ECOHYDROLOGICAL CHARACTERISATION OF OTAKAIRANGI WETLAND, NORTHLAND

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Abstract

Otakairangi wetland is a 2.6 km^2 remnant wetland located on the western edge of the Wairua River floodplain, around 15 km north-west from Whangarei. Otakairangi has experienced over 100 years of degradation with long term hydrological modifications aimed at enabling agricultural land use. This includes a large central drain that has effectively divided the wetland into two sections, removed the natural diffuse flows of the Otakairangi stream and lowered water tables. This central drain, coupled with border drainage and frequent fire events caused the peat within the wetland to become severely degraded and decreased biodiversity.

The ecohydrology of representative areas of the wetland was studied to assess the relationship between the anthropogenic disturbances and the wetland ecology and water level regime. The primary goal of this research was to characterise the wetland condition, which would aid in the development of future management goals and planning through measurement of wetland degradation and natural recovery.

The study focused on five key areas of the wetland, with several transect lines established to determine the extent of degradation and recovery. This includes two transects across the central drain and three extending from the wetland margins. Peat physical and chemical characteristics were measured in the field and laboratory, and then classified through statistical ordination techniques. Vegetation compositions and patterns were assessed along with foliage chemistry, and then compared to peat characteristics. Hydrological data were retrieved and analysed from eight automatic water level sites along a transect perpendicular to the central drain, while meteorological data was collected at a nearby Northland Regional Council rain gauge. Peat surface oscillation in the restiad bog and water chemistry were also examined.

Nutrients, isotopes ($\delta^{15}N$), heavy metals, and physical soil characteristics (such as bulk density and water pH) were highest in close proximity to the upper central drain and the northern border drain. A gradient was observed from the entrance

of the central drain into the central north-eastern section of the wetland, with high fertility substrate and swamp conditions transitioning to low fertility bog-like conditions. This was also shown by the vegetation composition, with larger swamp species such as flax (*Phormium tenax*) transitioning to sedges (*Machaerina teretifolia*) and ferns (*Gleichenia dicarpa*) and then to the restiad bog species *Empodisma robustum.* Low nutrient conditions were present elsewhere in the wetland, such as the southern section by the southern border drain and southern central drain. Both exhibited low fertility, the area around the southern central drain was dominated by large patches of *M. teretifolia* and small patches of *E. robustum* (fen-bog transitional), while the area by the southern drain was predominantly *G. dicarpa* and *Leptospermum scoparium* (fen). A natural spread of *E. robustum* is restoring the wetland, in terms of natural vegetation composition and peat substrate functionality, as indicated by analysis of peat cores from beneath sites in which it dominated.

The water levels in central areas of the wetland (100 - 290 m from the central drain) were relatively stable, with fluctuations around large rainfall events. These areas had rainfall as the primary water input, as they were independent from any surficial flows originating from either the central drainage channel or border drainage. However, several flood events impacted the marginal wetland water tables (February 2018, May-July 2018) with the drainage channel flooding and inundating between 50-100 m of the wetland area north-east of the central drain. In contrast, water table drawdown extended less than 20 m from the central drain, with the 20 m site experiencing mean water tables 1.5 m above that of the mean central drain water level.

The major risk to the wetland is continued flood inundation from the nutrient and sediment-rich waters from the surrounding catchment. Recommendations for management include improving the water quality from the headwaters to the wetland entrance through onsite farm management, engineering structures such as constructed wetlands or buffer zones to reduce flood inundation of the wetland. Internal restoration includes spreading the main peat forming species, *Empodisma robustum*, to areas in which it is currently absent, and a feasibility study for the re-introduction of the locally extinct *Sporadanthus ferrugineus* to the bog area.

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Midday in the southern section of Otakairangi wetland.

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Chapter 1. Introduction

1.1 Background

Wetlands are complex ecosystems which exist across a wide range of environmental conditions, but are under threat. In New Zealand, wetlands are defined as areas which are permanently or intermittently wet, with shallow water margins that provides habitat for adapted animal and plant species (Johnson & Gerbeaux, 2004). Approximately 90% of New Zealand wetland area has been lost in the last 150 years (Cromarty & Scott, 1996), as the historically recognised 'waste land' was often drained and burned to convert it for agricultural uses. However, this has slowed drastically in recent decades with international recognition of the benefits the ecosystem can have (Halabuk, 2006), including water storage, flood mitigation, and CO_2 storage.

A significant portion of research in New Zealand has been focused on wetland biodiversity, with the outcomes generally supporting sustainable, functioning wetlands and ecosystem services, such as the Arawai Kākāriki Project, and the Biodiversity Advice and Condition Fund (Clarkson *et al.*, 2013). Other researched topics have included the drainage of wetlands (for agricultural purposes), the addition of alternate water sources to wetlands, and carbon storage and emissions from peat wetlands.

1.2 Otakairangi Wetland

Otakairangi is a 2.6 km^2 remnant wetland located on the western edge of the former Wairua River floodplain, around 15 km north of Whangarei, Northland. While the remnant wetland is mainly a restiad peatland, there are areas inside and near the margins that are fen and swamp like. While the majority of the wetland receives rainfall as its primary water input, due to its low lying nature in a valley it periodically receives flows from the upper catchment and surrounding land, especially during inundation events such as floods (Campbell, 2017). This brings water, sediment and nutrients into the wetland that natural rain inputs cannot provide.

The primary threat to the wetland is the altered hydrologic regime following extensive drainage both around and within the wetland to create productive farmland and reduce flood risk. Other threats include burning events (evidence of historical fires), invasion of exotic plants that can outcompete native plants, pest animals such as pigs destroying wetland vegetation, and other indirect threats such as nutrient runoff entering the wetland through waterways (Clarkson *et al.*, 2015).

1.3 New Zealand Wetlands

New Zealand has a variety of wetland classes, broadly classed as swamps, fens and bogs (further described in Chapter 2). These wetlands are important habitat for a range of flora and fauna, with species present at Otakairangi including native galaxiids and eels, the Australasian Bittern (*Botaurus poiciloptilus*), the Spotless Crake (*Porzana tabuensis plumbea*), the North Island Fernbird (*Bowdleria punctata vealeae*), and the native orb weaving spider *Backobourkia brounii* (Clarkson *et al.*, 2015).

In 1976 New Zealand became a signatory to the Ramsar Convention on Wetlands of International Importance, with six New Zealand sites being recognised (Department of Conservation, 2014). These are: Whangamarino wetland, the Firth of Thames and Kopuatai Peat Dome in the Waikato region, the Manawatu River Mouth and Estuary in the Manawatu-Wanganui region, Farewell Spit in the Tasman region, and Awarua (Waituna Lagoon) in Southland. These six sites hold more than 20 wetland classes, with each having more than one wetland class (Maranhão & Sant'Ana, 2017).

Research studies conducted in New Zealand wetlands have shown that major threats to the condition and ecosystem processes in wetlands can come from different sources, as no two wetlands are the same. Opuatia wetland was threatened by exotic species invasion and stock access due to unfenced margins (Browne, 2005). In contrast, the major threat to Whangamarino wetland was ongoing flooding from the Whangamarino River, which increased nutrient levels in the ombrotrophic wetland and caused a shift in vegetation cover from restiad species to *Leptospermum scoparium* forest (Blyth *et al.*, 2013).

1.4 Aim and Objectives

Figure 1.1. Aerial drone photography of the southern section of Otakairangi wetland, facing north from the road.

This research was funded by the joint Department of Conservation and Fonterra Living Waters Project, with the intent of establishing the major threats to the wetland.

The primary aim of this research is to investigate the ecohydrological characteristics of Otakairangi wetland. This is in order to determine the magnitude of influence the drainage system has on modifying the natural hydrological processes, peat degradation state and vegetation composition, as well as identifying the effects of decades of fire and vegetation change. The findings will be interpreted in the context of the present state of the wetland to identify the main issues affecting the site, and what management options may be best suited to deal with the issues.

The main objectives are to:

- Measure the annual and seasonal hydrological regimes along a transect line in a central portion of Otakairangi wetland.
- Identify how the flow of the central drain influences the hydrological processes operating along the transect line, to determine the lateral effect of the drain.
- Characterise the vegetation composition and peat chemical and physical traits along several representative transect lines that span the wetland.
- Identify and attempt to explain the relationships between the hydrological processes and water inputs, vegetation composition, and the physical and chemical condition of the peat.
- Provide scientifically sound information on the current state of wetland condition and the external influencing factors, in order to formulate better management approaches for restoration and enhancement.

1.5 Thesis Outline

Chapter 2 reviews the literature on the current state of knowledge about wetland physical formation, hydrology, vegetation and chemistry to provide a background understanding for this study and the issues related to this wetland.

Chapter 3 will describe the location of Otakairangi Wetland, along with climate and geographical information for the area. The research sites in the wetland will be described, including the hydrological transect, vegetation plots, and major drainage systems.

Chapter 4 will describe the characteristics of peat along the transect lines, and discuss possible causes for the observed trends seen by comparing to other relevant studies. This includes current state of the peat in terms of physical degradation and natural recovery, as well as the concentrations of nutrients and metals.

Chapter 5 will present the patterns of vegetation across the wetland based on surveys along the transect lines. The vegetation composition will be interpreted relative to the peat and foliage chemistry at each location, while the changes to the dominant vegetation cover will be discussed.

Chapter 6 will analyse the hydrological patterns observed along the central transect line, based on data received from the pressure transducer network. Water level regimes are examined to identify the lateral extent of effects caused by the central drain, such as flood events and low flows. Peat surface oscillation in the wetland will also be described.

Chapter 7 is a discussion that will link the main findings of the research in the context of the outlined objectives. The composition and patterns of the surface vegetation will be compared to the chemical balance and inputs to the wetland, as well as the hydrological patterns, and zonation of wetland classes across the wetland will be discussed. The overall impact of the modified hydrological regime and surrounding land uses on the wetland will be outlined, followed by recommendations for future research and wetland management.

Chapter 2. Wetland Biogeochemistry

Wetlands provide a range of ecosystem services including controlling water levels during flood events, improving the quality of water that flows through them, and regulating global atmospheric carbon levels. They provide habitat for unique flora and fauna that are adapted to wet conditions, and are also valued for both cultural and recreational aspects. Wetland restoration is needed to reverse the decline in wetland area and condition to restore ecosystem services, and a variety of methods have been applied globally with differing success.

This chapter reviews the current state of wetland biogeochemical research, discussing wetland development, the distinction between wetland classes, and the current state of wetlands globally. The wetland water balance will be described by each individual component, while vegetation composition and functional characteristics will also be reviewed. Nutrient pathways in wetlands will be discussed along with the concept of ecohydrology. Aspects of anthropogenic and natural disturbances will be outlined, and the chapter will conclude with a discussion on wetland ecosystem management and restoration.

2.1 Introduction

Wetlands provide a number of natural services, including supporting services (nutrient cycling, primary production), provisioning services (production of food and materials), regulating services (carbon storage, flood attenuation, water filtering), and cultural services (traditional values, recreation and education) (Millenium Ecosystem Assessment, 2005). In New Zealand, wetlands are important to Māori for cultural and historical reasons, as well as being a traditional source of food and materials (Clarkson *et al*., 2013). Wetlands are also significant internationally for biodiversity values, creating habitat for a multitude of rare species. A significant number of birds and mammals depend on wetlands for breeding or feeding, while also locations of spawning grounds for fish and other groups such as insects and amphibians. The goal of wetland restoration is to return biodiversity and ecosystem functions to a functioning level comparable to that of an intact reference. It has been estimated that restored systems have 36% higher levels of provisioning, regulating and supporting ecosystem services than degraded wetlands, similar to those of natural wetlands (Meli *et al.*, 2014).

Major advances in restoration theory and practices have been evident in recent decades (Bonn *et al.*, 2016). However, theories on wetland development dynamics and restoration are largely based on Northern Hemisphere experiences, which does not always correlate to the development processes and dynamics of certain peatlands types found in the Southern Hemisphere, due to the different dominant vegetation types. For example, peat bogs in New Zealand and Australia are regionally classified as 'Restiad bogs', which are dominated by members of the Southern Hemisphere vascular plant family, Restionaceae (Judd *et al.*, 1999).

2.2 Ecohydrology

Ecohydrology is an interdisciplinary science that studies the interactions of the physical hydrology and the ecosystem (Zalewski, 2007). This includes both simple and complex interactions between the biota (flora, fauna and microbes), substrate, and water, as well as the storage and flows of nutrients and chemical elements. The term ecohydrology is relevant to wetland research as any study in a wetland context needs to account for the interactions between hydrology and the vegetation (Jørgensen, 2016). The hydrology, or the movement and storage of water, is the primary influencer on the ecology (such as vegetation and nutrient pathways). The ecology then influences the formation and maintenance of substrate, which can then in turn influence the movement of the water (Zalewski, 2007).

In wetlands, vegetation composition and abundance are directly controlled by the quantity and movement of water. Vegetation with functional adaptations to survive in different hydrological regimes will outcompete other species (which may also affect other processes, such as the rate of peat accumulation or degradation). This may be in relation to the height and fluctuation of the water table, or the frequency and duration of surface inundation. The fauna in wetlands, such as aquatic fish or macroinvertebrates, will also change in abundance based on the hydrology and vegetation (Sorrell & Gerbeaux, 2004). The hydrological regime provides the primary path for nutrients, while vegetation and detritus provide shelter, habitat and food for the fauna.

A key aspect of wetland research is the use of integrative science, with a basin or catchment scale being the most appropriate way to establish key degrading factors to an ecosystem. This knowledge allows identification of necessary technical solutions to increase the health of the ecosystem (Zalewski, 2007). Wetland ecohydrology of a given region can be conceptualized using three separate components: human disturbances, the hydrological dynamics and the response of the wetland natural ecosystems (Zhou *et al.*, 2016). The issue in ecohydrology is balancing the limited water resources between natural wetland ecosystems and human activities, and the scientific estimation of the proportion of natural wetland ecosystems to preserve in agricultural areas. To propose sustainable solutions, the major threats need to be identified and quantified (Zhou *et al.*, 2016).

Overseas studies of wetland ecosystems have revealed that it is impossible to understand their ecology without considering hydrology, as there are deep relationships between the organic components and the water (Bragg, 2002; Zalewski, 2007). Bragg (2002) described how many wetland studies conducted in Scotland only focused on the internal functioning of the wetland, treating them as areas isolated hydrologically from the rest of the catchment. He then promoted the use of wider catchment scale hydrology and functioning, such as catchment runoff generation and the consequent nutrient contribution entering the wetland and altering the ecology.

An example of how integrated science can influence management is the study Trepel & Kluge (2002) undertook to characterise the ecohydrology of a 150 ha degenerated valley peatland in Northern Germany, bisected by the Eider River. Degradation of the wetland was primarily caused by land use intensification, drainage and river regulation attempting to reduce flood impacts during the winter season. The upland catchment was primarily farming, with 60% land area dedicated to crops, 20% to pasture, and 10% to forest. The main finding of the study was that restoring natural flood flows and water regimes could aid in regeneration of the wetland over a long term period, but as there was irreversible damage to peat following decomposition, immediate changes would only restore surface flows and create a shallow lake system (Trepel & Kluge, 2002).

There have been a variety of New Zealand studies on wetland ecohydrological characteristics, with some being conducted in the Lower Waikato. Browne (2005) studied the ecohydrological characteristics of Opuatia wetland, analysing the vegetation composition, peat chemistry, wetland and river water levels, along with the meteorological conditions. Blyth (2011) studied the ecohydrological characteristics of Whangamarino wetland using similar methods, also including features such as vegetation chemistry and flood regimes. Clarkson *et al*., 2004b used the patterns of ecology, chemistry and hydrology to establish a New Zealand index defining wetland health. Opuatia was found to be in good health, although there was nutrient runoff from surrounding agricultural land, with high concentrations found in the wetland which promoted invasion from pest weeds, such as *S. cinerea* (grey willow), along with stock accessing the wetland through unfenced margins (Browne, 2005). Whangamarino wetland was found to be at risk due to continued flood inundation from the Whangamarino River, which brought nutrient and sediment rich waters deep into the wetland. This allowed the mineralised edge band to move further into the wetland, changing the vegetation composition with manuka invading an area that was previously restiad-dominated (Blyth, 2011; Blyth *et al*., 2013).

2.3 Wetland classes

Every wetland occurs with different influencing factors, and as such it can be difficult to classify them. This includes variations in the hydrological, biological and geomorphological functions which all influence how the wetland forms, or the degree of biological and mineral components (peat and mineral soils). A fundamental prerequisite for a wetland to develop is for an area of land to be almost permanently wet or saturated. The New Zealand Resource Management Act (1991) defines wetlands as "permanently or intermittently wet areas, shallow water or land/water margin that support a natural ecosystem of plants and animals adapted to living in wet conditions". This usually occurs in an area of restricted topography; a hollow or low lying land that has runoff from the surrounding area and/or impeded drainage (Johnson & Gerbeaux, 2004).

The palustrine hydro system wetland classification refers to wetlands that exists outside of the normal boundaries of lakes and river systems. Palustrine systems are classified into three main types in New Zealand (swamp, fen and bog), which differ primarily due to their different primary water sources, but are often developed from the previous types as the hydrologic or geomorphic conditions change (Johnson & Gerbeaux, 2004).

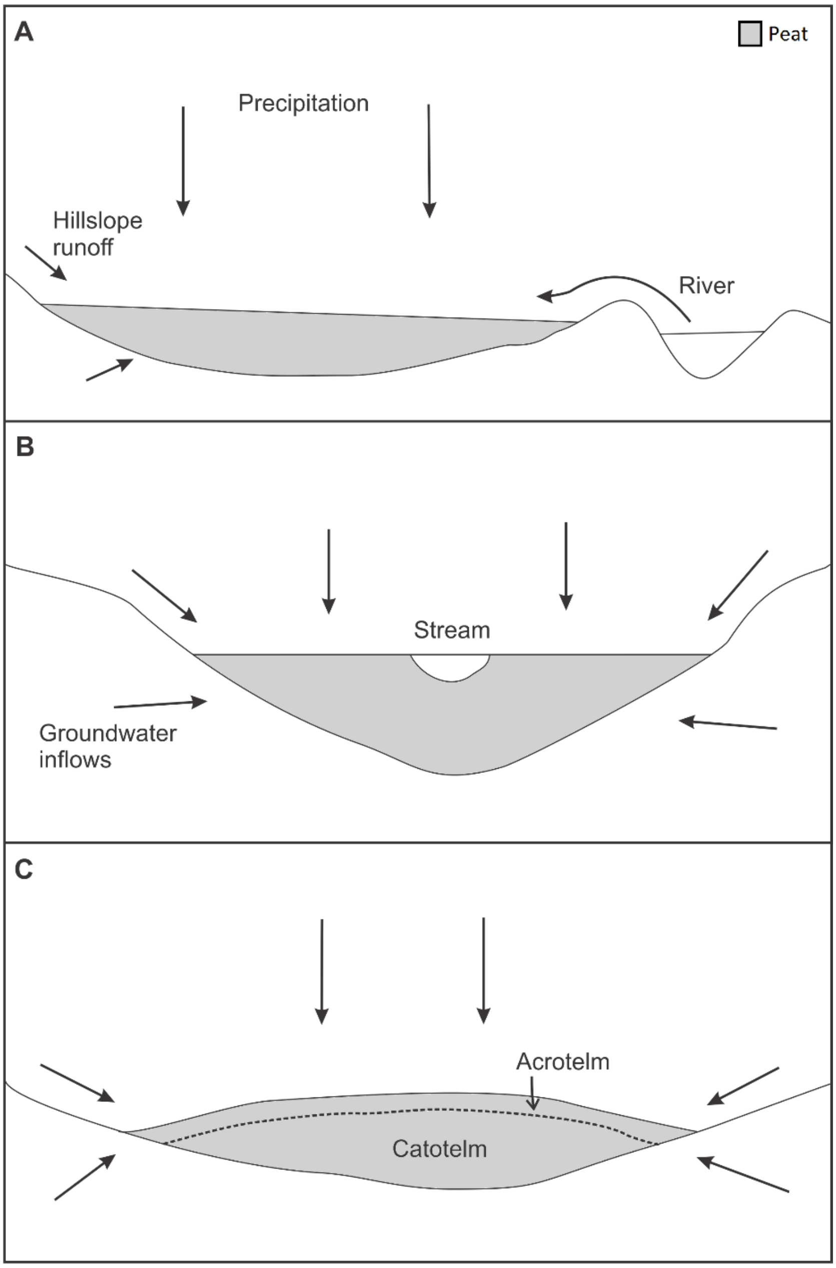

Figure 2.1. The main New Zealand wetland types and their water sources (arrows); (A) swamps, (B) fens, and (C) bogs. (Adapted from Campbell & Jackson, 2004).

Swamps (Figure 2.1 A) are usually partially connected to water systems, such as lakes and rivers, and as such they are periodically inundated with surface water. They have large inputs of mineralised sediment and nutrients, and tend to have pH levels closer to neutral or slightly acidic values, which can lead to increased vegetation productivity and diversity, as well as eutrophic conditions. They have low rates of peat accumulation as organic matter decomposes at a greater rate relative to other peat-forming wetlands (Peters & Clarkson, 2010).

Fens (Figure 2.1 B) are peat accumulating wetlands that can develop from swamps, usually by isolation from the inundating water system. Hydrological inputs are composed of rainfall, groundwater and lateral seepage from surrounding hill country, which result in lower nutrient levels (Johnson & Gerbeaux, 2004). As they receive water from sources other than just rainfall, they are termed minerotrophic.

Bogs (Figure 2.1 C) form when saturated anaerobic conditions cause the organic matter decomposition rate to decrease below the deposition rate, which results in peat accumulation. Due to this isolation from groundwater systems and external surface water, the wetland becomes primarily rain fed (ombrotrophic). This results in the wetland having lower plant production and specialised, wetland adapted plant species such as reeds and sedges that can withstand the lower nutrient inputs. Peat bogs are generally composed of layers of organic matter (peat) which is situated over bedrock or a mineral soil profile (Martini *et al.*, 2006). Peatlands have been estimated to cover 3% of the global terrestrial surface in at least 175 countries, predominantly in boreal or temperate regions (Leifeld & Menichetti, 2018).

2.4 Wetland hydrology

The hydrology of palustrine wetland types differs due to many influencing factors. Wetland hydrological processes are complex in nature due to the different water inputs (precipitation, groundwater and overland surface flows) and outputs (evaporation, surface and subsurface water discharge). Other influencers on the hydrology include the regional climate, microclimates, local topography, the geologic substructures, and the vegetation composition.

2.4.1 Water balance and flows in wetlands

The wetland water balance takes the general form of:

Change in storage = Inputs – Outputs

Where the equation is:

$\Delta S = (P + Q_{in} + G_{in}) - (E + Q_{out} + G_{out})$ (Equation 2.1)

ΔS= change in water storage, P= precipitation, Q_{in}= surface inflows, G_{in}= groundwater inflows, E= evaporation, Q_{out}= surface outflows, G_{out} = groundwater outflows. (Campbell & Jackson, 2004)

As the inputs to a wetland (rainfall, groundwater and surface inflows) are assumed to be balanced by the outflows (evaporation (including plant transpiration), groundwater and surface outflows) over the long term, the change in water storage in a wetland is assumed to be negligible over annual periods. If there is long-term change in water storage, it is likely there is some alteration to the hydrological regime such as physical changes to the wetland itself, which therefore raises or lowers the water table.

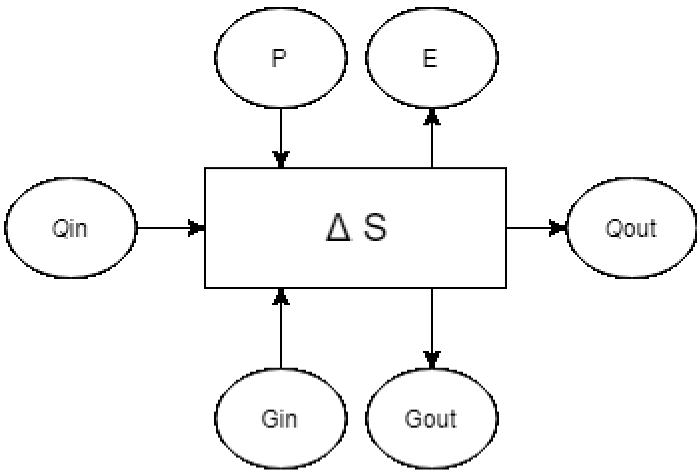

Figure 2.2. Wetland water balance using terms from Equation 2.1. (Adapted from Campbell & Jackson, 2004).

2.4.2 Precipitation and evaporation

As all wetlands are different, they have variable inputs from terrestrial water sources, but all have strong atmospheric water linkages. Bogs are primarily fed by rainfall which is usually low in nutrients (termed ombrotrophic), and therefore the regime of precipitation will dictate the growth of plants as well as the spatial extent of the wetland (Charman, 2002; Campbell & Jackson, 2004).

Many studies have concluded that evaporation is one of the main forms of water movement out of a wetland (depending on the class) (Mitsch & Gosselink, 2007). While wetlands naturally have high water content, plants often control evaporation rates by restricting water loss from the wetland surface. When plants become stressed from increasing solar radiation or decreasing water tables, photosynthesis may be reduced or stop, as stomata on leaves close to prevent gas exchange and water loss (Mitsch & Gosselink, 2007). Campbell & Williamson (1997) studied evaporation in Kopuatai wetland, New Zealand, and found that even when the peat was close to saturation the evaporation rates from *Empodisma robustum* dominated bog communities were far lower than what was expected, while Thompson *et al.*, (1999) found similar results with *Sporadanthus ferrugineus* in Kopuatai and Moanatuatua wetlands. This was primarily attributed to the strong physiological control by stomata, as well as canopy structural resistance, as dense growth allowed little diffusion of water vapour from the peat surface. These physiological factors prevented evaporation from the peat surface and reduced the evaporation rates in bogs, sustaining moisture in the peat and provided favourable (wet and anaerobic) conditions for peat formation, meaning that these plant species are important for reducing evaporation and increasing peat accumulation in peatlands.

2.4.3 Saturation excess and surface runoff

Overland surficial water flow occurs as a result of two main hydrological processes. Infiltration-excess overland flow occurs when a rainfall event is so intense that the majority of the water cannot infiltrate into the soil (Acreman & Holden, 2013), and is relatively rare in wetlands unless severely damaged. Wetlands reduce flood peaks by temporarily storing water in the substrate until the water table rises to the surface, at which time overland flow can occur via saturation excess overland flow. There is greater water storage in substrate when the water table is lower, such as during dry periods, while wet periods result in a high water table due to soil stores already being partially or completely saturated. Peat has a high water storage capacity due to high porosity within macro and micropores, or the network of cracks and spaces in between roots and organic matter (Acreman & Holden, 2013). The total porosity of peat can range from 71-95%, but often exceeds 80% (Rezanezhad *et al.*, 2016).

2.4.4 Groundwater and water storage

Groundwater movement in wetlands can be described by Darcy's Law, which states that groundwater velocity is determined by the gradient of potential or head (slope of the water table), and the hydraulic conductivity of the substrate. The measurement of wetland water tables is commonly undertaken using dip wells containing pressure transducers (Daniels *et al.*, 2008; Wilson *et al*., 2010), such as the work completed by Luscombe *et al*. (2016), who measured water tables in drained peatland areas in Exmoor National Park, England. They suggested the use of distributed monitoring networks to improve understanding of the hydrological processes regulating water tables, as the depth to water table is controlled by the size (width, depth) and position (local slope gradient and form) of anthropogenic drainage, as well as the variability of rainfall-runoff relating to its topographic contributing area (stream order, drainage density).

In peat material, the flow and storage of groundwater is determined by the extent of decomposition, as decomposition changes the structure, particle size and porosity of the peat. The general rule is that as peat becomes progressively more decomposed, the hydraulic conductivity of the peat decreases (Rezanezhad *et al*., 2016). Groundwater flow in highly humified peat has been observed to be non-Darcian, due to the pores being blocked by changing concentrations of gas bubbles which reduces the hydraulic conductivity. When water tables are high runoff will be rapid and flood mitigation will decrease as the storage capacity of the peat substrate nears its limit. Lower water tables will allow greater amounts of water to be stored, as the peat is not yet completely saturated (Acreman & Holden, 2013).

2.4.5 Wetland pH and electrical conductivity

Wetland soils are naturally acidic, with different wetland types varying in acidity. A primary cause for the acidity is the uptake of ammonium (NH_4+) by plant roots, which results in the release of H+ ions to balance valence charges, while peat and some peatland plants have a high cation exchange capacity (CEC). This means they naturally remove cations (positive ions) and replace them with H+ ions, which also reduces pH (Charman, 2002). The decomposition of organic matter also leads to acidification as microbes breaking down the matter cause mineralization and nitrification, decreasing the pH. Bogs are generally more acidic due to rainwater being the primary water source, which has little ability to counteract increases in H+ concentrations as it has lower levels of cations (Sorrell & Gerbeaux, 2004). Swamps and fens are influenced by groundwater and overland water flows, which can buffer the low pH as surface waters are often alkaline in nature. Measuring pH of wetland water can indicate differential water sources.

Electrical conductivity is a measure of a solution’s ability to conduct an electrical current, due to the presence of ions in the solution (such as Na+). Since swamps and marshes are prone to higher deposition of sediments and nutrients from surface water flows, they have higher electrical conductivities, as sediment is the primary carrier of bonded ions which can become soluble in water (Brown *et al*., 2015). Bogs, being primarily fed by rainfall, therefore have lower electrical conductivities, and measurements can provide insight into water sources and peat condition variables such as cation exchange capacity, although by itself cannot be used to quantify peat decomposition rates (Walter *et al.*, 2015). Measurement of electrical conductivity can indicate sediment deposition or differential water inputs (Brown *et al.*, 2015).

2.5 Peat physical processes

Wetlands can be composed of many different substrates. In peatlands, the primary substrate is peat, which consists mainly of partially decomposed organic matter (generally plant material) that has accumulated under anaerobic conditions over long periods of time. Peat as a substrate is important as it can influence a wide variety of physical and chemical processes in the ecosystems in which it is present.

2.5.1 Peat formation

Waterlogged conditions are characteristic of wetlands, with specialised plant species dominating due to functional adaptations that allow them to survive. Decomposition of organic matter by microbial activity is impeded in waterlogged wetland soils. Due to this reduced decomposition, the rate of accumulation of partially decayed organic matter is increased (compared to other ecosystems), as the layer of material is subsequently covered by newer organic matter, causing the nutrients and minerals to remain bound inside the underlying material. This new substrate, formed mainly of semi-decomposed organic matter, is called peat.

In peat bogs, the surface domes upwards in the centre where there is higher net accumulation of plant matter due to the imbalance between organic matter production and decomposition. This occurs with highly specialised plant types that can access the otherwise unobtainable bound nutrients and minerals, including mosses, sedges, and shrubs, although in New Zealand and Australia the dominant species are restiads, such as wire rush (*Empodisma* spp.) (Clarkson *et al.*, 2004).

2.5.2 Physical characteristics

Peat accumulates from organic matter under sustained anaerobic conditions, often with only minor conversion with constituents of mineral soils (mineralisation) (Kechavarzi *et al.*, 2007). As a result of this, peat is often composed of more than 60% organic matter, with less than 20–35% inorganic or mineral content. Fully formed peat that has not experienced enhanced degradation is typically composed of 88–97% water, 2–10% dry matter and 1–7% gas (Charman, 2002).

Wetlands vary in depth of the peat profile, which are dependent on multiple factors including the age. Young peatlands usually have peat accumulated to greater than 0.4 m in depth, while older peatlands exceed 3 m depth. In the Waikato region, the younger peat bogs such as Whangamarino wetland have peat depths up to 3.5 m, while older bogs such as Kopuatai peat dome in the Hauraki Plains have peat depths up to 12 m (Shearer & Clarkson, 1998). Otakairangi wetland in Northland has peat depths recorded up to 5.3 m (Clarkson *et al.*, 2015).

Peat structure and texture is primarily influenced by the overlying wetland plant composition which degrades into peat (Beheim, 2006). Peat can be formed by a variety of vegetative types, each of which forms different types of peat, which are broadly categorized into four groups: mosses, formed by genera such as *Sphagnum*, primarily in the Northern Hemisphere; herbaceous, formed by plants such as the restiad species; woody or secondary growth peats, found in a variety of settings including tropical peatlands, and detrital or humified peat (completely decomposed with unrecognisable plant material) (Charman, 2002). Each plant group has different structures which are mineralised by microbes to various degrees, resulting in different peat characteristics.

The dominant peat forming plants in New Zealand are *Empodisma* spp. (Sorrell & Gerbeaux, 2004, Hodges & Rapson, 2010), which forms peat from its fibrous roots making a dense mat. Other species, such as the fern *Gleichenia dicarpa,* have increased woody mass in stems and result in a coarser peat.

Ingram (1978) proposed a one-dimensional, conceptual framework consisting of two ordinal peat layers, which is still commonly used. The surficial layer of a peat column is known as the acrotelm, which is the layer of younger peat that exchanges gases and water between the wetland and the atmosphere. This layer is permeable, periodically aerobic, and has a high hydraulic conductivity. It experiences seasonal water table fluctuations, and allows for greater discharges from the wetland during high precipitation or flood events. This layer sits over the catotelm, the bulk of the peat body, which is usually permanently saturated and anaerobic, has low permeability (and therefore has lower hydraulic conductivity, making it relatively hydrologically inert), and comprises the bulk of the substrate in the wetland. The boundary between the acrotelm and catotelm is determined by the seasonal minimum water table (Holden & Burt, 2003).

2.5.3 Peat hydraulic processes

Water often flows across wetlands along a downslope gradient until it reaches the water table, the edge of the wetland or a body of water such as a stream or lake. The relative rate of water flow often implies that the primary path of water flow is through the acrotelm layer, which has a higher hydraulic conductivity than the catotelm (Holden & Burt, 2003; Bowden *et al.*, 2001).

Devito *et al.*, (1996) measured the hydraulic conductivity within a Canadian wetland. The average annual outflows of the wetland suggested that the majority of water movement occurred through the near surface layers, such as the top 10 cm, as peat layers from 20–50 cm experienced low lateral water movements. Halabuk (2006) showed similar results, concluding that the majority of water movement and discharge from wetlands occur from the surface layers (10 cm) of humified peat. However, Rizzuti *et al.*, (2004) found that the hydraulic conductivity is greatest in the upper 25 cm, where the peat layers are fibric to hemic, and in peat with a high percentage of macropores (pores >50 µm), while underlying hemic to sapric peat layers have high water holding capacities and low infiltration rates (Fraser *et al.*, 2001; Letts *et al.*, 2000).

2.5.4 Decomposition

Peat organic matter decomposes naturally over time under anaerobic conditions when natural water tables keep the majority of the substrate in a saturated condition. However, the rate of decomposition can be increased due to increases in microbial activity with increasing aerobic conditions (as the metabolic processes of microbes are more efficient in aerobic conditions). Exposure to oxygen allows aerobic decomposition to take place, occuring naturally through extreme events such as droughts, or with anthropogenic disturbance such as artificial drainage. Humification is the decay and conversion of organic compounds to a decayed state (humin). Peat that has been greatly decomposed implies significant decomposition over time, or dry, warm conditions that assist in increasing humification rates, while less humification indicates the conditions have been wet and cool with reduced decay, such as conditions found in natural peatlands (Charman, 2002). As peat forms from reduced decomposition in anaerobic conditions, moisture content is a key control on the rate of decomposition in peat (Grønlund *et al.*, 2008). With drainage and other mechanisms leading to lowered water tables, the peat becomes aerated, switching the microbial activity from anaerobic to aerobic, resulting in decomposition rates up to fifty times greater than under anaerobic conditions (Clymo, 1983; Holden *et al*., 2007).

Temperature is another key factor controlling the rate of microbial metabolisms, and therefore decomposition rates. Drainage is often associated with increasing temperature (due to resultant decrease in the water table), which can result in enhanced microbial activity and therefore peat degradation (Worrall *et al.*, 2004)

2.5.5 Peat surface oscillation

The peat surface in wetlands has been shown to reversibly oscillate in elevation (peat surface oscillation, PSO) (Nuttle & Hemond, 1988). PSO occurs where there is significant build-up of peat layers, such as in bogs. This is due to peat having the ability to expand and shrink in response to changes to the water level. As peat is made up of fibrous organic matter, it has a low density and a large water capacity, and therefore higher rainfall and saturation will lead to peat surface levels swelling on top of the water. Alternatively, when the water table lowers, the peat will compress due to its weak peat structural matrix no longer having support from pore water pressure (Fritz *et al.*, 2008).

Fritz *et al*. (2008) measured surface oscillation in Opuatia peatland, to investigate if the relationship between surface elevation (SE) and the absolute water level (AWL) was indeed linear, or seasonably variable. The dominant peat former at Opuatia is *Empodisma robustum* (species redefinition from *Empodisma minus* by Wagstaff & Clarkson (2012)), a restiad species which forms a dense lattice of specialised roots in the upper 7-10cm of the peat. The investigation found that while there was indeed a linear trend at some sites, others showed nonlinear relationships, which was suggested to be caused by floatation of peat layers during wetter seasons. It was also found that some sites experienced hysteresis, which was identified as a separation between a more linear 'drying curve' of peat as water tables receded, and nonlinear curves from the raising water table wetting the peat.

Absolute water level is the water level elevation with respect to mean sea level or another datum (such as bedrock), and is measured from a fixed reference point, generally anchored into the mineral substrate below peat layers. The relative water level (RWL) is the vertical distance between the surface of the peat and the water table below (the thickness of the unsaturated zone). Fritz *et al*. (2008) determined the surface elevation (SE) of the peat by subtracting AWL from RWL, and the surface oscillation by comparing the changes in surface elevation to changes in AWL. The surface oscillation is thought to be caused by a set of mechanisms, including flotation, compression or shrinkage, gas volume changes and freezing (Fritz *et al*., 2008). PSO influences the wetland hydrological regime, as it regulates water fluxes. PSO reduces the fluctuations of the RWL, which in turn increases the storability of the peat, reducing surface flows in the wetland.

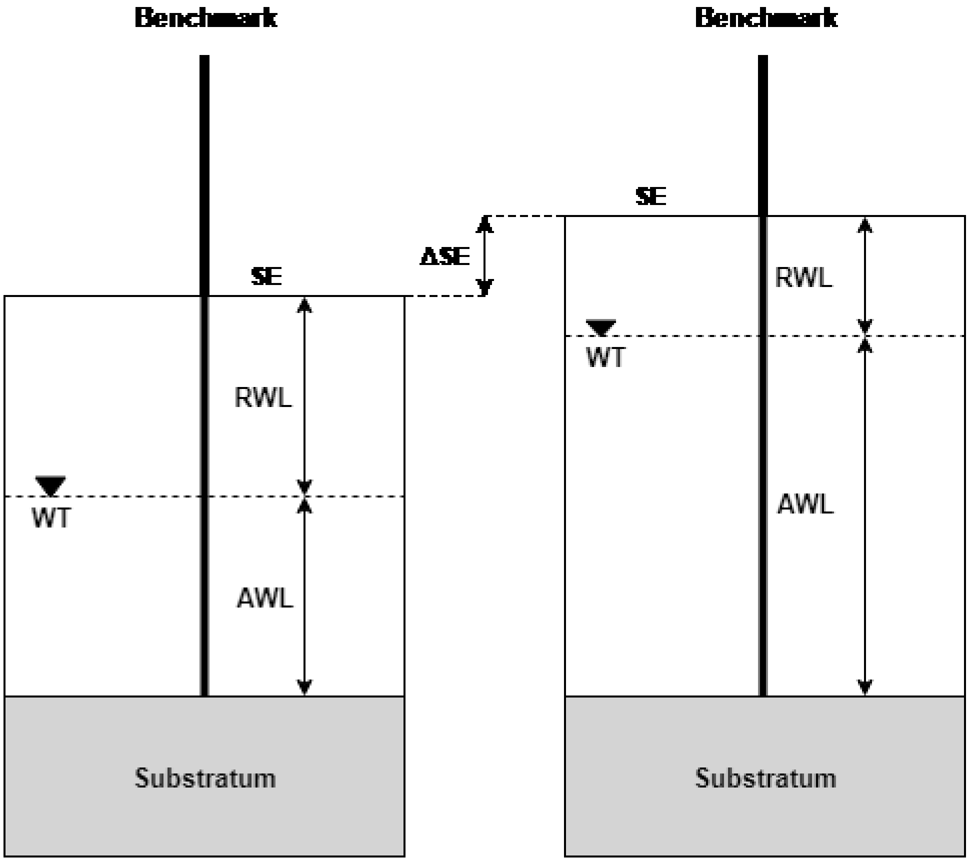

Figure 2.3. Peat surface oscillation, with changes to surface elevation with the water table position. SE= surface elevation, RWL= relative water level, AWL= absolute water level, WT= water table (modified from Fritz *et al*., 2008).

2.6 Vegetation

Vegetation species are generally organized along environmental gradients. The factors influencing the vegetation can include pH, cation concentrations, temperature, topography, substrate, moisture and shade. *Sphagnum* species are ecologically prominent wetland plants due to xerophytic adaptations and succession-directing acidification capabilities, and are the primary peat forming species in the northern hemisphere. Unlike *Sphagnum* mosses, the plant sustaining waterlogged conditions in New Zealand bogs is a herbaceous plant, *Empodisma* (Agnew *et al.*, 1993).

2.6.1 Functional adaptations of vegetation

Empodisma, the primary peat forming vegetation in New Zealand, has been subdivided into two species. *Empodisma robustum* is found north of 38^{o}S, while *Empodisma minus* is found south of that latitude (Wagstaff & Clarkson, 2012). *Empodisma* develops masses of cluster roots covered with fine root-hairs, which grow vertically towards the bog surface. Peat accumulation has been attributed to the high water-holding capacity and exchange properties of these roots. A combination of the slow decomposition of lignified tissues, allelopathic properties of phenolic acids and tannins, and the xeromorphic adaptations of *Empodisma* may explain the accumulation of peat in climatically unfavourable sites in the North Island (Kuder *et al.*, 1998; McGlone, 2009).

Empodisma and *Sporadanthus* (both Restionaceae) coexist in New Zealand raised bogs, yet *Sporadanthus* have significantly more depleted $\delta^{15}N$ natural abundance signatures than coexisting *Empodisma,* suggesting different nitrogen sources. Their root systems are spatially separated from *Empodisma,* which have a thick surface layer of about 50 mm of cluster roots in contrast to the deeper *Sporadanthus* roots (Clarkson *et al.*, 2009). It has been suggested that *Empodisma* and *Sporadanthus* acquire nutrients (primarily nitrogen) from different rooting zones, with *Empodisma* accessing nutrients at the surface from rainfall while *Sporadanthus* accesses nutrients from mineralization in deeper peat layers (Clarkson *et al*., 2009).

Other plants that survive in variable hydrological regimes have different adaptations to help them survive. Large airspaces (aerenchyma) in roots and stems allow oxygen to be transported from aerial parts of plant to the underground tissues (Sorrell & Gerbeaux, 2004), while emergent macrophytes have increased shoot biomass above the water. Others may have leaves or shoots that elongate very rapidly in response to water level changes, to ensure photosynthesis can still occur above the water (Sorrell & Gerbeaux, 2004).

2.6.2 New Zealand wetland vegetation

A chronosequence of restiad peat bogs in the lowland Waikato region, New Zealand, identified a major vegetation pattern with a dynamic hierarchical sequence from early successional sedges, to mid successional species such as *Empodisma*, the main peat-forming restiad species, to phases dominated by later successional *Sporadanthus ferrugineus* (Clarkson *et al*., 2004a). However, sites that were modified by drainage, fire, or weed invasion were dominated by non-restiad species, and as such age of disturbed bogs itself cannot be used to determine vegetation succession. As described in Chapter 1, New Zealand wetlands are primarily classified into three main types: swamps, fens and bogs. The wetland type is determined by the type of vegetation and the hydrology, which in turn are interlinked, and as such due to hydrologic changes across distance, wetlands can be composed of more than one class.

Swamps are wetlands that are influenced by water flows from natural systems, and as such the vegetation must be able to survive in dynamic conditions with high concentrations of sediment and nutrients. With a highly fluctuating water table, swamps frequently become inundated during winter periods. A common species, *Typha orientalis* (raupo) can survive a range of conditions and water levels, while sedge species (*Carex*) occur in areas of lower water tables. Kahikatea (*Dacrycarpus dacrydioides*) forests, although diminished, occur in many swamps. Invasive species include willows (*Salix* spp.), which outcompete natives and shade shorter vegetation, resulting in potential domination of the swamp area (Clarkson *et al*., 2004a).

Fens have intermediate levels of nutrients between bogs and swamps, as they receive water from surface water, ground water and precipitation. The vegetation therefore must be able to survive in the mesotrophic conditions (intermediate level of dissolved nutrients). Fens of New Zealand are typically dominated by *Schoenus* spp and *Machaerina* spp, while other common species that tolerate these conditions include *Phormium tenax* (New Zealand flax), *Gleichenia dicarpa* (tangle fern), *Leptospermum scoparium* (manuka) and *Coprosma teniculis* (Swamp Coprosma), and occasionally *E. robustum* (Clarkson, 2002).

Bogs are solely fed by rainfall, and as such provide conditions favourable for oligotrophic plant communities. The species adapted to these conditions in New Zealand are primarily restiad species (jointed rushes), such as *Empodisma* spp and *Sporadanthus* spp.

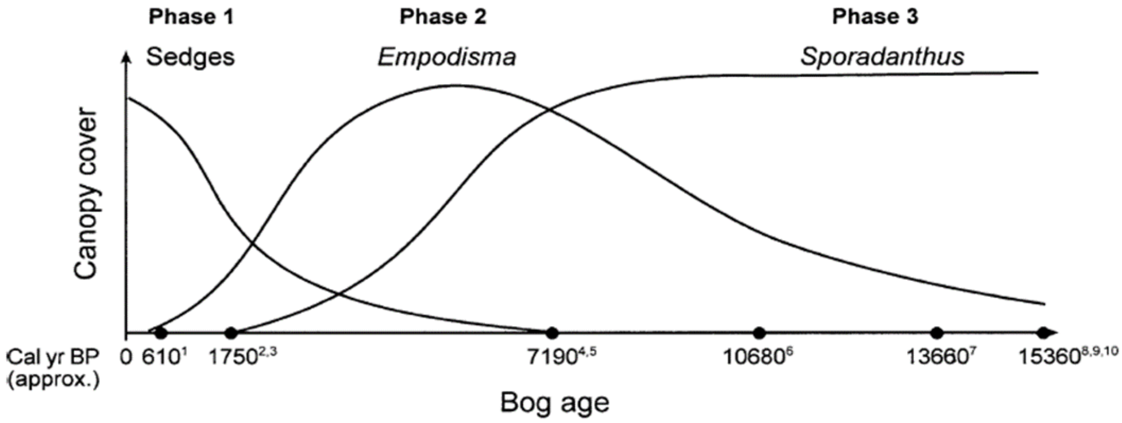

Figure 2.4. Species dominance and succession during the development of restiad bogs, with Waikato bogs marked. 1 = Duck Creek, 2 = Whangamarino, 3 = Opuatia, 4 = Torehape, 5 = Kopuatai (northern section), 6 = Lake Maratoto, 7 = Kopuatai (southern and central), 8 = Te Mimiha, 9 = Moanatuatua North, 10 = Moanatuatua (Clarkson *et al*., 2004a).

The vegetation composition in wetlands is directly influenced by the hydrology. As all vegetation requires water, the relative depth of the water table is vital in determining which species of plant will exist across the wetland area. However, a change in nutrient levels caused by an additional inflow of surface water to a wetland can cause a change in the vegetation community from bog species to those of a fen (Sorrell & Gerbeaux, 2004). Flood events can move a large amount of eroded sediment and dissolved nutrients from the upper catchment into the wetland as they can overtop surrounding river banks and artificial drainage ditches, changing the marginal vegetation composition (Charman, 2002). Therefore, vegetation is controlled by a combination of the physical and chemical hydrology.

The hydrology is a primary environmental condition which has significant effects on vegetation. Some species are better adapted to a range of conditions, while others favour specific hydrological regimes, causing the better adapted species to outcompete and survive. In bog ecosystems where nutrients levels are low, restiad species such as *Empodisma* dominate (Hodges & Rapson, 2010).

2.6.3 Vegetation indices

Indices are a common statistical method for examining vegetation communities in different ecosystems, such as the species richness index or the Shannon-Wiener index (Hill, 1973; Jost, 2006). They are used in wetlands to examine vegetation communities across spatial gradients and determine relationships between vegetation with other environmental variables (Xu *et al.*, 2015). The species richness index determines the total number of species, while the Shannon-Wiener index indicates diversity. Values for the Shannon-Wiener index are generally between 1.5 and 3.5 in most ecological studies, with the value increasing as both the richness and the evenness of the community increase. Species evenness index determines how close in cover each species is (are all the species evenly spread or is a single species dominating), while the importance value index (IVI) is a measure of how dominant a species is in a given area using the relative frequency, density and basal area of the species.

2.7 Wetland chemistry

Chemistry in wetlands can be used as an indicator for external inputs of nutrients, metals and inorganic elements, as well as for peat degradation and condition. Nutrients (such as nitrogen and phosphorus) are a major controlling factor for wetland ecology, as the biological productivity is limited by the availability of nutrients in substrate and water. Other elements such as metals may not normally be found in rain-fed wetlands, and therefore can often be directly linked to the hydrological regime.

2.7.1 Carbon

Carbon (C) is the primary component of organic matter, and as peat is primarily composed of organic matter, the majority of its chemical structure is composed of C and C compounds. Peatlands are characterized by an incomplete cycling of matter and therefore have a positive C balance, which results in peatlands contain 30% of all global soil C (Erwin, 2009). Peat organic matter decomposes from enhanced microbial activity, burning or draining, and with this it releases the previously locked C as CO_2 into the atmosphere. It has been shown in New Zealand that warm, temperate bogs dominated by *E. robustum* is a strong sink for carbon even during drought years (Goodrich *et al.*, 2017). Re-establishment and regeneration of wetlands containing peat may act as carbon sink to offset future emissions, acting as both a regulating and supporting service.

2.7.2 Nitrogen

Nitrogen (N) is a fundamental nutrient in environments that sustain life, constituting 78% of the Earth's atmosphere in the form of N_2 gas. The N cycle involves many complex pathways such as conversions of N into useable forms for vegetation. In wetlands the two main forms of N are nitrate and ammonium (NO_3– and NH_4+) (Brown *et al.*, 2015).

Nitrogen can be more abundant in swamps and some fens, due to inputs from surface and groundwater systems, while bogs have lower concentrations due to low concentrations in rainwater, a bogs primary water source. However, over the last century these ecosystems in the northern hemisphere have received inputs of atmospheric N up to 10 times larger than the pre-industrial levels (Van Aardenne *et al.*, 2001). The lack of oxygen in saturated wetland soils restricts nitrification (the process of creating nitrates) while also enhancing denitrification, meaning nitrate is limiting in wetlands (Sorrell & Gerbeaux, 2004). The low concentrations of N in wetland soils generally means it is a limiting nutrient for plant growth, although some studies show that any inorganic N is rapidly taken up by plants before it can be accessed by microbes (Bridgham *et al.*, 1998).

2.7.3 Phosphorus

Phosphorus (P) is an important nutrient that influences photosynthesis, respiration, energy transfer and other processes in vegetation. It naturally occurs in P mineral deposits, but is often present in small quantities in most ecosystems. Therefore, as the limited amount of P in most ecosystems can control the pace at which vegetation can grow, it is termed a limiting nutrient (Güsewell & Koerselman, 2002). The principal source for P is from the mining of phosphate rocks and ores, primarily for use in industry or as fertiliser application in agriculture. Most of the P retrieved from the minerals is insoluble and for use in fertilisers must be converted to a soluble form. The mineral form of P can be strongly bound to soil particles, which not only results in it being unavailable to primary producers, but also allows its transport with movement of sediment during erosion. The availability of P in ecosystems is also determined by plant uptake and the rate of conversion of organic matter to orthophosphate (soluble phosphate which can be used by plants, but also leached) by microbes (Sorrell & Gerbeaux, 2004).

Phosphorus, like N, can also be locked in organic matter and become unavailable for use by the rest of the ecosystem.

Sediments can also reduce the amount of available P in a system by binding it to ions present in the soils. With low water tables and the resultant aerobic conditions, metal oxides (such as Al and Fe) strongly bond with P, reducing its abundance for plant uptake. The bonds between P and sediment is strong, and are usually greater in clays which have a high surface area and large cation exchange capacity (Bridgham *et al*., 1998). Reducing conditions caused by high water tables can lead to P binding to another substance through adsorption with the ions in soil, removing it from solution. Absorption (one substance is dissolved by another and forms a new compound) occurs when the phosphate ions diffuse into the solid sediment particles. Studies have indicated that minerotrophic wetlands that have a higher total P content appear to be offset by greater immobilisation caused by geochemical sorption, reducing the amount available for use even though concentrations are relatively high. Phosphorus that is available for use has been found to be much greater in bogs totalling around 33% of the total P, while in aerobic swamps values were only 1% (Bridgham *et al*., 1998).

Phosphorus can also be lost from wetlands, moving into waterways in which it can become a potential threat to biological water quality by stimulating algal blooms. Literature suggests that alternating flood and drainage cycles can accelerate the rate of nutrient cycling and transport within the substrate, such as the release of P bound to Fe in anoxic conditions, and mineralization during aerobic drainage conditions (Aldous *et al.*, 2005). Niedermeier & Robinson (2009) investigated the loss and dynamics of soil P in a recently re-wetted, eutrophic fen peat, where management of the adjacent ditch water levels influenced the water table. Episodic P losses (higher drainage ditch water P concentrations) occurred during summer rain events, during autumn re-flooding, and during a 7-day period of water table drawdown by intermittent pump drainage.

2.7.4 Heavy metals and other elements

Environmental analysis in wetlands uses indicative metals and other elements, as substrates and solutions naturally have heavy metals and a variety of other elements present. Soil particles commonly have high levels of iron (Fe), aluminium (Al), copper (Cu) and zinc (Zn) as they have an ionic structure which is attracted to the negatively charged soil particles. Under low pH and anoxic conditions, such as the conditions present in many wetlands, these heavy metals can dissolve and can result in toxicity, affecting the biota (Brown *et al*., 2015), and can determine redox state of peat (Lindsay, 2010). Other elements, such as bromine and titanium, are indictors for decomposition rates because of residual enrichment (Martínez *et al.*, 2007).

Soil processes that control the availability of P in substrate also influence the concentration of potassium (K) and calcium (Ca). Potassium is used as a primary nutrient in fertiliser, used to improve the growth of vegetation stems, and higher concentrations in wetlands can result in a change in species composition and increases in productivity. Along with P and K, magnesium (Mg), lead (Pb) and mercury (Hg) can also be used for analysing species composition (Pyatt *et al.*, 1979; Norton *et al.*, 1997). Calcium concentrations are likely increased where pastoral farming has inputs of lime ($CaCO_3$) to increase soil pH. Erosion of the substrate in upland catchment areas often results in the movement of the Ca attached to soil particles down through the catchment, and can be used as a proxy for fertiliser movement.

Metals which are not used as nutrients, such as cadmium (Cd) and uranium (U), occur naturally in all environments. However, in peatlands the main sources include agricultural soils, bio solids, and P fertilizers moved through waterways, and as such they are a proxy for P based fertilisers in waterways (Roberts & Longhurst, 2002). The phosphate rock that is commonly used to manufacture fertilisers is known to have high concentrations of these toxic heavy metals, including Cd, Hg, Pb, and U. Cadmium and U are often considered among the worst elements for environmental health, and as they are relatively immobile, persist in the environment and the uptake and accumulation of these elements in the food chain make them a public health concern (Roberts, 2014). At Whangamarino wetland, Cd, U and the stable isotope $\delta^{15}N$ increased in abundance towards the Whangamarino River, which indicated that there were fertiliser inputs from pastoral farming in the upper catchment, which potentially had an effect on plants in Whangamarino wetland (Blyth, 2011).

2.7.5 Stable isotope $\delta^{15}N$

The vast majority of N in the natural system is in the form of $\delta^{14}N$, composing 99.63% of all N. The remaining 0.37% is in the form of $\delta^{15}N$, a stable isotope of N. Stable isotopes concentrations are usually expressed in ratios with units of delta (δ) or parts per thousand (‰), with $\delta^{15}N$ being reported relative to atmospheric air concentrations. A positive value indicates enrichment of the heavier isotope relative to atmopsheric, while negative values indicates depletion (Xue *et al*., 2009). Natural biological processes such as denitrification under anaerobic conditions can lead to enrichment of the heavier $\delta^{15}N$ isotope as microbes break $\delta^{14}N$ bonds easier than $\delta^{15}N$. Humified organic matter is enriched in $\delta^{15}N$ compared to recent organic compounds, and tends to increase with depth in a soil profile as humification increases (Kramer *et al.*, 2003). Fertilisers are often refined under strict procedures and hence usually have a consistent range of isotope enrichment: 0.5 to 5‰ for oxidized N (NO_3–), and lower values for urea (NH_4+). Therefore, $\delta^{15}N$ can be used as a tracer of nutrient inputs, but due to the variety of different biological processes which can contribute to isotope enrichment, identification of the direct causes cannot always be made (Xue *et al*., 2009).

In a natural terrestrial rain-fed ecosystems, the atmosphere is the primary source of N, and therefore $\delta^{15}N$ values in bogs are assumed to scatter around 0 ‰ (Broder *et al.*, 2012). However, vegetation in peatlands vary substantially in their $\delta^{15}N$ signature which could influence the $\delta^{15}N$ signature of the remaining peat material (from −11.3 to +2.7 ‰) (Krüger *et al.*, 2015). Drained or degraded peatlands possibly have increased $\delta^{15}N$ values with depth, due to the above mentioned processes (Krüger *et al*., 2015).

2.7.6 Peat chemistry in New Zealand wetlands

To characterise and distinguish between wetland classes in New Zealand, the chemistry of the substrate is analysed for total C, N and P, as well as pH and the ratios for C:N, C:P, and N:P (Table 2.1). Swamps and fens are partially fed by surface flows and/or groundwater, resulting in pH averaging around 5.2 and greater concentrations of nutrients such as N and P. Bogs, which are primarily fed by rainwater, are characterised by low concentrations of these nutrients and acidic pH levels (averaging around 4.0 for New Zealand bogs)(Clarkson *et al*., 2004b).

Table 2.1. Comparison of wetland means and ranges (in brackets) for key substrate parameters in swamps and bogs. New Zealand data obtained from 17 swamps and six bogs (Clarkson *et al*., 2004b). Opuatia wetland from six swamp and 22 bog sites (Browne, 2005), and Whangamarino wetland from 11 swamp and 12 bog sites (Blyth, 2011). TN= total nitrogen, TC= total carbon, TP= total phosphorus.

	New Zealand		Opuatia		Whangamarino	
	Bogs	Swamps	Bogs	Swamps	Bogs	Swamps
Soil pH	4.0 (3.7-4.4)	5.2 (4.1-5.9)	5.0 (4.3-5.3)	5.05 (4.8-5.4)	4.0 (3.5-4.3)	5.14 (4.3-6.4)
TC (mg cm3)	92.7 (24.1-239.8)	39.8 (5.2-100.6)	33.3 (24.2-42.29)	37.8 (29.8-47.4)	26.9 (18.1-41.4)	36.5 (20.0-52.6)
TN (mg cm3)	0.82 (0.02-1.83)	2.12 (1.15-3.24)	1.35 (0.7-1.98)	2.4 (1.7-2.8)	0.76 (0.37-1.57)	2.5 (1.5-4.0)
TP (mg cm3)	0.08 (0.01-0.20)	0.28 (0.15-0.69)	0.08 (0.03-0.13)	0.26 (0.18-0.33)	0.02 (0.01-0.04)	0.14 (0.03-0.21)
C:N	48.5 (35.9-79.7)	18.0 (14.2-30.6)	26.4 (17.0-49.0)	16.7 (14.0-19.0)	37.8 (26.4-48.7)	14.9 (11.3-21.6)
C:P	1904 (533-4221)	163 (45-435)	507.1 (236.9-1041.8)	161.4 (116.3-212.7)	2022.4 (947.4-2971.5)	332.8 (146.2-920.3)
N:P	39.0 (20.6-81.6)	9.1 (4.0-20.6)	18.8 (13.7-27.3)	9.54 (8.3-11.7)	52.4 (35.9-70.1)	20.7 (12.9-43.2)

The lower decomposition rates and the correspondingly high amounts of peat matter in bogs results in the total carbon (TC) present being larger than other wetland ecosystems. Compared to this, the increased rates of decomposition and larger mineral inputs from the variable hydrological regime in swamp and fen environments results in decreased concentrations of TC (Sorrell & Gerbeaux, 2004). Due to lower availability of N and P in rainwater and the high C content, bogs have high ratios of C:N, C:P and N:P, meaning that the nutrients are limited in bogs and reduced biological production. Compared to this, fens and swamps have slight additions of P from ground and surface water, resulting in lower ratios which suggest they are capable of greater biological production (Sorrell & Gerbeaux, 2004).

Vegetation only needs small amounts of P to aid in foliage and root growth, which leads to the development of communities of vegetation which are mainly limited by N in bog environments. Additions of nutrients to these wetlands by altered hydrological regimes or external sources cause the vegetation to switch from P to N limitations and change the communities from bog to swamp vegetation (Sorrell & Gerbeaux, 2004).

2.8 Anthropogenic disturbance of wetlands

Approximately 90% of New Zealand wetlands have been degraded or eliminated by anthropogenic causes (Clarkson *et al.*, 2013). The main form of degradation is through drainage to increase the amount of productive agricultural land, due to their positions in low lying land. The remaining wetlands are still relatively diverse with unique flora and fauna. However, they are still at risk from ongoing activities, including drainage, fire events, nutrient enrichment through intensive land use, and invasion of exotic species. Land development in the Hikurangi swamp area has eliminated over 96% of the former wetland extent in the area, as well as eroded the local biodiversity and ecosystem services (Figure 2.5).

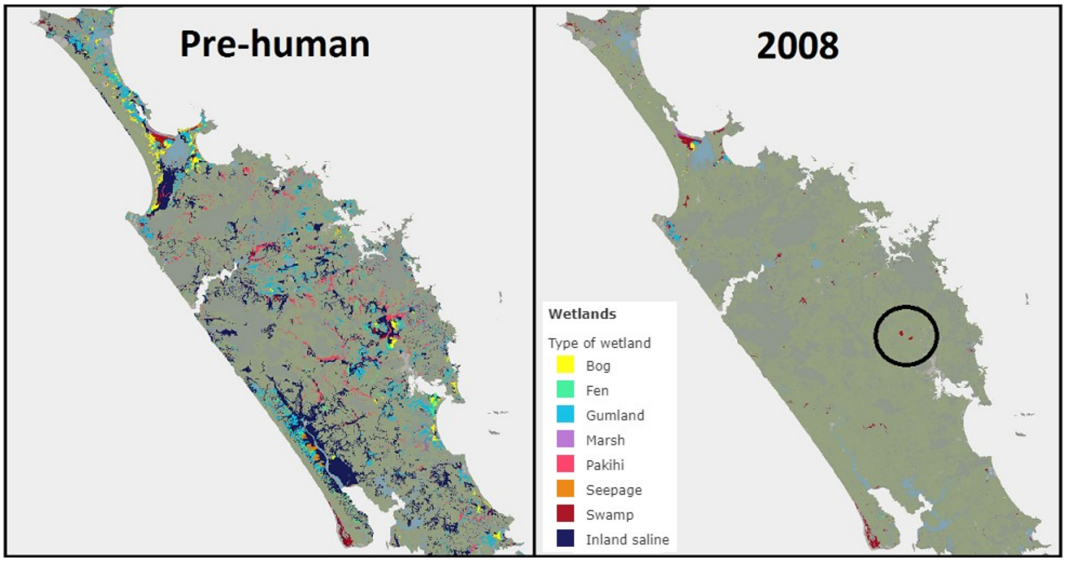

Figure 2.5. Northland pre-human wetland extent compared to 2008 recorded extent, with Otakairangi wetland visible in 2008 as one of two red spots circled (the other being the Wairua River wildlife management reserve) (http://statsnz.maps.arcgis.com).

2.8.1 Hydrological modification

Drainage of wetlands directly impacts upon their natural hydrology. Naturally diffuse flows of water, both as groundwater or surface overland flow are altered as water is directed into man-made channels. Lower water tables will induce vegetation change, promote C loss and (other than in the wettest mires) reduce the rates of peat formation (Price *et al.*, 2016). Biomass changes both in composition and abundance, and alters the nutrient pool in the area, which can lead to further ecosystem changes with dominance of other species. Drainage also allows oxygen to enter the peat, which can rapidly alter the microbial processes from anaerobic to aerobic (Pronger *et al.*, 2014). Enhanced peat degradation following this change results in surface subsidence, and can reduce the acrotelm hydraulic conductivity by two to three orders of magnitude.

Human influenced changes in water tables and nutrient content of surface waters result in surficial degradation of peat (Shearer & Clarkson, 1998). Lowering water table and/or allowing introduction of minerotrophic water will increase rates of peat degradation, but while the degraded peat most evidently occurs at the surface adjacent to the drainage ditch where the water table is depressed, the effects of the ditch may also be evident beyond the immediately adjacent peat. This has been shown where Kopuatai peat bog has been little affected by human influence, while Moanatuatua and Whangamarino wetlands were subjected to peat degradation that extended spatially past the extent of the drainage system (Shearer & Clarkson, 1998).

Shearer & Clarkson (1998) investigated the effects of water table decrease on the vegetation in Whangamarino wetland. A causeway built across the northern bog increased the nutrient supply to the wetland, while sand extraction and a flood control scheme in the Whangamarino River lowered the water table. New Zealand peat bog vegetation succession begins with sedge communities, and then progresses to ecological compositions including the restiad species *Empodisma* spp and *Sporadanthus*. The drainage and degradation of the wetland resulted in conditions similar to early stage vegetation compositions, which increased the wetland susceptibility to invasion from *Salix cinera*, an invasive weed which is able to establish in the early stage community.

Water table drawdown is greatest in direct proximity to the drain, and commonly extends tens of metres from the drains (Price *et al*., 2016). However, the effects of the drainage can be seen at much larger distances depending on the composition and the structure of the peat, as well as the underlying substrate (Landry & Rochefort, 2012). Poulin *et al.* (1999) showed that drainage can impact vegetation up to 60 m from the drainage source, while Trettin *et al.* (1991) showed drainage impacts up to 150-200 m from drainage ditches in peatlands which are situated on top of sand deposits.

The major impact of drainage is the lowering of the water table that leads to an increase in the air-filled porosity of the peat, which in turn affects microbial processes and thus decomposition rates. Prevost *et al.* (1999) investigated the impact of drainage on wetland soil solutions collected from 1.5, 5 and 15 m distances from the centre of multiple ditches, at a peatland site in eastern Québec, Canada. They observed that the solute content of soil solution was enhanced by drainage, with the effect generally proportional to ditch proximity for S and Mg, while increases in N, Na, K and Ca were mainly observed within 5 m of ditches.

2.8.2 Fire

Peatland fires are a natural destructive force that have multiple effects on the peat, vegetation, hydrology and nutrients. The rate of fire events increased significantly with human occupation events, as they attempted to remove areas of wetlands to create space for agriculture.

Burning peatlands results in the removal of surficial vegetation and biomass, as well as the current surface plant litter that would contribute to a new layer of peat. The fire dries the surface layer, which lowers the water table and over time will reduce the total plant cover. However, the main concern with the burning of peat is that it will severely impact the structure of the peat, allowing invasive species to colonise, while also releasing the nutrients that were bound in the organic structure (Norton & De Lange, 2003). Mineralisation or oxidation of the peat matter releases C into the atmosphere primarily as CO_2, but also as CH_4, and as peat wetlands are the world's largest sink of C, this poses a major threat to atmospheric C levels.

While burning events may have negative impacts, they are also a natural mechanism for disturbance and recolonization of wetland species, and can be an important factor for species diversity. Clarkson (1997) investigated the effects of three fires in Waikato wetlands, two in Whangamarino wetland (1984 and 1989), and one at Moanatuatua bog (1972), where the rates of vegetation recovery were established by monitoring the vegetation following the fire events. Rhizomatous species recovered rapidly following each fire event, while seeding species that were wiped out completely took longer to recolonise. Early colonising and adventive species rapidly established and dominated in the following two years due to open space and temporarily high nutrient levels, while the original state and community composition was reached in six and twelve years (Whangamarino wetland and Moanatuatua, respectively).

Norton & de Lange (2003) ignited controlled fires in Whangamarino wetland and recording the effects on vegetation, peat and microclimate, and found that fires can increase species diversity, as well as support critically endangered species. After burning, there was an increase in soil temperature as increased solar radiation reached the surface, and similar to Clarkson (1997), rhizome species dominated while obligate species (restricted to wetland conditions) did not immediately recolonise to previous numbers. Both species richness and diversity increased post-burning, but began to decline to pre fire composition over 4.5 years.

Corybas carsei, a rare orchid species, was killed during the fire, but re-established one year after burning with greater abundance, flowering and germination (Norton & De Lange, 2003).

Leptospermum scoparium (manuka) is the only New Zealand indigenous tree species which is known to utilise serotiny, an ecological adaptation where the seeds respond to a trigger rather than maturation. For manuka, whose seeds are fire resistant, the trigger is a burning event which allows the light, wind dispersed seeds to rapidly colonise early successional sites. In burnt peatlands, this process allows manuka to rapidly colonise over a wide area before other species can (Perry *et al.*, 2014).

These studies show that while fire can have negative impacts, it also leads to increased species diversity with recolonization of plants which would otherwise be outcompeted. Burned areas return to similar health and increased species diversity compared to pre-burning after significant time. Fires reset vegetation development, but do not alter the recovery and successional pathways (Clarkson, 1997).

2.8.3 Peat subsidence

The fibrous structure of peat allows it to hold a high volume fraction of water. However, as it dries following drainage, the structure of the peat collapses, with the pore spaces in the structure reducing both in size and quantity (Charman, 2002). The weight of the overlying matter is supported by water that is in the peat structure. As the water is removed with reduced pore space, the weight compresses the peat, making it more compact, breaking down the weakened structure and resulting in an increase in bulk density (Beheim, 2006). This is an irreversible change to the structure, as once compacted it cannot be reversed to the original form and results in surface subsidence. The parallel decrease in hydraulic conductivity with the lowered surface can result in increased flooding risk.

2.9 Management and restoration of degraded wetlands

The management of wetlands is vital to maintaining or improving the relative health of ecosystems. Human aided restoration processes need to target both the internal and external factors that are causes of degradation, including invasive species, excess nutrient inflows, and changes to hydrology, while also attempting to improve the ecosystem functioning.

2.9.1 Hydrological changes

The re-establishment of high water tables (commonly termed re-wetting) is one of the primary goals of peat restoration, but can be problematic since engineered solutions to deactivating drains cannot replicate the natural water table regimes (Holden *et al.*, 2004). Water regimes can be altered by drainage or the diversion of surface water and groundwater, while declines in water quality are generally associated with changes in land use and particularly the intensification of agriculture with increased nutrient and sediment loads in surface waters. If drainage ditches are not regularly maintained they can fill in with vegetation and sediment, causing them to lose their effectiveness in water removal (Fisher *et al.*, 1996). This neglect of drains can potentially be one of the simplest management strategies proposed to return peats to favourable conditions, however, the tendency of drains to infill depends on the type of material forming the floor, the slope angle and hence the resistance to scouring (Fisher *et al.*, 1996). Natural healing of ditches only seems to occur under certain conditions, including gentle slopes and in peats with extremely low hydraulic conductivities.

Price (1997) tested a range of water management approaches that attempted to ameliorate factors limiting *Sphagnum* regeneration in North America. They concluded that while simply blocking the drains caused good water table recovery during the wet spring period, the water table recession was much faster and greater in extent than in an undisturbed area, resulting in an unbalanced water table regime. The rewetting of the Cacouna Bog, Canada, by blocking the primary drainage network, caused the water table to rise and flood lower-lying trenches. Topographical variability and the location of the peat dams strongly influenced the magnitude of the water table rise at any given location, with the site-average water table rising by 32 cm. The higher water tables resulted in increased runoff variability, dependent upon antecedent conditions and event-based precipitation dynamics, but also produced hydrological conditions more favourable for the recolonization of *Sphagnum* mosses (Ketcheson & Price, 2011).

In New Zealand, a notable restoration example is Dunearn peat bog in Southland, which was modified with deep internal drains and an external ring drain. Plugging of internal drains with peat sods was conducted to restore wetland hydrology, with groundwater and peat levels increasing after blocking. However, increased water pressure behind one of the peat sods resulted in a “blowout”, again lowering the water table in one part of the peat bog and damaging adjacent farmland (Peters & Clarkson, 2010)

In Revonneva Nature Reserve, Finland, a swampy mire was drained by a drainage channel from the 1960s onwards, which caused significant drying with water table drawdown. While the drain itself was 1-2 m wide and deep enough to reach the underlying mineral soils in places, subsidence of the surrounding peat caused the channel to widen to about 50 m in width. Due to this extensive subsidence, the restoration planners decided that blocking the channel itself would only led to partial success in restoration, and instead constructed sizeable peat embankments across the channel extending across the whole of the area affected by subsidence. The result was a 2.5-km long chain of consecutive pools, with significant increases in open water throughout the peatland. Near natural hydrological conditions have been restored in a total of some 200 ha of drained peatland, and within two years the vegetation community rapidly reverted towards those of natural flark fens (Similiä *et al.*, 2014).

2.9.2 Restoring degraded peat

Landscape components influence each other in two directions. Modification of a peatland’s hydrology will directly affect the area’s flora and fauna, while also influencing the hydraulic soil properties, the relief, and eventually the composition of the peat body (Nakamura *et al.*, 1997). The degradation stages of peat therefore differ fundamentally from each other, with further degradation stages implying a more intense modification of the same components, while also affecting other components of ecosystem functioning. In undecomposed peat, the organic fraction (i.e. C content of the peat) is naturally high compared to that of the mineral fraction. However, as peat degrades and compresses over time, TC reduces, and the mineral fraction therefore increases. Therefore, peat core analysis for the nutrient and mineral balance of the substrate is a good indicator of the degradation of the wetland. For this reason, more degraded peatlands are more difficult to restore. They require explicit attention to components that might not have been directly impacted, but that have degraded as the longer-term but inevitable result of impacts upon other components (Schumann & Joosten, 2008). To restore systems with degraded peat, possible options include removing sections of top degraded peat layers to indirectly raise water tables, or block nearby drains to directly raise them (Schumann & Joosten, 2008), or to revegetate the surface by re-seeding with peat forming species (Zhang *et al.*, 2011).

2.9.3 Restoring peat accumulation

The final outcome of a restoration effort in a peat wetland is to have restored ecological and hydrological functioning, an indicator of which is accumulation of new peat. Lucchese *et al*. (2010) proposed that a *Sphagnum*-dominated peatlands could only be considered functionally 'restored' once organic matter accumulation achieved a thickness where the mean water table position in a drought year did not extend into the underlying formerly cutover peat surface. Immirzi *et al.* (1992) suggested that only peatlands that are sufficiently intact hydrologically can form more peat, while the recolonization of important peat-forming species is an important tool in restoring degraded peat (Holden *et al*., 2004). In the Northern Hemisphere the genus *Sphagnum* is ecologically essential for peat growth, and hence water tables must be maintained at a high level without great fluctuation. However, as the primary peat forming species in New Zealand is *E. robustum*, restoring peat accumulation should involve creating conditions beneficial to the spread and growth of *E. robustum*.

Chapter 3. Site Description

Located near the headwaters of the Kaipara Harbour catchment, the Hikurangi floodplain of the Wairua River was once a mosaic of diverse wetland types (collectively called the Hikurangi Swamp). This floodplain has been largely drained and developed for agriculture, forestry, horticulture, and lifestyle blocks. It is a biodiversity hotspot which is characterised by fragmented wetland and riparian forest remnants that hold several nationally threatened and at risk species (Clarkson *et al*., 2015). Past drainage and hydrological diversions to enable land development on the floodplain resulted in less than 4% of the wetland extent of the former Hikurangi "Swamp" remaining today (Clarkson *et al*., 2015).

Despite the small size of the Otakairangi wetland (260 ha, or 2.6 km^2), it is the largest remnant wetland on the Hikurangi floodplain, having experienced more than a century of modification, and is currently of interest for its restoration potential. Otakairangi stream is assumed to have entered in the northwest, fanned diffusely over the wetland area and then outflowed in the south east, creating a range of swamp and fen conditions. A large central zone would have been isolated from these surface flows, being primarily fed by rainfall, but due to the artificial central drainage, this has changed (Campbell, 2017). This hydrological modification to the wetland was performed to allow farming around and upstream of the wetland sometime around the 1920's or 1930's, with articles from the 1920 Northern Advocate documenting the efforts local landowners were undertaking to drain the swamp land (Campbell, 2017). The wetland is now bordered by drains which intercept any overland flow from the adjacent hill country, as well as being divided by a larger drain that runs through the centre of the wetland (Figure 3.1). The Otakairangi stream drains a catchment of around 20 km^2 of hill and valley country, and now runs through the large central drain in the wetland. While historical aerial photographs show the stream entering the wetland near the present day location, it was more likely to have run diffusely across the surface of the wetland, rather than being concentrated into a single linear path (Campbell, 2017).

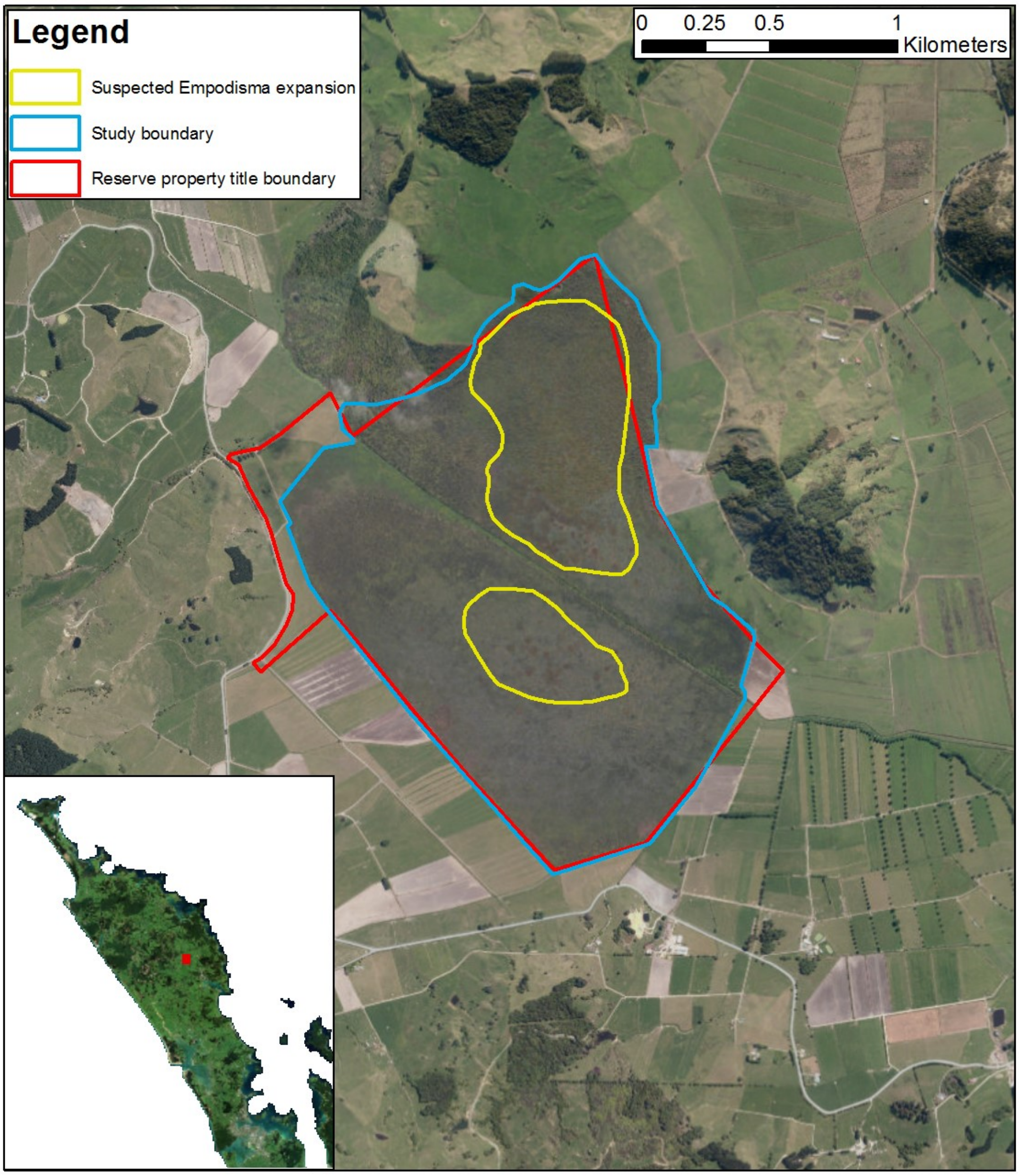

Figure 3.1. Otakairangi wetland reserve, showing the approximate study area, the suspected areas of *Empodisma robustum* expansion, and the reserve property boundary (Kendal, 2016).

3.1 Physical Environment

3.1.1 Developmental history

The development of the peatland was described by Newnham (1992) through analysis of a single core which was retrieved from an area south of the current conservation reserve. The area was the site of a lake prior to 30,000 years BP, presumably part of the larger lake system that covered the current Hikurangi plains area, shown by aquatic sediments and deposits of blue clay. These sediments was overlain by alternating layers of clays and peat, representing either gradual infilling of the lake, or alternating drying sequences. From 29,000 years BP, pollen from several wetland species were present, such as wire rush (*Empodisma robustum*) and tangle fern (*Gleichenia dicarpa*), indicating the creation of fen-like conditions. Similar to other New Zealand wetlands, natural fires (marked by charcoal) occurred which altered the plant assemblages and changed the developmental trajectory of the wetland. The earliest records of fire were around 21,500 years BP, while the greatest charcoal concentrations occurred between 13,600 BP and 7,820 BP (while *Empodisma robustum*/*Gleichenia dicarpa* communities were dominant), indicating that bog conditions had fully developed (Newnham, 1992).

The Otakairangi Stream, at the point it enters the wetland, drains a 20 km^2 headwater catchment, composed of hill country and valley bottoms which are primarily used for pastoral farming and supplementary feed crops. The stream, which once would have passed diffusely through the wetland, now passes through the wetland as a long, deep drain. The wetland is comprised of two large sections which are separated by the central drainage ditch which runs north-west to south-east. The south-western section is completely bordered by drains, which separate the relatively elevated wetland from the surrounding farmland. The north-eastern section lies below hill slopes with varying land uses. Similar to the south-western section, the primary land use is farming, with pasture dominating the majority of the eastern and north-eastern hill country. However, to the north of the wetland lies a hillslope dominated by a regenerating native bush, with many native and invasive plant species slowly encroaching onto the wetland where the substrate becomes more mineralised.

3.1.2 Climate

The wetland, being situated in the sub-tropical Northland region, frequently experiences droughts during the generally hot and dry summers, while the winters are mild, with only localised ground frosts (Chappell, 2013). The mean annual rainfall at Otakairangi wetland for the period 1990-2017, as recorded by the Northland Regional Council rain gauge at Rowland Road (adjacent to Otakairangi wetland), was 1259 mm per year. While the precipitation generally occurs in the cooler seasons, summer is prone to intense rainfalls during the passage of ex-tropical cyclones (Campbell, 2017).

3.1.3 Vegetation and substrate

The wetland shows zonation of plant species, with the edge banding near the drains being characterised by modified wetland and exotic species, such as raupō (*Typha orientalis*), reed sweetgrass (*Glyceria maxima*), flaxes (*Phormium tenax*), pastoral weeds and grasses. However, in the central, less modified sections, the vegetation comprises of a mosaic of wetland vegetation types. While primarily tall mānuka (*Leptospermum scoparium*) shrubland (large patches of which are dying) over a dense groundcover of tangle fern (*Gleichenia dicarpa)*, there are patches of wire rush (*Empodisma robustum*), Pakihi rush (*Machaerina teretifolia*), and *Coprosma* species.

The primary substrate of the wetland is peat, which is formed from the accumulation of plant remains following low decomposition rates under higher water table conditions. Evidence indicates this wetland started forming as far back as 29,000 years ago, and as such there are areas of deep peat, with over 5 m depth of peat in the centre of the wetland near the drain (Clarkson *et al*., 2015). However, this peat has been highly modified due to fires and changing hydrology, as well as mineral influences from the exterior of the wetland along the edge margins.

3.2 Hydrological and ecosystem sampling design

A single transect line was established in late 2017 for the measurement of the hydrological regime focusing on the influence of the central drain. This extended from the mid-point of the central drain to 280 m laterally (north-east) from the drain into the wetland along a pre-established path (Figure 3.2). This was composed of five dip-well sites (OT 1 to 6, positioned 0, 20, 50, 100 and 280 m from the drain) (Figure 3.2) with self-logging pressure transducers (INW level scout, Seametrics). A single site (280 m) was equipped with three transducers to provide atmospheric pressure data along with absolute and relative water levels. Later additions include an additional site at 290 m (OT 7, 10 m past the original transect) to act as an additional relative water level site but in a different vegetation type, and four sites mirroring the original transect line on the opposite side of the drain (south-west), (OT 8, 9, 10 and 11, positioned 20, 75, 140 and 280 m from the drain) (Figure 3.2).

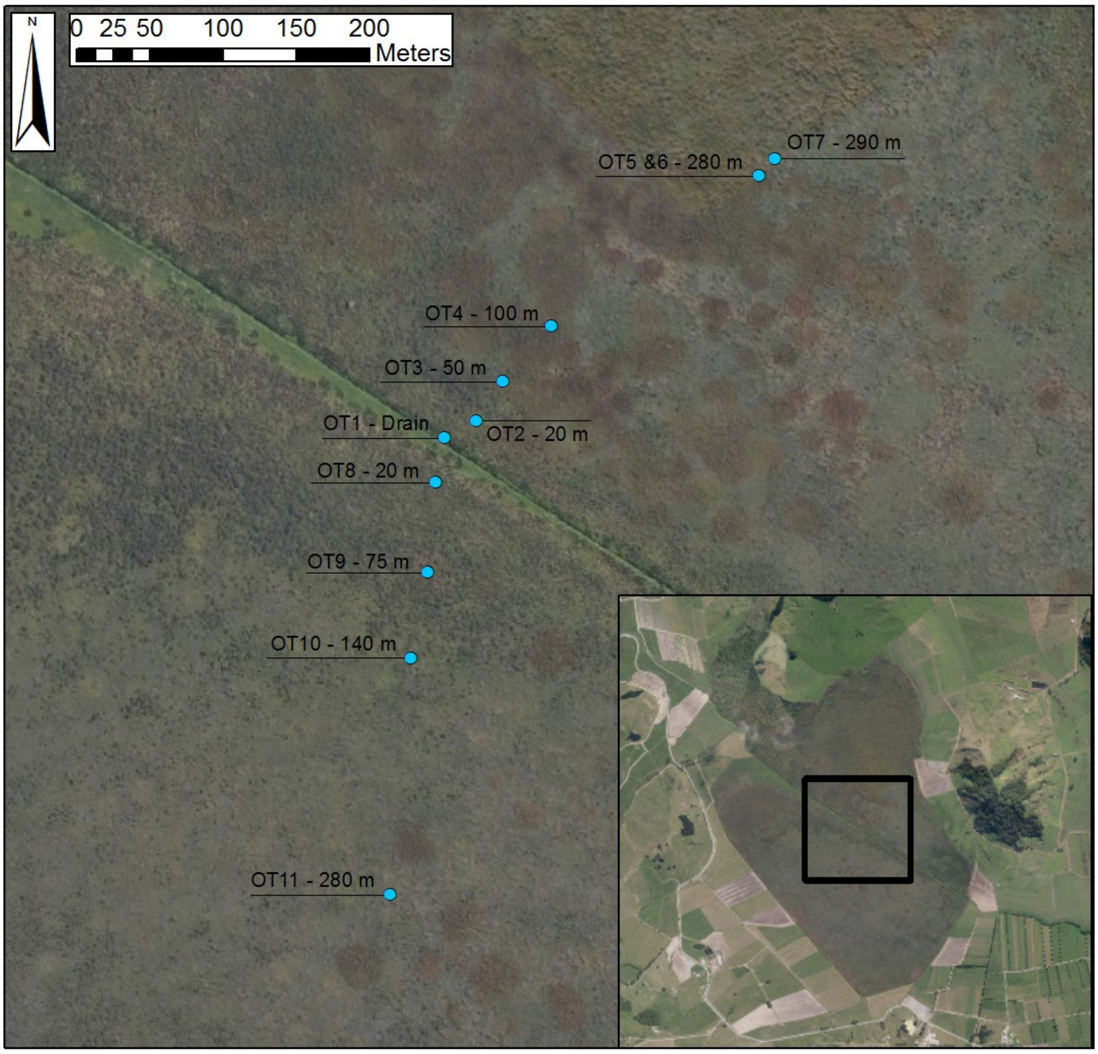

Figure 3.2. Positions of the hydrological network crossing the central drain.

Five transect lines were established for ecological and geochemical analyses over the duration of this study (Figure 3.3) using the methodology described by (Clarkson *et al.*, 2004b). Each transect line was designed with the intent of measuring the changes in hydrology, vegetation, and chemistry from a point of major influence in the wetland, such as drains or contrasting land use. Each transect line covered a total distance of 375 m, with six plots established at intervals of 75 m (see Section 5.2.1 for further details).

Transect A was established crossing the central drain utilising the hydrological transect line. This was established with the intent of measuring the effect of the central drain at what could be seen as the core of the wetland area, with three plots on either side of the drain.

Transect B was established extending into the wetland from the southernmost point of the wetland, where deep circuit drains border the wetland. The objective of this transect line was to determine the effect of edge drains on the southern side, as well as investigate the visible stress and dieback seen in *L. scoparium* across the southern section of the wetland.

Similar to Transect B, the objective of Transect C was to measure the effect of the border drains, but aimed at the northern side of the wetland where steep hill pastures can potentially cause increased runoff of nutrients into the wetland over the shallower northern drainage ditch.

The intent of Transect D was to establish the extent of effect that the northern native bush has on the wetland. Transport of mineralised sediment into the wetland can cause a change in vegetation community, and result in invasion of species from the hillslope in what is suspected to be a swamp-like area.

Transect E replicated Transect A, measuring the effect of the central drain. However, this line was positioned 500 metres south from Transect A, and included two plots to the north-east of the drain, and four to the south-west. This section of the drain was shallower, wider and seemingly infilling when compared to Transect A where the drain was deep and fast flowing drain. Therefore, this section was thought to be experiencing different hydrologic regimes and patterns.

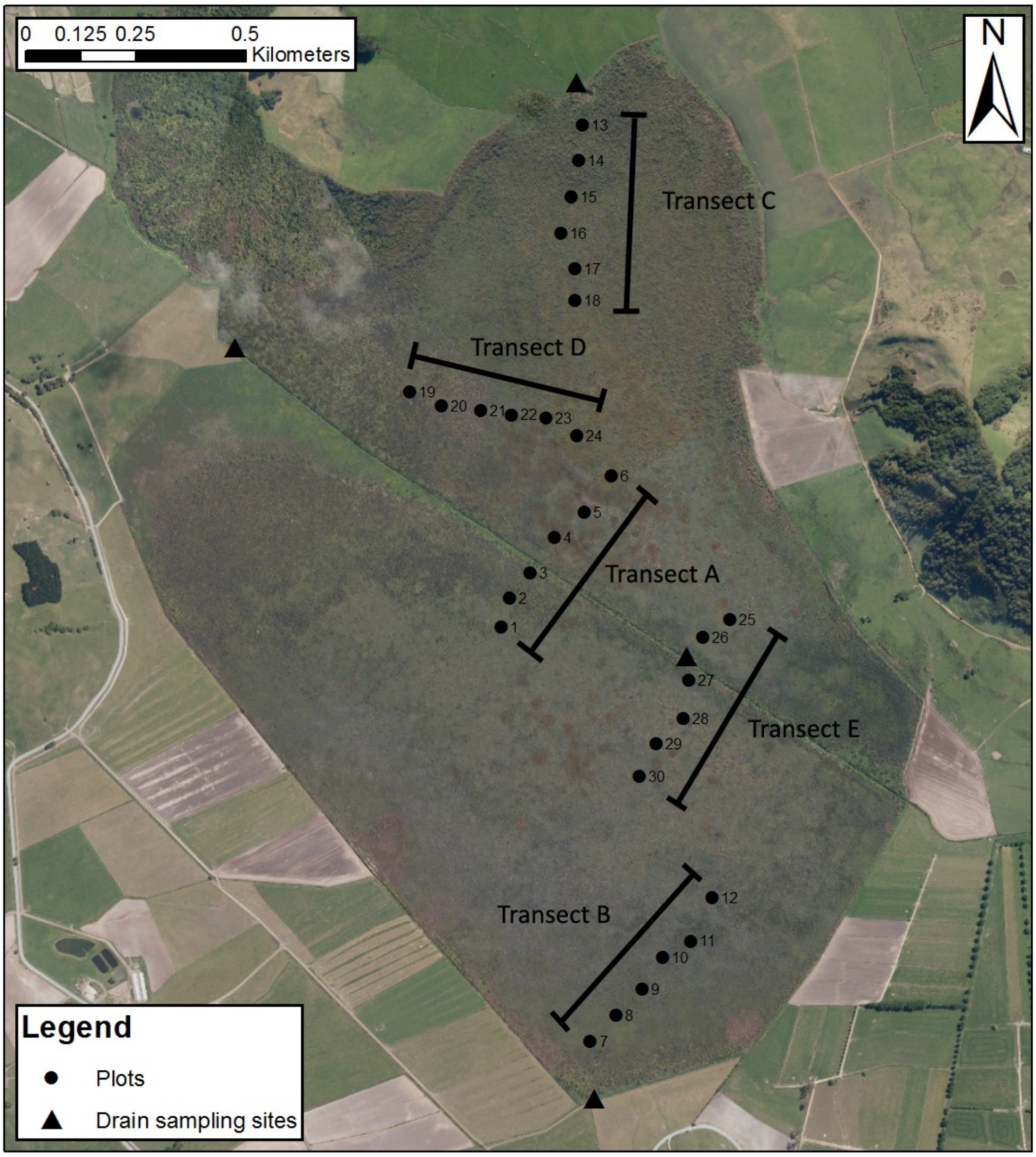

Figure 3.3. Location of the 30 vegetation plots along the five transect lines, as well as the four drain sampling sites.

Chapter 4. State and patterns of peat material

4.1 Introduction

The wetland peat was assessed along each of the transect lines in order to identify how differing hydrological regimes and nutrient inputs were influencing the physical and chemical state of the peat. A variety of environmental indicators were used to describe the current state of wetland substrate, as the assessment of peat using standard methods provides an understanding of the wetland and can be used to classify wetland classes relative to New Zealand studies (Clarkson *et al*., 2004b). Field assessments and measurements such as the von Post index, pH and electrical conductivity were taken while peat core samples were taken for laboratory analysis, including physical measurements of dry bulk density, volumetric moisture content and mineral content. Elemental analysis of nutrients (carbon (C), nitrogen (N) and phosphorus (P)) and inorganic elements that are associated with inputs of surface water, sediment and fertiliser (uranium (U), iron (Fe), calcium (Ca) etc.) were also utilised for the characterisation of peat.

4.2 Methodology

Peat samples were collected from each of the established sites along the transect lines (see Chapter 3.2 for details on plot establishment). Two samples were collected per site for a total of 60 cores over the entire wetland. The methodology components include:

- Collection of peat samples from established plots
- Measurement of degradation using the peat humification index
- Bulk density measurements
- Subsampling and processing
- Chemical analyses of nutrients and metals

4.2.1 Peat core collection

At each vegetation plot, two samples were collected from the upper 10 cm of peat material (the acrotelm, or active layer of peat) following the methods outlined by Clarkson *et al*. (2004b) during late January and early February of 2018. The peat samples were collected from the upper 10 cm of substrate as it is the most recently formed peat (or deposited sediment), which allowed an accurate observation of the physical and chemical state of the wetland across a range of areas. A cylindrical stainless steel corer (100 mm diameter by 75 mm tall) was used to extract the samples, which had been slightly modified for use in different types of peat material. The bottom of the corer was filed to give a sharper edge, and then precision filed to give small serrations to provide improved cutting for areas of more fibrous, tough peat.

Once at the site, a small patch of vegetation was cleared using a serrated knife to expose the peat surface. The corer was used by slowly rotating the corer into the upper 10 cm of peat material, so that the sharpened serrated edge cut through the material rather than compressing it. In some areas the peat material was made of densely packed, fibrous stem material and was too tough for the corer to cut through (Figure 4.1). In these cases the serrated knife was used to assist by cutting through the larger roots and woody material. Once the sample was removed from the ground, the upper and lower edges were trimmed using the serrated knife to ensure the volume matched the stainless steel corer. The samples were stored in labelled zip lock bags, and then transported in a large plastic bin until they were placed in a chilly bin at the vehicle. Two samples were collected per site to ensure enough peat material was collected for chemical analysis.

Figure 4.1. Example of a peat core, collected from the upper 10 cm in a dense patch of *Gleichenia dicarpa*.

4.2.2 Peat humification index

Humification of the peat was assessed in the field by the use of the von Post technique. This technique is a common method of assessing the relative degree of peat decomposition by visual observation of the peat structure, plant remains, peat and water colour as well as the textural feel of the peat matter. The extent of decomposition is relative to a scale from 1 to 10, in which a low rating of 1 represents little to no decomposition, and 10 indicates completely decomposed peat material (Figure 4.2). The peat assessment was undertaken at every site by taking a small peat sub sample from the upper 10 cm, although one site was shown to have strong mineral soil characteristics due to its location. Refer to Clarkson *et al*. (2004b) for further details on this method.

Figure 4.2. Range of substrate types. Highly mineralised peat (left) rated 7 on the von Post scale, *Gleichenia* based peat (centre) rated 4, and *Empodisma* 'proto-peat' core (right) rated 2.

4.2.3 Groundwater pH and electrical conductivity

Field pH and electrical conductivity measurements were taken at each of the vegetation plots during late January to early February of 2018. The hole remaining from where each peat core was collected was excavated further and left to fill with groundwater (approximately 10 minutes). A pH and conductivity meter then measured the relative levels of each, once readings were stable.

4.2.4 Bulk density

The dry bulk density (DBD) of peat was established by measuring volumetric weights and before and after drying the core using standard wetland methodology (Clarkson *et al*., 2004b). Each core was individually weighed after collection to the nearest 10 micrograms. Once transported to Waikato University, the samples were transferred to Al trays and dried at 105 °C over a period of two to three days, until there was no change in dry weight (no moisture remaining in the core). This was then reweighed to determine the dry weight, and dry bulk density was then determined using the volume of the core (589.05 cm^3).

4.2.5 Gravimetric and volumetric moisture contents

Gravimetric moisture content (% of mass in the peat core comprised of water) was calculated by dividing the mass of water in the sample (wet weight minus dry weight) by the dry weight. Volumetric moisture content (VMC) was calculated by multiplying the moisture content by the final dry bulk density (assuming that the density of water is 1.0 g cm^{-3})

4.2.6 Subsampling for analysis

The second peat core for each site was used for chemical analysis. This peat core was oven dried at 30 °C for 24 hours and then left to air dry for several days. Once the core was completely dried, it was evenly sliced into four sections to ensure each section had a representative amount of each horizontal layer of the peat core. These were then sectioned into smaller subsamples for use in different analysis (Figure 3.6).

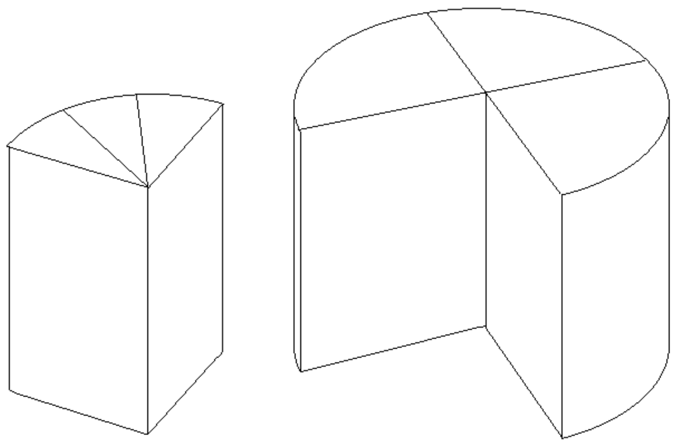

Figure 4.3. Peat core subsampling for chemical analysis.

4.2.7 Chemical analysis of carbon, nitrogen and $\delta^{15}N$

The analysis for the organic material (total carbon (TC), total nitrogen (TN) and $\delta^{15}N$) present in the samples required similar sample preparation techniques. The dried samples were ground to a fine powder in an agate and ball on a shaking machine, and stored in air-tight containers.

A small portion of these samples were weighed and analysed by an Elementar C&N analyser (through the University of Waikato laboratory). Percent C and N were adjusted using bulk density to calculate volumetric concentrations (mg cm^{-3}). $\delta^{15}N$ was analysed by a fully automated Europa Scientific 20/20 isotope analyser (University of Waikato Stable Isotope Lab).

4.2.8 Ash and organic content

Core subsamples were combusted (ashed) to remove organic matter and to determine the proportion of mineral components present. The sub samples were heated in a muffle furnace at 450 °C for 6 hours. This also prepared the peat samples for digestion to be used in the analysis of the mineral constituents via inductively coupled plasma mass spectrometry (ICP-MS).

4.2.9 Chemical analysis of metals and other elements

For the analysis of other elements in the peat sample, subsamples needed to be converted to an aqueous solution to be run through ICP-MS. Ashed subsamples weighing 0.25 g were added to a solution of 6 ml HNO_3 and 2 ml HCl (3:1 ratio) in 50 ml Falcon tubes, and were then left to passively digest over a 48 hour period. The solution was then heated on a hot block for one hour at a temperature of 80 °C, and left to cool for 30 minutes. Following cooling, 0.9 ml of concentrated H_2O_2 (30%) was added to the solution in 0.3 ml increments, with an hour of heating and 30 minutes of cooling in between each addition. Type one water (purified for laboratory use) was added to the solution until it reached 50 ml, and centrifuged at 4000 rpm for 10 minutes. 7.5 ml was filtered using a 0.45 micron filter into separate 50 ml Falcon tubes and then type one water was added for dilution to reduce the acid concentration below 2% for ICP-MS analysis (method adapted from Krachler *et al.*, 2002; Enders & Lehmann, 2012).

There was not complete dissolution of samples during peat digestion, as there were some mineral constituents that would not break down without use of dangerous acids (such as hydrofluoric acid which was avoided due to safety concerns). Any sample digested over the specified time frame was deemed as close to the total content in the substrate that is potentially available to plants; the portion that was unable to be digested is essentially bound and would stay that way under natural conditions.

4.2.10 Statistical analyses

Using standardised data obtained from peat analysis and plot condition (such as nutrients, inorganic elements, pH and von Post index), statistical analyses were undertaken using the software packages CANOCO (Windows V.4.5) and

STATISTICA (V.13.3). These analyses grouped sites together based on the commonality in these variables to determine clustering and then identify zones of different characteristics across the wetland.

The statistical program CANOCO used cluster analysis to identify the best clusters with the lowest stress value, from which a dendrogram (tree diagram) was created. This displayed the relationship between all 30 sites with Euclidean linkage distances. From this, six main groups were identified, as the ordination selected a cut-off value from which statistically valid groups were created.

A principal component analysis (PCA) ordination was also run. PCA uses multiple linear regression to interpolate the data variables into a 2-dimensional ordination space, with the environmental variables added over the ordination. This ordination was then compared to the clustered groups identified in the dendrogram, while nutrient patterns were reanalysed using cluster groups as bins for a boxplot.

4.3 Results

4.3.1 Peat physical characteristics

Peat samples were assessed for the physical characteristics of dry bulk density, von Post decomposition, volumetric moisture content, and mineral content. DBD and von Post measurements exhibited similar trends as physical degradation (higher von Post values) can be measured with decomposition and consolidation of the substrate (higher DBD values). Larger values were found close to the northern drainage ditch and upper central drain, while lower values were found in the south. While Transects B and E did not show much change between sites (maximum ranges for DBD 0.034 and 0.035 g/cm^{-3} respectively), Transects A, C and D all showed differences both between sites and between measurements. The upper central drain transect (A) showed variations on both sides of the drain, with the western edge having higher values of both DBD and von Post. The northern drain transect (C) had higher von Post measurements, while the northern swamp or native bush slope transect (D) showed a transition from high values to low values (largest range for DBD of 0.148 g/cm^{-3}) (Figure 4.4).

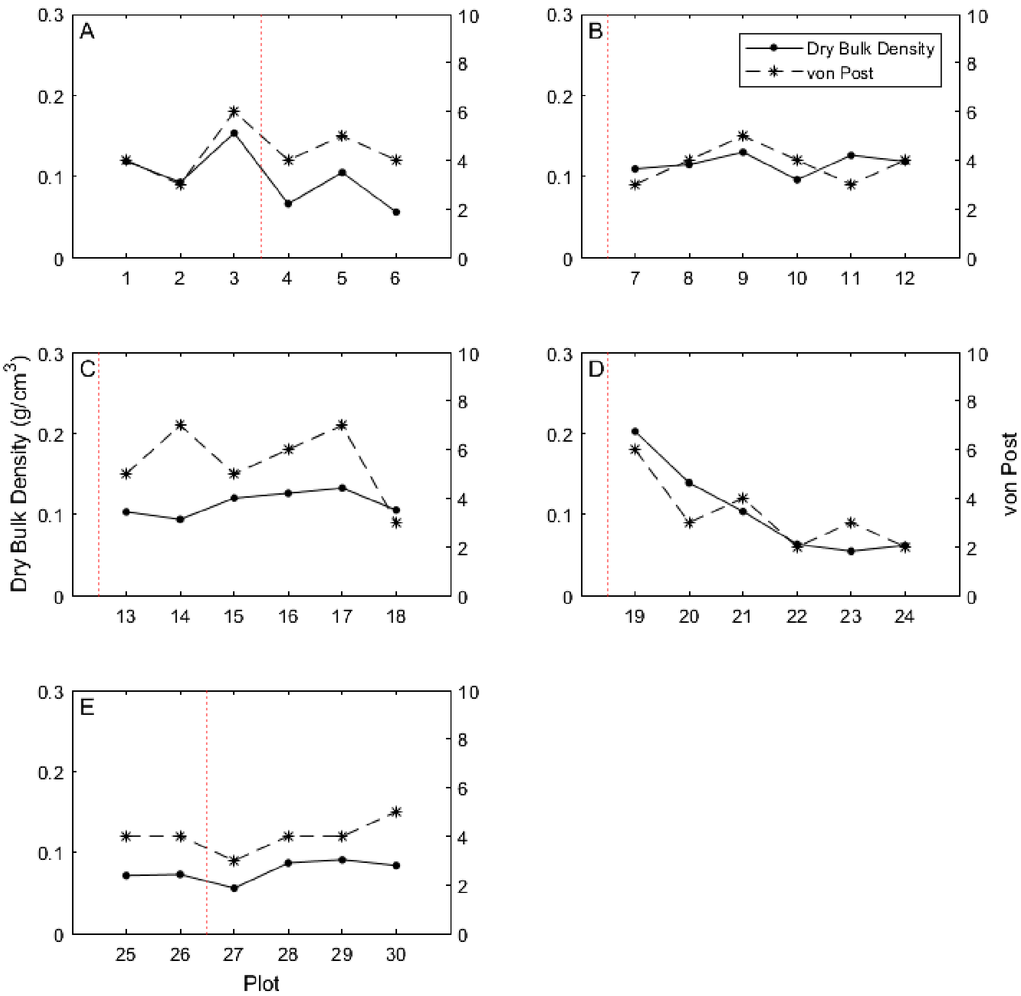

Figure 4.4. Dry bulk density of peat samples and corresponding von Post measurements plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

The VMC of peat samples generally decreased in areas of higher bulk density (with the exception of Plots 3 and 19), as greater mineral particles or increased decomposition of the peat reduces pore space for water. This was seen in the wetland as areas near drains had lower VMC (Figure 4.5). Transects A, C and E had the lowest ranges of 0.2, 0.35 and 0.29, while Transects B and D had larger ranges of 0.49 and 0.48.

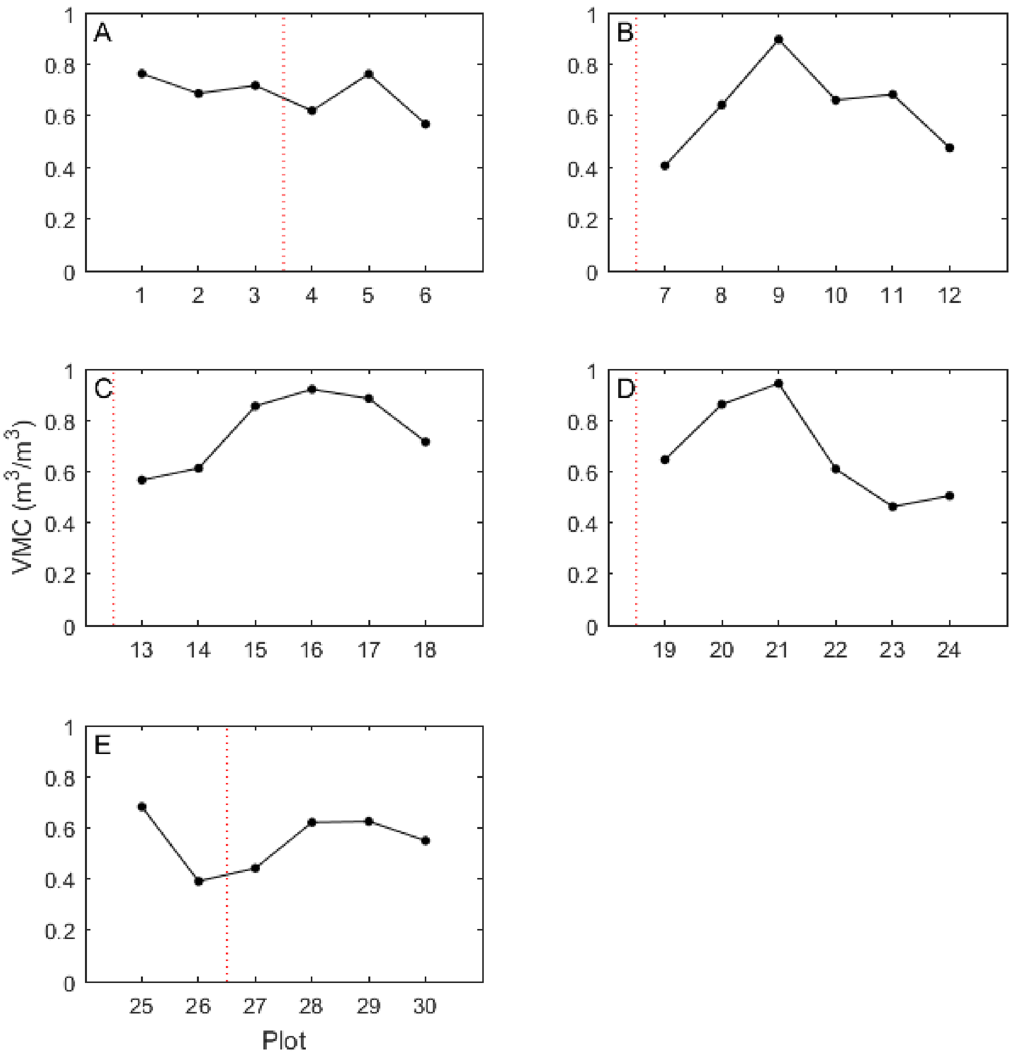

Figure 4.5. Volumetric moisture content of peat samples. Vertical lines indicate the relative position of the major influence to each transect.

Mineral content (% by weight) was highest near the northern and central drains (Figure 4.6). Site 19 experienced the highest mineral content (57.9%) while the sites west of the central drain on Transect A (sites 2 and 3) and most of Transect C (sites 13-16, 18) all showed high mineral contents.

The lowest mineral abundance was measured at site 24 (5.53 %), while Transect B and E also showed lower mineral content (Figure 4.6). Transect D showed a strong influence of disturbance with a high mineral content transitioning to low mineral content over the length of the transect.

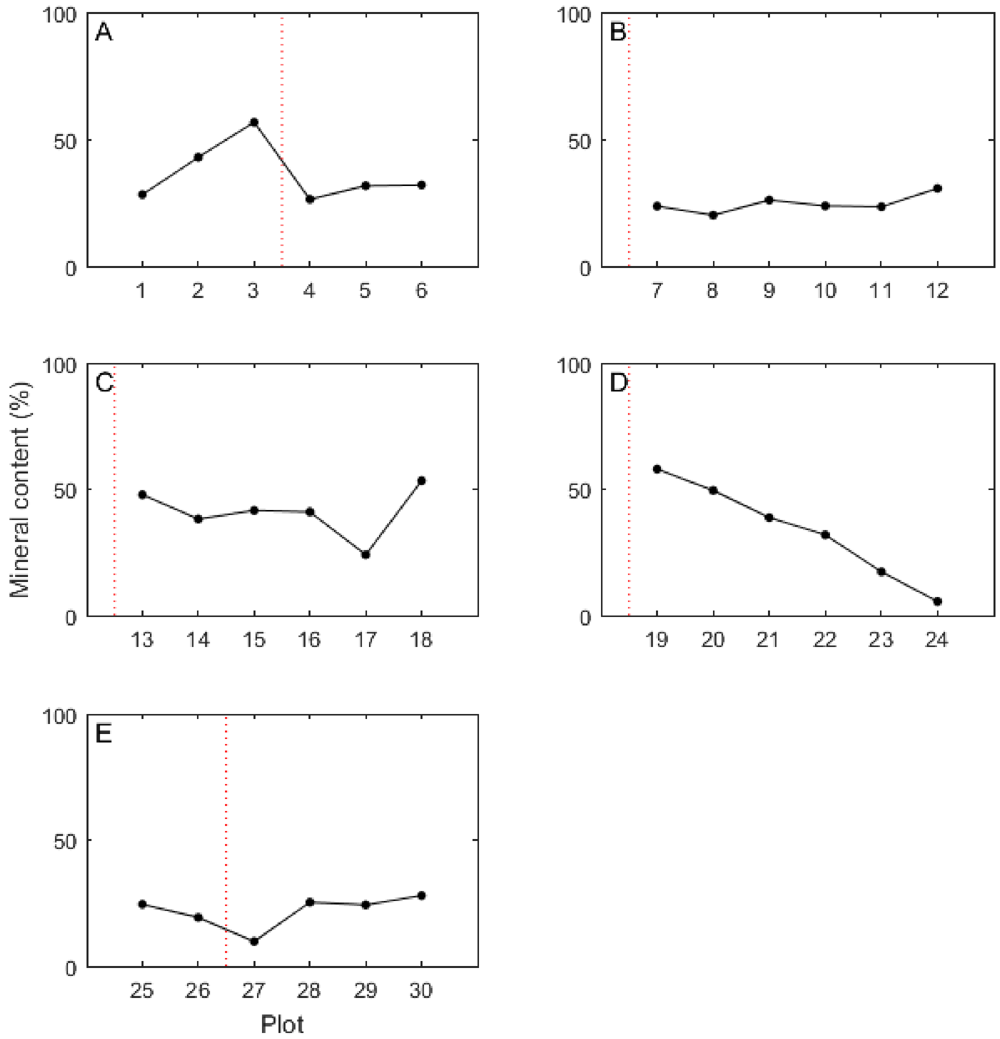

Figure 4.6. Mineral content by mass (%) present in peat samples plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

4.3.2 pH and electrical conductivity

Peat water pH and electrical conductivity (EC) changed significantly across the wetland. There was little change in EC along the southern transects, ranging from 72.5 µS to 86.2 µS, while the northern transects exhibited higher values up to 200.1 µS. Transect D (Plots 19-24) showed the greatest range, with a strong gradient from the highest to the lowest EC values (200.1 and 71.6 µS). pH followed similar patterns, with the southern plots generally having lower values (4.04 at Plot 30 to 4.41 at Plot 11), while the northern plots generally had elevated pH values (highest value of 5.52 located at Plot 13). The lowest pH of 3.82 was found at Plot 24 (Figure 4.7).

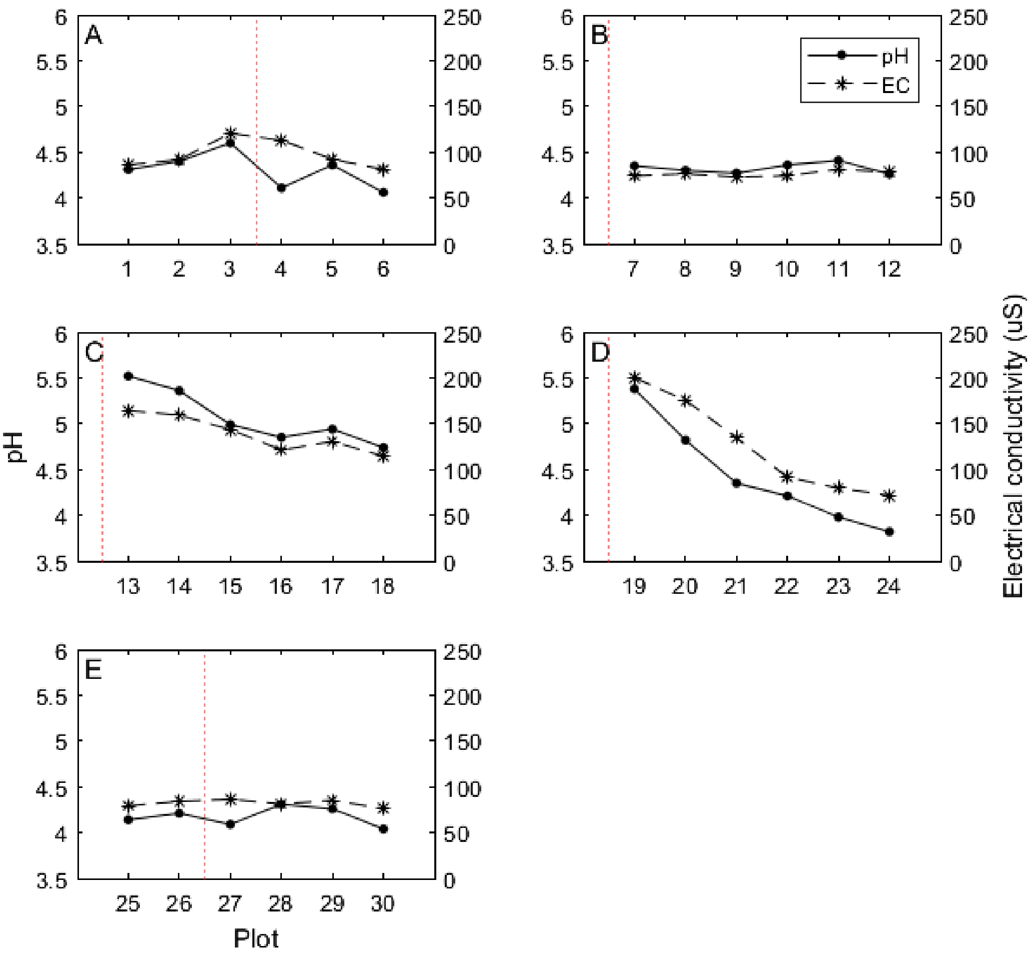

Figure 4.7. pH and electrical conductivity measurements plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

4.3.3 Carbon, Nitrogen and Phosphorus

Total gravimetric C (expressed as % by dry weight) ranged from a maximum of 44.21% at Plot 24 to a minimum of 18.74 % at Plot 19 (Figure 4.8). Total volumetric C (measured as mg cm^3) ranged from a maximum of 52.39 mg cm^3 at Plot 9 and to a minimum of 17.50 mg cm^3 at Plot 6 (Figure 4.8). These C values are lower than what was found in other New Zealand wetlands, with Clarkson *et al*. (2004) finding C values for swamps and bogs averaging 39.8 and 92.7 mg cm^{-3} and ranging from 5.2 to 239.8 mg cm^{-3} overall. Transect D shows a contrasting pattern between total gravimetric and volumetric C, with low gravimetric values giving way to higher values as volumetric drops (Figure 4.8).

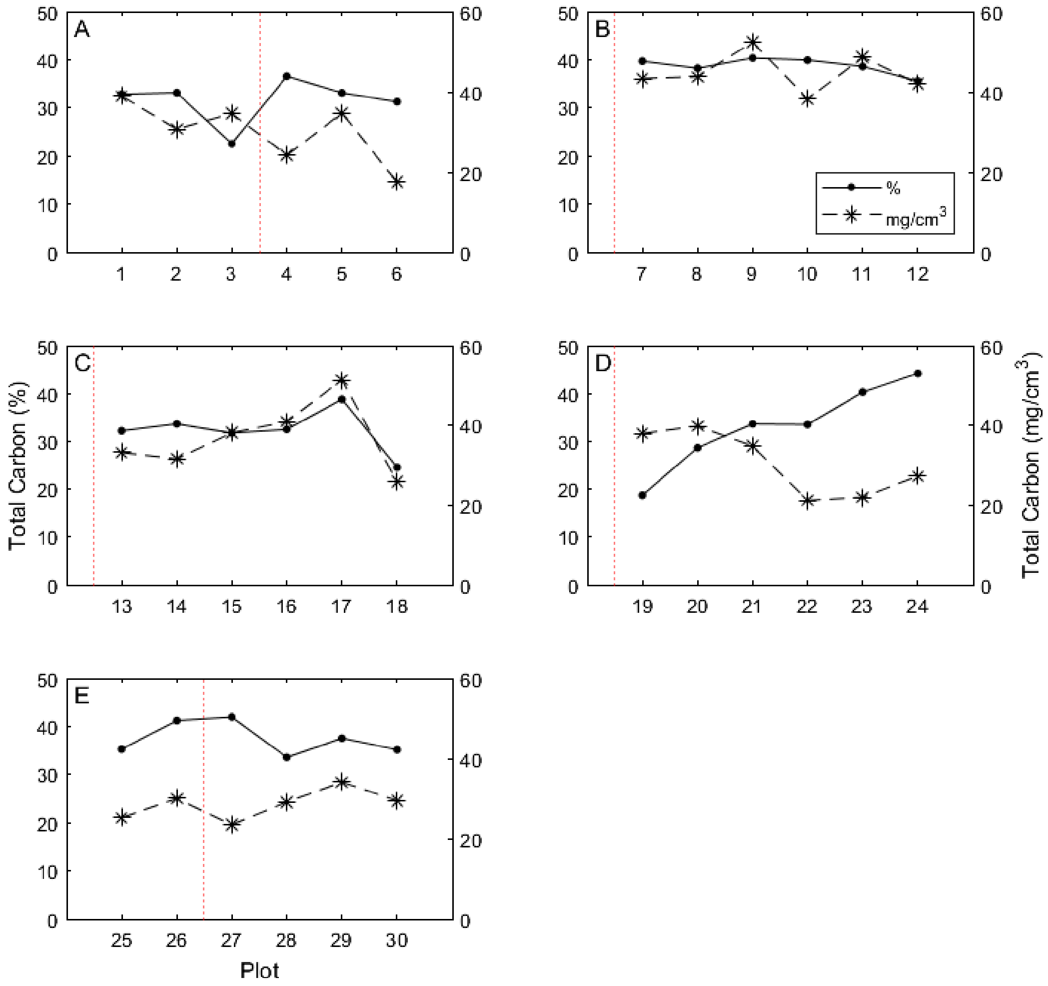

Figure 4.8. Total carbon content in peat samples expressed on gravimetric and volumetric basis, plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

TN of peat at Otakairangi varied significantly based on the different areas being measured. While most areas showed consistent values that sat within the swamp-bog range (between 1 to 2 mg cm^3), Transect D, which extended from a swamp marginal area to restiad interior, showed variation in concentrations (2.13 mg cm^3 range). Nitrogen was highest at site 19 (2.64 mg cm^3) and transitioned to the lowest value at site 24 (0.51 mg cm^3) (Figure 4.9).

Total P concentrations generally showed similar trends to TN, with the Transects B, C and E expressing similar P concentrations. However, the effects of flood inputs were shown in the values of Transects A and D in the northern area, with highest P concentrations near the central drain and swamp area. The maximum value occurred at site 19 (0.194 mg cm^3), while the minimum value was at site 23 (0.006 mg cm^3) (Figure 4.9).

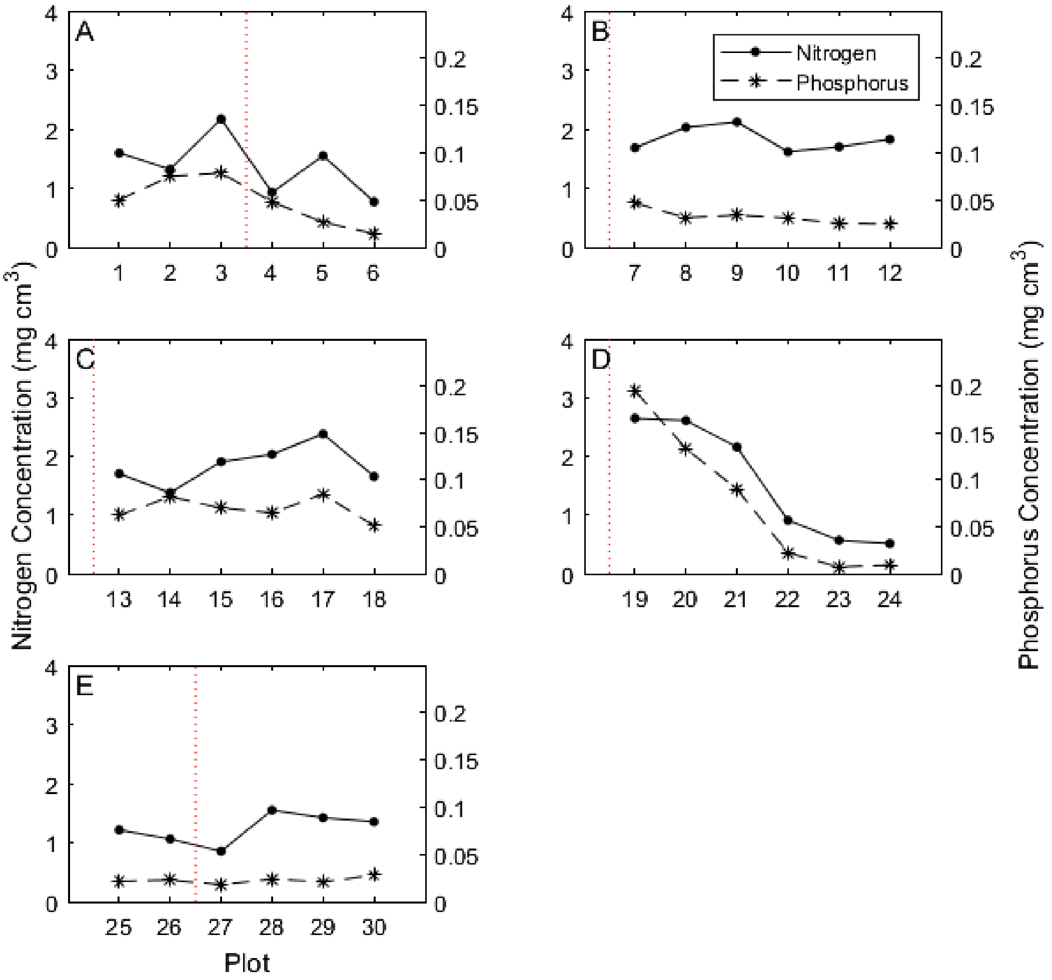

Figure 4.9. Total nitrogen and phosphorus content plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

The C:N ratio for the majority of the plots fell around the New Zealand average swamp values (Clarkson *et al.*, 2004b), with the restiad section at the end of Transect D showing an increase from low C:N ratios up to higher bog-like values. The lowest occurred at site 19 (14.33), while the maximum C:N ratio occurred at site 24 (53.06) (Figure 4.10).

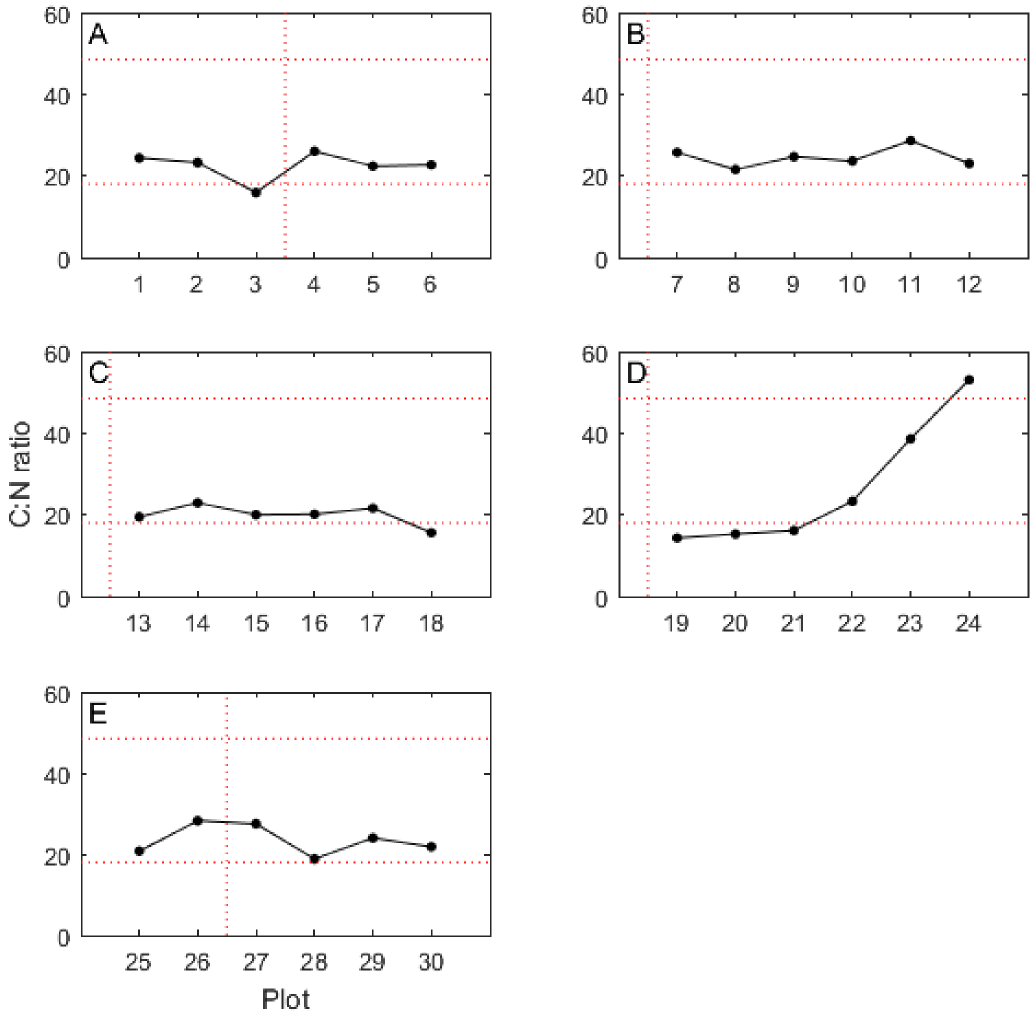

Figure 4.10. Carbon to nitrogen ratio of peat samples plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect. Horizontal lines indicate the average New Zealand swamp C:N ratio (18) and average bog C: N ratio (48.5) (Clarkson *et al*., 2004b).

The nutrients N and P are both used in plant growth, and ratios between them (and C) can be indicative of whether the systems are being limited by these nutrients. The highest N:P and C:P ratios were found at site 23, located at the end of Transect D (Plots 23 and 24), while the lowest ratios were found at site 19 in the swamp area at the start of this transect (Figure 4.11).

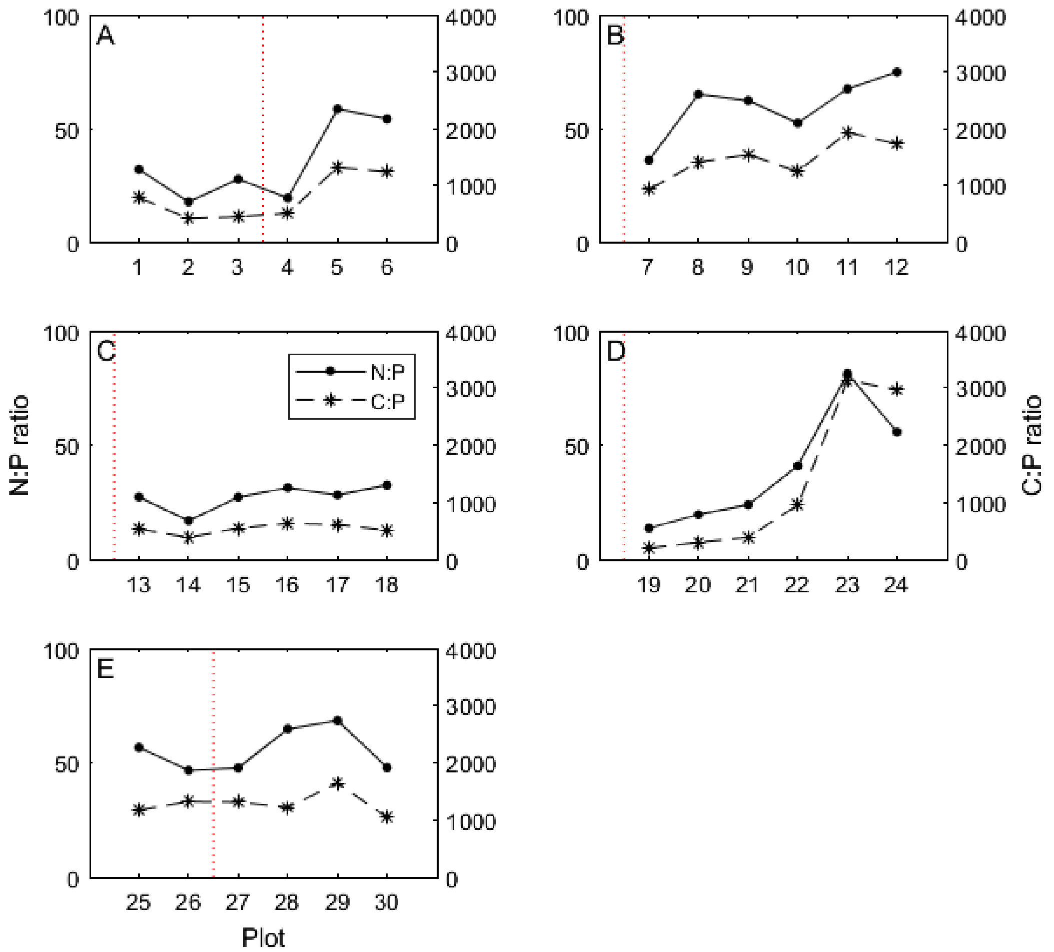

Figure 4.11. Nitrogen to phosphorus and carbon to phosphorus ratios plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

4.3.4 Peat Nitrogen 15 Isotope

$\delta^{15}N$ was generally enriched relative to atmospheric concentrations, with a noticeable increase in abundance in the swamp area of Transect D, peaking at 6.97 ‰. However, Transect B hovered around 0 ‰ with a depleted minimum value of -0.15 ‰. Transect D showed a gradient in $\delta^{15}N$ values, with enrichment in the first plots (19 and 20) and transitioning almost to depletion at the other end (Plots 23 and 4) (Figure 4.12).

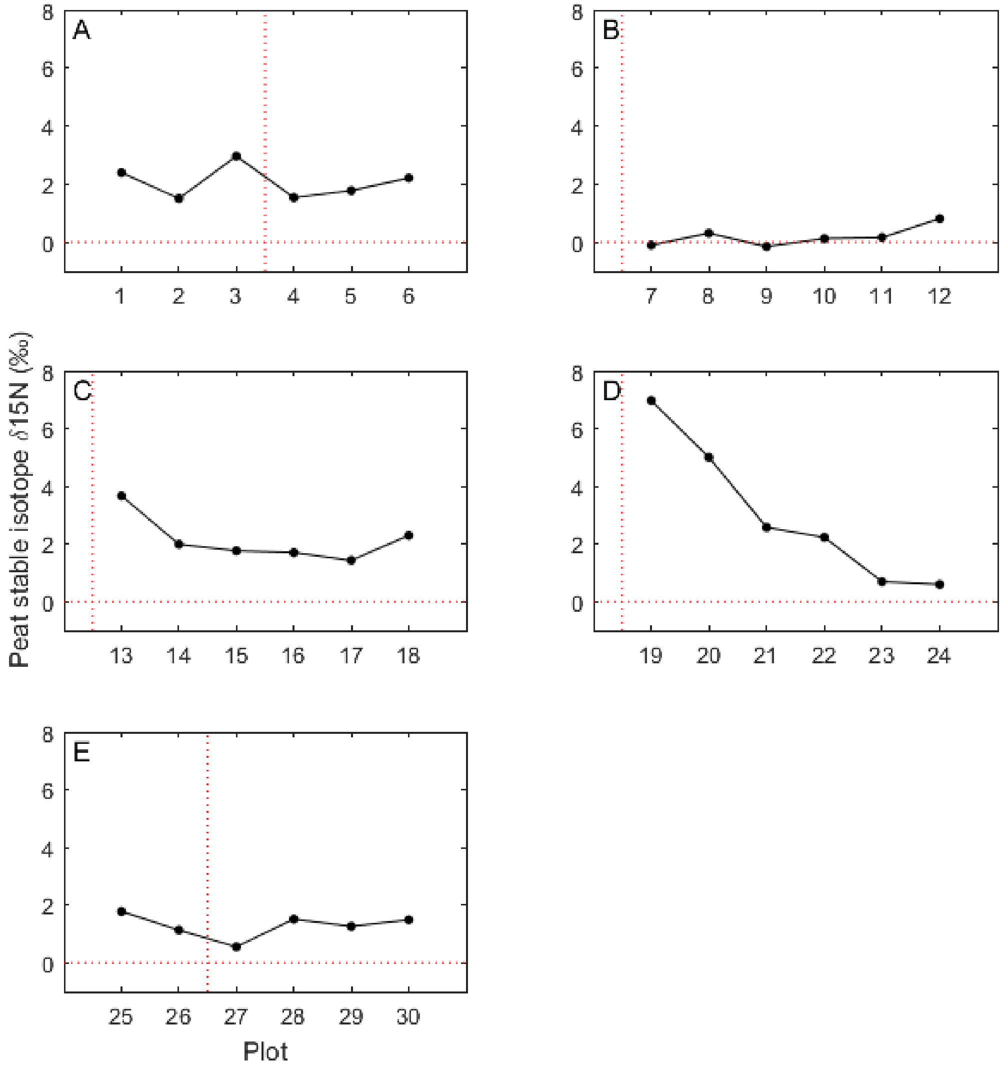

Figure 4.12. Nitrogen 15 isotope in peat samples plotted by transect (A-E) and plot number. Horizontal lines indicate atmospheric levels (Kramer *et al.*, 2003), with points below indicating depletion and points above indicating enrichment. Vertical lines indicate the relative position of the assumed influence along each transect.

4.3.5 Inorganic material

Concentrations of U were higher around the northern central drain, peaking at 0.103 µg cm^3 (Plot 19), a 100 fold increase compared to the lowest concentration of 0.001 µg cm^3 (Plot 24), which lay at opposite ends of the transect line. The southern transects exhibited lower concentrations, averaging around 0.01 µg cm^3 (Figure 4.13).

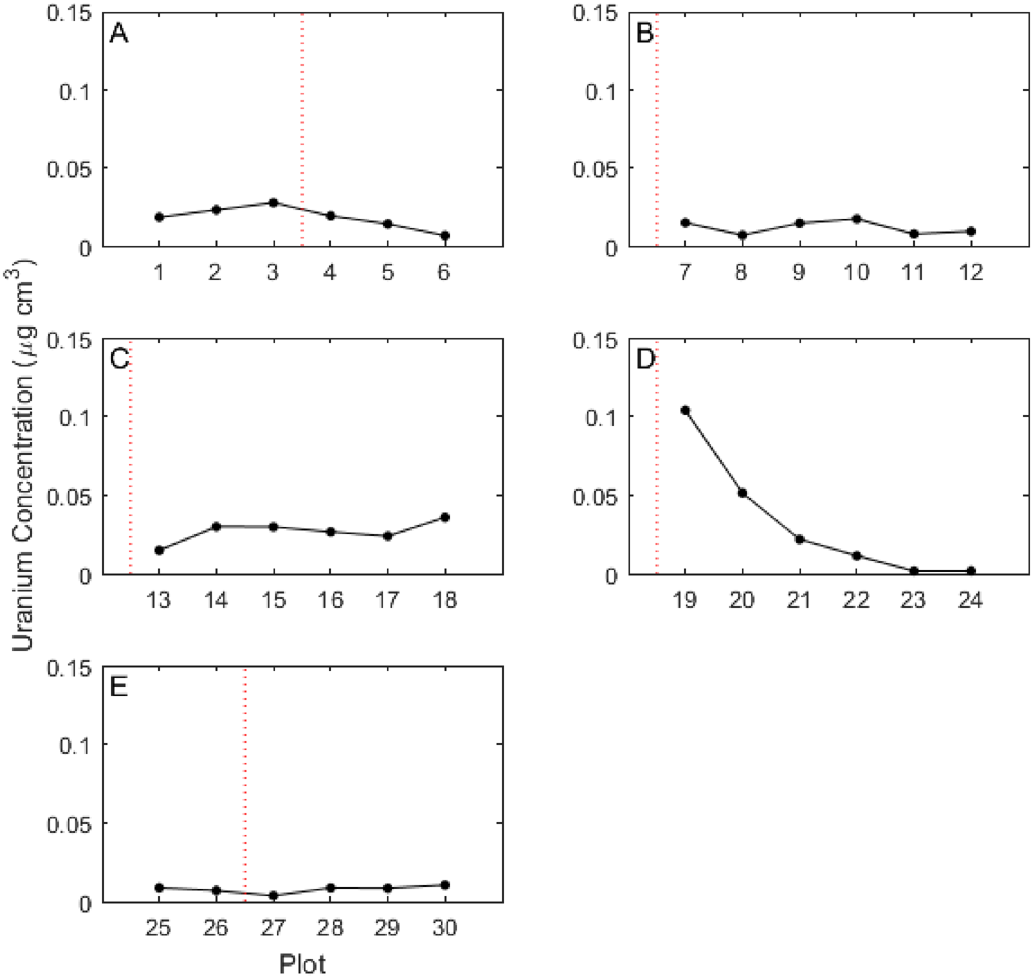

Figure 4.13. Peat uranium concentration plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

The concentrations of other heavy metals and nutrients (potassium (K), calcium (Ca), copper (Cu), zinc (Zn), aluminium (Al), and iron (Fe)) showed similar patterns to the other nutrients, with some exceptions. The northern central drain (A) showed higher concentrations of all elements (except for Fe) on the western side of the drain, decreasing with distance from the drain (Figures 4.14, 4.15 & 4.16. The southern drain transect (B) showed little variation with the exception of slightly elevated concentrations nearest to the drain. However, Fe showed two spikes, once near the drain (Plot 7) and once again further along the transect (Plot 11) (Figure 4.16). The northern drain transect (C) showed increasing concentrations of most elements with increased distance from the drain, with a notable exception of Ca which had extremely high values for the central four sites (Plots 14-17) (Figure 4.14). The northern swamp transect (D) extending from the native bush slope showed a strong trend in all recorded elements with concentrations decreasing with distance from the disturbance. The transect crossing the central drain in the southern section (E) showed low concentrations of all values, and little variation between sites, with the exception of slightly elevated Fe and Ca concentrations near the drain (Figures 4.14 & 4.16).

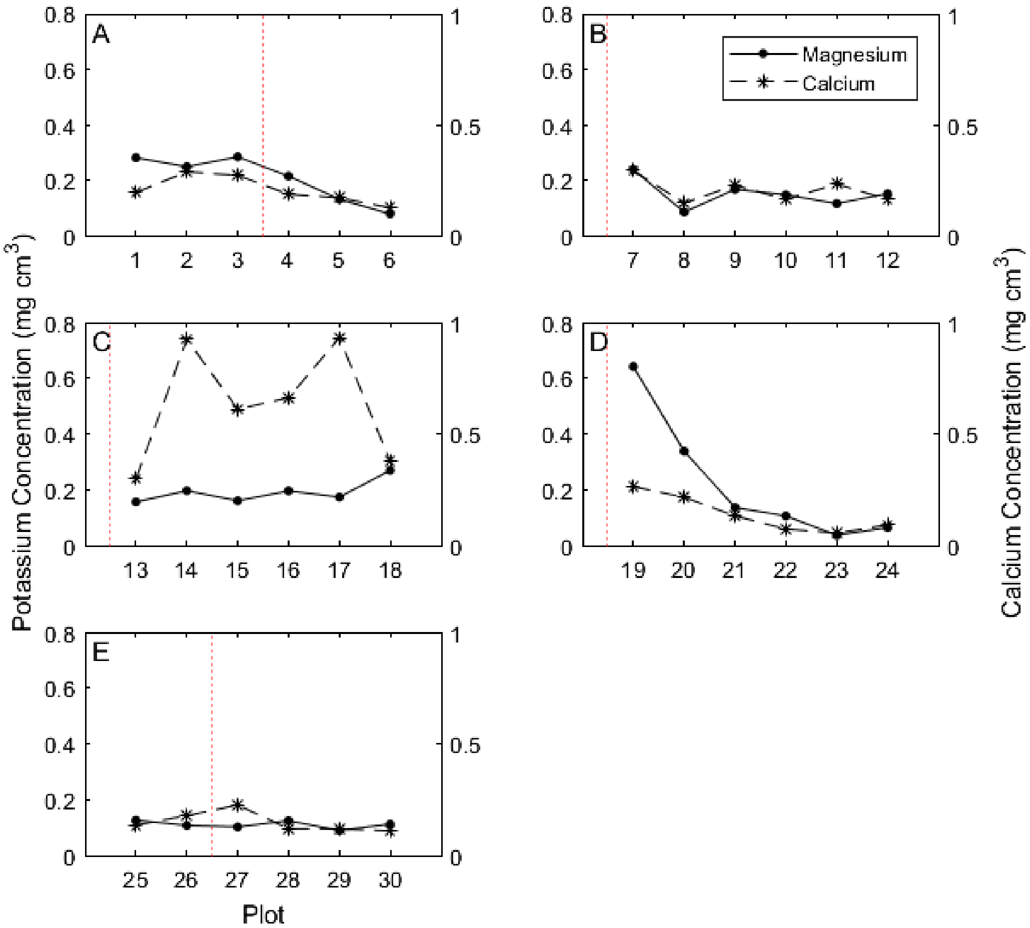

Figure 4.14. Peat potassium and calcium concentrations plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

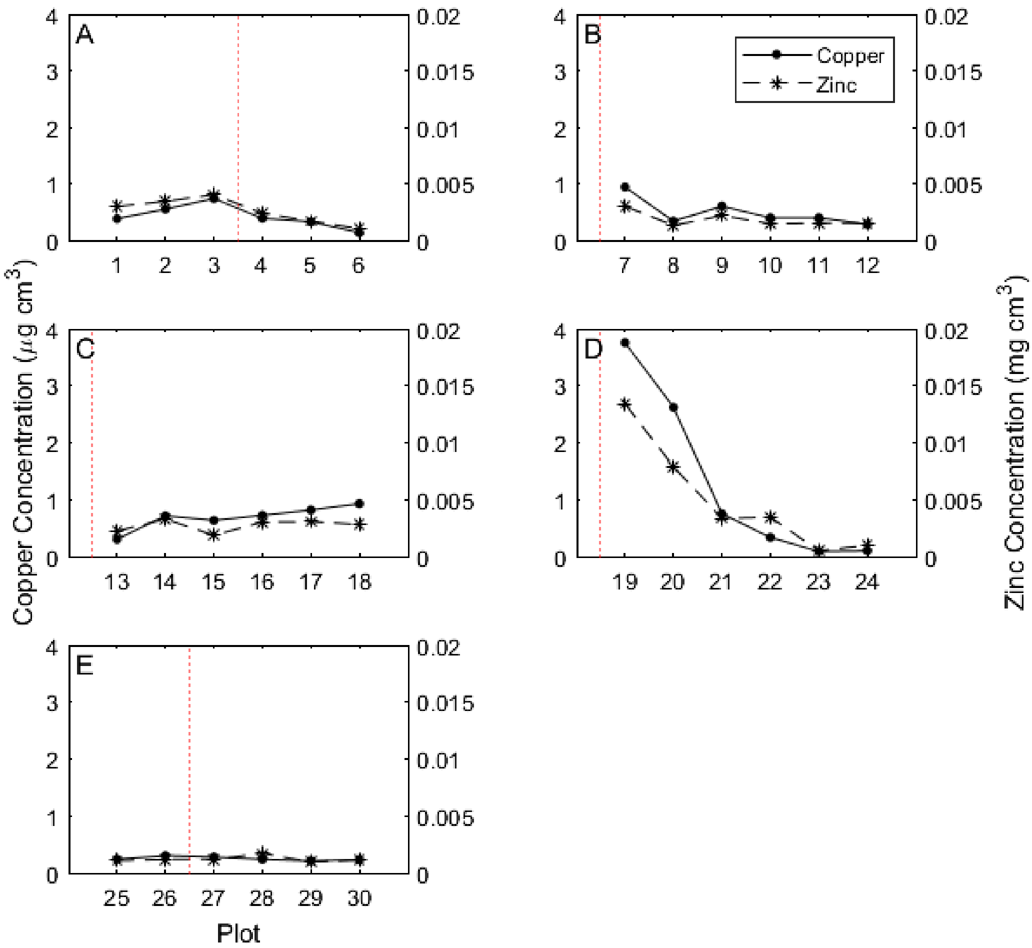

Figure 4.15. Peat copper and zinc concentrations plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

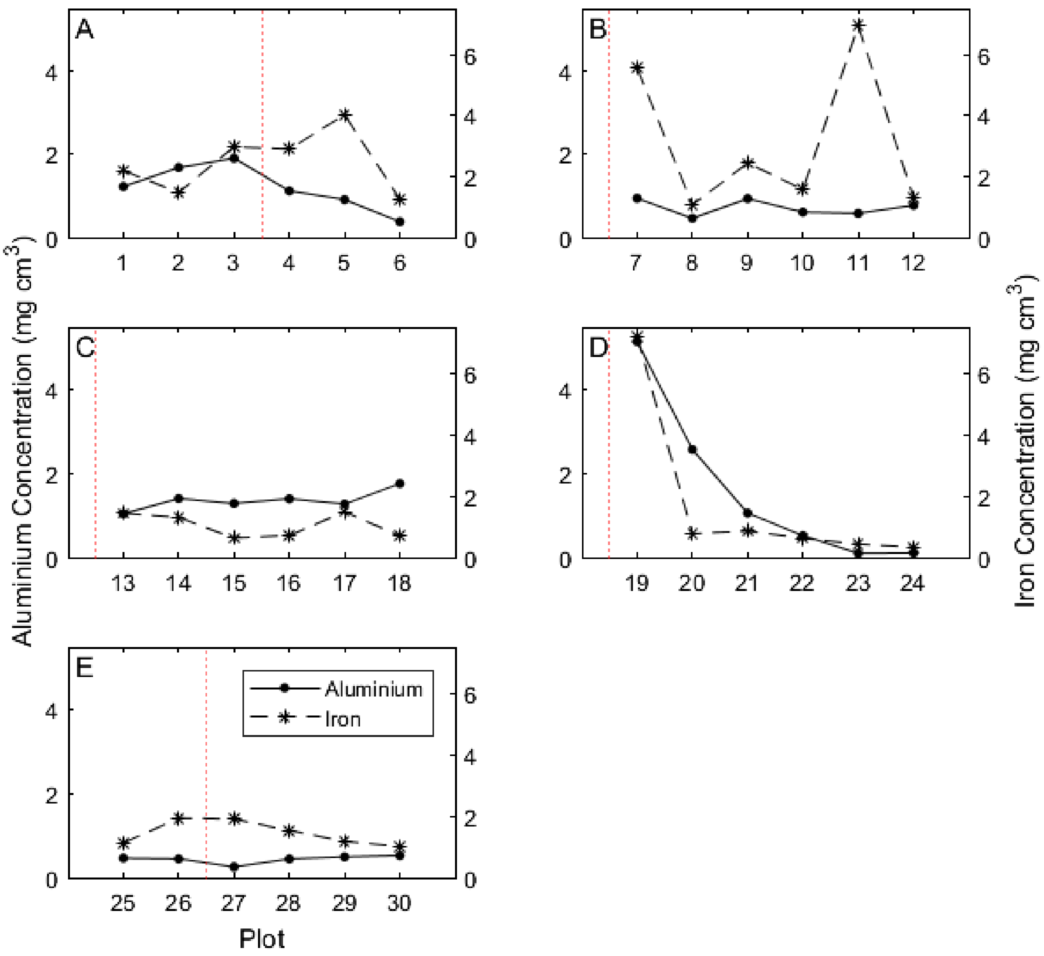

Figure 4.16. Peat aluminium and iron concentrations plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

4.3.6 Ordination statistics

Statistical ordinations were created to determine group relationship and clustering of sites based on peat physical and chemical values. The outputs were a dendrogram to identify clusters and a PCA ordination to express the relationship between them. Six primary groups were identified by the dendrogram (Figure 4.17), with a clear transition from highly mineralised, fertile peat through to low fertility, presumably rain-fed sites. These groupings corresponded to field observations, with the higher fertility sites having high decomposition ratings, while the low fertility sites were less degraded.

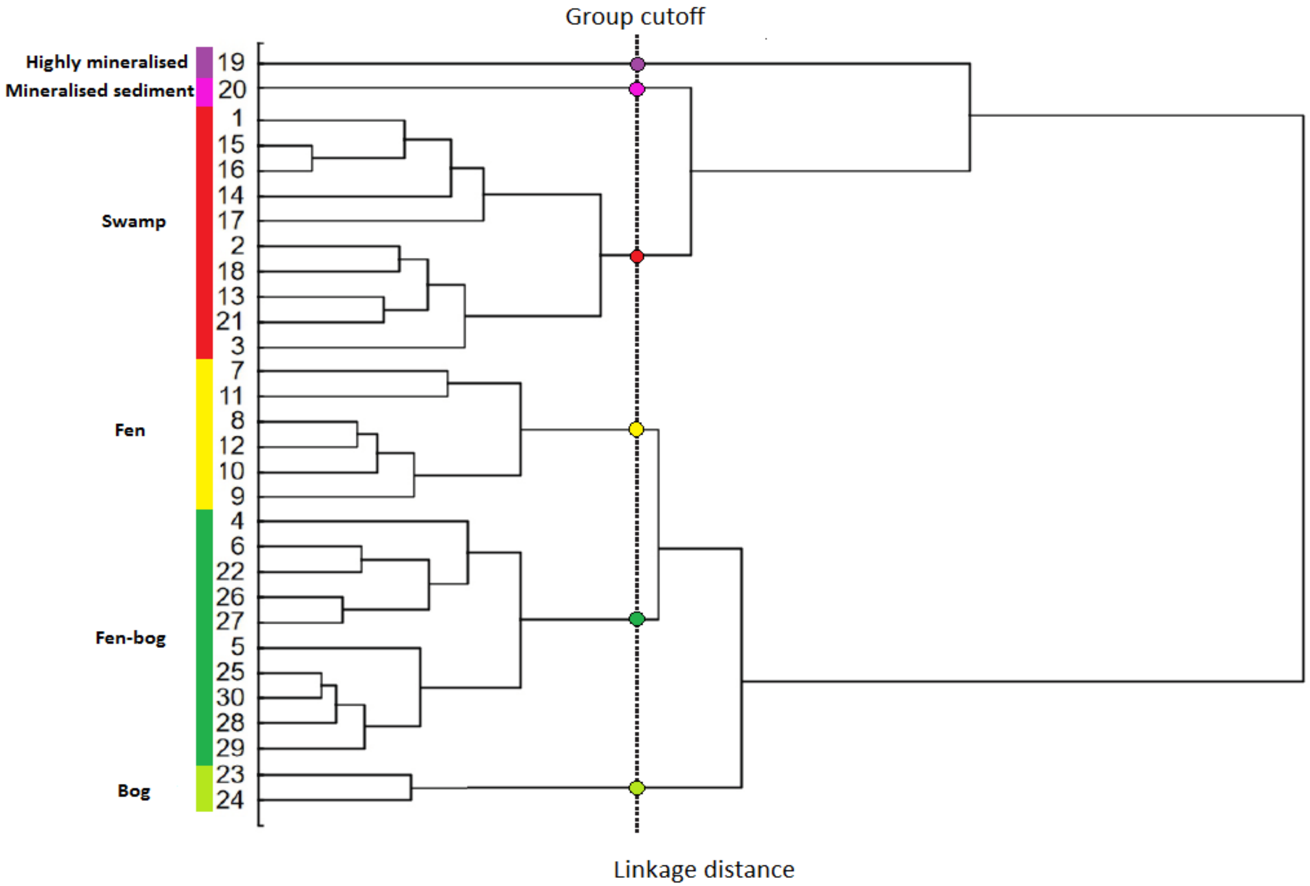

Figure 4.17. Tree dendrogram showing the relationships between the six major clusters derived through cluster analysis, with the group cut-off marked by the vertical line. Final groups were labelled bog, fen-bog transitional, fen, swamp, mineralised sediment, and highly mineralised sediment, based on values from Clarkson *et al*. (2004b) and terminology from Hodges & Rapson (2010).

The normalised data for all 30 sites were also plotted into an empty space, with the axis having no units or meaning (PCA axes 1 and 2, values -1 to 1) (Figure 4.18). The closer the sites were positioned, the more closely they resembled each other; vice versa, if they were positioned far apart they were dissimilar. Tight groupings of sites indicate that they are similar, and are most likely influenced by the same factors. For example, the swamp sites identified in Figure 4.17 were clustered together on one side of the ordination, while bog and fen-bog sites were clustered together on the opposite side.

The broad range of directions taken by the PCA vectors indicate that there is a wide range of relationships between sites, due to a wide range in different disturbance and environmental factors. Nutrient ratios (high ratios indicate good wetland plot condition) were directed to the left, while dry bulk density (DBD) was directed to the upper right (higher DBD indicates mineralisation and decomposition). The resultant fertility arrows are therefore a generalised indication only, as the x axis (PCA 1) explains four times the variance in data compared to the y-axis (PCA 2), as well as the key nutrient ratios and mineral content vectors tending to follow the x-axis direction.

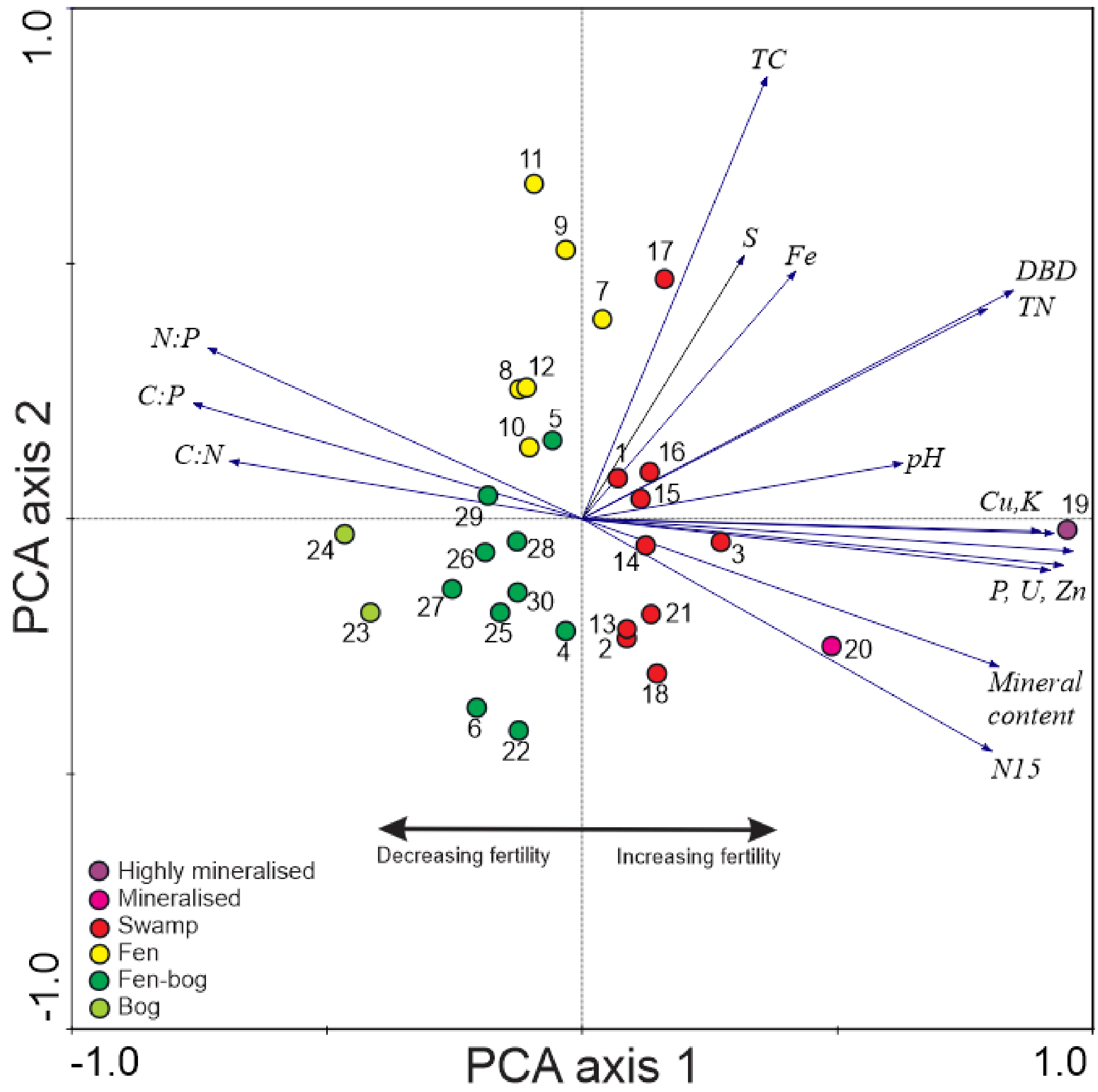

Figure 4.18. Ordination plot showing the relationships between individual sites. The figure shows a trend of increasing fertility from left to right, while the six major clusters derived from Figure 4.17 are marked. The x axis explains 60% of the variance, while the y axis explains 14%, showing that variation along the x axis has four times the weighting of the y axis. Vectors show the correlation for the relevant environmental variables to the plot positions, with closer vectors indicating high correlation and vectors at opposite angles show negative correlation. Vectors at 90 degrees to each other show no correlation (such as TC and N15).

Boxplots of the nutrient data binned by the ordination groups identified in Figure 4.17 shows a clear gradient between groups, with the mineralised and swamp plots having higher concentrations of all nutrients, but resultant lower ratios when compared to the bog and fen sites (Figure 4.19).

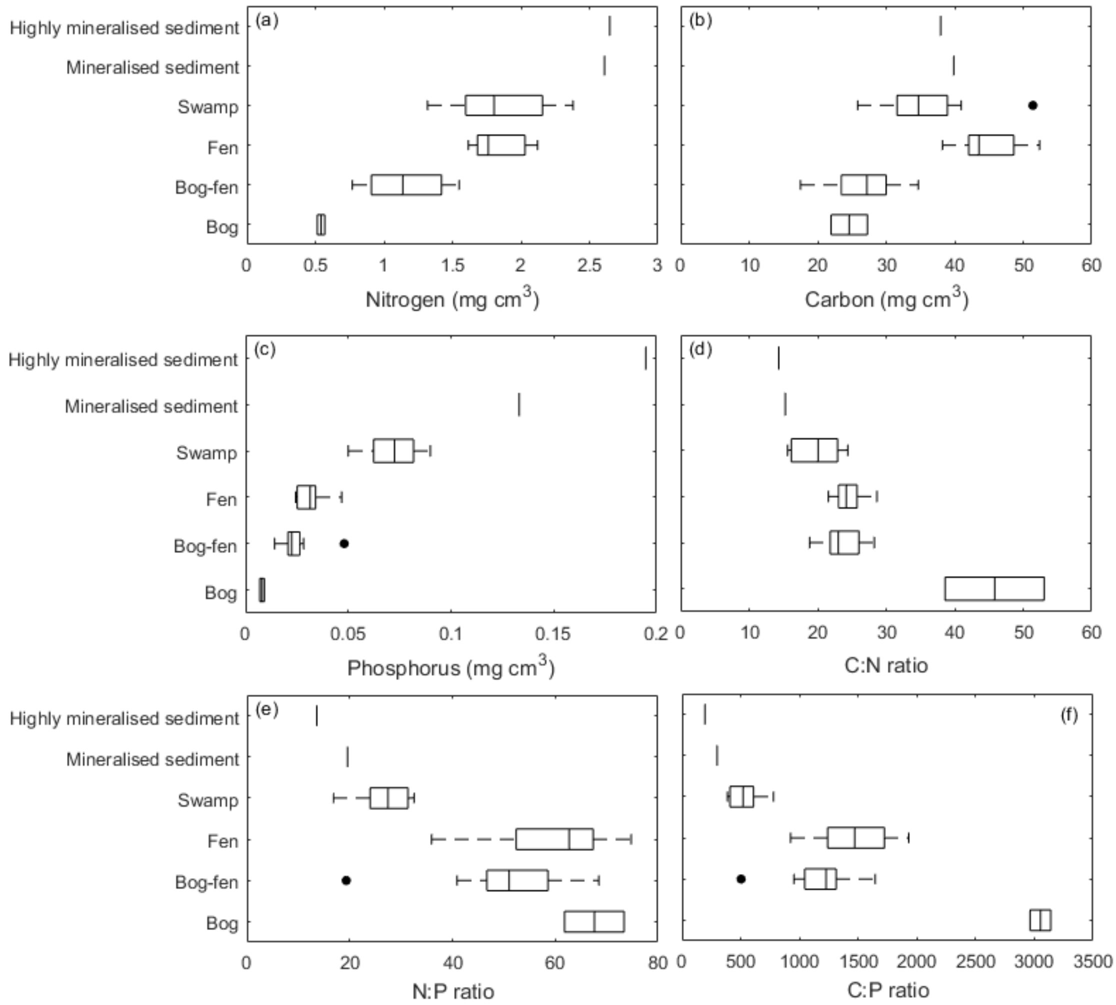

Figure 4.19. Box plot summaries of peat nutrients and nutrient ratios binned by cluster analysis groupings. Box plots show the median, upper and lower quartiles, and outliers. Bog (n) = 2, fen-bog (n) = 10, fen (n) = 6, swamp (n) = 10, mineralised sediment (n) = 1 and highly mineralised sediment (n) = 1 (where n = number of plots in each grouping).

4.4 Discussion

From the peat samples collected from the five transect lines across Otakairangi wetland, simple trends were revealed. Areas in the northern wetland, particularly near the central drain and northern border drain, exhibit higher nutrient and inorganic element concentration, as well as higher degradation implying frequent inundation. Other areas, such as the southern section and inner wetland zones have lower nutrients, higher nutrient ratios and less degraded peat, indicating primarily rain-fed systems.

4.4.1 Physical characteristics

The surficial substrate varied across the entire wetland, from what is considered pristine, newly formed peat through to mineralised mud and soil profiles. The primary peat former in New Zealand is *Empodisma spp.*, which forms light, fibrous peat from its complex root structure, which rates between 1 and 3 (Clarkson *et al*., 2004a). This was seen in scattered areas across the wetland where the species was found, such as the large patch in the north-eastern section, or in the smaller patches in the south-west. However, the majority of the peat analysed was in areas dominated by *G. dicarpa*, and as such the bulk of the peat was influenced in some way by this species. This peat consists of *G. dicarpa* rhizomes, and rated around 4 on the von Post scale. Other areas, such as the northern section of the wetland or below the native bush hill slope, experienced high inputs of mineralised sediments and nutrients, and as such the substrate transitioned to the point where it is may not be considered peat anymore, but rather a type of soil.

Overall, the peat in Otakairangi wetland showed repair or formation of new peat in most areas depending on the surficial vegetation, all of which was overlying degraded peat of varying severity. An example is Plot 7 near the southern drain, where the overlying peat was a 3 on the Von Post Scale (very weakly decomposed) (Figure 4.4), above a layer which was rated a 9 (almost completely decomposed). The majority of the overlying peat material (which was sampled in 4.2.1) was in the range of 3 to 6 on the von Post scale, with some exceptions above or below. Higher bulk density recorded in peat generally indicates stronger decomposition or greater mineral concentrations. The dry bulk density matched the overall patterns seen in the von Post measurements, due to higher von Post measurements indicating reduced pore space size and quantity, compacting the structure which results in increased bulk density. Volumetric moisture content of the peat was variable across the transect lines, as this correlates both to the bulk density and the level of degradation. The different peat forming species, the resultant peat structures, as well as the relative degradation of the site sampled all affect the moisture content (Charman, 2002).

Transect A which crossed the central drain near the centre of the wetland showed increased degradation of peat closer to the drain, with peat showing improvement in plant structure and water clarity with increased distance from the drain. The increase in bulk density and VMC was likely due to flood events (Charman, 2002). Transect B, which extended from the south-western drain, showed a slight undulating pattern, with peat structure generally increasing with distance from the drain. Transect C, which originated near the northern drainage ditch, showed strong decomposition of surficial peat (von Post values between 5 and 7) for the majority of sites, with the final site positioned deep in the wetland showing a far lower value of 3. This transect showed differences with the von Post measurements being higher as the substrate was strongly affected by muddy mineralised sediments (Figure 4.4). Transect D, extending from the edge of the more swamp like mineral soils near the base of the native bush, showed a strong improvement in peat structure, with the von Post values decreasing from 6 down to 2 as it extended into the wetland. Transect E, which crossed the central drain in the southern section, showed an opposing trend to Transect A, with sites close to the drain having lower von Post and DBD values.

Wetland classes are often used to examine and compare sites, with restiad bogs often characterized as being older, nutrient poor, and having low pH and EC values (Sorrell & Gerbeaux, 2004). By comparison, swamps are generally younger, with nutrients and sediment inputs from surface waters that can lead to higher pH and EC measurements (Campbell & Jackson, 2004). This was shown by Clarkson *et al*. (2004b), who utilized field pH in conjunction with other variables to help classify New Zealand wetlands. The restiad bogs measured in this study averaged 4.0 for pH (ranging between 3.7 and 4.4), while swamps averaged 5.2 (ranging between 4.1 and 5.9). Field measurements of pH and electrical conductivity (EC) (Figure 4.7) indicates that changes to peat in Otakairangi wetland can be associated with the hydrological regime. A significant proportion of Otakairangi field plots fell within the range for a restiad bog, with the southern plots and some of the north-eastern section having pH ranging from 3.8 to 4.4, and averaging 4.1, slightly higher than what was found in Clarkson *et al*. (2004b). The northern section, which was primarily composed of swampland (based on pH, nutrient concentrations, and ratios) showed similar, although slightly lower swamp values, ranging from 4.3 to

5.5 and averaging 4.9. EC was higher around the northern border drain and upper central drainage channels (Figure 4.7), where high amounts of sediment and nutrients enter the wetland during flooding events. An increase in the abundance of ions associated with the sediment leads to a high electrical conductivity (Walter *et al.*, 2015).

4.4.2 Carbon

While C is an important aspect in characterising a wetland (such as in nutrient ratios), the use of TC by itself is not particularly useful. This is due to issues between swamp and bog values, and the relative overlap between them. Clarkson *et al*. (2004b) found bog ranged from 24.1 to 239. 8 mg cm^3, while swamps ranged from 5.2 to 100.6 mg cm^3 C. While the values obtained in Otakairangi are within these ranges, the values for bogs and swamps have a large overlap, meaning that they cannot be used alone for characterisation purposes. The conversion of gravimetric C (percentage of peat substance composed of C) to volumetric (weight of C by volume, mg cm^3) utilises bulk density, so while peat in a restiad bog may have high gravimetric C, due to low bulk density from low mineral abundance or reduced consolidation, it will have lower volumetric C. This was shown in Transect D, where in the swamp area (sites 19-20) there was low gravimetric C (20-30% C), but due to high dry bulk density, the gravimetric C was high (~40 mg/cm^3) (Figure 4.8). At the other end of the transect (sites 22-24), the gravimetric C was high (35-45%) but due to low dry bulk density the volumetric C was low (20-30 mg/cm^3).

Bridgham *et al.* (1998) studied 16 different wetlands in Minnesota and determined the effect of bulk density on volumetric C mass by investigating mineralisation rates in relation to nutrients. Due to the distinction between nutrient turnover and availability, it was determined that bulk density alone explained 75% to 83% of the variation in C mineralisation. Due to the low bulk densities, bogs and acidic fens have a low mineralisation rate (high turnover rate in the acrotelm, but low nutrient concentrations).

4.4.3 Nitrogen and phosphorus

Nitrogen and P are key nutrients that are required for plant growth, and therefore influence the growth rates of vegetation.

Total nitrogen in Otakairangi varied significantly across the different areas being sampled. While many of the sites fell into the range of New Zealand bogs (0.02-1.83, average of 0.82 mg cm^3), they more closely resembled values found in New Zealand swamps (1.15-3.24, average of 2.12 mg cm^3) (Clarkson *et al*., 2004b). Transect A showed significant differences between sites, with higher N on the west side of the central drain (ranging from 1.32 to 2.17 mg cm^3) and lower N concentrations on the east side (0.77 to 1.55 mg cm^3) (Figure 4.9). The N levels generally increased with distance south of the northern drain on Transect C, but were more stable than Transect A. Transects B and E showed low variability in N, although Transect B (1.83 mg cm^3 average) experienced significantly higher values than Transect E (1.24 mg cm^3 average). Transect D showed a strong trend with the swamp area having high N (maximum value of 2.64 mg cm^3), and the inner wetland area having far lower values (minimum 0.51 mg cm^3).

Total P concentrations showed similar trends to N, with Transects B, C and E exhibiting P concentrations (averages 0.032, 0.069, and 0.022 mg cm^3 respectively) (Figure 4.9). However, the effects of flood inputs were shown in the values of Transects A and D in the northern area, with increases in P. Transect A saw higher P values on the west side of the central drain (averaging 0.068 mg cm^3), and lower on the east side (average 0.030 mg cm^3) similar to what was seen for N (implying more frequent inundation from the drain), while Transect D saw a large decrease from site 19 to 24 (0.009-0.195 mg cm^3 range). The two southern transects showed lower P concentrations, indicating that external inputs of nutrients to the wetland are less frequent or less intense compared to the northern areas.

4.4.4 Nutrient ratios

The availability of N in a soil or peat system is often expressed using C:N ratios, as the C:N ratio declines when N accumulates or C decreases. The C:N ratios found across Otakairangi were consistent with those found across New Zealand by Clarkson *et al*. (2004b), with restiad bogs averaging 48.5 (35.9–79.7), and swamps 18 (14-30.6). Almost all plots were found to be near the swamp average and outside of the bog range, with site values ranging from 14.33 to 28.65 (Figure 4.10). The only exceptions to this were sites 23 and 24 which were located in dense *Empodisma*, which had elevated C:N ratios of 38.62 and 53.07. This implies that the majority of the wetland has received input of N from an external source other than rainwater, such as periodic flooding events from the drainage channels surrounding the wetland area (Blyth *et al*., 2013).

Other nutrient ratios which are useful for distinguishing nutrient availability or limitation is N:P (whether the substrate is limited by either N or P) and C:P (availability of P). Phosphorus and N are often limited in bogs due to a lack of surface or groundwater inputs, and as such high N:P ratios indicate no P is being imported by surficial water movement, while N is still being brought in as small amounts with rainwater (Sorrell & Gerbeaux, 2004). Plants need P to grow, and in most wetlands are generally limited by P more than N (Sorrell & Gerbeaux, 2004). Additionally, as inputs from surface water increase (such as from flood events), P becomes increasingly abundant and accessible through transport of sediments, resulting in N limiting vegetation growth. This was found in Whangamarino wetland with flooding of the Whangamarino river entering the wetland (Blyth *et al*., 2013).

Clarkson *et al*. (2004b) recorded New Zealand wetland nutrient ratios, with bogs averaging 39 for N:P ratios (range 20.6–81.6), and averaging 1904 for C:P (range 533-4221). For swamps N:P ratios averaged 9.1 (range 4.0–20.6) while C:P averaged 163 (range 45-435). These two ratios showed similar patterns across Otakairangi (Figure 4.11), with Transects B, E and the end of Transect D expressing higher ratios, which indicates lower availability of P (N:P ranging from 35.94 to 81.30, C:P ranging from 925 to 3140), falling in Clarkson's range for bog values. Lower ratios were recorded on the west end of Transect A, across all of Transect C, and the beginning of Transect D (N:P ranging from 13.60 to 32.53, C:P ranging from 195 to 779), falling above Clarkson's range for swamp values. This indicates that while these areas of the wetland have higher abundances of P than the bog-like areas, they are still potentially P limited. Deposition of sediments and increasing concentrations of Al, Fe, K and Ca closer to the central and northern drains could potentially be adsorbing P from solution, making it unavailable to plants (Bridgham *et al*., 1998). However, it is widely regarded that N:P ratios in plant biomass is a better approach for determining nutrient limitation in wetlands, and is examined in the vegetation chapter.

4.4.5 Nitrogen 15 isotope

The stable isotope $\delta^{15}N$ can be used to determine external inputs of N into wetland areas, as N can be washed into waterways and from there could enter the wetland through flooding. Fertilizer $\delta^{15}N$ ranges from 0.5 to 5‰, while animal waste is generally enriched between 10 and 20‰ $\delta^{15}N$ (Xue *et al*., 2009).

The variations in $\delta^{15}N$ followed a similar pattern to previously measured variables, with areas near to drains having elevated $\delta^{15}N$ levels and most transects otherwise showing little variation. The notable exception was Transect D, which had elevated levels near the hillslope/drainage ditch, decreasing with distance into the wetland area (range 0.60-6.97‰). While Transects A, C and E all showed elevated levels of $\delta^{15}N$ (averages 2.06, 2.14, and 1.29‰ respectively), Transect B showed near-atmospheric levels of $\delta^{15}N$, with an average of 0.19‰.

Overall, the increase in $\delta^{15}N$ across the wetland was clear, with the surrounding agricultural practises being an important contribution. However, other factors contribute to $\delta^{15}N$ changes, such as variations through biological processes and therefore identifying the direct cause for the $\delta^{15}N$ increase can be difficult. Enrichment generally occurs through fertiliser, manure and effluent application to soils, which is likely the cause for higher $\delta^{15}N$ levels observed in the upper wetland area. Cross examination of $\delta^{15}N$ found in plant biomass is an approach that provides indications of nutrient limitation, due to the strong correlation with P availability in both soil and plant biomass (Clarkson *et al.*, 2005), and is investigated in Chapter 6.

4.4.6 Other elements

Uranium is naturally occurring, and therefore low concentrations would be expected to be found in wetlands through atmospheric deposition. However, U can be used as an indicator for fertiliser inputs, as phosphate rock (commonly used in pastoral fertilisers) is known to have high concentrations of heavy metals (including U), and technologies to completely remove them from fertilisers are not yet utilised worldwide (including New Zealand) (Roberts & Longhurst, 2002; Roberts, 2014). Any erosion from the areas the fertilisers are applied could then move U that is bound to organic matter and sediment through water systems into the wetland (Takeda *et al.*, 2006).

Increases in U concentrations were seen in the western side of Transect A, in Transect C extending from the northern drain, and in the northern section of Transect D (Figure 4.13). These increases are all found next to the northern and upper central drain, which is likely due to the main upper catchment and the northern hill slopes consisting of pastoral farming, suggesting there is some artificial input of P coming directly from fertilisers.

It has been shown in other wetlands that sediments that are transported from water ways by flood deposition, bringing with it large amounts of fine particles containing high levels of Al, Fe, Ca and K (Blyth *et al*., 2013). This was shown in the data collected across Otakairangi, where the concentrations of elements generally showed the same pattern, with increased values near to drainage ditches that are assumed to experience flooding and transport of sediments into the wetland (Figures 4.14, 4.15 & 4.16). This includes large increases to elemental concentrations along the northern drain below the pastoral hillslopes and below the native bush, and on the western side of the upper central drain. Other areas, such as near the southern central drain and southern drain showed slightly elevated concentrations, indicating that they do not receive flood inputs as often or as large as the other areas. However, there were exceptions to the overall pattern observed, with higher concentrations of some elements in different areas. Across the northern transect (C) Ca showed high concentrations (Figure 4.14), indicating potential aerial input of dust during lime fertiliser application. The southern transect (B) had two sites with elevated Fe concentrations, only matched by site 19 (swamp section beneath native bush) (Figure 4.16), potentially indicating areas of past inundation, submerged bog Fe ores, or sites of diffuse groundwater upwelling (Tiner, 2016).

4.4.7 Wetland zonation

The use of cluster analysis and ordinations to cluster sites by similarity represents a transformation of multiple sets of data into easily interpretable sets of sites. This in turn allowed the zonation of the wetland based on specific conditions (Figure

4.20). Browne (2005) utilised peat characteristics in ordinations to describe nutrient enrichment, while Blyth (2011) used ordinations to express the effect how flood regime and peat environmental variables influence wetland vegetation communities. Variables such as nutrients, concentrations of metals and N15, and physical peat properties contributed to the ordinations, showing that different areas of the wetland have different primary water inputs, nutrient availabilities, and rates of sediment input through flood inundation, which in turn influences the vegetation cover.

The ordinations showed six major clusters which generally showed a trend of moving from low nutrient, bog-like conditions to highly mineralised, nutrient rich sediment, but also showed a distinct zonation of low nutrient plots by vegetation cover (Figure 4.17). While secondary ordinations based around plant cover did not show any strong patterns as the majority of plots had *Gleichenia dicarpa* as the primary or secondary vegetation species, peat condition ordinations showed separation by vegetation in low nutrient plots (fen and fen-bog clusters)(Figure 4.18).

The northern area of the wetland showed far higher nutrient levels, metal concentrations, peat bulk density and von Post scores, indicating that this area is frequently inundated with flood waters from the central and border drains and is prone to peat degradation and mineralisation (Clarkson *et al*., 2004; Blyth, 2011). This is split between three areas; the area extending from the northern drain, the area below the native bush next to the central drain entrance, and the either side of the upper central drain. The area below the native bush (Plots 19 and 20) showed the highest values of mineralisation, implying it receives frequent inundation. This is most likely caused by its proximity to the entrance of the central drain to the wetland, meaning it is the first area to be inundated during flood events. The area beyond this (Plot 21), the area below the northern hill slopes and border drain (Plots 13-18), and the area west of the central drain (Plots 1, 2 and 3) show lower values and less degraded peat, meaning that while these areas are still frequently inundated and receive high nutrient inputs, the frequency of inundation is less than Plots 19 and 20. Compared to the northern areas, the southern transects (Plots 7-12 and 25-30) show low nutrient and metal concentrations and higher nutrient ratios. This implies that they are primarily fed by rain water and are infrequently (to never) inundated by drain water (Charman, 2002). However, these plots are split into two groups of similar characteristics, with the main differences being C:N ratios. Plot 22 is similar to these sites, and exhibits transitional values between the swamp-like western side of Transect D and the lower nutrient conditions in the east. The eastern end of this transect (Plots 23-24) shows far lower nutrient and metal concentrations, as well as lower bulk densities and von post ratings, implying that these sites are isolated from the water table and drain water and can be classified as bog sites (Clarkson *et al*., 2004b).

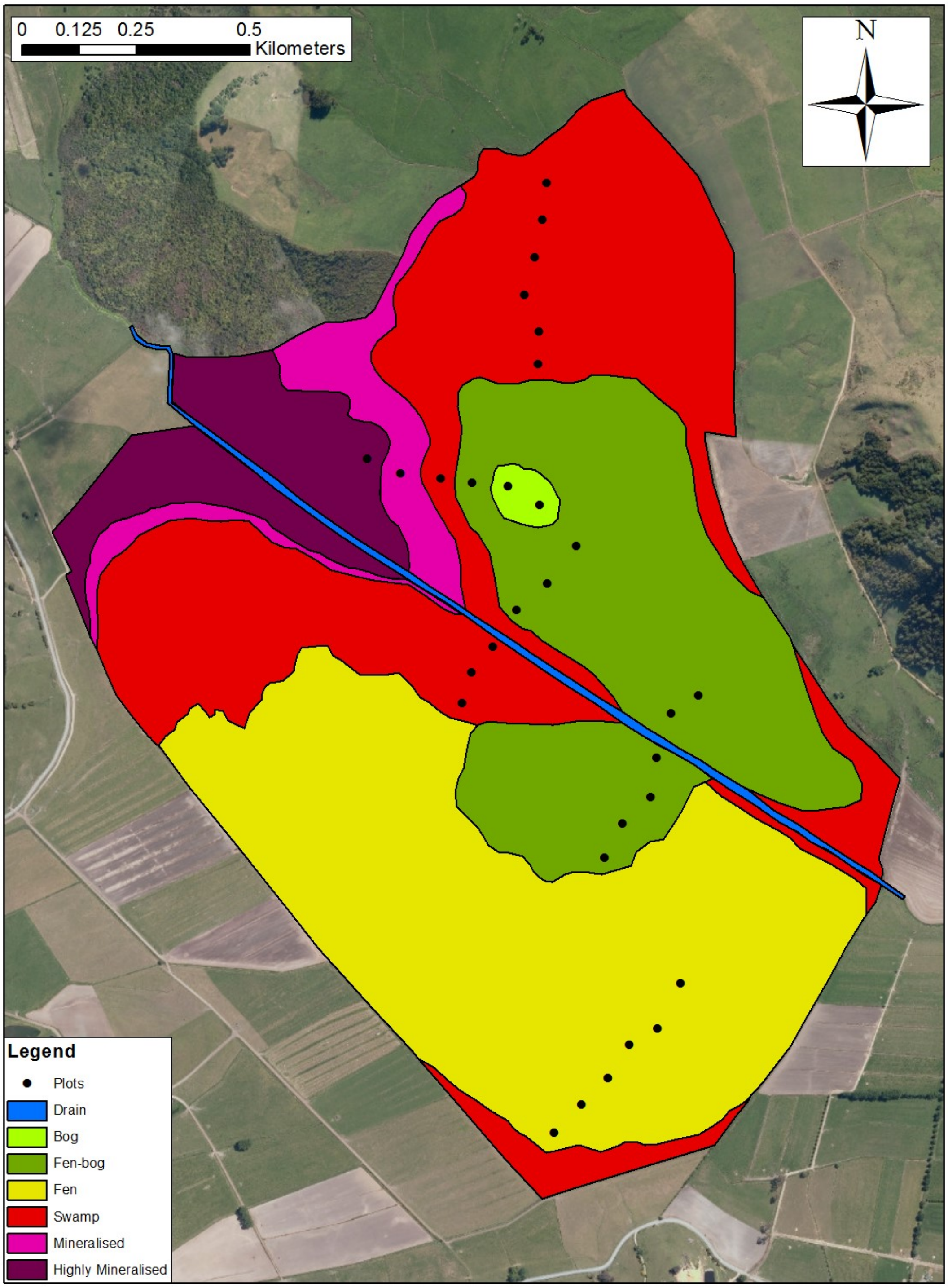

Figure 4.20. Conceptual wetland type zonation, based on ordination statistics and clustering of groups using field and laboratory assessments of vegetation and peat. Note, areas beyond transect lines estimated using vegetation composition (Kendal, 2016).

4.5 Summary

- 30 sites were established and measured for a variety of different chemical and physical attributes, the locations of which are visible in Figures 3.3 and 4.20.
- Transect A which crosses the upper central drain had more degraded substrate on the south-west side of the drain. Measurements of von Post, DBD, and mineral content were higher on the SW side and lower on the NE side. Concentrations of nutrients and metals also indicate increased degradation on the SW side of the central drain.
- Transect B had low von Post, DBD, and mineral content values, while P was also low in concentration. C and N concentrations were high, giving high C:N and C:P ratios, indicating that the area is predominantly rain-fed. Elevated Fe concentrations may indicate input from groundwater seepage or erosion of submerged Fe ore.
- Transect C is significantly degraded with higher measurements of von Post, DBD, and mineral content, as well as high P values and N15 enrichment, implying frequent inundation of surficial flood water from the northern drain. Increased abundance of Ca potentially indicate aerial deposition of lime fertiliser dust.
- Transect D shows a significant gradient from Plot 19 to 24 in all environmental variables measured, indicating a change from frequently inundated areas near Plots 19 and 20 through to rain-fed bog like areas at Plots 23 and 24.
- Transect E had lower nutrients and inorganic elements when compared to Transect A, and also show less degradation implying that the southern section of the central drain does not experience frequent inundation and may be primarily rain-fed.
- Multivariate statistical analyses and ordinations show groupings of wetland plots by peat characteristics, with a resulting environmental gradient of low fertility bog-like to high fertility swamp-like conditions.
- The most mineralised substrate was found beneath the native bush hillslope and beside the central drain. Swampland was found to be around

the central drain and around the northern drain where it is assume to flood most frequently. These zones were the highest in mineral content and nutrients, and were more highly degraded compared to the rest of the wetland. Metals such as Ca, K, Al and Fe were elevated in concentration near to drainage channels along with U and $\delta^{15}N$, most likely attributed to deposition with floodwaters.

- Restiad bog was found in the centre of the northern section at two sites, while fen-bog areas were found through the north-eastern zone and into the south-western zone of the wetland, crossing the southern section of the central drain. These zones were the lowest in nutrients and inorganic elements, and also had the lowest degradation values, indicating they are infrequently or never impacted by surficial water flows from the drainage channels.
- Fen-like sites were located in the far south of the wetland, where low nutrient conditions prevent encroachment of invasive vegetation and swamp species. This area is assumed to be rain-fed, infrequently flooded by the deep southern drain, and potentially influenced by groundwater upwelling.

Chapter 5. Vegetation characteristics

5.1 Introduction

The composition and associated chemistry of vegetation was assessed along five transect lines to determine how the hydrological regime influences the distribution of surface vegetation within Otakairangi wetland.

Along each transect line, six vegetation plots were established at intervals of 75 m, for a total of 30 plots over 2250 m. Each plot was assessed as per the Atkinson method for structure and composition of vegetation, while the plot health was determined by the presence and absence of invasive species, and the relative stress of living plants and dieback (Clarkson *et al*., 2004b). At each plot vegetation samples were retrieved for chemical analysis as analyses of plant tissue chemistry provide useful information related to fundamental processes such as nutrient retention and plant nutrition.

5.2 Methodology

Vegetation was analysed in terms of composition by plot area in the field, as well as chemically in a laboratory. The methods used for analysing the vegetation are summarised below.

- The establishment of the plots along transect lines
- Collection of foliage samples within or near the plots
- Analysis of the chemical composition of samples
- Multivariate analysis of the collected data

5.2.1 Plot establishment

As a single transect would not allow a true representative analysis of the whole wetland, five smaller transect lines were established originating from or across a point of major influence to the wetland, each supporting six vegetation plots. The transect lines were subjectively established in areas of high influence to the wetland, such as across the central drain or extending from the base of the native bush to the north. These plots were separated by 75 m distance to ensure that the plots captured any changes in vegetation along the transect line, but did not extend onto another transect line (Figure 3).

The plots were 4x4 m in size and were established at random distances (5-25 m) either side of the transect line, chosen by randomly generating a number on a handheld calculator. The plot establishment and assessment of the vegetation was undertaken using the technique outlined in the Handbook for Monitoring Wetland Condition (Clarkson *et al*., 2004b). The percentage cover of species in plots was given as raw assessments by the same observer as this was a subjective procedure. Only one layer (canopy) was used as the vast majority of plots were in areas where the only tall vegetation species was manuka, which was never taller than 5 m.

Vegetation plots were assessed by determining the dominant plant species first, followed by the cover of subsequent species. The wetland plot sheets were completed during time in the field, including plot health indicators, peat decomposition, and field measurements of water table.

Figure 5.1. Photo of plot 24, with markers used to indicate the extent of the 4 x 4 m plots. Pictured plot dominated by *Empodisma robustum, Gleichenia dicarpa, Machaerina teretifolia* and *Leptospermum scoparium.*

5.2.2 Foliage sampling

Samples of vegetation material were taken at every site along each transect, either from within the plot itself, or from a nearby plant. Small brown paper envelopes were filled with the leaves and end stem material of the most abundant species across the wetland, *Gleichenia dicarpa*, where it was present. *G. dicarpa* was collected at 28 of the 30 sites, and in one of the remaining two plots where *G. dicarpa* was absent the next most common species, *Leptospermum scoparium* (mānuka), was sampled. However, the remaining plot was heavily influenced by the northern native bush and mineral soils, and as such the vegetation was significantly different, being comprised of other species. Therefore, two vegetation samples were collected, one from the most abundant species, *Coprosma tenuicaulis*, and one sample from the most abundant fern species, *Paesia scaberula*.

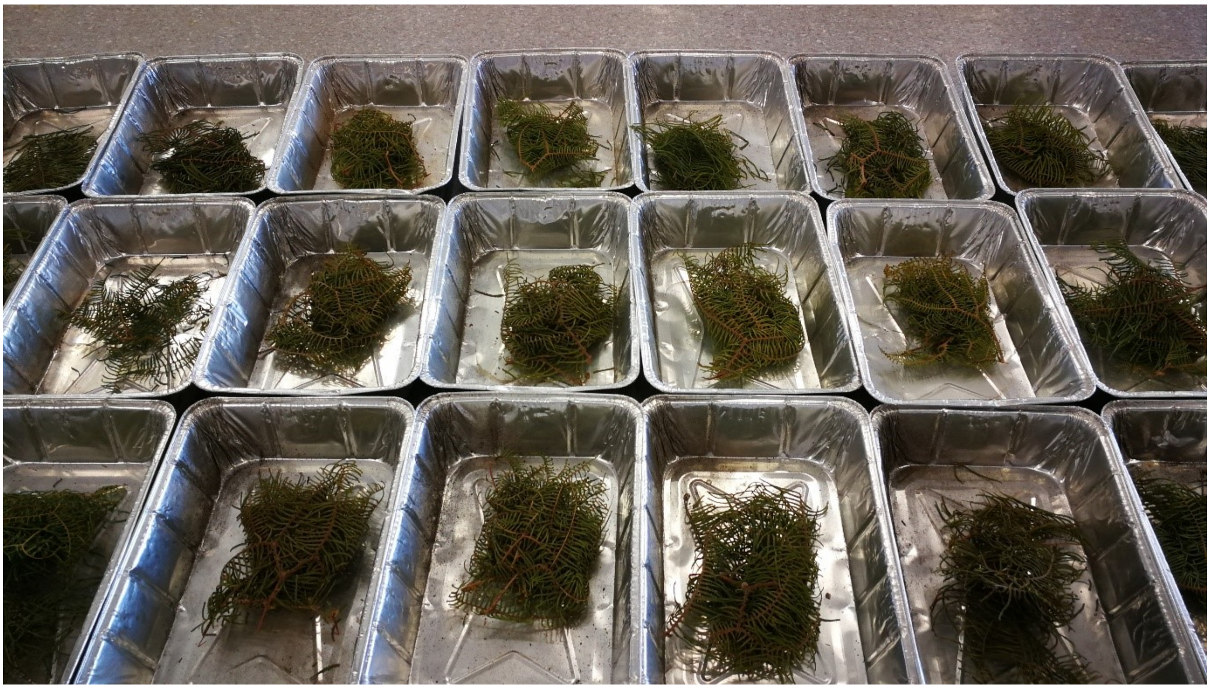

Figure 5.2. Dried *Gleichenia dicarpa* foliage used for chemical analysis.

5.2.3 Chemical analysis of carbon, nitrogen and $\delta^{15}N$

The analysis for the organic material (C, TN and $\delta^{15}N$) present in the foliage samples required similar sample preparation techniques to peat (described in Chapter 4). The samples were oven dried at 30 °C for a 24 hour period, and then left to air dry 2–3 days. The leaves were stripped from the stems and were then ground to a fine powder in an agate and ball on a shaking machine, and stored in air-tight containers.

Small portions of these samples were weighed and analysed by an Elementar C&N analyser (through the University of Waikato laboratory) for C and N concentrations, while $\delta^{15}N$ was analysed by a fully automated Europa Scientific 20/20 isotope analyser (the University of Waikato Stable Isotope Lab).

5.2.4 Chemical analysis of metals and other elements

For the concentrations of other elements in the foliage, such as P, dried leaf matter was digested into an aqueous solution and analysed in the ICP-MS lab at the University of Waikato. The digestion process involved weighing 200 mg of dried and finely ground leaf tissue from each sample into 50ml Falcon tubes. 1 ml of concentrated HNO_3 (65%) and 0.4 ml of concentrated H_2O_2 (30%) was then added to the sample and left to pre-digest overnight at room temperature. The falcon tubes were then heated at 80 °C for one hour with the tube caps loosened, followed by half an hour of cooling. A further 0.4 ml of H_2O_2 was added, followed by a further half-hour of re-heating. This step was repeated two more times. After the samples were fully digested, type one water was added to bring the final solution to 50ml (2% HNO_3). 15 ml of this solution was filtered using a syringe and 0.45 micron filter into a 15 ml Falcon tube suitable for ICP-MS analysis.

5.2.5 Vegetation community analysis

Major wetland vegetation community groupings were established by allocating each site a label using the dominant vegetation species. The five final groups were: *Coprosma tenuicaulis/Phormium tenax*, *Gleichenia dicarpa*/*Leptospermum scoparium*, *G. dicarpa, Machaerina teretifolia/G. dicarpa* and *G. dicarpa/Empodisma robustum*. Substrate chemical characteristics (Chapter 4), including bulk nutrients, nutrient ratios, and inorganic elements were then plotted as box plots with sites binned by these vegetation community groups to determine substrate preference.

A map based on drone photography of the wetland (Kendal, 2016) was ground-truthed with onsite visual observations along the transect lines and access routes, and adjusted by satellite imagery to assist in measuring the present spread of

Empodisma robustum. This allowed the vegetation patterns to be compared to peat zonation mapping from Chapter 4.

The five dominant vegetation species located across the wetland area (*G. dicarpa, E. robustum, M. teretifolia, L. scoparium,* and *C. tenuicaulis*) were individually analysed for dominant locations by interpolating individual plot % cover onto the wetland substrate ordination (Section 4.3.6, Figure 4.18).

5.2.6 Vegetation indexing

Species diversity indices are statistical variables which describe the vegetation community. These were calculated for each vegetation plot using the following methods:

(1) Species richness index **(R)**:

$$\mathbf{R = S}$$

Where **S** is the number of individual vegetation species found within the bounds of the vegetation plots.

(2) Shannon-Wiener index **(H)**:

$$\mathbf{H = -\sum P_i \ln P_i}$$

Where $\mathbf{P_i}$ is the proportion of the plot area covered by species i.

(3) Species evenness index **(E)**:

$$\mathbf{E = H / \ln(R)}$$

Average vegetation cover by the groupings determined in Chapter 4 give relative species importance values using the following equation:

(4) Vegetation cover averages by ordination group **(V):**
(Adapted from Importance Value Index (IVI))

$$\mathbf{V = (n_i / t) \times (\sum c_i / n_i)}$$

Where $\mathbf{n_i}$ is the number of plots in which species i occurs (within a single ordination group), **t** is the total number of plots (in the group), and $\mathbf{c_i}$ is the cover of species i (by plot).

5.3 Results

5.3.1 Foliage chemistry

The total N in foliage samples was elevated for sites near to drains compared to other plots on the transect (Figure 5.3). Nitrogen peaked for *Gleichenia dicarpa* foliage at plot 13 with 1.58%, while the highest TN was recorded in the site 19 *Paesia scaberula* sample with 1.95%. The southern transects (B and E) did not vary greatly (both ranging 0.23%), while the northern transects showed far greater range (0.46% and 0.72% ranges for Transects A and C, respectively).

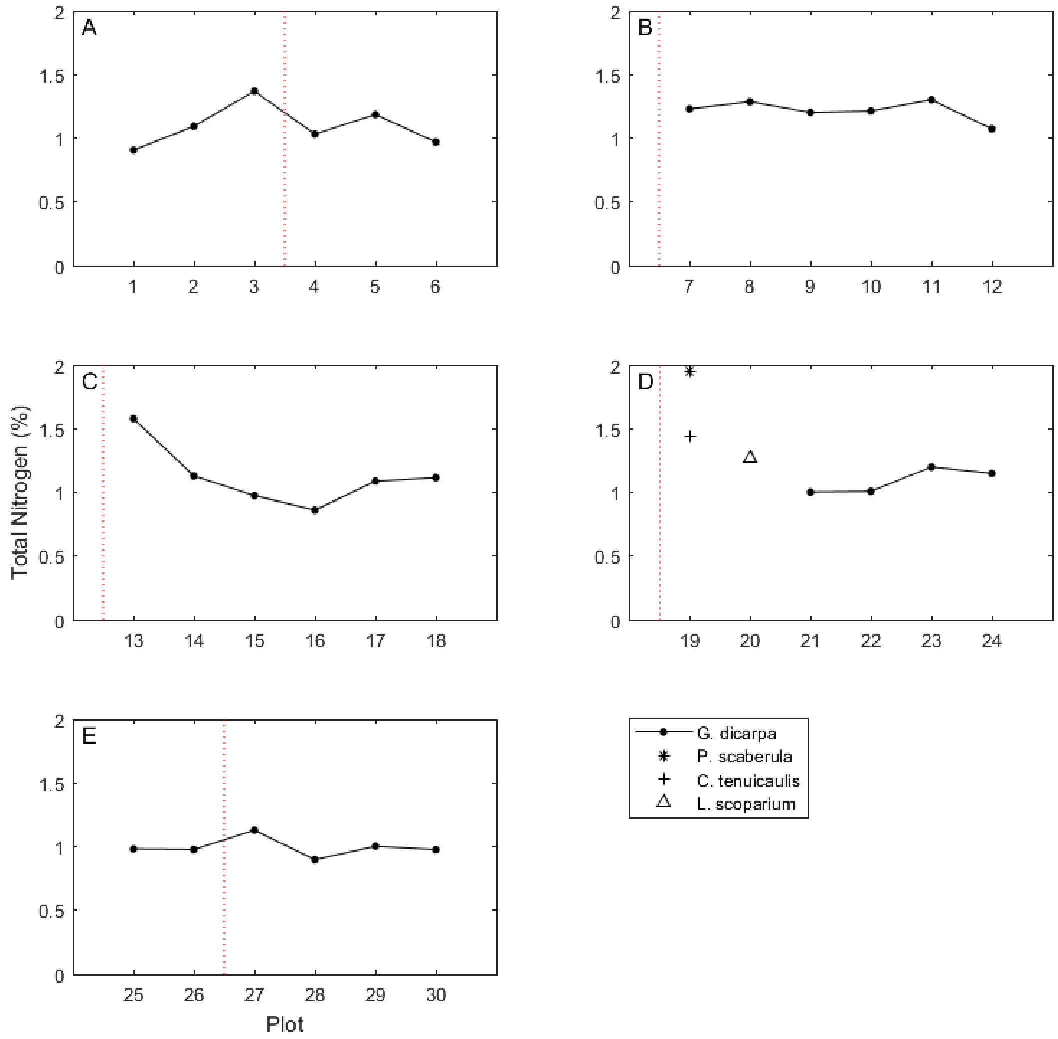

Figure 5.3. Nitrogen concentration (%) in foliage samples plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

Nitrogen 15 isotope concentrations were mostly depleted in vegetation samples compared to atmospheric values, with enriched values in vegetation near to drainage channels. The highest value was recorded at plot 13 nearest to the northern drain at 4.89 ‰, while the lowest value was recorded at plot 28 with a depleted minimum value of -7.64 ‰ (Figure 5.4). Transect C showed the greatest range of 10.7 ‰.

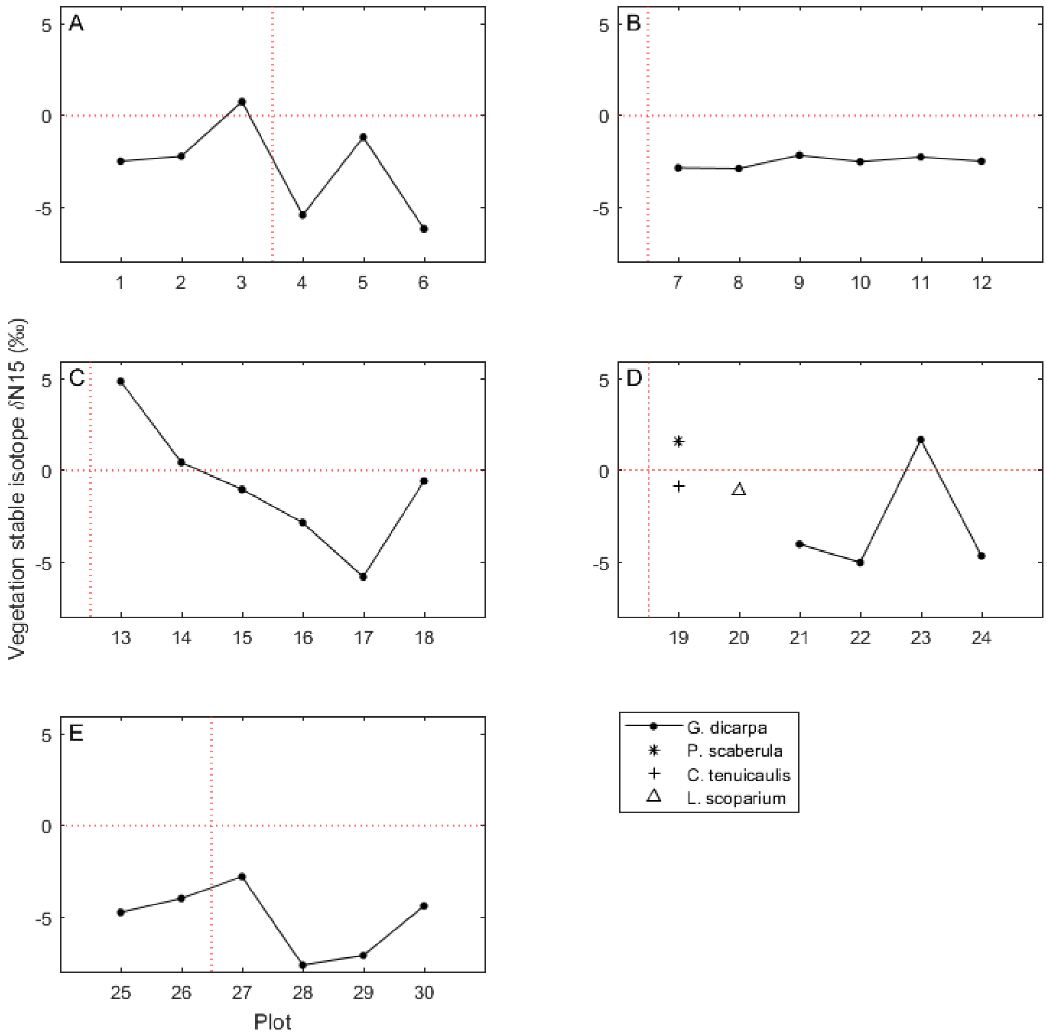

Figure 5.4. Nitrogen 15 isotope in foliage samples plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect. Horizontal lines indicates the position of atmospheric levels, with points below indicating depletion and points above indicating accumulation.

Phosphorus concentrations in foliage followed the same pattern as N, with higher concentrations near to the northern drainage channels and bush slope, decreasing with distance into the wetland, while the southern plots were lower consistently (Figure 5.5). The maximum concentration of P occurred at site 13 near to the shallow northern drain with a value of 0.12%, while the lowest value occurred in the south at site 12 with a value of 0.02%.

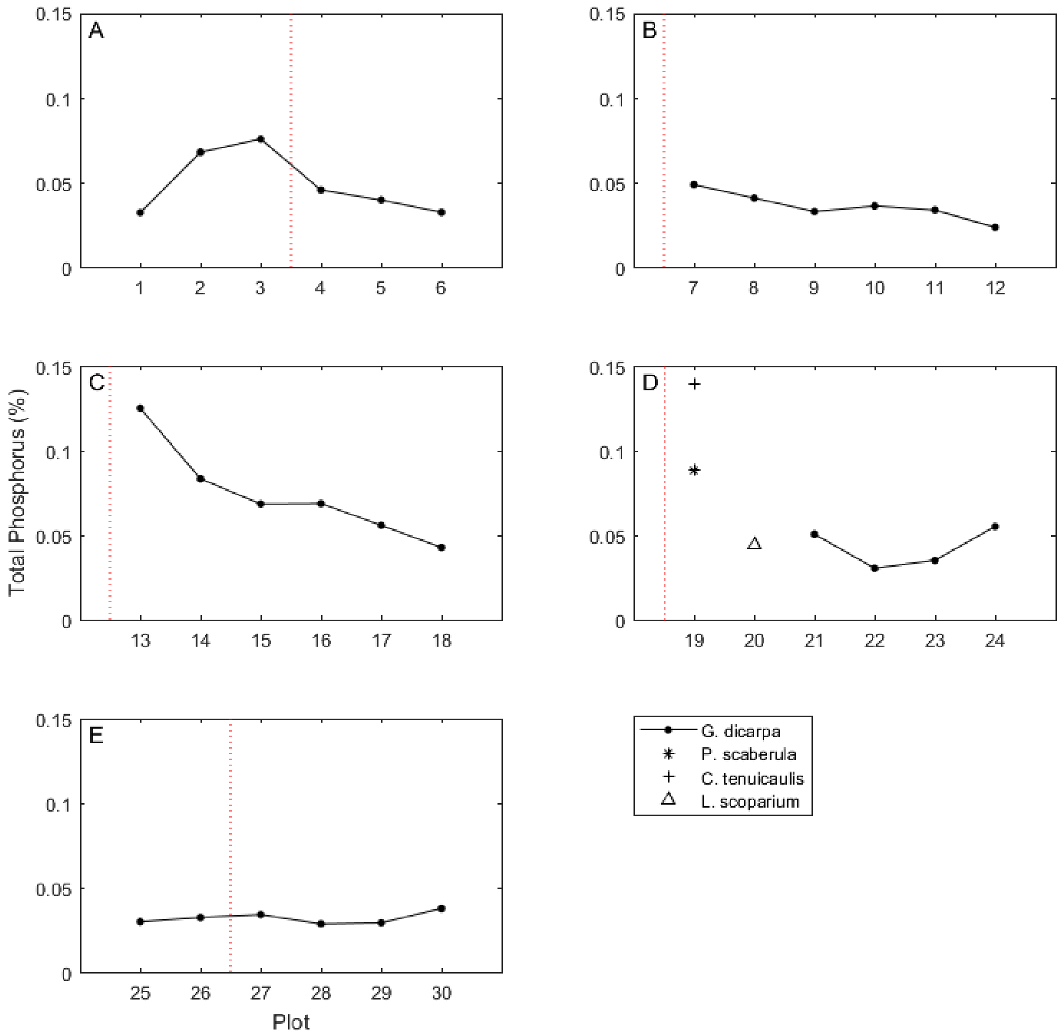

Figure 5.5. Phosphorus concentration (%) in foliage samples plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect.

N:P ratios in plant biomass is a commonly used indicator to determine whether a wetland is limited by either nutrients (Güsewell *et al.*, 2003). The ratios express that Otakairangi wetland is most likely limited by P (southern areas and inner wetland zones), while some areas near to drainage channels (northern drain, bush slope and upper central drain) are limited by both N and P (Figure 5.6). No plots were purely N-limited.

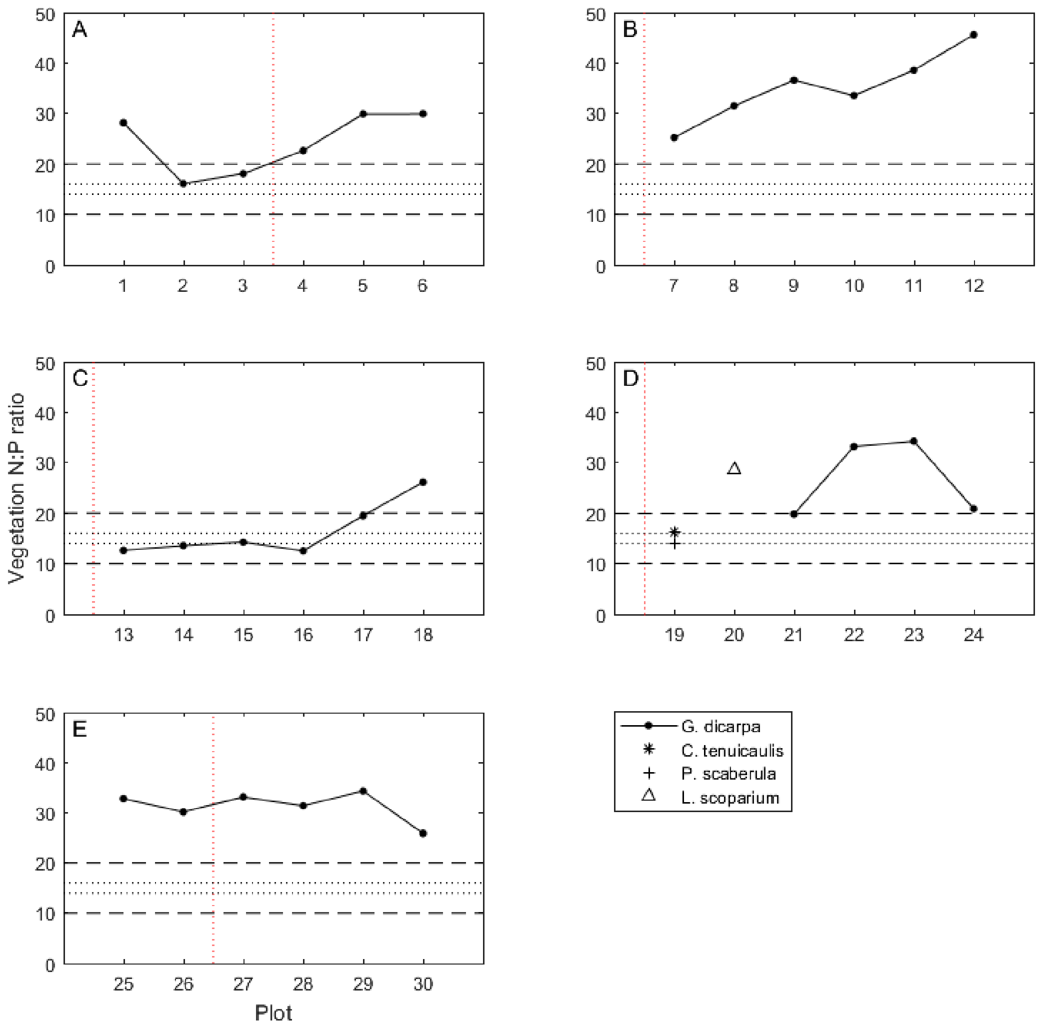

Figure 5.6. Nitrogen to phosphorus ratios in foliage samples plotted by transect (A-E) and plot number. Vertical lines indicate the relative position of the assumed influence along each transect. Horizontal lines are indications of nutrient limiting conditions based on Güsewell *et al.*, (2003); Cusell *et al*., (2014); and Emsens *et al.* (2017).

5.3.2 Substrate nutrients and ratios by vegetation cover

Substrate nutrients show strong variation between vegetation cover types. This zonation shows that plots dominated in some part by *Empodisma robustum* tend to have lower N and C, while sections dominated by *Gleichenia dicarpa* and *Coprosma tenuicaulis/Phormium tenax* have higher values (Figure 5.7 (a) and (b)). Phosphorus shows more overlap in boxplots, with plots dominated by *E. robustum* or *Machaerina teretifolia* having the lowest P values (Figure 5.7 (c) and (d)).

G. dicarpa dominated plots had the highest median C:N and C:P ratios, the *M. teretifolia/G. dicarpa* grouping had the highest median N:P ratio. Groups partially dominated by *E. robustum* had the highest maximum C:N and C:P ratios, also with high N:P ratios, indicating that the restiad bog areas exhibited conditions with greater accumulation of organic matter and lower nutrient inputs/P limitation. The highest N:P ratios were seen in the *M. teretifolia/G. dicarpa* group, showing that this group was nutrient limited, but does not have bog like accumulation of organic matter. Groups with *C. tenuicaulis* or *Leptospermum scoparium* experienced the lowest ratios, implying high nutrient input and availability, and/or higher decomposition rates.

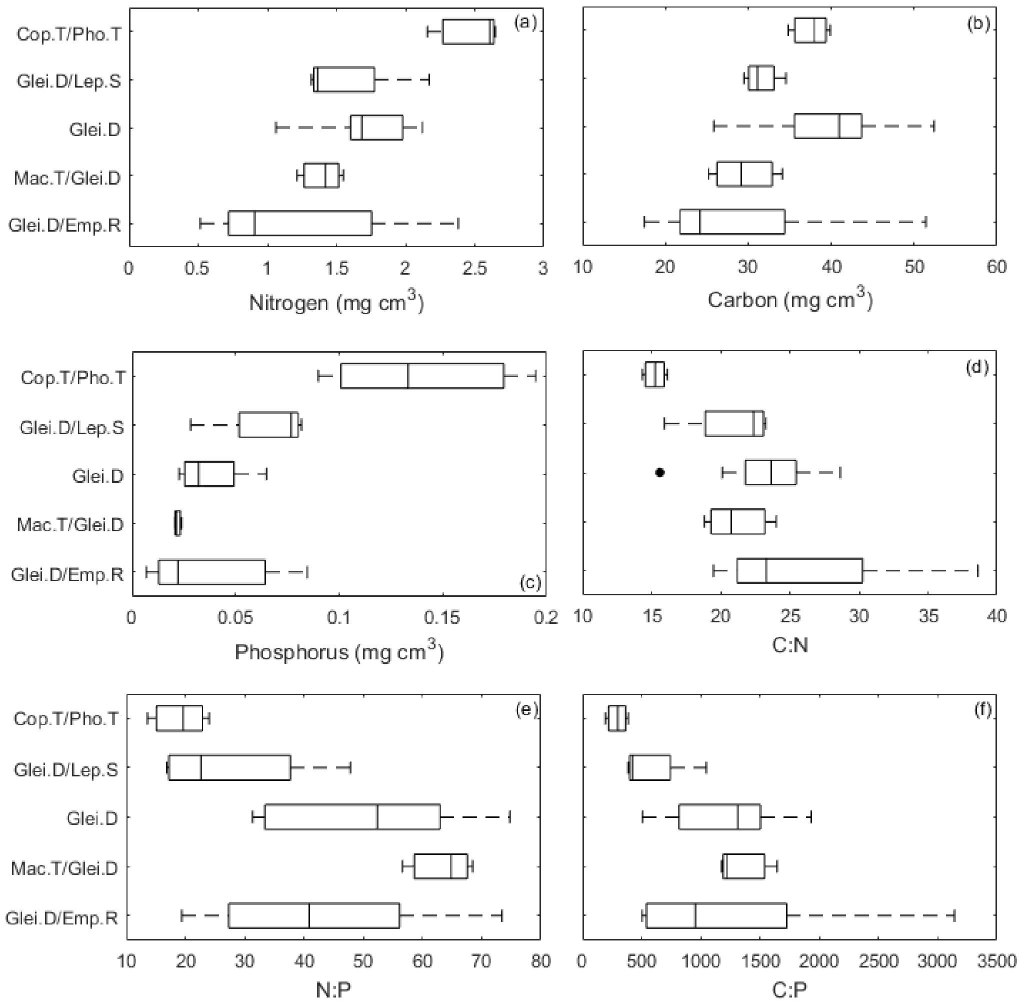

Figure 5.7. Box plot summaries of peat nutrients (a) nitrogen, (b) carbon, (c) phosphorus, and peat nutrient ratios (d) C:N, (e) N:P, and (f) C:P for each of the five dominant communities. Box plots show the median, upper and lower quartiles, and outliers. Glei.D = *Gleichenia dicarpa*, Lep.S = *Leptospermum scoparium*, Emp.R = *Empodisma robustum*, Cop.T = *Coprosma tenuicaulis,* Pho.T = *Phormium tenax*, Mac.T = *Machaerina teretifolia.*

5.3.3 Trace elements

Concentrations of some trace elements show greater variation within groups, making the vegetation cover groups less distinct from each other. For all elements, the *E. robustum* group had low concentrations with some higher values drawing out the upper quartiles, while the *C. tenuicaulis/P. tenax* groups had highly variable concentrations (with the exception of Ca, which experienced elevated outliers for the *G. dicarpa* groups due to the northern transect) (Figure 5.8). The variation between trace elements indicate highly variable hydrology for the *C. tenuicaulis/P. tenax* group with input from surface waters, while the lower values for *E. robustum* and *Machaerina teretifolia* groups indicating precipitation as the dominant water source.

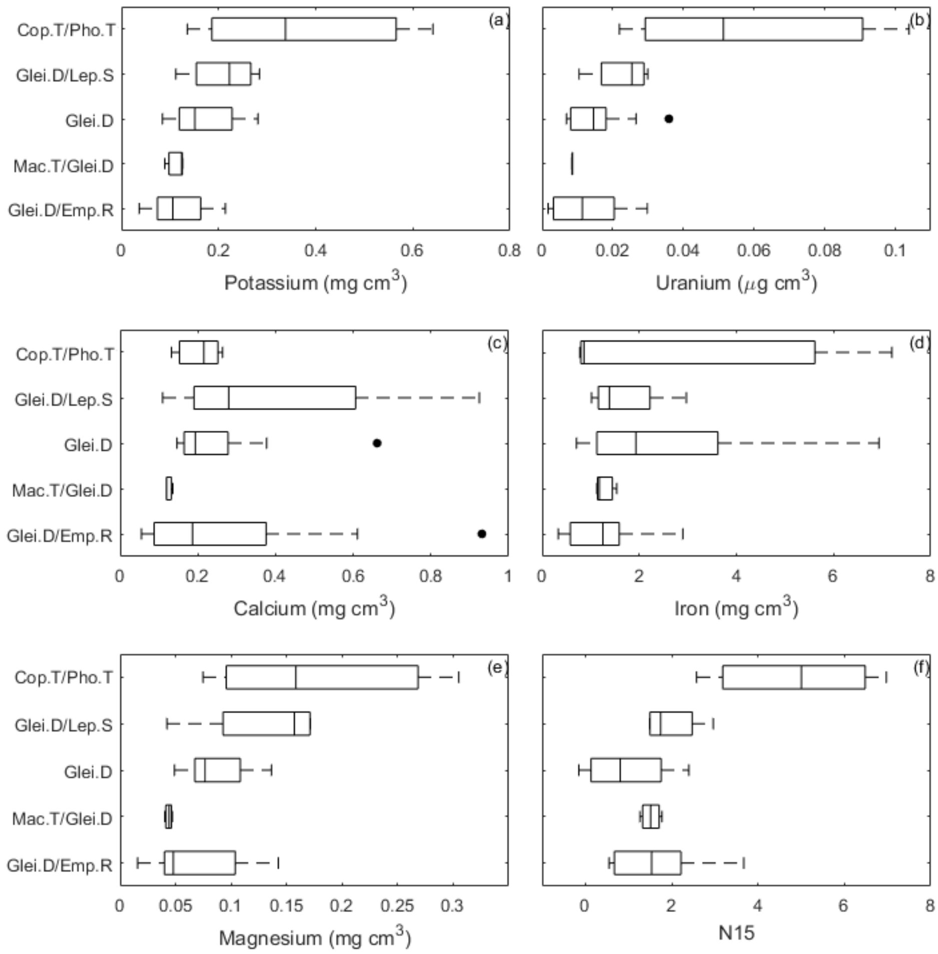

Figure 5.8. Box plot summaries of peat trace elements (a) potassium, (b) uranium, (c) calcium, (d) iron, (e) magnesium, and (f) nitrogen 15 isotope for each of the five dominant communities. Box plots show the median, upper and lower quartiles, and outliers. Glei.D = *Gleichenia dicarpa*, Lep.S = *Leptospermum scoparium*, Emp.R = *Empodisma robustum*, Cop.T = *Coprosma tenuicaulis,* Pho.T = *Phormium tenax*, Mac.T = *Machaerina teretifolia.*

5.3.4 Vegetation patterns

The wetland vegetation map (Figure 5.9) showed zonation of vegetation cover, with wetland plants in the central zones of the wetland, and having invasive species in the marginal banding around the edges of the wetland area where the ring drains separated the wetland from the surrounding farmland.

The marginal areas, or 'swamp belts', were characterised by high diversity and generally taller vegetation due to the additional inputs of water and nutrients from the drains. These areas were covered predominantly by *Phormium tenax*, although other species that were found include *Convolvulus arvensis*, *Glyceria maxima* and *Ranunculus repens*. Larger tree species also occurred, such as *Cyathea dealbata* and *Dicksonia squarrosa*.

Further into the wetland the vegetation is shorter with decreased diversity, creating fern and restiad lands that are more characteristic of peat bogs. These areas had four main vegetation species: *G. dicarpa*, *M. teretifolia*, and *E. robustum* as the groundcover, and *L. scoparium* as both groundcover and canopy. *G. dicarpa* was the most common and widespread species, found almost everywhere in the wetland with the exception of some areas in the northern swamp zone. *M. teretifolia* was also common across large swaths of the wetland area, such as Transect E, but was also sparsely spread in several areas, such as Transect B. *L. scoparium* occurred across the majority of the wetland area, but varied in density. It occurred as both singular, isolated trees and as dense forests stands, and was more frequently found as a secondary ring inside the marginal bands. The centre of the north-eastern section of the wetland was dominated by large patches of *E. robustum*, with sections now being converted into restiadland. It was also found in smaller patches sporadically across the wetland area, both in open areas (such as in the south-western section) and in the high diversity marginal bands (central drain, northern drain). Some areas showed elevated stress levels in vegetation, such as in some parts of Transect B extending from the southern drain. In this area a significant portion of the taller manuka trees were dying or already dead, with the tangle fern groundcover often broken with fallen manuka.

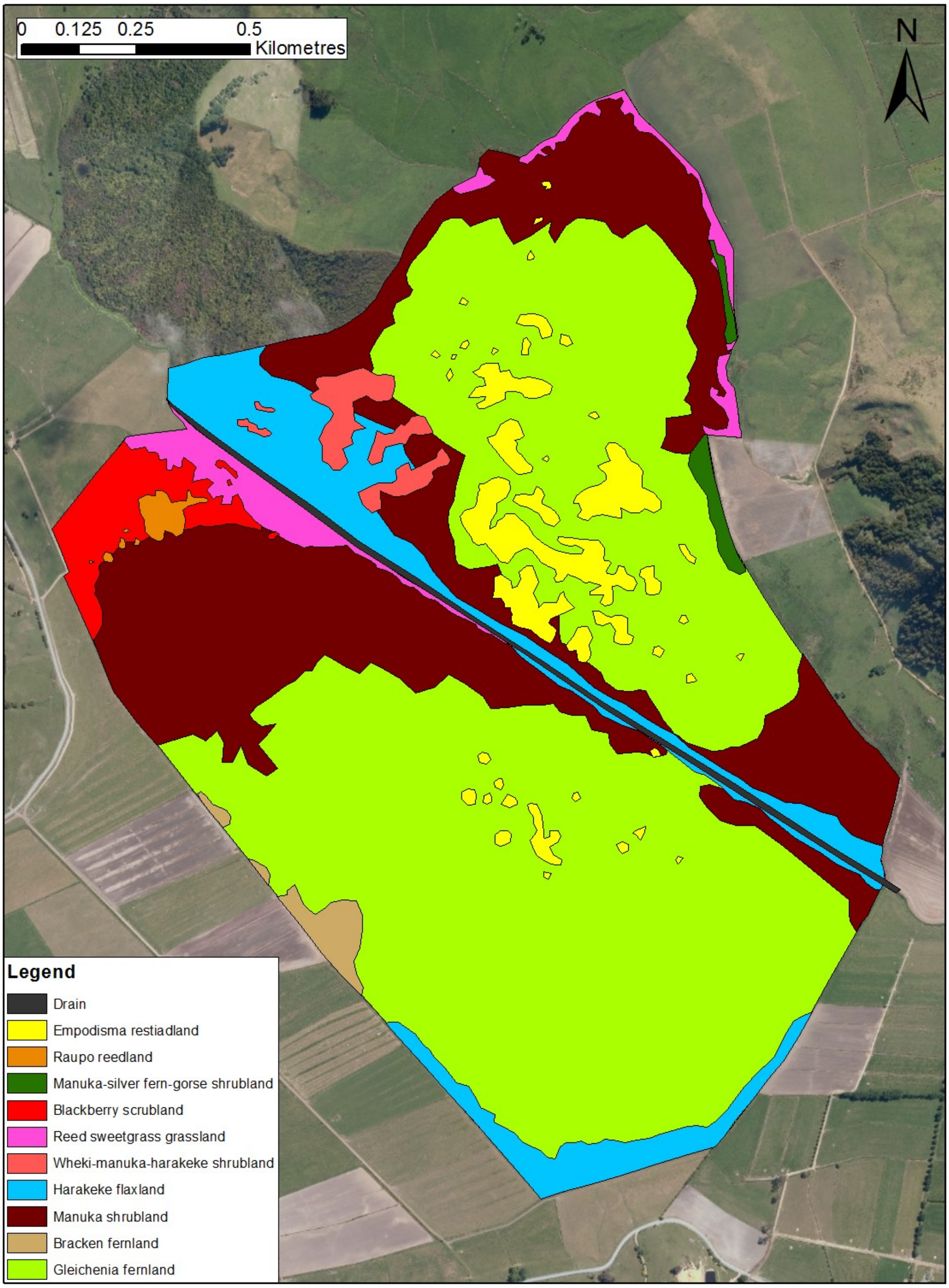

Figure 5.9. Wetland vegetation cover. Based on aerial drone photography by Kendal (2016) with additional interpretation of satellite imagery and ground-truthed using onsite visual observations.

Using the base ordination plot (Figure 5.10 f) created using principle component analysis of the peat (chapter 4.3.5), vegetation cover was interpolated to exhibit vegetation zonation with substrate condition and fertility. *G. dicarpa* was the most widespread of the vegetation species, but was more abundant in the upper section of the ordination where other species were absent or lower in abundance (Figure 5.10 a). *E. robustum* and *M. teretifolia* were both found to have greatest cover in the lower left of the plot, where there was low fertility and higher nutrient ratios (Figure 5.10 b & c). *L. scoparium* had higher coverage in the lower centre to lower right, where increasing fertility reduces the cover of bog species (Figure 5.10 d). *C. tenuicaulis* had the highest cover in the far right of the ordination, where the fertility was greatest, while also having some high cover in lower fertility plots (Figure 5.10 e).

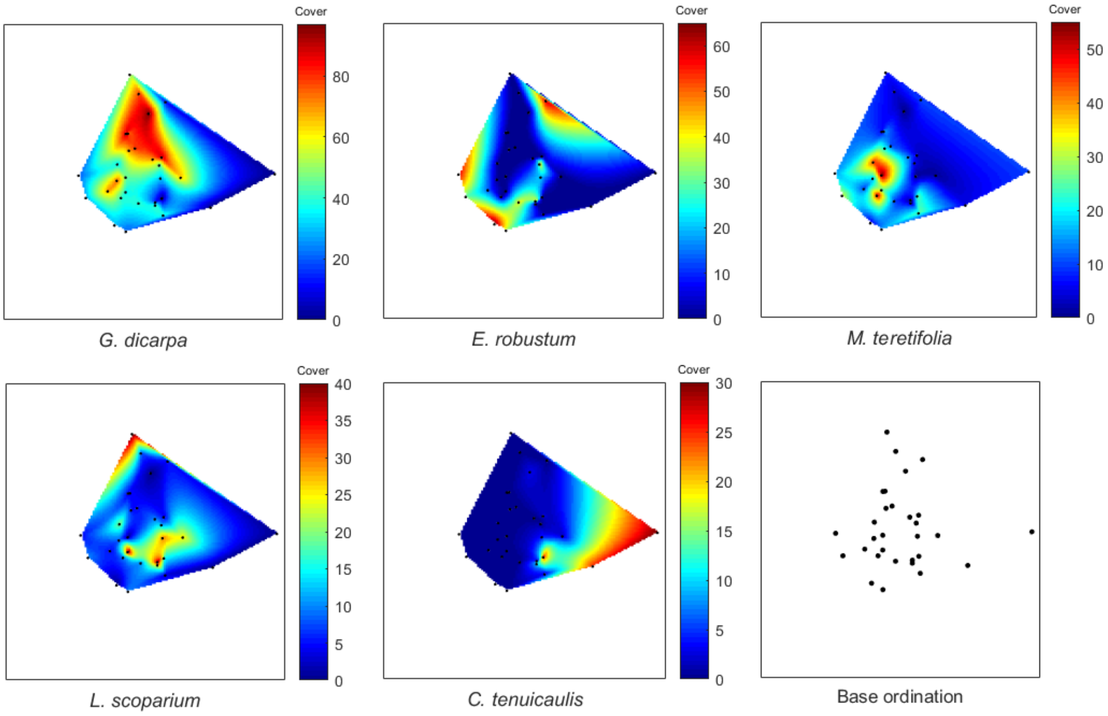

Figure 5.10. Percentage cover of vegetation species using the base ordination plot created in Section 4.3.5. Figures show cover for a) *G. dicarpa*, b) *E. robustum*, c) *M. teretifolia*, d) *L. scoparium*, e) *C. tenuicaulis,* and f) the base ordination plot.

Using the ordination groupings described in Section 4.3.6, vegetation cover was averaged to give relative species importance values for each wetland type. Between these groupings species compositions differed, with major vegetation species showing an increase in cover with either increasing or decreasing nutrients, while other species which were infrequently found did not show any major patterns (Table 5.1). *E. robustum* and *M. teretifolia* became more abundant as fertility decreased, with changes from 18% and 8% in swamp-like plots to 44% and 23% cover in bog-like plots, respectively. These two species were the dominant vegetation in the lower fertility plots, competing with the more widespread and abundant *G. dicarpa*. The vegetation species *L. scoparium* and *C. tenuicaulis* increased in cover with increasing fertility, increasing from 8% to 17% cover for *L. scoparium* (bog-like to swamp-like plots), and <1% to 30% for *C. tenuicaulis* (fen-bog to highly mineralised plots). The most abundant plant species across a wide range of the wetland was *G. dicarpa,* which showed a trend of peaking in the fen group (77%) and decreasing in the other groupings (24% in bog-like plots, 40% in swamp-like plots and down to 0% in mineralised plots)(Table 5.1).

The wetland vegetation is primarily dominated by *G. dicarpa*, with *M. teretifolia and L. scoparium* generally being secondary species. *E. robustum* has spread across a wide swath of the wetland and become locally dominant in several areas (Table 5.1). Average species richness decreased from 4.5 in bog-like plots to 3 in the fen-like plots, and then increased to a high value of 7 for the highly mineralised plot (Table 5.2). This trend was also seen in the Shannon-Wiener index and Evenness index, with values decreasing from the bog-like plots to the fen-bog and fen-like plots, and then increasing towards the mineralised plots (Table 5.2).

Table 5.1. Plot vegetation cover (mean ± 1 standard deviation) by ordination groupings (columns). The number of plots in each group is indicated in brackets. *Indicates the presence of the species in a single plot (excluding the mineralised and highly mineralised groups, which were comprised of single plots).

Species	Bog (2)	Fen-bog (10)	Fen (6)	Swamp (10)	Mineralised (1)	Highly mineralised (1)
Machaerina teretifolia	23 ± 11	24 ± 19	9 ± 7	8 ± 10	5	10
Gleichenia dicarpa	24 ± 7	47 ± 21	77 ± 16	40 ± 27	-	-
Leptospermum scoparium	8 ± 4	14 ± 24	10 ± 16	17 ± 13	-	-
Empodisma robustum	44 ± 19	15 ± 21	-	18 ± 28	-	-
Parablechnum minus	<1*	-	-	2 ± 4	-	-
Coprosma teniculis	-	<1*	1*	7 ± 9	25	30
Schoenus brevifolius	-	<1*	1*	9 ± 14	-	-
Usnea sp.	-	<1*	-	-	5	2
Phormium tenax	-	-	-	4 ± 8	35	20
Isachne globosa	-	-	-	<1*	-	8
Convolvulus arvensis	-	-	-	1 ± 2	-	15
Carex secta	-	-	-	<1*	-	-
Campylopus introflexus	-	-	-	<1*	-	-
Glyceria maxima	-	-	-	<1*	-	-
Ranunculus repens	-	-	-	<1*	-	-
Coprosma propinqua	-	-	-	<1*	-	-
Ulex europaeus	-	-	-	<1*	-	-
Paesia scaberula	-	-	-	-	20	15
Cyathea dealbata	-	-	-	-	10	-

Table 5.2. Vegetation indices by ordination grouping (mean ± standard deviation). Species richness indicates the amount of species recorded, Shannon-Wiener characterises communities by abundance and evenness of species, while the Evenness index indicates how close in abundance each species is (0 to 1, with 1 indicating perfect evenness).

Ordination group	Species richness index (R)	Shannon-Wiener index (H)	Evenness Index (E)
Bog (2)	4.5 ± 0.7	1.25 ± 0.09	0.84 ± 0.15
Fen-bog (10)	3.5 ± 1	0.97 ± 0.21	0.81 ± 0.21
Fen (6)	3 ± 0.6	0.58 ± 0.28	0.51 ± 0.22
Swamp (10)	5.5 ± 2.1	1.15 ± 0.35	0.70 ± 0.21
Mineralised (1)	6	1.57	0.87
Highly mineralised (1)	7	1.76	0.91

5.4 Discussion

The vegetation across the wetland showed distinct banding and zonation, with the majority of the border areas close to drains having taller woody vegetation or swamp species, while the central areas are characterised by fernlands and patches of *Empodisma* restiadlands.

5.4.1 Foliage chemistry

In wetland substrates, C:N and N:P ratios are indicators of nutrient limitations and wetland class (Clarkson *et al.*, 2004a). However, these do not necessarily reflect on vegetation biomass uptake of nutrients as substrate is affected by different factors. Vegetation nutrient uptake, while primarily determined by the availability of nutrients, is also strongly affected by temperature, competition, pH and microbial activity (Güsewell & Koerselman, 2002). As bogs are primarily rain fed and therefore are nutrient poor, it is assumed that they also have low nutrient availability for vegetation uptake. Examining N:P ratios in plant biomass (tissue samples) is recognised as being a better indicator of plant available nutrients, including N saturation or P limitation (Güsewell, 2004), as it takes into account the variability and relative changes of both key nutrients (Güsewell & Koerselman, 2002; Güsewell *et al.*, 2003). In wetlands, foliage N:P ratios generally explain at least 75% of nutrient limitation whereas N and P concentrations on their own can only explain 5% and 50%, respectively (Güsewell & Koerselman, 2002; Güsewell *et al.*, 2003).

The indicators of P or N limitation in New Zealand wetland systems has not yet been confirmed with fertilisation experiments, and despite different species having different responses, there are accepted boundaries for N:P ratio nutrient limitation. Broadly, ratios less than 10:1 indicates N limitation, with greater than 10:1 gradually transitioning to N and P co-limitation (14:1 - 16:1), while ratios greater than 20:1 indicate P limitation (Sorrell & Gerbeaux, 2004; Cusell *et al*., 2014; Emsens *et al*., 2017). Figure 5.6 indicated that the majority of sites sampled at in Otakairangi wetland were limited by P (southern transects and inner wetland zones), while plots closer to drainage channels where large sediment inputs were present lowered the N:P ratio and became nutrient co-limited. The southern wetland being limited by P is potentially caused by infrequent flood inundations, or P immobilisation with sorption to the substrate. Structural, compositional and ecosystem functional changes can be indicated with the use of N:P ratios, since continuous depositions of higher nutrient concentrations will change site characteristics. This may have occurred in the northern section due to regular flood inundation and the intensification of agriculture (with the resultant application of fertilisers) in the upper catchment area providing the northern section with higher nutrients.

Plant tissue N isotopes can indicate past N availability, while also indicating the current state and potential trajectory of N availability (Gerhart & McLauchlan, 2014). Variation in $\delta^{15}N$ among plants within an ecosystem has been interpreted as reflecting potential differences in mycorrhizal dependence, depth of acquisition, fixation and the utilization of depositional N, and therefore understanding patterns in data may not necessarily be straightforward. However, past studies have demonstrated that vegetation becomes enriched in $\delta^{15}N$ as the availability of N increases and vegetative processes preferentially utilise ^{14}N (Craine *et al*., 2015).

Restiad species in bogs such as *E. minus* usually have stable foliage $\delta^{15}N$ abundances, due to structural adaptations. Wiry stems direct rainfall to large cluster roots which are capable of utilising the limited nutrients available, and therefore taking *E. minus* foliage samples for $\delta^{15}N$ analysis would likely yield similar values to other New Zealand bogs (Clarkson *et al*., 2005).The majority of the foliage samples were *G. dicarpa* (found across the majority of the wetland), but two plots where it was absent necessitated sampling of other species (*L. scoparium, P. scaberula, and C. tenuicaulis*). The $\delta^{15}N$ (‰) found in vegetation samples across Otakairangi foliage samples showed depletion across most of the wetland, likely due to low availabilities of N for plant uptake (Figure 5.4). The areas of $\delta^{15}N$ enrichment were positioned near to drainage channels, and was likely attributed to greater nutrient inputs during past flood events.

Clarkson *et al*. (2005) studied the $\delta^{15}N$ abundance of foliage in New Zealand bogs and showed a positive correlation with foliage P concentrations, where enrichment of the $\delta^{15}N$ isotope correlated to increasing P abundance in plant matter (Clarkson *et al*., 2005). It was determined that near the centre of New Zealand bogs, *L. scoparium* had significantly depleted $\delta^{15}N$ levels around -14.96 ‰, while closer to the fringes higher $\delta^{15}N$ was measured, averaging around -3.71 ‰. Plot 20 at Otakairangi, where *L. scoparium* foliage was collected, showed slightly depleted values of -1.07 ‰, implying that this area has high flood inputs and differential N usage (Figure 5.4).

5.4.2 Vegetation patterns

G. dicarpa becomes outcompeted in either lower or higher fertility conditions, where other species which are more adapted to different nutrient inputs are present (Table 5.1). Past studies showed that *G. dicarpa* sprouts rapidly post-burning events as it regenerates from rhizomes in the substrate (McQueen & Forester, 2000). *G. dicarpa* was located across almost all of the wetland as a primary groundcover vegetation species, although it was less abundant in high and low nutrient conditions (Figure 5.9 and 5.10). The fen-like plots were similar to fen-bog in nutrient levels, and therefore it is probable that *G. dicarpa* is so prevalent in the south due to low nutrient conditions (Figure 5.8) inhibiting invasive species (and the resulting competition) and the lack of other bog species (*E. robustum* and *M. teretifolia*) due to slow dispersal rates from the northern section (Table 5.1). Given time, this section may convert to fen-bog and then to bog-like conditions with the arrival and associated peat construction of *E. robustum*.

E. robustum, while not as widespread as *G. dicarpa,* was found across a wide range of the wetland (Figure 5.9, Table 5.1). This is presumed to be due to it spreading radially from this origin point in the north-eastern section during the species mast events. This meant that it was not only found in the bog-like and fen-bog transitional plots, but also in swamp-like plots along the northern drain and near the central drain (Figure 5.10 and Table 5.1). In the areas where it was dominant the nutrient concentrations were low (assumed to be solely rain-fed) and there was significant build-up of new peat layers above the older, more degraded layers. This follows the idea that *E. robustum* is an ecosystem engineer, changing the availability of resources to other species through physical changes in biotic and abiotic material (Hodges & Rapson, 2010). The current spread of *E. robustum* indicates that the wetland is naturally recovering in terms of vegetation cover.

L. scoparium (manuka) was a common species across the whole wetland (Figure 5.10 and Table 5.1), providing light to medium cover across many areas due to its tolerance for different growing conditions. In some areas, such as the northern area and along the central drainage ditch, manuka grew in dense clusters with heavy foliage (Figure 5.9). This species (which is known to have synchronised growth of populations due to rapid colonisation utilising secondary succession in wetlands) (Perry *et al*., 2014) also exhibited group dieback in the southern area, with many fallen and decaying trunks along with dead individuals that were still standing (cracked and brittle trunks).

M. teretifolia was primarily located in low fertility areas of the wetland that are presumed to be rain-fed (Figure 5.8 and 5.10). However, while it was found mainly in bog and fen-bog sites, it was also present in the fen and swamp sites as well as the mineralised sites, making it the only species to be present in all ordination groupings (Table 5.1). While the is primarily found in moderately to extremely acidic peat bogs, it is also found in gum land and alongside lakes, streams and drainage ditches. It is able to survive in a range of conditions, and therefore was found across the wetland in all different conditions.

C. tenuicaulis is a common New Zealand wetland species, which showed a strong preference for higher fertility areas in Otakairangi (Table 5.1). It was found primarily along the northern transects where frequent inundation events brought nutrients into the wetland, and near to the southern and central border drains. However, it was also located within the central wetland area with *E. robustum* and *M. teretifolia*. *P. tenax* was present in the marginal areas of the wetland that were in close proximity to drainage channels, and as such it was only found in the mineralised sediment and swamp zones (Table 5.1).

Vegetation indices were used to determine vegetation patterns between the groupings identified in Section 4.3.6. For all three indices there was a transition from high values in the mineralised sites (high fertility, encroachment of invasive and native bush species) through to low in the fen site, and then increasing again into the bog sites (Table 5.2). This is due to a transition from high fertility to low fertility, with vegetation communities changing from larger swamp species (*P. tenax*) through to low nutrient bog species (*E. robustum).* The fen zone had the lowest values due to the absence of both invasive and swamp species (low nutrient conditions not supporting them), while also lacking bog species (which have not reached the area yet). The swamp zone expressed the largest range of any group for all three indices (excluding mineralised groups which were single sites), as it encompassed areas that are heavily influenced by drainage flooding but are also close to the inner wetland bog sites. Therefore they had a large range of species, with bog species (*E. robustum*) being found in some northern sites alongside swamp species (*P. tenax*) and invasive species (*Ulex europaeus*).

5.5 Summary

- Foliage N:P ratios indicated that this wetland is likely limited by P, even with significant P inputs around the marginal areas where drainage ditches overflow during rain events.
- Stable isotope nitrogen 15 indicated that areas near the upper central drain and the northern drain experience frequent flooding, while the inner wetland areas are primarily rain-fed.
- Swampland was present in a band around the northern and upper central drainage channels. This band was characterised by dense, tall *P. tenax* and *C. tenuicaulis*. The substrate in this zone had large ranges for both chemical and physical properties, commonly found to have higher values for nutrient and mineral concentrations than the rest of the wetland area. The edges of this band nearest to the drainage channels was also characterised by the presence of invasive grasses and weeds, such as *G. maxima* and *C. arvensis*. This implies that the area has a dynamic hydrological regime, with frequent inundation events as well as high input of nutrients.
- The restiad bog zone in the central northern zone was characterised by dense patches of *E. robustum*, which dominated over the commonly found *G. dicarpa* understory found elsewhere. Taller woody plants such as *L. scoparium* were very infrequent in this area compared to other sections of the wetland. The substrate was newly formed ‘proto-peat’ (Figure 4.2), which implies that the area is predominantly rain-fed restiad peat bog.
- The fen zone was bog-like in terms of nutrients, resulting in a lack of invasive species encroaching into the area, but did not exhibit larger bog index values due to the absence of the primary bog species (*E. robustum* has not expanded that far south presently). It was therefore dominated by two main species, *G. dicarpa* as ground cover and *L. scoparium* as canopy, with *M. teretifolia* dispersed between these two.

- Vegetation indexing found that there is a transition in vegetation composition across the wetland class gradient. Species richness, Shannon-Wiener and evenness indices all showed high values for the mineralised and swamp sites, decreasing for the fen and then increasing again into the bog sites. This indicates that in areas of high nutrients, a large number of species that can tolerate dynamic hydrological regimes outcompete other wetland species, whereas only bog species can survive in the rain-fed bog conditions, with the fen zone acting as an intermediary zone with low nutrients but absent of bog species.
- *E. robustum*, the primary peat forming species in New Zealand, has spread across a wide range of the northern section, and is present in small patches south of the central drain. This natural spread of *Empodisma* is followed by a recovery in the wetland substrate, with lower nutrients and inorganic elements recorded even when in close proximity to a drain.
- Vegetation composition is directly linked to the hydrology and nutrient inputs along the drainage channels. The main risk to the wetland vegetation is a change in biodiversity likely caused by flood inundation, resulting in a reduction of restiad bog area.

Chapter 6. Hydrological processes and influences

6.1 Introduction

The defining feature of wetlands is their hydrological functioning, which creates ecological communities composed of many plant and animal species. Hydrology is also the key factor in the formation of wetlands, influencing the size and type of wetland (Chapter 2).

Across Otakairangi wetland, the differences in vegetation and nutrients are likely to be explained partly by varying hydrological processes. Anthropogenic drainage has altered the hydrological regime, due to both radial drainage surrounding the wetland, as well as the deep central drain effectively splitting the wetland in two. This has caused a dual ring effect around the wetland sections, with invasive and swamp species dominating near the edges of all the drains, and more natural wetland communities occurring in the central areas of the two main sections.

This chapter will describe the hydrological processes operating in the wetland. This includes records of the hydrological regime across the study period, the peat surface oscillation in an intact *Empodisma robustum* patch (movement of the peat surface as a response to water levels), and chemical analysis of wetland water. This section will also focus on the water table drawdown caused by the central drain, both in terms of depth and lateral extent.

6.2 Methodology

The hydrological processes and influences on the wetland were analysed using a variety of methods. These included:

- Monitoring the water table with pressure transducers along a transect
- Analysis of data collected from a Northland Regional Council rain gauge near to the wetland
- Measurement of peat surface oscillation
- Chemical analysis of water samples from drains and within the wetland

6.2.1 Water table monitoring

Pressure transducers (INW LevelScout) were installed along a transect crossing the central drain. This was conducted in two sections, with the drain and north-eastern section being installed in August 2017, and the south-western section in September 2018. These were utilised to determine the characteristics of the water table regime and the relative influence the central drain had on the wetland water table.

Seven transducers were installed along the north-eastern section, with the first being installed in the central drain as a reference for drain water levels, and the others being installed at distances of 20 m, 50 m, 100 m and 280 m from the drain. At the 280 m site, three transducers were installed with one kept above ground as a barometric logger for atmospheric pressure, one installed to measure the relative water table (suspended from the peat surface), while the other was used to measure the absolute water table (attached to a fixed reference pole).

Figure 6.1. Andrew Kirk downloading data from the INW LevelScout pressure transducer at site OT2 on 29th November, 2018.

Five additional transducers were later installed on 12th September 2018, with one further relative water level site at 290 m NE, and four on the south-western section to extend the transect into this area of the wetland. These transducers were installed at distances of 20 m, 75 m, 140 m and 280 m from the drain. However, due to technical issues occurring at OT 8 and 9 the datasets were found to be unusable, and with the late installation date for these sites the problem was not able to be solved before the conclusion of this research, and are therefore excluded from analysis.

The data these transducers provided were downloaded on a quarterly basis, with manual water table readings being undertaken to ensure data quality and identify if the instruments were drifting. Raw water level datasets were barometrically compensated using data from the barometric logger. Elevations of each site was surveyed using a Leica CS20 controller paired with a GS16 GNSS receiver, using RTK corrections provided by Smartfix to allow the water level data to be compared against each other with respect to reference datum (sea level). Rain datasets were collected from the Northland Regional Council rain gauge at Rowlands Road, approximately one km to the south-west of the wetland. The data was analysed for long term averages (30 years) as well for comparison of rainfall events to the wetland and drain water levels. Water level datasets utilised for this research were obtained between 6th October, 2017 and 11th January, 2019 (final download). The first month of measurements (to the 6th October, 2017) were unusable due to a faulty barometric probe.

6.2.2 Peat surface oscillation

Peat surface oscillation was measured using the paired transducers at the 280 m (north-east) site (OT5, OT6) (Figure 6.2). Absolute water levels (AWL) were determined through a reference rebar post inserted through into the peat down to the mineral soil (rebar total length 7.75 m), and surveyed using a Leica CS20 controller paired with a GS16 GNSS receiver. The submerged transducer was suspended from a cable attached to this reference post, and the water level readings were then calculated relative to the surveyed elevation. Surface elevation (SE) of the peat was determined by subtracting the AWL readings from the relative water level (RWL) (Fritz *et al.*, 2008). RWL refers to the unsaturated zone, which is the depth of the water table from the peat surface, and was measured by attaching a second submerged transducer to a wooden board that was set on the peat surface. This ensured the probe measured RWL (water table depth) as the peat surface oscillated (shrinkage and swelling).

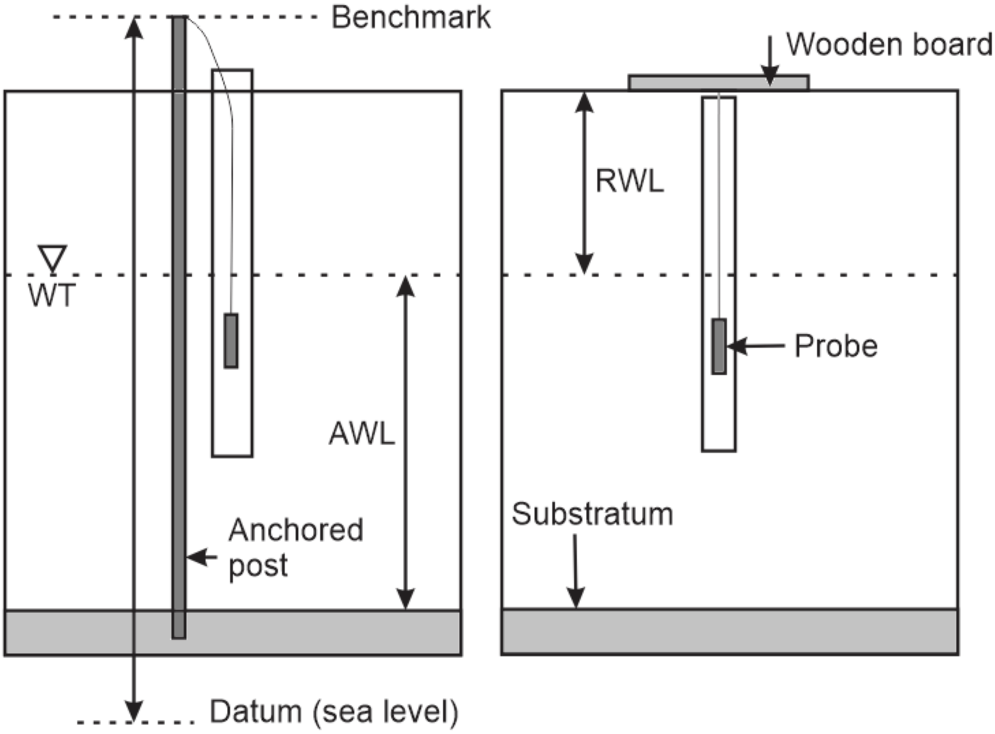

Figure 6.2. Diagram showing the layout of the automatic dip wells used to measure surface oscillation (adapted from Blyth (2011)). Absolute water level (AWL) was calculated by measuring water table fluctuations in relation to a fixed reference point (an anchored rebar post reaching to the substratum) surveyed to sea level. Relative water level (RWL) was calculated by measuring water table fluctuations in relation to a 'floating' wooden board which was attached to the peat surface. Surface elevation (SE) was calculated from the difference between these two measurements.

6.2.3 Water sampling and chemical analysis

Water samples were taken from the water table at each of the thirty plots, and from four locations along major drainage channels (Figure 3.3). These drain locations were chosen subjectively to give a representative sample of the water bordering and passing through the wetland area. These include two samples from the main central drain (one at the northern entrance and one 1250 m down the drain towards the southern exit of the wetland), one from the north-eastern drain skirting the wetland, and one from the south-western drain.

For plot water samples, the hole created from the first peat core being collected was excavated further and allowed to fill with groundwater. After sufficient time for disturbed sediments in the water to settle, 30 ml syringes were rinsed with water from the well twice before being used to extract between 20 to 30 ml of the surface water from the well. For all water samples, the water in the syringe was then filtered through 0.45 µm membrane filters into 50 ml Falcon tubes for transport (only 10ml was required, but between 15 ml to 20 ml was filtered). For some sites, the water was extremely turbid and required use of two or more filters to extract the required amount of water. The falcon tubes were then stored in the bins with peat samples until out of the wetland. Water samples were then processed for storage and transport by using a smaller 15 ml pipette syringe to move 9.8 ml of the filtered water into a 15ml falcon tube which had been prepared with 0.2 ml of concentrated nitric acid (HNO_3). Water samples were analysed for heavy metals and key nutrients using inductively coupled plasma mass spectrometry (ICP-MS) at Waikato University.

6.3 Results

6.3.1 Rainfall and water table regime

The seasonal pattern of 30-year monthly average rainfall showed that peak rainfall normally occurs in the winter period (May to August). The 2018 winter period involved higher rainfall during June, but lower than normal for May, July, and August. The summer period of 2018 had far above normal rainfall, with January to March experiencing rainfall between 124-233 mm when compared to the normal 78-106 mm (Figure 6.3). This was due to significant rainfall events that occurred throughout summer (including two ex-tropical cyclones), each of which caused increased water levels in the central drain and flooding into the wetland (Figure 6.3). While six months of the 2018 year were below average rainfall, the large rainfall events in February raised the annual rainfall value 28% higher than the 30-year average (Table 6.1).

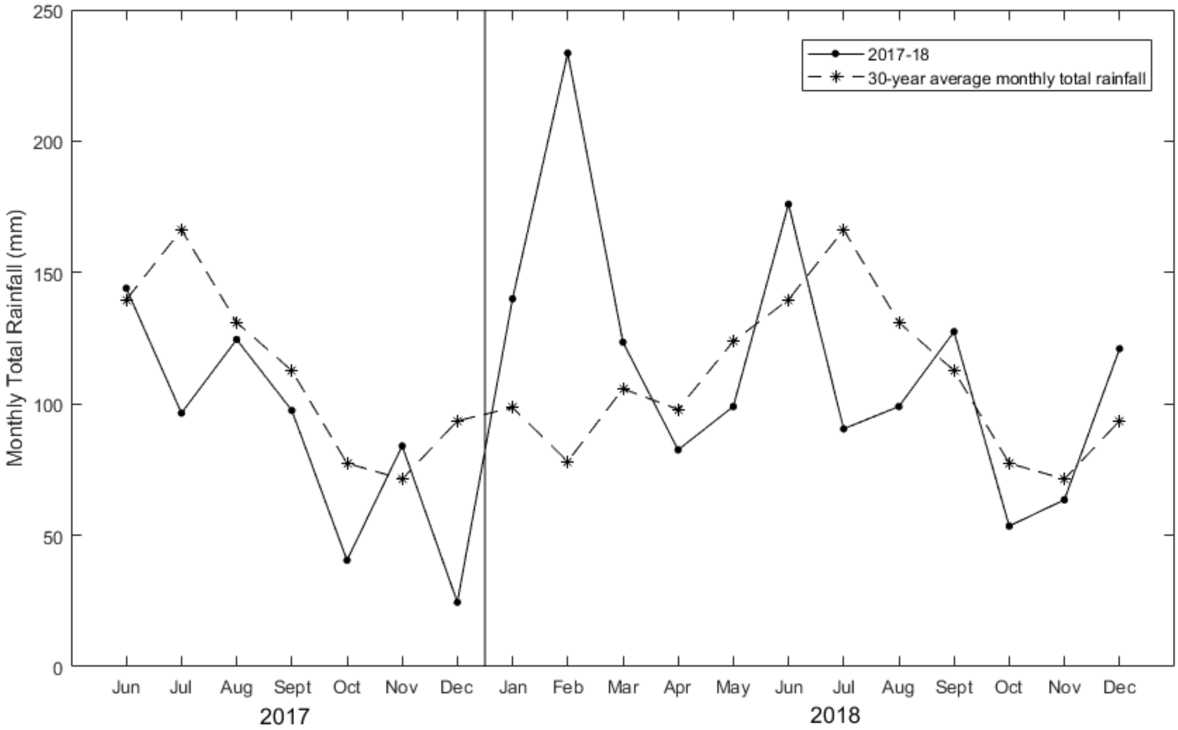

Figure 6.3. Monthly total rainfall for 2017-2018 and the 30-year monthly average (1986-2016) rainfall (for the respective months).

Table 6.1. Total cumulative yearly rainfall for 2017, 2018, and the 30-year average (1986-2016).

Timeframe	30 year average	2017	2018
Rainfall (mm)	1267	1188	1521

The central drainage ditch showed a classic stage hydrograph, with large peaks during rainfall events and reduced water levels during low rainfall periods (2.82 m range). While the wetland water tables were far more stable (ranges between 0.39 to 0.69 m) they showed the same response to rain as the drain hydrograph, to a lesser extent (Figure 6.4). The height of flood peaks indicate that surface inundation is experienced far into the wetland, with the drain water level exceeding and influencing the water table at OT 2 and OT 3 (20 and 50 m from the central drain) (Figures 6.4 and 6.5).

When wetland water levels were plotted against the drainage ditch levels, the inundation elevation is evident due to the response at each site changing from more or less horizontal to following the 1:1 line (Figure 6.5). This is also evident in Figure 6.6, where the water table regimes are plotted as box plots against the surface elevation. The drainage ditch (OT1) and the two sites closest to the drain (20 and 50 m sites) show surface inundation of the same height, implying that the drain water is flooding into these sites, while OT4 and 5 were not influenced by the drain water level. The 20 m site (closest site to the central drain) did not experience water table drawdown, indicating that water table drawdown caused by the central drain extends less than 20 m laterally from the drain banks (Figure 6.6 and 6.8).

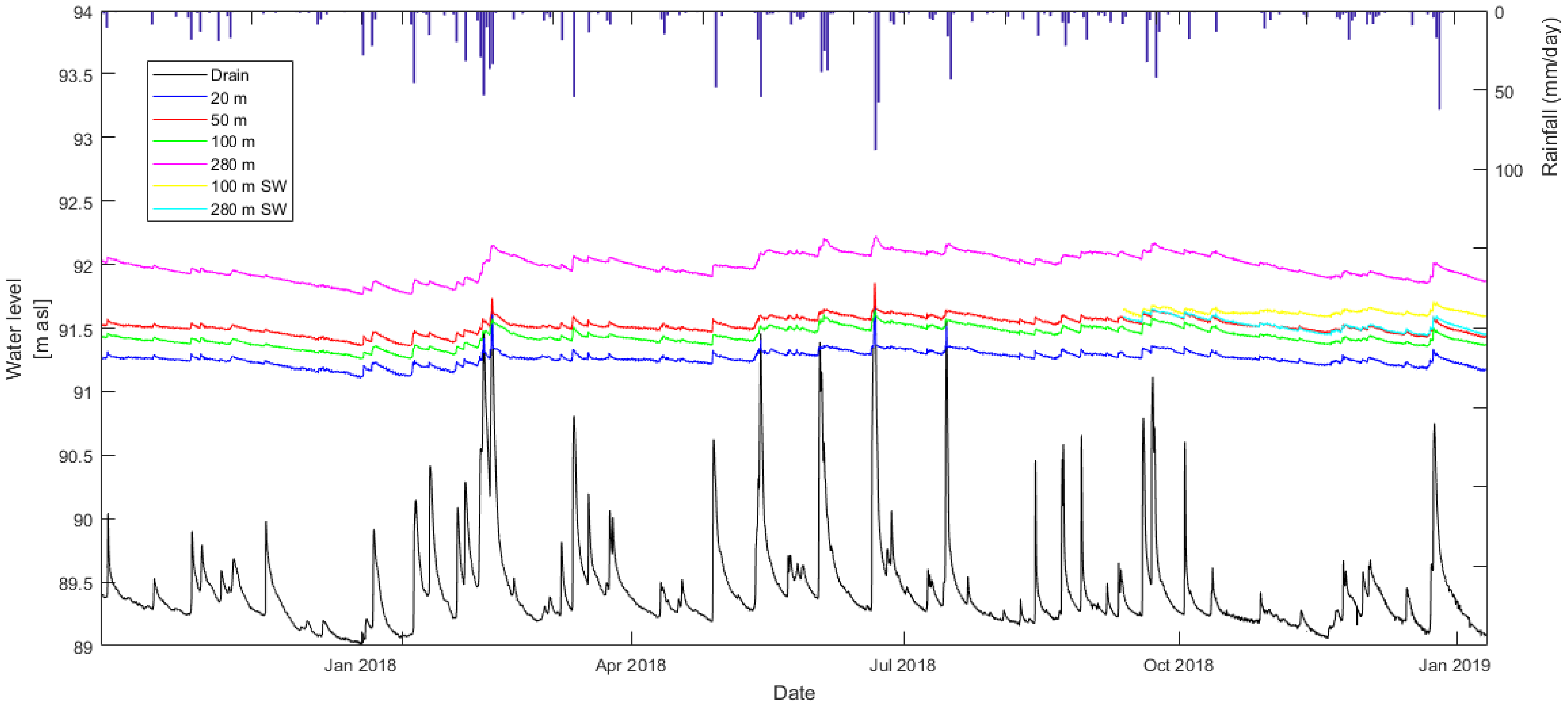

Figure 6.4. Water table regime across the wetland over the 15 month study period, with flood inundation from the drain visible during several high rainfall events (February, May-July 2018).

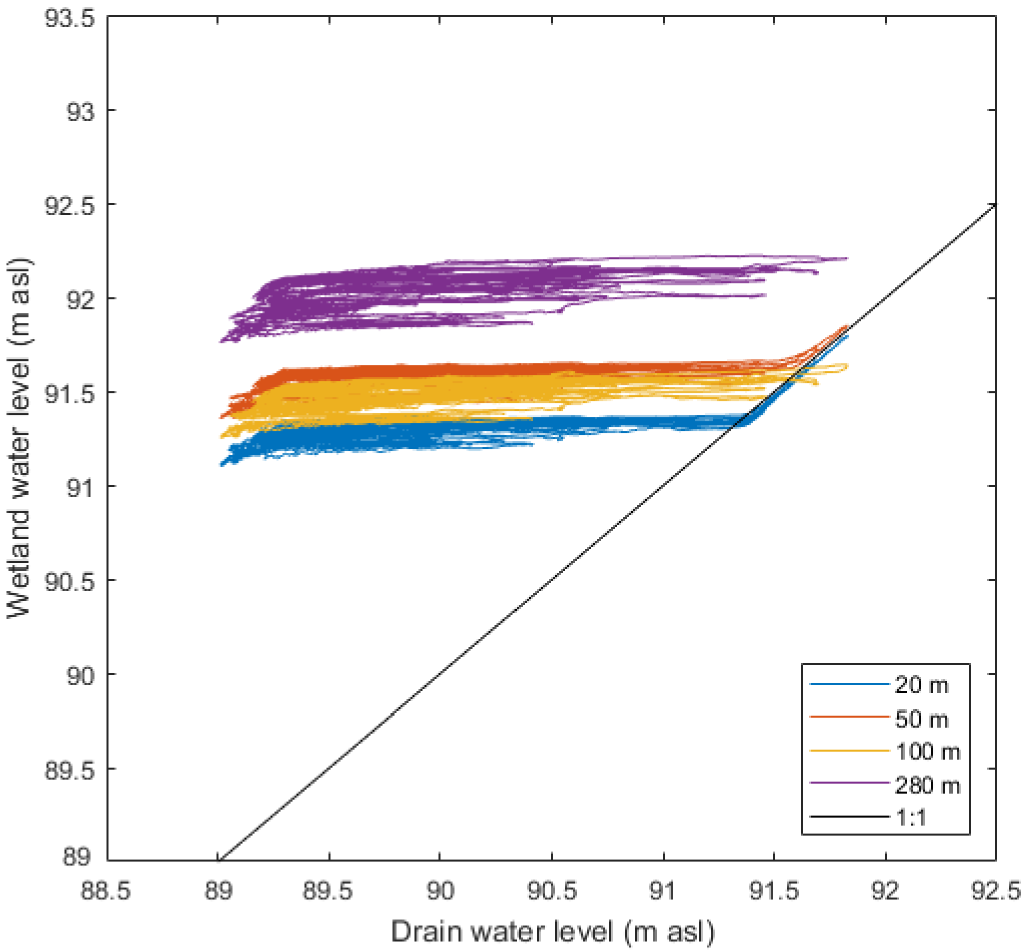

Figure 6.5. Wetland water tables plotted against drain water level. Water level response parallel to the 1:1 line indicates the wetland water table is being influenced by the drainage ditch water level, due to flood inundation.

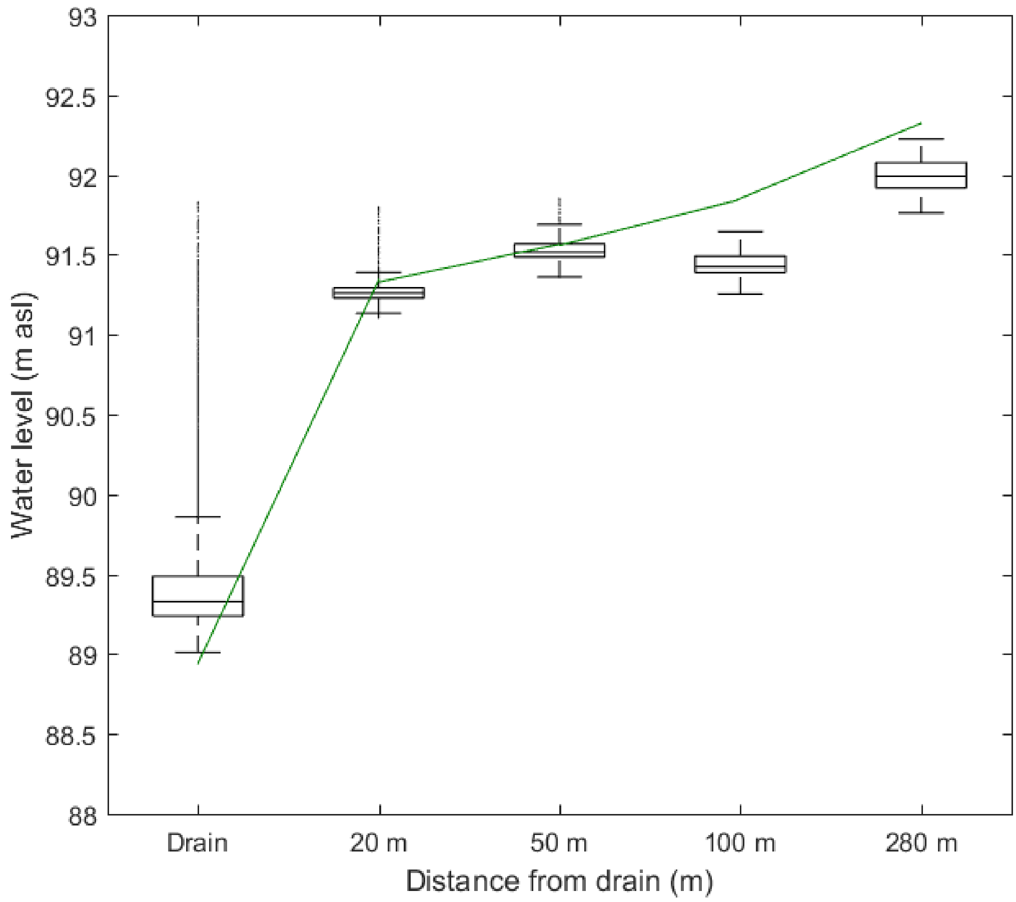

Figure 6.6. Box plots of water table regime by site, with outliers presented as dots. Green line indicates peat surface position relative to the water table. Note, the x-axis is not to scale.

The relative water levels show that surface inundation occurs at 20 m and 50 m from the drain, with water levels exceeding the surface by up to 0.4 m. 100 m and 280 m sites have lower water tables, and were not inundated throughout the study period (Figure 6.7).

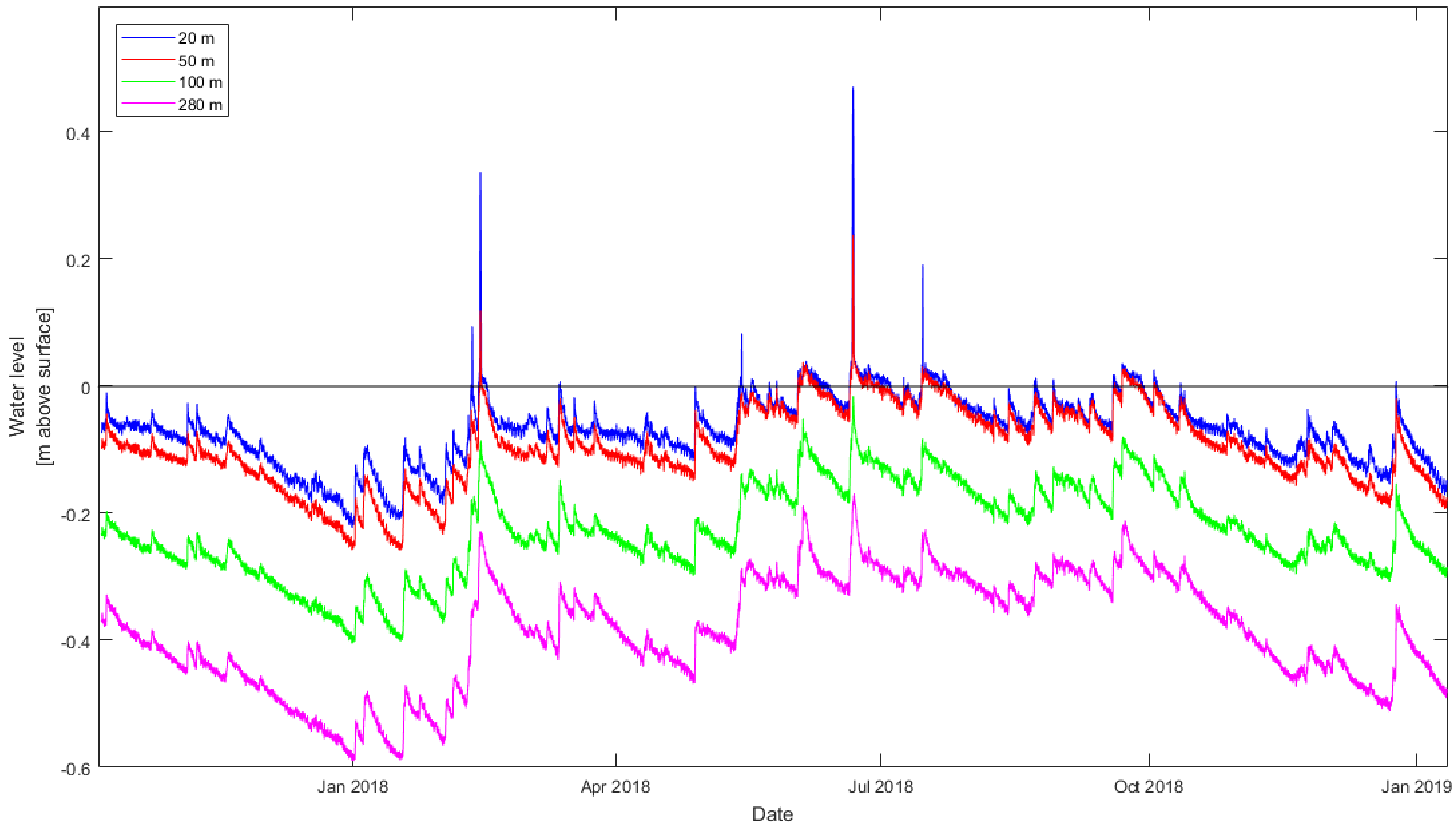

Figure 6.7. Relative water levels. Horizontal line indicates the surface of the peat substrate.

The water level range decreased with increasing distance from the drain, indicating decreasing effect from inundation and water table drawdown, and increasing internal regulation. The maximum range of the water level in the drain was 2.75 m due to extreme low and high flows, while 20 m from the drain it decreases to 0.69 m, and then to 0.49 m at 50 m from the drain (Figure 6.8). The average (Figure 6.8) and median water level (Figure 6.6) for the wetland sites (100 and 280 m) sit below ground level (indicating an unsaturated zone), but the sites 20 and 50 m from the drain experienced inundation events often (Figure 6.9). The 20 m site also experienced a smaller and shallower normal range compared to the 50 m site.

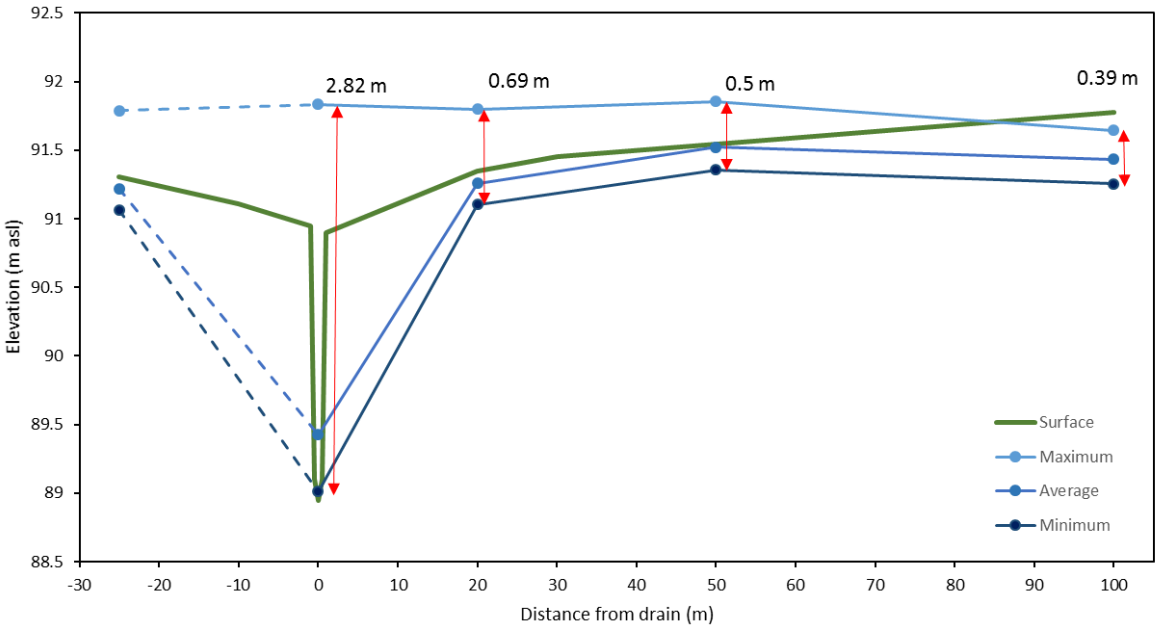

Figure 6.8. Maximum, minimum and average water level for sites within 100m from the central drain, with water level ranges given by numbers.

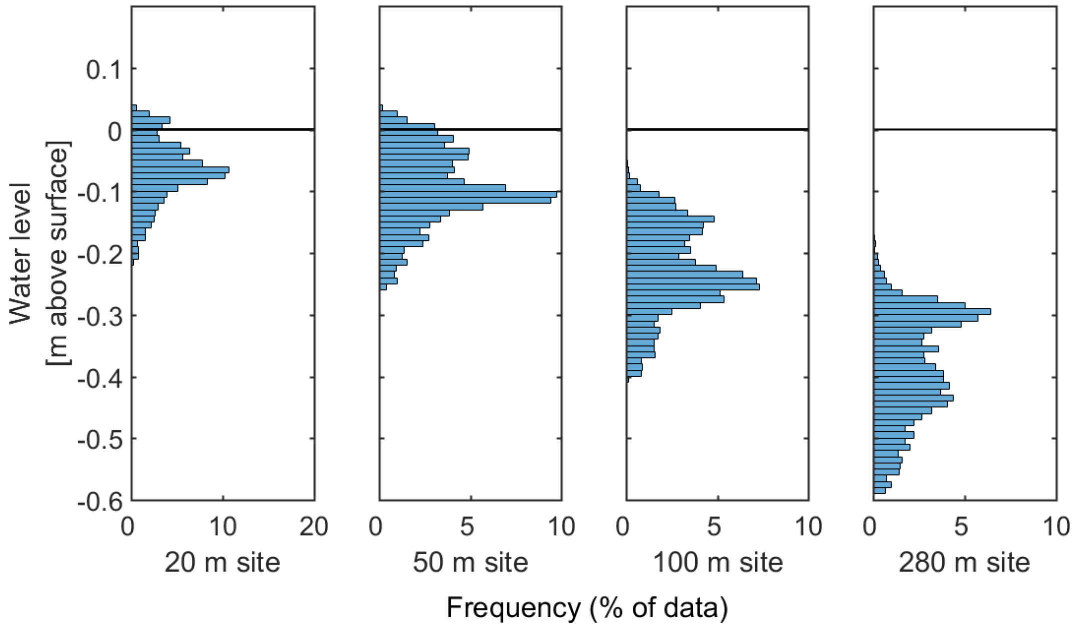

Figure 6.9. Frequency distribution of water table levels (excluding outlier values) relative to peat surface.

The water table regime of the 280 m site in Otakairangi sits between the other New Zealand bog water tables. Moanatuatua had a deep water table during 2015-2017 period, ranging between 200 and 950 mm deep, while Kopuatai had a shallow water table, ranging between 300 mm deep to 90 mm inundation (Figure 6.10). When split into individual years, the wetland water tables show different regimes, with the water table of Moanatuatua deepening during 2016, while Kopuatai was more frequently inundated during 2016 and 2017 (Figure 6.11).

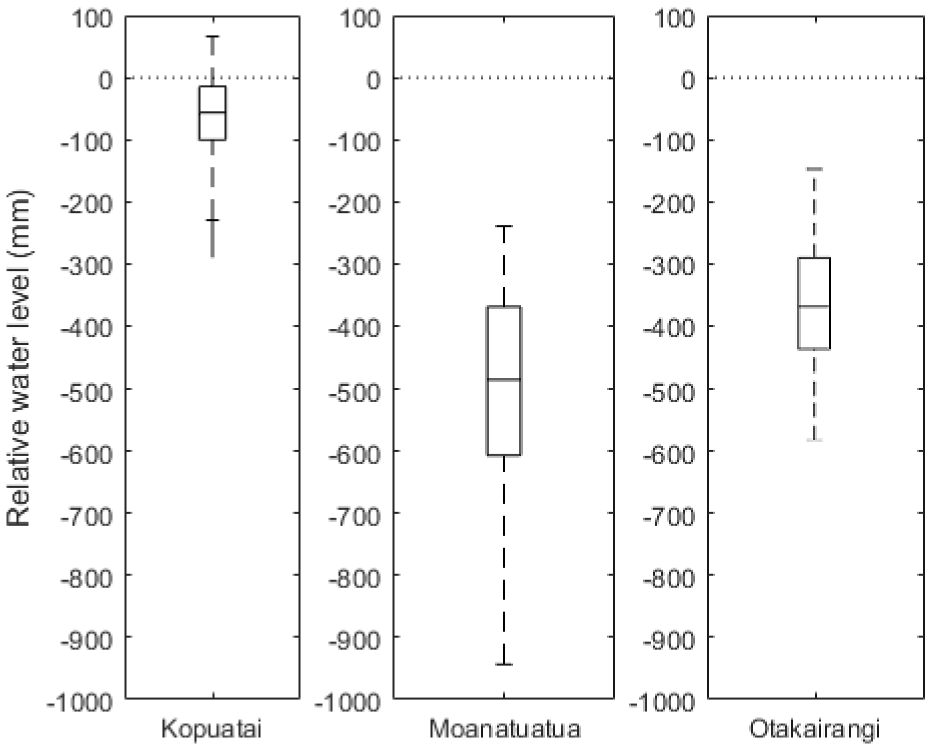

Figure 6.10. Comparison of New Zealand wetland 'bog' water tables against Otakairangi (280 m site). Wetlands used for comparison include Kopuatai, New Zealands largest (10,201 ha) unaltered raised bog located in the central Hauraki plains, and Moanatuatua, a degraded remnant peat bog located 17 km south of Hamilton (120 ha) (comparison data from January 1st, 2015 - December 31st, 2017, Otakairangi data from 6th October, 2017 - 11th January, 2019).

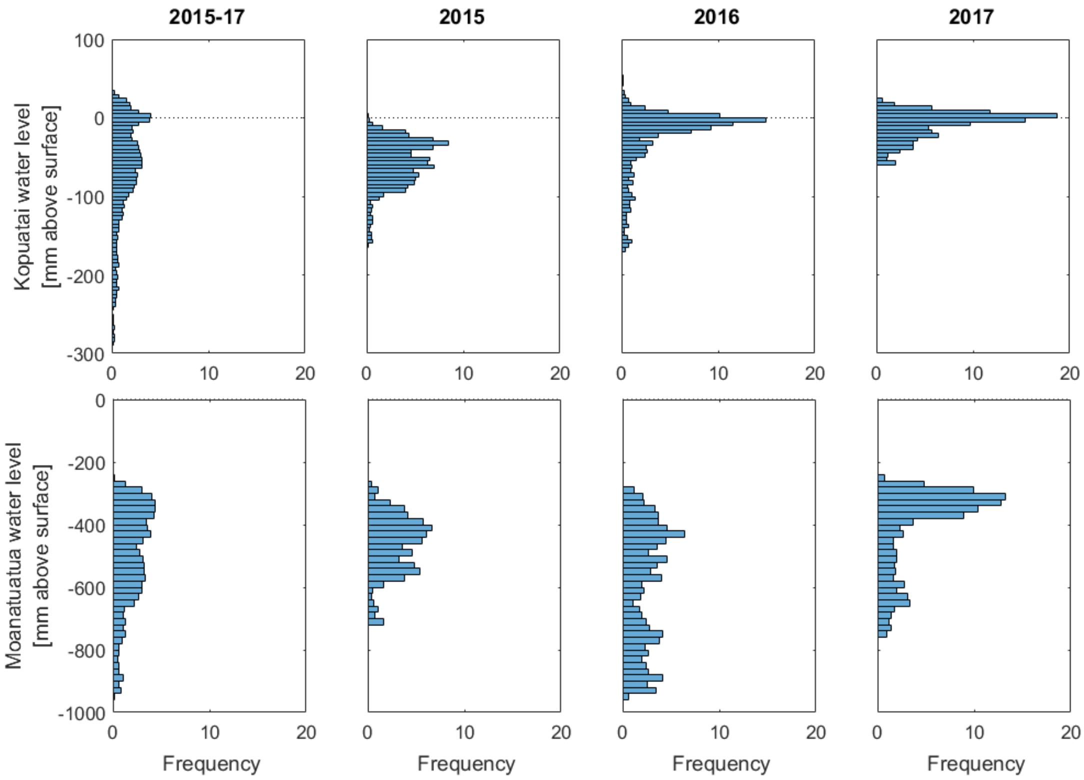

Figure 6.11. Water table depth frequency of Kopuatai and Moanatuatua wetlands over the 2015-17 period, expressing the variability of wetland water tables.

6.3.2 Peat surface oscillation

Water level measurements to determine PSO were obtained ~280 m from the drain using two pressure transducers that recorded measuring RWL and AWL. Following the method used by Fritz *et al.* (2008), the relationship between the peat surface elevation (SE) and the water level (AWL) was examined. Surface elevation responded to the water table with the minimum in early summer and the maximum elevation in late summer. The total range of surface elevation was 49 mm from October 2017 to January 2019 (*E. robustum*) and 70 mm from September 2018 to January 2019 (*G. dicarpa*) (Figure 6.12). The *E. robustum* dominated RWL site showed a deeper water table and greater hysteretic curves, while the *G. dicarpa* dominated site was much closer to the surface, implying that it is more likely to be inundated (Figure 6.13).

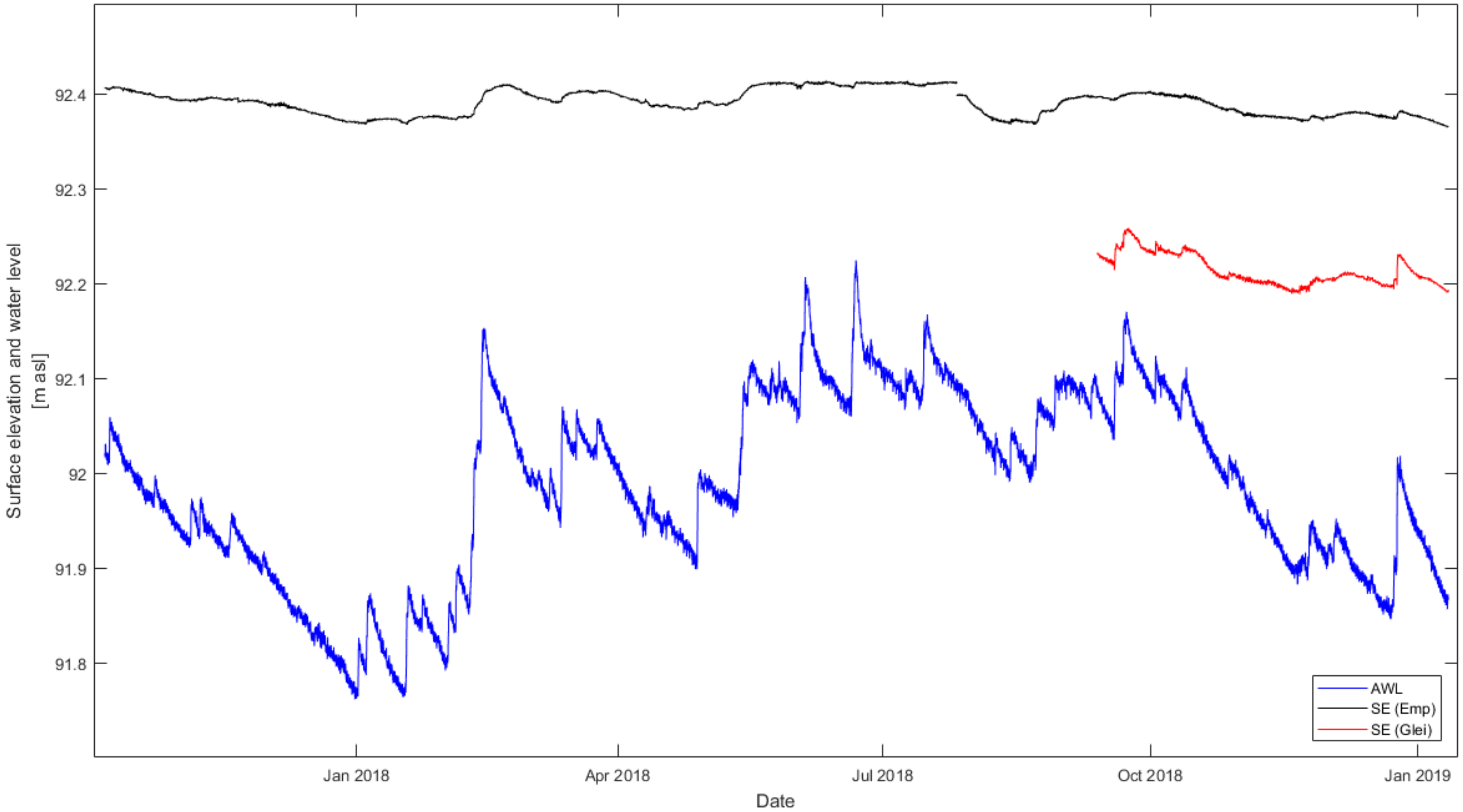

Figure 6.12. Surface elevation (SE) at sites OT5 and OT7 (*E. robustum* and *G. dicarpa* dominated plots), and the absolute water level. Vertical distance between the two SE lines and the AWL line is the relative water level or the thickness of the unsaturated zone.

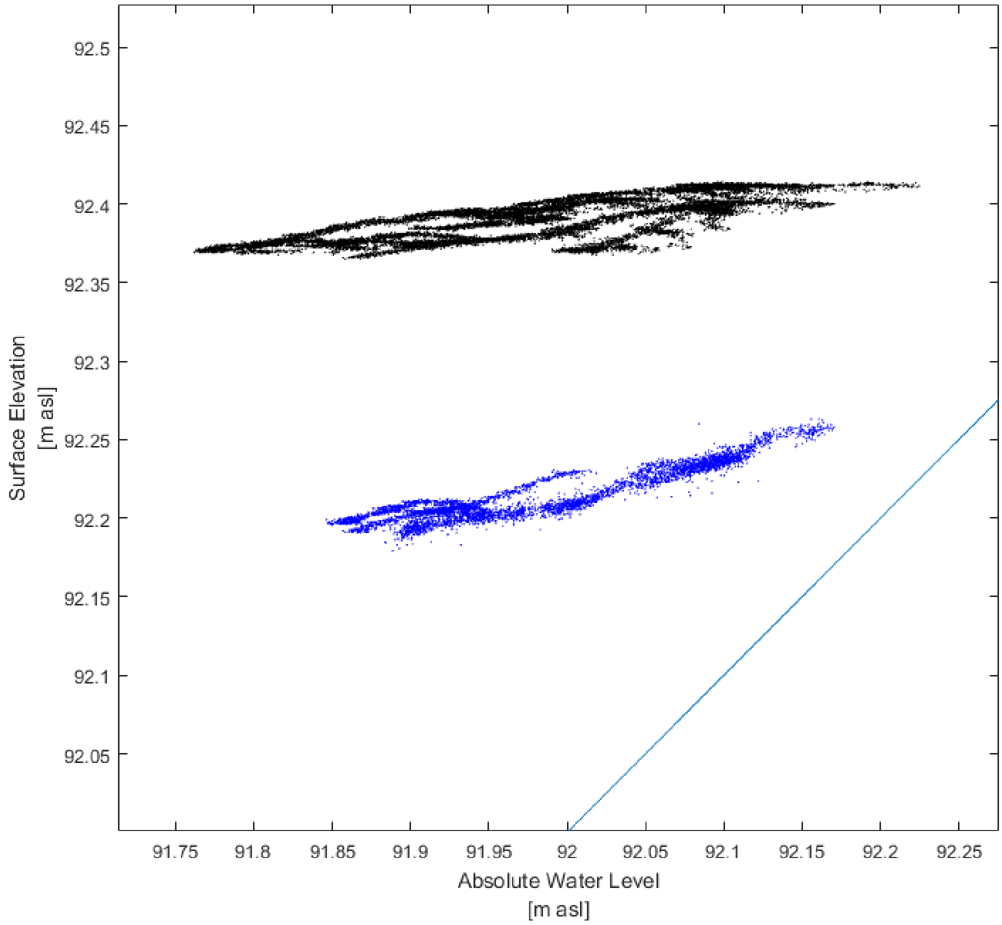

Figure 6.13. Surface elevation against absolute water level, showing linear relationships between surface elevation and absolute water level. Black dots indicate the relationship in an *E. robustum* dominated site, while blue indicates a *G. dicarpa* dominated site. Note that the time scales for each site is different (*E. robustum* from October 2017 to January 2019, *G. dicarpa* from September 2018 to January 2019).

6.3.3 Water sample analysis

Samples of the groundwater showed indicative patterns of metals and nutrients across the wetland. Samples collected near the drains and the native bush area generally had elevated levels of inorganic elements when compared to the inner wetland values. While the drains, the central wetland bog and the fen-bog areas showed natural background levels (or below) for Cd, the areas near drainage ditches and the northern bush (swamp) showed elevated levels above the national background levels, while the mineralised substrates had extremely high values compared to the rest (Figure 6.14a). Phosphorus levels in the drain samples were high, ranging from 61.5 to 84.5 µg /L (Figure 6.14b). As a result of this, the P levels in the plots nearest to the drainage ditches were also elevated (primarily swamp plots). Potassium showed elevated concentrations in drain, swamp, fen-bog and bog plots, while lower values were found in the mineralised and fen plots (Figure 6.14c).

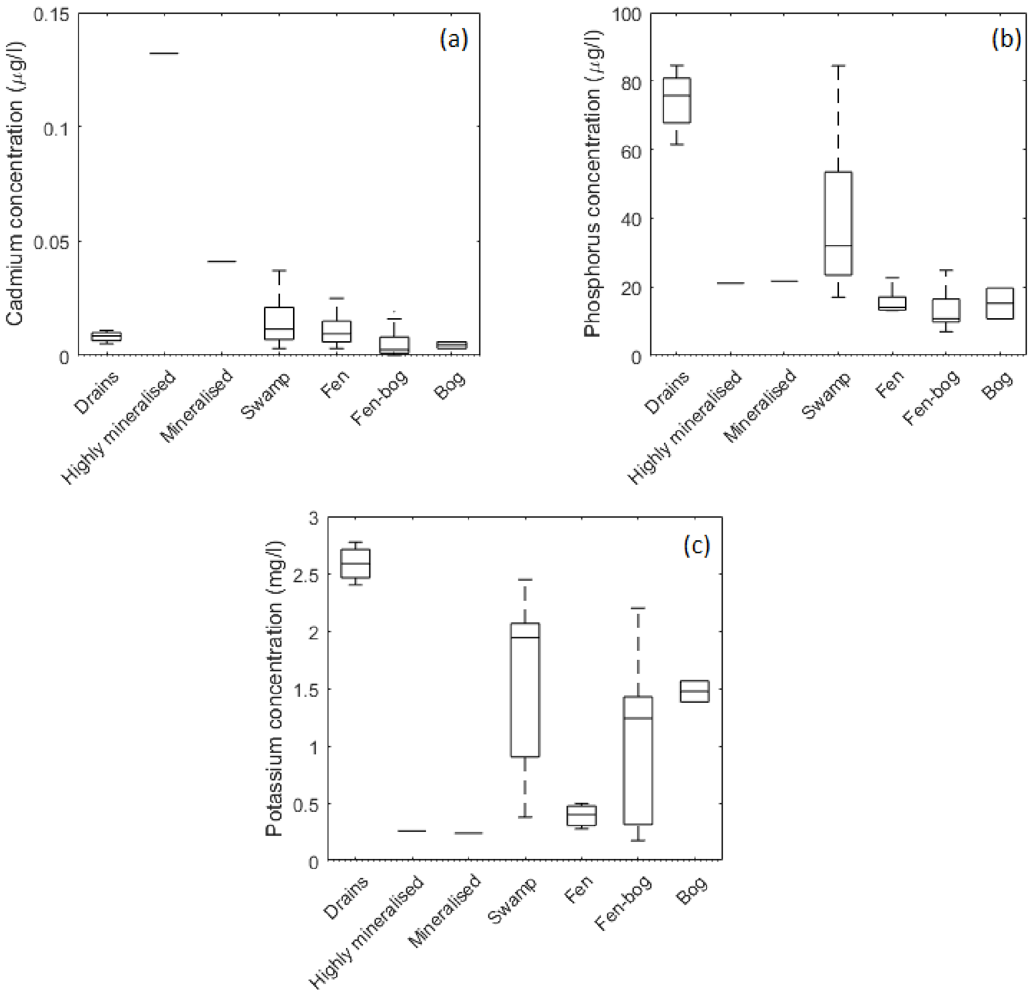

Figure 6.14. Concentrations of cadmium, phosphorus and potassium in water samples.

Copper and Zn showed the same pattern, with concentrations increasing from bog plots through to the mineralised plots (Figure 6.15a and b). Magnesium and Ca were found to have elevated concentrations in drain samples as well as swamp sites, with the other wetland groupings showing the same trend as Cu and Zn (decreasing from mineralised to bog plots) (Figure 6.15c and d). Aluminium (Al) and iron (Fe) were found in low concentrations in all drain samples, and were variable within the other groupings (Figure 6.15e and f). The fen grouping was consistently high for these metals.

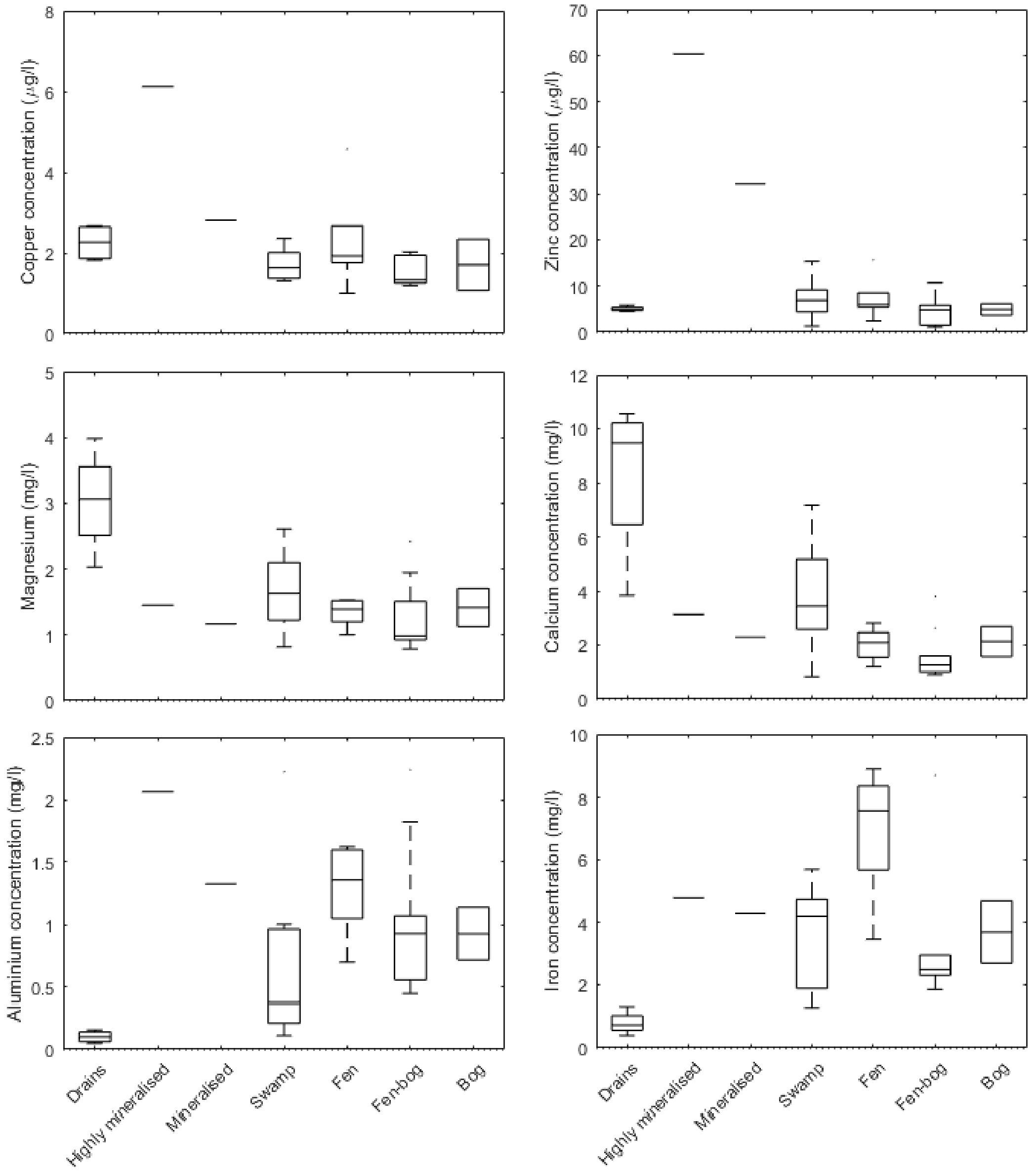

Figure 6.15. Concentrations of copper, zinc, magnesium, calcium, aluminium and iron in water samples.

6.4 Discussion

6.4.1 Water table regime and drainage drawdown effects

The summer period spanning 2017 and 2018 was the second wettest January on record, while February experienced 300% of the 30 year average rainfall for that month (Figure 6.3). This was caused by several tropical air masses over Northland during January, and rainfall from two ex-tropical cyclones (Fehi and Gita) affecting Northland during February 2018.

The high rainfall inputs were reflected in the water table regime measured across the wetland, with large spikes to the water table during the summer period (Figure

6.4). Other large rainfall events, such as those throughout early winter, also caused significant increases in the water table throughout the wetland. Some of these events also caused inundation from the drain that extended into the wetland area past the marginal banding, with evidence indicating that surface water made its way somewhere between the 50 m transducer and the 100 m transducer (Figures 6.5, 6.6, & 6.8). This was due to the flow capacity of the central drainage ditch being exceeded due to the large amount of precipitation falling in the upper catchment, resulting in the channelized flow through the wetland being ~0.8 m higher than the banks of the drainage channel. This overflow through the drain appears to have entered the upper areas of the wetland first, into the swamp-like area beneath the native bush slope and the areas on either side of the first km of the central drain. Central areas of the wetland where the vegetation composition changes from woody or swamp species to *Empodisma robustum* and *Gleichenia dicarpa* experienced higher absolute water tables, but lower relative water tables (Figures 6.4 and 6.7). This means that while the peat surface and water table are higher with increasing distance from drainage, the thickness of the unsaturated zone increased as well (Figure 6.7 & 6.8).

One of the primary issues with drainage channels within or near wetlands is the potential effect of water table drawdown (Price *et al*., 2016). The drain banks at OT1 were between 1.5 to 2 m tall, which indicates that a strong head gradient should be occurring between the higher wetland groundwater levels and the lower drain water levels, which would remove water from the wetland and increase the depth of the unsaturated zone. Water table drawdown is greatest in direct proximity to the drain (Price *et al*., 2016), but effects can be seen at much larger distances based on the structure of the peat, such as changes to vegetation up to 60 m away (Poulin *et al*., 1999; Landry & Rochefort, 2012). However, the 20 m site had an average water level far higher than that of the central drain, with the total difference between them being 1.83 m (Figures 6.6 and 6.8). This indicates that the drawdown effect of the central drain extends less than 20 m, less than originally anticipated.

Comparison between the inner wetland at Otakairangi and other bogs in the Waikato region using water tables can indicate the wetland condition. When compared to Kopuatai, New Zealands largest (10,201 ha) unaltered raised bog, the water table at Otakairangi was deeper with some overlap around 150 to 200 mm depth. Moanatuatua, a small degraded remnant peat bog south of Hamilton (120 ha) which is drained on all sides shows far lower water tables than Otakairangi. Kopuatai exhibited the lowest range (~375 mm), while Moanatuatua has a significantly larger range (~700 mm) when compared to Otakairangi (~450 mm) (Figure 6.10). This possibly indicates that Otakairangi currently resides in a state of degradation and recovery, exhibiting a mid-ground between an unaltered bog system and a heavily degraded remnant.

6.4.2 Peat surface oscillation

In Otakairangi, PSO was measured at 280 and 290 m sites, where the peat surface was found to oscillate by 49 mm and 70 mm over the duration of the study (Figure 6.12). This PSO is lower than what has previously been observed, such as (Fritz *et al*., 2008)(Fritz, 2007), who found that PSO ranged from 32 mm to 280 mm across 23 sites in Opuatia wetland, averaging at 149 mm, or Blyth (2011), who observed a maximum PSO range of 125 mm in Whangamarino wetland. The PSO observed in Otakairangi follows the type B relationship as described by (Fritz, 2007)(Fritz *et al*., 2008). While a type B PSO (at Opuatia wetland) implies that surface elevation changes by 32% of what absolute water level does (oscillation coefficient (OSC) = 0.32), and that the surface cover at the site is around 44% (16% standard deviation) restiad species (Fritz *et al*., 2008), it was found that surface elevation changed by 49 mm to the water level changes of 462 mm (OSC= 0.1), even though surface cover of the area was >50% restiad, and 70 mm (OSC = 0.15) for a zone that was <10 restiad. Whangamarino exhibited an OSC of 0.2 (AWL range of 620 mm to SE range of 125 mm) (Blyth, 2011). Unlike these previous studies which observed clear hysteresis in the wetting and drying response of the peat surface, where the surface decreased during summer following drying and increased through the winter season following re-wetting, only slight hysteresis was observed at Otakairangi. This can potentially be attributed to the lack of a distinct drying phase over the 2017-2018 summer period (Figures 6.4 & 6.12) due to the above average summer rainfall, causing the unsaturated zone to shrink rather than grow as it would normally do in a dry summer period. As surface oscillation shows a positive relationship to peat depth (Fritz *et al*., 2008), the reduced PSO may also be due to the depth of the current active peat layer in the area being shallow over the lower layer of heavily degraded peat, as the *E. robustum* community has not had enough time to build up a significant layer of new ‘proto-peat’.

Comparison between RWL sites of different vegetation compositions indicate that while the *E. robustum* dominated site did not exhibit the strong surface oscillation seen in other wetlands, it has a large unsaturated zone, reducing the potential inundation frequency when compared to the site dominated by *G. dicarpa*, where the water level was much closer to the surface (Figure 6.12 and 6.13). The relatively close oscillation coefficient between sites (0.1 and 0.15) indicate that the new peat layers are adding to the depth of the unsaturated zone but not yet increasing PSO, with the hydrology being determined by layers of underlying degraded peat. Stronger peat surface oscillation might occur in the area identified as exhibiting more bog-like characteristics (sites 23 and 24), whereas the current site is located in a fen-bog transitional zone (Figures 4.18 & 4.20).

6.4.3 Water chemistry

The chemistry of drain water and the groundwater found in the wetland are potential indicators of external inputs from flooding and transport of sediments. Some elements, such as Cd, P, and Ca are indicators of fertiliser inputs that are bound to sediment (Roberts, 2014), while Fe, Al, Cu and Zn are indicators of sediment movement into the wetland (Brown *et al*., 2015).

Cadmium levels in the wetland water samples indicate fertiliser based inputs as some areas of the wetland had concentrations above the expected atmospheric concentration of 0.01 μgL^{-1}. The drain samples were low in concentration, but this is potentially due to dilution in the large flows, and the wetland concentrations being higher due to long term accumulation. This includes the areas that are believed to experience frequent inundation events, such as Transect D which was highly mineralised, as well as the western side of the Transect A, and the start of Transect C below the northern hill slopes. This was shown in Figure 6.14a where the drain concentrations were low but the mineralised and swamp sites had higher values. This trend follows results found by Blyth *et al*. (2013), who found Cd concentrations within Whangamarino wetland increased towards the Whangamarino river, where more frequent inundation resulted in the transport of upper catchment fertilisers.

The P concentrations in the drain waters around the wetland areas were far higher than what would be in normal wetland water systems, primarily due to the application of P based fertilisers in the primarily agricultural catchment surrounding the wetland (ranging between 61.5 and 84.5 μgL^{-1}) (Figure 6.14). This P binds to sediment that erodes off the landscape, making its way into the water and is transported to the wetland area. This was seen in all of the drains where the P concentration was far higher than most of the wetland concentrations (Figure 6.14b), while higher concentrations were also seen in the water samples taken from the northern transect (swamp). This is potentially due to the shallow northern drain providing an access way for the sediment-bound P to easily flood into the northern section, as well as the steep hillslope north of the wetland providing a source of eroded sediments. Other sections that showed elevated P in groundwater samples include the western section of the Transect A, implying it is prone to more frequent flooding and transport of P that the eastern section, as well as the beginning of Transect B. Transect E, crossing the southern section of the central drain, showed low P levels implying that the southern section of the drain is inundated less frequently than Transect A. These trends shown in ground water samples were also seen in the P concentrations of peat samples (Figure 4.9).

Potassium in water samples was highest in the drain samples, ranging between 2.4 to 2.77 mgL^{-1}. This was followed by high values in both swamp, fen-bog, and bog zones, while the mineralised and fen zones had low concentrations (Figure 6.14c). This contrasted to the K concentrations found in peat samples, where higher values were found in the mineralised and swamp sites (Figure 4.14). However, for both ground water and peat samples the high concentrations were isolated to the north-east section of the wetland, with values on the south-west side of the central drain being lower. Copper and Zn showed a trend of decreasing concentrations from the mineralised plots to the bog sites, with the drain samples having low concentrations similar to fen and fen-bog zones (Figure 6.15a and b). This pattern was matched by the peat sample concentrations (Figure 4.15), as both

Cu and Zn have an ionic structure which is attracted to the negatively charged soil particles (Brown *et al*., 2015), and are likely spread through the wetland following inundation events. Magnesium and Ca showed the same trend but with higher concentrations in drain samples than the wetland samples (Figure 6.15c and d). Calcium concentrations were high in the drains (ranging from 3.8 to 10.6 mgL^{-1}), with the swamp zone also having high values, primarily due to the sites found along Transect C, similar to concentrations found in peat samples (Figure 4.14). High concentrations of Al were found in the mineralised, fen-bog and fen sites (Figure 6.15e), which was dissimilar to what was found in peat samples (high for mineralised and swamp sites, low for fen, fen-bog and bog sites)(Figure 4.16), which could be potentially attributed to leaching (Similia *et al*., 2014). High Fe concentrations were found in the fen sites (Figure 6.15f). While no Fe seeps (chalybeate springs) were visible, the increased Fe concentration found in the southern section may be caused by localised groundwater seepage containing dissolved Fe discharges (Tiner, 2016). This was also seen in peat sample analysis, where sites 7 and 11 had higher Fe concentrations than most of the wetland (excluding the two mineralised sites) (Figure 4.16).

6.5 Summary

- The drain exhibits a classic stage hydrograph similar to a river or stream, with large peaks in water level during heavy or extended rain events. The maximum range of water levels exhibited by the central drain was 2.82 m.
- The central drain flooded several times throughout the study period, with the water level rising above the drain banks. This caused inundation of adjacent parts of the wetland, with the extent of inundation being determined by the height of the drain water level. The largest flood event caused inundation between 50 to 100 m into the wetland (north-east).
- The central drain did not cause significant water table drawdown on the north-east side 20 m from the drain. The total difference between the average drain and 20 m site water levels was 1.83 m, indicating that the drawdown effect of the central drain extends less than 20 m.
- The 20 m and 50 m sites were frequently inundated, while the 100 m and 280 m sites were never inundated. The relative water table depth

(unsaturated zone) increased with increasing distance from the central drain.

- The water table regime at the 280 m site in Otakairangi sits between those of Moanatuatua (degraded remnant peat bog) and Kopuatai (New Zealand's largest unaltered raised peat bog). Moanatuatua has deep water tables and a large unsaturated zone, whereas Kopuatai has a high water table that periodically inundates the surface. Otakairangi has a shallower water table than Moanatuatua, but deeper than Kopuatai.
- The peat surface oscillation (PSO) observed in Otakairangi was smaller than in untouched natural peat bog systems, with the surface oscillating only 49 and 70mm (OSC = 0.1 and 0.15). Higher oscillation values may be observed in the locations identified as exhibiting true peat bog characteristics.
- The PSO observed in the patch of *Gleichenia dicarpa* was similar to that of the *E. robustum*, but was far shallower.
- Phosphorus values in drain water samples were high (61.5 to 84.5 μgL^{-1}), followed by a large range in the swamp concentrations (primarily the northern border drain transect). This is likely due to flood inundation bringing in phosphorus.
- Cadmium, copper, and zinc showed low concentration in the drains and high concentrations in the mineralised sites, decreasing through to the bog-like sites, indicating transport by sediment and fertiliser during flood events. Increased Fe concentrations in the southern section may indicate groundwater seepage.

Chapter 7. Discussion, conclusions and recommendations

7.1 Introduction

This study of Otakairangi wetland involved investigation of peat substrate physio-chemical properties, vegetation composition, and the hydrological regime of the central drainage channel and adjacent wetland area. The characterisation of the ecohydrological function required linking these components together which give an understanding of the current state of degradation, recovery, and the major influencing factors which contribute to the present status.

7.2 Wetland classes and zonation

Previous chapters described the physical and chemical characteristics of the peat substrate, and how this allowed zoning of Otakairangi into different classes (Figure 4.17) based on previous research (Clarkson *et al*., 2004b; Browne, 2005; Blyth, 2011), as well as vegetation composition across the wetland (Figure 5.9). Otakairangi consists of a range of wetland types including swamp, fen and young bog, each having corresponding water regimes, vegetation communities, and peat physical and chemical characteristics.

Swampland was identified as being present in a marginal fringe band of varying widths around the border of the wetland and along most of the central drain. This extended between 10-30 m from border drainage for many areas (such as southern borders and the lower half of the central drain), but also extended much further in other areas, such as beside the northern entrance of the central drain, where it extended several hundreds of metres into the wetland area. The study sites that were included in this zone include the southern side of the central drain (plots 1-3) and most of the sites in the northern band (plots 13-21). This area was dominated by large vegetation species, such as *P. tenax* and *C. tenuicaulis,* and was also frequented by invasive species such as *U. europaeus* and *G. maxima* were also present. Nutrient concentrations in the swampland areas were consistent with those found in other New Zealand swamps (Table 7.1). The primary factor influencing this zone is likely to be frequent inputs of surficial water during flood events, which deposits suspended sediment and nutrients. The nutrients and sediment in the water are likely products of the surrounding catchment agricultural practices, with erosion of pasture and channel banks increasing the amount of sediment entering the waterways, or directly by cattle entering the fringe margins (northern drain).

Table 7.1 extends Table 2.1 with the addition of measurements taken from Otakairangi wetland in zones identified as restiad bog (bog and fen/bog transitional) and swampland (swamp, mineralised and highly mineralised) (Chapter 4). Transect B which measured the impact of the southern drain was termed fen, and for the purposes of this table the sites from this zone have been excluded due to their unique characteristics.

Table 7.1. Comparison of key wetland substrate parameters as swamp and bog means and ranges (in brackets). New Zealand data obtained from 17 swamps and six bogs (Clarkson *et al*., 2004b), Opuatia wetland from six swamp and 22 bog sites (Browne, 2005), Whangamarino wetland from 11 swamp and 12 bog sites (Blyth, 2011), and Otakairangi from 12 swamp and 12 bog sites. TN= total nitrogen, TC= total carbon, TP= total phosphorus.

	New Zealand		Opuatia		Whangamarino		Otakairangi	
	Bogs	Swamps	Bogs	Swamps	Bogs	Swamps	Bogs	Swamps
Soil pH	4.0 (3.7-4.4)	5.2 (4.1-5.9)	5.0 (4.3-5.3)	5.05 (4.8-5.4)	4.0 (3.5-4.3)	5.14 (4.3-6.4)	4.1 (3.8-4.4)	4.86 (4.3-5.5)
TC (mg cm3)	92.7 (24.1-239.8)	39.8 (5.2-100.6)	33.3 (24.2-42.29)	37.8 (29.8-47.4)	26.9 (18.1-41.4)	36.5 (20.0-52.6)	26.52 (17.50-34.69)	36.5 (25.9-52.4)
TN (mg cm3)	0.82 (0.02-1.83)	2.12 (1.15-3.24)	1.35 (0.7-1.98)	2.4 (1.7-2.8)	0.76 (0.37-1.57)	2.5 (1.5-4.0)	1.06 (0.51-1.55)	1.96 (1.32-2.65)
TP (mg cm3)	0.08 (0.01-0.20)	0.28 (0.15-0.69)	0.08 (0.03-0.13)	0.26 (0.18-0.33)	0.02 (0.01-0.04)	0.14 (0.03-0.21)	0.02 (0.01-0.05)	0.08 (0.05-0.19)
C:N	48.5 (35.9-79.7)	18.0 (14.2-30.6)	26.4 (17.0-49.0)	16.7 (14.0-19.0)	37.8 (26.4-48.7)	14.9 (11.3-21.6)	27.3 (18.8-53.1)	19.1 (14.3-24.4)
C:P	1904 (533-4221)	163 (45-435)	507.1 (236.9-1041.8)	161.4 (116.3-212.7)	2022.4 (947.4-2971.5)	332.8 (146.2-920.3)	1485.3 (503.5-3139.8)	476.6 (194.9-778.8)
N:P	39.0 (20.6-81.6)	9.1 (4.0-20.6)	18.8 (13.7-27.3)	9.54 (8.3-11.7)	52.4 (35.9-70.1)	20.7 (12.9-43.2)	53.4 (19.3-73.4)	24.8 (13.6-32.5)

The majority of the wetland area on the northern side of the central drain which lies beyond the swamp marginal bands was classified as either bog or fen-bog transitional. These zones were located beyond the commonly inundated swampland areas, marking a change from mixed water inputs of surficial flows and rain to a primarily rain dominated system. The true bog sites (plots 23 and 24) were located in the central area of the north-eastern wetland section, surrounded by a fen-bog transition zone. The bog sites exhibited little degradation of the surficial peat layer, with low von Post readings and bulk density values, while groundwater pH was consistent with other wetlands. These two plots had the lowest nutrient values and the highest nutrient ratios of the wetland, and although the C values were lower than what might be expected, they still fell within the New Zealand bog range. While the N values were consistent with the New Zealand bog average, the P values were low for New Zealand conditions, which caused the C:P ratio to be higher than the New Zealand average. In the bog area, the dominant vegetation was *E. robustum,* with *G. dicarpa* and *M. teretifolia* also being found occasionally. The presence of newly formed peat from *E. robustum* roots and a low decomposition state indicate that this section is a raised peat bog. This is supported by the low nutrient, rain-fed condition, as well as a high water table consistent with bogs.

The fen-bog transition zone was distinguished from the bog sites by having increased degradation (higher von Post and pH), as well as presumed inputs of surficial water which caused higher nutrient concentrations, lower nutrient ratios, and increased inorganic element concentrations (lower central drain, eastern border drain). This zone was mostly dominated by the same vegetation species as the bog sites, but with *E. robustum* having less cover and other species such as *G. dicarpa, M. teretifolia* and *L. scoparium* becoming more common. This zone includes the area around the bog zone (plots 4-5, 22) as well as the area around the southern central drain (plots 25-30). It is plausible that *E. robustum* was left in a small patch in the north-east section (bog zone) following burning events in the past, and is now expanding out across the wetland (primarily fen-bog zone).

The southern area of the wetland above the southern border drain was classified as fen, as it had indicator values intermediate between the bog and swamp sites, similar to the fen-bog transitional zone. However, it was distinguished from the fen-bog zone as it had different nutrient and inorganic element concentrations, such as higher C values. This zone also exhibited a different vegetation composition, with the main two vegetation species in the area being *G. dicarpa* as ground cover and *L. scoparium* as canopy cover. The manuka canopy cover appears to be rapidly collapsing, with standing dead trunks emerging from the ground cover frequently, which is believed to be attributed to a group colonisation following a historical burn event that has now reached the end of its life span. The absence of groundcover other than *G. dicarpa* may be caused by the manuka canopy previously shading out the competition, or potentially due to the area being unsuitable for many invasive species (low in nutrients). However, this has given rise to a large area of the wetland being suitable for future colonisation by *E. robustum*.

7.3 Impacts and disturbance of artificial modification

7.3.1 Central drainage channel

Clearly visible disturbance to the wetland which could be mitigated or removed, the central drain does not have as great an effect as previously thought. Initial concepts of drainage disturbance included water table drawdown which potentially extended far into the wetland, and a pathway for nutrients to be moved into the wetland, changing the ecological balance. Analysis of water levels in the drain and in the wetland 20 m NE shows that there is no significant water table drawdown past 20 m, potentially due to degradation and compression of peat substrate along the margins of the drain creating a semi-impermeable barrier.

While the central drain provides the primary passage of all water exiting the upper catchment, it also periodically floods during heavy rainfall events. This allows water to move further into the wetland than it would normally move, and as such provides increased nutrients and inorganic elements to the wetland. This was seen in the northern section of the central drain (plots 1-3, 19-21) where it floods frequently and a change in vegetation community and substrate composition was observed. This may also be attributed to sediment being washed down from the native bush slope and into the drainage channel, not solely from agricultural inputs.

The drain has created a swamp belt directly through the centre of the wetland area, which differs in width. In the lower section of the central drain, this margin is 10-30 m wide, whereas in the upper sections of the drain the margin is significantly wider, extending over 200 m in some places. This is presumed to be caused by frequent and heavy inundation of the upper central drain during storm events, as it is the first area where the drain banks overtop. The lower section does not experience the same inundation as the excess water has already been diffused across the upper section.

With the current data available it is hard to identify an accurate long term flood regime and determine the extent of flood events across the wetland, as the data collected over the period of this study is based on chemical analysis of substrate and water table regime over a set transect line. However, because the vegetation clearly transitions from flood tolerant species to those which are more suited to low fertility conditions (as explained above), use of vegetation as an indicator for flood extent is possible.

Figure 7.1. The central drain following a rain event, with Ben Herbert clearing the pressure transducer system of debris (12th September, 2018).

7.3.2 Northern drainage channel

The northern drainage channel that extends from the central drain entrance across the northern border of Otakairangi lies at the base of pasture hillslopes. This drain, unlike the southern and central drainage channels, is shallow and carries only a small quantity of water. While this would normally be viewed in a positive aspect as it is not causing water table drawdown (a primary issue in wetlands worldwide), it is creating a conduit for runoff (sediment and nutrients from the hillslope pasture) to directly enter the wetland.

Deeply cut drainage channels have high banks which provide a barrier preventing small flood events from inundating the wetland area, whereas the shallow northern drain does not. While it has not mineralised the substrate as much as it has near the entrance of the central drain, the presumed frequent flooding of the northern section appears to have extended the marginal swamp belt further into the wetland area than in any other area of the wetland.

This high fertility swamp zone appears to be encroaching upon the restiad bog zone within the northern section of Otakairangi wetland. Invasive vegetation species that are commonly found in more fertile ecosystems are found scattered across this zone, including *U. europaeus.* Larger swamp species which are generally found in the hydrologically dynamic margins are found far into the wetland area, such as *P. tenax.* However, as this area of the wetland remnant sits below the hillslope, it may have historically been a large swampland zone, with shallow mineralised peat layers characteristic of swampland.

Figure 7.2. The northern drain at the base of the pasture hillslopes.

This study indicates that the periodic flooding of the wetland is a primary concern for wetland health and condition, and therefore has to be highly controlled. Flooding provides a conduit for nutrients and sediment to enter the wetland, which can alter the wetland condition by reducing the current extend of the ombrotrophic restiad peatland and increasing the extent of the marginal minerotrophic swampland, a primary vector for invasive weeds.

7.4 Recommendations

Holden *et al*. (2004) raised concern over wetland restoration, questioning what the end destination of restoration efforts are, both in terms of hydrology and ecology. As the climate today is different from that when many peatlands initially formed, a peatland restored in today's climate may well develop along an entirely different trajectory than what the original peatlands did thousands of years ago, and therefore raises the question: should restoration targets aim to re-establish pre-human ecosystems, or simply to improve upon the current state? Development in new directions may be unavoidable, so potential aims should include the preservation or enhancement of the environmental functioning of the wetland rather than trying to recreate past ecosystems. This can be done through internal or external efforts, in what Zalewski (2015) termed dual regulation. Restoration plans should use knowledge of the present state of the site to define objectives, assess the feasibility of each objective, outline the processes to be applied, and define how the efforts and impacts will be monitored (Similia *et al.*, 2014).

Zalewski (2015) suggested that instead of trying to balance the social and economic priorities of catchment management with ecosystem attributes, a preferable alternative would be to focus on ecosystem potential. Rather than focusing on the protection of pristine, undisturbed ecosystems, regulating the physical processes in novel ecosystems could increase their ecological potential (water resource value, biodiversity, ecosystem services, and resilience to stress). The new focus should be on the use of catchment scale dual regulation of the hydrology and biology to assist conservation and restoration. However, the complexity of this task will require multi-disciplinary input and communication between different specialisations (such as ecologists and engineers), as the elimination of threats and human assisted recovery of the ecosystem cannot be solved by a single scientific field.

External threats to Otakairangi wetland include water table drawdown by drainage both around and within, the increased inputs of nutrients from the drains into the wetland, and the invasion of external plant species and pest animals. Internally, the major threats are decreases in the diversity of native wetland species and the encroachment of the exotic border vegetation further into the wetland. Degradation of peat and reductions in peat formation are also major threats to the internal functioning of the wetland. Fire is both an external and internal threat. While a natural driver of ecosystem change in large wetlands, it can result in negative change in smaller remnant wetlands, such as the possibility of removing the current areas of *E. robustum* expansion within Otakairangi wetland.

For a restoration strategy to succeed, it must be based on two fundamental objectives: the elimination of threats, and the amplification of opportunities that guarantee reaching the goal (Zalewski, 2015). For Otakairangi wetland, this research has outlined the major threats to the wetland which can be mitigated or managed in some shape or form. However, these all have different costs and potential benefits, and as such they should be chosen based on cost-benefit analysis, as well as the likelihood of success. Dual regulation of wetland and catchment hydrology and biology is crucial to improving the ecosystem potential of the target area.

7.4.1 Internal restoration and management

Recovery to pre-disturbance vegetation communities can be successful once protective microclimates have been established, either artificially or naturally (Clarkson *et al.*, 2017). As restoration has been equated to accelerated succession (Van der Valk, 1998), to protect the wetland area and restore it to natural bog conditions, the natural spread of *E. robustum* could be enhanced by human assisted activities. Clarkson *et al*. (2017) described restoration efforts in both New Zealand and Australia, stating that patchwork transplants of wetland species shows increased vegetation recovery on burnt and cut sites. The current wetland area is dominated by a ground cover of *G. dicarpa*, with scattered patches of *E. robustum* that have presumably resulted from natural dispersal processes. By assisting the spread of *E. robustum* across the wetland area, it could increase the natural ecosystem succession process and allow for transition towards peat bogs. This includes increasing the rate of peat accumulation, raising the water table and bog surface elevation, and sequestering C in organic matter. This process could be conducted by the patchwork spread of *E. robustum* seedlings across a wide swath of the wetland area (where it is not currently present such as the western, south-western, or south-eastern sections), creating a mosaic which can eventually spread and unify (Clarkson *et al*., 2017). This can be completed in two main ways, the capture and spread of seeds during a mast event (potentially long-term and difficult to complete), or transplanting whole *E. robustum* plants from the northern section (easier but potentially may cause slight damage to areas where plants will be transplanted from).

The reintroduction of the locally extinct *Sporadanthus ferrugineus* into zones that are already occupied by *E. robustum* would allow the bog to recreate past peat bog climax communities, with both the primary peat former *E. robustum* coexisting with *S. ferrugineus*. Currently, the majority of the *E. robustum* is clustered in a large patch in the north-east, and therefore this is most likely the best location to begin reintroduction. Reintroduction of *S. ferrugineus* would also allow introduction of the 'at risk' endemic moth species *Houdinia flexilissima* (commonly called Fred the Thread), which mine and pupate inside living *S. ferrugineus* stems.

Control of invasive animals and predators will prevent ongoing degradation of both the wetland condition (pigs eroding the drain banks and trampling vegetation) and the ecosystem (pest animals predating on wetland bird species). This can also include the removal of fast spreading pest weeds, such as Royal fern (*Osmunda regalis*), or clearing standing trees along the margins of the wetland (long term ecological phasing of vegetation). The clearing of drainage ditch banks is a common practice, as it both removes the invasive species from the marginal areas, as well as reducing evaporation and water table drawdown from these species. It also increases the amount of available decaying wood in the area, helping to ease the transition of substrates between the wetland peat and the adjoining marginal areas (Similia *et al*., 2014). As any fire event within the wetland could potentially set back the natural spread of *E. robustum*, any fires that start within the wetland should quickly be contained and suppressed to prevent further damage to the wetland.

Further research can be conducted within the wetland to expand upon this thesis. Measurement of peat depths in the northern wetland (Transect C) would be beneficial as it would show whether the high nutrient values recorded there are indicative of ongoing degradation caused by the shallow northern drain (if deeper peat layers are present), or if the historic condition of the area has always been swamp-like with higher inputs of sediment and nutrients (shallow peat layers).

Vegetation composition is an important indicator of wetland class and is a simple metric to measure, the implementation of regular monitoring of vegetation through key areas of Otakairangi would ensure changes and trends are identified over medium to long term. Future drone surveys could potentially repeat the vegetation map provided by Kendal (2016), focusing on changes from the past survey, as well as identifying *E. robustum* expansion across the wetland, as this was not previously noted in past surveys.

The current peat surface oscillation pressure transducers can be moved into the area identified as having strong bog characteristics, to potentially see if the surface oscillation there is stronger than the current location. As peat surface oscillation is a potential indicator for wetland health, with substantial surface oscillation only occurring in pristine wetland conditions, monitoring the oscillation where peat layers may be thicker and exhibit stronger natural surface oscillation would give a long term monitoring scheme establishing the functioning of the ombrotrophic peat bog area.

Lidar can be conducted to acquire a wetland wide map, giving elevation data, overland flow paths and topographic differences that cause differences in wetland condition.

7.4.2 External land management and projects

Work outside of the wetland in the nearby catchment through improved farm management and protective practises (riparian planting, retirement of runoff source areas, headwater forest planting to reduce quick flows) could be accomplished to reduce the amount of sediment entering the waterways, as well as nutrients that are bound to them. The area around the wetland is separated into three main catchments, a large catchment area feeding the water for the central drain, a mid-sized catchment to the south-west of the wetland feeding into the western border drain, and a smaller catchment to the north and east feeding into the respective border drain. The smallest catchment contains pasture and forested areas, and makes up an area of ~140 ha (1.38 km^2). The second catchment to the south-west of the wetland measures ~550 ha (5.5 km^2), and is comprised of pasture, crops, forested areas and hill country. The largest catchment, which is the source for the water running into the central drain, measures ~ 2000 ha (20 km^2), and contains primarily pasture and crops, with the bordering hill slopes being partially covered by forest and bush (Figure 7.3).

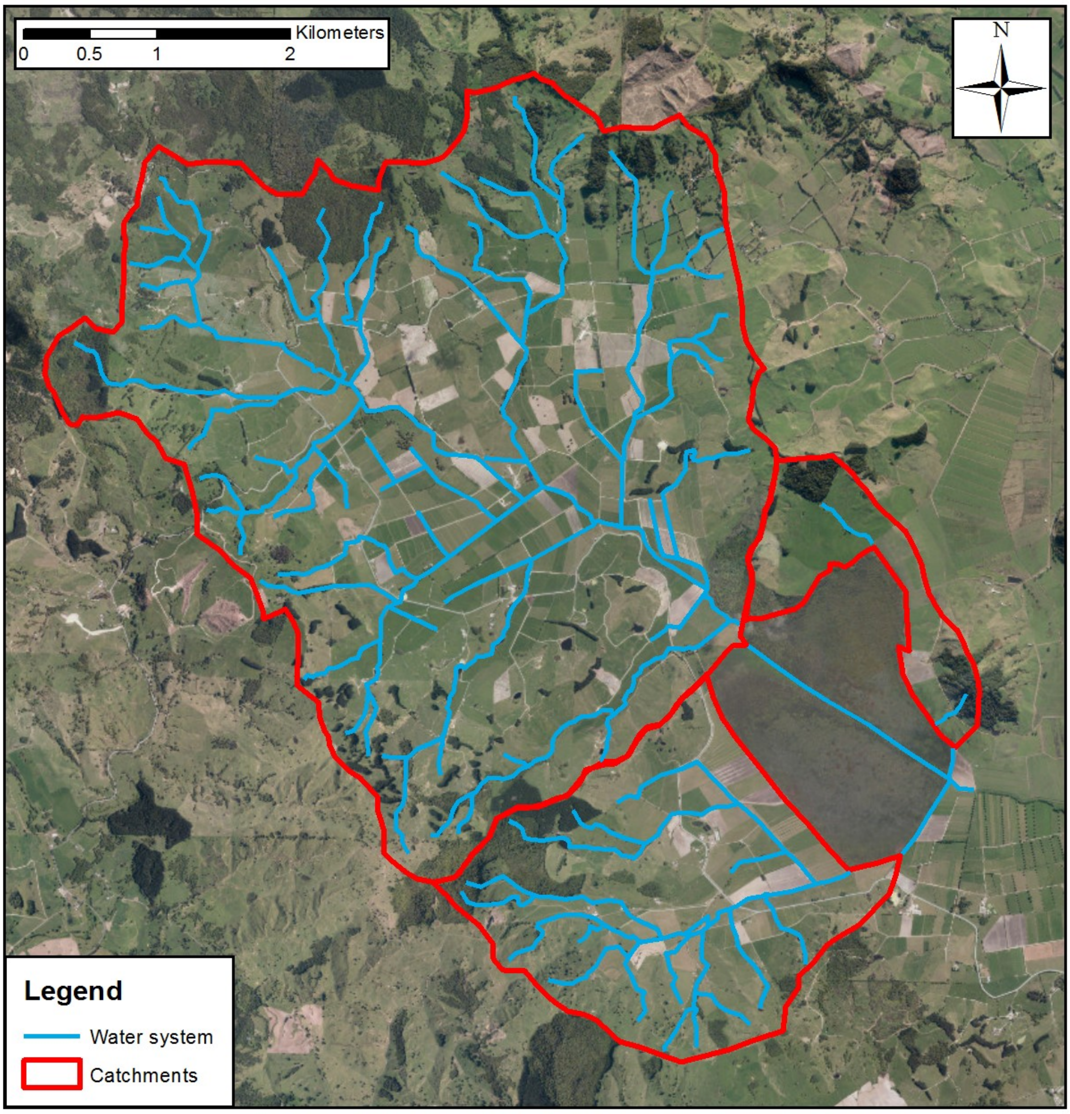

Figure 7.3. Otakairangi wetland with contributing surface water catchments and major waterways delineated. Note the catchment boundary around the wetland is composed of border drains so should also be considered part of the water system.

Establishment of sediment traps in the tributaries of the upper catchment and ongoing water monitoring will determine where the greatest sources of nutrients and sediments entering the central drain are. This knowledge will allow external management efforts to be focused in the most urgent locations, such as farm management plans, riparian planting or improving the current extent of fencing.

A constructed wetland could be established above the entrance of the central drain to intercept and hold the nutrient rich water from the upper catchment. This would provide a location for the water currently passing along the central drain to be passively filtered with sediment settling out in the lower energy environment, while also potentially providing flood water storage, assisting in preventing large scale flooding further down the catchment and in the wetland area. However, the current flooding evident in the farmland north of the wetland would need to be addressed as it would likely be influenced by the constructed wetland.

Despite the issues that this shallow drainage channel causes, it should not be dug deeper, as this would create other adverse effects, and as such other alternatives should be found. Similar to the idea of a constructed wetland, buffer zones could be constructed along the northern drain. These could utilise sward species such as *Carex secta, Machaerina teretifolia,* or *Phormium tenax*, which are both natural in wetland environments and suitable for the interception and holding of sediment and nutrients. This would prevent the hillslope erosion and the agricultural runoff from directly running into the wetland, preventing further degradation, while improved fencing would ensure stock are excluded from the wetland area.

7.4.3 Potential central drain projects

Common restoration efforts in wetlands that have been drained involve damming or infilling drainage channels. This allows the internal water table to rise back towards natural levels, which in turn allows natural diffuse flows to spread across the wetlands rather than being channelized to one path. However, in the case of Otakairangi, there are both border drains and a large central drain which were created to improve the surrounding land for farming, as well as reduce flooding in the area. The central drain has experienced high flows during storm events, overtopping and flowing into the wetland, while down the valley productive farmland was also flooded. Ideally, the central drain would be removed in the future to restore a large section through the central wetland back into a more natural state, but that will require significant inputs of money, energy and further research.

Diverting water from the central drain to a border drain could allow for its infilling or other such projects that could potentially mitigate the effects of water table drawdown and the increased nutrient levels in the centre of the wetland area. However, this would require widening or deepening a border drain to compensate for the increased flow that normally passes through the current central drain, potentially creating further adverse effects on both the neighbouring farmland and the border of the wetland that is utilised for the diversion.

Blocking or damming the central drain may recreate natural water flows, but the water that would be diverted across the wetland would be nutrient and sediment rich, due to the upper catchment being primarily farming. Farming practises such as application of fertiliser and grazing animals cause the water that runs through the wetland to be higher in nutrients than natural freshwater, and by spreading this across the wetland it could potentially do more harm by increasing the probability of invasive species moving across the area.

Future research and long term monitoring should be conducted, such as creating a 3D profile of the central drain, giving information of the head gradient (from the entrance to exit), flow patterns, speeds and discharges, as well as nutrient and sediment loads during different flow events. Changes along the drain from bank erosion may give an indication of natural recovery with the drain infilling and widening to compensate for high levels of energy in the water. A detailed study of water quality during different flood events would be useful to determine possible sources and relative concentrations of nutrients and sediment entering the wetland during flood events.

7.5 Conclusions

Otakairangi is the largest remaining remnant of the Hikurangi swamp and is a significant wetland in the Northland region due to its range of ecohydrological characteristics, ecological heritage, and ecosystem services. The use of environmental indicators identified zones of different wetland classes, primarily driven by inputs of surface water from the central drainage channel and the surrounding border drainage, but also by the vegetation community.

The intensive agricultural land usage that dominates the catchment, as well as farmland directly next to the wetland, has led to mineralisation of the edges of Otakairangi wetland with a swamp belt characterised by higher nutrient availability and a wide range of vegetation species. However, this also acts as a buffer zone that in some ways prevents nutrients from further penetrating into the inner wetland which exhibits restiad fen and bog characteristics.

The restiad bog area that was located along Transect D was small in extent, presumably due to the drainage scheme and past fires which would have altered the vegetation composition. This zone was low in nutrients and was dominated by the peat-forming vegetation species *E. robustum*. This area is at risk from further nutrient inputs with surface water inundation during flood events, and the resulting encroachment of other vegetation species which will outcompete *E. robustum.* However, there is evidence that despite the degradation of the wetland, *E. robustum* is in fact expanding naturally, which has led to improvement in wetland condition, and other bog areas may be present outside of the measured transects.

Increased flood inundation of the central drainage channel is the primary cause for nutrient and sediment inputs at the peat surface and changes in vegetation composition near the central drain. Secondarily, the shallower northern drain acts as a conduit for water coming off the hillslopes to enter the northern section of the wetland. A change in catchment land use (reduction in nutrient and sediment loading to the freshwater system) in conjunction to engineering solutions around the central drain to reduce the flood frequency is recommended.

While there seems to be natural recovery of the wetland condition due to the spread of *E. robustum*, Otakairangi wetland is still at risk, as a decline in wetland condition followed by a loss of biodiversity will continue if the frequent flooding of the drains and associated sediment transport is not mitigated. Potential fire events could also set back the current recovery of wetland vegetation, despite the insurance created by the wide ranging, patchwork spread of *E. robustum*. The wetland requires the establishment of an ongoing monitoring regime, as well as the implementation of mitigation methods to reduce or remove the anthropogenic degradation factors to the wetland. Assisting the spread of *E. robustum* into the south-west will also mitigate the threat of fire.

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Appendix A

Table A. 1: Von Post decomposition index (adapted from Clymo 1983).

Degree of decomposition	Description
H1	Undecomposed. Plant structure unaltered, fibrous. Yields only clear colourless water.
H2	Almost undecomposed. Plant structure distinct, almost unaltered Yields only clear water coloured light yellow-brown.
H3	Very weakly decomposed. Plant structure distinct, remains easily identified. Yields distinctly turbid brown water.
H4	Weakly decomposed. Plant structure distinct, most remains easily identifiable. Yields strongly turbid water, residue rather mushy.
H5	Moderately decomposed. Plant strucuture clear but becoming indistinct, most remains difficult to identify. Yields much turbid brown water, residue very mushy.
H6	Well decomposed. Plant structure indistinct but clearer in the squeezed residue than in the undisturbed peat. About a third of the peat escapes between the fingers, strongly mushy.
H7	Strongly decomposed. Plant structure indistinct but still recognisable About half the peat escapes between the fingers, slurry of peat in suspension.
H8	Very strongly decomposed. Plant structures very indistinct, only remnant root fibres and wood identifiable. About two-thirds of the peat escapes between the fingers, thick slurry, little free water.
H9	Almost completely decomposed. Plant structure almost unrecognisable. Almost all the peat escapes between the fingers, no free water.
H10	Completely decomposed. Plant structure unrecognisable, amorphous. All the peat escapes between the fingers, no free water.

Appendix B

Table B. 1: Species list of flora found within the vegetation plots in Otakairangi wetland (alphabetical order). For an extensive vegetation list of the entire wetland and surrounding area, refer to Clarkson, Bartram & Price (2015).

Vegetation species	
Scientific name	Common name
Campylopus introflexus	Heath star moss
Carex secta	Purei sedge
Convolvulus arvensis	Field bindweed
Coprosma propinqua	Mingimingi
Coprosma tenuicaulis	Swamp coprosma
Cyathea dealbata	Silver fern
Empodisma robustum	Jointed wire rush
Gleichenia dicarpa	Tanglefern
Glyceria maxima	Reed sweet-grass
Isachne globosa	Swamp millet
Leptospermum scoparium	Mānuka
Machaerina teretifolia	Sedge
Paesia scaberula	Hard fern
Parablechnum minus	Swamp kiokio
Phormium tenax	Common flax
Ranunculus repens	Creeping buttercup
Schoenus brevifolius	Bog-rush
Ulex europaeus	Gorse
Usnea sp.	Old man's beard

Appendix C

Table C. 1: Average, median, maximum, minimum and ranges for water level monitoring sites at Otakairangi wetland. Units are water level relative to sea level (m asl). * indicates sites that were measured as relative water level (relative to peat surface). Measurements were taken from 6th October, 2017 to 11th January, 2019 (OT 1 to OT6), and 12th September, 2018 and 11th January, 2019 (for OT7, OT10 and OT11).

Site	Distance from central drain (m)	Average	Median	Max	Min	Range
OT1	0	89.43	89.33	91.83	89.01	2.82
OT2	20 (NE)	91.26	91.26	91.80	91.10	0.69
OT3	50 (NE)	91.52	91.52	91.85	91.36	0.50
OT4	100 (NE)	91.43	91.43	91.64	91.25	0.39
OT5*	280 (NE)	91.84	91.84	92.05	91.63	0.42
OT6	280 (NE)	91.99	91.99	92.22	91.76	0.46
OT7*	290 (NE)	91.85	91.83	92.00	91.73	0.27
OT10	140 (SW)	91.63	91.62	91.71	91.58	0.13
OT11	280 (SW)	91.53	91.52	91.65	91.44	0.22

Appendix D

Contents of the supplementary files include peat, vegetation and hydrological data.

The file labelled ‘Vegetation’ includes vegetation plot data (species cover and indicator scores).

The file labelled ‘Chemical and physical’ includes peat physical and chemical data, vegetation chemical data and community analyses, as well as water chemical data.

The file labelled ‘Hydrology’ includes the water table data retrieved from the automatic dip well pressure transducers.