25.1 mg/L), silt loam sites 6.3-12.2 mg/L (average 10.6 mg/L), and sandy sites 6.0-6.3 mg/L (average 6.2 mg/L).

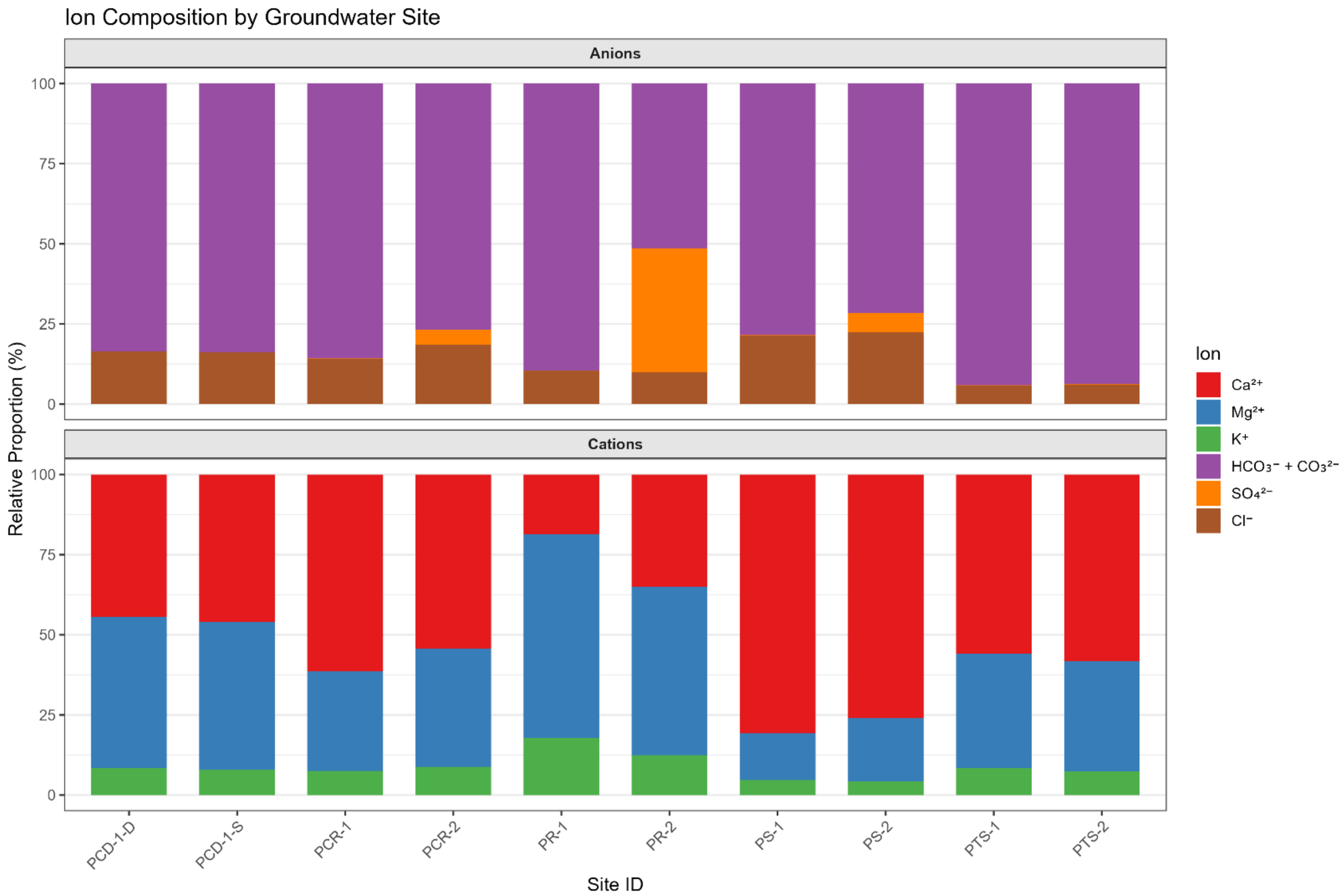

Figure 17: Relative ion proportions within groundwater across all Moutoa piezometer monitoring sites.

Dunn's post-hoc tests confirmed that groundwater at peaty soil sites differed highly significantly from both sandy soils (adjusted $p=4.34\times10^{-18}$) and silt loam soils (adjusted $p=1.03\times10^{-10}$) for magnesium, while sandy and silt loam sites also differed significantly from each other (adjusted $p=0.002526$) (Table 12). Potassium levels within shallow groundwater at peaty soil sites also differed highly significantly from both sandy (adjusted $p=2.66\times10^{-17}$) and silt loam (adjusted $p=2.26\times10^{-10}$), while sandy and silt loam also differed significantly (adjusted $p=0.00372$) (Table 12).

The elevation of both magnesium and potassium in organic soil groundwater reflects the cation exchange capacity (CEC), decomposition and mobilisation processes characteristic of organic matter decomposition. Organic soils typically possess very high CEC due to the abundance of negatively charged functional groups on organic matter surfaces, enabling substantial cation retention (Younger, 2007). However, under the artificial drainage regime operating in the Moutoa-Whirokino catchment, aerobic decomposition of soil organic matter releases these previously bound cations into solution.

Table 12: Dunn's post-hoc test showing pairwise comparisons for which groundwater samples differ from each other in relation to ion composition, when grouped by soil type.

Ion type	Group 1	Group 2	Sample Size 1	Sample Size 2	Adj. p-value	Adj. p-value significance level
Ca^{2+} (meq)	Sandy	Silt loam	22	43	0.000928	***
Ca^{2+} (meq)	Sandy	Peaty	22	44	1	ns
Ca^{2+} (meq)	Silt loam	Peaty	43	44	0.002702	**
Mg^{2+} (meq)	Sandy	Silt loam	22	43	0.002526	**
Mg^{2+} (meq)	Sandy	Peaty	22	44	4.34E-18	****
Mg^{2+} (meq)	Silt loam	Peaty	43	44	1.03E-10	****
K^{+} (meq)	Sandy	Silt loam	22	43	0.00372	**
K^{+} (meq)	Sandy	Peaty	22	44	2.66E-17	****
K^{+} (meq)	Silt loam	Peaty	43	44	2.26E-10	****
Cl^{-} (meq)	Sandy	Silt loam	22	43	0.001024	**
Cl^{-} (meq)	Sandy	Peaty	22	44	0.001195	**
Cl^{-} (meq)	Silt loam	Peaty	43	44	1.08E-17	****
SO_4^{2-} (meq)	Sandy	Silt loam	22	43	0.024366	*
SO_4^{2-} (meq)	Sandy	Peaty	22	44	0.071498	ns
SO_4^{2-} (meq)	Silt loam	Peaty	43	44	1	ns
HCO_3^{-} + CO_3^{2-} (meq)	Sandy	Silt loam	22	43	0.053699	ns
HCO_3^{-} + CO_3^{2-} (meq)	Sandy	Peaty	22	44	6.89E-11	****
HCO_3^{-} + CO_3^{2-} (meq)	Silt loam	Peaty	43	44	4.64E-07	****

This pattern aligns with international research on drained peatlands. Tiemeyer et al. (2007) documented elevated cation concentrations in drainage ditches from degraded peatland under intensive grassland use in Germany, attributing this to enhanced mineralisation following drainage. Van Beek et al. (2007) similarly reported that in intensively managed Dutch dairy grasslands on peat soils, a major portion of nutrient and cation loading to surface water originated from deeper peat layers undergoing mineralisation, rather than from contemporary surface applications. The mobilisation of magnesium and potassium from decomposing peat represents a fundamental alteration of biogeochemical cycling, converting these soils from long-term cation sinks to sources.

Calcium

Calcium concentrations within shallow groundwater displayed more complex spatial patterns compared to magnesium and potassium, with a moderate effect size (η^2=0.142) indicating that soil type only explains approximately 14% of calcium variability (Table 11). Groundwater calcium concentrations at sandy soil sites exhibited the highest median calcium concentrations overall, with PS-1 at 53.4 mg/L (IQR: 52.4-54.4 mg/L) and PS-2 at 59.1 mg/L (IQR: 55.2-61.8 mg/L), averaging 56.3 mg/L across both sandy sites. Silt loam soil sites showed moderate and variable calcium, and peaty soil sites exhibited the most striking within-group heterogeneity for calcium. The PCD-1 piezometers (both deep and shallow) showed very high calcium concentrations (median 75.0 mg/L and 73.8 mg/L respectively), exceeding the sandy soil sites concentrations, while the PR-1 and PR-2 piezometers in Makerua peaty silt loam displayed very low calcium concentrations in groundwater (median 14.0 mg/L and 12.8 mg/L respectively), representing the lowest calcium concentrations observed across the groundwater monitoring network.

Chloride

Chloride concentrations in groundwater showed large soil type effects (η^2=0.694), with peaty soil sites generally having elevated chloride concentrations (average 45.9 mg/L across peaty sites) compared to silt loam soil sites (average 16.3 mg/L across all sites). Sandy soil sites displayed intermediate groundwater chloride levels (average 32.4 mg/L). Dunn's post-hoc tests confirmed that peaty soils differed highly significantly from both sandy soils (adjusted p=0.001195) and silt loam soils (adjusted $p=1.08\times10^{-17}$), with peaty sites consistently showing elevated chloride.

Sulfate

Sulfate concentrations within groundwater were generally very low across all sites, with an overall average of 1.5 mg/L and the majority of samples at or near the reported result limits of 0.19 mg/L. Notable exceptions to this general sulfate depletion included PS-2 (median 7.16 mg/L, IQR: 2.09-16.50 mg/L) and PCR-2 (median 6.47 mg/L, IQR: 2.10-7.91 mg/L), both of which are shallower piezometers that also exhibited elevated dissolved oxygen. The co-occurrence of higher sulfate and higher dissolved oxygen at these specific sites suggests a connection, as sulfate is the terminal electron acceptor in microbial respiration under strongly reducing conditions, with sulfate reduction converting SO_4^{2-} to sulfide (S^{2-}) under anoxia.

This widespread sulfate depletion in groundwater across the catchment provides additional evidence of strongly reducing conditions. Under anaerobic conditions with dissolved oxygen below approximately 0.5 mg/L, sulfate becomes an electron acceptor for microbial respiration following the sequential utilisation of electron acceptors ($O_2 \rightarrow NO_3^- \rightarrow Mn^{4+} \rightarrow Fe^{3+} \rightarrow SO_4^{2-}$) (McMahon & Chapelle, 2008). The near-absence of sulfate indicates that sulfate reduction is occurring at most sites, placing them on the strongly reducing end of the redox spectrum. This is consistent with the previously-discussed anoxic classifications and has important implications for nitrogen transformations, as sulfate-reducing conditions favour DNRA over denitrification (Friedl et al., 2018).

Alkalinity

Alkalinity concentration (as $CaCO_3$) within groundwater showed large soil type effects (η^2=0.475), with peaty soil sites exhibiting dramatically elevated values (medians 480-530 mg/L) compared to sandy (medians 180-200 mg/L) and silt loam (medians 110-310 mg/L) sites. This elevation reflects enhanced CO_2 production from microbial decomposition of organic matter (35% in Makerua peaty silt loam; Cowie, 1978). The CO_2 reacts with water to generate bicarbonate and carbonate, creating the high alkalinity documented in organic soil groundwater.

The elevated alkalinity in groundwater at peaty soil sites provides substantial buffering capacity against pH changes, helping to maintain the near-neutral to slightly alkaline pH (median 7.35-7.68) observed in these groundwaters, despite the production of organic acids during peat decomposition. This buffering is particularly important in drained peatlands where decomposition processes might otherwise lead to acidification (Younger, 2007).

Integrated interpretation

The major ion patterns reveal that organic soil groundwater possesses fundamentally different hydrochemical characteristics compared to mineral and sandy soil environments, driven primarily by organic matter decomposition under reducing conditions. The same processes mobilising Mg^{2+} and K^+ also release NH_4^+ and DRP (examined in Section 5.1.3), creating an integrated hydrochemical-nutrient signature characteristic of drained peatland systems. The elevated alkalinity, consistently low dissolved oxygen, and widespread sulfate depletion collectively establish conditions favouring ammonium persistence and phosphorus mobilisation, as demonstrated quantitatively in subsequent sections.

5.1.2.4 Groundwater hydrochemical classification

Multivariate statistical analysis provides systematic classification of groundwater samples into distinct hydrochemical types, revealing how geological setting and biogeochemical processes create characteristic groundwater chemistries influencing nutrient behaviour in the subsurface environment.

Cluster analysis and principal component analysis

Cluster analysis performed on major ion percentage data (expressed on meq/L basis) identified five distinct groundwater types (Appendix Table A8, Table 13, Figure 18). Principal Component Analysis effectively separated these clusters, with the first two dimensions explaining 88.3% of total variance (Dim1: 56.1%, Dim2: 32.2%). This high explanatory power indicates that major ion chemistry is controlled by limited dominant processes, primarily related to parent material dissolution, organic matter decomposition, and redox-associated transformations.

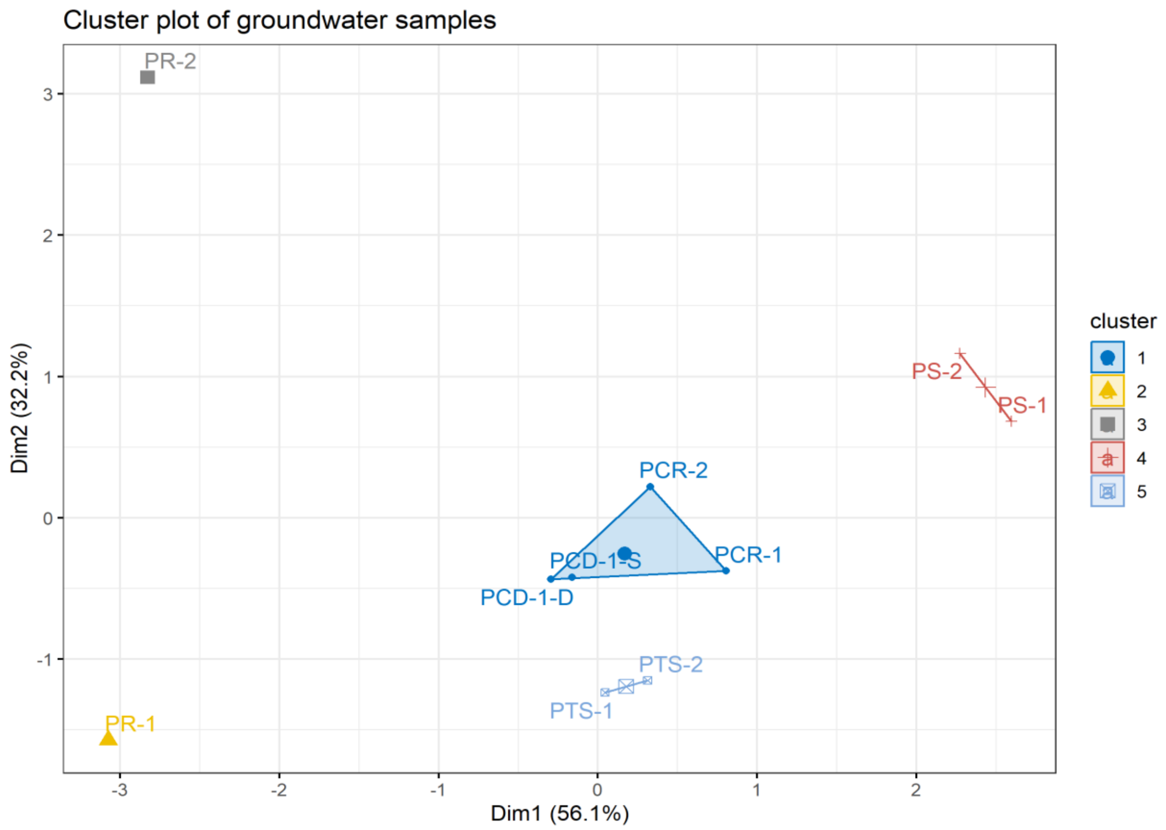

Figure 18: Principal component analysis showing clusters of groundwater samples based on ion composition at these sites.

Hydrochemical facies identification

Based on the cluster analysis and following established hydrochemical classification principles (Cherry & Freeze, 1979; Younger, 2007), five distinct water types are identified:

1. **Cluster 1 - Ca-rich mixed / HCO_3^- dominant water**: This most common water type, encompassing both organic-rich (Kairanga Peaty Silt Loam) and mineral (Kairanga Silt Loam) alluvial soils, shows balanced cation composition (Ca^{2+} 51.50%, Mg^{2+} 48.50%) with strong bicarbonate dominance (91.94%). The approximate 1:1 Ca:Mg ratio suggests mixed carbonate sources, consistent with diverse river alluvium deposits. This signature is typical of groundwater in Quaternary alluvial systems in New Zealand lowland settings (Rosen & White, 2001).

2. **Cluster 2 - Mixed cations / HCO_3^- dominant water**: Characterised by pronounced magnesium enrichment (81.41% Mg^{2+}, 18.59% Ca^{2+}) combined with the highest bicarbonate proportion (95.41%) and elevated K^+ (17.92%). This unique chemistry, found in the deep piezometer in Makerua peaty silt loam, reflects intensive organic matter mineralisation. Van Beek et al. (2007) documented similar Mg-enriched groundwater in Dutch peat grasslands, attributing it to preferential desorption of exchangeable Mg^{2+} from decomposing organic matter. The elevated K^+ further supports active organic matter decomposition as the primary control on ion composition.

Table 13: Groundwater hydrochemical characteristics and classification, based on dominant ions per cluster and linked to soil type.

Cluster	1	2	3	4	5
Sites	PCD-1-D, PCD-1-S, PCR-1, PCR-2	PR-1	PR-2	PS-1, PS-2	PTS-1, PTS-2
Ca^{2+} (% mean)	51.50	18.59	35.01	78.36	57.12
Mg^{2+} (% mean)	48.50	81.41	64.99	21.64	42.88
Cl^- (% mean)	7.48	4.55	6.46	10.52	2.52
SO_4^{2-} (% mean)	0.58	0.04	35.80	1.65	0.04
HCO_3^- + CO_3^{2-} (% mean)	91.94	95.41	57.74	87.84	97.43
Dominant Soil	Organic/Peaty	Organic/Peaty	Organic/Peaty	Sandy	Silt Loam
Dominant Cation	Ca-rich mixed	Mixed cations	Mixed cations	Ca dominant	Ca-rich mixed
Dominant Anion	HCO_3^- + CO_3^{2-} dominant	HCO_3^- + CO_3^{2-} dominant	HCO_3^- + CO_3^{2-} -rich mixed	HCO_3^- + CO_3^{2-} dominant	HCO_3^- + CO_3^{2-} dominant
Water Type Name	Ca-rich mixed / HCO_3^- + CO_3^{2-} dominant	Mixed cations / HCO_3^- + CO_3^{2-} dominant	Mixed cations / HCO_3^- + CO_3^{2-} -rich mixed	Ca dominant / HCO_3^- + CO_3^{2-} dominant	Ca-rich mixed / HCO_3^- + CO_3^{2-} dominant

3. **Cluster 3 - Mixed cations / HCO_3^--rich mixed water**: Despite sharing the same soil type as PR-1, this shallow piezometer displays distinctly different chemistry, most notably sulfate enrichment (35.80% of anions) coupled with reduced bicarbonate (57.74%). This pattern suggests oscillating redox conditions. The persistence of measurable sulfate alongside reduced bicarbonate indicates incomplete sulfate reduction or periodic oxidation events, a pattern with important implications for nitrogen transformations, as sulfate-reducing conditions favour DNRA over denitrification (Friedl et al., 2018).

4. **Cluster 4 - Ca dominant / HCO_3^- dominant water**: The sandy soil sites display the most calcium-dominant chemistry (Ca^{2+} 78.36%, Mg^{2+} 21.64%) paired with high bicarbonate (87.84%). This composition reflects the simpler mineralogy of wind-blown coastal sands (Clement et al., 2010), with rapid recharge and limited water-rock interaction time (Younger, 2007).

5. **Cluster 5 - Ca-rich mixed / HCO_3^- dominant water**: Similar to Cluster 1 in cation composition but showing the highest bicarbonate dominance (97.43%)

and lowest chloride (2.52%) in the catchment, suggesting minimal marine or contamination influence and potentially greater inland recharge contribution.

The Piper diagram (Figure 19) provides integrated visualisation of these relationships, plotting median values of major cations and anions in groundwater across the sampling period.

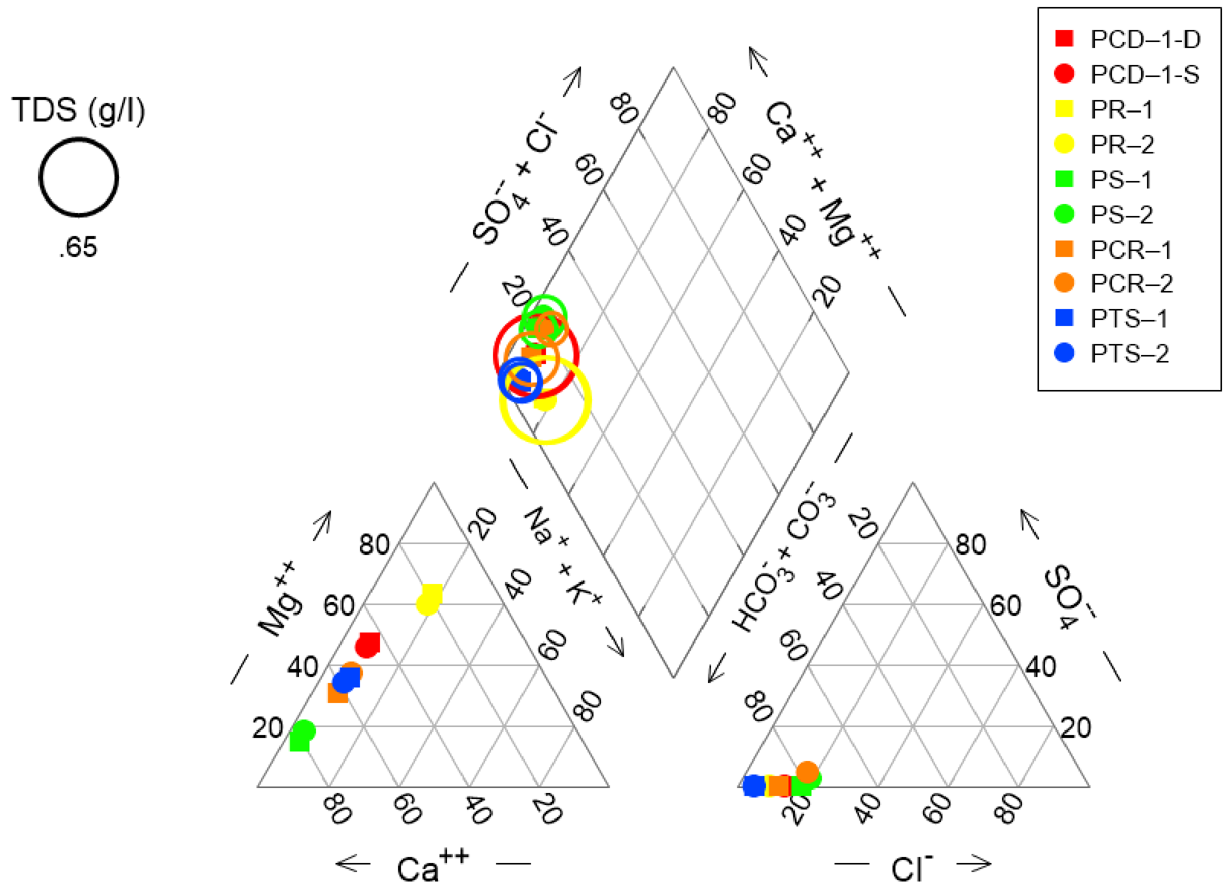

Figure 19: Piper diagram with median results across the sampling period from all groundwater sites. Note that sodium (Na+) was not sampled during this study.

The Piper diagram clearly illustrates cation variability (left triangle) contrasting with relative anion uniformity (right triangle), where nearly all sites cluster near the bicarbonate apex. In the central diamond, most sites occupy calcium-bicarbonate or mixed Ca-Mg-HCO_3 fields, characteristic of fresh groundwater in alluvial systems.

Vertical stratification observations

Paired piezometers at most sites fell within the same cluster (PCD-1-S and PCD-1-D: Cluster 1; PS-1 and PS-2: Cluster 4; PTS-1 and PTS-2: Cluster 5), suggesting vertical similarity in major ion composition over the sampled depth range (1.0-5.5 m bgl). This implies relatively uniform parent materials and transformation processes through the shallow groundwater zone.

The exception occurs at the PR site, where PR-1 (deeper, screen from 4.0-5.0 m bgl) and PR-2 (shallower, screen from 1.0-4.0 m bgl) occupy completely different clusters. The deeper piezometer PR-1 shows the highest Mg-enrichment and K^+ elevation in the dataset, while the shallow PR-2 displays unique sulfate enrichment. This depth stratification suggests fundamentally different redox regimes operating at different depths within the same soil profile. The deeper zones may experience more consistently reducing conditions favouring organic matter decomposition and cation mobilisation, while shallower zones may undergo redox oscillations that drive sulfur cycling. The deeper PR-1 had all 11 groundwater samples classified as 'Anoxic', while the shallower PR-2 had the most variability in the 'Mixed (oxic-anoxic)' category with 5 samples being 'Anoxic' and 6 'Mixed'.

Implications for nutrient dynamics

The identified hydrochemical facies create distinct biogeochemical environments with varying capacities for nutrient transformation and retention. Organic soil groundwaters (Clusters 2 and 3) exhibit elevated Mg^{2+} and K^+ signatures indicating active organic matter mineralisation, which is the same process that releases organic nitrogen and phosphorus (examined in Section 5.1.3).

The high alkalinity across all water types (85-97% bicarbonate dominance) indicates substantial buffering capacity. Van Beek et al. (2007) demonstrated in Dutch peat grasslands that high groundwater alkalinity (300-800 mg/L HCO_3^-) contributed near-neutral pH (6.5-7.5) that reduced phosphorus sorption capacity and enhanced DRP mobility. The similarly high alkalinity in Moutoa-Whirokino groundwater likely influences phosphorus behaviour, particularly in the low anion storage capacity of organic soils (McDowell & Monaghan, 2015).

Sandy soil groundwaters (Cluster 4) display simple Ca-HCO_3 chemistry with low K^+ (4.33-4.70%) and absence of sulfate enrichment, indicating rapid drainage, limited organic matter content (~7% versus ~35% in peaty soils; Cowie & Smith, 1958; Cowie, 1978), and shorter residence times. Therefore, these sites likely serve as useful reference conditions representing minimal organic matter influence on nutrient dynamics.

River alluvium groundwaters (Clusters 1 and 5) occupy an intermediate biogeochemical position, with balanced Ca-Mg compositions and moderate organic matter content (4.6-6.3%; Cowie, 1978) suggesting mixed controls, being carbonate dissolution, clay weathering, and moderate organic matter decomposition. Their nutrient transformation capacities likely fall between the extremes represented by organic and sandy soils.

Limitations

An important limitation of this classification is the absence of sodium (Na^+) measurements in the analytical suite. Sodium is often the dominant cation in marine-influenced or older groundwater systems (Younger, 2007), and its exclusion prevents definitive classification using standard international schemes (Cherry & Freeze, 1979; Hem, 1985). The classification presented here therefore characterises the "non-sodium cation signature" rather than complete hydrochemistry. Given the catchment's position on a coastal floodplain developed on marine sediments (Clement et al., 2010), sodium may contribute significantly to total cation content, particularly in deeper or older groundwater. However, for the shallow systems sampled here (1.0-5.5 m depth), the measured cation suite likely captures the primary controls on groundwater chemistry, as sodium typically increases with groundwater age and depth (Hiscock & Bense, 2014).

Despite this limitation, the identified hydrochemical facies provide valuable context for interpreting the redox conditions and nutrient concentrations, as major ion chemistry both reflects and influences the biogeochemical processes controlling nitrogen and phosphorus transformations in this artificially drained wetland landscape.

5.1.3 Groundwater nutrient dynamics and mobilisation

Having established the hydrological context (Section 5.1.1), redox conditions (Section 5.1.2.2), and hydrochemical framework (Sections 5.1.2.3-4), this section examines nutrient concentrations in shallow groundwater. The analysis begins by establishing mechanistic controls, then explores spatial, vertical, and temporal patterns demonstrating how these mechanisms manifest in the field data. It is worth noting that concentration-based analysis without comprehensive flow measurements prevents definitive load calculations in this study.

5.1.3.1 Mechanistic framework: Controls on nitrogen and phosphorus mobilisation

Three interconnected mechanisms drive nutrient mobilisation in the Moutoa-Whirokino catchment, operating simultaneously with relative contributions varying by location, depth, and season. These mechanisms, discussed extensively in the Literature Review, are established here as the framework for interpreting observed nutrient patterns.

Mechanism 1: Organic matter decomposition and nitrogen mineralisation

In Moutoa's peaty soils containing up to 35% organic matter (Cowie, 1978), microbial decomposition releases both inorganic nitrogen and phosphorus. This aligns with documented processes in drained organic soils internationally, where aerobic conditions within peaty soils follow water table drawdown and artificial drainage, and decomposition rates accelerate leading to increases in nitrogen mineralisation (Tiemeyer et al., 2007; Van Beek et al., 2007; Becher et al., 2023), producing CO_2, NH_4^+, and organic acids. As detailed in the Literature Review, microbial decomposition of organic nitrogen in soil organic matter releases plant-available inorganic nitrogen, particularly ammonium, when microbial carbon and energy requirements are met (Hart et al., 1994; Robertson & Groffman, 2024). The CO_2 reacts with water to generate bicarbonate and carbonate, creating the elevated alkalinity documented in Section 5.1.2.3. This same process mobilises organic nitrogen through mineralisation, converting organic-N to NH_4^+.

This interpretation is strongly supported by the hydrochemical analysis, which demonstrated that peaty soil groundwater exhibits elevated magnesium (up to 81.4% of cations at PR-1) and potassium (up to 17.9% at PR-1), which are both clear indicators of organic matter decomposition and cation mobilisation (Hem, 1985; Appelo & Postma, 2005).

Drainage-enhanced decomposition is a key change from natural peatland conditions. Under natural permanently saturated conditions, slow anaerobic decomposition preserves organic matter. Artificial drainage lowering water tables introduces periodic oxygen availability to previously saturated peat, which can accelerate decomposition rates and nutrient release (Van Beek et al., 2007). This creates ongoing mobilisation of nutrients accumulated over centuries of peat formation.

Mechanism 2: Redox association with nitrogen speciation

In typical aerobic agricultural soils, mineralisation-derived ammonium is rapidly oxidised to nitrate through nitrification, creating the nitrate-dominated leaching patterns documented across New Zealand dairy farming systems (Monaghan et al., 2016; Ledgard et al., 1999). Under the predominantly anoxic conditions documented in Section 5.1.2.2, nitrogen transformations strongly favour ammonium persistence over nitrate formation or accumulation. Multiple processes combine to create ammonium-dominated systems:

- **Nitrification suppression:** Nitrifying bacteria require molecular oxygen (O_2) to oxidise NH_4^+ to NO_3^-. Under anoxic conditions (O_2 <0.5 mg/L documented at most sites), nitrification cannot proceed, allowing mineralisation-derived ammonium to persist in groundwater. Boonman et al. (2024) demonstrated that in Dutch agricultural peatlands, oxygen penetration remains severely limited even after drainage due to rapid oxygen consumption during organic matter decomposition. Under such conditions, the ammonium-oxidising bacteria and archaea responsible for nitrification, which are obligate aerobes requiring DO > 0.5 mg/L (Robertson & Groffman, 2024), cannot function.
- **Denitrification:** Any nitrate present or produced during transient oxidation events can be rapidly reduced to N_2 gas by denitrifying bacteria using NO_3^- as terminal electron acceptor under oxygen-limited conditions. This attenuates nitrate from solution, further suppressing nitrate accumulation. The anoxic classification of most groundwater samples indicates conditions where denitrification actively attenuates nitrate, converting it to gaseous N_2 or N_2O (Robertson & Groffman, 2024; Wolters et al., 2022).
- **Dissimilatory nitrate reduction to ammonium (DNRA):** Under deeply reducing conditions with high organic carbon availability (characteristic of many organic soil sites), some microbial communities reduce nitrate to ammonium rather than N_2 gas (Friedl et al., 2018). The sulfate-depleted conditions documented at most sites (Section 5.1.2.3) may favour DNRA over denitrification, meaning any nitrate formed or infiltrating may be converted back to ammonium rather than lost as N_2 gas. Friedl et al. (2018) found that DNRA exceeded denitrification rates in subtropical Australian pasture soils upon rewetting, with soil redox potential (not carbon to nitrate ratio) having the dominant associated effect.

The net result is groundwater dominated by NH_4^+ with minimal NO_3^-, contrasting with well-aerated mineral soil systems where nitrate typically dominates.

Mechanism 3: Phosphorus mobilisation through multiple pathways

Organic soils exhibit three characteristics promoting phosphorus release:

1. **Low anion storage capacity:** Organic soils in New Zealand typically have very low anion storage capacity (ASC <20%; McDowell & Monaghan, 2015), limiting phosphorus sorption to soil particles. This contrasts with mineral soils containing iron and aluminium oxides that strongly sorb phosphate. The result is enhanced phosphorus mobility in drainage waters from organic soils, with dissolved phosphorus losses from low ASC organic soils ranging from 1.7 to 87 kg P/ha, representing up to 89% of applied fertiliser phosphorus (McDowell & Monaghan, 2015).
2. **Organic phosphorus mineralisation:** The same decomposition processes releasing NH_4^+ also mobilise organic phosphorus. Sapek et al. (2021) found that in Polish meadow complexes on peat soils, phosphate-P and ammonia-N concentrations in groundwater were significantly positively correlated ($r=0.67$, $p<0.001$), indicating peat mineralisation was a common driver for both nutrients. The magnitude of phosphorus stored in organic soil profiles means ongoing mineralisation can sustain elevated DRP concentrations for decades. Vermaat and Hellmann (2010) found Dutch peat polders under agricultural use containing 1400 kg P/ha in the upper 50 cm compared to 700 kg P/ha in nature reserves after over 50 years of dairy farming. This illustrates legacy phosphorus reservoirs continuing to be mobilised through drainage-enhanced decomposition even with reduced contemporary fertiliser applications.
3. **Reductive dissolution of iron-phosphorus complexes:** Under anoxic conditions (Section 5.1.2.2), iron oxyhydroxides undergo reductive dissolution, releasing previously bound phosphorus (Scalenghe et al., 2002; Herndon et al., 2019). Herndon et al. (2019) found that iron oxyhydroxides sequester approximately 46% of inorganic phosphorus in organic soils. When Fe^{3+} is reduced to soluble Fe^{2+} under anoxic conditions, this phosphorus is released to solution. Van Beek et al. (2004) documented that in Dutch dairy grassland on peat soils, 33-82% of phosphorus loading originated from nutrient-rich deeper peat layers undergoing mineralisation rather than surface fertiliser applications. The fluctuating water tables characteristic of artificially drained peatlands (Section 5.1.1) create repeated oxidation-reduction cycles that can progressively deplete iron-bound phosphorus pools, with P release rates in rewetted agricultural peat potentially orders of magnitude higher than natural peatlands (Zak et al., 2008).

Integrated mechanistic framework

These three mechanisms operate simultaneously and interact. Organic matter decomposition provides the substrate (organic N and P) and creates the reducing conditions (through oxygen consumption) that both suppress nitrification and promote reductive P release. The result is coupled NH_4^+-DRP mobilisation under persistently anoxic conditions, with spatial patterns controlled by soil organic matter content, temporal patterns influenced by water table dynamics, and vertical patterns reflecting oxygen availability gradients. The following sections examine how this mechanistic framework manifests in observed nutrient concentrations.

5.1.3.2 Site-specific groundwater nutrient concentrations

Groundwater nutrient analysis spanning 11 months across ten monitoring sites revealed systematic spatial patterns controlled by soil type, with dissolved nitrogen occurring predominantly as ammonia-N rather than nitrate-N throughout the catchment (Appendix Table A9, Figures 20-23).

Ammonium dominance

Soil type association exerted strong control on ammonia-N concentrations in groundwater (Kruskal-Wallis H=51.58, df=2, p=6.29x10-12, η^2=0.468, large effect size). The groundwater at peaty soil sites (PCD-1-D, PCD-1-S, PR-1, PR-2) had elevated nutrient concentrations compared to sandy and silt loam sites. Groundwater ammonia-N medians ranged from 6.79 g/m^3 (PR-2) to 18.30 g/m^3 (PCD-1-D) in peaty soil sites, compared to just 1.47-1.54 g/m^3 in sandy soils (PS-1, PS-2) and 1.17-9.45 g/m^3 in silt loam soils (PCR-1, PCR-2, PTS-1, PTS-2) (Figure 20). This is up to a 12-fold difference in ammonium concentrations within shallow groundwater between soil types. Post-hoc tests confirmed ammonia-N concentrations at peaty soil sites differed significantly from both sandy (adjusted p=1.09×10^{-11}) and silt loam (adjusted p=2.29×10^{-5}) sites, while sandy and silt loam sites also showed a significant difference, but at a lower significance level (adjusted p=0.003) (Appendix Table A10).

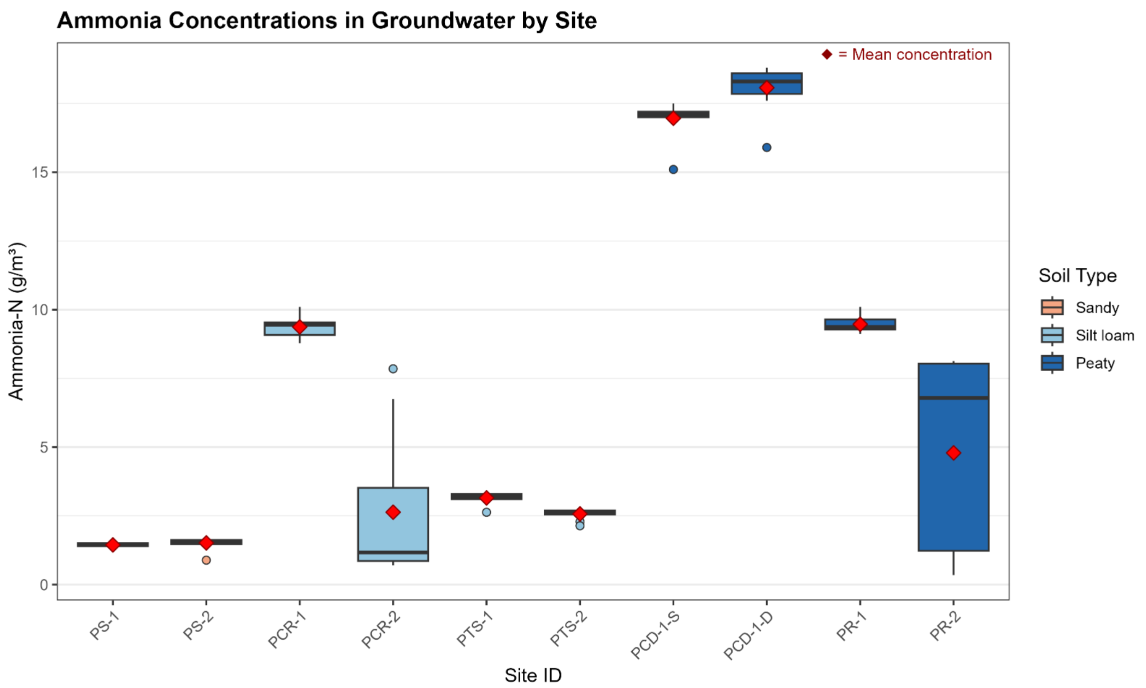

Figure 20: Ammonia-N concentrations in groundwater by site and soil type.

This near-absence of nitrate, coupled with elevated ammonium, represents a nitrogen speciation pattern markedly different from the nitrate-dominated losses typically documented in well-drained agricultural systems (Monaghan et al., 2016; Di & Cameron, 2002).

These spatial patterns directly reflect the mechanistic framework established in Section 5.1.3.1. Peaty soils with up to 35% organic matter content provide substantial substrate for mineralisation-derived NH_4^+ production, while maintaining the persistently anoxic conditions (Section 5.1.2.2) that suppress nitrification. Sandy soils with only ~7% organic matter (Cowie & Smith, 1958) produce less mineralisation-derived ammonium and exhibit more variable redox conditions. The high variability of groundwater ammonia-N within the silt loam category likely reflects site-specific differences in organic matter content, drainage effectiveness, and proximity to organic soil inclusions.

Nitrate-nitrogen

Nitrate-N in groundwater remained uniformly low across all sites (median <0.020 g/m^3, Figure 21), with only PCR-2 showing the occasional detectable variation (IQR 0.020-0.075 g/m^3). PR-2 also had a single large nitrate-N reading of 5.71g/m^3 in June 2024, and a subsequent reading of 0.516g/m^3 in July, giving this site a nitrate-N mean of 0.582g/m^3. This near-complete absence of nitrate despite substantial total inorganic nitrogen availability suggests the effectiveness of anoxic conditions in suppressing nitrification or rapidly attenuating any nitrate formed through denitrification or DNRA.

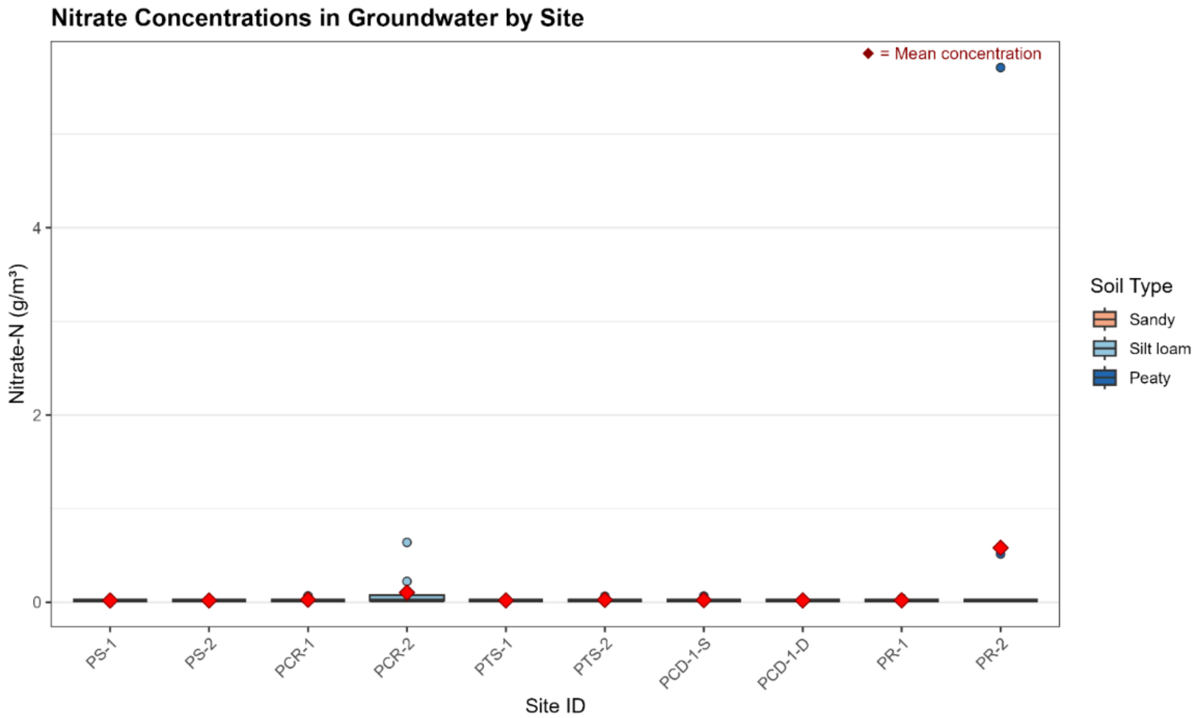

Figure 21: Nitrate-N concentrations in groundwater by site.

The stark contrast between high ammonium and negligible nitrate demonstrates the fundamental importance of the redox status association with nitrogen speciation in this system. This nitrogen speciation pattern is markedly different from the nitrate-dominated losses typically documented in well-drained agricultural systems. In typical aerobic agricultural soils, mineralisation-derived ammonium is rapidly oxidised to nitrate through nitrification, creating the nitrate-dominated leaching patterns documented across New Zealand dairy farming systems (Monaghan et al., 2016; Di & Cameron, 2002; Ledgard et al., 1999). Under aerobic conditions typical of well-drained mineral soils, nitrate typically comprises >80% of dissolved nitrogen exports (Monaghan et al., 2016). The pattern is somewhat reversed in the Moutoa-Whirokino catchment, with ammonium comprising most of inorganic nitrogen which reflects the persistent oxygen limitation documented in Section 5.1.2.2.

Phosphorus patterns

Dissolved reactive phosphorus (DRP) exhibited spatial patterns similar to ammonia-N, with strong soil type control ($H=57.27$, $df=2$, $p=3.67\times10^{-13}$, $\eta^2=0.521$, large effect) (Figure 22). Peaty soils exhibited the highest DRP levels (medians 3.15-4.56 g/m^3), followed by silt loams (0.09-2.61 g/m^3) and sandy soils (0.14-0.18 g/m^3). The PR-2 site showed particularly high variability (IQR 0.06-5.16 g/m^3), suggesting dynamic phosphorus mobilisation processes at this location. These elevated DRP concentrations in peaty groundwater indicate that drainage-enhanced mineralisation is mobilising both nitrogen and phosphorus simultaneously from the organic soil reservoir. The parallel spatial patterns of ammonia-N and DRP (both elevated in peaty soil groundwater) support the coupled-release mechanism described in Section 5.1.3.1, where organic matter decomposition simultaneously mobilises both nutrients. This coupling is quantified in Section 5.1.3.3 through correlation analysis.

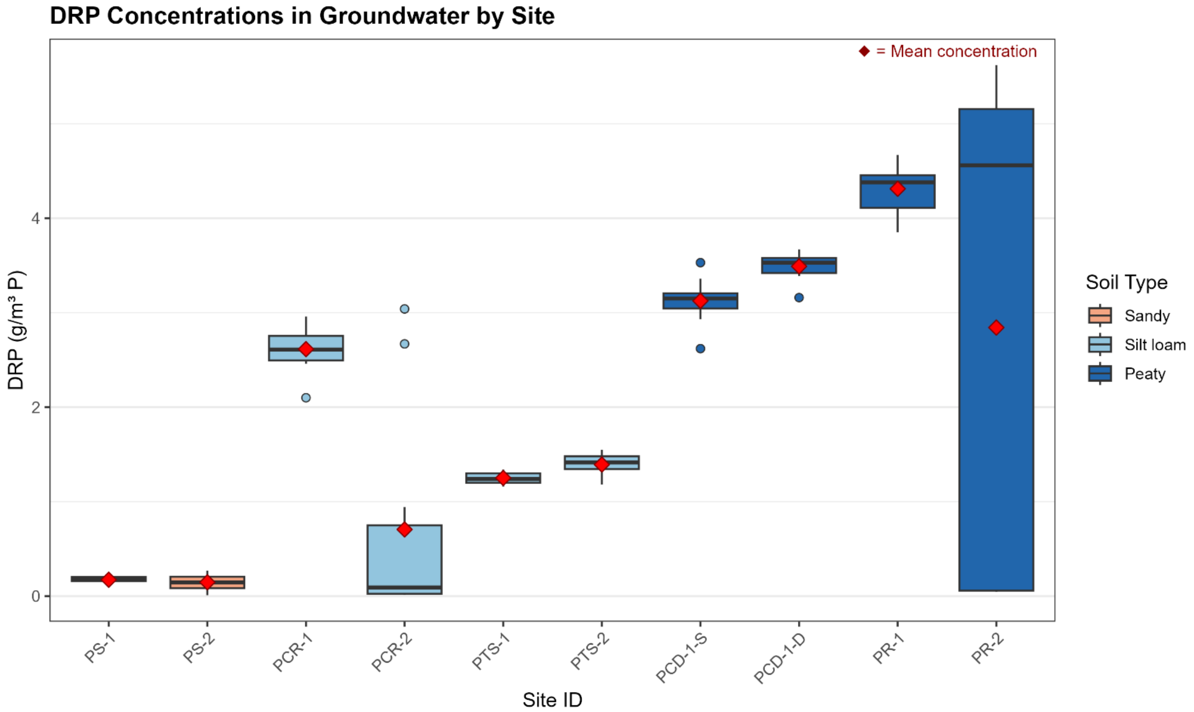

Figure 22: Dissolved reactive phosphorus concentrations in groundwater by site and soil type.

Dissolved organic carbon patterns

DOC concentrations showed similar soil type controls ($H=65.07$, $df=2$, $p=7.43\times10^{-15}$, $\eta^2=0.595$, large effect) (Figure 23), reinforcing the interpretation that organic matter decomposition is the common driver. Peaty soils showed substantially higher DOC values (medians 11.1-21.1 g/m^3) than sandy (5.0-5.3 g/m^3) or silt loam soils (3.6-6.8 g/m^3). The PR-2 site exhibited exceptional DOC concentrations with a median of 21.1 g/m^3 and an upper IQR reaching 41.5 g/m^3, which indicates extreme organic matter decomposition is occurring at this location. This site also showed the anoxic to methanogenic conditions documented in Section 5.1.2.2, suggesting the most intensely reducing environment in the monitoring network. The elevated DOC in organic soil groundwater reflects active decomposition of the substantial organic matter pool, with DOC serving as both an indicator of decomposition activity and a substrate for microbial respiration that consumes oxygen and maintains reducing conditions.

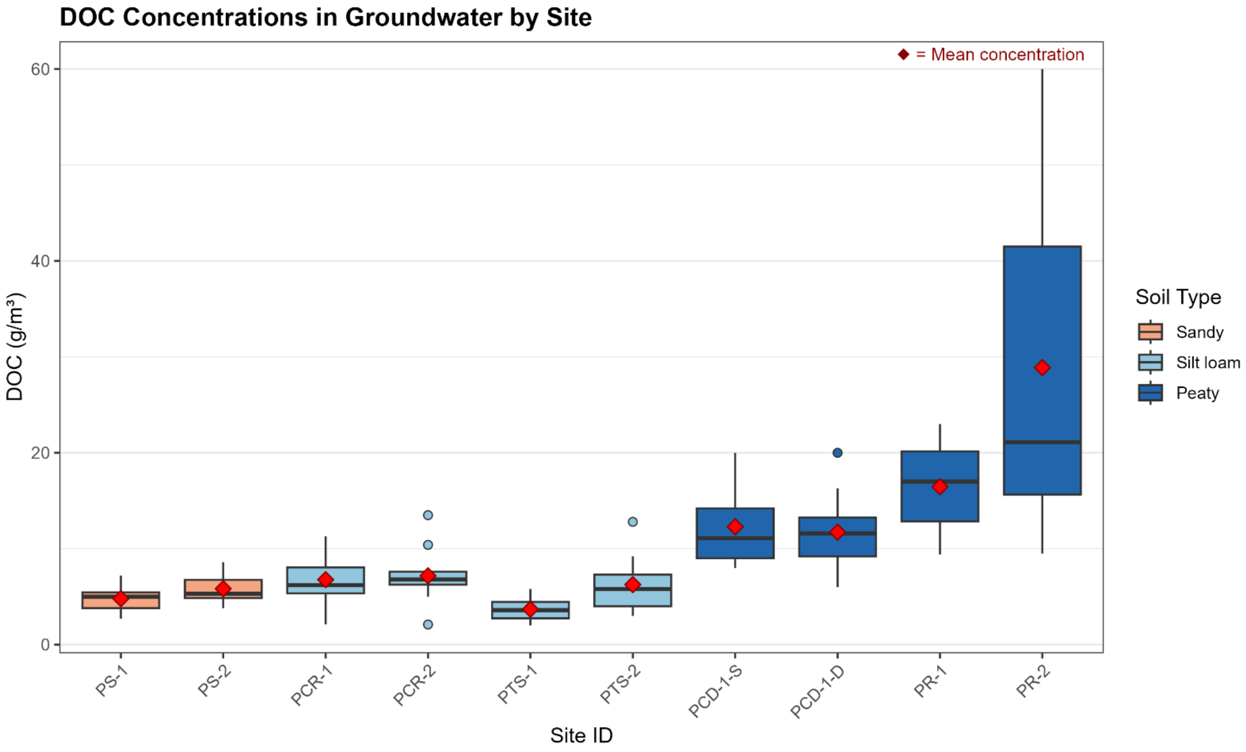

Figure 23: Dissolved organic carbon concentrations in groundwater by site and soil type.

The pattern observed at Moutoa-Whirokino, with elevated ammonium and negligible nitrate, is a different nitrogen regime from well-drained agricultural soils. Rather than the "mineralisation → nitrification → nitrate leaching" pathway typical of aerobic systems, the Moutoa system exhibits "mineralisation → ammonium accumulation → direct ammonium transport" in anaerobic groundwater, with DNRA potentially providing a feedback mechanism that reinforces ammonium dominance (Barkle et al., 2021).

Soil type as the primary control on groundwater nutrient concentrations

Grouping results by soil type clarifies the systematic differences in groundwater nutrient concentrations between the three main soil groups and quantifies the magnitude of soil type effects (Table 14).

Shallow groundwater piezometers in peaty soils exhibited median concentrations of ammonia-N, DRP, and DOC 8.5, 20.6, and 2.8 times higher respectively than the piezometers in sandy soils (Table 14). Silt loam soils showed intermediate values for most parameters (ammonia-N 3.12 g/m^3, DRP 1.34 g/m^3, DOC 5.8 g/m^3), though with considerable within-group variability reflecting the variation of alluvial parent materials and varying degrees of organic matter accumulation across the Kairanga and Parewanui soil series. Kruskal-Wallis tests revealed significant differences across soil types for

DOC, SIN, ammonia-N, and DRP (Table 18). The large Eta-squared values (0.47-0.60) indicate that soil type accounts for 47-60% of the variability in these critical nutrient parameters, demonstrating that parent material and organic matter content are the dominant controls on shallow groundwater chemistry in this catchment.

Table 14: Median groundwater nutrient concentrations, by soil type at piezometer location.

Soil Type	Number of samples	DOC median (g/m^3)	SIN median (g/m^3)	Ammonia-N median (g/m^3)	Nitrite-N median (g/m^3)	Nitrate-N median (g/m^3)	DRP median (g/m^3)
Peaty	44	13.9	12.6	12.6	<0.020	<0.020	3.53
Sandy	22	5.05	1.53	1.49	<0.020	<0.020	0.171
Silt loam	43	5.8	3.21	3.12	<0.020	<0.020	1.34

The near-identical medians for SIN and ammonia-N within each soil type (differing by only 0.01-0.08 g/m^3) confirm that ammonia-N comprises essentially all the soluble inorganic nitrogen. This is in stark contrast to typical New Zealand agricultural drainage studies, where nitrate typically dominates dissolved inorganic nitrogen. For example, Monaghan et al. (2000, 2016) documented that subsurface drainage from intensively grazed dairy pastures in Southland carried primarily nitrate, accounting for approximately 80% of total soluble inorganic nitrogen discharged, with mean annual losses of 30-56 kg N/ha dominated by the nitrate form.

Table 15: Kruskal-Wallis test comparing groundwater nutrients across all soil types.

Parameter	H statistic	Degrees of freedom	p value	Eta squared (η^2)	Effect size magnitude
DOC (g/m^3)	65.06706	2	7.43E-15	0.594972	large
SIN (g/m^3 N)	57.22326	2	3.75E-13	0.520974	large
Ammonia-N (g/m^3)	51.58303	2	6.29E-12	0.467764	large
Nitrite-N (g/m^3)	1.477273	2	0.478	-0.00493	small
Nitrate-N (g/m^3)	4.724673	2	0.0942	0.025704	small
DRP (g/m^3 P)	57.26701	2	3.67E-13	0.521387	large

In contrast, nitrite-N and nitrate-N showed no significant differences across soil types, with negligible effect sizes (Table 15). The consistent absence of nitrate across most samples from all soil types, regardless of drainage characteristics or land use intensity, is particularly noteworthy and reinforces the interpretation that oxygen limitation and redox processes, rather than nitrogen loading per se, effect the nitrogen speciation in groundwater at the study sites.

5.1.3.3 Groundwater nutrients and redox control correlations

Correlation analysis quantifies relationships between nutrients and redox-sensitive parameters, providing statistical support for the mechanistic framework established in Section 5.1.3.1.

Spearman's correlation analysis revealed strong relationships between nutrient parameters and indicators of organic matter content and redox status (Figure 24). DOC showed strong positive correlations with sum of inorganic nitrogen (ρ=0.45), ammonia-N (ρ=0.46), and particularly DRP (ρ=0.91), confirming that dissolved organic carbon as an indicator of organic matter decomposition is closely linked to nutrient mobilisation. The exceptionally strong DOC-DRP correlation (ρ=0.91) suggests organic phosphorus mineralisation is tightly coupled to overall organic matter decomposition, consistent with findings from Polish peat systems (Sapek et al., 2021). It is also of note that both DOC and DRP independently peak in peaty soils, therefore soil type may be a driver of this correlation rather than a direct causal DOC–DRP link.

Ammonia-N showed a strong positive correlation with DRP (ρ=0.84), supporting the coupled-release interpretation where both nutrients are co-mobilised during organic matter decomposition. This quantifies the parallel spatial patterns documented in Section 5.1.3.2.

Grouping samples by their McMahon and Chapelle (2008) redox classification revealed differences in nutrient concentrations between anoxic and mixed redox categories (Table 16), providing support that redox status is associated with nutrient behaviour.

Table 16: Redox categories of samples grouped by Mixed or Anoxic, with some key nutrient statistics for each category.

Redox Category	Mixed	Anoxic
N samples	28	81
Ammonia-N mean (mg/L)	3.808	8.156
Ammonia-N median (mg/L)	2.645	8.93
Ammonia-N Inter-Quartile Range (IQR) (mg/L)	1.79 - 4.66	1.59 - 15.1
Nitrate-N mean (mg/L)	0.276	0.021
Nitrate-N median (mg/L)	<0.020	<0.020
Nitrate-N IQR (mg/L)	0.020 - 0.027	<0.020
DRP mean (mg/L)	1.615	2.148
DRP median (mg/L)	1.35	2.56
DRP IQR (mg/L)	0.1 - 1.83	0.21 - 3.49

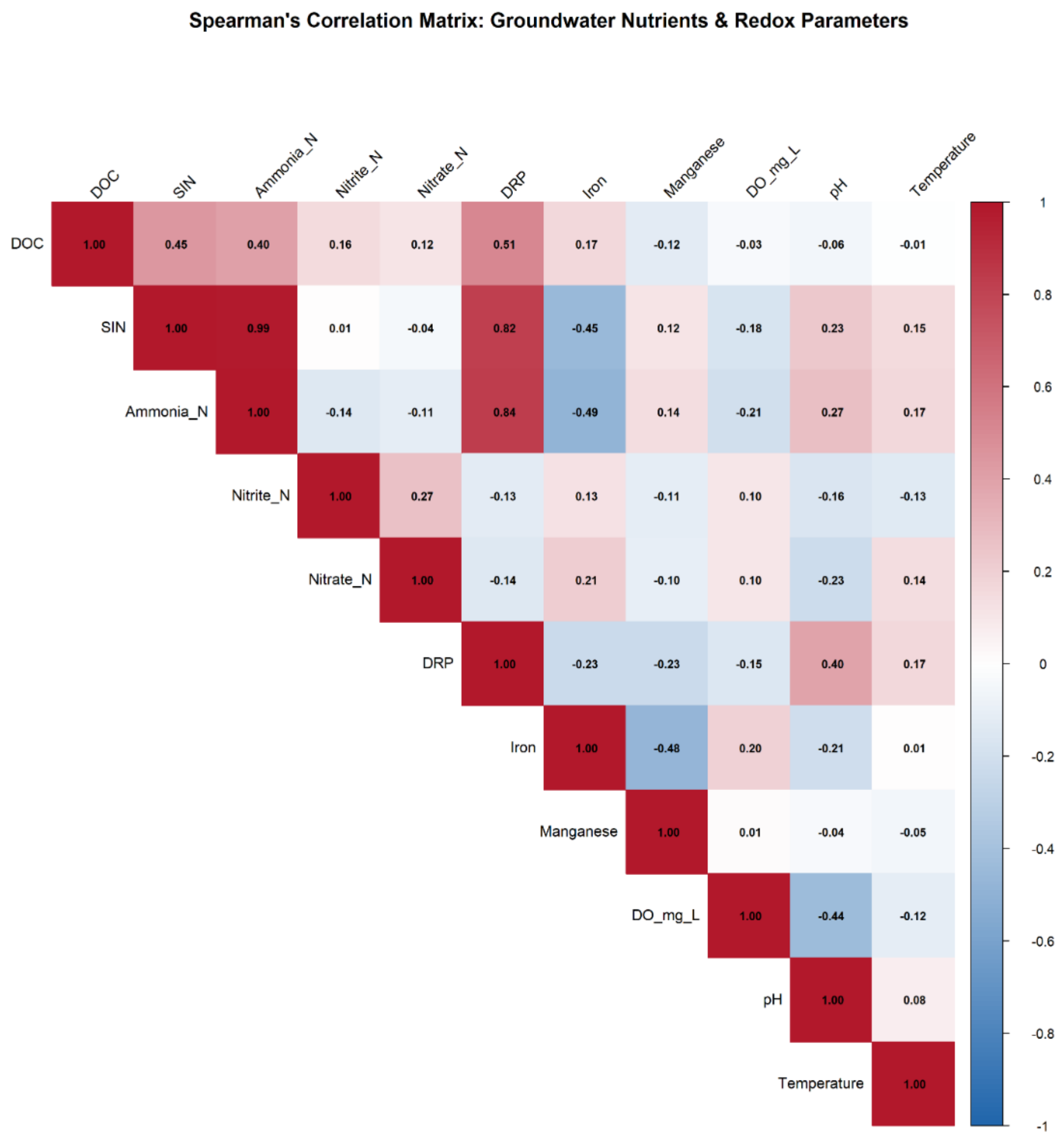

Figure 24: Spearman's correlation matrix for groundwater nutrients and redox parameters.

Anoxic samples showed substantially higher ammonia-N (mean 8.16 g/m^3, median 8.93 g/m^3) compared to mixed redox (mean 3.81 g/m^3, median 2.65 g/m^3), representing 2.1-fold difference in means and 3.4-fold difference in medians. The wider interquartile range for ammonia-N measured in the anoxic samples (1.59-15.1 g/m^3) versus mixed (1.79-4.66 g/m^3) indicates both higher central tendency and greater variability under reducing conditions.

Groundwater nitrate-N concentrations were minimal in both categories but showed differences consistent with redox theory. Mixed samples had mean 0.28 g/m^3 (though median remained at or near reported result limits, <0.020g/m^3), reflecting occasional detectable nitrate at PCR-2, while anoxic samples averaged only 0.02 g/m^3. The near-universal absence of nitrate in anoxic samples suggests effective suppression of nitrification and/or rapid attenuation through denitrification or DNRA under oxygen-limited conditions.

Groundwater DRP was elevated in anoxic samples (mean 2.15 g/m^3, median 2.56 g/m^3) compared to mixed (mean 1.61 g/m^3, median 1.35 g/m^3), with broader interquartile range (0.21-3.49 g/m^3 versus 0.10-1.83 g/m^3). This pattern is consistent with enhanced phosphorus release under reducing conditions through reductive dissolution of iron oxyhydroxides (Scalenghe et al., 2002), as established in the mechanistic framework (Section 5.1.3.1).

Integration with hydrochemistry

The correlation between ammonia-N and major ions in groundwater provides additional mechanistic insight. The elevated TDS and specific cation patterns (Mg^{2+}, K^+) in organic soil groundwater documented in Sections 5.1.2.1 and 5.1.2.3 reflect the same organic matter decomposition and reducing conditions generating ammonia-N and DRP in groundwater. This integrated hydrochemical-nutrient signature characterises drained peatland groundwater, where high oxygen demand from organic matter decomposition creates reducing conditions, suppresses nitrification, promotes reductive dissolution, and mobilises cations, all at the same time.

This correlation analysis confirms the mechanistic framework operates as predicted, where organic matter decomposition (indexed by DOC) drives coupled NH_4^+-DRP release, while redox conditions (anoxic versus mixed) effect the magnitude of nutrient accumulation and the near-complete suppression of nitrate-N.

5.1.3.4 Depth gradients in groundwater nutrients

Paired shallow (screen from 1.0-4.0 m bgl) and deep (screen from 4.0-5.0 m bgl) piezometers at five sites enable assessment of vertical nutrient stratification. Understanding depth patterns is important because oxygen penetration may be limited by diffusion and consumption through organic matter decomposition (Boonman et al., 2024).

Kassim and Yaacob (2019) documented in Malaysian tropical peatlands that different nutrients also respond distinctly to water table changes, with both nitrate-N and ammonia-N concentrations maximised when water tables were maintained at 40 cm below ground surface. In the Moutoa-Whirokino system, sites maintaining water tables in the 0.3-0.5 m depth range (CD sites, PCD sites, JJ1) are very similar to these conditions for nutrient mobilisation throughout much of the year.

Depth comparison results

Comparison of groundwater chemistry between shallow and deep piezometers revealed consistent directional patterns across the catchment, though statistical testing indicated these differences were not significant at individual paired sites (Figure 25). Deep piezometers had higher mean concentrations of ammonia-N (8.32 g/m^3) compared to shallow piezometers (5.74 g/m^3), a 45% increase at depth (Table 17). Nitrate-N concentrations were higher in shallow groundwater (mean 0.15 g/m^3) than deep (0.02 g/m^3). DRP followed a similar pattern to ammonia-N, with deep samples averaging 2.36 g/m^3 compared to 1.65 g/m^3 in shallow samples (43% increase at depth).

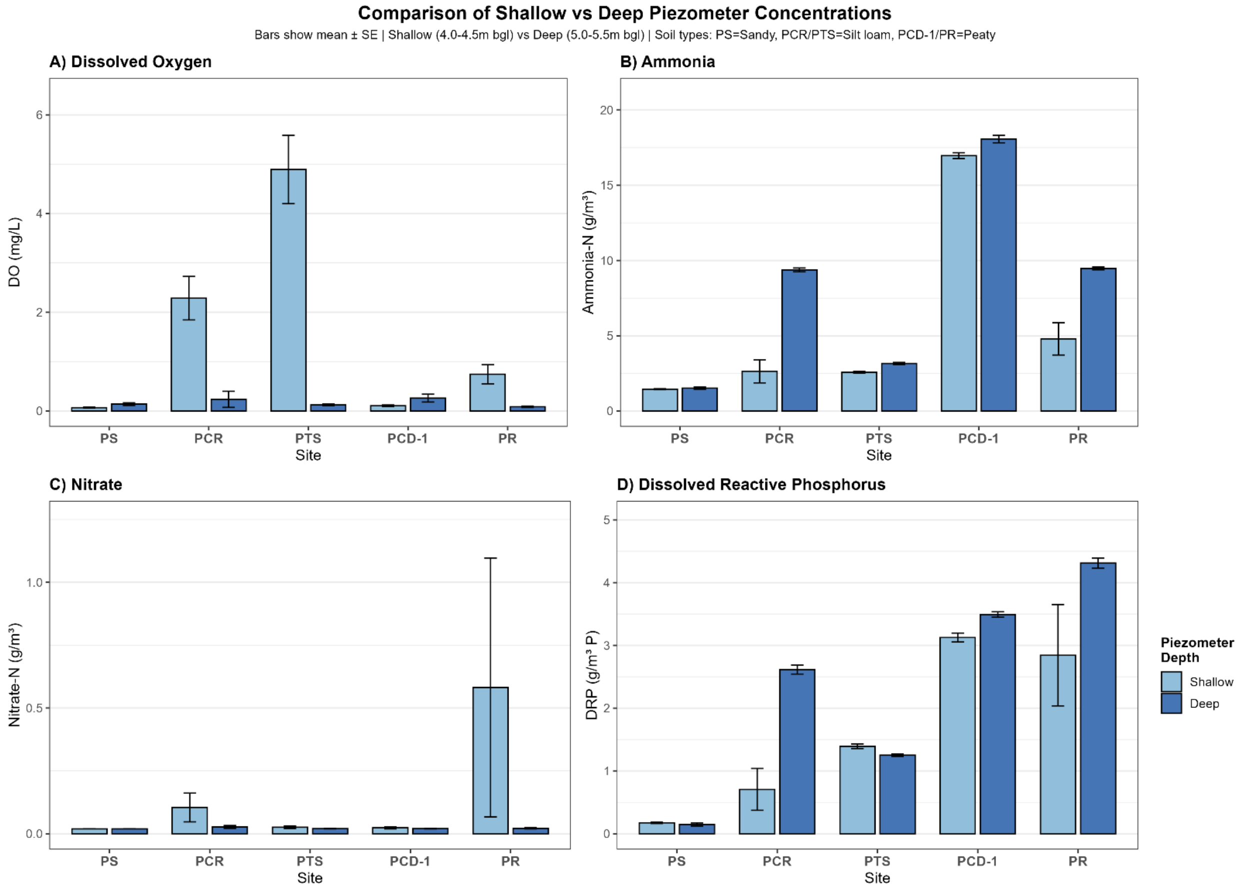

Figure 25: Comparison of key nutrients in groundwater samples - shallow versus deep piezometer comparison.

The most pronounced depth-related difference in groundwater was dissolved oxygen, where shallow piezometers averaged 1.56 mg/L compared to only 0.17 mg/L in deep piezometers (Table 17). This difference in dissolved oxygen between depths supports the understanding of nitrogen speciation patterns at the groundwater sites. This oxygen gradient supports the nitrogen speciation patterns, where deeper zones with minimal oxygen maintain higher ammonia-N through suppressed nitrification, while shallow zones with occasionally elevated oxygen show slightly higher (though still very low) nitrate-N concentrations.

Iron showed substantially higher concentrations in shallow samples (mean 0.70 g/m^3) than deep (0.12 g/m^3), with much higher variability (SD 1.23 versus 0.16 g/m^3). This likely reflects fluctuating redox conditions near the surface where alternating wet-dry cycles promote iron oxide dissolution during reduction and precipitation during oxidation. Manganese concentrations were nearly identical between depths (mean 0.41-0.44 g/m^3), indicating minimal depth stratification in manganese cycling.

Table 17: Summary statistics for groundwater samples, differentiating by shallow versus deep piezometers.

Parameter	Deep piezometer	Shallow piezometer
n samples	55	54
Temp mean	15.59	15.63
Temp SD	0.78	1.18
DO mean	0.17	1.56
DO SD	0.27	2.13
pH mean	7.40	7.04
pH SD	0.31	0.67
DOC mean	8.90	11.98
DOC SD	5.53	12.00
Ammonia-N mean	8.32	5.74
Ammonia-N SD	5.91	6.15
Nitrate-N mean	0.02	0.15
Nitrate-N SD	0.01	0.78
DRP mean	2.36	1.65
DRP SD	1.53	1.73
Fe mean	0.12	0.70
Fe SD	0.16	1.23
Mn mean	0.41	0.44
Mn SD	0.26	0.18

Statistical assessment

Paired statistical tests found no significant differences between shallow and deep piezometers at individual sites ($p>0.10$ for all parameters, Appendix Table A11). The lack of statistical significance despite clear directional trends reflects the dominance of horizontal spatial variability (particularly soil type effects documented in Section 5.1.3.2) over vertical stratification within this relatively shallow monitoring depth range (1.0-5.0 m bgl).

The absence of site-by-site significance should not overshadow the biological and biogeochemical importance of observed patterns. When all shallow samples are compared against all deep samples across the catchment (n=54-55 per depth category), directional trends emerge showing deep zones as more reducing (lower DO, higher NH_4^+ and DRP) and shallow zones as more variable (higher DO variability, occasional NO_3^-, fluctuating Fe).

Mechanistic interpretation

The depth patterns support the mechanistic framework (Section 5.1.3.1) where oxygen availability controls nitrogen speciation. Deeper groundwater zones remain buffered from atmospheric oxygen input, maintaining persistently anoxic conditions favourable for ammonia-N accumulation and phosphorus mobilisation. Shallow zones experience greater coupling to surface-driven processes including rainfall infiltration carrying dissolved oxygen, creating more variable redox conditions that effect occasional nitrification (explaining higher mean nitrate in shallow samples) while still maintaining overall oxygen limitation.

The elevated DOC in shallow groundwater samples (mean 11.98 g/m^3 versus 8.90 g/m^3 at depth) suggests more active organic matter decomposition in shallower zones where water table fluctuations create repeated oxidation-reduction cycles. This interpretation is consistent with the conceptual model where reductions in groundwater levels during summer exposes organic horizons primarily in the shallow zone, promoting decomposition, while deeper zones remain continuously saturated.

5.1.3.5 Temporal variation in groundwater nutrient concentrations

Temporal analysis of nutrient concentrations reveals distinct patterns controlled by depth, with deep piezometers exhibiting remarkable stability while shallow piezometers show greater variability and site-specific events (Figures 26-27).

Deep piezometer stability

Deep piezometers across all sites maintained temporal stability in ammonia-N, nitrate-N, and DRP concentrations throughout the 11-month monitoring period (Figure 26). PCD-1-D sustained consistently elevated ammonia-N at 17-20 g/m^3, PCR-1 and PR-1 both maintained stable concentrations at 9-10 g/m^3, while PS-1 and PTS-1 showed low but steady concentrations at 1-3 g/m^3. These deep zone concentrations remained stable despite water table fluctuations exceeding 1m at sites PR-1 and PTS-1 (from approximately 2.5 m to 1.5 m bgl documented in Section 5.1.1).

This stability reflects continuous organic matter mineralisation and persistent anoxic conditions in deeper groundwater zones, buffered from surface influences. The substantial organic matter pool (up to 35% in peaty soils; Cowie, 1978) provides ongoing substrate for decomposition-derived ammonium production, while deep zone oxygen limitation (mean DO 0.17 mg/L, Section 5.1.3.4) prevents nitrification regardless of water table position.

Shallow piezometer variability

In contrast, shallow piezometers exhibited greater temporal variability and site-specific trends (Figure 27). PCR-2 showed declining ammonia-N from approximately 8-9 g/m^3 in January 2024 to 4-5 g/m^3 by September 2024, suggesting either dilution from increased recharge or enhanced flushing. PCD-1-S maintained elevated ammonia-N throughout (16-19 g/m^3), demonstrating that shallow zones in highly organic soils can sustain high concentrations comparable to deep zones.

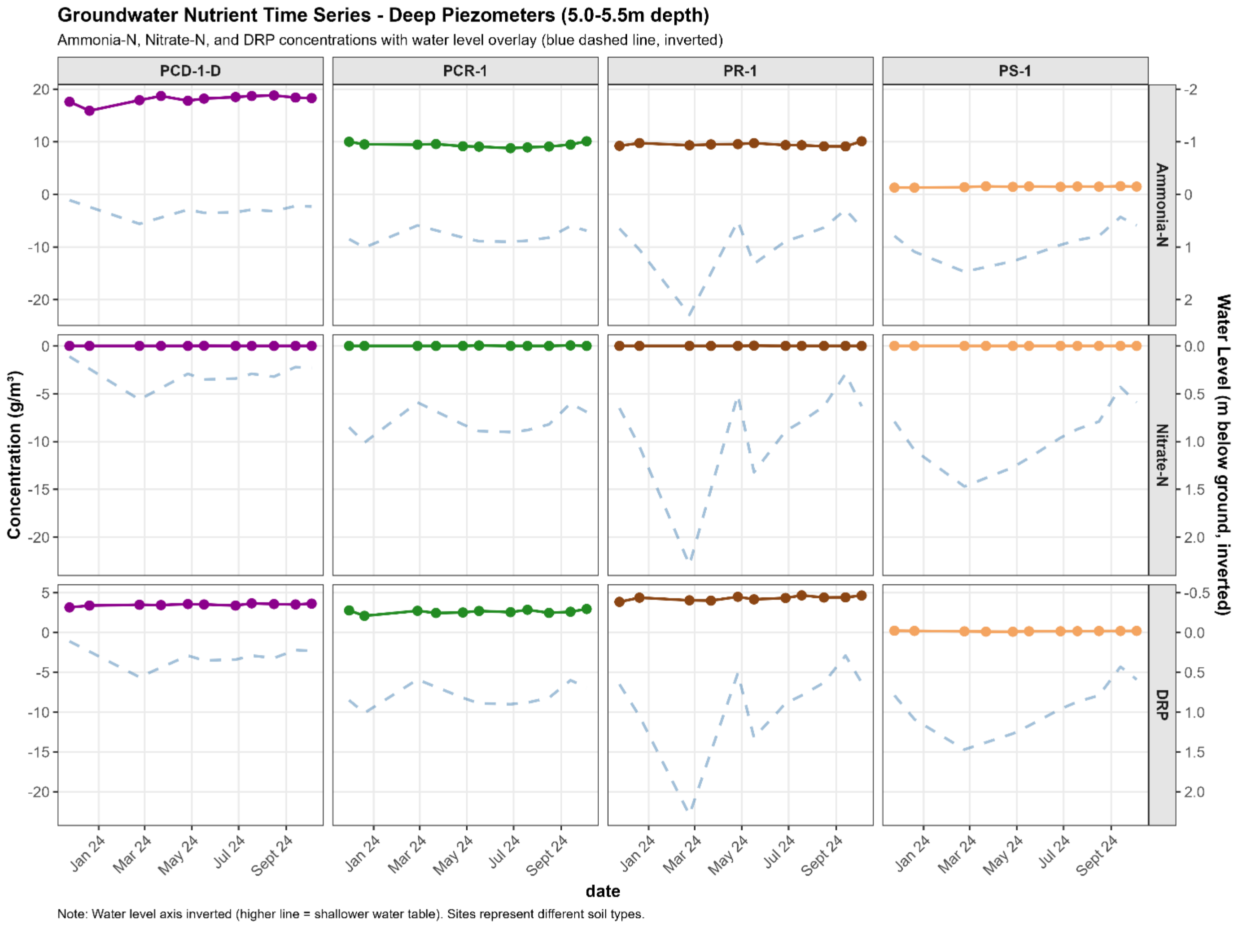

Figure 26: Groundwater nutrient time series at deep piezometer sites, with groundwater level depth shown on secondary axis.

Nitrate-N concentrations in groundwater remained consistently below 0.5 g/m^3 across both depths throughout the monitoring period, with one notable exception discussed below. This persistent nitrate-N limitation confirms the effectiveness of anoxic conditions in suppressing nitrification system-wide.

The PR-2 nutrient crash event

The most significant temporal dynamic was the June-July 2024 nutrient crash at PR-2 shallow (Figure 27). Ammonia-N maintained at 9-10 g/m^3 from November 2023 through May 2024 before declining quickly to near 0 g/m^3 around June-July 2024, with only brief recovery to 2-3 g/m^3 before final decline to near 0 g/m^3 by September. This crash was mirrored in DRP, declining from 2-3 g/m^3 to near 0 g/m^3 over the same period.

Concurrent with the ammonia-N and DRP crash, PR-2 exhibited a transient nitrate-N spike to approximately 5.71 g/m^3 (first sample) and 0.516 g/m^3 (second sample) during June-July 2024, representing the only substantial nitrate-N detection in the entire monitoring dataset. This unique nitrate appearance coincided with water table recovery from approximately 2.5 m to 1.0 m bgl (Section 5.1.1) during the period of rainfall in May-June 2024.

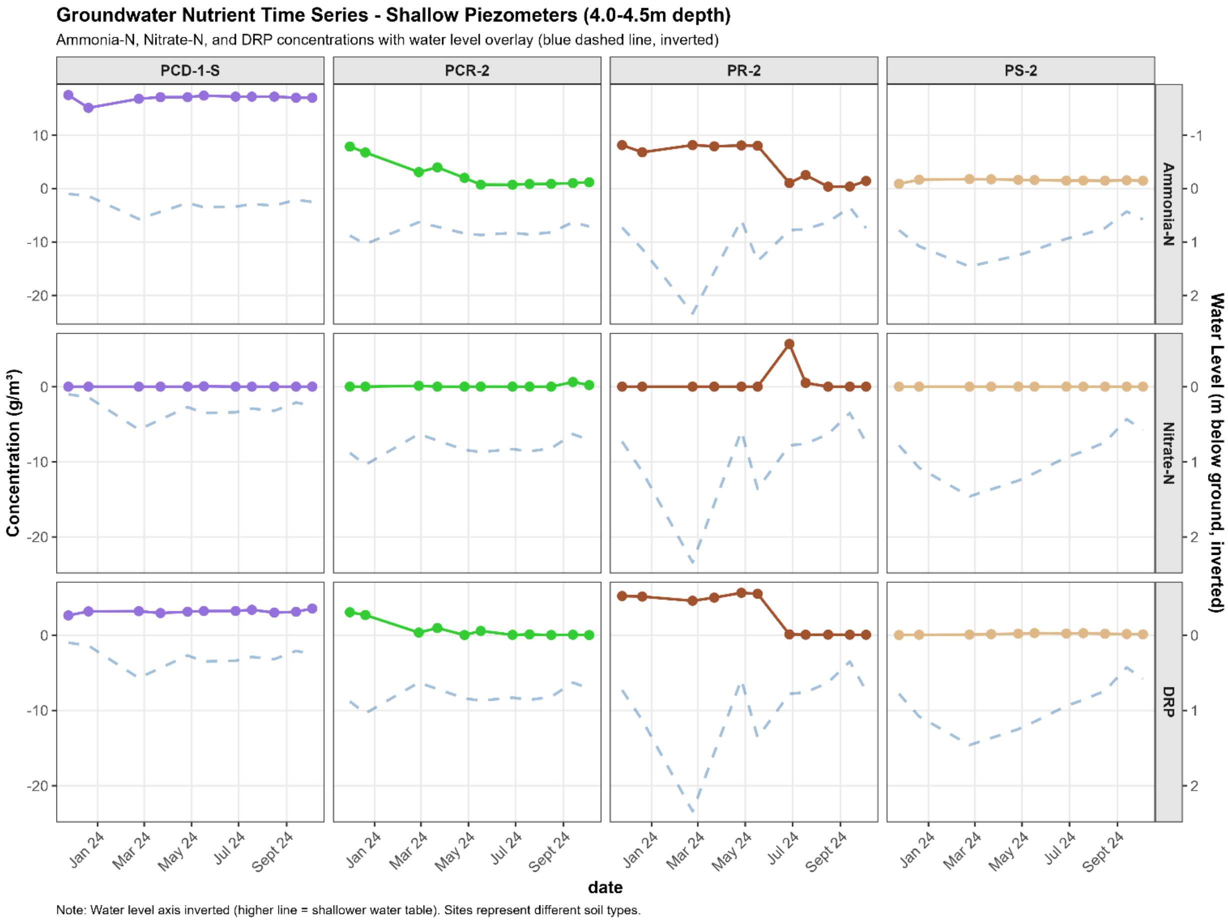

Figure 27: Groundwater nutrient time series at shallow piezometer sites, with groundwater level depth shown on secondary axis.

The co-occurrence of nutrient decline and nitrate-N appearance suggests rapid flushing or dilution of the shallow groundwater layer during water table rise. Possible mechanisms include: (1) displacement of nutrient-rich shallow groundwater by infiltrating rainfall with lower nutrient concentrations, (2) enhanced lateral flow during water table recovery, or (3) temporary oxidation of shallow groundwater during the infiltration event, permitting transient nitrification before conditions returned to anoxic status.

Critically, this event was site-specific. The paired deep piezometer PR-1 maintained stable ammonia-N (9-10 g/m^3) and DRP (2-3 g/m^3) throughout the same period despite experiencing identical water table fluctuations. This depth-specific response reinforces that shallow groundwater zones experience greater coupling to hydrological dynamics and surface-driven processes, while deep zones remain buffered by stable conditions and continuous nutrient supply from ongoing organic matter mineralisation.

The PR-2 event demonstrates that while nutrients generally show temporal stability (particularly at depth), specific combinations of hydrological conditions and site characteristics can produce dramatic short-term variations in shallow groundwater. However, the rarity of this event (occurring at only one site once during 11 months monitoring across ten sites) indicates that temporal stability rather than variability characterises nutrient concentrations in this system.

5.1.3.6 Comparison of soil water and groundwater nutrient concentrations

Soil water sampling via suction cups (60 cm depth) was implemented to investigate nutrient dynamics in the shallow vadose zone. While only one successful sampling round was achieved (16 November 2024), the data provide insights into vertical nitrogen speciation gradients across the soil-groundwater continuum.

Nitrogen speciation gradients through the soil profile

The comparison of nitrogen speciation between soil water (60 cm depth) and groundwater (1.0-5.0 m depth) has shown a stark contrast in the dominant inorganic nitrogen forms (Figure 28). Soil water showed nitrate-N dominance (1.09-4.49 g/m^3) with relatively low ammonia-N (0.166-0.531 g/m^3). Groundwater exhibited the inverse pattern, with ammonia-N strongly dominating (1.13-18.7 g/m^3) and nitrate-N at trace levels (0.029-0.107 g/m^3).

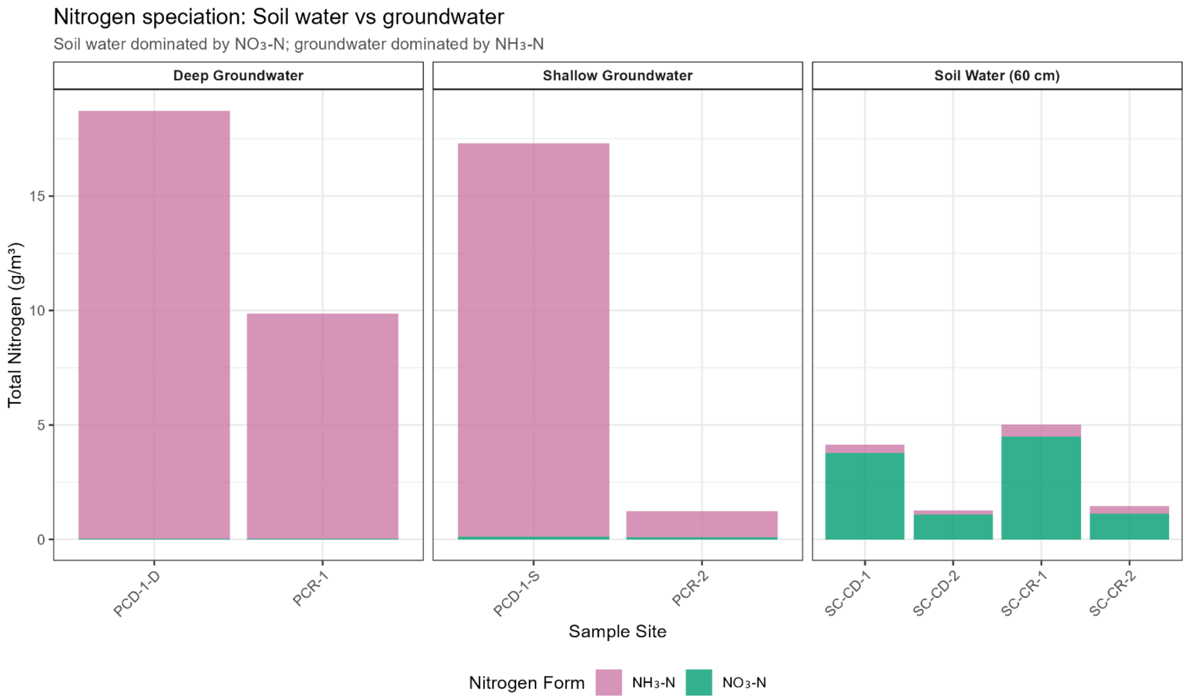

Figure 28: Stacked bar chart showing absolute concentrations of both Nitrate-N and Ammonia-N in shallow and deep piezometers, versus soil water suction cup samples.

This nitrogen speciation reversal indicates a critical transition zone exists between 0.6 m and 4.0 m depth where redox conditions shift from oxygen-available (where nitrification in soil water occurs) to persistently anoxic (effecting the suppression of nitrification in groundwater). The transition likely reflects progressive oxygen depletion with depth as infiltrating rainfall carrying dissolved oxygen encounters increasing oxygen demand from microbial decomposition of organic matter.

This pattern aligns with findings from Van Beek et al. (2007) in Dutch peat grasslands, where nitrate leaching to surface waters predominantly originated from near-surface horizons rather than deep peat layers. The elevated nitrate-N in shallow soil water is likely as result of nitrogen inputs from livestock urine deposition and fertiliser applications, which undergo nitrification in the aerobic surface soil. As discussed in the literature review (Section 2.3.2.2), nitrification (the microbial oxidation of ammonium to nitrate) is primarily controlled by ammonium availability and oxygen availability (Robertson & Groffman, 2024; Beeckman et al., 2018).

Mechanistic interpretation

The shallow soil water zone (0-1 m depth) experiences regular atmospheric oxygen input through rainfall infiltration, diffusion, and gas exchange. This oxygen availability permits nitrifying bacteria to convert mineralisation-derived ammonium to nitrate, explaining nitrate dominance in soil water samples. However, in organic-rich soils with high oxygen demand, this aerobic zone is shallow and transient.

Below approximately 1-2 m depth, oxygen consumption through organic matter decomposition exceeds atmospheric oxygen supply. This creates the persistently anoxic conditions documented in groundwater (Section 5.1.2.2), where nitrate cannot be produced (nitrification inhibited) and any infiltrating nitrate is rapidly consumed through denitrification or DNRA. The result is ammonium-dominated groundwater despite nitrate presence in shallow soil water.

Implications for nutrient transport pathways

This vertical stratification has important implications for understanding nitrogen export pathways. Under baseflow conditions with normal water table depths, the nitrate-rich shallow zone (0-1 m) does not connect directly to drainage systems. Instead, nitrogen export occurs through deeper groundwater discharge carrying predominantly ammonium to drains. However, during periods of intense rainfall or water table rise to near-surface, preferential flow paths (macropores, tile drains, soil cracks) may bypass the anoxic transformation zone, allowing direct transport of nitrate-rich soil water to drainage networks. This mechanism likely explains the episodic nitrate pulses observed in surface water monitoring (Section 5.2), contrasting with the predominantly ammonium-dominated baseflow signature.

The single-date sampling represents a limitation, as temporal variability in soil water chemistry could not be assessed. Future monitoring incorporating regular soil water sampling across seasons would provide valuable insights into how the aerobic-anaerobic transition zone responds to changing water table positions and rainfall patterns documented in Section 5.1.1.

5.2. Drain water nutrient patterns and groundwater linkages

Surface water monitoring at 16 researcher-sampled sites (fortnightly sampling, November 2023-October 2024, n=255 samples) combined with seven long-term catchment group sites (monthly sampling, December 2022-September 2025) provides comprehensive assessment of nutrient export dynamics. This section examines spatial patterns, temporal variability including rainfall-lag relationships, and connections between groundwater sources and surface water quality.

5.2.1 Overall catchment patterns and key sites

Catchment-wide surface water nutrient concentrations exhibited strong positive skewness across all parameters. Ammonia-N ranged from <0.02 to 14.40 g/m^3 with median 0.291 g/m^3 (mean 0.911 g/m^3), while nitrate-N displayed much lower median (0.080 g/m^3) despite wider range (<0.020-20.3 g/m^3), indicating highly episodic nitrate pulses rather than sustained elevation. DRP showed median 0.125 g/m^3, while total phosphorus median was 0.490 g/m^3, with DRP comprising approximately 26% of total P loads.

Site-specific analysis showed distinct spatial differences in surface water nutrient concentrations (Table 18). Three hotspot sites exhibited particularly elevated concentrations: STS-2 had highest nitrate-N (mean 5.65 g/m^3, median 0.52 g/m^3) and total N (mean 19.80 g/m^3, the highest in the dataset) alongside elevated ammonia-N (mean 3.59 g/m^3); Spres-1 showed consistently high ammonia-N (mean 3.36 g/m^3, median 3.20 g/m^3) with elevated DRP (mean 1.04 g/m^3, median 0.71 g/m^3, highest DRP in the catchment); SR-1 displayed episodic elevated total P concentrations (mean 11.73 g/m^3 versus median 0.19 g/m^3).

In contrast, several sites had consistently low nutrient concentrations. SBP-4 showed uniformly low levels (ammonia-N mean 0.02 g/m^3, DRP mean 0.04 g/m^3, total N mean 1.12 g/m^3, total P mean 0.10 g/m^3), while SS-1 and Levin Road similarly displayed minimal nutrient enrichment. Notably, the sites with the lowest nutrient concentrations were predominantly borrow pit locations (SBP-4, SBP-3, SBP-1, Levin Road), indicating significant attenuation capacity in these systems examined in Section 4.5.

5.2.2 Nitrogen patterns and soil type effects

Ammonia-N surface water concentrations exhibited soil type associations with silt loam associated sites showing highest concentrations and greatest variability (Figure 29). Statistical analysis confirmed significant differences across soil types (Kruskal-Wallis $H=31.91$, $p=5.9\times10^{-7}$, $\eta^2=0.113$), though effects were weaker than in groundwater ($\eta^2=0.113$ versus 0.568 in Section 5.1.3.2). Dunn's post-hoc tests revealed ammonia-N concentrations in surface water silt loam sites differed significantly from both sandy (adjusted $p=0.007$) and peaty (adjusted $p=2.81\times10^{-7}$) sites. This ammonia-N dominance in both groundwater and surface waters contrasts with nitrate-dominated losses typical of well-drained mineral agricultural soils, where nitrate-N typically comprises >80% of dissolved nitrogen exports (Monaghan et al., 2016; Di & Cameron, 2002).

The elevated ammonia-N at silt loam sites is somewhat unexpected given that silt loam groundwaters showed intermediate concentrations (median 3.12 g/m^3 versus 6.79-18.30 g/m^3 in peaty sites, Section 5.1.3.2). This surface water pattern likely reflects both groundwater inputs and surface-derived inputs from nutrient management practices. Silt loam soils derived from river alluvium have lower organic matter content (~4-6%; Cowie, 1978) than peaty soils but may be more vulnerable to surface-applied nutrient losses, particularly when farm dairy effluent applications occur on saturated soils, when soil water holding capacity is low, or when fertiliser application method, rate, or timing

Table 18: Surface water quality parameters and summary statistics by sampling site.

Site ID	n	Ammonia-N median (g/m^3)	Ammonia-N mean (g/m^3)	Nitrate-N median (g/m^3)	Nitrate-N mean (g/m^3)	DRP median (g/m^3)	DRP mean (g/m^3)	Total N median (g/m^3)	Total N mean (g/m^3)	Total P median (g/m^3)	Total P mean (g/m^3)
SCR-1	20	0.31	0.46	0.05	0.62	0.17	0.21	3.15	3.42	1.35	1.64
SCR-2	21	1.79	2.16	0.03	0.50	0.47	0.41	4.40	4.34	1.40	2.05
SBR-1	8	0.27	0.84	0.12	0.60	0.21	0.46	4.70	4.51	0.84	1.03
SCD-1	21	0.16	0.24	0.04	0.34	0.06	0.08	2.60	3.36	0.37	0.57
Spres-1	23	3.20	3.36	0.18	1.31	0.71	1.04	6.60	8.41	1.20	1.75
SKK-1	21	0.43	0.55	0.29	0.85	0.24	0.32	4.20	5.07	0.89	1.12
STS-1	9	0.10	1.00	0.59	0.87	0.06	0.12	2.90	5.12	0.35	0.82
STS-2	11	0.11	3.59	0.52	5.65	0.21	0.41	17.00	19.80	1.10	2.64
SR-1	17	0.35	1.28	0.34	1.36	0.02	0.11	3.00	8.08	0.19	11.73
SS-1	14	0.07	0.17	0.25	0.35	0.02	0.03	1.40	1.76	0.12	0.15
SBP-1	21	0.08	0.17	0.16	0.34	0.16	0.21	3.40	3.70	0.52	0.62
SBP-4	14	0.02	0.02	0.02	0.05	0.03	0.04	1.10	1.12	0.09	0.10
SBP-2	19	0.53	0.93	0.02	0.09	0.09	0.12	2.30	2.41	0.29	0.49
SPS-1	15	0.10	0.27	0.04	0.16	0.12	0.13	2.00	2.56	0.43	0.48
SBP-3	15	0.03	0.07	0.02	0.08	0.09	0.11	2.30	2.56	0.35	0.43
SPS-2	6	1.11	1.18	0.11	0.19	0.11	0.12	3.15	3.02	0.52	0.62
Whirokino Pump	28	0.46	0.89	0.25	0.31	0.06	0.09	NA	NA	NA	NA
Levin Road	32	0.04	0.07	0.02	0.23	0.10	0.11	NA	NA	NA	NA
Diagonal Pump	32	0.43	0.67	0.14	1.38	0.09	0.24	NA	NA	NA	NA
Cooks Pump	32	0.56	0.88	0.15	1.08	0.32	0.67	NA	NA	NA	NA
Tutoko Dairy	19	0.16	0.32	0.02	0.59	0.20	0.22	NA	NA	NA	NA
Top Tributary	32	1.04	1.21	0.19	1.30	0.48	0.63	NA	NA	NA	NA
Kerekere Road	32	1.02	1.35	0.02	0.16	0.14	0.14	NA	NA	NA	NA

are suboptimal. The high variability at silt loam sites (multiple substantial outliers at STS-2, SCR-2, SBR-1) supports episodic management-related impacts alongside groundwater-derived loads. In order to understand the contribution and combination of surface versus soil derived inputs in silt loam soils, further testing should be undertaken to consider fertiliser and effluent inputs (utilising available records) alongside suction cup sampling of soil water.

Ammonia-N concentration in surface water sites associated with peaty soils showed considerable variability rather than uniformly high concentrations despite elevated groundwater sources. Some peaty sites showed elevated ammonia-N (Top Tributary ~1.5 g/m^3, Kerekere Road ~1.5 g/m^3), while others maintained low levels (SBP-4, SBP-3, Levin Road). This suggests drainage water management and treatment processes substantially modify groundwater nutrient concentrations before reaching surface sampling points, with borrow pit systems providing particularly effective attenuation, as may be the case with SBP-4, SBP-3, SBP-1 and Levin Road.

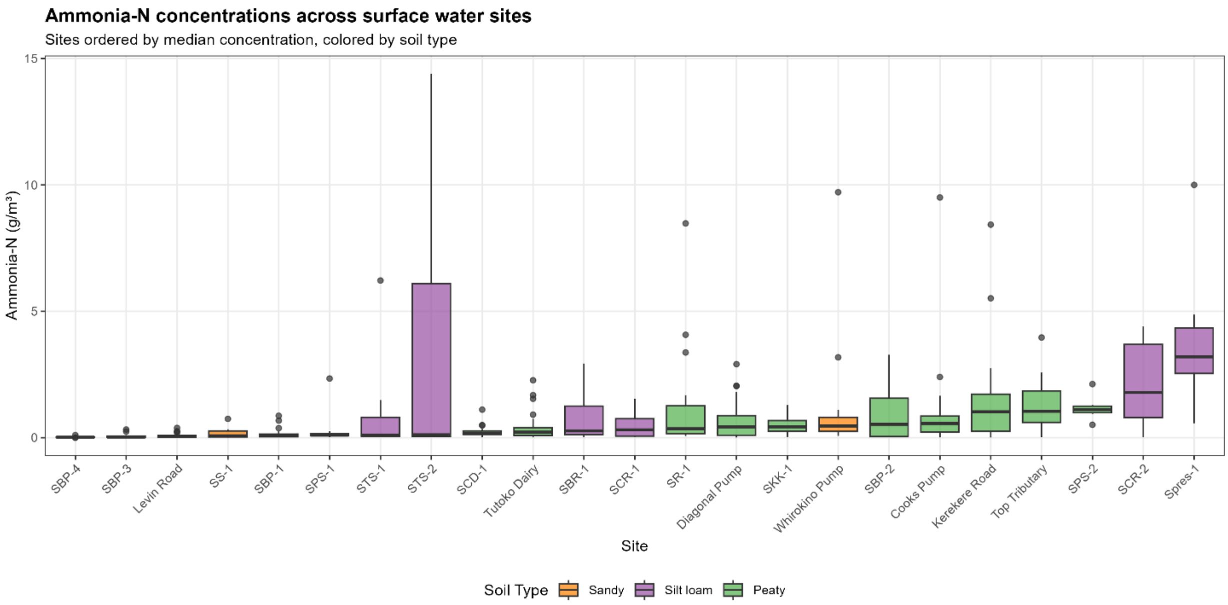

Figure 29: Ammonia-N concentrations across all surface water sites, ordered by median values and coloured by soil type association.

Nitrate-N concentrations contrasted sharply with ammonia-N patterns across the catchment (Figure 30). Most sites across all soil types displayed very low median concentrations (<0.5 g/m^3) but with some elevated episodic outliers. STS-2 featured the highest value (approximately 20.3 g/m^3), while several peaty soil associated surface water sampling sites had multiple high outliers including Diagonal Pump (up to ~13 g/m^3), Cooks Pump, Top Tributary, and Tutoko Dairy (multiple outliers reaching ~7 g/m^3). Statistical analysis confirmed significant soil type effects (Kruskal-Wallis p=0.00013), with peaty soils differing significantly from both sandy (adjusted p=0.007) and silt loam (adjusted p=0.002) sites, driven by high outliers despite low medians.

The inverse relationship between ammonia-N and nitrate-N concentrations reflects the redox gradient associated with nitrogen speciation established in Sections 5.1.2.2 and

5.1.3.6. Sites with high ammonia-N typically showed low nitrate-N (Spres-1, SCR-2, Kerekere Road), while sites with nitrate-N outliers often had lower ammonia-N (STS-1, STS-2). This mirrors the soil water versus groundwater contrast where soil water at 60 cm depth was nitrate-dominated (1.09-4.49 g/m^3) while groundwater at 4.0-5.5 m was ammonium-dominated (1.13-18.7 g/m^3), with the transition zone between 0.6-4.0 m depth largely preventing nitrate-N from reaching surface drains under baseflow conditions.

The episodic nitrate-N pulses in surface water likely originate from direct surface runoff mobilising nutrients from fertiliser applications or effluent during storm events, preferential flow through tile drainage systems and soil macropores bypassing the anoxic transformation zone (Jarvis, 2007), or temporary oxidation of shallow groundwater following water table reductions creating transient nitrification conditions. The extreme outliers (up to 20.3 g/m^3) approach concentrations documented in tile drainage from conventionally managed systems (Monaghan et al., 2016 reported 30-56 kg N ha^{-1} year^{-1} losses, predominantly as nitrate-N), indicating that during high-intensity rainfall events, the catchment can generate nitrate-N pulses comparable to mineral soil systems despite the predominantly ammonium-dominated baseflow signature.

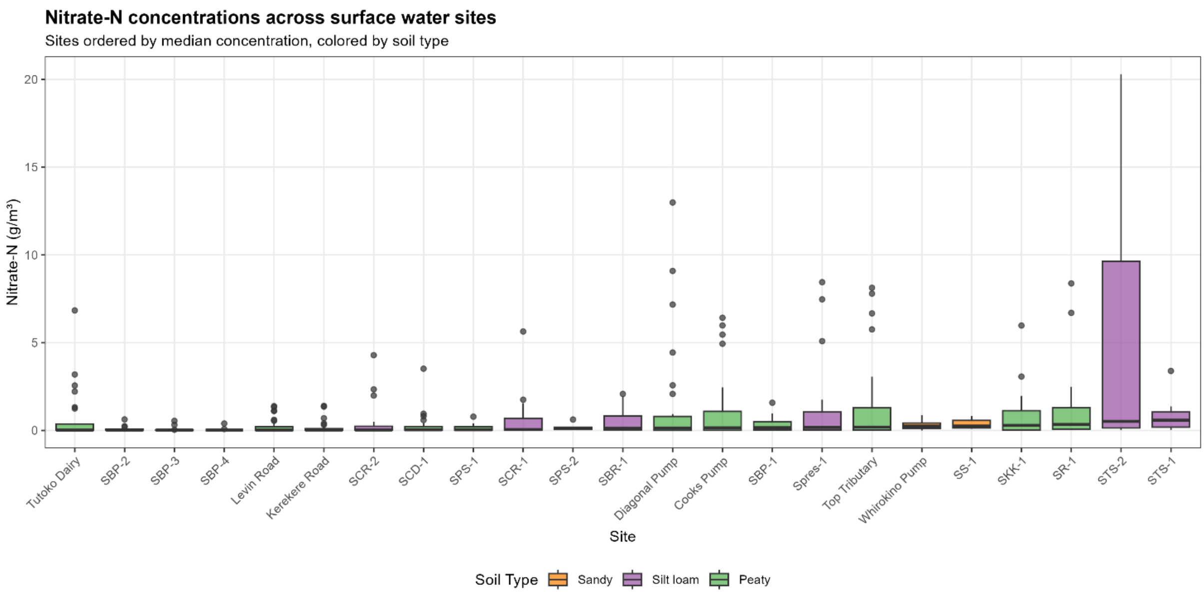

Figure 30: Nitrate-N concentrations across all surface water sites, ordered by median values and coloured by soil type association.

5.2.3 Phosphorus patterns and transport mechanisms

DRP showed complex spatial patterns less controlled by soil type than ammonia-N (Figure 31). Statistical analysis confirmed highly significant soil type effects (Kruskal-Wallis $p=2.2\times10^{-16}$) with all three pairwise comparisons significant: sandy versus silt loam (adjusted $p=1.00\times10^{-16}$), sandy versus peaty (adjusted $p=8.97\times10^{-9}$), and silt loam versus peaty (adjusted $p=9.50\times10^{-7}$). However, unlike ammonia-N where silt loam was consistently highest, DRP patterns showed the highest site (Spres-1, silt loam: median 0.71 g/m^3) contrasting with many low-concentration peaty soil associated sites (SBP series: medians near detection limits), demonstrating DRP export is not primarily soil-type driven but influenced by site-specific management and treatment capacity.

The correlation between ammonia-N and DRP in surface waters (ρ=0.476, p=3.59×10^{-28}) supports coupled mobilisation from groundwater sources, though the relationship is weaker than in groundwater itself (ρ=0.84, Section 5.1.3.3), indicating surface processes modify groundwater-derived concentrations. Total N and total P displayed the strongest correlation of all nutrient pairs (ρ=0.762, p=2.56×10^{-49}), suggesting common sources delivering both nutrients together.

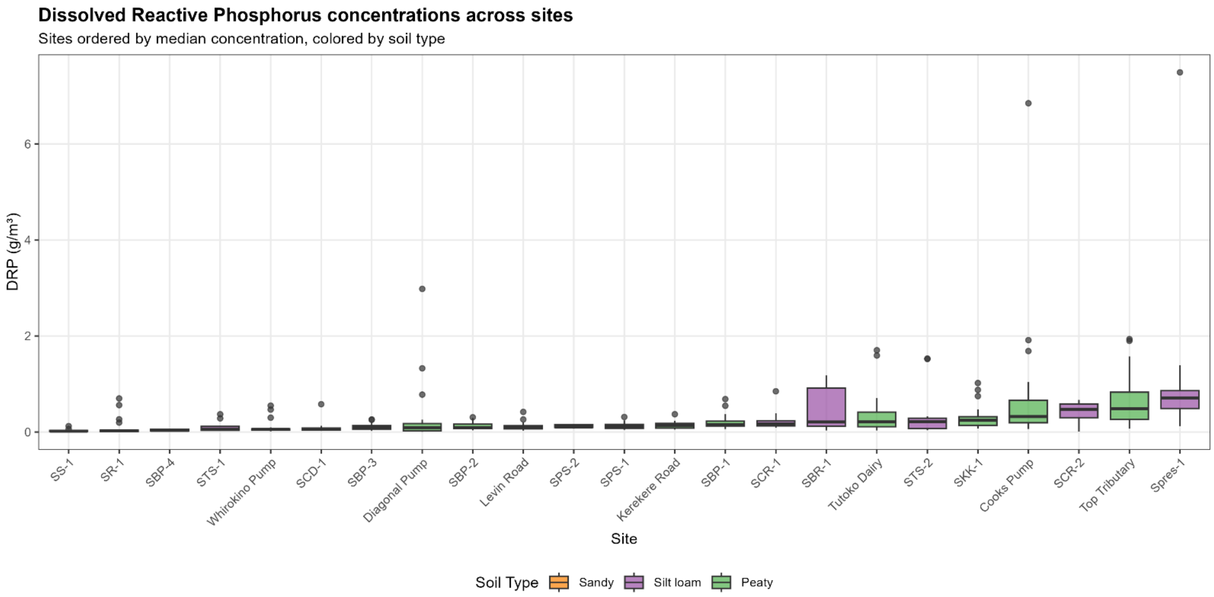

Figure 31: DRP concentrations across all surface water sites, ordered by median values and coloured by soil type association.

DRP comprised 26% of total phosphorus loads, indicating dissolved reactive phosphorus represents a substantial but not dominant fraction. This phosphorus speciation pattern has important implications for mitigation strategy selection. In well-drained pastoral systems on mineral soils, surface runoff typically transports the majority of phosphorus as particulate forms attached to eroded sediment (Monaghan et al., 2016; Simmonds et al., 2016). However, in drained organic soils with inherently low anion storage capacity (ASC typically <20%; McDowell & Monaghan, 2015), dissolved phosphorus transport through subsurface drainage becomes more important. McDowell and Monaghan (2015) documented that dissolved phosphorus losses from low ASC organic soils ranged from 1.7 to 87 kg P ha^{-1}, representing up to 89% of applied fertiliser phosphorus. Gray et al. (2024) found that subsurface drainage was a key phosphorus pathway in organic soils regardless of moisture conditions, due to the combination of low ASC and high hydraulic conductivity. Sapek et al. (2021) found phosphate-phosphorus and ammonia-nitrogen concentrations in groundwater from Polish peat meadows were significantly positively correlated, indicating peat mineralisation was the common driver.

5.2.4 Temporal dynamics and rainfall-nutrient relationships

Time series analysis revealed episodic nutrient pulses within surface water, superimposed on relatively stable baseline concentrations (Figure 32). The early monitoring period (December 2023-February 2024) showed elevated ammonia-N at several sites, with STS-2 particularly high, declining by March 2024. A mid-study event in May 2024 produced sharp concentration spikes in both ammonia-N and DRP at Spres-1, while most other sites remained stable, indicating site-specific vulnerability to nutrient mobilisation during saturated soil conditions, potentially from effluent irrigation or fertiliser application followed by rainfall-driven transport. The late winter period (July-August 2024) showed increases in nitrate-N concentrations in surface water across multiple sites, suggesting catchment-scale rainfall events with widespread nutrient mobilisation, while DRP remained relatively stable during this higher rainfall period.

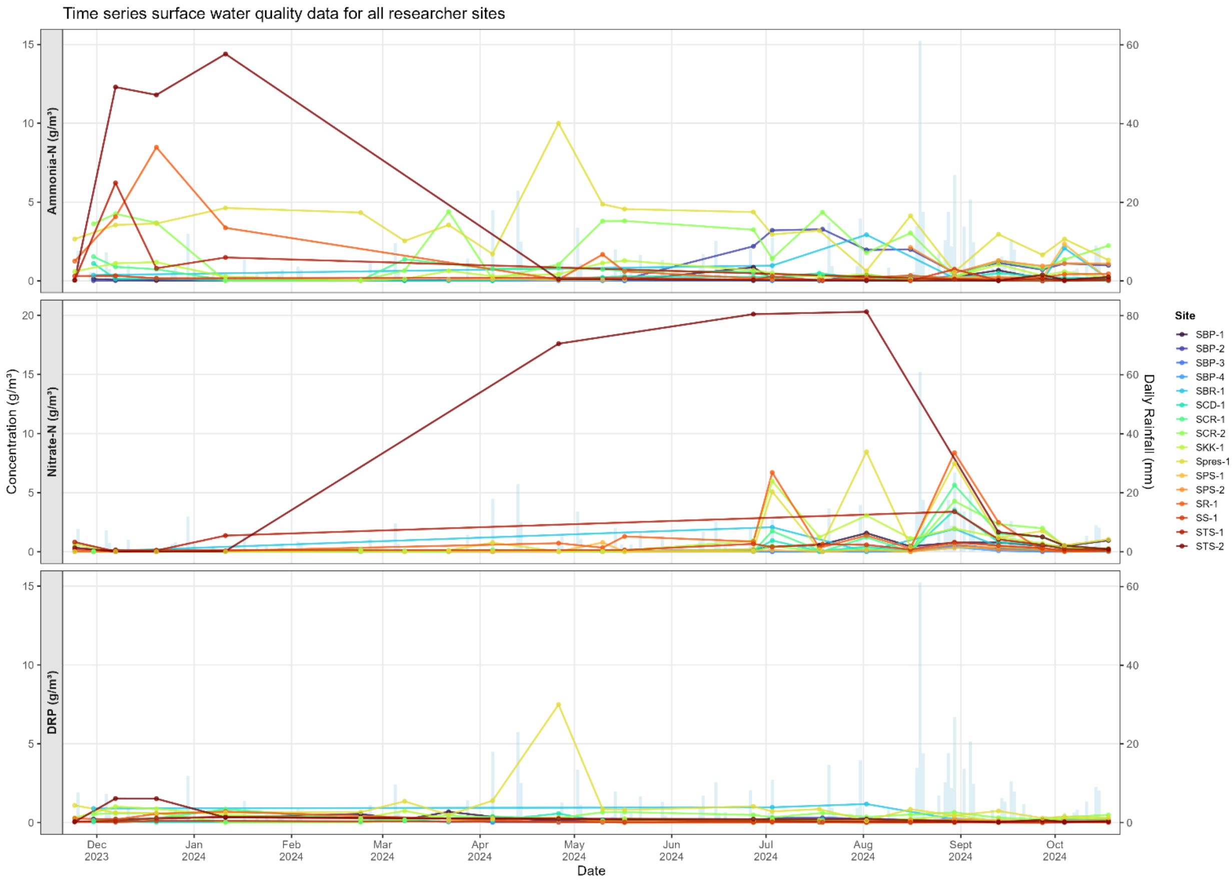

Figure 32: Time series surface water quality data for all researcher sampling sites during the sampling period.

Spearman's correlation analysis between surface water nutrient concentrations and rainfall at 3-day, 7-day, and 14-day lag periods revealed differing responses among nutrients, indicating multiple transport pathways operating at different timescales. Nitrate-N showed the strongest and most consistent rainfall relationship, with 50% of sites exhibiting significant correlations at 14-day lag, 31.2% at 3-day lag, and 12.5% at 7-day lag. This response pattern, with significant correlations at both short (3-day) and long (14-day) lags, suggests nitrate-N transport through dual pathways: rapid delivery by preferential flow, macropores, and tile drainage systems during rainfall events (3-day lag), and slower sustained delivery via groundwater discharge as elevated water tables connect to the drainage network (14-day lag). This interpretation aligns with Jarvis (2007), who documented that preferential flow through macropores and structural cracks can rapidly transport solutes from surface soils to drainage systems within hours to days, bypassing the soil matrix and its potential for retention or transformation.

Ammonia-N showed peak correlation strength at intermediate (7-day) lag, with 25% of sites exhibiting significant relationships, compared to 18.8% at 14-day lag and only 6.2% at 3-day lag. This intermediate-lag response indicates ammonia-N mobilisation operates through slower transport pathways than nitrate-N, with weak immediate (3-day) response suggesting ammonia-N is not readily available for rapid surface runoff or preferential flow transport, but rather requires subsurface pathways that develop over several days following rainfall. This pattern is consistent with ammonia-N being produced primarily through decomposition processes in saturated, oxygen-limited soil zones where it accumulates due to nitrification inhibition (Section 5.1.2.2), then transported laterally through tile and mole-pipe drainage networks as water tables rise following rainfall.

DRP showed significant correlations exclusively at 14-day lag (25% of sites), with no significant relationships at 3-day or 7-day lags. This delayed response indicates dissolved phosphorus mobilisation through the slowest transport pathways, likely reflecting phosphorus release from soil profiles during prolonged saturation that allows slow lateral transport through groundwater and subsurface drainage. The absence of short-lag correlations suggests direct surface runoff and rapid preferential flow are not dominant phosphorus transport mechanisms in this artificially drained landscape, contrasting with many agricultural systems where surface runoff provides a major phosphorus pathway. Instead, the 14-day lag response supports the interpretation that subsurface drainage is the dominant nutrient export pathway, with phosphorus mobilised from soil profiles through redox-sensitive processes (Section 5.1.3.1) then transported via groundwater discharge to the surface drainage network.

5.2.5 Seasonal patterns and long-term context

Analysis of long-term catchment group surface water quality data (December 2022-September 2025) across seven sites revealed seasonal patterns varying by nutrient species (Figure 33). Ammonia-N showed moderate seasonal variability with winter having highest median (0.628 g/m^3) and mean (1.083 g/m^3) compared to summer (median 0.327 g/m^3), autumn (median 0.102 g/m^3), and spring (median 0.431 g/m^3), with most observations clustered below 2 g/m^3 across all seasons, suggesting consistent baseline ammonia-N production with episodic elevations.

Nitrate-N had the most pronounced seasonal pattern, with winter having elevated concentrations (mean 2.022 g/m^3, median 0.835 g/m^3) compared to other seasons. The winter maximum reached approximately 13.0 g/m^3, the highest nitrate-N value recorded across the entire catchment group monitoring program. DRP concentrations decreased progressively from summer (mean 0.516 g/m^3, median 0.255 g/m^3) through autumn (mean 0.382 g/m^3, median 0.093 g/m^3) and spring (mean 0.229 g/m^3, median 0.137 g/m^3) to winter (mean 0.15 g/m^3, median 0.097 g/m^3).

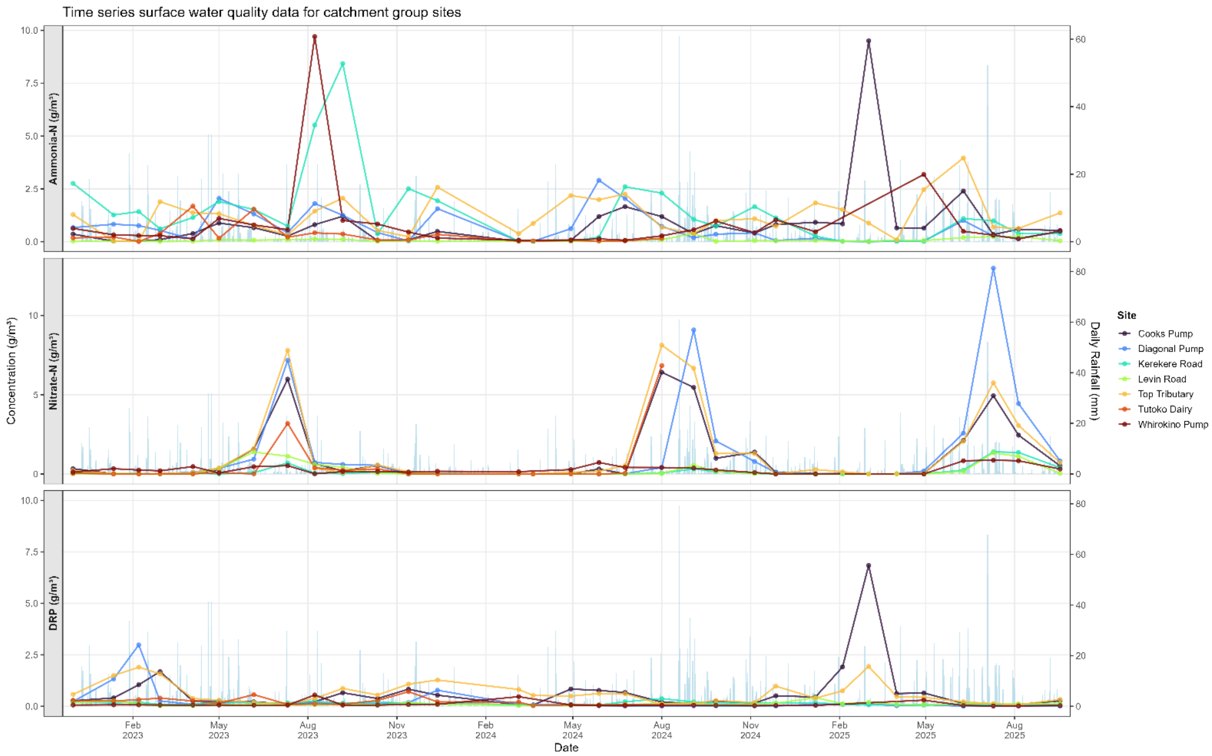

Figure 33: Time series surface water quality data for all catchment group sampling sites during the sampling period.

The contrasting seasonal nutrient concentration patterns in surface water between nitrate-N (winter maximum) and DRP (summer maximum) indicate different controls and transport mechanisms. The winter elevation in nitrate-N reflects the convergence of elevated rainfall, reduced evapotranspiration, rising water tables, and increased drainage flows that mobilise accumulated soil nitrate-N. Petry et al. (2002) found that approximately 75% of nitrogen loss from agricultural catchments occurs during autumn and early winter periods of higher rainfall, when hydrological connectivity between surface soils and drainage systems is maximised. The summer DRP maximum may reflect enhanced organic matter decomposition under warmer temperatures combined with periodic irrigation maintaining drainage flow despite low rainfall.

The extended time series revealed episodic nutrient spikes within relatively stable baseline conditions. Ammonia-N spikes occurred during August 2023 (reaching approximately 10 g/m^3) and February 2025 (approximately 9.5 g/m^3), separated by approximately 18 months and occurring in different seasons, suggesting sporadic events rather than seasonal patterns. Nitrate-N showed three spike events in July 2023 (maximum approximately 8 g/m^3), August 2024 (maximum approximately 9 g/m^3), and August 2025 (maximum approximately 12 g/m^3, the highest concentration observed), demonstrating apparent annual recurrence pattern during winter months correlating with high-rainfall periods. This multi-year data confirms the event-driven nutrient dynamics documented in the 2023-2024 intensive study represent continuous catchment characteristics rather than transient conditions specific to the study year. The intensive study period (November 2023-October 2024) captured conditions during reasonably stable to improving water quality for ammonia and phosphorus, with stable conditions for total inorganic nitrogen, representing typical rather than anomalous catchment behaviour.

5.2.6 Water quality parameters and ecological implications

Surface water quality field parameter measurements (n=255 samples) revealed some water quality concerns alongside nutrient enrichment. Dissolved oxygen concentrations had the largest variation of all measured parameters, with median DO ranging from 1.09 mg/L at SR-1 to 11.37 mg/L at SBP-3. Seven of sixteen sites (44%) had median DO below 4 mg/L, indicating widespread water quality impairment. DO concentrations below 5 mg/L can stress sensitive aquatic organisms, while levels below 2-3 mg/L create hypoxic conditions incompatible with most aquatic life (Diaz & Rosenberg, 2008). The prevalence of oxygen-limited conditions has important implications for nitrogen transformations, as DO levels below 2 mg/L can support denitrification (Wolters et al., 2022), potentially explaining some nitrate-N depletion observed in the system. However, the persistence of elevated ammonia-N concentrations in these low-oxygen drainage waters indicates that oxygen limitation inhibits nitrification more effectively than denitrification attenuates nitrogen from the system.

Statistical analysis confirmed significant differences in most field parameters across soil types. TDS ($H=47.18$, $df=2$, $p=5.68\times10^{-11}$, $\eta^2=0.181$), conductivity ($H=42.71$, $df=2$, $p=5.31\times10^{-10}$, $\eta^2=0.163$), and pH ($H=39.22$, $df=2$, $p=3.05\times10^{-9}$, $\eta^2=0.149$) showed large effect sizes. Surface water sites associated with peaty soils showed the highest conductivity and TDS (median conductivity approximately 700 μS/cm, distribution extending to 3500 μS/cm), while silt loam sites had lowest values (approximately 400 μS/cm). These patterns correspond with shallow groundwater (Section 5.1.2.1), where peaty soils also had highest conductivity (median 773-914 μS/cm), indicating connectivity between groundwater and surface drain waters (Van Beek et al., 2007).

pH patterns showed progression from acidic in silt loam soils (median pH 5.7-6.7) through neutral in sandy soils (median 7.3) to alkaline in peaty soils (median 7.2-8.1). Site SBP-3 displayed an outlier exceeding pH 9.5 coinciding with highest median DO (11.37 mg/L), suggesting substantial primary productivity. Such high pH events have important implications for ammonia toxicity, as the equilibrium between ammonium (NH_4^+) and toxic unionised ammonia (NH_3) shifts dramatically with pH. At pH 9.5 combined with elevated ammonia concentrations, the proportion of toxic NH_3 increases substantially, creating potential ecological risk to aquatic organisms even where total ammonia concentrations are moderate (Emerson et al., 1975). This represents an additional water quality concern beyond nutrient enrichment alone, particularly at sites experiencing algal blooms driving pH elevation.

5.2.7 Comparison to Regional and National standards

This analysis compares median concentrations at each site against the Horizons Regional Council One Plan targets and the National Policy Statement for Freshwater Management (NPS-FM) National Bottom Lines (NBL), where applicable. These standards represent regulatory thresholds designed to protect ecosystem health and maintain water quality suitable for downstream uses including the internationally significant Manawatu Estuary.

Soluble Inorganic Nitrogen (SIN)

SIN concentrations are the sum of all bioavailable nitrogen forms (ammonia-N, nitrate-N, and nitrite-N). SIN values generally exceeded the Horizons Regional Council target of 0.444 mg/L across the catchment (Figure 34). Three sites were critical inorganic nitrogen hotspots with medians exceeding 2.5 mg/L: STS-2 had very high SIN at approximately 11.8 mg/L; Spres-1 at approximately 4.3 mg/L; and SCR-2 at approximately 3.0 mg/L.

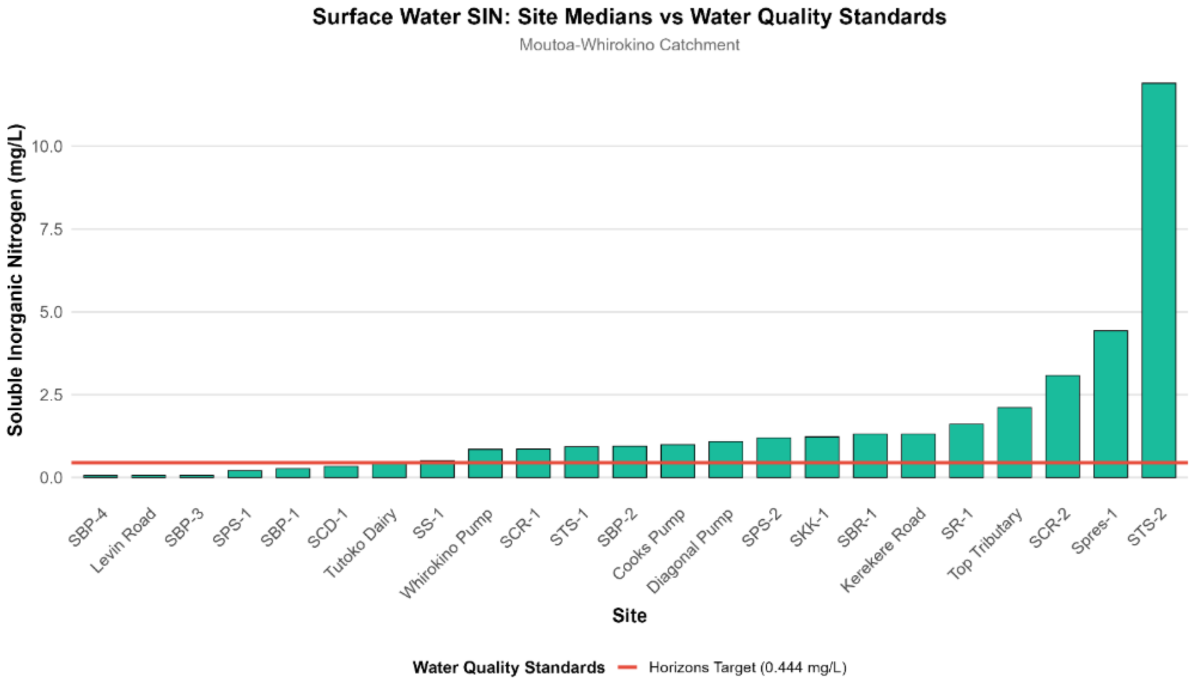

Figure 34: Soluble Inorganic Nitrogen surface water values across the catchment in relation to water quality standards.

Nitrate-Nitrogen

In contrast to SIN, nitrate-N concentrations were generally at or below water quality standard thresholds across the catchment (Figure 35). All 16 researcher-monitored sites plus the 7 community group sites were below the NPS-FM National Bottom Line of 2.4 mg/L. Only two sites (STS-1 and STS-2) marginally exceeded the more stringent Horizons Regional Council target of 0.444 mg /L, though concentrations remained far below levels typically associated with severe eutrophication risk. This pattern aligns with those previously discussed, that nitrogen in Moutoa drainage waters exists predominantly as ammonia rather than nitrate.

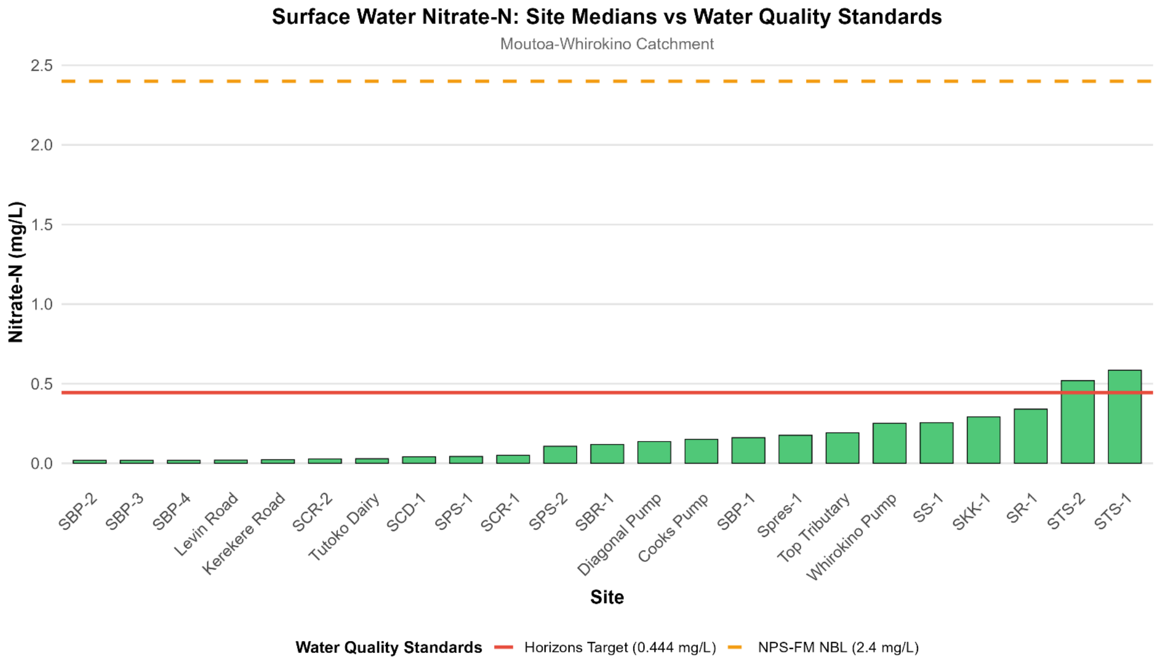

Figure 35: Nitrate-N surface water values across the catchment in relation to water quality standards.

Dissolved Oxygen

Dissolved oxygen concentrations varied in relation to standards, with eight sites failing to meet the NPS-FM National Bottom Line of 5 mg/L (Figure 36). SR-1 displayed hypoxic conditions at approximately 1.0 mg/L (10.5% saturation), falling below even the Horizons Regional Council target of 2 mg/L.

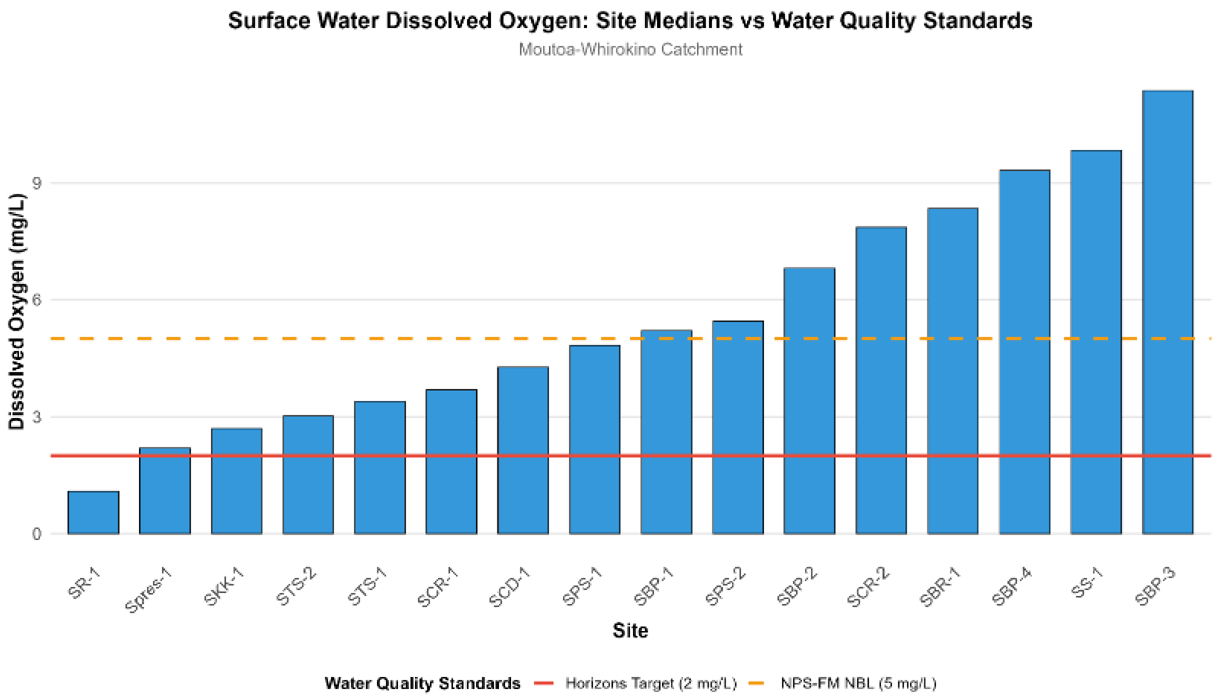

Figure 36: Dissolved oxygen surface water values across the catchment in relation to water quality standards.

Seven additional sites fell between the Horizons target and the NPS-FM bottom line (2-5 mg/L range): Spres-1, SKK-1, STS-2, STS-1, SCR-1, SCD-1, and SPS-1. These sites experience compromised oxygenation that would stress or exclude many aquatic organisms (Diaz & Rosenberg, 2008). Eight sites met the NPS-FM bottom line, with SBP-3 showing the highest dissolved oxygen at approximately 10.8 mg/L.

Ammonia-Nitrogen

Ammonia-N concentrations showed more variable compliance than SIN, with ten sites exceeding both the NPS-FM National Bottom Line (0.24 mg/L) and the Horizons Regional Council target (0.4 mg/L) (Figure 37). Spres-1 had the most elevated ammonia-N concentrations with a median of approximately 3.2 mg/L, or 13 times the national bottom line and 8 times the regional target. Additional sites with severe exceedances included SCR-2 (~1.75 mg/L), SPS-2 (~1.1 mg/L), Top Tributary (~1.05 mg/L), and Kerekere Road (~1.0 mg/L). Three sites (SBR-1, SCR-1, SR-1) exceeded the national bottom line but remained below the regional target, while ten sites remained compliant with both standards.

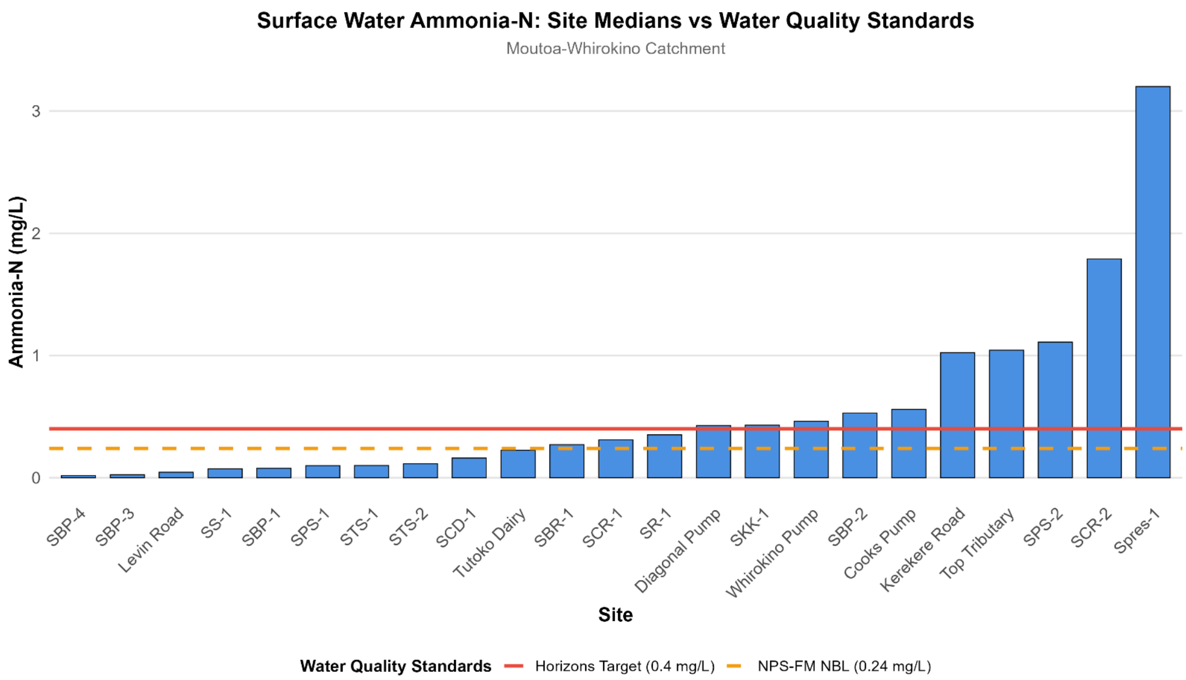

Figure 37: Ammonia-N surface water values across the catchment in relation to water quality standards.

Dissolved Reactive Phosphorus

DRP concentrations across all 23 monitoring sites exceeded the Horizons Regional Council target of 0.01 mg/L (Figure 38). While no national bottom line exists for DRP under the NPS-FM, the universal exceedance of the regional target suggests severe issues with phosphorus enrichment throughout the drainage network. Three sites emerged as critical phosphorus hotspots: Spres-1 (~0.7 mg/L), Top Tributary (~0.5 mg/L), and SCR-2 (~0.5 mg/L).

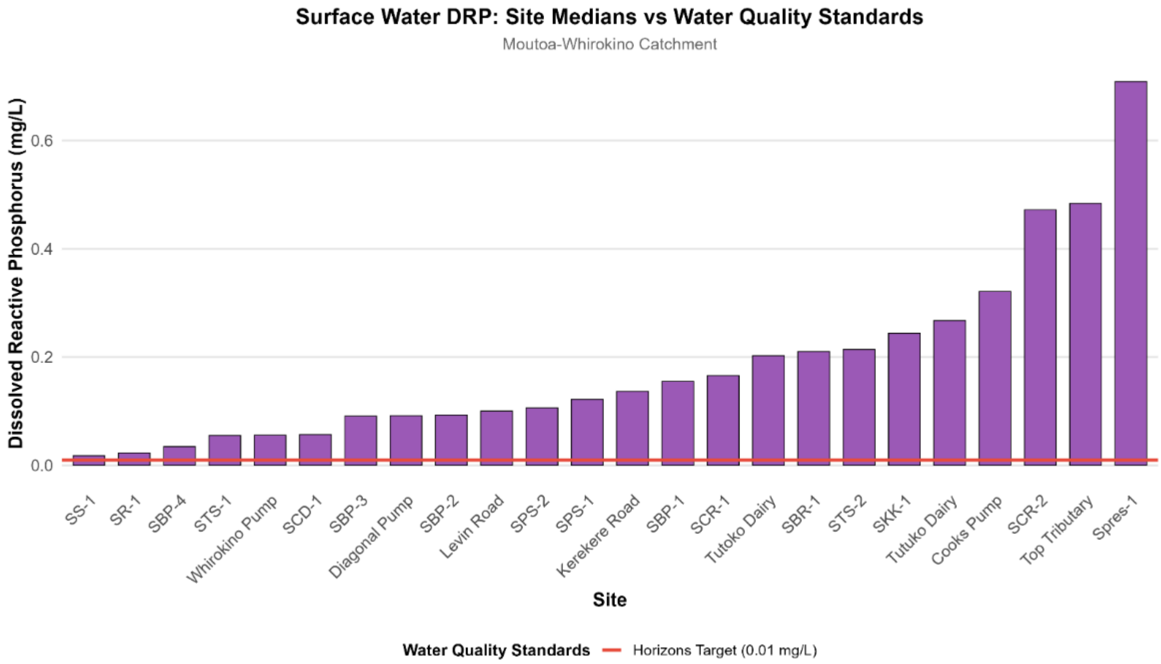

Figure 38: Dissolved reactive phosphorus surface water values across the catchment in relation to water quality standards.

The universal DRP exceedance reflects the combination of low anion storage capacity in organic soils (particularly peaty and organic-enriched silt loam soils), ongoing phosphorus mineralisation from organic matter decomposition, and redox-associated phosphorus mobilisation documented in the groundwater analysis (Section 5.1.3). As discussed in the Literature Review (Section 2.3.2.5), drained organic soils with low anion sorption capacity are among the highest-risk landscapes for phosphorus export to receiving waters, and the surface water data from Moutoa confirm this assessment.

Regional and National Context

The nutrient patterns observed in Moutoa-Whirokino drainage waters are consistent with broader regional water quality challenges documented for the Manawatu catchment. As noted in the Literature Review (Section 2.2.3), between June 2017 and June 2022, only 32% of river monitoring sites in the Manawatu catchment met One Plan targets for SIN, while just 22% of sites met targets for DRP. The Moutoa-Whirokino subcatchment appears to perform slightly better than the catchment average for SIN (with several sites achieving compliance), but substantially worse for DRP (with zero sites achieving compliance). This suggests that phosphorus management represents a significant water quality challenge in this artificially drained organic soil landscape.

Further to this, the LAWA (Land Air Water Aotearoa) monitoring site at Manawatu River at Whirokino shows a 5-year median DRP concentration that places the lower Manawatu River in the worst 25% of sites nationally for phosphorus. The surface drain data from Moutoa-Whirokino suggests that this degraded river water quality reflects, at least in part, elevated phosphorus loads delivered via the artificial drainage network from intensively managed organic soil systems. The concentration of DRP in individual surface drains (ranging up to 0.7 mg/L at Spres-1) exceeds the river concentrations substantially, indicating that dilution by the main stem Manawatu River moderates but does not eliminate the phosphorus enrichment from this type of catchment.

It is important to emphasise that the concentration data presented here represent samples collected from surface drains and small tributaries within the catchment, not from the main stem of the Manawatu River. These drain waters will be diluted substantially upon discharge to larger receiving waters, though dilution does not reduce total nutrient loads. Therefore, while the concentration data identifies priority areas for management intervention, it does not directly represent the water quality experienced in the Manawatu River or Estuary.

5.2.8 Integration with groundwater sources and implications for management

Surface water nutrient patterns show substantial site-to-site variability, with some sites having particularly high elevation (STS-2, Spres-1, SR-1) while neighbouring sites on similar soils maintain consistently low concentrations (SBP-4, SBP-3, SS-1). This variability demonstrates that management practices and drainage infrastructure configuration can modify nutrient export beyond what soil type alone predicts. The ammonium-dominated signature in surface waters (median NH_4-N 0.291 g/m^3 versus NO_3-N 0.080 g/m^3) mirrors the pattern observed in shallow groundwater (Section 5.1.3.2), though with substantially lower absolute concentrations. Quantitative analysis of groundwater-surface water nutrient relationships at paired monitoring sites is presented in the Section 5.3, while treatment performance in borrow pit systems is examined in Section 6.2.

Having established surface water patterns and their variability, this section quantitatively examines whether groundwater nutrient concentrations predict surface water quality at paired monitoring sites.

5.3 Relationships between surface drains and groundwaters nutrient concentrations

The preceding sections have established that shallow groundwater contains elevated nutrient concentrations (Section 5.1.3) while surface drainage waters display substantial spatial variability (Section 5.2). To quantitatively assess whether groundwater nutrient concentrations predict surface water quality at spatially proximate sites, Spearman correlation analysis was conducted on five paired groundwater-surface water site groupings: PS-1&PS-2/SS-1 (sandy soils), PR-1&PR-2/SR-1 (peaty soils), PTS-1&PTS-2/STS-1&STS-2 (silt loam soils), PCD-1-S&PCD-1-D/SCD-1 (peaty soils), and PCR-1&PCR-2/SCR-1&SCR-2 (silt loam soils).

5.3.1 Overall correlation patterns and interpretation

Aggregated correlation analysis across all paired sites (n=56 observations) revealed weak positive relationships for nitrogen species but no relationship for phosphorus. Ammonia-N showed weak positive correlation between groundwater and surface water concentrations ($\rho=0.287$, $p=0.032$), as did nitrate-N ($\rho=0.271$, $p=0.043$). In contrast, DRP exhibited essentially zero correlation (ρ=-0.002, p=0.987), while soluble inorganic nitrogen showed no significant relationship (ρ=-0.059, p=0.664).

These weak overall correlations indicate that while shallow groundwater represents a nutrient source to the drainage network, surface water concentrations are not simply predicted by proximate groundwater chemistry. This disconnect reflects the complexity of hydrological and biogeochemical processes operating between groundwater discharge and surface water sampling points, consistent with conceptual models of groundwater-surface water interactions in artificially drained agricultural catchments (Winter et al., 1998; Sophocleous, 2002).

5.3.2 Site-specific correlation patterns

Individual paired site analysis revealed that most site-specific correlations were non-significant, with only two exceptions showing statistically significant relationships. The PCR/SCR-2 pairing displayed moderate positive nitrate-N correlation (ρ=0.706, p=0.022, n=10), suggesting that groundwater nitrate concentrations at this silt loam site influence surface water patterns, though sample size limits confidence in this relationship. More surprisingly, the PTS/STS-2 pairing exhibited significant inverse ammonia-N correlation (ρ=-0.829, p=0.042, n=6), where higher groundwater ammonia concentrations corresponded with lower surface water concentrations.

Most paired sites showed no significant correlations despite substantial differences in groundwater nutrient concentrations. For example, PCD-1 piezometers (both shallow and deep) maintained median ammonia-N of 17.1-18.3 g/m^3 and DRP of 3.15-3.53 g/m^3, yet the proximate surface water site SCD-1 showed median ammonia-N of only 0.16 g/m^3 and DRP of 0.06 g/m^3 (representing 99% and 98% reductions respectively) with no significant correlation between the paired concentrations (all p>0.16). Similarly, PR piezometers showed groundwater ammonia-N of 6.79-9.37 g/m^3 and DRP of 4.38-4.56 g/m^3, while SR-1 displayed surface concentrations of 0.35 g/m^3 and 0.02 g/m^3 respectively, again with no significant correlations (all p>0.25).

5.3.3 Mechanisms contributing to weak groundwater-surface water relationships

The weak correlations and substantial concentration reductions between groundwater sources and surface water sampling points reflect multiple interacting processes. Dilution is a primary mechanism, with surface drainage waters receiving inputs from multiple sources including direct rainfall, surface runoff from surrounding paddocks, lateral inflows from adjacent sub-catchments, and groundwater discharge from areas beyond the immediate monitoring piezometers (Tiemeyer et al., 2007). Groundwater and surface water are interconnected components of a single hydrological system, with the direction and magnitude of exchanges controlled by hydraulic gradients that vary spatially and temporally in response to precipitation, drainage operations, and evapotranspiration (Winter, 1999).

Temporal irregularity between groundwater nutrient production and surface water sampling likely contributes to weak correlations. Van Beek et al. (2007) documented in

Dutch peat grasslands that approximately 90% of nutrient leaching occurred during winter months, highlighting substantial temporal variability in nutrient mobilisation and transport.

Variable subsurface hydrologic connectivity exerts strong control on nutrient transport at catchment scales (Jencso et al., 2009). The artificial drainage infrastructure in the Moutoa-Whirokino catchment creates complex flow pathways, with tile drains, mole drains, and open drains intercepting shallow groundwater and shifting it through networks that may or may not discharge directly to surface water monitoring locations. Schilling and Helmers (2008) demonstrated that artificial subsurface drainage fundamentally alters hydrological pathways, reducing mean groundwater travel times from approximately 30 years in naturally drained systems to less than 10 years in tile-drained catchments (Schilling et al., 2020), while simultaneously creating preferential pathways that bypass certain landscape positions.

In-channel biogeochemical transformations may substantially modify nutrient concentrations between groundwater discharge and surface water sampling. Nitrogen transformations including nitrification-denitrification cycles, biological uptake by aquatic vegetation, and volatilisation of ammonia-N at elevated pH can reduce dissolved nitrogen concentrations during transport. Phosphorus retention through sorption to channel sediments, co-precipitation with iron and aluminium oxyhydroxides, and sedimentation of particulate forms may attenuate dissolved reactive phosphorus loads, particularly in vegetated drainage channels or borrow pit systems that provide extended hydraulic residence time and enhanced treatment capacity (Section 6.2).

The inverse ammonia-N correlation observed at the PTS/STS-2 pairing ($\rho=-0.829$, $p=0.042$) warrants cautious interpretation given the small sample size ($n=6$). Potential explanations include dominance of surface-derived ammonia-N sources (such as farm dairy effluent applications) that vary independently of groundwater concentrations, episodic dilution events that differentially affect surface versus groundwater sampling, or complex flow routing where STS-2 receives drainage from areas distinct from the PTS piezometer locations. This anomaly emphasises that spatial proximity does not guarantee direct hydrological connectivity in artificially drained catchments with engineered flow networks.

5.3.4 Implications for source tracking and nutrient management

The weak correlations between groundwater and surface water nutrient concentrations demonstrate that while shallow groundwater represents a major nutrient source to the drainage network, surface water quality is not simply determined by local groundwater chemistry. The substantial concentration reductions observed at most paired sites (commonly 90-99% for ammonia-N and DRP) indicate that multiple processes attenuate groundwater-derived nutrient loads before they reach surface monitoring points.

This complexity has important implications for nutrient management strategy development. Source control measures targeting groundwater nutrient concentrations (such as controlled drainage to reduce organic matter mineralisation in peaty soils) remain essential for reducing baseline nutrient mobilisation. However, the weak groundwater-surface water correlations suggest that in-system treatment approaches leveraging dilution, biogeochemical transformation, and physical retention processes offer substantial opportunities for improving surface water quality even where groundwater sources remain elevated. The borrow pit systems examined in Section 6.2, vegetated drainage channels, and constructed wetlands can provide sequential treatment of groundwater-derived nutrient loads, with the weak correlations documented here demonstrating that treatment effectiveness can substantially modify the relationship between source concentrations and receiving water quality.

Chapter 6

6. Nutrient mitigation strategies

The preceding sections have documented elevated nutrient concentrations in shallow groundwater (Section 5.1.3), substantial spatial variability in surface water quality (Section 5.2), and weak groundwater-surface water correlations indicating significant modification of nutrient loads during transport (Section 5.3). This section examines two complementary mitigation approaches: controlled drainage for source reduction, and borrow pit systems for in-stream treatment.

6.1 Controlled drainage: Water table management for nutrient source control

Controlled drainage is the practice of managing artificial drainage outlet elevations to manipulate water table position. It can reduce nutrient mobilisation at source by maintaining elevated soil moisture and limit the aerobic decomposition that drives nutrient release. This section evaluates controlled drainage effects on water table dynamics, hydraulic gradients, and nutrient export using data from the PCD-1 transect where drainage management was actively manipulated, and examines limited available nutrient data from the associated surface drain.

6.1.1 Spatial patterns and hydraulic controls on nutrient transport

A cross-sectional transect of piezometers was established at the PCD-1 site to characterise spatial patterns of water table position and drainage management responses. The approximately 700-metre transect extends across Kairanga Peaty Silt Loam and includes four surface drains at 0 m (Drain 1), ~275 m (Drain 2), ~600 m (Drain 3), and ~710 m (Drain 4). Monitoring across eleven dates from 24 November 2023 to 4 October 2024 captured the full seasonal cycle documented in Section 5.1.1, providing additional spatial resolution unavailable from the catchment-wide network.

Cross-sectional profiles showed consistent spatial patterns throughout the monitoring period (Figure 39). Water table elevations formed an undulating profile, with depressions adjacent to drain locations and elevated water midway between drains, illustrating hydraulic control by artificial drainage. At drain locations (vertical dashed blue lines in Figure 39), water table depressions varied with drainage management status (blocked versus unblocked) and rainfall.

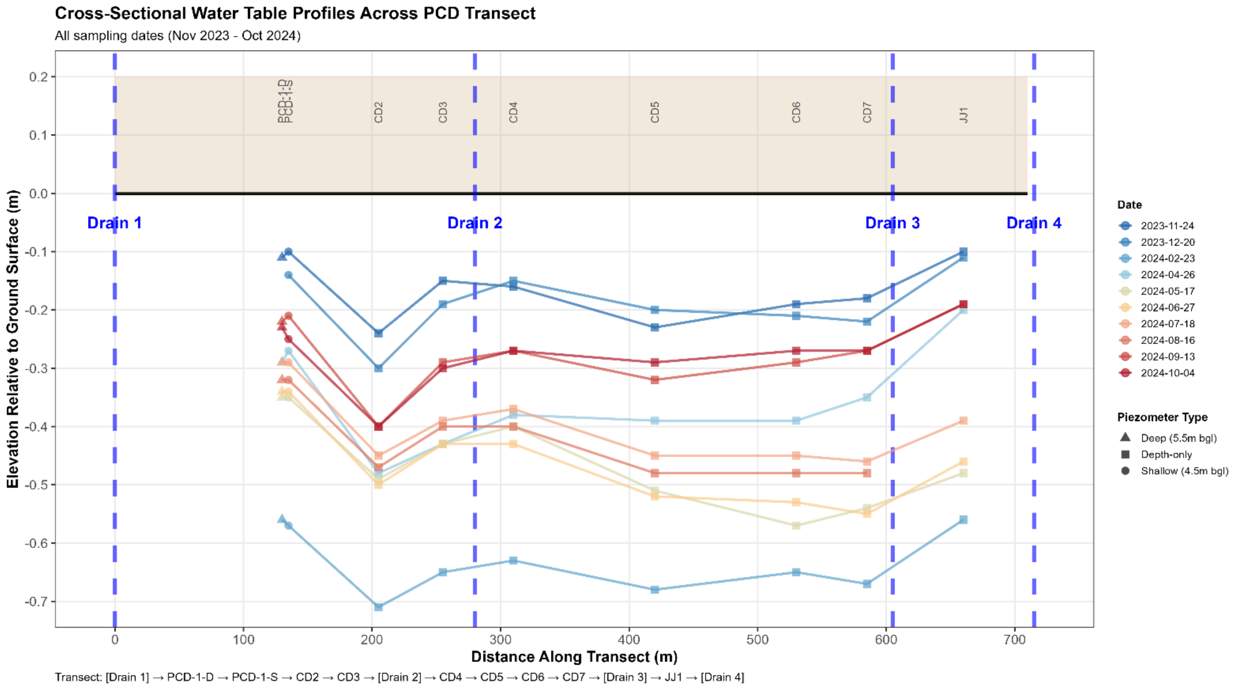

Figure 39: Cross-sectional water table profiles across PCD transect.

Monitoring commenced in late spring 2023 under blocked conditions. The 24 November 2023 measurement following substantial rainfall (41.59 mm in previous seven days) showed elevated water tables across the transect, indicating blocked drains successfully retained water. By 20 December 2023, despite minimal rainfall (0.93 mm in previous seven days), water tables remained elevated, demonstrating controlled drainage water retention effects.

During February 2024 drought (6.82 mm total rainfall over 23 days, including 11 days with zero precipitation), unblocked drains permitted substantial water table decline. The Drain 1-2 section reached approximately 0.70 m bgl, which was the transect maximum depth, approximately 0.4-0.5 m lower than November 2023 blocked conditions. This drawdown may have exposed organic soil horizons to atmospheric oxygen, potentially accelerating mineralisation and nutrient mobilisation (Section 5.1.3.1).

Following April 2024 rainfall (63.72 mm total) with drainage shifted to blocked configuration, water tables recovered by 0.2-0.4 m despite modest preceding-week rainfall (2.88 mm), demonstrating that blocked drainage captured and retained autumn precipitation.

During May-August 2024, progressive water table decline occurred despite substantial cumulative rainfall (143.81 mm over May-July). With drains unblocked, winter rainfall was discharged rather than retained, and by August water tables approached February minimum levels. This pattern confirms findings by Tähtikarhu et al. (2025) that controlled drainage can reduce drain discharge by hundreds of millimetres over 18-month periods, with corresponding groundwater level increases depending on peat thickness and hydrogeological connectivity.

The persistent undulating water table profile demonstrates hydraulic gradients controlling groundwater flow and nutrient transport. Groundwater flows from higher to lower water table elevations (Younger, 2007), and the pattern of elevated water midway between drains with depressions at drain locations creates hydraulic gradients driving lateral flow convergence on the drainage network. Elevated ammonia-N concentrations in groundwater (Section 5.1.3) flow laterally following these gradients, concentrating nutrient loads into tile and surface drains. Monaghan et al. (2016) documented that subsurface drainage accounted for approximately 80% of total soluble nitrogen discharged in New Zealand pastoral systems, with mean annual losses of 30-56 kg N ha^{-1}.

A field photograph (Figure 40) captured on 30 November 2023 at PCD-1 shows surface drain 2 during blocked drainage, with elevated water maintained by controlled outlet. The exposed drain bank had recently been mechanically cleared and reveals a vertical moisture gradient:

- Lower section (near water level): Dark, fully saturated soil indicating continuous contact with ponded water in the drain;
- Middle section: Progressively lighter-coloured soil showing the capillary fringe zone where water is drawn upward from the drain through capillary forces;
- Upper section: Drier surface soil, though still showing evidence of moisture likely maintained through both capillary rise and rainfall.

This demonstrates how blocked drainage maintains near-saturated conditions throughout the organic-rich surface horizon, which is the zone of maximum nutrient production through decomposition (Section 5.1.3.1).

6.1.2 Effects on surface water nutrient concentrations

Surface water samples collected at SCD-1 (the surface drain receiving discharge from the PCD transect area) were analysed to evaluate potential controlled drainage effects on nutrient export. To ensure data accuracy, samples were conservatively categorized as "blocked" or "unblocked" only when collected within seven days of a documented blockage status check, yielding a limited dataset of three blocked samples and ten unblocked samples suitable for comparison.

Statistical comparison revealed no significant differences in nutrient concentrations between drainage management regimes for ammonia-N, nitrate-N, DRP, total nitrogen, or total phosphorus. The only significant difference was dissolved oxygen, which was substantially lower during blocked periods (median 0.28 mg/L) compared to unblocked periods (median 4.01 mg/L) (Mann-Whitney $U=3.0$, $p=0.049$). This dissolved oxygen reduction is mechanistically consistent with blocked drainage maintaining more anoxic conditions in the soil profile and drainage water.

Figure 40: Controlled drainage in operation during summer season, with artificially-elevated drainage water effects seen in the soil profile moisture at this drain cutting. Taken November 30, 2023 near PCD-1 site.

The absence of statistically significant nutrient differences must be interpreted cautiously given very small sample size for blocked periods (n=3), which provides limited statistical power. Nitrate-N showed a trend toward lower concentrations during blocked periods (median 0.019 versus 0.064 g/m^3), consistent with mechanistic expectations that anoxic conditions suppress nitrification (Section 5.1.3.1), but this did not reach statistical significance (p=0.139).

Additional data collection with frequent blockage documentation and synchronised sampling would be needed to properly quantify controlled drainage nutrient effects. Despite limited direct nutrient evidence, the demonstrated water table manipulation (0.4-0.5 m differences), maintenance of anoxic conditions (lower dissolved oxygen, p=0.049), and established mechanistic understanding (Section 5.1.3.1) support controlled drainage as theoretically sound nutrient source control.

6.2 Borrow pit treatment potential and optimisation

Borrow pits are large excavated water bodies created during construction of the Moutoa Floodway stopbanks in the early 1960s using a walking dragline (Evans, 1964). These excavations reached depths of 4.9-5.5 m and widths of 27-30 m. Following excavation, the pits filled with water from rainfall, surface runoff, and groundwater seepage, becoming permanent features within the drainage network.

Four sampling sites were established within floodway borrow pits (SBP-1 to SBP-4), plus one long-term site (Levin Road, monitored since December 2022). The hydrological characteristics of these borrow pits differs from the surrounding drainage network. Their greater depth relative to surface drains creates opportunities for stratification and potentially longer hydraulic residence times compared to the rapidly flowing drainage channels. The transition from shallow, fast-flowing agricultural drains to deeper, more static water bodies changes hydraulic conditions that may influence nutrient transformation and attenuation processes.

6.2.1 Current water quality comparison

Statistical comparison between borrow pit water quality (n=5 sites, 101 samples) and other surface water sites (n=21 sites, 356 samples) revealed significantly lower nutrient concentrations in borrow pits across all measured parameters (Table 19, Figure 41).

Table 19: Kruskal-Wallis test results and median differences for borrow pit water quality parameters versus all other water quality sampling sites.

Parameter	n	n	Median Borrow Pit	Median Other	Median Diff	H statistic	p value
Ammonia-N (g/m^3)	101	356	0.042	0.43	0.389	79.583	<2e-16
Nitrate-N (g/m^3)	101	356	<0.020	0.114	0.095	15.657	7.59E-05
DRP (g/m^3)	101	356	0.097	0.15	0.052	13.439	0.000246
Total P (g/m^3)	69	182	0.33	0.685	0.355	19.058	1.27E-05
Dissolved Oxygen (mg/L)	69	184	7.87	3.725	-4.145	39.451	3.36E-10
pH	69	184	7.62	6.81	-0.81	55.204	1.09E-13
Temperature (°C)	69	184	13.84	14.625	0.785	1.184	0.277

Ammonia-N concentrations in borrow pits (median 0.042 g/m^3) were significantly lower than other surface water sites (median 0.43 g/m^3), suggesting a potential attenuation option in the context of the catchment's characteristic elevated ammonium levels. Nitrate-N concentration was similarly lower, while phosphorus concentration was also lower. Most notably, borrow pits maintained substantially higher dissolved oxygen (median 7.87 mg/L versus 3.73 mg/L) and pH (median 7.62 versus 6.81). The well-oxygenated, near-neutral conditions contrast with the often hypoxic, acidic drainage waters, suggesting favourable environments for biological nutrient transformation. Temperature showed no significant difference between borrow pits and other sites.

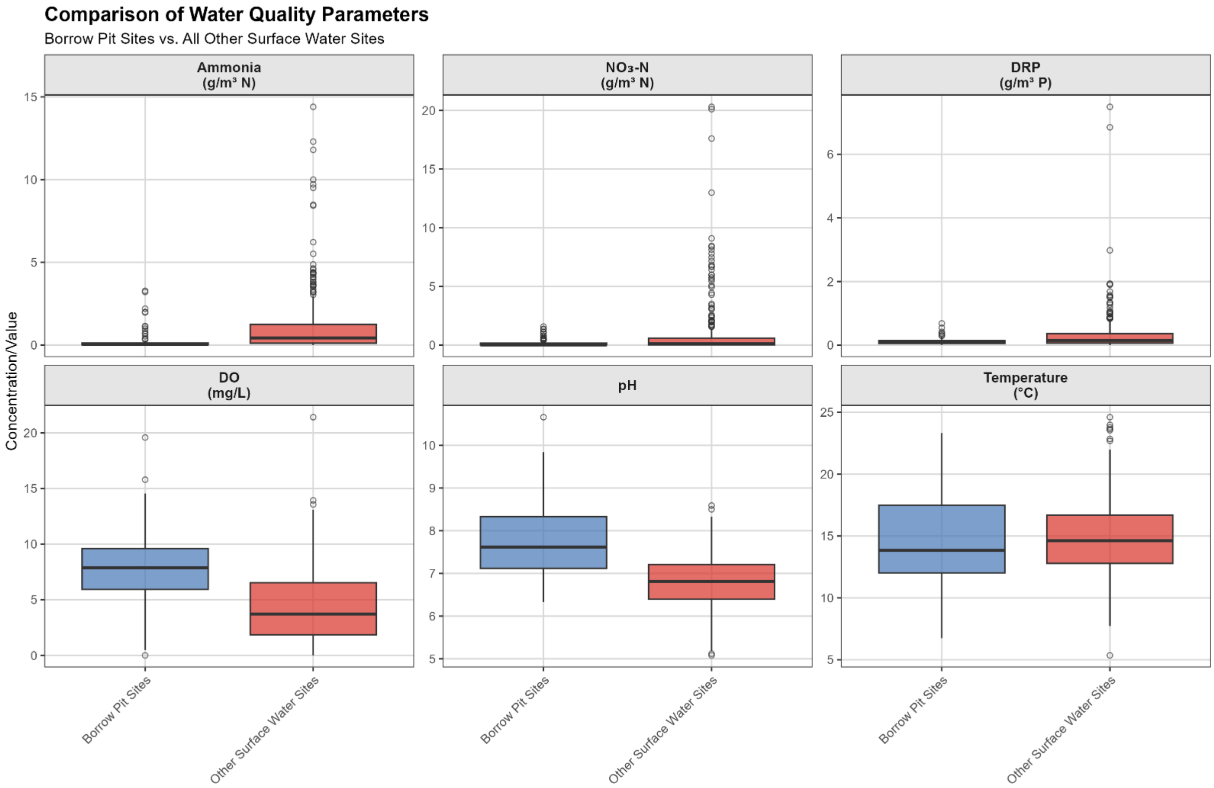

Figure 41: Comparison of water quality parameters in borrow pit sites against all other surface water sampling sites.

6.2.2 Potential treatment mechanisms

The significantly lower nutrient concentrations in borrow pits compared to the surrounding drainage network shows the potential for these water bodies to function as nutrient sinks or attenuators within the agricultural landscape.

Percentage reductions in median nutrient concentrations between the drainage network and borrow pits (Table 20) were calculated as: [(Median drains – Median borrow pits) / Median drains] × 100. This metric provides an order-of-magnitude indication of nutrient difference, though it represents a comparative assessment rather than precise attenuation efficiency. True attenuation efficiency would require paired influent-effluent sampling at individual pits with known residence times and water balance data. The calculation assumes borrow pit catchments experience similar land use and groundwater conditions to surrounding agricultural areas, and that drainage water entering pits originates primarily from comparable source areas. Despite these limitations, the concentration differences provide useful context for considering treatment options across nutrient species.

Table 20: Percentage reductions in median nutrient concentrations between the borrow pits and surface drain.

Nutrient	Median Drains (mg/L)	Median Borrow Pits (mg/L)	Reduction Percent
Ammonia-N	0.43	0.042	94.5
Nitrate-N	0.039	0.019	51.3
DRP	0.206	0.097	52.8
SIN	1.5	0.081	94.6
Total N	3.4	2.2	35.3
Total P	0.69	0.33	52.2

Several treatment mechanisms may contribute to this nutrient concentration difference:

Sedimentation and particulate removal

The transition from shallow, rapidly flowing drains to deeper, static borrow pits creates ideal conditions for gravitational settling of particulate-bound nutrients. The reduced flow velocity in borrow pits allows suspended sediments, organic matter, and associated phosphorus to settle out of the water column. This physical removal mechanism is particularly effective for Total Phosphorus, as evidenced by the 52.2% reduction in median TP concentrations (0.33 g/m^3 in borrow pits versus 0.69 g/m^3 in drains). The deeper profile of borrow pits (4.9-5.5 m) compared to surface drains provides settling depth, enhancing removal efficiency.

Biological uptake and transformation

Elevated dissolved oxygen (median 7.87 mg/L) and near-neutral pH (median 7.62) support aerobic biological communities that assimilate dissolved nutrients. Aquatic vegetation can directly uptake nitrogen and phosphorus, temporarily sequestering nutrients in biomass. The 94.5% difference in ammonia-N concentrations may be caused by active nitrification processes converting ammonia-N to nitrate-N in the aerobic water column, although these mechanisms are not properly understood in this area and further research would be required to confirm any other processes which may be occurring. However, simultaneously low nitrate-N concentrations (83% lower median) indicate that nitrification alone cannot explain the difference in nitrogen patterns. This suggests coupled nitrification-denitrification processes, with nitrate-N converted to nitrogen gas in anoxic sediment layers, which is a well-documented nitrogen attenuation pathway in treatment ponds and wetlands (Vymazal, 2007).

Hydraulic retention and dilution

Greater water volume in borrow pits relative to drainage channels increases hydraulic retention time, providing extended opportunity for biological and chemical transformation processes. Large surface area facilitates atmospheric oxygen exchange, maintaining elevated DO levels. Direct rainfall or groundwater inputs may also contribute to dilution of incoming nutrient-rich drainage water.

Comparison to treatment pond literature

Brunet et al. (2021) documented 36% Total Nitrogen and 8% Total Phosphorus removal in eutrophic farm ponds, though nitrate-nitrogen attenuation could reach 64% through particulate nitrogen removal as algal biomass. The borrow pit results show substantial differences in nitrogen concentration (approximately 83-90% difference in dissolved inorganic nitrogen species), though direct comparison is complicated by methodological differences, where the present study compares borrow pit water quality against surrounding drains rather than measuring influent-effluent differences within individual ponds. This performance without design optimisation suggests potential for enhanced nutrient mitigation through strategic management.

6.2.3 Borrow pit treatment potential: optimisation and expansion

The nutrient concentration differences between drainage waters and borrow pits suggests potential for enhanced treatment through strategic optimisation. This must be hedged by acknowledging the limitations in this research, particularly given that only nutrient concentrations were measured, rather than flows. However, several targeted interventions could improve effectiveness while maintaining passive operation advantages.

Hydraulic retention time management

Hydraulic retention time (HRT) is critical for treatment efficiency. Optimal HRT of 24-48 hours has been identified for effective nutrient retention (Bai et al., 2025), though this must be balanced against flood control requirements. Current borrow pits lack controlled outlets, meaning HRT varies with inflow and likely decreases during high-flow events when nutrient loads are greatest. Installing adjustable outlet structures could extend residence times during elevated nutrient loading periods while maintaining drainage capacity during floods. Simple gate structures would allow seasonal water level adjustments, raising levels during low-flow periods to increase volume and residence time, while lowering during high-flow events to maintain flood protection.

Vegetation establishment and management

Macrophyte establishment is a cost-effective optimisation strategy with multiple benefits. Emergent and submerged vegetation would directly assimilate dissolved nutrients, sequestering nitrogen and phosphorus in biomass that could be periodically harvested. Beyond direct uptake, vegetation provides substrate for beneficial microbial communities, enhances sedimentation through flow velocity reduction, and creates habitat for aquatic organisms contributing to ecosystem-based treatment. NIWA guidelines emphasise vegetation as essential for nutrient removal, with planted systems substantially outperforming unvegetated controls (Tanner et al., 2020). However, vegetation establishment in deeper borrow pits (4.9-5.5 m) would require shallow zones or marginal shelves where plants can establish, achieved through selective excavation or installation of submerged platforms.

Sediment management

Over time, sedimentation will gradually reduce borrow pit volume and treatment capacity. Periodic dredging to remove accumulated nutrient-rich sediments would restore treatment volume and prevent the transition from nutrient sink to source that occurs in aging ponds as sediment sorption capacity saturates (Land et al., 2016). Given 60 years of operation since excavation, surveys comparing current depths to original records would quantify accumulation rates and inform maintenance timing.

Chemical enhancement

Aluminium or iron-based coagulants have demonstrated treatment improvements in agricultural drainage systems. While chemical enhancement could significantly improve phosphorus removal (Bachand et al., 2019a,b), it introduces operational complexity, ongoing costs, and potential risks requiring careful management. The passive, low-maintenance nature of current systems represents a key advantage, while chemical treatment would require dosing infrastructure, regular chemical supply, and monitoring. This intervention might be most appropriate if other optimisation strategies prove insufficient to meet water quality targets.

6.2.4 Expansion potential: pumping additional catchment areas

Current drainage configuration and expansion opportunity

The Moutoa-Whirokino drainage scheme currently routes some sub-catchment areas to discharge through borrow pits, while other areas bypass them entirely, draining directly to pump stations and the river. This presents an opportunity to trial borrow pit treatment efficacy by redirecting high-nutrient drainage waters to flow through borrow pits before final discharge.

The spatial analysis of surface water nutrient patterns found substantial variety across the catchment, with some areas having consistently elevated nutrient concentrations. Surface water monitoring revealed that silt loam-associated sites showed the highest ammonia-N concentrations, and groundwater analysis found even stronger soil type associations, with peaty soils found to have elevated nutrient concentrations. Ammonia-N in peaty soil groundwater ranged from 6.79 to 18.30 g/m^3 (median values), compared to 1.47-1.54 g/m^3 in sandy soils and 1.17-9.45 g/m^3 in silt loam groundwaters. Dissolved reactive phosphorus followed similar patterns, with peaty groundwater medians (3.15-4.56 g/m^3) substantially exceeding silt loam (0.09-2.61 g/m^3) and sandy soils (0.14-0.18 g/m^3).

These spatial patterns indicate that targeting organic soil sub-catchments and those surface water sampling areas with consistently elevated nutrient levels for routing through borrow pits, could maximise nutrient load reduction per unit of infrastructure investment. Drainage waters from peaty areas carry the highest baseline nutrient concentrations, meaning treatment of these waters could achieve the greatest absolute mass attenuation even with modest percentage reductions.

Infrastructure requirements and considerations

Expanding borrow pit utilisation would require several infrastructure modifications:

- Existing pump stations may require capacity upgrades or additional pumps to handle increased flows if drainage is redirected. The specific requirements would depend on existing pump capacities relative to flows to be redirected, drainage network hydraulics, and whether pumping can be staged to avoid peak simultaneous demands.

- New infrastructure would be needed to route drainage from target areas to borrow pits. The choice between pipelines and open channels depends on distances involved, elevation changes, land availability for channel construction, and maintenance considerations. Channels offer lower construction costs but require regular vegetation management, while pipelines involve higher capital costs but lower ongoing maintenance.
- Flow distribution infrastructure would allow operators to manage which sub-catchments drain to borrow pits versus direct discharge. This flexibility enables adaptive management, directing flows to treatment during periods of adequate residence time while bypassing during extreme flood events when flood control takes priority over treatment.

Detailed cost assessment is beyond the scope of this research, but expansion would require capital investment in pumping, conveyance, and control infrastructure, plus ongoing operational costs for electricity and maintenance. Economic analysis would need to compare these costs against alternative nutrient mitigation approaches and the value of nutrient load reductions achieved.

Residence time and capacity trade-offs

A critical consideration is balancing increased inflow against hydraulic residence time. Current treatment performance reflects existing residence time from current inflow volumes. Substantially increasing inflow while maintaining fixed volume would proportionally reduce residence time, potentially degrading treatment efficiency. This could be addressed through: additional excavation creating new treatment cells; configuring multiple pits as treatment trains maintaining total residence time with operational flexibility; routing high-nutrient drainage during lower flow periods while bypassing during floods; or implementing controlled drainage principles to direct flows during critical periods.

Geographic targeting for maximum benefit

Strategic geographic targeting should prioritise drainage routing based on nutrient concentrations and load contributions:

1. First priority: Organic soil sub-catchments with demonstrated high ammonium and DRP concentrations in both groundwater and surface water. These areas contribute the greatest baseline loads and would benefit most from treatment.
2. Second priority: Areas with episodic high nutrient spikes identified in the temporal analysis, particularly locations showing the highest maximum concentrations during storm events, even if median values are moderate.
3. Third priority: Areas with intensive management or known point sources (such as effluent application zones) where localised treatment could intercept nutrient loads before catchment-scale mobilisation.

The existing spatial dataset (Section 4.3.2) provides a foundation for this targeting, though additional sampling to characterise currently un-monitored sub-catchments may reveal additional high-priority areas.

6.3 Integration with other mitigation strategies

Borrow pit optimisation and expansion should be viewed as complementary to, rather than replacing, other nutrient mitigation approaches:

- Controlled drainage: Managing water table elevations in organic soil areas through controlled drainage (Section 2.4.1 Literature Review) could reduce baseline nutrient mobilisation, decreasing the treatment load on downstream borrow pits and other mitigation features. The combination of source control (controlled drainage) and treatment (borrow pits) may achieve greater overall reduction than either approach alone.
- Vegetated drainage ditches: The extensive drainage network feeding borrow pits could incorporate vegetation to provide preliminary treatment before borrow pit discharge. Research shows vegetated ditches achieve 44% total nitrogen and 43% total phosphorus removal (Bai et al., 2025), meaning sequential treatment (ditch → borrow pit → river) could compound removal efficiency.
- Constructed wetlands: Purpose-built wetlands designed specifically for ammonium treatment (requiring aerobic-anaerobic sequential zones, Toet et al., 2005) could treat high-concentration drainage from organic soil areas. Borrow pits could then provide polishing treatment of wetland discharge before final discharge.
- Treatment ponds with chemical enhancement: If dissolved phosphorus removal becomes the priority management objective and passive approaches prove insufficient, chemical enhancement of borrow pits or construction of dedicated alum-dosed treatment ponds (Bachand et al., 2019a & b) could target this specific challenge.

Catchment-scale nutrient management requires multiple, strategically deployed interventions working together. Borrow pits represent an underutilised existing asset that, with modest optimisation and strategic expansion, could contribute meaningfully to catchment nutrient reduction targets while maintaining the advantages of passive operation, minimal ongoing costs, and integration with essential drainage infrastructure.

Chapter 7

7. Synthesis and Conclusion

7.1 Research context and objectives

This research investigated nutrient dynamics and transport pathways in the Moutoa-Whirokino catchment, a 6,000-hectare artificially drained former wetland in the lower Manawatu River system. The study addressed critical knowledge gaps regarding nutrient behaviour in intensively managed drained organic soils under New Zealand's pastoral dairy systems, where international research has predominantly focused on nitrate-nitrogen losses from Northern European and North American peatlands.

Preliminary monitoring revealed an unusual pattern: elevated ammonia-nitrogen and dissolved reactive phosphorus in surface drains, alongside unexpectedly low nitrate-nitrogen levels. This nitrogen speciation pattern differed markedly from typical New Zealand agricultural drainage systems, suggesting unique nutrient transformation pathways. The Manawatu Estuary's designation as a Wetland of International Importance under the Ramsar Convention creates particular urgency for understanding and mitigating nutrient losses from these artificially drained systems.

7.2 Key findings

This research confirmed an ammonium-dominated nitrogen regime in both shallow groundwater and surface drainage waters, fundamentally distinct from typical New Zealand agricultural systems. The mechanistic framework identified three interconnected processes: organic matter decomposition releasing ammonium from peaty soils, predominantly anoxic groundwater conditions suppressing nitrification, and potential DNRA activity converting nitrate back to ammonium. Soil type emerged as the dominant control, with peaty soil groundwater showing substantially higher nutrient concentrations than sandy soils.

Despite elevated groundwater nutrient concentrations, weak correlations with surface water quality revealed modification during transport, creating opportunities for in-system treatment approaches.

7.3 Mitigation strategy potential

Controlled drainage demonstrated water table manipulation capacity and maintained lower dissolved oxygen during blocked periods, supporting its theoretical basis as nutrient source control. The most significant finding was the potential passive treatment performance of floodway borrow pits, where 94.5% ammonia-N concentration and 52.2% phosphorus differences were found between borrow pits and other drains. Potential expansion opportunities exist through routing high-nutrient drainage water from organic soil sub-catchments through borrow pits before final discharge.

7.4 Contribution to scientific understanding

The characterisation of an ammonium-dominated nitrogen system in New Zealand's drained organic soils extends international understanding beyond the nitrate-focused research from Northern European and North American peatlands. The mechanistic framework linking organic matter decomposition, redox effect, and potential DNRA activity provides explanation for nitrogen speciation patterns that differ from both mineral agricultural soils and northern hemisphere drained peatlands.

The identification of vertical nitrogen speciation gradients (nitrate-dominated at shallow soil depth transitioning to ammonium-dominated at groundwater depth) reveals critical redox transition zones associated with nitrogen export forms. This has important implications for mitigation strategy selection, as conventional denitrification-based treatment systems designed for nitrate attenuation perform differently when treating ammonium-dominated drainage.

The weak correlations between groundwater and surface water nutrient concentrations demonstrate fundamental challenges in predicting receiving water quality from source area characterisation alone, emphasising the importance of integrated monitoring approaches that capture both source dynamics and transformation processes operating during transport.

The potential for borrow pit treatment performance represents one of the few documented assessments of passive agricultural drainage treatment in New Zealand drained organic soil systems, demonstrating that retrofitting existing drainage infrastructure may provide water quality improvements without requiring purpose-built treatment wetlands or intensive operational inputs.

7.5 Management implications and recommendations

7.5.1 Prioritising organic soil sub-catchments

Management efforts should prioritise peaty and organic-enriched silt loam sub-catchments, where groundwater serves as a persistent source delivering ammonia-N and DRP through baseflow conditions. In these areas, source control through water table management offers greatest potential for reducing baseline nutrient generation. Sandy soil areas exhibit substantially lower groundwater nutrient concentrations and may require less intensive intervention.

7.5.2 Integrated source control and treatment approach

The research findings support a dual strategy combining source reduction with in-system treatment. Controlled drainage managing water table elevations in organic soil areas can reduce baseline nutrient mobilisation by limiting aerobic decomposition. Sequential treatment through borrow pit systems can provide substantial load reduction even where source control proves incomplete. Complementary interventions including vegetated drainage ditches and purpose-built wetlands could compound overall attenuation efficiency.

7.5.3 Adaptive management and monitoring

The episodic nature of nutrient export events indicates that adaptive drainage management could reduce storm-driven loads. Flow distribution infrastructure allowing operators to direct high-nutrient drainage to treatment during periods of adequate residence time while bypassing during extreme floods would provide operational flexibility. Long-term monitoring should continue to capture inter-annual variability and evaluate management intervention effectiveness, with flow measurements integrated to enable load-based assessment.

7.5.4 Protecting the Manawatu Estuary

The universal exceedance of dissolved reactive phosphorus targets across all monitoring sites indicates that phosphorus management represents a critical catchment-wide challenge requiring coordinated intervention. Strategic investment in borrow pit optimisation and expansion, combined with controlled drainage implementation in organic soil areas, could meaningfully contribute to catchment-scale nutrient reduction targets while maintaining drainage capability essential for continued agricultural land use.

7.6 Research limitations and future direction

7.6.1 Study limitations

The 11-month intensive monitoring period captured seasonal patterns but may not represent long-term conditions, though the extended catchment group dataset demonstrated the study period represented typical conditions. Limited controlled drainage nutrient data provided insufficient statistical power to definitively quantify treatment effects, though demonstrated water table manipulation and dissolved oxygen reduction support mechanistic expectations. Concentration-based analysis without comprehensive flow measurements prevents definitive load calculations, while the borrow pit assessment compared water quality in pits against surrounding drains rather than measuring influent-effluent differences at individual pits with known residence times.

7.6.2 Priority future research

Key priorities include: controlled drainage trials with replicated treatments and frequent monitoring to definitively quantify nutrient reduction potential; borrow pit hydraulic studies through influent-effluent monitoring and water balance determination; load-based assessment through flow monitoring equipment; DNRA quantification through isotope tracer studies; long-term monitoring of legacy nutrient dynamics; catchment-scale modelling integrating soil type, groundwater depth, and drainage infrastructure; and comparative assessment across other drained organic soil catchments to test whether the ammonium-dominated signature is widespread or site-specific.

7.7 Concluding statement

This research has addressed critical knowledge gaps regarding nutrient dynamics in New Zealand's drained organic soil systems under intensive pastoral dairy farming. The identification of an ammonium-dominated nitrogen regime fundamentally distinct from typical New Zealand agricultural drainage provides new understanding of how organic matter content, redox conditions, and artificial drainage infrastructure interact to create unique nutrient transformation pathways.

The mechanistic framework established through integrated groundwater and surface water monitoring explains both spatial and temporal patterns, while the weak correlations between groundwater sources and surface water quality demonstrate that substantial nutrient modification occurs during transport, creating opportunities for intervention beyond source control alone.

The potential for passive treatment options in existing borrow pit infrastructure combined with the water table manipulation capacity of controlled drainage provides pathways for further investigation of reduction nutrient loads from drained organic soil catchments while maintaining essential drainage capability.

Most importantly, this research has direct relevance for protecting the Manawatu Estuary's status as a Wetland of International Importance. The universal exceedance of dissolved reactive phosphorus targets combined with widespread ammonia-N exceedances confirms the urgency of coordinated nutrient management. The spatial targeting framework developed through this research, which prioritises organic soil sub-catchments with the highest baseline groundwater concentrations provides clear guidance for strategic intervention deployment that could meaningfully reduce cumulative nutrient loading to this internationally significant receiving environment.

By advancing scientific understanding of drained peatland nutrient dynamics in New Zealand contexts, exploring treatment potential in existing infrastructure, and identifying practical pathways for integrated source control and treatment approaches, this research contributes to the broader effort to balance intensive agricultural productivity with the protection of freshwater ecosystems and internationally valued estuarine environments.

Bibliography

An, S., & Verhoeven, J. T. (2019). Wetland functions and ecosystem services: Implications for wetland restoration and wise use. In Wetlands: Ecosystem Services, Restoration and Wise Use (pp. 1-10). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-030-14861-4_1

Appelo, C. A. J., & Postma, D. (2005). Geochemistry, groundwater and pollution. 2nd. *Ed. Balkema, Rotterdam*.

Baalbergen, J. (2024, May 14). Moutoa floodgates to get upgrade later this year. *Horowhenua Chronicle*. https://www.nzherald.co.nz/horowhenua-chronicle/news/moutoa-floodgates-to-get-upgrade-later-this-year/Y326QI7WPJDDFHZLVHSZ7DKPMY/

Bachand, P. A., Bachand, S. M., Kraus, T. E., Stern, D., Liang, Y. L., & Horwath, W. R. (2019b). Sequestration and transformation in chemically enhanced treatment wetlands: DOC, DBPPs, and nutrients. Journal of Environmental Engineering, 145(8), 04019044. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001536

Bachand, S. M., Kraus, T. E., Stern, D., Liang, Y. L., Horwath, W. R., & Bachand, P. A. (2019a). Aluminum-and iron-based coagulation for in-situ removal of dissolved organic carbon, disinfection byproducts, mercury and other constituents from agricultural drain water. Ecological Engineering, 134, 26-38. https://doi.org/10.1016/j.ecoleng.2019.02.015

Bai, J., Qin, Y., Zhao, J., & Song, Y. (2025). Investigation of agricultural nutrient removal by ecological ditches using meta-analysis. Agriculture, Ecosystems & Environment, 380, 109401. https://doi.org/10.1016/j.agee.2024.109401

Ballantine, D. J., & Davies-Colley, R. J. (2010). Water quality trends at NRWQN sites for the period 1989-2007. https://www.environment.govt.nz/assets/Publications/Files/final-report-water-quality-trends-NRWQN.doc

Barkle, G., Stenger, R., Moorhead, B., & Clague, J. (2021). The importance of the hydrological pathways in exporting nitrogen from grazed artificially drained land. *Journal of Hydrology*, *597*, 126218. https://doi.org/10.1016/j.jhydrol.2021.126218

Bataille, C. Y., Malinen, S. K., Yletyinen, J., Scott, N., & Lyver, P. O. B. (2021). Relational values provide common ground and expose multi-level constraints to cross-cultural wetland management. People and Nature, 3(4), 941-960. https://doi.org/10.1002/pan3.10244

Becher, M., Kalembasa, D., Kalembasa, S., Symanowicz, B., Jaremko, D., & Matyszczak, A. (2023). A new method for sequential fractionation of nitrogen in drained organic (Peat) soils. International Journal of Environmental Research and Public Health, 20(3), 2367. https://doi.org/10.3390/ijerph20032367

Beeckman, F., Motte, H., & Beeckman, T. (2018). Nitrification in agricultural soils: impact, actors and mitigation. Current opinion in Biotechnology, 50, 166-173. https://doi.org/10.1016/j.copbio.2018.01.014

Bethune, M. G., Selle, B., & Wang, Q. J. (2008). Understanding and predicting deep percolation under surface irrigation. Water Resources Research, 44(12). https://doi.org/10.1029/2007WR006380

Boonman, J., Harpenslager, S. F., van Dijk, G., Smolders, A. J., Hefting, M. M., van de Riet, B., & van der Velde, Y. (2024). Redox potential is a robust indicator for decomposition processes in drained agricultural peat soils: A valuable tool in monitoring peatland wetting efforts. Geoderma, 441, 116728. https://doi.org/10.1016/j.geoderma.2023.116728

Briggs, L. J., & McCall, A. G. (1904). An artificial root for inducing capillary movement of soil moisture. Science, 20(513), 566-569. https://doi.org/10.1126/science.20.513.566

Brunet, C. E., Gemrich, E. R., Biedermann, S., Jacobson, P. J., Schilling, K. E., Jones, C. S., & Graham, A. M. (2021). Nutrient capture in an Iowa farm pond: Insights from high-frequency observations. Journal of environmental management, 299, 113647. https://doi.org/10.1016/j.jenvman.2021.113647

Bullock, A., & Acreman, M. (2003). The role of wetlands in the hydrological cycle. Hydrology and Earth system sciences, 7(3), 358-389. https://doi.org/10.5194/hess-7-358-2003

Burkitt, L. L., Winters, J. L., & Horne, D. J. (2017). Sediment and nutrient losses under winter cropping on two Manawatu hill country soils. Journal of New Zealand Grasslands, 27-33. https://doi.org/10.33584/jnzg.2017.79.554

Carlson, W. T., Houlbrooke, D. J., Lucci, G. L. & Sprosen M. S. (2018). Drainage water quality on two Waikato peat dairy farms. Farm environmental planning – Science, policy and practice. https://www.massey.ac.nz/~flrc/workshops/18/Manuscripts/Paper_Carlson_2018.pdf

Chappell, P. R. (2015). The Climate and Weather of Manawatu-Wanganui. *NIWA Science and Technology Series 66*, 40. https://niwa.co.nz/sites/default/files/NIWA_ManawatuWanganui_Climate_WEB.PDF

Cheng, Y., Elrys, A. S., Merwad, A. R. M., Zhang, H., Chen, Z., Zhang, J., Cai, Z., & Müller, C. (2022). Global patterns and drivers of soil dissimilatory nitrate reduction to ammonium. Environmental Science & Technology, 56(6), 3791-3800. https://doi.org/10.1021/acs.est.1c07997

Cherry, J. A., & Freeze, R. A. (1979). *Groundwater* (Vol. 370). Englewood Cliffs, NJ: Prentice-Hall.

Clapcott, J.E., & Young, R.G. (2009). Temporal variability in ecosystem metabolism of rivers in the Manawatu-Wanganui Region. Cawthron Report No. 1672. https://www.envirolink.govt.nz/assets/Envirolink/702-HZLC66.pdf

Clement, A. J., Sloss, C. R., & Fuller, I. C. (2010). Late quaternary geomorphology of the Manawatu coastal plain, North Island, New Zealand. *Quaternary International*, *221*(1-2), 36-45. https://doi.org/10.1016/j.quaint.2009.07.005

Cohen, J. (1988). Statistical power analysis for the behavioral sciences (2nd ed.). Lawrence Erlbaum Associates.

Cowie, J. D. (1978). Soils and agriculture of Kairanga County, North Island, New Zealand. https://doi.org/10.7931/dl1-sbb-033

Cowie, J. D., & Rijkse, W. C. (1977). Soils of Manawatu County, North Island, New Zealand. https://doi.org/10.7931/DL1-SSR-30-1

Cowie, J. D., & Smith, B. A. J. (1958). Soils and agriculture of Oroua Downs, Taikorea, and Glen Oroua Districts, Manawatu County. https://doi.org/10.7931/dl1-sbb-016

Daughney, C. J. (2006). A national protocol for state of the environment groundwater sampling in New Zealand. Ministry for the Environment. https://environment.govt.nz/assets/Publications/Files/national-protocol-groundwater-dec06-updated.pdf

Davies-Colley, R.J. (2013). River water quality in New Zealand: an introduction and overview. Ecosystem services in New Zealand: conditions and trends. 432–447. https://www.landcareresearch.co.nz/assets/Publications/Ecosystem-services-in-New-Zealand/2_12_Davie-Colley.pdf

Department of Conservation. (n.d.). Manawatu Estuary. https://www.doc.govt.nz/parks-and-recreation/places-to-go/manawatu-whanganui/places/manawatu-estuary/

Di, H. J., & Cameron, K. C. (2002). Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient cycling in agroecosystems, 64(3), 237-256. https://doi.org/10.1023/A:1021471531188

Diaz, R. J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *science*, *321*(5891), 926-929. DOI:10.1126/science.1156401

Dudley, B. D., Burge, O. R., Plew, D., & Zeldis, J. (2020). Effects of agricultural and urban land cover on New Zealand's estuarine water quality. New Zealand Journal of Marine and Freshwater Research, 54(3), 372-392. https://doi.org/10.1080/00288330.2020.1729819

Elliott, A. H., Alexander, R. B., Schwarz, G. E., Shankar, U., Sukias, J. P. S., & McBride, G. B. (2005). Estimation of nutrient sources and transport for New Zealand using the hybrid mechanistic-statistical model SPARROW. Journal of Hydrology (New

Zealand), 44(1), 1-27.
https://search.informit.org/doi/abs/10.3316/INFORMIT.585640954449381

Elsayed, A., Rixon, S., Zeuner, C., Levison, J., Binns, A., & Goel, P. (2023). Text mining-aided meta-research on nutrient dynamics in surface water and groundwater: Popular topics and perceived gaps. Journal of Hydrology, 626, 130338. https://doi.org/10.1016/j.jhydrol.2023.130338

Emerson, K., Russo, R. C., Lund, R. E., & Thurston, R. V. (1975). Aqueous ammonia equilibrium calculations: effect of pH and temperature. *Journal of the Fisheries Board of Canada*, *32*(12), 2379-2383. https://doi.org/10.1139/f75-274

Evans, P. G. (1964). The lower Manawatu river control scheme. *New Zealand Engineering*, *19*(11), 411-420.
https://search.informit.org/doi/10.3316/informit.164593277328691

Forest and Bird. (n.d.). Manawatu River Clean-Up.
https://www.forestandbird.org.nz/branches/horowhenua/manawatu-river-clean

Foxton Historical Society. (n.d.). *Kete Horowhenua*.
https://horowhenua.kete.net.nz/

Friedl, J., De Rosa, D., Rowlings, D. W., Grace, P. R., Müller, C., & Scheer, C. (2018). Dissimilatory nitrate reduction to ammonium (DNRA), not denitrification dominates nitrate reduction in subtropical pasture soils upon rewetting. *Soil Biology and Biochemistry*, *125*, 340-349. https://doi.org/10.1016/j.soilbio.2018.07.024

Garnier, S., Ross, N., Rudis, R., Camargo, A. P., Sciaini, M., & Scherer, C. (2023). viridis: Colorblind-Friendly Color Maps for R. R package version 0.6.4. https://CRAN.R-project.org/package=viridis

Gray, C. W., Talbot, W. D., Manning, M. M., Murray, R., & Roberts, A. H. C. (2024). Phosphorus Leaching from Grazed Pasture Systems in Aotearoa New Zealand: Sources, Forms, Extent and Risk Factors. Opportunities for improved farm and catchment outcomes. Occasional Report No. 36.
https://flrc.massey.ac.nz/workshops/24/Manuscripts/Gray_Col_1.pdf

Grenon, G., Madramootoo, C. A., von Sperber, C., Ebtehaj, I., Bonakdari, H., & Singh, B. (2023). Nutrient release in drainage discharge from organic soils under two different agricultural water management systems. Hydrological Processes, 37(8), e14953. https://doi.org/10.1002/hyp.14953

Grolemund, G., & Wickham, H. (2011). Dates and times made easy with lubridate. Journal of Statistical Software, 40(3), 1-25.
https://doi.org/10.18637/jss.v040.i03

Grønlund, A., Hauge, A., Hovde, A., & Rasse, D. P. (2008). Carbon loss estimates from cultivated peat soils in Norway: a comparison of three methods. Nutrient Cycling in Agroecosystems, 81(2), 157-167. https://doi.org/10.1007/s10705-008-9171-5

Guth, P., Gao, C., & Knorr, K. H. (2023). Electron accepting capacities of a wide variety of peat materials from around the globe similarly explain CO2 and CH4 formation. Global Biogeochemical Cycles, 37(1), e2022GB007459. https://doi.org/10.1029/2022GB007459

Han, L. L., Wang, H., Ge, L., Xu, M. N., Tang, J. M., Luo, L., Li, P., & Kao, S. J. (2023). Transition of source/sink processes and fate of ammonium in groundwater along with redox gradients. Water Research, 231, 119600. https://doi.org/10.1016/j.watres.2023.119600

Harmsworth, G. (2002). Coordinated monitoring of New Zealand wetlands, Phase 2, Goal 2: Maori environmental performance indicators for wetland condition and trend. A Ministry for the Environment SMF project–5105. Landcare Research Contract Report LC, 102(099), 65. https://www.researchgate.net/publication/237144561_Harmsworth_GR_2002_Maori_environmental_performance_indicators_for_wetland_condition_and_trend_Coordinated_Monitoring_of_New_Zealand_Wetlands_Phase_2_Goal_2_A_Ministry_for_the_environment_SMF_Project_5

Hart, S. C., Stark, J. M., Davidson, E. A., & Firestone, M. K. (1994). Nitrogen mineralization, immobilization, and nitrification. Methods of soil analysis: Part 2 microbiological and biochemical properties, 5, 985-1018. https://doi.org/10.2136/sssabookser5.2.c42

Hem, J. D. (1985). *Study and interpretation of the chemical characteristics of natural water* (Vol. 2254). Department of the Interior, US Geological Survey.

Herndon, E. M., Kinsman-Costello, L., Duroe, K. A., Mills, J., Kane, E. S., Sebestyen, S. D., Thompson, A. A., & Wullschleger, S. D. (2019). Iron (oxyhydr)oxides serve as phosphate traps in tundra and boreal peat soils. Journal of Geophysical Research: Biogeosciences, 124(2), 227-246. https://doi.org/10.1029/2018JG004776

Hiscock, K. M., & Bense, V. F. (2014). *Hydrogeology: principles and practice*. John Wiley & Sons.

Horizons Regional Council. (2025). *Horowhenua District Schemes*. [Horowhenua Scheme Meeting Presentation]. https://www.horizons.govt.nz/HRC/media/Media/Flood%20protection/Horowhenua-Schemes-Meeting-Presentation.pdf?ext=.pdf

Horizons Regional Council. (2025a). State of the Environment | Horizons Region 2025. (Report No. 2025/EXT/1962). Horizons Regional Council. https://www.horizons.govt.nz/HRC/media/Media/State-of-the-Environment-Horizons-Region-2025.pdf

Horizons Regional Council. (2025b). Horowhenua District Schemes. [Horowhenua Scheme Meeting Presentation]. https://www.horizons.govt.nz/HRC/media/Media/Flood%20protection/Horowhenua-Schemes-Meeting-Presentation.pdf?ext=.pdf

Horizons Regional Council. (n.d.). Moutoa Sluice Gates & Floodway. https://www.horizons.govt.nz/HRC/media/Media/Flood%20protection/Moutoafloodgates.pdf?ext=.pdf

Horizons Regional Council. (n.d.). *Moutoa Sluice Gates & Floodway*. https://www.horizons.govt.nz/HRC/media/Media/Flood%20protection/Moutoafloodgates.pdf?ext=.pdf

Jarvis, N. J. (2007). A review of non-equilibrium water flow and solute transport in soil macropores: Principles, controlling factors and consequences for water quality. European Journal of Soil Science, 58(3), 523-546. https://doi.org/10.1111/j.1365-2389.2007.00915.x

Kassambara, A. (2023). rstatix: Pipe-Friendly Framework for Basic Statistical Tests. R package version 0.7.2. https://CRAN.R-project.org/package=rstatix

Kassambara, A., & Mundt, F. (2020). factoextra: Extract and Visualize the Results of Multivariate Data Analyses. R package version 1.0.7. https://CRAN.R-project.org/package=factoextra

Kassim, N. Q. B., & Yaacob, A. (2019, September). Nutrients dynamics in peat soil: Influence of fluctuating water table. IOP Conference Series: Earth and Environmental Science, 327(1), 012024. https://doi.org/10.1088/1755-1315/327/1/012024

King, K. W., Williams, M. R., Macrae, M. L., Fausey, N. R., Frankenberger, J., Smith, D. R., Kleinman, P. J. A., & Brown, L. C. (2015). Phosphorus transport in agricultural subsurface drainage: A review. Journal of environmental quality, 44(2), 467-485. https://doi.org/10.2134/jeq2014.04.0163

Kirkland, G. J. (1970). *The Challenge of Flax: A Study of New Zealand Woolpack and Textiles Limited, Foxton and its Employees*. [A Thesis Presented in Partial Fulfilment of the Requirements for the Degree of Master of Arts in Geography at Massey University]. Massey Research Online. https://mro.massey.ac.nz/items/3d785dd5-15e9-4d04-b2bd-18aab4af2660

Knight, C. (2018). Ravaged beauty : an environmental history of the Manawatu. Tōtara Press.

Kovacic, D. A., David, M. B., Gentry, L. E., Starks, K. M., & Cooke, R. A. (2000). Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage (Vol. 29, No. 4, pp. 1262-1274). American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America. https://doi.org/10.2134/jeq2000.00472425002900040033x

Kumwimba, M. N., Meng, F., Iseyemi, O., Moore, M. T., Zhu, B., Tao, W., Liang, T. J., & Ilunga, L. (2018). Removal of non-point source pollutants from domestic sewage and agricultural runoff by vegetated drainage ditches (VDDs): Design, mechanism, management strategies, and future directions. Science of the Total Environment, 639, 742-759. https://doi.org/10.1016/j.scitotenv.2018.05.184

Kynkäänniemi, P., Ulén, B., Torstensson, G., & Tonderski, K. S. (2013). Phosphorus retention in a newly constructed wetland receiving agricultural tile drainage water. Journal of Environmental Quality, 42(2), 596-605. https://doi.org/10.2134/jeq2012.0266

Land, Air, Water Aotearoa (LAWA). (n.d.-a). Manawatū-Whanganui region, Surface Water Zone: Manawatū. https://www.lawa.org.nz/explore-data/manawatu-whanganui-region/water-quantity/surface-water-zones/manawatu

Land, Air, Water Aotearoa (LAWA). (n.d.-b). Manawatū at Whirokino. https://www.lawa.org.nz/explore-data/manawatu-whanganui-region/river-quality/manawatu/manawatu-at-whirokino

Land, Air, Water Aotearoa (LAWA). (n.d.-c). Manawatu at Ōpiki Bridge. https://www.lawa.org.nz/explore-data/manawatu-whanganui-region/river-quality/manawatu/manawatu-at-opiki-bridge

Land, Air, Water Aotearoa (LAWA). (n.d.-d). Manawatū at Teachers College. https://www.lawa.org.nz/explore-data/manawatu-whanganui-region/river-quality/manawatu/manawatu-at-teachers-college

Land, Air, Water Aotearoa (LAWA). (n.d.-e). Manawatū at Upper Gorge (Ferry Reserve). https://www.lawa.org.nz/explore-data/manawatu-whanganui-region/river-quality/manawatu/manawatu-at-upper-gorge-ferry-reserve

Land, Air, Water Aotearoa (LAWA). (n.d.-f). Manawatū at Hopelands. https://www.lawa.org.nz/explore-data/manawatu-whanganui-region/river-quality/manawatu/manawatu-at-hopelands

Land, M., Granéli, W., Grimvall, A., Hoffmann, C. C., Mitsch, W. J., Tonderski, K. S., & Verhoeven, J. T. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environmental Evidence, 5(1), 9. https://doi.org/10.1186/s13750-016-0060-0

Landcare Research New Zealand Ltd. (2020). *LCDB v5.0 - Land Cover Database version 5.0, Mainland, New Zealand*. Landcare Research New Zealand Ltd. Accessed July 18 2023. https://lris.scinfo.org.nz/layer/104400-lcdb-v50-land-cover-database-version-50-mainland-new-zealand/

Ledgard, S. F., Penno, J. W., & Sprosen, M. S. (1999). Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. The Journal of Agricultural Science, 132(2), 215-225. https://doi.org/10.1017/S002185969800625X

Leifeld, J., Wüst-Galley, C., & Page, S. (2019). Intact and managed peatland soils as a source and sink of GHGs from 1850 to 2100. Nature Climate Change, 9(12), 945-947. https://doi.org/10.1038/s41558-019-0615-5

Lemke, A. M., Kirkham, K. G., Wallace, M. P., VanZomeren, C. M., Berkowitz, J. F., & Kovacic, D. A. (2022). Nitrogen and phosphorus removal using tile-treatment wetlands: A 12-year study from the midwestern United States. Journal of Environmental Quality, 51(5), 797-810. https://doi.org/10.1002/jeq2.20316

Maechler, M., Rousseeuw, P., Struyf, A., Hubert, M., & Hornik, K. (2023). cluster: Cluster Analysis Basics and Extensions. R package version 2.1.4. https://CRAN.R-project.org/package=cluster

Manawatu Estuary Trust. (2025, March). Papangaio Te Wharangi Manawatū Estuary Management Plan. https://www.metrust.org.nz/_files/ugd/ef14f9_acbf938c3db74cce8ffbfacf309b07a3.pdf

Manderson, A. (2018). Mapping the extent of artificial drainage in New Zealand. Report prepared for Lincoln Agritech. https://www.landcareresearch.co.nz/assets/Discover-Our-Research/Projects/IDA/Manderson_2018_mapping_extent_artificial_drainage_NZ.pdf

McArthur, K. (2009). Section 42A Report of Mrs Kathryn Jane McArthur on behalf of Horizons Regional Council. https://www.horizons.govt.nz/HRC/media/Media/One%20Plan%20Documents/Mrs-Kathryn-Jane-McArthur.pdf?ext=.pdf

McArthur, K. J., Roygard, J., & Clark, M. (2010). Understanding variations in the limiting nitrogen and phosphorus status of rivers in the Manawatu-Wanganui Region, New Zealand. Journal of Hydrology (New Zealand), 49(1), 15-33. https://www.jstor.org/stable/43944984

McArthur, K., & Patterson, M. (2007). Nitrogen and Phosphorus Loads to Rivers in the Manawatu-Wanganui Region: An Analysis of Low Flow State: Technical Report to Support Policy Development. https://www.horizons.govt.nz/HRC/media/Media/One%20Plan%20Documents/nutrient-loading-technical-report3-final-draft_Jemmas-edits.pdf?ext=.pdf

McDowell, R. W., & Haygarth, P. M. (2024). Reducing phosphorus losses from agricultural land to surface water. Current Opinion in Biotechnology, 89, 103181. https://doi.org/10.1016/j.copbio.2024.103181

McDowell, R. W., & Monaghan, R. M. (2015). Extreme phosphorus losses in drainage from grazed dairy pastures on marginal land. Journal of environmental quality, 44(2), 545-551. https://doi.org/10.2134/jeq2014.04.0160

McDowell, R. W., & Wilcock, R. J. (2008). Water quality and the effects of different pastoral animals. New Zealand Veterinary Journal, 56(6), 289-296. https://doi.org/10.1080/00480169.2008.36849

McMahon, P. B., & Chapelle, F. H. (2008). Redox processes and water quality of selected principal aquifer systems. Groundwater, 46(2), 259-271. https://doi.org/10.1111/j.1745-6584.2007.00385.x

McNeill-Adams, J. (1968). *Effects of the Lower Manawatu River Flood Control Scheme on the Farming in the Lower Manawatu*. [A Thesis Presented in Partial Fulfilment of the Requirements for the Degree of Master of Arts in Geography at Massey University]. Massey Research Online. https://mro.massey.ac.nz/bitstreams/dee101b8-0dd0-489a-8d1e-0a0fed868501/download

McWethy, D. B., Wilmshurst, J. M., McGlone, M. S., Fromont, M., Burrows, C., Wood, J. R., & Whitlock, C. (2010). Rapid landscape transformation in South Island, New Zealand, following initial Polynesian settlement. Proceedings of the National Academy of Sciences, 107(51), 21343-21348. https://doi.org/10.1073/pnas.1011801107

Ministry for Primary Industries. (2017). Analysis of Drivers and Barriers to Land Use Change. https://www.mpi.govt.nz/dmsdocument/23056-ANALYSIS-OF-DRIVERS-AND-BARRIERS-TO-LAND-USE-CHANGE/

Ministry for the Environment & Stats NZ (2023, April). New Zealand's Environmental Reporting Series: Our freshwater 2023. https://environment.govt.nz/assets/publications/our-freshwater-2023.pdf

Ministry for the Environment & Stats NZ (2025). New Zealand's Environmental Reporting Series: Our environment 2025. https://environment.govt.nz/assets/publications/our-environment-2025.pdf

Monaghan, R. M., Paton, R. J., Smith, L. C., & Binet, C. (2000). Nutrient losses in drainage and surface runoff from a cattle-grazed pasture in Southland. In Proceedings of the Conference-New Zealand Grassland Association (pp. 99-104). https://www.researchgate.net/profile/Leo-Smith-6/publication/266415224_Nutrient_losses_in_drainage_and_surface_runoff_from_a_cattle-grazed_pasture_in_Southland/links/54b41c2c0cf28ebe92e453d6/Nutrient-losses-in-drainage-and-surface-runoff-from-a-cattle-grazed-pasture-in-Southland.pdf

Monaghan, R. M., Smith, L. C., & Muirhead, R. W. (2016). Pathways of contaminant transfers to water from an artificially-drained soil under intensive grazing by dairy cows. Agriculture, Ecosystems & Environment, 220, 76-88. https://doi.org/10.1016/j.agee.2015.12.024

Morgan, J., & Burns, K. (2009, December 2). Manawatu River 'among worst in the West'. The Dominion Post. https://www.stuff.co.nz/environment/3097651/Manawatu-River-among-worst-in-the-West

Mulder, L., Van de Graaf, A. A., Robertson, L. A., & Kuenen, J. G. (1995). Anaerobic ammonium oxidation discovered in a denitrifying fluidized bed reactor. FEMS microbiology ecology, 16(3), 177-183. https://doi.org/10.1111/j.1574-6941.1995.tb00281.x

National Institute of Water and Atmospheric Research (NIWA). (2010). *River Environment Classification New Zealand (2010)*. https://data.mfe.govt.nz/layer/51845-river-environment-classification-new-zealand-2010/

National Institute of Water and Atmospheric Research. (2017, June 6). How clean are our rivers? https://niwa.co.nz/water-atmosphere/water-atmosphere-1-july-2010/how-clean-are-our-rivers

Ogle, D. H., Doll, J. C., Wheeler, A. P., & Dinno, A. (2023). FSA Methods. R package version 0.9.5. https://CRAN.R-project.org/package=FSA

Pan, H., Qin, Y., Wang, Y., Liu, S., Yu, B., Song, Y., ... & Zhu, G. (2020). Dissimilatory nitrate/nitrite reduction to ammonium (DNRA) pathway dominates nitrate reduction processes in rhizosphere and non-rhizosphere of four fertilized farmland soil. Environmental Research, 186, 109612. https://doi.org/10.1016/j.envres.2020.109612

Pandey, A., Suter, H., He, J. Z., Hu, H. W., & Chen, D. (2019). Dissimilatory nitrate reduction to ammonium dominates nitrate reduction in long-term low nitrogen fertilized rice paddies. Soil Biology and Biochemistry, 131, 149-156. https://doi.org/10.1016/j.soilbio.2019.01.007

Parliamentary Commissioner for the Environment. (2004). Growing for good: Intensive farming, sustainability and New Zealand's environment. https://pce.parliament.nz/media/ml4pcrou/growing-for-good-full.pdf

Peden, R. (2009). Fire and agriculture. Te Ara - the Encyclopedia of New Zealand. https://teara.govt.nz/en/fire-and-agriculture

Pedersen, T. L. (2024). patchwork: The Composer of Plots. R package version 1.2.0. https://CRAN.R-project.org/package=patchwork

Petry, J., Soulsby, C., Malcolm, I. A., & Youngson, A. F. (2002). Hydrological controls on nutrient concentrations and fluxes in agricultural catchments. Science of the Total Environment, 294(1-3), 95-110. https://doi.org/10.1016/S0048-9697(02)00058-X

Pham, T., Yli-Halla, M., Marttila, H., Lötjönen, T., Liimatainen, M., Kekkonen, J., Läpikivi, M., Klöve, B., & Joki-Tokola, E. (2023). Leaching of nitrogen, phosphorus and other solutes from a controlled drainage cultivated peatland in Ruukki, Finland. Science of the Total Environment, 904, 166769. https://doi.org/10.1016/j.scitotenv.2023.166769

Quinn, J. M., & Stroud, M. J. (2002). Water quality and sediment and nutrient export from New Zealand hill-land catchments of contrasting land use. New Zealand journal of marine and freshwater research, 36(2), 409-429. https://doi.org/10.1080/00288330.2002.9517097

Ramsar Sites Information Service. (2023, April 13). Manawatū River mouth and estuary. https://rsis.ramsar.org/ris/1491

Rivett, M. O., Buss, S. R., Morgan, P., Smith, J. W., & Bemment, C. D. (2008). Nitrate attenuation in groundwater: a review of biogeochemical controlling processes. *Water research*, *42*(16), 4215-4232. https://doi.org/10.1016/j.watres.2008.07.020

Robertson, G. P., & Groffman, P. M. (2024). Nitrogen transformations. Soil microbiology, ecology and biochemistry, 407-438. https://doi.org/10.1016/B978-0-12-822941-5.00014-4

Rosen, M. R., & White, P. A. (2001). *Groundwaters of New Zealand* (pp. xiii+-498).

Roygard, J. K. F., McArthur, K. J., & Clark, M. E. (2012). Diffuse contributions dominate over point sources of soluble nutrients in two sub-catchments of the Manawatu River, New Zealand. New Zealand Journal of Marine and Freshwater Research, 46(2), 219-241. https://doi.org/10.1080/00288330.2011.632425

Sapek, A., Pietrzak, S., Juszkowska, D., & Urbaniak, M. (2021). Phosphate and ammonia concentrations in waters from a meadow complex located on peat soils in various range of groundwater level. Journal of Water and Land Development, 51, 188-198. https://doi.org/10.24425/jwld.2021.139030

Sapek, A., Sapek, B., Chrzanowski, S., & Jaszczyński, J. (2007). Mobilization of substances in peat soils and their transfer within the groundwater and into surface water. Agron. Res, 5(2), 155-163. https://agronomy.emu.ee/vol052/p5206.pdf

Scalenghe, R., Edwards, A. C., Ajmone Marsan, F., & Barberis, E. (2002). The effect of reducing conditions on the solubility of phosphorus in a diverse range of European agricultural soils. European Journal of Soil Science, 53(3), 439-447. https://doi.org/10.1046/j.1365-2389.2002.00462.x

Schilling, K. E., & Helmers, M. (2008). Effects of subsurface drainage tiles on streamflow in Iowa agricultural watersheds: Exploratory hydrograph analysis. Hydrological Processes: An International Journal, 22(23), 4497-4506. https://doi.org/10.1002/hyp.7052

Schilling, K. E., Streeter, M. T., Vogelgesang, J., Jones, C. S., & Seeman, A. (2020). Subsurface nutrient export from a cropped field to an agricultural stream: Implications for targeting edge-of-field practices. Agricultural Water Management, 241, 106339. https://doi.org/10.1016/j.agwat.2020.106339

Silver, W. L., Herman, D. J., & Firestone, M. K. (2001). Dissimilatory nitrate reduction to ammonium in upland tropical forest soils. Ecology, 82(9), 2410-2416. https://doi.org/10.1890/0012-9658(2001)082[2410:DNRTAI]2.0.CO;2

Simmonds, B., McDowell, R. W., & Condron, L. M. (2016). The effect of soil moisture extremes on the pathways and forms of phosphorus lost in runoff from two contrasting soil types. Soil Research, 55(1), 19-27. https://doi.org/10.1071/SR15324

Smith, D. R., King, K. W., Johnson, L., Francesconi, W., Richards, P., Baker, D., & Sharpley, A. N. (2015). Surface runoff and tile drainage transport of phosphorus in the midwestern United States. Journal of environmental quality, 44(2), 495-502. https://doi.org/10.2134/jeq2014.04.0176

Smith, G., Singh, R., & Matthews, A. (2017). Assessing farm-scale nutrient flow pathways and nitrate attenuation in rangitikei sand country. Science and Policy: Nutrient

Management Challenges for the Next Generation, Occasional Report, (30), 12. https://flrc.massey.ac.nz/workshops/17/Manuscripts/Paper_Smith_2017.pdf

Snelder, T. H., Fraser, C., Larned, S. T., Monaghan, R., De Malmanche, S., & Whitehead, A. L. (2021). Attribution of river water-quality trends to agricultural land use and climate variability in New Zealand. Marine and Freshwater Research, 73(1), 1-19. https://doi.org/10.1071/MF21086

Sophocleous, M. (2002). Interactions between groundwater and surface water: the state of the science. Hydrogeology journal, 10(1), 52-67. https://doi.org/10.1007/s10040-001-0170-8

Stamm, C. H., Flühler, H., Gächter, R., Leuenberger, J., & Wunderli, H. (1998). Preferential transport of phosphorus in drained grassland soils, 27(3), 515-522. https://doi.org/10.2134/jeq1998.00472425002700030006x

Stats NZ. (2021a, April 15). Agricultural and horticultural land use. https://www.stats.govt.nz/indicators/agricultural-and-horticultural-land-use/

Stats NZ. (2021b, April 15). Exotic land cover. https://www.stats.govt.nz/indicators/exotic-land-cover/

Stats NZ. (2021c, April 15). Fertilisers – nitrogen and phosphorus. https://www.stats.govt.nz/indicators/fertilisers-nitrogen-and-phosphorus/

Stats NZ. (2022a, April 14). River water quality: nitrogen. https://www.stats.govt.nz/indicators/river-water-quality-nitrogen/

Stats NZ. (2022b, April 14). River water quality: phosphorus. https://www.stats.govt.nz/indicators/river-water-quality-phosphorus/

Stats NZ. (2024, December 12). Livestock numbers: Data to 2023. https://www.stats.govt.nz/indicators/livestock-numbers-data-to-2023/

Stats NZ. (2025, June 24). Groundwater quality: Data to 2024. https://www.stats.govt.nz/indicators/groundwater-quality-data-to-2024/

Tähtikarhu, M., Räsänen, T. A., Hyväluoma, J., Piayda, A., & Myllys, M. (2025). Analysing hydrological impacts of controlled drainage, peat thickness and groundwater fluxes in cultivated peat soils. Acta Agriculturae Scandinavica, Section B—Soil & Plant Science, 75(1), 2454388. https://doi.org/10.1080/09064710.2025.2454388

Tanner, C. C., McKergow, L. A., Goeller, B. C., Woodward, K. B., Sukias, J. P., Craggs, R. J., & Matheson, F. E. (2020). The spectrum of edge-of-field to waterway mitigation options for nutrient management in New Zealand's farmed landscapes. Nutrient management in farmed landscapes. Occasional Report, (33). https://www.researchgate.net/publication/372245271_THE_SPECTRUM_OF_EDGE-OF-FIELD_TO_WATERWAY_MITIGATION_OPTIONS_FOR_NUTRIENT_MANAGEMENT_IN_NEW_ZEALAND'S_FARMED_LANDSCAPES

Tanner, C., Sukias, J. & Woodward, B. (2021). Technical guidelines for constructed wetland treatment of pastoral farm run-off. NIWA Client Report No: 2020208.1HN for DairyNZ (May 2021). NIWA Project: DNZ18203. https://niwa.co.nz/sites/default/files/CW%20Technical%20Guidelines-Final.1%2020.5.21%2Bnote.pdf

Tannock, J. L. McC. (1968). *The Growth of Shannon: The Impact of Economic Change on a Country Town*. [A Thesis Presented in Partial Fulfilment of the Requirements for the Degree of Master of Arts in Geography at Massey University]. Massey Research Online. https://mro.massey.ac.nz/items/98b3b0c8-5a13-48c5-8526-bb1a8d355a3e

Tiemeyer, B., Frings, J., Kahle, P., Köhne, S., & Lennartz, B. (2007). A comprehensive study of nutrient losses, soil properties and groundwater concentrations in a degraded peatland used as an intensive meadow–implications for re-wetting. Journal of Hydrology, 345(1-2), 80-101. https://doi.org/10.1016/j.jhydrol.2007.08.002

Toet, S., Van Logtestijn, R. S., Kampf, R., Schreijer, M., & Verhoeven, J. T. (2005). The effect of hydraulic retention time on the removal of pollutants from sewage treatment plant effluent in a surface-flow wetland system. Wetlands, 25(2), 375-391. https://doi.org/10.1672/13

Vale, S. S., Fuller, I. C., Procter, J. N., Basher, L. R., & Smith, I. E. (2016). Characterization and quantification of suspended sediment sources to the Manawatu River, New Zealand. *Science of the Total Environment*, *543*, 171-186. https://doi.org/10.1016/j.scitotenv.2015.11.003

Van Beek, C. L., Droogers, P., Van Hardeveld, H. A., Van den Eertwegh, G. A. P. H., Velthof, G. L., & Oenema, O. (2007). Leaching of solutes from an intensively managed peat soil to surface water. Water, air, and soil pollution, 182(1), 291-301. https://doi.org/10.1007/s11270-007-9339-7

Van Beek, C. L., Van den Eertwegh, G. A. P. H., Van Schaik, F. H., Velthof, G. L., & Oenema, O. (2004). The contribution of dairy farming on peat soil to N and P loading of surface water. Nutrient Cycling in Agroecosystems, 70(1), 85-95. https://doi.org/10.1023/B:FRES.0000045984.93498.c3

Van Esbroeck, C. J., Macrae, M. L., Brunke, R. I., & McKague, K. (2016). Annual and seasonal phosphorus export in surface runoff and tile drainage from agricultural fields with cold temperate climates. Journal of Great Lakes Research, 42(6), 1271-1280. https://doi.org/10.1016/j.jglr.2015.12.014

Vassiljev, A., & Blinova, I. (2012). The influence of drained peat soils on diffuse nitrogen pollution of surface water. Hydrology Research, 43(4), 352-358. https://doi.org/10.2166/nh.2012.117

Vermaat, J. E., & Hellmann, F. (2010). Covariance in water-and nutrient budgets of Dutch peat polders: what governs nutrient retention?. Biogeochemistry, 99(1), 109-126. https://doi.org/10.1007/s10533-009-9395-8

Vymazal, J. (2007). Removal of nutrients in various types of constructed wetlands. Science of the total environment, 380(1-3), 48-65. https://doi.org/10.1016/j.scitotenv.2006.09.014

Wei, T., & Simko, V. (2021). R package 'corrplot': Visualization of a Correlation Matrix. Version 0.92. https://github.com/taiyun/corrplot

Wheeler, W. (2011). *Moutoa Drainage Scheme Audit*. (Report No. 2011/EXT/1138). Horizons Regional Council.

Wickham, H. (2016). ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York. https://ggplot2.tidyverse.org

Wickham, H., & Bryan, J. (2023). readxl: Read Excel Files. R package version 1.4.3. https://CRAN.R-project.org/package=readxl

Wickham, H., & Seidel, D. (2022). scales: Scale Functions for Visualization. R package version 1.2.1. https://CRAN.R-project.org/package=scales

Wickham, H., François, R., Henry, L., Müller, K., & Vaughan, D. (2023a). dplyr: A Grammar of Data Manipulation. R package version 1.1.2. https://CRAN.R-project.org/package=dplyr

Wickham, H., Vaughan, D., & Girlich, M. (2023b). tidyr: Tidy Messy Data. R package version 1.3.0. https://CRAN.R-project.org/package=tidyr

Williams, M. R., King, K. W., & Fausey, N. R. (2015). Contribution of tile drains to basin discharge and nitrogen export in a headwater agricultural watershed. Agricultural water management, 158, 42-50. https://doi.org/10.1016/j.agwat.2015.04.009

Winter, T. C. (1999). Relation of streams, lakes, and wetlands to groundwater flow systems. Hydrogeology Journal, 7(1), 28-45. https://doi.org/10.1007/s100400050178

Winter, T. C., Harvey, J. W., Franke, O. L., & Alley, W. M. (1998). Ground water and surface water: A single resource (Vol. 1139). US Geological Survey Circular.

Wolters, T., Bach, T., Eisele, M., Eschenbach, W., Kunkel, R., McNamara, I., Well, R., & Wendland, F. (2022). The derivation of denitrification conditions in groundwater: Combined method approach and application for Germany. Ecological indicators, 144, 109564. https://doi.org/10.1016/j.ecolind.2022.109564

Xin, J., Liu, Y., Chen, F., Duan, Y., Wei, G., Zheng, X., & Li, M. (2019). The missing nitrogen pieces: A critical review on the distribution, transformation, and budget of nitrogen in the vadose zone-groundwater system. Water research, 165, 114977. https://doi.org/10.1016/j.watres.2019.114977

Younger, P. L. (2007). Groundwater in the Environment. Malden, MA: Blackwell Publishing.

Zak, D., Gelbrecht, J., Wagner, C., & Steinberg, C. E. W. (2008). Evaluation of phosphorus mobilization potential in rewetted fens by an improved sequential chemical extraction procedure. European journal of soil science, 59(6), 1191-1201. https://doi.org/10.1111/j.1365-2389.2008.01081.x

Appendices

Appendix Table A1: Shallow groundwater level variability summary statistics, as measured at piezometer sites across the Moutoa catchment (November 2023-October 2024).

Site ID	Soil Type	Mean level (m, bgl)	Min level (m, bgl)	Max level (m, bgl)	Range (m, bgl)	Standard deviation	Coefficient of variation
CD2	Peaty soils	0.44	0.24	0.71	0.47	0.13	28.51
CD3	Peaty soils	0.37	0.15	0.65	0.50	0.14	38.90
CD4	Peaty soils	0.35	0.15	0.63	0.48	0.14	40.86
CD5	Peaty soils	0.41	0.20	0.68	0.48	0.15	36.70
CD6	Peaty soils	0.40	0.19	0.65	0.46	0.16	39.36
CD7	Peaty soils	0.40	0.18	0.67	0.49	0.16	41.10
JJ1	Peaty soils	0.30	0.10	0.56	0.46	0.17	58.64
PCD-1-D	Peaty soils	0.30	0.11	0.56	0.45	0.12	40.62
PCD-1-S	Peaty soils	0.28	0.10	0.57	0.47	0.13	45.77
PR-1	Peaty soils	0.91	0.29	2.29	2.00	0.57	62.46
PR-2	Peaty soils	0.94	0.35	2.34	1.99	0.57	60.13
PS-1	Sandy soils	0.94	0.43	1.47	1.04	0.32	33.50
PS-2	Sandy soils	0.93	0.43	1.46	1.03	0.31	33.93
PCR-1	Silt loam	0.80	0.59	1.01	0.42	0.14	16.96

Site ID	Soil Type	Mean level (m, bgl)	Min level (m, bgl)	Max level (m, bgl)	Range (m, bgl)	Standard deviation	Coefficient of variation
PCR-2	Silt loam	0.81	0.63	1.04	0.41	0.12	15.38
PTS-1	Silt loam	1.70	0.85	2.45	1.60	0.67	39.26
PTS-2	Silt loam	1.78	0.85	2.56	1.71	0.65	36.60

Appendix Table A2: Dunn's test with Bonferroni correction results for groudwater field parameters, by associated soil type.

Parameter	Group 1	Group 2	Sample Size 1	Sample Size 2	p-value	Adj. p-value	Adj. p-value significance level
Water Level (m, bgl)	Sandy	Silt loam	20	40	0.235111	0.705333	Non-significant (ns)
Water Level (m, bgl)	Sandy	Peaty	20	39	0.000832	0.002496	**
Water Level (m, bgl)	Silt loam	Peaty	40	39	3.22E-08	9.65E-08	****
Conductivity (µS/cm)	Sandy	Silt loam	22	43	0.794171	1	ns
Conductivity (µS/cm)	Sandy	Peaty	22	44	8.39E-09	2.52E-08	****
Conductivity (µS/cm)	Silt loam	Peaty	43	44	2.15E-11	6.45E-11	****
pH	Sandy	Silt loam	22	43	0.001448	0.004344	**
pH	Sandy	Peaty	22	44	0.764045	1	ns
pH	Silt loam	Peaty	43	44	0.000419	0.001258	**
TDS (ppt)	Sandy	Silt loam	22	43	0.767603	1	ns

Parameter	Group 1	Group 2	Sample Size 1	Sample Size 2	p-value	Adj. p-value	Adj. p-value significance level
TDS (ppt)	Sandy	Peaty	22	44	7.04E-09	2.11E-08	****
TDS (ppt)	Silt loam	Peaty	43	44	2.24E-11	6.73E-11	****
DO (mg/L)	Sandy	Silt loam	22	43	8.47E-05	0.000254	***
DO (mg/L)	Sandy	Peaty	22	44	0.042552	0.127655	ns
DO (mg/L)	Silt loam	Peaty	43	44	0.019533	0.058599	ns
DO Saturation (%)	Sandy	Silt loam	22	43	7.87E-05	0.000236	***
DO Saturation (%)	Sandy	Peaty	22	44	0.038636	0.115908	ns
DO Saturation (%)	Silt loam	Peaty	43	44	0.020978	0.062934	ns

Appendix Table A3: Kruskal-Wallis test results comparing shallow groundwater piezometer field sampling parameters across associated soil types (sandy, silt loam, organic/peaty soil types).

Parameter	H statistic	Degrees of freedom (df)	p value	Eta squared (η^2)	Effect size magnitude
Water Level (m, bgl)	31.91	2	1.18E-07	0.282131	large
Temperature (°C)	0.12	2	0.944	~0	negligible
Conductivity (µS/cm)	55.91	2	7.22E-13	0.508625	large
pH	16.04	2	0.000329	0.132416	moderate
TDS (ppt)	56.06	2	6.69E-13	0.510047	large
DO (mg/L)	16.06	2	0.000326	0.132605	moderate

Parameter	H statistic	Degrees of freedom (df)	p value	Eta squared (η^2)	Effect size magnitude
DO Saturation (%)	16.14	2	0.000313	0.133394	moderate

Appendix Table A4: Descriptive statistics for groundwater sample field parameters, by site.

Parameter	Statistics	Site Name									
		PCD-1-D	PCD-1-S	PR-1	PR-2	PS-1	PS-2	PTS-1	PTS-2	PCR-1	PCR-2
Soil Type		Peaty	Peaty	Peaty	Peaty	Sandy	Sandy	Silt loam	Silt loam	Silt loam	Silt loam
Water Level (m, bgl)	Mean	0.3	0.28	0.91	0.94	0.94	0.93	1.7	1.78	0.81	0.81
	SD	0.12	0.13	0.57	0.57	0.32	0.31	0.67	0.65	0.14	0.12
	Median	0.29	0.28	0.72	0.75	0.92	0.9	1.82	1.83	0.83	0.83
	IQR	0.23-0.34	0.22-0.34	0.63-1.01	0.66-1.04	0.79-1.15	0.75-1.13	1.04-2.34	1.27-2.41	0.72-0.89	0.74-0.87
	Range	0.11-0.56	0.1-0.57	0.29-2.29	0.35-2.34	0.43-1.47	0.43-1.46	0.85-2.45	0.85-2.56	0.59-1.01	0.63-1.04
Temperature (°C)	Mean	15.63	15.64	15.62	15.29	15.68	15.67	15.25	15.04	15.81	16.44
	SD	0.74	0.8	0.67	1.47	0.68	1.26	0.42	1.08	0.57	1.38
	Median	15.83	15.84	15.61	15.81	15.65	15.73	15.39	14.86	15.98	16.73
	IQR	15.25-16.20	15.18-16.16	15.03-16.06	14.23-16.56	15.11-16.06	14.61-16.64	14.91-15.50	14.21-16.08	15.24-16.23	15.20-17.20
	Range	14.36-16.32	14.22-16.76	14.84-16.86	13.12-17.34	14.76-16.86	14.06-17.56	14.59-15.93	13.62-16.55	15.03-16.64	14.34-18.54
	Mean	1266.43	1243.6	1331.57	1295.06	530.25	625.71	558.77	612.62	790.73	417.75

Parameter	Statistics	Site Name									
		PCD-1-D	PCD-1-S	PR-1	PR-2	PS-1	PS-2	PTS-1	PTS-2	PCR-1	PCR-2
Soil Type		**Peaty**	**Peaty**	**Peaty**	**Peaty**	**Sandy**	**Sandy**	**Silt loam**	**Silt loam**	**Silt loam**	**Silt loam**
Conductivity (µS/cm)	SD	571.8	553.43	543.79	597.71	233.94	251.48	247.46	268.82	357.15	182.87
	Median	1076.4	1073.7	1128.2	1124.7	465.5	531.4	486.4	534.5	686	393.5
	IQR	926.8-1265.7	900.5-1226.3	941.5-1664.1	922.0-1279.7	388.2-521.0	459.1-726.9	404.4-555.9	435.7-603.7	588.8-742.0	272.9-593.5
	Range	795.33-2591.2	779.51-2525.7	797.41-2309.9	850.4-2765.0	323.67-1073.4	341.31-1157.4	350.08-1130.3	367.72-1199.0	504.7-1634.4	197.02-668.99
pH	Mean	7.4	7.37	7.73	6.67	7.55	7.3	7.45	7.17	7.12	6.45
	SD	0.24	0.23	0.28	0.98	0.26	0.28	0.22	0.26	0.2	0.51
	Median	7.39	7.35	7.68	7.39	7.56	7.28	7.42	7.22	7.12	6.58
	IQR	7.29-7.58	7.29-7.53	7.62-7.97	5.92-7.46	7.45-7.74	7.06-7.52	7.40-7.51	6.93-7.32	7.00-7.19	6.10-6.82
	Range	6.95-7.74	6.93-7.7	7.25-8.11	5.26-7.7	7.07-7.87	6.86-7.73	7.04-7.85	6.78-7.57	6.84-7.5	5.57-7.09
TDS (ppt)	Mean	0.83	0.81	0.83	0.86	0.35	0.41	0.37	0.4	0.53	0.28
	SD	0.36	0.36	0.31	0.38	0.15	0.16	0.15	0.17	0.22	0.12

Parameter	Statistics	Site Name									
		PCD-1-D	**PCD-1-S**	**PR-1**	**PR-2**	**PS-1**	**PS-2**	**PTS-1**	**PTS-2**	**PCR-1**	**PCR-2**
Soil Type		**Peaty**	**Peaty**	**Peaty**	**Peaty**	**Sandy**	**Sandy**	**Silt loam**	**Silt loam**	**Silt loam**	**Silt loam**
	Median	0.7	0.69	0.73	0.73	0.3	0.35	0.32	0.345	0.45	0.27
	IQR	0.610-0.780	0.565-0.780	0.655-0.825	0.640-0.830	0.270-0.340	0.310-0.470	0.280-0.360	0.293-0.395	0.410-0.480	0.180-0.385
	Range	0.57-1.68	0.55-1.64	0.57-1.49	0.55-1.79	0.23-0.7	0.25-0.75	0.24-0.73	0.26-0.78	0.36-1.06	0.14-0.43
DO (mg/L)	Mean	0.26	0.1	0.08	0.74	0.07	0.14	0.12	4.89	0.24	2.28
	SD	0.26	0.05	0.05	0.64	0.03	0.09	0.06	2.19	0.54	1.46
	Median	0.19	0.1	0.08	0.6	0.06	0.12	0.12	4.96	0.09	1.98
	IQR	0.13-0.28	0.06-0.14	0.06-0.11	0.24-1.16	0.04-0.09	0.08-0.16	0.08-0.16	3.27-6.75	0.06-0.10	1.54-3.04
	Range	0.07-1.01	0.04-0.19	0.01-0.16	0.06-1.89	0.02-0.13	0.05-0.36	0.05-0.23	1.88-7.73	0.02-1.87	0.22-5.16
DO Saturation (%)	Mean	2.6	1.03	0.85	7.41	0.65	1.38	1.25	48.4	2.42	23.11
	SD	2.65	0.54	0.47	6.36	0.35	0.9	0.59	21.65	5.63	14.27
	Median	1.88	1.02	0.86	6.26	0.62	1.2	1.18	49.83	0.89	21.15

Parameter	Statistics	Site Name									
		PCD-1-D	PCD-1-S	PR-1	PR-2	PS-1	PS-2	PTS-1	PTS-2	PCR-1	PCR-2
Soil Type		**Peaty**	**Peaty**	**Peaty**	**Peaty**	**Sandy**	**Sandy**	**Silt loam**	**Silt loam**	**Silt loam**	**Silt loam**
	IQR	1.27-2.78	0.60-1.35	0.57-1.10	2.31-11.49	0.36-0.90	0.86-1.58	0.82-1.56	31.85-66.22	0.60-0.97	15.63-30.74
	Range	0.68-10.15	0.31-1.89	0.14-1.64	0.65-19.1	0.19-1.34	0.48-3.66	0.52-2.38	19.12-78.11	0.16-19.37	2.44-50.24

Appendix Table A5: Groundwater site redox classification, based on McMahon and Chapelle (2008) framework.

Redox Variables		**Dissolved O_2**	**NO_3-N (as Nitrogen)**	**Mn^{2+}**	**Fe^{2+}**	**SO_4^{2-}**	**Redox Category**	**Dominant Redox Process**	**n samples Anoxic**	**n samples Mixed**
Unit		**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**				
Threshold Value		**0.5**	**0.5**	**0.05**	**0.1**	**0.5**				
PCD-1-D	Mean	0.260	0.020	0.841	0.018	0.221	Anoxic	Mn(IV)	10	1
	Median	0.190	<0.020	0.838	0.017	0.190				
PCD-1-S	Mean	0.105	0.023	0.687	0.017	0.334	Anoxic	Mn(IV)	11	0
	Median	0.100	<0.020	0.691	0.017	0.190				
PCR-1	Mean	0.235	0.027	0.297	0.151	0.260	Anoxic	Mn(IV)	10	1
	Median	0.090	<0.020	0.298	0.060	0.190				

Redox Variables		Dissolved O_2	NO_3-N (as Nitrogen)	Mn^{2+}	Fe^{2+}	SO_4^{2-}	Redox Category	Dominant Redox Process	n samples Anoxic	n samples Mixed
Unit		**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**				
Threshold Value		**0.5**	**0.5**	**0.05**	**0.1**	**0.5**				
PCR-2	Mean	2.285	0.104	0.442	1.751	5.576	Mixed(oxic-anoxic)	O2-Fe(III)/SO4	1	10
	Median	1.980	<0.020	0.385	1.310	6.470				
PR-1	Mean	0.085	0.021	0.063	0.207	0.418	Anoxic	CH4gen	11	0
	Median	0.080	<0.020	0.063	0.163	0.190				
PR-2	Mean	0.743	0.582	0.223	1.576	243.558	Mixed(oxic-anoxic)	O2-Fe(III)/SO4	5	6
	Median	0.600	<0.020	0.079	1.040	0.190				
PS-1	Mean	0.065	<0.020	0.487	0.072	0.293	Anoxic	Mn(IV)	11	0
	Median	0.060	<0.020	0.487	0.064	0.190				
PS-2	Mean	0.138	<0.020	0.431	0.161	15.997	Anoxic	Fe(III)/SO4	11	0
	Median	0.120	<0.020	0.402	0.105	7.160				
PTS-1	Mean	0.125	0.020	0.395	0.043	0.190	Anoxic	Mn(IV)	11	0
	Median	0.120	<0.020	0.396	0.028	0.190				
PTS-2	Mean	4.893	0.026	0.366	0.031	0.237	Mixed(oxic-anoxic)	O2-Mn(IV)	10	0
	Median	4.955	<0.020	0.360	0.028	0.190				

Appendix Table A6: Total dissolved solids classification and descriptive statistics across groundwater sampling sites (Younger, 2007).

Site ID	Soil Type	Number of samples	TDS mean (mg/L)	TDS median (mg/L)	TDS min (mg/L)	TDS max (mg/L)	Primary Category
PS-1	Sandy	11	350.9091	300	230	700	Fresh Water
PS-2	Sandy	11	411.8182	350	250	750	Fresh Water
PCR-1	Silt loam	11	525.4545	450	360	1060	Fresh Water
PCR-2	Silt loam	11	275.4545	270	140	430	Fresh Water
PTS-1	Silt loam	11	369.0909	320	240	730	Fresh Water
PTS-2	Silt loam	10	403	345	260	780	Fresh Water
PCD-1-D	Peaty	11	827.2727	700	570	1680	Fresh Water
PCD-1-S	Peaty	11	805.4545	690	550	1640	Fresh Water
PR-1	Peaty	11	831.8182	730	570	1490	Fresh Water
PR-2	Peaty	11	856.3636	730	550	1790	Fresh Water

Appendix Table A7: Major ion concentrations (median and IQR) in shallow groundwater samples, by site.

Site ID	Soil Type	Number of samples	Ca^{2+} (mg/L)	Mg^{2+} (mg/L)	K^{+} (mg/L)	Cl^{-} (mg/L)	SO_4^{2-} (mg/L)	Alkalinity (mg/L as CaCO3)
PS-1	Sandy	11	53.4 (52.4-54.4)	5.8 (5.7-6.0)	6.0 (5.8-6.1)	29.1 (28.7-29.5)	<0.19 (0.19-0.21)	180 (180-190)
PS-2	Sandy	11	59.1 (55.2-61.8)	8.3 (7.8-8.9)	6.3 (6.2-6.8)	35.7 (34.3-38.5)	7.16 (2.09-16.50)	200 (195-210)
PCR-1	Silt loam	11	52.1 (50.0-52.8)	15.7 (15.6-16.3)	12.2 (11.9-12.6)	29.3 (29.1-30.6)	<0.19	310 (305-320)
PCR-2	Silt loam	11	21.1 (19.8-28.0)	8.8 (8.6-11.3)	6.3 (6.0-9.7)	17.7 (16.0-19.6)	6.47 (2.10-7.91)	110 (105-175)
PTS-1	Silt loam	11	40.9 (40.3-41.6)	15.7 (15.6-16.1)	12.0 (11.9-12.2)	8.6 (8.2-9.0)	<0.19	230 (230-240)
PTS-2	Silt loam	10	48.0 (47.7-49.4)	17.1 (16.7-17.3)	11.9 (11.8-12.3)	9.4 (8.8-9.7)	<0.19	250 (242-250)
PCD-1-D	Peaty	11	75.0 (74.2-76.2)	48.4 (47.4-49.5)	27.7 (27.1-28.6)	56.6 (54.9-57.3)	<0.19 (0.19-0.24)	500 (490-500)
PCD-1-S	Peaty	11	73.8 (72.8-75.1)	44.6 (44.0-45.6)	25.2 (24.6-25.5)	53.9 (52.9-55.0)	<0.19 (0.19-0.36)	480 (470-490)
PR-1	Peaty	11	14.0 (13.8-14.2)	28.7 (28.6-29.9)	26.1 (25.7-26.8)	36.0 (35.0-36.5)	<0.19 (0.19-0.53)	530 (530-535)

Site ID	Soil Type	Number of samples	Ca^{2+} (mg/L)	Mg^{2+} (mg/L)	K^{+} (mg/L)	Cl^{-} (mg/L)	SO_4^{2-} (mg/L)	Alkalinity (mg/L as CaCO3)
PR-2	Peaty	11	12.8 (12.4-109.5)	21.6 (21.1-57.8)	21.5 (16.6-22.1)	36.9 (34.8-45.5)	<0.19 (0.19-559.00)	510 (38-520)

Appendix Table A8: Cluster analysis for groundwater major ions by site.

Site ID	Soil Type	Cluster	Ca^{2+} (%)	Mg^{2+} (%)	K^{+} (%)	HCO_3^{-} + CO_3^{2-} (%)	SO_4^{2-} (%)	Cl^{-} (%)
PCD-1-D	Peaty	1	44.35	47.20	8.44	92.56	0.02	7.42
PCD-1-S	Peaty	1	45.97	46.00	8.03	92.64	0.03	7.32
PCR-1	Silt loam	1	61.41	31.13	7.46	93.61	0.04	6.35
PCR-2	Silt loam	1	54.26	36.97	8.77	88.94	2.22	8.84
PR-1	Peaty	2	18.59	63.49	17.92	95.41	0.04	4.55
PR-2	Peaty	3	35.01	52.57	12.42	57.74	35.80	6.46
PS-1	Sandy	4	80.75	14.55	4.70	89.90	0.07	10.02
PS-2	Sandy	4	75.97	19.70	4.33	85.77	3.22	11.01
PTS-1	Silt loam	5	55.92	35.61	8.47	97.51	0.04	2.45
PTS-2	Silt loam	5	58.31	34.31	7.37	97.36	0.05	2.59

Appendix Table A9: Median and IQR groundwater nutrient concentrations by site and soil type association.

Site ID	Soil Type	Number of samples	DOC (g/m^3)	SIN (g/m^3 N)	Ammonia-N (g/m^3)	Nitrite-N (g/m^3)	Nitrate-N (g/m^3)	DRP (g/m^3 P)
PS-1	Sandy	11	5.0 (3.8-5.4)	1.51 (1.44-1.55)	1.47 (1.40-1.51)	<0.020	<0.020	0.18 (0.16-0.20)
PS-2	Sandy	11	5.3 (4.8-6.8)	1.58 (1.52-1.66)	1.54 (1.48-1.62)	<0.020	<0.020	0.14 (0.08-0.20)
PCR-1	Silt loam	11	6.2 (5.3-8.1)	9.49 (9.14-9.58)	9.45 (9.08-9.54)	<0.020	<0.020	2.61 (2.50-2.75)
PCR-2	Silt loam	11	6.8 (6.2-7.6)	1.68 (0.90-3.62)	1.17 (0.86-3.52)	<0.020	<0.020 (0.020-0.075)	0.09 (0.02-0.75)
PTS-1	Silt loam	11	3.6 (2.8-4.4)	3.24 (3.15-3.34)	3.20 (3.11-3.30)	<0.020	<0.020	1.24 (1.20-1.30)
PTS-2	Silt loam	10	5.8 (4.0-7.3)	2.68 (2.59-2.74)	2.64 (2.55-2.70)	<0.020	<0.020	1.42 (1.35-1.48)
PCD-1-D	Peaty	11	11.6 (9.2-13.2)	18.30 (17.95-18.60)	18.30 (17.85-18.60)	<0.020	<0.020	3.53 (3.42-3.58)

Site ID	Soil Type	Number of samples	DOC (g/m^3)	SIN (g/m^3 N)	Ammonia-N (g/m^3)	Nitrite-N (g/m^3)	Nitrate-N (g/m^3)	DRP (g/m^3 P)
PCD-1-S	Peaty	11	11.1 (9.0-14.2)	17.20 (17.00-17.20)	17.10 (17.00-17.20)	<0.020	<0.020	3.15 (3.04-3.21)
PR-1	Peaty	11	17.0 (12.8-20.1)	9.41 (9.32-9.69)	9.37 (9.28-9.64)	<0.020	<0.020	4.38 (4.11-4.46)
PR-2	Peaty	11	21.1 (15.6-41.5)	6.83 (2.26-8.07)	6.79 (1.23-8.04)	<0.020	<0.020	4.56 (0.06-5.16)

Appendix Table A10: Dunn's test showing pairwise comparisons for which groundwater samples differ from each other in nutrient composition, when grouped by soil type.

Nutrient type	Group 1	Group 2	Sample Size 1	Sample Size 2	p-value	Adj. p-value	Adj. p-value significance level
DOC	Sandy	Silt loam	22	43	0.589915	1	ns
DOC	Sandy	Peaty	22	44	1.83E-10	5.48E-10	****
DOC	Silt loam	Peaty	43	44	1.21E-12	3.63E-12	****
SIN	Sandy	Silt loam	22	43	0.000537	0.001611	**
SIN	Sandy	Peaty	22	44	2.34E-13	7.03E-13	****
SIN	Silt loam	Peaty	43	44	2.72E-06	8.15E-06	****

Nutrient type	Group 1	Group 2	Sample Size 1	Sample Size 2	p-value	Adj. p-value	Adj. p-value significance level
Ammonia-N	Sandy	Silt loam	22	43	0.001106	0.003318	**
Ammonia-N	Sandy	Peaty	22	44	3.65E-12	1.09E-11	****
Ammonia-N	Silt loam	Peaty	43	44	7.62E-06	2.29E-05	****
DRP	Sandy	Silt loam	22	43	0.003523	0.010568	*
DRP	Sandy	Peaty	22	44	8.18E-13	2.45E-12	****
DRP	Silt loam	Peaty	43	44	2.61E-07	7.84E-07	****
Manganese	Sandy	Silt loam	22	43	0.001785	0.005356	**
Manganese	Sandy	Peaty	22	44	0.186235	0.558705	ns
Manganese	Silt loam	Peaty	43	44	0.027176	0.081528	ns

Appendix Table A11: Direction of depth effects on sample parameters for shallow versus deep piezometers, indicating which parameters are higher or lower with depth and the statistical significance of these differences.

Parameter	Mean Shallow	Mean Deep	Mean Difference	P value	Direction
Temperature (°C)	15.617	15.595	0.023	0.897	Higher in Shallow (not significant)
DO (mg/L)	1.618	0.169	1.450	0.190	Higher in Shallow (not significant)
pH	7.040	7.399	-0.359	0.197	Higher in Deep (not significant)
DOC (g/m^3)	11.879	8.896	2.983	0.281	Higher in Shallow (not significant)

Parameter	Mean Shallow	Mean Deep	Mean Difference	P value	Direction
Ammonia-N (g/m^3)	5.681	8.317	-2.636	0.114	Higher in Deep (not significant)
Nitrate-N (g/m^3)	0.151	0.022	0.129	0.106	Higher in Shallow (not significant)
DRP (g/m^3 P)	1.648	2.363	-0.715	0.158	Higher in Deep (not significant)
Iron (g/m^3)	0.689	0.116	0.574	0.787	Higher in Shallow (not significant)
Manganese (g/m^3)	0.441	0.406	0.035	0.579	Higher in Shallow (not significant)
Conductivity (µS/cm)	819.858	914.642	-94.784	0.267	Higher in Deep (not significant)
TDS (ppt)	0.538	0.593	-0.055	0.348	Higher in Deep (not significant)