ECOHYDROLOGICAL CHARACTERISATION OF WHANGAMARINO WETLAND

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Abstract

The Whangamarino wetland is internationally recognised and one of the most important lowland wetland ecosystems in the Waikato Region. The wetland's hydrology has been altered by reduced river base levels, the installation of a weir to raise minimum water levels and the Lower Waikato Waipa Flood Control Scheme, which is linked via the (hypertrophic) Lake Waikare and affected by varying catchment land use practices.

When water levels exceed capacity, the overflow is released into the Whangamarino wetland, which also receives flood waters from Whangamarino River. Water levels in the wetland are also affected at high stage, by a control structure near Meremere at the confluence of Waikato and Whangamarino Rivers, and at low stage by a weir a short distance upstream.

The ecohydrology of a representative part of the wetland was studied to assess the linkage between wetland ecology and the natural and anthropogenic modification of the flood regime and land use. The primary goal of this research was to characterise the present state of the wetland, which will aid in developing future goals and approaches for restoration.

The study focused on a 2.3 km transect extending from the Whangamarino River, through the wetland to adjacent farmed hillsides. Hydrological and meteorological data were retrieved and analysed from an automated weather station and seven water level sites along the transect. Historical water level records (over 46 years) were used to determine changes in the hydrological regime and the impact of the flood control scheme, through a flood inundation and frequency analysis. During a winter flood event, river water quality was assessed. Peat surface oscillation in the restiad bog was examined. Vegetation patterns were assessed and classified through ordination and statistical techniques. Peat, soil and foliage physical and chemical quality were measured. Atmospheric ammonia (NH_3) deposition rates of N into the wetland were measured.

Water levels in the inland 0–1.1 km of the transect line (restiad bog) were relatively stable and consistent, rising and falling through winter and summer. This area had rainfall as the primary water input and was independent from the Whangamarino River, except during large flood events where the fringe of the restiad bog was inundated. Closer to the Whangamarino River water levels were more variable and strongly responsive to the river's hydrological regime. A flood inundation event in September 2010 impacted on wetland water level regimes up to 1.4 km from the river and had a return period of 3.3 years. Frequency analysis showed sites up to 500 m from the river will likely be inundated by floods every year. A 100 year flood was estimated to inundate 1.75 km from the river, but would not cover the entire wetland. River water samples collected during a flood event showed total suspended solids within the Whangamarino River peaking at 260 mg L^{-1}, double the concentration from Pungarehu Canal (86 mg L^{-1}). Nutrient concentrations (such as dissolved reactive phosphorus) followed a similar pattern to the flood hydrograph. Minimum water levels have increased since the development of the artificial weir, but before this occurred water table lowering may have encouraged manuka invasion towards the restiad bog. Increased flood inundation is now the most likely threat to continued wetland degradation and manuka invasion into the restiad bog, due to the change in water levels and the deposition of sediment and nutrients.

Nutrients, heavy metals, isotopes ($\delta^{15}N$) and physical soil characteristics (such as bulk density) increased from the start of the manuka belt (1100 m) and were greatest near the Whangamarino River (2300 m). A gradient was observed in peat and soil chemistry patterns, with increasing fertility and a change from bog to swamp-type environments along the transect line towards the river. A mineralised swamp fringe belt was present next to the farmland (0–50 m). From 50–1100 m a restiad bog (dominated by *Empodisma minus*) was present and changed to a manuka transition zone from 1100–1500 m. From 1500–1900 m, a swamp environment was present with a dominant canopy of manuka changing to *Coprosma tenuicaulis* closer to the river. *C. tenuicaulis* appears to be acting as a buffer zone over 150 m, removing a large amount of nutrients and sediment from flood waters. The remaining 400 m (1900–2300 m) of the transect line was a marshland, with the highest nutrient and sediment abundances and the most variable water level patterns. This area was colonised primarily by *Polygonum persicaria* (willow weed).

The major risk to the wetland is from continued flood inundation with nutrient and sediment rich waters. Recommendations for future management include restoring catchment water quality and better management of the flood control regime.

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Chapter 1: Introduction

1.1 Background

Wetlands are important ecosystems and throughout the world are under threat. They cover around 6% of the terrestrial surface (Bullock & Acreman 2003). Wetlands have historically been recognised as 'waste' land, often being drained and converted to pasture for agricultural purposes. In New Zealand approximately 90% of wetlands have been lost (Sorrell *et al.* 2004b). This vast drainage often for agricultural purposes, has been slowed dramatically in recent years and in some situations, wetlands are being restored and enhanced (Mitsch & Gosselink 2007).

In recent decades, wetland studies have come to identify the benefits these ecosystems can have (Halabuk 2006). Benefits range from hydrological and hydro-chemical components such as increasing water quality ("natures' kidneys") to biological components such as uniquely adapted and threatened plant and animal life (Jiang *et al.* 2007). Various human uses of wetlands include water storage, flood mitigation and as carbon sinks for increasing global atmospheric CO_2.

The majority of wetland studies have been undertaken overseas, in Canada, Britain and the USA. Wetland hydrological studies in New Zealand have been limited, with a significant amount of research still needed. The Waikato region has approximately 25% of original wetlands remaining with some of New Zealand's largest wetland ecosystems (Clarkson 2002; Department of Conservation 2010a). These areas are under similar pressure from intensive agriculture, and in depth hydrological characterisations are being undertaken to gain an idea of how the wetlands function as an ecosystem (Mitsch & Gosselink 2007). With this knowledge, management issues can be dealt with more thoroughly and steps can be taken to preserve and protect these valuable ecosystems.

1.2 Waikato and New Zealand wetlands

Most Waikato wetlands have been destroyed or significantly modified by drainage (for agriculture) but there are still a number of substantial wetlands that include young swamps to developed restiad bogs. Restiad bogs are usually dominated by two plant species of the Restionaceae family; wire rush (*Empodisma minus*) and cane rush (*Sporadanthus ferrugineus*). *E. minus* is the primary peat forming species in these wetlands, due to a number of adaptations (Hodges & Rapson 2010). Bogs develop as rain fed systems, with no additional input from surface and groundwater (Johnson & Gerbeaux 2004). These remaining restiad bog wetlands have been confined in the Waikato between latitudes 37º S and 38º S due to artificial drainage (Figure 1.1) (Clarkson *et al.* 2004a).

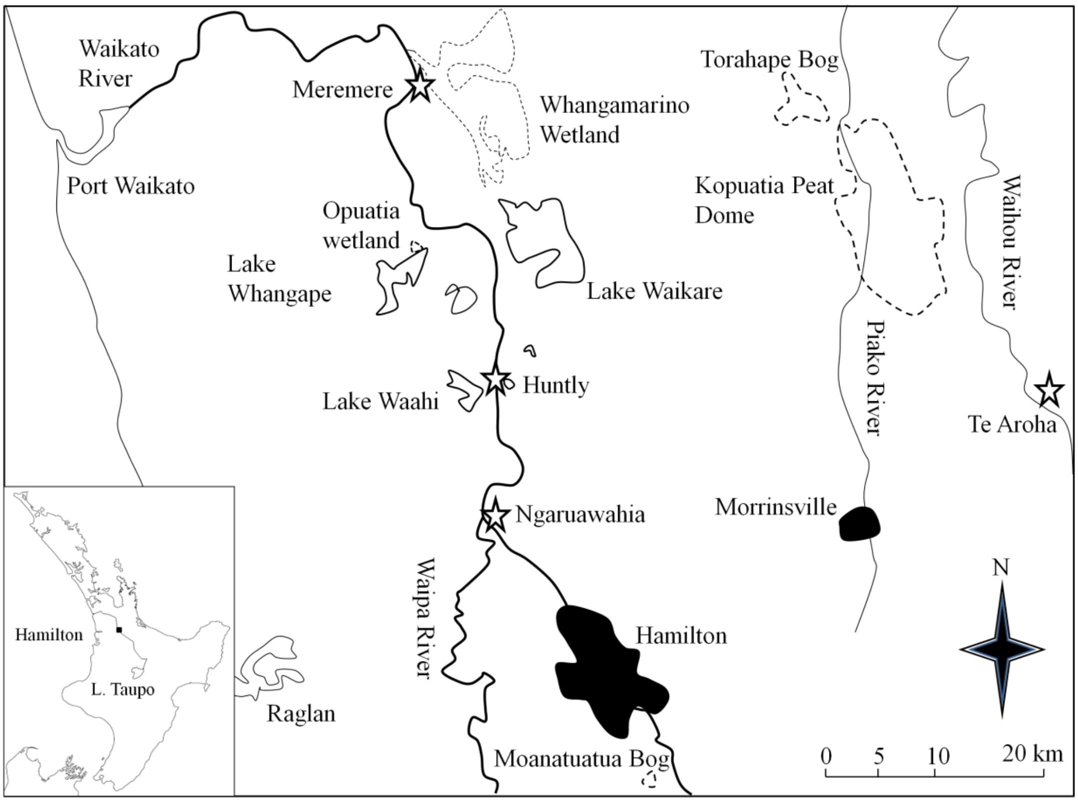


Figure 1.1: Location of wetlands throughout the Waikato region.

Fens and swamps are also present in the Waikato and are the most common wetland classes found (all three classes shall be described in greater detail in Chapter 2). Fens often receive hydrological inputs from groundwater, surface flows and precipitation which increase the availability of nutrients, ultimately allowing more diverse plant species to be found (Johnson & Gerbeaux 2004).

Both woody and herbacious plant species may be found, including *Baumea spp* (twig rush), *Schoenus spp*, *Gleichenia dicarpa* (tangle fern), *Leptospermum scoparium* (manuka), *Coprosma tenuicaulis* (swamp coprosma) and *Phormium tenax* (flax) (Clarkson 2002). Fens are often present in surrounding boundaries of developed restiad wetlands, where in Whangamarino they are common in close proximity to restiad bogs.

Swamps were less widespread in the Waikato before European settlement, as the dominant wetland was restiad bogs. Generally swamps have surface water and precipitation as the main form of input, and have a water table which resides above the ground surface for long periods of time. Often they are nutrient rich and have a range of species, particularly invasive species which can outcompete natives (Johnson & Gerbeaux 2004). Common plants in swamps include *Typha orientalis* (raupo), sedges such as *Carex secta* and tree species such as *Cordyline australis* (cabbage tree). Invasive species that are common include *Salix cinerea* (grey willow) and *Salix fragilis* (crack willow) (Clarkson 2002). Lakes Waahi, Whangape and Waikare (Department of Conservation 2010a). The latter of which flows directly into the Whangamarino wetland as part of Environment Waikato’s flood storage scheme (Department of Conservation 2010a).

These wetlands in the Waikato are important areas for a range of fish species, such as native Galaxiids (whitebait) and Long-finned and Short-finned Eels (*Anguilla* spp.). Various species of birds inhabit these wetlands including, North Island Fernbird (*Bowdleria punctata vealeae*), Australasian Bittern (*Botaurus Poiciloptilus*), Brown Teal (*Anas aucklandica chlorotis*), and the Spotless Crake (*Porzana tabuensis plumbea*). Large numbers and various species of waterfowl (both native and introduced) are also found (National Wetland Trust 2009; Department of Conservation 2010a). Many rare flora are also found in the remaining Waikato wetlands, including the very rare helmet orchid (*Anzybas carsei*) only found in the Whangamarino wetland (Department of Conservation 2010a).

1.3 Whangamarino wetland

Whangamarino is a large wetland (over 7000 ha) located approximately 60km south of Auckland (Figure 1.1). Swamps, fens and restiad bogs are all present in various locations in the wetland. The Whangamarino wetland receives water from the Whangamarino and Maramarua Rivers and additionally has hydrological inputs from the Waikato River, through Lake Waikare (separate from the catchment inputs). The wetland is currently utilised as a flood storage area for the Waikato River, to prevent flooding of agricultural land (Waugh 2007; Department of Conservation 2010a).

Threats to this wetland include changes in the hydrological regime from the flood control scheme and nutrient laden flood waters entering from the Whangamarino River and Lake Waikare. Invasive plant species (such as willow) outcompete many natives due to their ability to survive and thrive in a range of harsh environments. Farming and cropping potentially can be impacting in the wetland directly through cattle access and grazing on the fringes of Whangamarino, and indirectly through nutrient runoff entering water ways and ultimately contributing to eutrophication.

In 2009 Department of Conservation established a transect line of automatic water level monitoring sites, supported by a meteorological station providing environmental data (Chapter 3).

Research at this site was supported by funding from the Department of Conservation as part of the Arawai Kākāriki (Green Waterway) project which aims to enhance the ecological restoration of three of New Zealand’s foremost wetland (Department of Conservation 2009). The overall aim is to provide an ecohydrological classification of the state of health of Whangamarino wetland which can then be used to provide information and recommendations for future management and protection.

1.4 Objectives

The primary aim of this research is to investigate the ecohydrological characteristics within a portion of the Whangamarino wetland and determine the internal and external factors impacting on the patterns and processes of water, peat and vegetation. Findings will be interpreted in the context of the present state of the wetland, identifying the hydrological and land management issues affecting the site. In more detail the objectives are to:

- Characterise the seasonal and annual hydrological regimes along a transect line through a portion of the southern Whangamarino wetland.
- Identify hydrological processes operating along the transect line with particular regard to the flow regime of the Whangamarino River and the operation of the Lower Waikato/Waipa Flood Control Scheme. Characterise the water quality of a flood event that leads to inundation in the wetland.
- Determine patterns in vegetation composition and peat chemical and physical characteristics along the transect line.
- Identify and attempt to explain the relationship between vegetation characteristics, foliage and peat physical and chemical quality, nutrient inputs and hydrological processes.
- Provide scientifically sound information on the current state of wetland condition and the influencing factors, in order to formulate better management approaches for restoration and enhancement.

1.5 Thesis outline

Chapter two provides a review of relevant wetland research and literature as a background to the thesis and Whangamarino wetland.

Chapter three describes the Whangamarino wetland research site in regards to its location, climate and geology. The meteorological and hydrological monitoring network along the wetland transect line is described, and relevant studies relating to the wetland are outlined.

Chapter four presents findings on peat and soil characteristics along the transect line at Whangamarino with discussions on what may be causing the observed trends. Comparisons are made to relevant New Zealand wetland literature.

Chapter five describes the vegetation composition and patterns along the transect line at Whangamarino wetland. Findings are interpreted relative to the peat and foliage chemistry at each site, and historical vegetation changes for the entire wetland are looked at in more detail.

Chapter six investigates the wetland hydrology along the transect line based on data from seven different water level monitoring sites. Transect water levels are examined to identify changes and the possible driving influences for the patterns observed. Flood inundation and frequency assessment is undertaken through comparison with Whangamarino River records and transect water levels. Additionally, water quality during a flood event and peat surface oscillation in the restiad bog is described.

Chapter seven discusses the main findings of the research in an ecohydrological context, relative to the research objectives. The patterns along the transect line in peat and vegetation are compared with hydrology, and wetland classes for different zones are identified. The impact of the modified hydrological regime on the wetland is outlined, and recommendations for future management and research are discussed.

Chapter 2: Literature review

2.1 Introduction

Wetlands have long been considered wastelands and throughout the ages have been modified and drained for primarily agricultural purposes. In the last half century, a change in peoples' perceptions of wetlands from wastelands, to valued ecosystems has resulted in a vast increase in research. Wetlands are now seen as dynamic environments that have unique hydrological and ecological functions, such as improving water quality or providing habitats for a diverse range of plants and animals (Mitsch & Gosselink 2007).

This chapter will discuss the relevant literature and current understanding relating to wetlands. It will identify the different wetland classes and describe the various hydrological, ecological and environmental processes occurring. Wetland and peat development will be described, followed by wetland hydrology. Nutrient presence and movement in wetlands shall also be discussed, linking with the concept of eco-hydrology as a framework for wetland research. Nutrients and hydrology are strong controlling factors for vegetation characteristics and diversity and will be elaborated on with particular regard to disturbance, human modification and management issues.

2.2 Ecohydrology

The term ecohydrology is often used in a range of different ecosystems and is an integrated approach that aims to analyse and quantify the interactions between biological and hydrological processes at a catchment scale (Trepel & Kluge 2002; Zalewski 2002). This term can also be applied in the context of wetland research and is based on the idea that a variety of components link with hydrology (Figure 2.1). Ecology is the principal component that is affected by hydrology, where aspects of ecology break down to include such things as vegetation, nutrients and soils. Hydrology is based on the study of water, which is a primary control on the complex interactions between biological and ecosystem function (Trepel & Kluge 2002; Campbell & Jackson 2004; Marani *et al.* 2006).

Vegetation composition in wetlands is controlled directly by water movement and abundance. Certain plant species will out-compete others in varying hydrological regimes, with hydrological inputs (such as surface water or rainwater) ultimately changing other components of the ecosystem (such as peat decomposition or accumulation). Aquatic biology will also change relative to hydrological processes, where macro-invertebrates and fish species will be less or more abundant depending on food sources (nutrients and vegetation detritus) and hydrological stability.

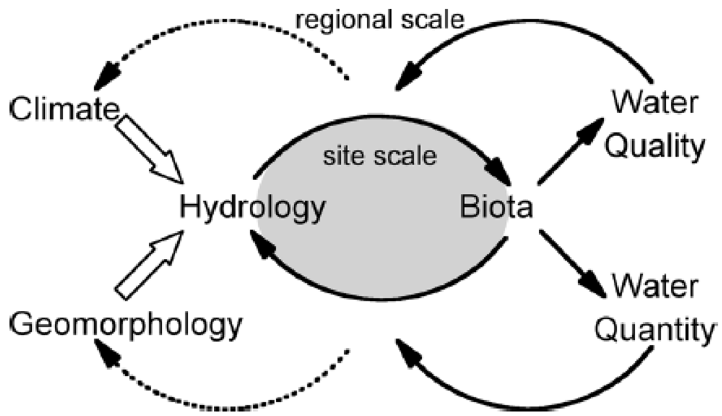


Figure 2.1: The concept of ecohydrology (Trepel & Kluge 2002).

Overseas ecohydrology studies of wetland ecosystems have revealed intimate relationships between their organic components and water, which means that it is impossible to understand their ecology without considering hydrology (Bragg 2002: Zalewski 2002; Zalewski 2007). For example, Bragg (2002) describes how many ecological wetland studies have been conducted in Scotland, where the only focus was on wetlands internal functioning, treating them as areas in isolation from catchment hydrology. Yet the knowledge obtained from these studies allowed insight to wider catchment scale hydrology and functioning, including runoff generation downstream and nutrient contribution altering wetland biology. Ecohydrology has now been accepted as a fundamental approach when pursuing wetland research.

Integrative science is an important component of ecohydrology, and on a basin or catchment scale is the most appropriate way for increasing the health of a range of ecosystems while incorporating harmonization with necessary technical solutions (i.e. dams, flood control and irrigation schemes) (Zalewski 2007).

Browne (2005) studied the ecohydrological characteristics of Opuatia wetland in the lower Waikato River catchment, partly as required by resource consent conditions from Environment Waikato relating to extension of the LWWFCS. Assessment included vegetation plots, peat chemistry, wetland and river water table measurements and installation of a meteorological station. Ecological, chemical and hydrological patterns were used to define the wetlands current state of health through a New Zealand index, known as the Handbook for Monitoring Wetland Condition (Clarkson *et al.* 2004b). Opuatia was found to be in good health, although was at risk from *S. cinerea* (grey willow), which had invaded the unfenced wetland margins over time. Nutrient runoff from surrounding agricultural land was also a cause for concern, with higher concentrations having entered the wetland and promoting weed invasion and degradation of wetland health.

Trepel & Kluge (2002) undertook an ecohydrological characterisation of a 150 ha degenerated valley peatland, dissected by the Eider River in Northern Germany. The peatland was affected by drainage, land use intensification and river regulation (such as deepening, straightening and stop banks) to reduce flood impacts during winter. The upland catchment draining into the wetland (similar to Whangamarino catchment) was comprised of 60% crops, 20% pasture and 10% forest. Nitrogen fertilisation in these areas was around 130-160 kg N ha^{-1} yr^{-1}. The ecohydrological study incorporated a transect line through the peatland, which assessed hydrology, peat characteristics and vegetation composition using manual measurements and techniques used in the present Whangamarino study. Essentially they found restoring flood flows and a more natural water level regime would not lead to regeneration of peatland due to irreversible damage (from enhanced decomposition).

2.3 Wetland classes and types

The classification of wetlands can be large and complex, often due to the range of hydrological, geomorphological and biological functions that contribute to their development. Wetlands often contain both mineral soil and organic (peat) constituents (Price *et al.* 2003). The following review will focus primarily on peatlands which are wetlands with organic matter layers (greater than 30 cm thickness), usually situated over a mineral soil or bedrock (Charman 2002; Martini *et al.* 2006).

Generally the fundamental prerequisite for the development of wetlands is wet or saturated land. This usually occurs in depressions relative to the surrounding topography, where water is trapped (or drainage impeded), or a limited slope results in very little runoff for all or part of the year (Ingram 1983; Murray-Hudson *et al.* 2006). 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".

The wetlands types that form the focus of this study belong to the palustrine hydrosystem. Palustrine wetlands are those which are fed by rain, groundwater or surface water and do not occur within the normal boundaries of lakes, rivers and estuaries (Campbell & Jackson 2004; Johnson & Gerbeaux 2004). With these three types of hydrological inputs, coupled with topographic position in a catchment, wetlands can often be classified into certain groups (Novitzki 1978; Gibson *et al.* 2000; Jolly *et al.* 2008). All studies mentioned will be some form of a peatland (subset of palustrine wetlands), and from this point on they shall all be referred to as wetlands.

The most common wetland types are swamps, fens and bogs (Table 2.1). Each of these wetlands often develops from the previous successive type, with swamps being the youngest and bogs the oldest. Class changes of wetlands can take thousands of years. Swamps will not always turn into fens or bogs as it depends on a variety of factors including location, hydrology and chemistry of the wetland.

Swamps often have surface water, precipitation and occasionally some groundwater contribution. They commonly contain both mineral and peat substrates, have poor drainage and often have a water table above the ground surface in places (Johnson & Gerbeaux 2004). Water pH is closer to neutral (and slightly acidic) and contains higher levels of nutrients compared to bogs (leading to swamps often being classified as eutrophic). Hence swamps are often extremely productive with a diverse range of species present.

Fens usually develop from swamps through the accumulation of peat resulting in the isolation of the site from significant surface water contributions (Clarkson 2002). Fen hydrological inputs are comprised of groundwater, with some surface runoff and rainfall. Nutrient levels and pH (acidity) are generally low to moderate in fen ecosystems and therefore are generally mesotrophic (Johnson & Gerbeaux 2004). Swamps and fens are also known as minerotrophic (where they receive additional water sources rather than only precipitation) (Clarkson 2002).

In these wetlands, specially adapted wetland plants such as reeds and sedges become an integral part of the ecosystems. Over time saturated anaerobic conditions increase the amount of peat (derived from organic material), which accumulates due to a slowed decomposition rate (lack of oxygen) and rises above the regional water table (Murray-Hudson *et al.* 2006). At this stage the wetland would be called ombrotrophic or a bog, with the peat surface typically being domed. Bogs are hydro-logically isolated from all minerotrophic water sources apart from rainwater, with some groundwater contributions to deep layers of peat at the original water table depth (Verry & Boelter 1978; Johnson & Gerbeaux 2004).

Table 2.1: Four main palustrine wetland systems in New Zealand (adapted from Johnson & Gerbeaux 2004).

Wetland class	Dominant hydrological input	Description
Marshland	Surface water and groundwater , precipitation	Mesotrophic to eutrophic, mineral wetland, large fluctuations in water table, better drainage than swamps, high nutrient and sediment levels, low to neutral acidity, presence of grasses, sedges, weeds.
Swamp	Surface water, groundwater and precipitation	Mesotrophic to eutrophic, mineral and peat substrates, high nutrient and sediment inputs, variable water table, low acidity, presence of shrubs, sedges, flaxes, scrub bushes, reeds and possibly forest.
Fen	Groundwater, precipitation and some surface water	Oligotrophic to mesotrophic, presence of peat, intermediate nutrient levels, high water table with slow to moderate water flow, moderate acidity, presence of sedges, restiads, ferns, scrub.
Restiad bog	Precipitation	Oligotrophic, peat, nutrient poor, saturated (high water table with little movement or flow) and poorly aerated, high acidity, presence of restiads, ferns, scrubs, lichens, cushion plants, trees.

2.4 Peat physical processes

Peat is the primary component which makes up the wetland substrate, and consists mainly of organic matter from decomposed plant material. Peat soils have been estimated to cover 2–3% (approximately 5 x 10^6 km^2) of the global terrestrial surface (Whittington & Price 2005; Martini *et al.* 2006; Gronlund *et al.* 2008). Due to the lack of mineral constituents and weathering, peat cannot be directly classed as a soil. While peat has a simple structure and texture, the physical and chemical processes it can influence and control are complex.

2.4.1 Peat formation

Through time, wetland depressions develop plant communities that are adapted to living in damp, water logged environments. Peat forms due to dead plant material accumulating beneath living vegetation. The main controlling factor for accumulating organic matter is microbial decomposition being impeded by anaerobic, waterlogged soils found in wetlands (Bragg & Tallis 2001).

Under natural conditions organic matter decay is primarily due to microbial decomposition and soil organisms (such as nematodes), with the main controls on the rate of degradation related to water content, temperature, plant material (i.e. lignin or cellulose) and oxygen supply. Generally peatlands have slow decay rates and hence an accumulation in organic matter, but modification of one of these controls can increase this decay rate. Water table (WT) height is a key controller in decomposition, where often the more oxygenated and less saturated the peat is then the higher the rate of degradation (Bragg & Tallis 2001; Charman 2002). The main area of rapid decomposition is the near surface layer (0 to 0.5 m), which is periodically saturated, aerobic and with a high hydraulic conductivity (water movement rate). This zone is the acrotelm (Ingram 1978). Below the acrotelm the peat is permanently saturated (anaerobic) with a low hydraulic conductivity and little degradation occurring (Beheim 2006; Biester *et al.* 2006; Martini *et al.* 2006). The deeper zone, making up the majority of peat volume in bogs is termed the catotelm (Ingram 1978).

2.4.2 Physical characteristics

Peat accumulates as a result of sustained anaerobic conditions in wetland soils, with often only a small amount of mineralisation (conversion of organic matter to mineral soil constituents) (Kechavarzi *et al.* 2007). Hence peat often contains greater than 60% organic matter, and less than 20–35% inorganic content. In its natural state peat should hold 88–97% water, 2–10% dry matter and 1–7% gas (Flaig 1986; Charman 2002). Wetlands containing peat have layers that are usually greater than 0.4 m in depth and in some instances can exceed 3 m. Kopuatai peat dome in the lower Hauraki area of the Waikato region has peat depths up to 12 m, and areas of Whangamarino wetland (a younger wetland hydrosystem) have peat depths up to 3.5 m (Shearer 1997; Shearer & Clarkson 1998). Peat depths across these wetlands were mapped out by Shearer (1997).

The major variation in peat structure and texture is due to the overlying wetland plant composition that degrades to peat (Beheim 2006). Peat is formed from various species of plant but is usually categorized into four groups; moss, herbaceous, wood and detrital or humified peat (entirely decomposed so that plant material is unrecognisable) (Fuchsman 1986; Charman 2002). Certain plant groups have differing makeups of organic structures which are mineralised by microbes to various degrees. The dominant peat forming plant in New Zealand is *E.minus* (Sorrell & Gerbeaux 2004; Hodges & Rapson 2010).

2.4.3 Peat acrotelm and catotelm

Hydraulic conductivity describes the ability of a substrate to transmit water. The initial classification of the acrotelm and catotelm layers by Ingram (1978) was later reclassified by Burt (1995), Haigh and Kilmartin (2003) and Haigh (2006). They classified the acrotelm as the layer which exchanges moisture and gases with the atmosphere, is permeable, aerobic, and has a high hydraulic conductivity which experiences seasonal water table fluctuations. Once the water table drops below permeable layers, water discharge is greatly reduced. The catotelm is a layer with very low hydraulic conductivity, low permeability and is anaerobic with low water flow rates. It is usually permanently saturated.

In 1978, Verry and Boelter analysed the horizons found in various peatlands. Table 2.2 shows the hydrological characteristics of these peat layers. The acrotelm was noted as the top 30 cm layer of surface peat, with the catotelm at 50 cm, and an intermediate layer (known as the hemic layer) between the two.

Table 2.2: Physical and hydrological characteristics of wetlands in Minnesota, North America (modified from Verry & Boelter 1978).

Layer	Fibre content (%)	Dry bulk density (g cm^3)	Hydraulic conductivity (m day^{-1})
Acrotelm (Fibric)	> 66%	0.09	>1.29
Middle (Hemic)	33–66%	0.09–0.2	1.04–1.29
Catotelm (Sapric)	<33%	0.2	<1.29

Burt (1995) & Haigh and Kilmartin (2003) found up to four times greater runoff in the acrotelm layer. Dry bulk density (DBD) in wetlands is a key controller of hydraulic conductivity, where decreasing hydraulic conductivity occurs with increasing DBD. Peat with a low DBD of 0.05 g cm^{-3} could have a hydraulic conductivity of up to 145 m day^{-1}, while peat with a DBD of 0.25 g cm^{-3} might have a hydraulic conductivity of only 0.33 m day^{-1}. While this may not be applicable to all peat layers around the world, it shows that with increasing dry bulk density and decreasing hydraulic conductivity, water movement slows.

Soils and peat composition often vary across a wetland. Spatial variations are common in both peat structure and texture. Infiltration rates (absorption of water through the peat at rate which will not saturate or pond the surface) measured by Haigh and Kilmartin (2003) in the acrotelm and catotelm of a wetland in UK were found to vary considerably. Infiltration rates ranged from 1510 mm hr^{-1} in the acrotelm to 214 mm hr^{-1} 0.5 m below the surface (catotelm).

2.4.4 Peat hydraulics

Often water travels through a permeable surface (acrotelm) of wetland peat and hillslope soil until it reaches the saturated water table. In an Otago headwater catchment in New Zealand, Bowden *et al.* (2001) studied an upland wetland system and found water moved through the bog quickly (likely through the acrotelm) and could not sustain baseflow to a stream for more than a few days. Measurements of hydraulic conductivity were undertaken by Devito *et al.* (1996) in a Canadian wetland. Low lateral water movements were found in peat layers from 20–50 cm, and when compared to the average annual outflows of the swamps suggested most of the water movement occurs through the near surface peat layer or within the top 10 cm. This was supported by the higher hydraulic conductivity (1 x 10^{-2} cm s^{-1}) in the surface layer, compared to the low conductivity (1 x 10^{-5} cm s^{-1}) at 0.5 m depth. Similar results were found by Halabuk (2006) for the top 5 cm of peat layers. It was concluded that most water movement and discharge must occur near the surface (0–10 cm) through the humified peat.

2.4.5 Peat decomposition pathways

Water table position is the primary control on peat decomposition. Changes in water table will alter processes that can speed up decomposition. These changes occur naturally throughout annual cycles (summer to winter) but can be exacerbated by extreme events such as droughts or artificial drainage.

Temperature is one of the key factors that control microbial respiration and ultimately peat degradation in wetland systems. Increasing temperature is often associated with drainage and can result in greater microbial activity & peat degradation (Kalbitz & Geyer 2000; Worrall *et al.* 2004). A study of drained peat soils by Hogg *et al.* (1992) found the greatest loss of peat occurred in the upper 10 cm of the soil where temperatures were consistently warmer and the peat was more oxygenated. Alternatively, under extreme dry conditions during droughts, microbial activity may also decline (Worrall *et al.* 2004).

The degree of peat decay and conversion of compounds to a decayed state (known as humin) is defined as humification. Greater humification/decomposition of peat suggests dry, warmer conditions with increased decay (such as following drainage), whereas low humification suggests wet, cool conditions with reduced decay (natural peatland conditions) (Charman 2002). Generally humification occurs slowly under anaerobic conditions in the catotelm (Biester *et al.* 2006). This is due to microorganisms' biological reactions being reduced due to a lack of oxygen (Flaig 1986). After humification by microorganisms, humic and fluvic acids are produced, which may accumulate to around 25–35% of peat (dry weight). The acids contribute to the development of micropores and aid in water retention by coating pores with molecular absorbance sites. Hence water is attracted often resulting in drainage of peatlands taking a significant time (low hydraulic conductivity) (Flaig 1986; Fuchsman 1986).

Moisture content is a key control on the availability of oxygen in peat. Oxygen is fundamental for microbial activity which is enhanced under aerobic conditions and reduced under saturated anaerobic conditions (Gronlund *et al.* 2008). Peat has a high affinity to hold water and lowering a water table will still result in the presence of residual water, providing optimal conditions for microbial decomposition, with rates up to fifty times faster than in anaerobic conditions (Holden *et al.* 2007). In areas with fluctuating water tables the peat is often more degraded as the conditions allow greater variation in peat moisture content and oxygenation (Shearer 1997).

Mineralisation is a natural process which involves the conversion of organic matter to more labile, inorganic plant accessible nutrients. Labile nutrients will be taken up by plant roots and will not be incorporated into peat. Mineralisation can occur by physical and chemical reactions within the soil, but commonly involves microorganisms. Anaerobic wetland environments slow mineralisation to a significant degree (Charman 2002; Holden *et al.* 2007). The highest rates of mineralisation are in the aerobic acrotelm layer, where labile products (phosphorus and nitrogen) created from the breakdown of organic matter by microorganism are available to plants (Worrall *et al.* 2004; Holden *et al.* 2007). Lowering the water table leads to enhanced decomposition of organic matter to its mineral constituents (Bambalov 2005).

2.5 Wetland hydrology

Wetland hydrological processes can be complex due to the variety of inputs (surface rainfall, groundwater, overland flow) and outputs (evaporation, surface and groundwater outflow) (Campbell & Jackson 2004). Hydrology differs amongst the various palustrine wetland types due to many influencing components. Some of these include topography, geology, local and regional climate, and vegetation composition.

2.5.1 Water flows in wetlands

Hydrology is a key factor in determining wetland classification. Swamps are recharged by groundwater, surface water and rainfall. Fens are predominantly recharged by groundwater, with some limited inputs from surface water and rainfall, while bogs are recharged exclusively by precipitation. Table 2.1 gives a broad overview of the formation of some palustrine wetlands in New Zealand based on their hydrological properties.

Wetland hydrology is often classified and compared based on the water balance:

$$\text{Inputs} - \text{Outputs} = \text{Change in storage} \qquad 2.1$$

Where the general form of the equation for wetlands is:

$$(P + Q_{in} + G_{in}) - (E + Q_{out} + G_{out}) = \Delta S \qquad 2.2$$

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

ΔS in a wetland is often negligible over an annual period, where water seasonal inputs from precipitation, surface inflow and groundwater are matched by outputs via evaporation, groundwater and surface outflow. Where there is an ongoing change observed in the storage of water in a wetland, either increasing or decreasing, this is likely due to changes in the hydrological regime or physical changes to the wetland. A positive ΔS usually indicates a rising water table, while a negative ΔS indicates a lowering water table (Campbell & Jackson 2004).

Figure 2.2 shows the wetland water balance diagrammatically.

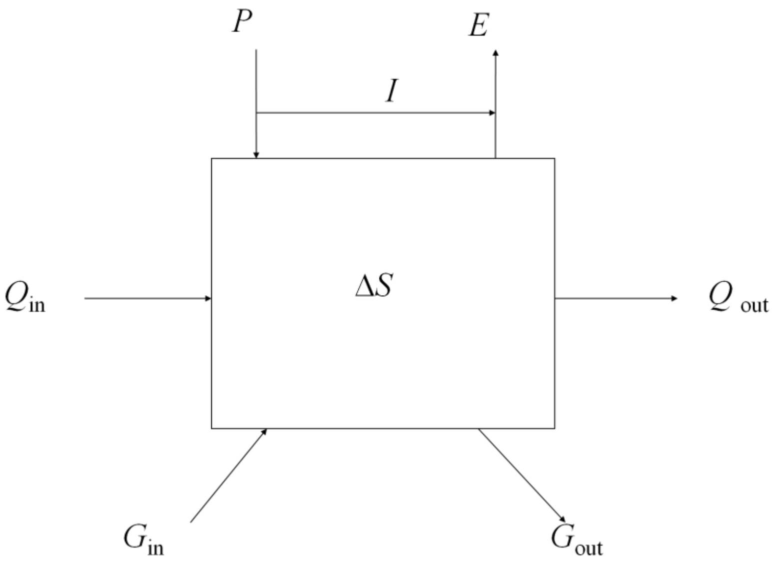


Figure 2.2: Water balance model for a wetland using terms from Equation 2.2. *I*= interception loss from foliage (Campbell & Jackson 2004).

2.5.2 Saturation excess overland flow and near surface runoff

Ultimately flooding is related directly to soil moisture conditions. Historical studies by Carter *et al.* (1978) and Verry and Boelter (1978) found wetlands across America (particularly in Minnesota) appeared to reduce flood peaks by storing water temporarily or until the water table was at the surface, after this point saturation is reached and runoff occurs. Therefore when the water table is lower during dry periods, greater storage is available until the water table is at the wetland surface and soil stores are saturated. This is supported by a study by Branfireun & Roulet (1998) in a wetland in Ontario-Canada, Emili & Price (2006) from a forest wetland in Canada, multiple African wetland studies reviewed by Von der Heyden (2004) and Andes headwater wetlands observed by Roa- García (2009). They all found that wetland runoff processes were directly related to antecedent moisture conditions, or the saturated moisture content of the soil before a rainfall event. During rainy seasons, flood storage (retardation) was low as wetland water storage was close to capacity. After a large storm event, water movement was via overland flow.

Evans *et al.* (1999), Brassard *et al.* (1999), Holden and Burt (2003a; 2003b) and Haigh (2006) observed various wetlands in the United Kingdom (UK). They found that rapid generation of near surface runoff is characteristic of wetlands, often occurring at water table depths within 5 cm of the surface (or in the acrotelm layer). This soil/peat layer is only saturated after high levels of antecedent precipitation, when the water table rises above the permanently saturated catotelm. Around 98% of the runoff collected was from the peat surface to 5 cm depth (Holden & Burt 2003a; 2003b).

2.5.3 Groundwater inputs in wetlands

Wetlands are often positioned in low topographical positions, such as depressions or basins. As these low points often intersect with the local or regional water table, wetlands develop. Hence groundwater (GW) is a significant component of water movement in, through and out of wetlands.

Wetlands have often been compared to sponges, releasing stored water during dry periods into surface water streams that exit downstream. There is some merit in this argument, where a modelling study was undertaken by Smakhtin & Batchelor (2005) in Rustenburg, South Africa of a catchment with a wetland (measured) and without a wetland (experimentally removed from the model). A total contribution of 13% baseflow to the stream from the existing wetland was identified. This contribution to stream baseflow was only obvious during dry years, whereas in wet years wetland water levels were sustained by groundwater, precipitation and catchment surface water inputs.

It is more likely that wetlands act as conduits for water stored elsewhere in the catchment and the sponge theory developed from what is actually groundwater flowing through a wetland and sustaining high flows. One such example of this occurring was in a British Columbia forest stream that was sustained by deep groundwater contributions flowing through a headwater swamp which contributed up to 73% of streamflow (Fitzgerald *et al.* 2003). This theory is also supported by

Bowden *et al.* (2001) who looked at two headwater catchments containing bog wetlands in Otago, New Zealand. They found water storage after rainfall events from these wetlands could only sustain baseflow for a few days, while groundwater flowing through the wetlands (from surrounding deep unsaturated loess horizons) could sustain flow for up to four weeks.

2.5.4 Water storage

Water storage is an important and major component of wetland hydrology. Many wetlands naturally have a deep layer of peat which has a low bulk density and a high water storage capacity.

The storage capacity of wetland soils is related to the position of the water table in the soil profile. Greatest runoff in a variety of American wetlands was found to occur when the water table was high due to the low availability of wetland storage capacity (Carter *et al.* 1978). A study of Wales headwater wetlands by Haigh and Kilmartin (2003) found the lowering water table in summer (by evaporation) allowed greater storage of water. Field measurements in a wetland in the UK by Haigh (2006) found peat can hold 25 times its own weight in water, and have a volumetric moisture content of 86–94%.

During dry conditions water tables in wetlands are lower and the available storage capacity is higher. Wetland depressions in Devils Lake Basin, North Dakota store 811,000 cubic decametres of water. They store approx 72% of runoff volume from a 2 year frequency flood and about 41% runoff volume from 100 year frequency flood (Ludden *et al.* 1983). This available storage capacity is reduced during long periods of rainfall where water tables rise to the surface. Kvaerner & Klove (2006) found available water storage ranged from 150–520 mm for various wetlands catchments in Oslo, Norway. An analysis of the flood mitigation capacity of wetland soils in Momoge Reserve, China, based on soil moisture capacity and soil water storage, found the total amount of water the wetland could store was 7.15×10^4 m^3 km^{-2} yr^{-1}. When translated into an economic benefit for flood mitigation the wetland was valued at \$5700 km^2 yr^{-1} (Jiang *et al.* 2007).

2.5.4.1 Peat oscillation

When water tables are high runoff will be rapid and flood mitigation will decrease. Lower water tables will allow greater amounts of water to be stored. This statement is supported by a study of Opuatia wetland, New Zealand by Fritz *et al.* (2008). This study looked at peat surface oscillation (PSO) over time in response to water level changes. They found PSO was up to four times greater during wet seasons than during dry seasons, and resulted in a large increase in the unsaturated zone during the dry season (and ultimately storage capacity for the wetland).

PSO can only occur in areas of peat build up, such as in bogs. As peat has a low density, is made up of fiborous organic matter and has large water content (greater than 85%), there is the ability for the bog to shrink and swell in response to water level changes. Higher rainfall and saturation will lead to peat surface levels swelling, or ‘floating’ on top of the water table. When the water table lowers, peat compresses due to the weak peat matrix (which is usually supported by pore water pressure). The total fluctuation is termed peat surface oscillation (PSO) and requires accurate and frequent water level measurements to detect (Price *et al.* 2005; Fritz *et al*. 2008).

A linear relationship between surface elevation (SE) and absolute water level (AWL) has been found in many studies for floating wetlands and implies peat surface elevation is driven entirely by changes in AWL. AWL is the water level elevation relevant to sea level or another datum and is measured from a fixed reference peg anchored into mineral substrate below peat layers (Figure 2.3). SE of the peat is determined by subtracting AWL from relative water level (RWL). RWL is the vertical distance between the peat surface and the water table (unsaturated zone). PSO is plotted based on the peat surface elevation (mm) vs. AWL (mm). The 1:1 linear relationship fails when water tables exceed surface elevation (Chapter 6) (Fritz *et al.* 2008).

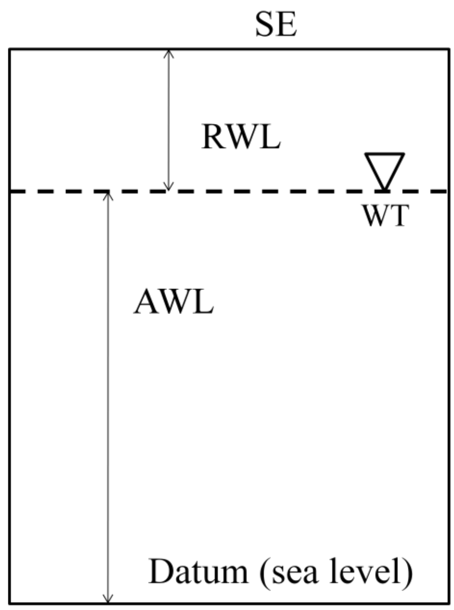


Figure 2.3: Diagram explaining the definition for AWL and RWL relative to the water table (WT) and surface elevation (SE).

2.5.5 Precipitation and evaporation

Precipitation is a key input of water for all wetland classes, and for bogs it is the only water source (Campbell & Jackson 2004). Precipitation is usually low in nutrients, hence wetlands receiving only precipitation as an input are often nutrient poor and termed ombrotrophic (Charman 2002; Campbell & Jackson 2004). Precipitation is seasonal and dependent on location. In New Zealand's North Island, highest rainfall usually occurs during winter.

Evaporation involves a phase change of water from a liquid to a gas, in the form of water vapour. This requires energy from the sun or the environment, and hence greater evaporation occurs on sunny days, especially during summer when day length is longer (Equation 2.3).

$$E = E_t + E_o + E_I + E_s \qquad 2.3$$

Evaporation (E) includes interception loss (E_i), transpiration related to plant photosynthetic activity (E_t), evaporation off open water surfaces (E_o) and from moist soil (E_s) (Campbell & Jackson 2004).

Interception loss involves the evaporation of precipitation intercepted by canopy foliage (i.e. leaves, branches) after a rainfall event. External conditions such as wind, saturation vapour pressure deficit and solar radiation (ultimately controlling temperature) are the main influences on evaporation rates in a wetland.

Many studies have concluded one of the main forms of water movement out of a wetland (depending on class) is due to evaporation. Evaporation in very dry summers caused a significant drop in the water table in wetlands of Antwerp, Belgium which meant less water flowed through the permeable acrotelm layer (Boeye & Verheyen 1992). Balek (2003) found evaporation in the Bangweulu Swamp in Central Africa led to a 60% loss of total inflow to the swamp. High interception loss and transpiration were found to contribute to the low flows. Eaton *et al.* (2001) found transpiration losses to be large in wetlands of the Canadian sub arctic in spite of a relatively low vapour-pressure deficit over the surface.

While wetland conditions naturally result in a high affinity for evaporation, plant physiological and structural controls often create restrictions on water loss from the wetland surface. For example, plants undergo atmospheric exchanges through photosynthesis, where water vapour and oxygen are substituted for CO_2. Under increasing stress (from higher solar energy, and lowering water tables), plant photosynthesis may be reduced and potentially could stop altogether. This is often observed when stomata, which are present on leaves and involved in gas substation with the atmosphere, are closed by guard cells (surrounding each stomata) to prevent gaseous exchange or water loss (Mitsch & Gosselink 2007).

Campbell & Williamson (1997) studied Kopuatai wetland in New Zealand and looked at evaporation rates in raised peat bogs. They found that even when the peat was close to saturation, evaporation rates were lower than expected. This was attributed to a high canopy resistance formed by *E.minus*, due to the dense growth of this species allowing little diffusion of water vapour from the peat surface and plant physiological controls. Similar findings were found for the restiad bog species *Sporadanthus ferrugineus* by Thompson *et al.* (1999) in Kopuatai and Moanatuatua wetlands of the Waikato region, New Zealand. Evaporation rates were constrained at these sites by a combination of plant physiological and canopy factors which prevented evaporation from the peat surface. Consequently, these species can reduce evaporation rates in bogs, sustaining a moist peat surface and provide favourable (wet and anaerobic) conditions for peat formation.

Maintaining these species in bogs is important, and is supported by a study of Kopuatai wetland, undertaken by Thornburrow *et al.* (2009), focussing mainly on the hydrology of the peat dome and how this influences vegetation composition. A conceptual hydrological model (water balance) was created and an in-depth understanding of the hydrological processes occurring at the site was obtained. Management recommendations were targeted towards maintaining high water levels (to reduce peat degradation), with methods such as drain damming and engineering solutions (weirs).

2.6 Vegetation– New Zealand wetland types

The vegetation composition in wetlands is strongly determined by hydrology.

Bogs are fed by precipitation which provides favourable growth for oligotrophic plant communities that eventually form into elevated peat domes and are termed ombrogenous wetlands (Mitsch & Gosselink 2007). Peat domes are low in nutrients, so are dominated by species which can survive these conditions. In New Zealand, restiad bogs are generally dominated by *E.minus* and *S. ferrugineus*. Additionally species such as *G. dicarpa* and *Baumea* are present (Clarkson 2002).

Fens have hydrological inputs from precipitation, groundwater and occasionally surface water. The concentrations of nutrients are higher than in bogs, but lower than what would be found in swamp conditions. Hence these fen systems are often classed as mesotrophic (Charman 2002; Ingram 1983). In New Zealand, and in particular the Waikato region, fens are dominated by *Baumea* spp. and *Schoenus* spp. Additional species which can tolerate a range of nutrient conditions are *G. dicarpa*, *P. tenax* (New Zealand flax), *L. scoparium* (manuka), *Dianella nigra* (turutu), and *C. tenuicaulis* (swamp coprosma) (Clarkson 2002).

New Zealand swamps are typically dominated by species which can survive in dynamic environments with higher concentrations of sediment and nutrients (such as *Polygonum persicaria*, also known as willow weed). When the water level is not too high, sedge species such as *Carex* are present. *T. Orientalis* (raupo) is present in swamps in a range of conditions, including deeper water. Willow species *S. cinerea* and *Salix fragilis* (crack willow) were introduced in New Zealand and are invasive. They can dominate swamps if uncontrolled, outcompeting natives and shading out shorter plant species (Clarkson 2002; Clarkson *et al.* 2004a). Also willows have high transpiration rates that can lead to wetlands drying out and a lowering of water tables (Charman 2002).

2.6.1 Vegetation functional adaptations

Wetland vegetation has to survive in saturated and sometimes anaerobic conditions. Often soils are oxygen deficient which can make it difficult for plants to exchange gases (Smith & Smith 2000). Plant roots cannot respire in saturated conditions and anaerobic metabolism for extended periods results in a build up of toxic compounds that lead to plant death. Many wetland plants (such as willows) avoid deoxygenation through shallow root systems that allow them to survive in anoxic and saturated conditions by exchanging oxygen with the atmosphere (Charman 2002; Sorrell & Gerbeaux 2004).

Plants that are saturated frequently or survive in variable hydrological regimes have internal anatomical adaptations which transport oxygen to below ground roots and rhizomes. These involve large airspaces (aerenchyma) in roots and stems that allow transport of oxygen from aerial parts of plant to the underground tissues (Sorrell & Gerbeaux 2004). Morphological adaptations in emergent macrophytes include the presence of large amounts of shoot biomass above water. Many have a depth accommodation response, in which leaves or shoots elongate very rapidly in response to water level changes, to maximise survival by ensuring biomass is above the water table and able to photosynthesise (Clevering *et al.* 1996).

Plants control the rate of diffusion and photosynthesis through stomatal pores which are located on aerial structures (such as leaves) and sit between epidermal cells. These cells are called guard cells and control the opening and closing of stomata pores. Stomata generally open during the day and close at night, as a response to light for photosynthesis (Smith & Smith 2000). Light intensity, temperature, atmospheric CO_2 concentrations and humidity are all environmental factors which affect stomatal pores opening and closing.

Species in New Zealand can survive in nutrient poor wetlands (bogs) because of physiological adaptations. Some plants are carnivorous (i.e. *Drosera spp.*), while others have special roots which can gather nutrients from the nutrient poor rainwater. One species that can do this is *E. minus* which can also exude the enzyme alkaline phosphatase to break down orthophosphate from organic matter so that it becomes available for plant uptake (Sorrell & Gerbeaux 2004).

Agnew *et al.* (1993) examined five different wetlands on flat terrains across New Zealand, focussing on ombrotrophic bogs and the presence of *E. minus*. They looked at this species' role in peat formation and its functional ecology. *E. minus* produces a surface mat of negative geotropic roots which form into peat. The base exchange capacity (ability to remove ions from rainwater) in *E. minus* roots was similar to a common bog species in the Northern Hemisphere, *Sphagnum cristatum*, even though *E. minus* has a lower water holding capacity. This was attributed to a functional adaptation, where *E. minus'* long wiry stems direct incoming rainfall (for nutrient adsorption) down to the roots (Figure 2.4).

Figure 2.4: *E. minus* wiry stems as a functional adaptation to direct rainfall to plant roots, while also providing ideal habitat for spiders.

A recent review by Hodges & Rapson (2010) examined the role *E. minus* has as an “ecosystem engineer”, through a fen-bog transition in New Zealand wetlands. The Northern Hemisphere, *Sphagnum spp.* appeared to be unimportant in the transition from fens to bogs in New Zealand, while the restiad species *E. minus* looks to have filled this role.

2.6.2 Hydrology and vegetation

Vegetation composition is directly influenced by changing hydrological regimes. The position of the water table is crucial in determining what vegetation will exist in an area. Hydrological inputs from various sources also control vegetation composition. A change in hydrological inputs to a wetland, such as addition of surface water inflow from a new source, can cause a change in species composition. Fens in particular often have groundwater inputs which bring in phosphorus and result in a change of vegetation composition (Sorrell & Gerbeaux 2004).

Wheeler & Shaw (1995) found plants can serve as indicators of hydrological regimes. Certain species outcompete and survive in conditions which are unfavourable to others. An example would be *E. minus* dominating in low nutrient conditions. A combination of physical hydrology (such as water table position) and chemical hydrology (nutrient inputs present in various hydrological sources) are the main contributing factors towards vegetation composition (Charman 2002; Sorrell & Gerbeaux 2004).

Flood events can play an important role in vegetation composition. Generally speaking, flood events often carry high amounts of eroded sediment (from headwater catchments) and dissolved nutrients from land that has entered waterways from infiltrating rainfall. These flood events have the potential to overtop river banks (and artificial stop banks) and transport nutrients and sediment into wetland sites, changing vegetation composition (Charman 2002).

A New Zealand study by Sorrell *et al.* (2007) looked at soil and vegetation responses to hydrological changes in a partially drained fen. They utilised a before-after-control experiment of restoring water levels to pre drainage levels to observe the wetland response. Soil profiles, vegetation plots (and foliage samples) and continuous water level records were taken at the site, along a moisture gradient. The wetland experienced frequent short but intense floods, and vegetation analysis showed a gradient in species from pastoral communities in dry marsh areas, to wetland sedges in the centre of the wetland. Weed species dominated the zone of large water table fluctuations and dry conditions over summer, and all vegetation showed a high correlation with environmental variables bulk density, soil nutrient content and redox potential. Re established (artificial) flooding reduced pastoral communities and redox potential in dry areas (similar to controlled flooded zones), but had no affect on N and P availability. This supported the idea of hydrological manipulation to reduce soil oxidation, peat break down and the potential to alter wetland plant communities (towards native vegetation).

2.7 Nutrients

Nutrients are a major controlling factor for many aspects of wetland ecology. Often nutrient concentrations are directly relevant to the hydrological regime present at the site. Nutrients are a fundamental building block for life and their presence in water and peat can directly influence vegetation composition (Sorrell & Gerbeaux 2004). Wetland plants are often adapted to unique environments, such as *E. minus* colonising bogs with few nutrients or alternatively *P. persicaria* surviving in periodically inundated soil with high nutrients (swamps).

Biological productivity in wetlands is limited by the amount of nutrients present in soil and water which is often determined by the breakdown of organic matter by microbes. In wetlands, the most common limiting nutrients are nitrogen and phosphorus (Sorrell & Gerbeaux 2004).

2.7.1 Microbial processes for nutrient removal and turnover

Ammonification is the first step in the nitrogen cycle and involves the conversion of organic nitrogen (such as amino acids and proteins from decaying matter) to inorganic ammonium (NH_4^+). Another process that commonly forms ammonium involves hydrolyses of ammonia (NH_3^+) in the soil (which occurs in high concentration in ruminant urine) (Di & Cameron 2008).

Nitrification occurs under aerobic conditions, and involves the conversion of ammonium (NH_4^+) to nitrate (NO_3^-) through oxidation by microbes. Nitrification requires aerobic conditions, so is generally limited in bog systems due to high water tables (Sorrell & Gerbeaux 2004).

Under saturated anaerobic conditions, denitrification occurs which involves the conversion of nitrates to nitrogen gas. The output from this process is essentially gaseous forms of nitrogen and nitrous oxides, N_2, N_2O, NO (Di & Cameron 2008). This process is the main form of nutrient removal in wetlands and is an important mechanism for restiad bogs having low nutrients availabilities.

Mineralisation converts organic nitrogen into inorganic forms available to plants and occurs under aerobic conditions. Under anaerobic conditions, immobilisation may occur, which locks nutrients away from plant uptake. If there is an abundance of carbon, but limited nitrogen, immobilisation will take place as microbes breaking down the organic remains will use the nitrogen and make it unavailable for plants (Sorrell & Gerbeaux 2004).

2.7.2 Wetland pH and conductivity

Naturally, most wetland soils are often acidic with different classes having varying degrees of acidity. pH is determined by the concentration of H^+ protons present in a solution. Higher concentration of H^+ protons results in an increasing acidity (pH 0–7). Neutral pH is 7 while alkalinity (basic) ranges from pH 8–14 and involves an increasing concentration of OH^- ions. Various molecules form under increasing acidity, but common acid forming molecules are HCl (hydrochloric), HNO_3 (nitric) and H_2SO_4 (sulfuric) (Brown *et al.* 2003).

Peat and some peatland plants have a high cation exchange capacity (CEC). In a broad sense, this generally means the ability to remove cations (positive ions such as NH_4^+) from solution and replace these with H^+ ions, which reduces pH. In New Zealand the root layer of *E. minus* has a CEC as high as *Sphagnum spp.* which are particularly efficient at removing cations from solution in raised mires (Charman 2002).

The cause for this acidity is due to a variety of processes, but the main mechanism is the uptake of the nutrient ammonium (NH_4^+) by plant roots. In swamps and fens, groundwater and riverine water (often alkaline in nature) can buffer the pH, resulting in decreased acidity, while bogs generally remain acidic because rainwater has little ability to buffer increases in H^+ concentrations (Sorrell & Gerbeaux 2004).

Electrical conductivity (EC) is a measure of the ability of a solution to conduct an electrical current, through the presence of ions (such as salt ions like Na^+ and Ca^+) in a solution. Higher EC may indicate more sediment deposition, where sediment is the primary carrier of bonded ions which can become soluble in water (Brown *et al.* 2003). Swamps and marshlands have higher EC, due to greater deposition of sediment and nutrients from surface waters.

2.7.3 Carbon

Carbon is the dominant constituent of organic matter (OM). OM decomposition, and mineralisation rates are higher in bogs near the top 10 cm of surface layers (acrotelm) but deeper than 10 cm decomposition slows dramatically. This is due to the perched water table sustained by bogs which is the main reason they accumulate peat (Verhoeven *et al.* 1990). Labile (readily available) pools of carbon are what could be considered available for microbial uptake and important for biotic life and are around 10% of total carbon availability (Bridgham *et al.* 1998).

2.7.4 Phosphorus

Phosphorus (P) occurs mainly in phosphate minerals. The principal source is phosphate rock. Most of the phosphate present in these sources are insoluble and is often converted to a soluble form for use in fertilizers. The mineral form of phosphate is often strongly bound in soil particles and unavailable to primary producers. When treated with phosphoric acid, triple superphosphate can be formed. Essentially this technique forms a soluble orthophosphate (PO_4^{3-}) which can be assimilated by plants, but also leached through soil to contribute to eutrophication (Brown *et al.* 2003). The availability of phosphorus is also determined by the difference between plant uptake and the rate of conversion of organic matter to orthophosphate by microbes (Sorrell & Gerbeaux 2004). Phosphorus can also be immobilised like nitrogen and become unavailable as it is locked in the organic matter fraction which generally accounts for 50–90% of total P (New Zealand Institute of Chemistry 2010).

P can also become unavailable through sorption with ions present in sediments. Under aerobic conditions such as during a low water table, Al and Fe oxides (in solution) bond very strongly with P and reduce its abundance for plant uptake (Craft 1996; Venterink *et al.* 2002). High water tables forming reducing conditions can lead to P becoming more available. Adsorption occurs when the ions in soil (Al, Fe, Ca, K) attach with phosphate ions and and are removed from solution, while absorption occurs when the phosphate ions then diffuse into the solid. Both of these mechanisms combined are known as sorption (New Zealand Institute of Chemistry 2010). Sorption of P could result in a reduction in plant available phosphorus even though volumetric concentrations are higher in mineralised swamplands (Craft 1996; Bridgham, *et al.* 1998).

The bonds between phosphorus and sediment are very strong, and usually are greater in clays which have a high surface area and large cation exchange capacity (positive ions such as Al and Fe imbedded in the surface of clays which bond with phosphorus) (McGill & Cole 1981). Wetland studies have indicated that higher TP content in minerotrophic wetlands appears to be offset by greater immobilisation due to geochemical sorption. Higher mineral content and aerobic conditions results in large amounts of TP being removed from the pool and hence while concentrations are high, availability is low. Additionally the low N:P and C:N ratios (less than 20) in mineralised swamps suggest higher amounts of mineralisation, but potentially due to the large amount of sediment and P binding, mineralisation may be low (Bridgham *et al.* 1998). Labile P (plant available) has been found to be much greater in bogs, around 33% of total P, while in aerobic swamps labile P values were at 1% of total P (Bridgham *et al.* 1998; Verhoeven *et al.* 1990). A New Zealand study by Cooke (1992) found wetlands which receive high levels of Fe and Al can remove large amounts of P, even concentrations which come in from sewage effluent. It was hypothesised that high P retention in wetlands can be sustained if Al and Fe inputs remain large.

2.7.5 Nitrogen

Nitrogen constitutes 78% of the earth's atmosphere in the form of N_2 and is a fundamental building block of life. The two most relevant forms of nitrogen in a wetland are nitrate (NO_3^-) and ammonium (NH_4^+) (Brown *et al.* 2003). The nitrogen cycle is complex and involves many steps through the conversion of nitrogen into useable plant products. A lack of oxygen in wetland soils also restricts nitrification (production of nitrates) and enhances denitrification, which subsequently means nitrates are limiting in wetlands. Optimum plant growth occurs when the NH_4^+: NO_3^- ratio is around 2:1 (Sorrell & Gerbeaux 2004). In swamps and some fens, nitrogen can be more abundant due to inputs from surface and groundwater systems, which can allow greater plant growth. Generally the lack of nitrogen in wetland soils means it is a limiting nutrient for plant growth.

N and P can be found in both organic and inorganic forms, the latter of which is directly available to plants. More than 95% of N stored in soils is typically in organic forms and is covalently bonded to carbon (Craft 1997; Schipper *et al.* 2004).

Bogs have low concentrations of N due to very low concentrations in rainwater. There are likely to be high levels of immobilisation due to large amounts of undecomposed organic matter and coupled with the low N concentrations. However, other wetland studies (in bogs) show that nitrogen can potentially be more abundant, as decomposition of plant material is slow. The anaerobic conditions usually sustained by high water tables result in low microbial activity, so any organic material broken down into inorganic N forms can potentially be rapidly taken up by plants, rather than being consumed by microbes (Verhoeven *et al.* 1990; Bridgham *et al.* 1998). In bogs N mineralization has been detected at C:N ratios between 60 and 100 (Damman 1988).

2.7.6 Heavy metals and elements

There are a variety of heavy metals which are present in solution and in strongly bound forms with soil particles. Copper (Cu), Zinc (Zn), Iron (Fe) and Aluminium (Al) are some metals commonly found in high concentrations when sediment is present, as their ionic structures are attracted to the negatively charged soil particles (Ministry of Environment 1999). Under low pH and anoxic conditions (present in wetland sites), many of these heavy metals can become soluble and potentially could lead to toxicity, affecting plants and animals (Brown *et al.* 2003). They can also bind strongly with P (section 2.7.4).

Potassium (K) and calcium (Ca) are strongly affected by the soil processes that control P availability. A study of 44 swamps and fens in Western Europe indicated that high concentrations of N, P and K along a nutrient gradient (poor-rich) also resulted in a change in species composition and increased productivity due to greater nutrient availability. Generally it was found that a lowering of the water table, resulting in increased aeration, led to greater soil turnover and nutrient availability and hence a change in wetland class (Venterink *et al.* 2002). Increases in Ca concentrations are likely attributed to fertiliser inputs coming in from pastoral farming. Lime ($CaCO_3$) is often applied with various fertiliser mixtures to increase soil pH (Roy 2001; Fertiliser New Zealand Ltd 2010). Erosion of soil particles in the upland Whangamarino catchment which are deposited in the wetland will likely lead to increased Ca concentrations and more sorption of P.

Phosphate rock commonly used in fertilisers is known to have relatively high concentrations of heavy metals such as arsenic, cadmium, chromium, lead, mercury, and uranium. Cadmium (Cd) and uranium (U) are often considered among the worst elements for human and environmental health (Roy 2001). Unfortunately, economic considerations have prevented the adoption of technologies that remove Cd and U from phosphate rock and/or phosphoric acid, hence high concentrations are observed where fertiliser inputs are occurring (Roy 2001). Cd is relatively immobile in the environment (and in solution) and has been shown to be highly correlated with total phosphorus concentrations in the soil, principally through the use of superphosphate fertilisers (Roberts *et al.* 1994;

Roberts & Longhurst 2002; Stevenson 2004). Additionally Cd can accumulate in foliage in pastures and lead to toxicity in animals (Roberts *et al.* 1994; Roberts & Longhurst 2002).

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

Naturally, most of nitrogen is in the $\delta^{14}N$ form, comprising 99.63% of all nitrogen on earth. The remainder is in the isotope form $\delta^{15}N$ (Robinson 2001). Biological processes such as denitrification (under anaerobic conditions) can lead to enrichment in the remaining nitrates with the slightly heavier $\delta^{15}N$ isotope, as microbes can break bonds (of nitrate to nitrogen gas) with the $\delta^{14}N$ form of N easier than with the $\delta^{15}N$ state (Robinson 2001; Bedard-Haughn *et al.* 2003). Stable isotope ratios are usually expressed in delta (δ) units and a per mil (‰) notation relative to the respective international standards. $\delta^{15}N$ is reported to atmospheric air (AIR). A positive value indicates enrichment in heavy isotopes, negative indicates depletion (Panno *et al.* 2001; Xue *et al.* 2009). Fertilizer $\delta^{15}N$ ranges from 0.5 to 5‰ for oxidized N (NO_3^-); with lower values for the reduced form (i.e. NH_4^+) or urea. The isotope can be used as a tracer to identify different sources of nutrient movement in or through a system. Fertilisers are often refined under strict procedures and hence usually have a consistent range of isotope enrichment. In a general sense, $\delta^{15}N$ can be a good indicator of potential nutrient inputs at a site (such as Whangamarino wetland), but due to a variety of different biological processes which can contribute to isotope enrichment, identification of the direct cause for such increases in nutrient concentrations cannot always be made (Panno *et al.* 2001; Xue *et al.* 2009).

2.7.8 Peat chemistry in New Zealand wetlands

Chemistry has been used to help characterise wetland types in New Zealand. Commonly used to distinguish between wetland class are total nitrogen and phosphorus, total carbon, pH, and the ratios C:N, C:P, N:P (Table 2.3).

Swamps (and fens which have an intermediate range between bogs) are fed by surface and groundwater. This generally results in a higher pH (averaging 5.2) and greater concentration of TN and TP. Bogs are fed by rainwater, which is low in nutrients and pH (4.0) (Clarkson *et al.* 2004b).

Table 2.3: Means and ranges (in brackets) for soil parameters at 17 swamps and six bogs sampled in New Zealand (Clarkson *et al.* 2004b), and 6 swamp and 22 bog sites in Opuatia wetland (Browne 2005). TN= total nitrogen, TC= total carbon, TP= total phosphorus.

	NZ Bogs	NZ Swamps	Opuatia Bog	Opuatia Swamp
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)
TC (mg cm^{-3})	92.7 (24.1-239.8)	39.8 (5.2-100.6)	33.3 (24.2-43.29)	37.8 (29.8-47.4)
TN (mg cm^{-3})	0.82 (0.02-1.83)	2.12 (1.15-3.24)	1.35 (0.7-1.98)	2.4 (1.7-2.8)
TP (mg cm^{-3})	0.08 (0.01-0.20)	0.28 (0.15-0.59)	0.08 (0.03-0.13)	0.26 (0.18-0.33)
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)
C:P	1904 (533-4221)	163 (45-435)	507.1 (236.9-1041.8)	161.4 (116.3-212.7)
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)

Bogs have slow decomposition rates and high amounts of peat (organic matter). Lack of decomposition means high amounts of total carbon are present. Greater decomposition rates and mineral inputs in swamps and fens (due to a more variable hydrological regime) results in decreased concentrations of total carbon (Sorrell & Gerbeaux 2004).

Additionally, bogs have high ratios of C:N, C:P and N:P (Table 2.3) due to lower availability of N and P. Higher ratios means biological production in bogs is limited due to anaerobic conditions and a lack of crucial nutrients. Swamps on the other hand have lower ratios which suggest they are capable of greater biological production (Sorrell & Gerbeaux 2004).

Generally vegetation growth in bogs is limited by lack of nitrogen and phosphorus. Fens have a slight addition of phosphorus from ground and surface water systems. As foliage only needs small amounts of phosphorus to aid in growth, this leads to the development of vegetation limited mainly by nitrogen. Addition of nutrients to a wetland (from various hydrological sources) will result in a switch from phosphorus to nitrogen limitation and a change from bog to swamp vegetation (Sorrell & Gerbeaux 2004).

2.8 Ammonia deposition

Ammonia (NH_3) enters wetlands through volatilisation off surrounding land. The dominant source of NH_3 (g) emissions to the atmosphere are through agricultural activities (animal wastes and fertilisers), contributing an estimated 90% in Western Europe (Sutton *et al.* 1995; Kirchmann *et al.* 1998). Other anthropogenic inputs of ammonia (such as motor vehicles and waste processing) are thought to be small in comparison to farmland, but are hard to quantify due to the large number of sources (Battye *et al.* 1994**;** Sutton *et al.* 2000). NH_3 (g) has a relatively short atmospheric lifespan, ranging from only a few hours up to 5 days. This is because it can be absorbed by water and react with acid gases (NO_x) to produce aerosols. Ammonia that is converted to aerosols has a longer residence time and can be dispersed over greater distances. This is primarily removed by wet deposition. It will also be dry deposited onto local vegetation or soil until water (i.e. from precipitation) converts it to aqueous ammonium (NH_4^+) (Chimka *et al.* 1997; Scudlark & Church 1999). This can lead to a reduction in vegetation diversity and eutrophication.

Factors which influence the rate of dry deposition of NH_3 are:

1) The properties of the chemical species being measured, such as phase (i.e. gas, liquid, particle), chemical reactivity and particle size. Dry flux of NH_3 is an order of magnitude larger than NH_4^+.
2) Meteorological factors such as wind speed, temperature and humidity.
3) Physical and chemical characteristics of the deposition surface. Deposition over land is greater than water due to higher turbulence, by a factor of 2–5 (Williams 1982).

Dry deposition appears to be heterogeneous in space and time, more likely linked to fluctuating meteorological conditions than changing airborne concentrations (Roadman *et al.* 2003).

2.9 Anthropogenic threats to wetlands

In New Zealand, approximately 90% of wetlands have been drained for various purposes, but mainly for agriculture due to their presence in low lying fertile land (Park 2002). The remaining wetlands are still relatively diverse and have many unique plants and animal species. Wetlands are valued for their biodiversity, food and fibre resources, flood protection and ability to store and remove nutrients (Campbell & Jackson 2004). The wetlands present in New Zealand are still at risk from anthropogenic activities including drainage for agriculture, nutrient enrichment from various land uses, fire and invasive species.

2.9.1 Drainage and hydrological modification

Drainage involves the lowering of a wetland water table, decreasing saturation and increasing the oxygen content in the soils. This allows wetlands to be farmed or used for cropping with greater productivity due to the improved conditions for plant growth. There are many types of processes that control peat degradation and most are modified or increased by drainage. All these processes are somewhat interrelated and can be potentially affected by a simple modification in the wetland environment (such as drainage). Generally as peat decomposes and water table is permanently lowered, plants that colonise and are specially adapted to survive in these environments will likely disappear from the area. These will often be replaced with invasive weed species.

Shearer and Clarkson (1998) investigated water level changes at two bogs on the Whangamarino wetland (Chapter 3). They found a change in water levels, nutrient inputs and regular fires contributed to peat degradation. This allowed the colonisation of invasive species in the bog fringes (such as *S. cinerea*) at an early stage.

Galat *et al.* (1998) studied the significantly modified Missouri River, USA where the lower section had been channelized (over 1200 km), leveed (stop banks) and the banks stabilised for flood control. By the 1980’s over 80% of the wetlands were lost through significant drainage and restriction of flood flows. After a 1:100 year flood event in 1993 overtopped stop banks and caused $13 billion of damages, 20,000 ha of flood plain was re instated for wetlands. Managed or controlled flooding is now actively practiced where controlled flooding is used in some wetlands to maintain habitat for waterbirds and wetland presence. A post flood assessment on these wetlands was undertaken, with an overall positive response in all the biota (fish, invertebrates, birds, aquatic plants, terrestrial vegetation) due to flood events over 5 years. Due to a large range in wetland types, controlled and highly managed flooding was recommended to ensure their survival, as some wetlands (bogs) required little nutrient inputs to maintain condition (and therefore reduced flooding), while floodplain swamps required greater inundation frequency.

2.9.2 Fire

Fires in peatlands can dry out surface layers, breaking down peat structure and directly converting peat to CO_2 which will be lost from the system. The main impacts fires have on peatlands are:

1) The destruction of living biomass: vegetation, animals and soil organisms.
2) The destruction of plant litter and surface peat: reduces the accumulation rate of peat into deeper layers.
3) Release of nutrients: ash from the burn releases nutrients and can change the composition of plants. Additional loss of nutrients by gaseous conversion (carbon dioxide).
4) Hydrological change: initial drying of peat surface from fire and a lowering of the water table. Over time the reduced plant cover lowers the transpiration rate. Eventually this can lead to greater presence of surface water (Charman 2002; Haigh 2006).

In the English peat district >300 wild peat fires were recorded from 1970–75 affecting 8% of the total area (Haigh 2006). The most significant effect was an increase in nutrients and decreased vegetation cover. This will lead to greater invasion of noxious plants and eventually dominance of grass like species (graminoids), resulting in an increase in peat erosion (Figure 2.7) (Bragg & Tallis 2001).

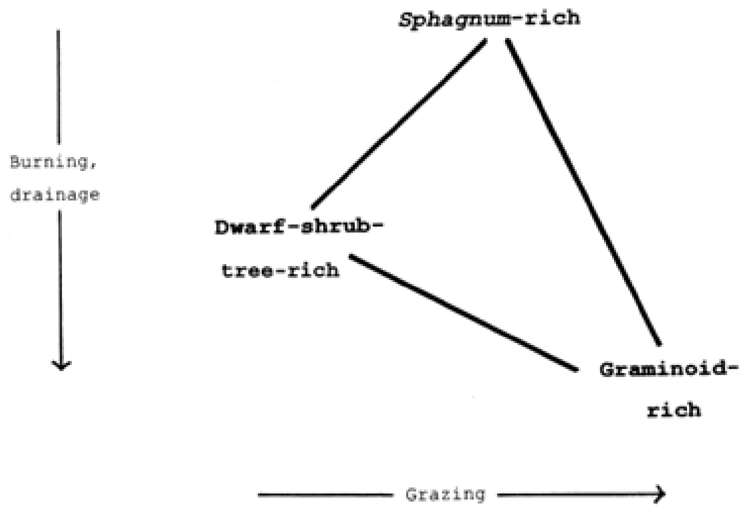


Figure 2.5: Effects of burning, drainage and grazing on wetland species composition (Bragg & Tallis 2001).

Fire disturbance has been a natural occurrence in many Waikato wetlands before anthropogenic burnings. Fire in natural, protected wetlands may be an important contribution to species diversity. Two fires in Whangamarino wetland in 1984 and 1989 and a fire at Moanatuatua bog in 1972, were studied by Clarkson (1997). Vegetation was monitored to determine rates of recovery. Rhizome species colonised rapidly after fires, while species that were eliminated had to recolonise by seed (and took longer). Early colonising, adventive species were present in the first one to two years, likely due to an increase in nutrients, but after 6 to 12 years the sites were back to a similar ecological state pre-fire. A diverse range of species were able to recolonise, and through natural succession in the wetland, original dominant vegetation such as *E. minus* and *Baumea spp.* eventually outcompeted these other plants (such as *Drosera binata* or sundew).

Similar results were found in a study by Timmins (1992), who looked at the impact fires had on Eweburn bog in Te Anau, New Zealand. She found total plant cover had returned 4.5 years after the fire, and species composition and diversity increased greatly in the first two years following the burning. Early colonisers were rhizome species and adventives, and by the end of the study there was still greater species diversity than before the fire. It was predicted that vegetation composition would eventually revert back to that observed pre fire.

The idea that fires can increase species diversity and sustain critically endangered species (such as the rare orchid *Corybas carsei*) was also supported by Norton & de Lange (2003). Controlled fires were lit in the Whangamarino wetland, and the effects on microclimate, vegetation and peat where monitored. The results showed a significant increase in solar radiation reaching the surface and day time soil temperature was higher for four years after the fire. Rhizome species dominated after the fire, but obligate species (such as *E. minus* and *L. scoparium*) did not recolonise to similar numbers. There were cases where invasive *Salix spp.* colonised the plots, when it was not present previous to the burning event. Species richness and diversity increased after the fire, but over 4.5 years began to decline to pre fire composition. *C. carsei* was killed during the fire, but re established 1 year after burning and at greater abundance. Also, flowering and germination of this rare plant increased (Norton & de Lange 2003).

In all studies it was shown that fire actually led to an increase in species diversity, due to rapid colonisation by plants that would otherwise be outcompeted. After a certain amount of time, burned areas return to similar health pre fire, but often with greater species diversity. Fires were also shown to sustain and promote the growth of some rare species (*C. carsei*), which suggests burning could be important in sustaining these species in the future. Yet at the same time large fires risk damaging peat structure and also allowing the colonisation of invasive species which are otherwise absent from the vegetation at that site (Clarkson 1997; Norton & de Lange 2003).

2.9.3 Grazing and erosion

Grazing ruminant animals on peatlands (not including drained areas converted to pasture) can lead to enhanced decomposition rates. Grazing is widely practiced on mires in the British Isles, where it has been suggested that 82% of areas being grazed are substantially modified (Bragg & Tallis 2001). Grazing results in animals eating wetland plant species and decreasing the vegetation cover on peat surfaces. Also they compact the peat surface, increasing subsidence. This severely increases the susceptibility of peat to erosion processes. Reduction in vegetation leads to an increase in common erosion processes including wind and fluvial (water) erosion, aided with freeze-thaw action (Holden *et al.* 2007). Vegetation initially provide some protection of the peat layers from the elements, but once removed by grazing the peat can be directly exposed to the erosion. This will often result in an increased degradation, where peat will be removed from the wetland in the form of suspended sediment or DOC in the water (Bragg & Tallis 2001; Charman 2002; Haigh 2006). Significant erosion in studies of peatlands in Scotland occurred between AD 1500-1700. This has been linked back to a major onset of high intensity storms during this period. This was coupled with many of these wetlands being grazed by stock, and increasing the risk of erosion (Stevenson *et al.* 1990).

Grazing often leads to an increase in weed plants that contribute less to the formation of peat. Seeds can be brought in through ruminant's hooves, or from vegetation removal allowing increased chances for weed plant establishment. These are often woody species such as the *Salix spp*. The presence of these plants can dry out a wetland which increases aerobic degradation and can slowly destroy wetlands from the outside in (Shearer 1997; Haigh 2006).

2.9.4 Subsidence

When peat dries following drainage, the fibrous structure which allows greater than 90% moisture content shrinks, resulting in pore spaces decreasing (Charman 2002; Gronlund *et al.* 2008). Bulk density of peat will increase following drainage as the structure breaks down and becomes more compact. The compression of peat occurs because the weight of the overlying surface is supported by water present within the peat structure (Beheim 2006). Following drainage, this weight is transferred to the weak peat structure, which in turn collapses under the pressure. Hence a surface lowering and subsidence is observed (Whittington & Price 2005; Beheim 2006). It eventually becomes irreversibly physically altered, and hydraulic conductivity decreases which can lead to flooding. This is often coupled with desiccated peat being coated in a hydrophobic resin (water repellent) (Haigh 2006). Studies by Whittington & Price (2005) have shown that water table fluctuations can increase to twice that of a normal wetland following drainage, due to increased compaction and a reduction in hydraulic conductivity.

Chapter 3: Site description

3.1 Introduction

Whangamarino is one of the largest remaining wetlands in the Waikato region, New Zealand. This large wetland (> 7000 ha) is ecologically important because it provides habitat to a range of indigenous species, both flora and fauna. On the fringes, close to farmland and riverine influences (such as the Whangamarino River), mineralised swamplands are commonly found. These swamps are dominated by many introduced species, including *S. cinerea*, which has been reduced in extent due to an intensive control operation by the Department of Conservation. Further into the wetland, the vegetation classes change gradually from a swamp system, to a dominant canopy of *L. scoparium*, and in the centre and areas most isolated from riverine inputs, restiad bogs dominated by *E. minus* & *Baume*a *spp*. have developed (Department of Conservation 2009).

Whangamarino is surrounded by drained former wetlands and upland areas used for pastoral farming and cropping. Additionally, in the upper catchment areas, native and exotic forests are also present. The wetland is bounded to the west by low hills and the Waikato River. The Whangamarino River bisects the wetland, flowing NW towards its confluence with the Maramarua River and the Waikato River. South of the wetland lies Lake Waikare. This lake acts as an artificial flood storage area for the Waikato River and drains into the wetland through the Pungarehu Canal (Figure 3.1). The combined catchment area for the wetland is around 48,900 ha (Department of Conservation 2010a).

This chapter gives an outline of the physical environment of the wetland and includes descriptions of geology, climate and anthropogenic influences. The wetland transect line used in this study will then be described

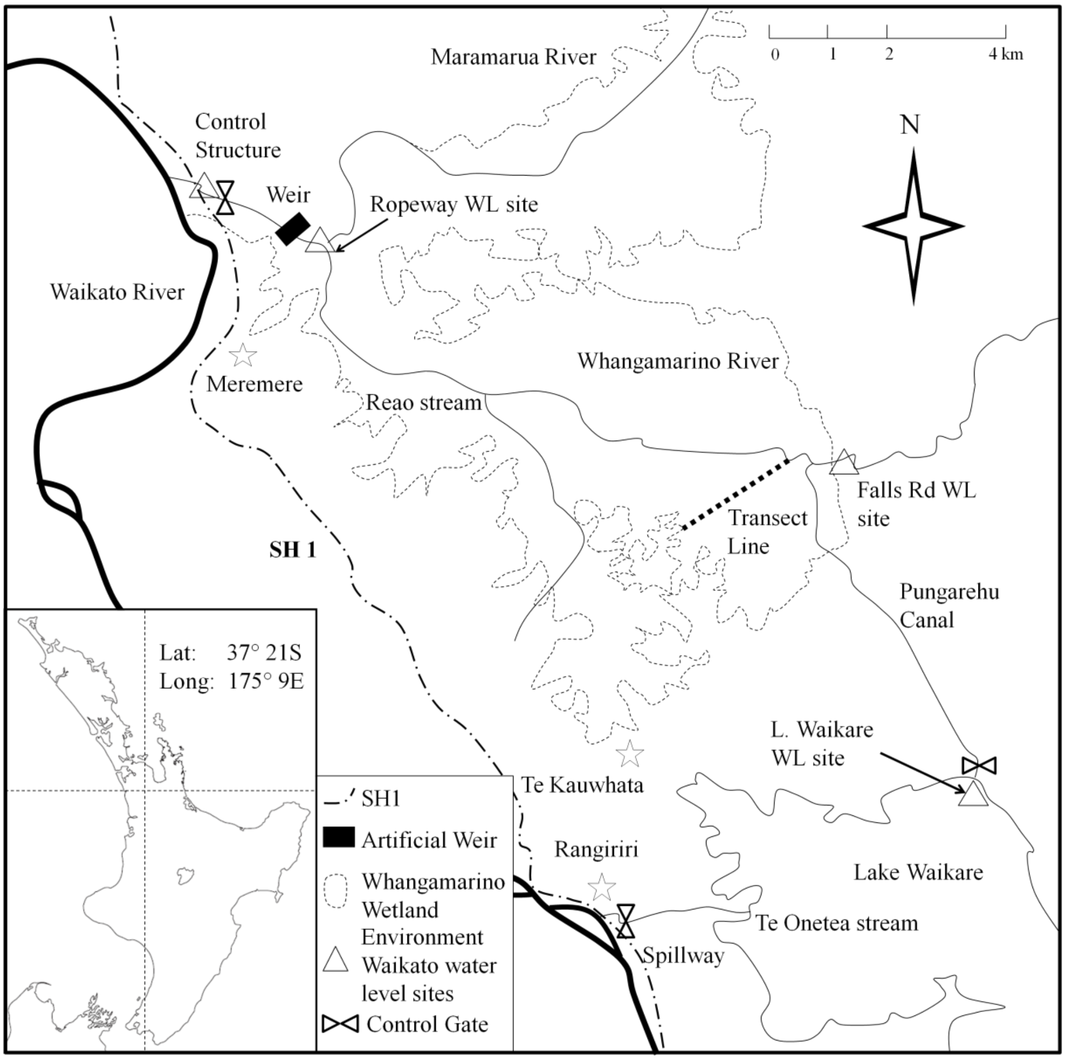


Figure 3.1: Whangamarino wetland and transect line in the northern Waikato region.

3.2 Ramsar convention

The Whangamarino wetland is part of the ‘Ramsar Convention’ (site no. 443) which is an intergovernmental treaty established for the common goal of conservation and wise use of wetlands (for sustainable development). The convention was adopted in 1971, and enforced in 1975. Parties subject to the convention meet every 3 years to make decisions on the management of wetlands. New Zealand signed the convention in 1976, and had a variety of commitments to undertake (Department of Conservation 2010b).

These included:

- To designate at least one wetland of international importance.
- To formulate and implement planning to promote the conservation of Ramsar sites and, as far as possible, the wise use of wetlands generally.
- To report on changes, or likely changes, in the ecological character of Ramsar sites.
- To establish nature reserves on wetlands.
- To promote training in wetland research and management.

New Zealand presently has six Ramsar sites, covering around 566 km^2. These sites are (with listing dates):

- Whangamarino (4-Dec-1989)
- Kopuatai Peat Dome (4-Dec-1989)
- Firth of Thames (29-Jan-1990)
- Manawatu River Mouth and Estuary (25-Jul-2005)
- Farewell Spit (13-Aug-1976)
- Awarua Wetlands (13-Aug-1976)

The 10th conference for the Ramsar convention was undertaken in 2008. The meeting urgently called for people to improve water use efficiency, prevent wetland degradation and loss, restore degraded wetlands and manage existing wetlands wisely (Department of Conservation 2010b). Particular issues and threats relevant to New Zealand were identified, and included:

- increasing water bird conservation
- climate change
- biofuel production
- extractive industries (coal, oil, gas)
- improve monitoring, reporting and management of wetland condition

3.3 Physical environment

The wetland is comprised primarily of three large basins, drained by the Maramarua and Whangamarino Rivers and the Reao Stream (Department of Conservation 2010a). The headwaters of the Whangamarino River include rolling to steep forested and pastoral land. The underlying rock type is greywacke. These streams and rivers flow through almost flat plains and hills of volcanic clays before entering the wetland (Gibbs 2009). The base of the wetland is formed by clay eroded from surrounding hills and varying amounts of elevated siltstone outcrops occur within it. Deep beneath the wetland is a large seam of coal, lying around 400 m below the surface at the Reao Arm, and increasing to around 800 m depth elsewhere (Department of Conservation 2010a). This geological presence is significant as its commercial value (for mining) is in contrast to the wetlands environmental value, resulting in interesting and challenging management issues.

Wetland substrates are primarily peat, which has accumulated from plant remains due to high water tables promoting anaerobic conditions and slowing decomposition rates. Closer to the rivers and stream margins, alluvial sediments mixed with organic matter are common, brought in and deposited during flooding events. The average surface elevation of the wetland is 4.2 m, while in the elevated bog areas heights can be above 5 m, and near the river below 3.5 m (Department of Conservation 2010a). The wetland peat is quite young, only beginning to form around 1400–1600 BP. Hence deep peat layers of greater than 6 m (such as found at Kopuatai wetland) are not present (Shearer & Clarkson 1998).

Approximately 5,000 ha of the wetland are administered by the Department of Conservation. Of this around 500 ha is leased for grazing on a temporary basis. Approximately 730 ha are privately owned by the Auckland/Waikato Fish and Game Council, while the remaining 1,600 ha of wetland are in private ownership (Department of Conservation 2010a).

The climate in this area is typical of the lower Waikato Basin. Average rainfall is around 1,200 mm, ranging from 1,100 to 1,500mm. Mild temperatures average 9°C in winter to 19°C in summer, with a high humidity often present. Ground frosts occur infrequently, although fog is common (Clarkson *et al.* 2004a).

Westerly winds prevail and average annual sunshine hours are about 2,050–2,150 (Department of Conservation 2010a). Wetland water pH ranges from around 3.7 to 6.0 (Clarkson *et al.* 2004a). Table 3.1 presents some climatic data for Whangamarino wetland, retrieved from the meteorological station. Rainfall trends are examined further in Chapter 6.

Table 3.1: Annual climate data at Whangamarino wetland from 1-Jan-2010 to 31-Dec-2011.

Mean annual air temperature (°C)	Predominant wind direction	Total rainfall (mm)	Mean wind speed (ms^{-1})	Mean relative humidity (%)	Mean soil temperature at 0.05 m and 0.2 m (°C)
13.9	W	1007	0.93	81	13, 12.7

3.4 Flood control scheme

The Whangamarino wetland receives water from a catchment of around 500 km^2, and additionally has hydrological inputs from the Waikato River, through Lake Waikare. The wetland currently is utilised as a flood storage area for the Waikato River, preventing flooding of agricultural land (Department of Conservation 2010a). Before the flood storage scheme was developed, flood water from the Waikato River would naturally flow through the Te Onetea stream into Lake Waikare, which at a certain water level would drain into the wetland. Additionally water would also backflow up the Whangamarino River (from the Waikato River), consequentially causing a large ponding area and limited scope for agricultural farming (Mullholland 1991).

In 1961, the lower Waikato-Waipa Flood Control Scheme (LWWFCS) began development of a control structure at the confluence between the Whangamarino and Waikato River (Mulholland 1991). This structure was put in place to prevent backflow up the Whangamarino River (when the Waikato River was in flood) and coupled with stop banks and pumping, agricultural farming could be undertaken (Waugh 2007). The Te Onetea and Rangiriri streams were blocked by the construction of the Rangiriri spillway, preventing the lake draining naturally into the Waikato River. A control flood gate was installed on the Te Onetea stream under the Rangiriri spillway, which can be opened when the Waikato River is high, allowing flow into Lake Waikare (Waugh 2007). The Te Onetea flood gate is only opened when the Waikato River discharge exceeds 1300 m^3 s^{-1} (Mulholland 1991). A third flood gate was installed at the new outlet of Lake Waikare into the constructed Pungarehu Canal, which flows into the Whangamarino River where it enters the wetland.

The wetland and lake can be managed independently for flood storage, holding approximately 94.8 million cubic metres of water, and reduce Waikato River flood peaks by 0.4 to 0.6 m (Waugh 2007; Department of Conservation 2010a). In doing so, water levels in the wetland are held below a maximum of 5.85 m and potentially can store water from flood events up to 1:100 year (Waugh 2007).

Total costs for the development of the LWWFCS over 22 years were estimated to be $195 million (2007 value). The benefit of this scheme for flood protection has been quantified for the significant flood event in 1998 (estimated return period of 45 years, Chapter 6). In 1998, the Waikato River discharge exceeded 1500 m^3s^{-1}, and spilled across the Rangiriri spillway. Wetland water levels reached 5.63 m, and inundation extended from the normal 17 km^2 to 67 km^2. Without the control scheme, some 73 km^2 of adjoining land would have been flooded. The potential damage to this land (if it was flooded) was estimated to be around $5.2 million in 2007 (Waugh 2007).

Due to the commercial extraction of sand in the Waikato River near Mercer, the river's bed has lowered by around 1.3 m since 1958. This resulted in minimum water levels in the wetland and Whangamarino River lowering to around 2.7 m RL, compared to the original height of 3.62 m. Coupled with the flood storage scheme, an artificial weir 450 m downstream of the confluence between the Maramarua and Whangamarino Rivers was proposed (Mulholland 1991). Installation of this rock weir began in 1994 and was completed, after setbacks, in late 1999. The final height of the weir was 2.95 m and allows the minimum water levels in the wetland to be kept around 3.0 to 3.5 m and additionally allow fish passage (Department of Conservation 2007; National Wetland Trust 2009). This ensures the health of the wetland is not adversely affected by excessive drainage.

In times of flood, water overtops and goes around the weir, to be stopped by the control structure at the confluence with the Waikato River (Mulholland 1991). Higher minimum water levels within the wetland only account for a 4% loss in potential flood storage capacity (Waugh 2007).

3.5 Lake Waikare

Lake Waikare is an integral component of the storage of flood water as part of the LWWFCS. The lake surface is maintained at a historically low level and water level fluctuations are reduced from 1.9 m to 0.35 m. As a result large areas of lake margin wetlands have been drained and converted to pasture. The lake currently has a hypertrophic status which is defined as being supersaturated with nitrogen and phosphorus, possessing a high algal biomass, low water clarity and a tendency for algal blooms during summer (Environment Waikato 2007). Nutrient and sediment inputs are dominated by the Matahura Stream (105 km^2), contributing around 5400 tonnes (75%) of sediment, 60% of the total nitrogen and 53% of the total phosphorus flowing into the lake. Te Kauwhata sewage treatment plant contributes 6–10% of total phosphorus into the lake (Environment Waikato 2007). The lake is shallow and undergoes constant re-suspension from wind turbulence resulting in little settling of sediment particles. This is exacerbated by pest fish such as Koi carp and a lack of aquatic macrophytes to reduce sedimentation and nutrient uptake (Environment Waikato 2007).

Since the construction of the Waikare control gate, the lake level is maintained at a maximum of 5.50–5.65 m.a.s.l. During the winter, the lake is often drained to much less than 5.50 m in order to allow greater storage during rainfall events.

3.6 Whangamarino transect line

A transect line (Figure 3.2) extending from farmland to the Whangamarino River was selected by the Department of Conservation. Selection was based on a variety of considerations, including access, health and diversity of the sampled wetland types and changing ecotones (variations in ecological indicators such as vegetation). The transect was initially established by Sinclair Knight Merz (SKM) Environmental Consultancy, where five water level sensors were positioned along the 2.3 km transect. Access to the site is through a farm at the end of Swan Road, or alternatively by walking (or boating) down the Whangamarino River from Falls Road.

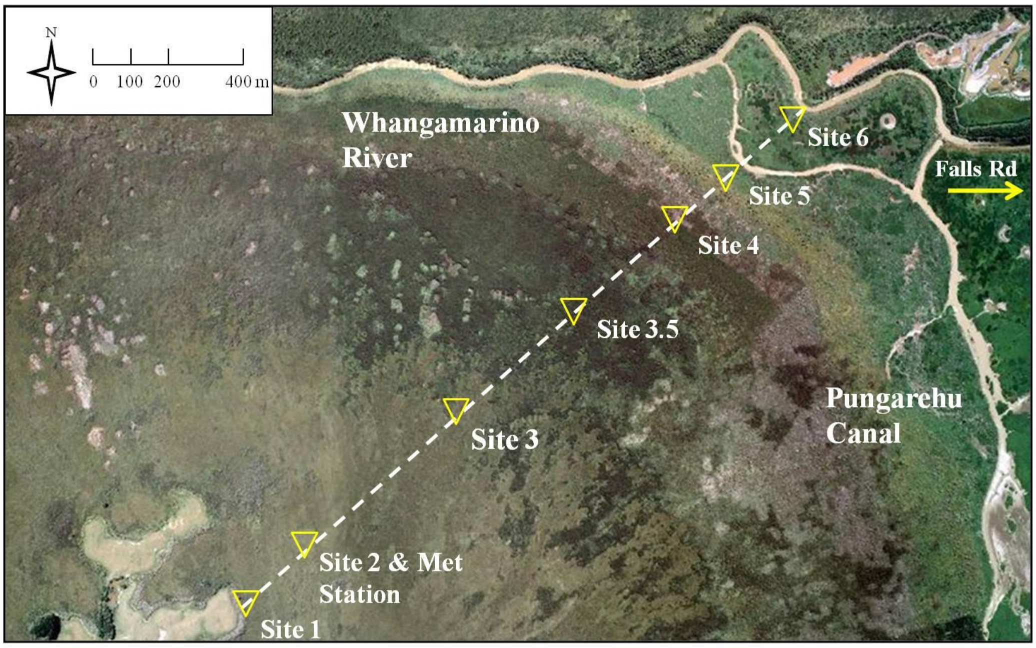


Figure 3.2: Transect line from farmland to the Whangamarino River (aerial photograph courtesy of Waikato District Council and Google Earth 2010).

The University of Waikato in conjunction with the Department of Conservation took over the monitoring of the transect line in early 2009. In April 2009, a meteorological station was installed next to site 2 (250 m from farmland). Downloads of data (stored in a data logger) occurs wirelessly through a cellphone modem with the battery being recharged via a solar panel (Figure 3.3). A range of measurements were taken at the site including relative water level (depth of water table below the peat surface) and absolute water level (Chapter 2), wind speed and direction, air temperature and humidity, solar radiation, rainfall and water/peat temperature (Table 3.2).

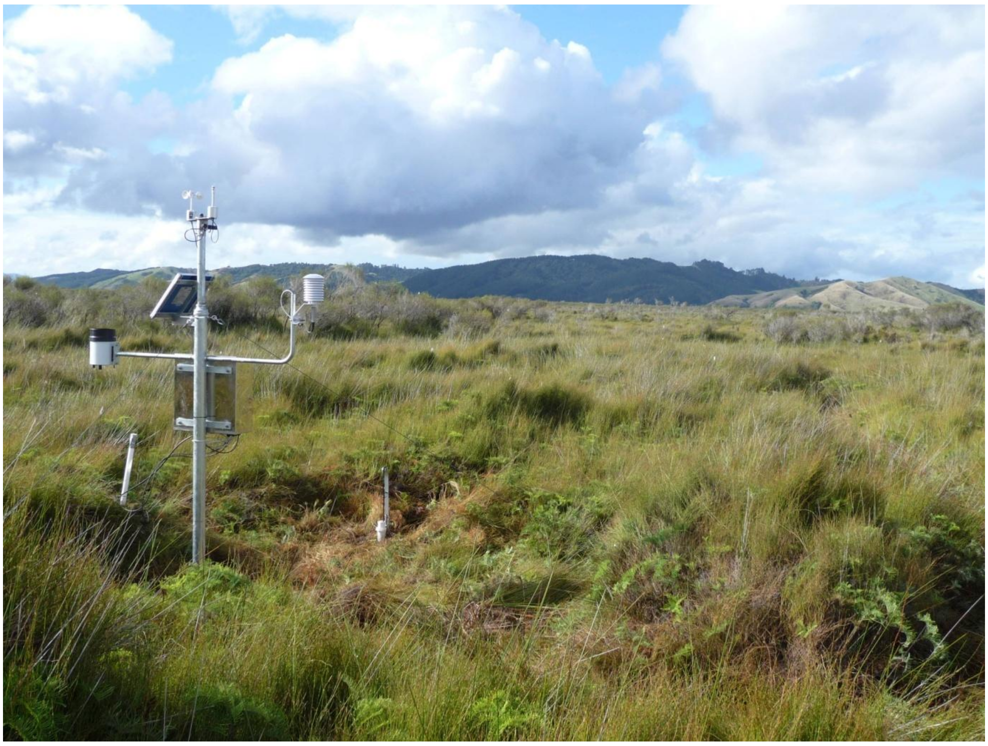


Figure 3.3: Meteorological station positioned 250 m into the wetland. The hills in the background are part of the upper catchment for the Whangamarino River (image courtesy of David Campbell).

Water levels and temperature at sites along the transect (other than site 2, meteorological station) were measured with self logging pressure transducers (Levelogger Gold, Solinst, Ontario, Canada) (Solinst 2009). Water level is the height of the water in a slotted dipwell (Johnson & Gerbeaux 2004). These probes calculate water level through temperature compensated pressure readings, and are barometrically compensated with the aid of a barometric probe (Barologger Gold, Solinst, Ontario, Candada) (which takes air pressure readings and was positioned at site 2, 1 m above the ground). A gap in the water level transect was identified in early 2010, where there appeared to be a dramatic change in water level regimes between sites 3 and 4. DOC funded the instalment of an extra water level site, which was positioned between these two monitoring locations (site 3.5).

Surface elevation changes due to peat surface oscillation in response to water levels were measured using a system similar to that described by Fritz *et al.* (2008) in Chapter 2, section 2.5.4.1. Absolute water levels are calculated relative to a datum (sea surface at 0 m) and are useful to look at elevation gradients along the transect, and also in sites which undergo surface oscillation.

Table 3.2: Instrumentation used at the Whangamarino meteorological station.

Variable Measured	Instrument	Make and Model
Wind speed	Cup anemometer	Vector A101M-11M
Wind direction	Wind vane	Vector W200P 3M
Solar radiation	Pyranometer	Apogee PYR-P Precision SP110
Peat temperature (5 and 20 cm)	Therm-107 probe	Scottech 107-L temperature sensor
Humidity and air temperature	T & RH Probe	Vaisala HMP 45A
Rainfall	Rain gauge	Texas electronics RG-3M 0.2 mm tipping bucket
Water levels	ISD Probes x 2 (RWL and AWL)	Scottech ISD-SS3-5M submersible pressure transducer
	Aerial and modem	Wavecom
	Datalogger	Campbell Scientific CR1000

Four of the water level sites had surveyed reference pegs (length of steel several metres long) installed into the mineral soil below the peat (sites 1, site 2, 3 and 3.5), to provide a fixed reference elevation. Dipwells were constructed from 50 mm slotted PVC pipe and were inserted into the peat adjacent to the reference pegs. The other 3 sites (4, 5 and 6) were installed in areas with no peat and hence did not require a reference peg as the surface was assumed not to oscillate (Table 3.3). Surveying at these sites was undertaken to the top of the dipwell pipe (using RTK GPS ± approximately 5 mm).

Table 3.3: Water level monitoring site information for the Whangamarino transect. SE = surface elevation, RP = reference peg elevation.

Water Level Site	SE/RP (m)	Instrumentation/manufacturer	Start & End of Records	Distance from farmland/river (m)
1	4.88, 6.36	Solinst Levelogger Gold (LLG), model 3001. Solinst Ltd, Ontario, Canada.	10-Aug-2009 to 15-Jan-2011	50, 2250
2 (Met)	4.97, 5.35	Barologger gold, model 3001 (Solinst Ltd, Ontario, Canada),2 x water level ISD-SS3 (Scottech)	17-Mar-2009 (Baro), 9-Apr-2009 (ISD) to 15-Jan-2011	250, 2050
3	4.81, 5.14	Solinst LLG, model 3001. Solinst Ltd, Ontario, Canada.	17-Mar-2009 to 15-Jan-2011	900, 1400
3.5	4.36, 4.85	Solinst LLG, model 3001. Solinst Ltd, Ontario, Canada.	22-Feb-2010 to 15-Jan-2011	1350, 950
4	3.74, na	Solinst LLG, model 3001. Solinst Ltd, Ontario, Canada.	17-Mar-2009 to 15-Jan-2011	1850, 450
5	3.26, na	Solinst LLG, model 3001. Solinst Ltd, Ontario, Canada.	17-Mar-2009 to 15-Jan-2011	1970, 330
6	2.9, na	Solinst LLG, model 3001. Solinst Ltd, Ontario, Canada.	17-Mar-2009 to 15-Jan-2012	2250, 50

Along the transect line, vegetation plots and ammonia samplers (Appendix C) were also established. Soil chemistry samples were taken within these vegetation plots.

Historical water level data for Falls Road (45 years), Ropeway (30 years), the Control Structure (30 years) and Lake Waikare (30 years) were provided by Environment Waikato (Figure 3.1). These water level records were used to describe the historical changes in hydrology and also to help identify the current hydrological issues affecting this wetland. They were particularly important for flood return period and inundation analysis (Chapter 6).

3.7 Human influences

The Whangamarino wetland, like many other wetlands in New Zealand, has been subjected to anthropogenic influences. The primary cause for degradation to the wetland is through drainage, which over the last 150 years has been extensive. Drainage was often undertaken to utilise wetlands for agriculture and cropping, and in doing so reduces the extent of wetlands dramatically (Clarkson *et al.* 2004a: Campbell & Jackson 2004). Around 2700 ha of Whangamarino wetland has been drained (Waugh 2007).

A decreased water table can have dramatic effects on vegetation composition and nutrient abundance in a wetland (Chapter 2). Increased microbial activity, peat breakdown and invasion of weed plants are common in wetlands following drainage. Lower water tables allow increased terrestrial predator abundance, which would otherwise not be able to reach and prey on birds due to high water levels (Sorrell *et al.* 2004).

Fire is a threat in wetlands and can potentially lead to peat degradation. Various studies have been undertaken in Whangamarino, and it also appears that fire can increase species diversity in the following regrowth period (Chapter 2). There are mixed feelings when determining the benefits of fires at Whangamarino. While native and rare species may recolonise (such as the fire affected orchid *Corybas carsei*), potentially invasive plants such as *Salix sp.* also may establish at the site (Norton & de Lange 2003).

A large scale effect on the Whangamarino wetland may be occurring from nutrient inputs, predominantly through hydrological processes. Agricultural farming around the wetland and in the river catchments often involves the addition of fertilizers to stimulate plant growth. Also decreased vegetation cover in the upper catchment results in increased erosion and leaching, which ultimately means sediment and nutrients end up in the water-ways (Parkyn & Wilcock 2004). Adding to the mix is the increased intensification of farming practices, particularly dairy, in the Waikato Region. Leaching of nutrients to the waterways leads to higher fluxes in the wetland, through both the Whangamarino River and the Pungarehu Canal. Coupled with the flood storage scheme, nutrients and sediment are likely having a greater impact on the wetland. Poor water quality affects vegetation (changes species composition), aquatic and terrestrial communities, and diminishes the overall health of the wetland (Sorrell & Gerbeaux 2004). Large flood events bring in sediment and nutrients and have the potential to reach further into zones of pristine, low nutrient bogs, which ultimately may diminish in size and extent.

Steps are being taken to ensure the health of Whangamarino wetland does not diminish, and part of this monitoring is directly related to the research being undertaken. During this study, the ecohydrological state of the wetland will be identified which will aid in informing management decisions and future mitigation and protection of an important and diminishing ecosystem type in New Zealand.

3.8 Previous Whangamarino wetland studies

Whangamarino is a large and important wetland in New Zealand, where a range of scientific studies have been undertaken. Some of the important studies relevant to this research are summarised.

Changes in vegetation composition were assessed by Reeves (1994) over a 50 year timeframe based on maps developed from aerial photos. This timeframe incorporated a variety of significant events which have impacted on this wetland, including sand abstraction (lowering minimum water levels), the development of the flood control scheme and the installation of the artificial weir (to increase minimum water levels). A steeper river gradient caused by sand abstraction resulted in the wetland draining faster and average water levels decreased by 1.3 m, leading to the wetland being dry for longer periods. Large changes were observed in vegetation cover as a response to these developments. The most important in the context of this thesis is the mass invasion of *L. scoparium* (over 500 ha) into the once pristine restiad bog area, which the current transect line runs through. This invasion began after only two years of the flood control scheme being developed and four years of sand abstraction (Chapter 5, Figure 5.13).

Shearer (1997) examined natural and anthropogenic influences on peat development in Waikato restiad bogs, included in this study was Kopouatai, Whangamarino and Moanatuatua wetlands. The latter two have been the most impacted by anthropogenic activities, mainly through lowered water tables affecting peat development. Surface peats in Whangamarino wetland are degraded, and Shearer (1997) attributed this to a lowered water table and raised nutrient inputs from increased surficial water inputs (Whangamarino River and Pungarehu Canal).

Shearer and Clarkson (1998) studied the Whangamarino wetland in greater detail, looking at two restiad bog areas in order to determine the possible effects lowered river levels may have had on peat development. On bog margins close to the river they found a reversal in bog condition from preserved (hemic) peat to degraded (sapric) peat and attributed this to the lower water table and increased nutrients from flooding. This was making the wetland more susceptible to weed invasion (as evidenced by the extent of grey willow on river fringes).

Clarkson *et al.* (2004a) studied vegetation and peat characteristics in the sequence of development of lowland restiad bogs in the North Island, including Whangamarino, Moanatuatua and Kopouatai. Using vegetation classification techniques, vegetation was assessed in all plots from a range of bog sites with varying ages. They identified a sequence of plant establishment through different aged bogs, from sedges to *E. minus,* to the final bog coloniser *S. ferrugineus.* As succession (and age) of bogs increased, von Post (decomposition level) and nutrient concentration decreased in surface layers. *E. minus* was found to have a wide environmental range and shows its ability to act as an early coloniser to established restiad bog species (and ultimately contribute to peat formation) (Figure 3.4).

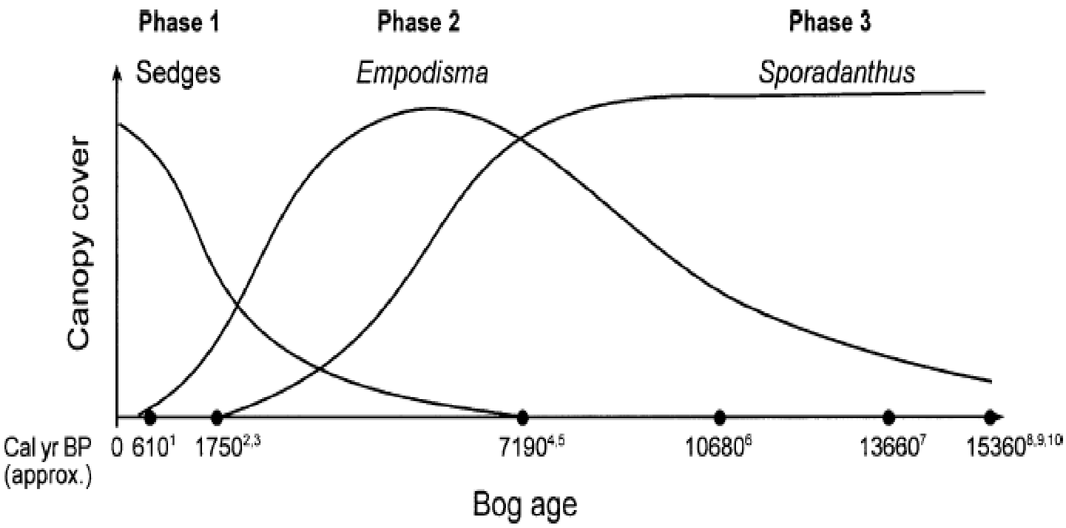


Figure 3.4: Diagram showing species dominance and succession over time during the development of restiad bogs, with the dominant bogs in the Waikato Region presented. 2 = Whangamarino, 3 = Opuatia, 5 = Kopuatai (northern part), 7 = Kopuatai (southern, main part), 9 =Moanatuatua North, 10 = Moanatuatua South (Clarkson *et al.* 2004a).

Chapter 4: Peat and soil characteristics

4.1 Introduction

Peat and soil was assessed along the transect line in order to identify physical and chemical variations that potentially were present due to the influence of a dynamic hydrological regime, natural and anthropogenic nutrient inputs.

A variety of indicators were chosen to aid in describing the current state of Whangamarino wetland and possibly to explain why the wetland is in that condition. These indicators provide insight into the changes along the transect line, most likely attributed to hydrological influences such as water level changes (enhancing decomposition), sediment and nutrient deposition.

Degree of peat decomposition was assessed along the transect line using the von Post technique, which is useful to identify physical changes through field observations. Assessment of soil and peat physical components were undertaken to gain an understanding on the change along the transect line relative to standard methods used in wetlands (Clarkson *et al.* 2004b). This included measuring bulk density, field pH and conductivity (µS cm^{-1}) and volumetric moisture content. Mineral content (% weight) in samples along the transect line were determined via ashing, to identify the sediment inputs from farmland and the Whangamarino River. A range of heavy metals were analysed in samples through ICP-MS, and give an insight into deposition occurring from river inputs (coming in bound with sediment or in a dissolved form). Cadmium (Cd) and Uranium (U) were compared with total phosphorus (also measured through ICP-MS) as these elements are potential indicators of P derived from fertiliser. Total carbon and nitrogen were also analysed through combustion to give an insight into nutrient changes along the transect line. Total carbon provides an indication of the amount of organic matter present in a sample. High levels of organic matter indicate the presence of peat, while lower levels indicate an increased mineral input.

Higher levels of N and P can provide greater nutrient availability for plant production and often lead to colonisation of invasive species which outcompete slow growing natives, of which some are specially adapted to survive in nutrient poor conditions (Sorrell & Gerbeaux 2004). C, N and P ratios (i.e. N:P) are good indicators of wetland nutrient limitations and helpful in classifying wetland classes relative to New Zealand studies (Clarkson *et al.* 2004b). Finally, the stable isotope $\delta^{15}N$ was also measured to determine possible fertiliser inputs and biological processes which may be influencing nitrogen abundances in the wetland

Soil and peat characteristics will later be compared with vegetation patterns and hydrological processes in Chapter 7.

4.2 Methods

Methods used during the analysis of peat and soil are summarised below. In some situations, new methods had to be developed, for which a greater description will be given. Peat cores were collected from the top 10 cm of the surface within each of the established vegetation plots (at 100 m intervals) (Chapter 5). In total, 27 cores were retrieved along the transect line. Three cores were taken at intervals less than 100 m, near water level sites (site 1 at 50 m, site 3 at 900 m and site 4 at 1850 m). The start of the transect began at the first core, taken at 0 m on the farmland/wetland fringe.

4.2.1 Peat and soil collection

At each sampling location, one core was retrieved from the top 10 cm of the peat/soil surface. A standard stainless steel soil corer was used to retrieve these samples, although modified slightly by filing a rough edge on the bottom of the core to slice through peat easily.

1) Sites at 100 m intervals along the transect line were selected using stratified random sampling where distance and direction (north or south) from the track were randomly selected (Chapter 5). Additional samples near each water level monitoring site were taken using the similar random approach.
2) At the sample location, a small patch of vegetation covering the peat or soil was cleared and a core from the top 10 cm was retrieved using a

stainless steel cylindrical soil corer (dimensions 100 mm deep x 75 mm diameter). Compaction was avoided by rotating the corer into the peat, so the filed edge cut through rather than compressed the sample. Hence an undisturbed peat sample could be collected. These samples were stored in a labelled bag (Figure 4.1).

Figure 4.1: Soil cores retrieved using a stainless steel soil corer.

3) Once the core had been removed, a hole (up to 0.6 m) was excavated and allowed to fill with groundwater. This water was used to analyse for field pH and conductivity (µS cm^{-1}) (method 4.2.3).
4) Samples were carried in a backpack until they were placed in a chilly bin at the vehicle. Samples were then stored at 4 °C in a laboratory refrigerator until required for analysis.

4.2.2 Peat humification index

Peat humification was assessed in the field using the von Post technique. This method assesses the degree of peat decomposition by observing the structure, plant remains, colour and wetness of the peat. Decomposition is related to a scale from 1–10, where 1 is very low decomposition while 10 is entirely decomposed. Assessment was undertaken at each site with a small sub sample of peat from the top 10 cm of the peat surface. 23 sites were assessed, stopping within 300 m from the river. The technique was abandoned closer to the river as no peat was present, only mineral soil. Refer to Clymo (1983) and Clarkson *et al.* (2004b) for more details on this method. The full method can be found in Appendix A.

4.2.3 Field pH and conductivity ($\mu S\ cm^{-1}$)

Field pH and conductivity were measured at each of the sample sites. A hole excavated below each peat core was left to fill with groundwater and to reach equilibrium (approximately 10–15 minutes). A pH and conductivity meter then measured the relative levels of each, once readings were stable.

4.2.4 Bulk density

Dry bulk density (DBD) was first measured using standard wetland methods of drying an entire core, working out gravimetric moisture content and then calculating DBD based on volume and dry weight. This involves dividing the oven dry mass of a sample by the volume. Usually two cores are taken, one for DBD, the other for chemical analysis. A trial was undertaken to identify if one core (which could be subsampled for DBD) could be used for both physical and chemical analyses.

The trial involved taking two cores from eight different locations. One of these cores were dried as per standard methods, while the second core was weighed to obtain a ‘fresh’ weight, and then approximately seven subsamples were removed evenly around the core with a serrated knife (similar to cutting a cake slice) (Figure 4.2). Subsamples were bulked together and dried through the normal gravimetric techniques. The moisture factor (MF) which is the wet sample weight (g) divided by the dry sample weight (g) was then be applied to the original ‘fresh’ weight of the entire core (by dividing the fresh weight by the MF to give the dry weight). This gave the oven dry weight of which DBD was then determined from using core volume (577.72 cm^3). One core could be used for all analyses, with very similar bulk density values to that of drying a full core (± 5 % of drying a full core to obtain bulk density).

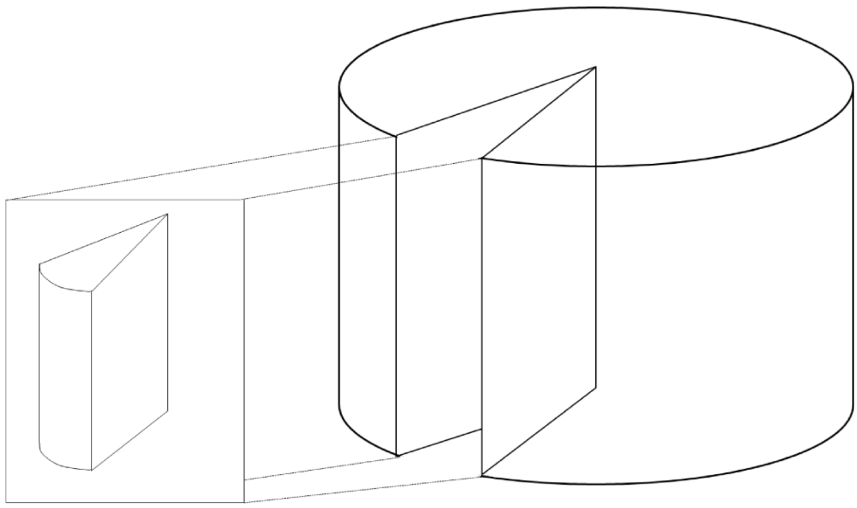


Figure 4.2: Sub-sampling from a peat core for bulk density analysis.

4.2.5 Gravimetric moisture content

Subsamples of the core (method 4.2.4) were weighed fresh, and then dried overnight at 105 °C (as per standard soil methods) until a constant weight was achieved. This dry weight was used to calculate bulk density for the whole core, based on its fresh weight and volume. Volumetric moisture content (VMC) is determined by multiplying the moisture content (% of water in the peat core) by the final dry bulk density (assuming density of water is 1.0 g $cm^{-3-)}$.

4.2.6 Carbon, Nitrogen, $\delta^{15}N$ analysis

The analyses for total carbon, total nitrogen and $\delta^{15}N$ present in the samples all required similar preparation techniques. This involved air drying approximately 50–70 g fresh peat slices (as in method 4.2.4) for 2–3 days. The air dried samples were then ground in an agate and ball on a shaking machine to a fine air dried powder and stored in air-tight containers.

A small portion of these samples were weighed and ground further in a ball mill to achieve a < 1 mm particle size, and analysed by a Truspec LECO CN machine, via combustion (through Waikato Universities' 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 (Waikato Universities stable isotope lab).

4.2.7 Ash content

Sub samples of dried peat were ashed at 460 °C for 6–12 hours to determine the proportion of mineral and organic matter present, for preparation of peat samples (mineral constituents) for ICP-MS digestion (Yafa & Farmer 2006).

4.2.8 Measurement of metals

Heavy metals (and nutrients such as calcium and phosphorus) were determined using inductively coupled plasma mass spectrometry (ICP-MS). This was undertaken following EPA Method 200.2; digestion of sediments for ICP-MS total recoverable elements, version 3 (Environmental Protection Agency 2010a). Some modifications have been made to this method to overcome specific issues observed during preliminary peat digestion trials. Peat samples were ashed prior to digestion to remove organic matter interferences (Yafa & Farmer 2006). This left the mineral constituents and all the elements present in the sample. Also the sample would be easier to digest and less volatile.

Ashed samples were digested using strong acids, nitric (70% HNO_3) and hydrochloric (70% HCl) to break down mineral constituents. These acids, along with heating, essentially convert most solids and metals into an aqueous form, which later once dilute, can be analysed through ICP-MS. Before the digestion was undertaken, all the equipment to be used was acid washed to ensure no cross contamination with samples.

Modifications of the EPA method 200.2 included making up three 100 ml volumetric flasks of nitric acid at 4M concentration (35.7 ml of 70 % [HNO_3]), which is less concentrated than original method. Also, only one 100 ml flask of hydrochloric solution (1:4 concentration) was made. Lower concentrations were used to prevent sample volatility. 10 ml of HNO_3 solution and 3 ml of HCl solution was added to each of the ashed samples and left to stand for 2 days, then were digested on a hotplate up to 90 °C for 3-4 hours (Figure 4.3). The remaining digestion followed the EPA method (Environmental Protection Agency 2010a).

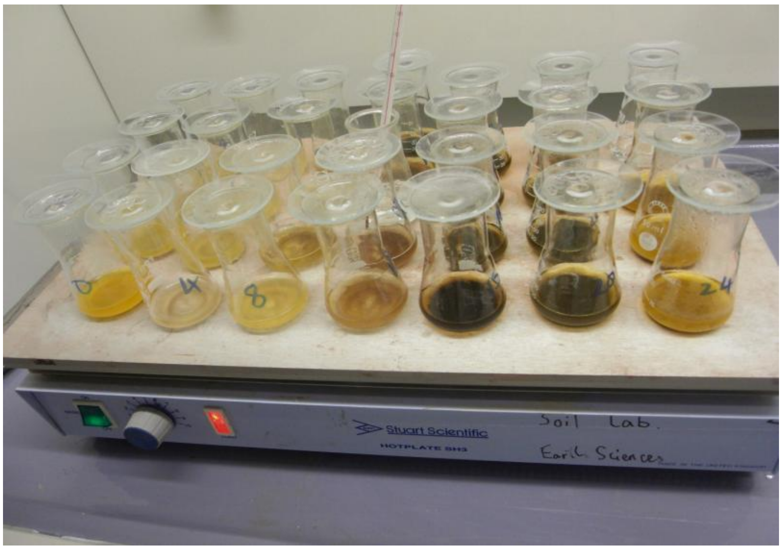


Figure 4.3: Hotplate digestion of ashed peat and soil samples from the Whangamarino transect line.

4.3 Results

Peat and soil results will be presented based on key trends and patterns along the transect line. All analyses were only undertaken once due time constraints, limited samples and resources. Results can also be viewed in Chapter 5, relative to different vegetation groups along the transect line. Appendix D presents all the collected raw data and concentrations.

4.3.1 Peat and soil physical analysis

Peat and soil were assessed for bulk density, volumetric moisture content, von Post decomposition and mineral content. In general, all of the parameters exhibited the same trends, being greatest at the edge of the transect near the farmland, low in the centre of the wetland and increasing significantly towards Whangamarino River.

Higher bulk density generally suggests greater amounts of mineral particles present in the sample, or a higher degree of peat decomposition and consolidation. When a sample has a higher bulk density, it will usually have lower VMC. Bulk density at Whangamarino wetland was lowest in the zone from 50 m to 1100 m (around 0.05–0.10 g cm^{-3}), and increased at the wetland edges towards the farmland and the Whangamarino River (peaking at 0.28 g cm^{-3}). Hence, VMC was slightly lower near the farmland (degraded peat), greater in the zone between 50–1100 m and decreased again towards the river (Figure 4.4).

Sample sites were assessed for relative peat decomposition based on the von Post technique (Figure 4.1b). Low decomposition is from 1–3, very high decomposition is from 8–10. The final four sites (400 m) near the Whangamarino River were composed of a mineral soil mixed in with decomposing vegetation, and von Post was not made at these sites. Minimum decomposition was found at 100 m (level 2), while maximum decomposition of 8 was observed from 1700 m.

Mineral content (% of weight) in samples was highest near the Whangamarino River (70.1 % at 2000 m), with initial increases in mineral content beginning at 1000 m, the start of the manuka belt (patchy from 900 m and dense from 1100 m). Lowest mineral abundance in all sites was at 100 m (2.6%), similar to the lowest peat decomposition (Figure 4.4). Close to the farmland (0–50 m) mineral content also increased significantly.

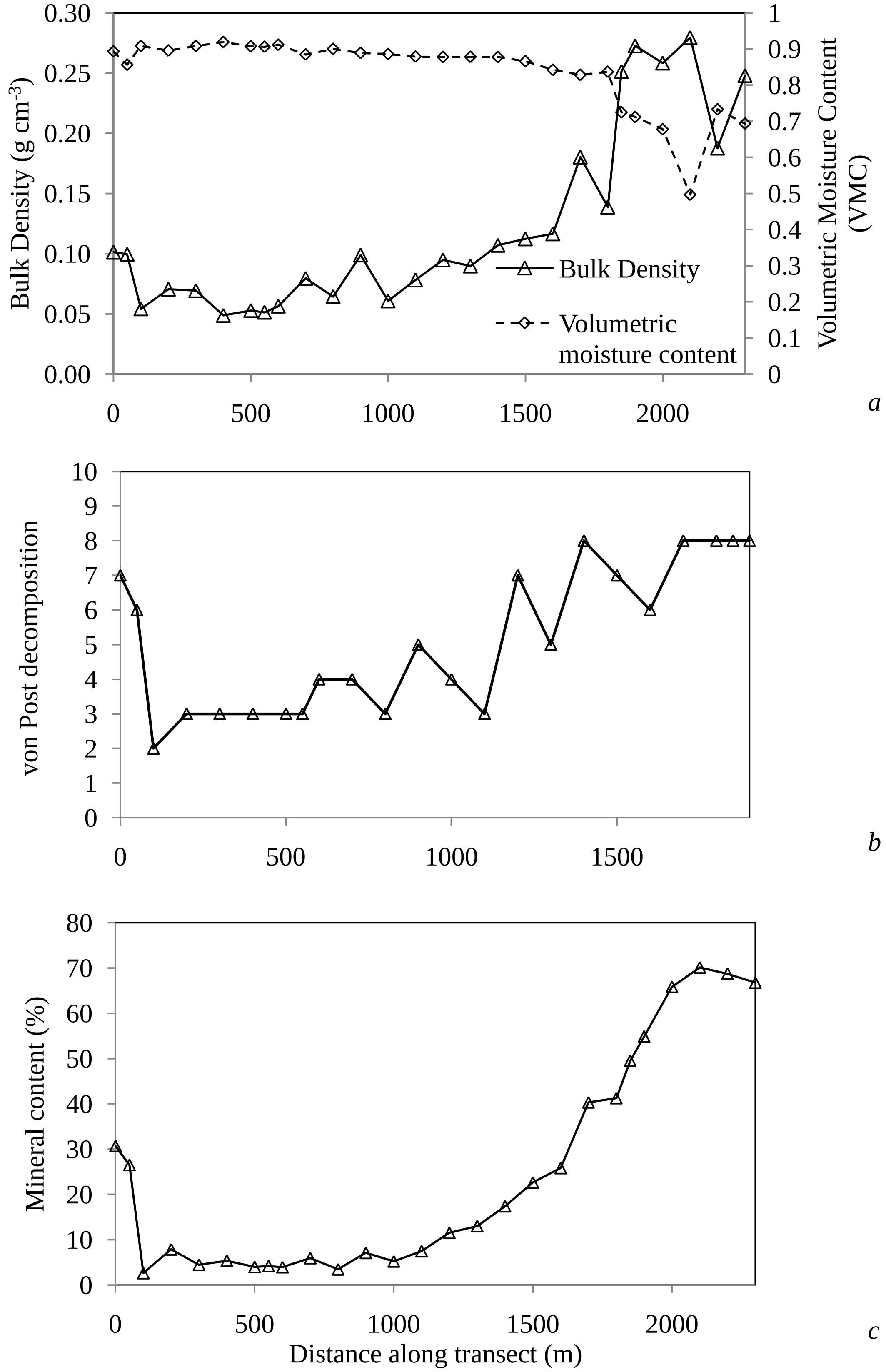


Figure 4.4: (*a*) Bulk density (g cm^{-3}) and VMC, (b) peat decomposition in the top 10 cm and (*c*) mineral content (% of mass) present in samples along the Whangamarino transect line (0–2300 m).

4.3.2 pH and electrical conductivity

Peat water pH and electrical conductivity (EC) increased along the transect line (Figure 4.5). There was little change in electrical conductivity from 50–1800 m, where it was 102.5 µS cm^{-1}. Over the next 200 m, conductivity rose to 165.7 µS cm^{-1}, then dramatically increased to 909 µS cm^{-1}. The final reading and highest value next to the river was 1193 µS cm^{-1} (2300 m), while the lowest measurement was found at 400 m (65.7 µS cm^{-1}).

pH on the transect line from 1100–1800 m increased from 3.6 to 4.8, but did not increase significantly until 1500 m. From 1900–2300 m pH rose from 4.8–6.7 over the space of 400 m. The lowest pH of 3.5 was found at 1000 m.

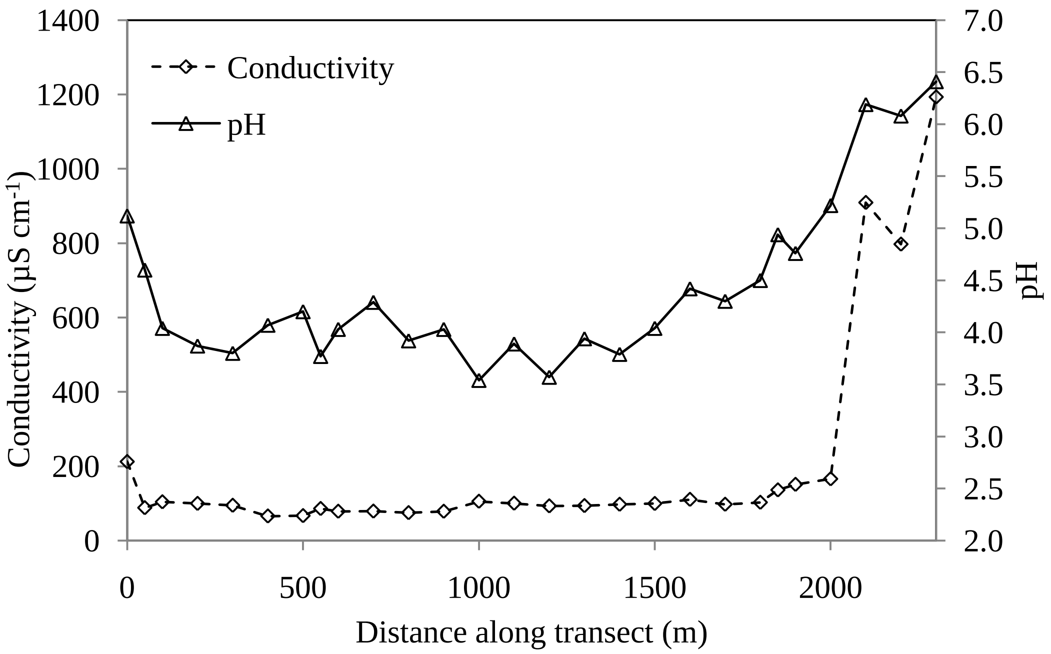


Figure 4.5: Electrical conductivity (µS cm^{-1}) and pH field measurements along the Whangamarino transect line.

4.3.3 Carbon, nitrogen and phosphorus

Gravimetric carbon (% by weight) ranged from a maximum of 42.9 % (550 m) to a minimum of 10.2 % (2100 m). Volumetric carbon (mg cm^{-3}) was lowest at 100 m (18.2 mg cm^{-3}) and highest at 1850 m (52.6 mg cm^{-3}). Total gravimetric carbon (%) was highest in the zone from 50 m – 1100 m and decreased towards the river (Figure 4.6).

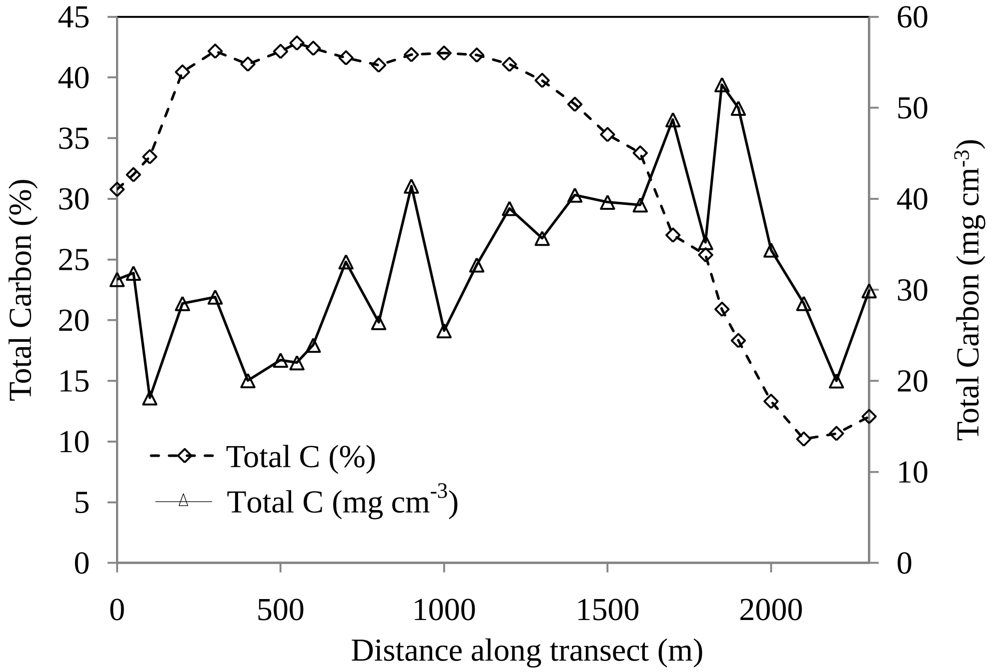


Figure 4.6: Total carbon gravimetric (%) & volumetric (mg cm^{-3}) in samples along the Whangamarino transect line.

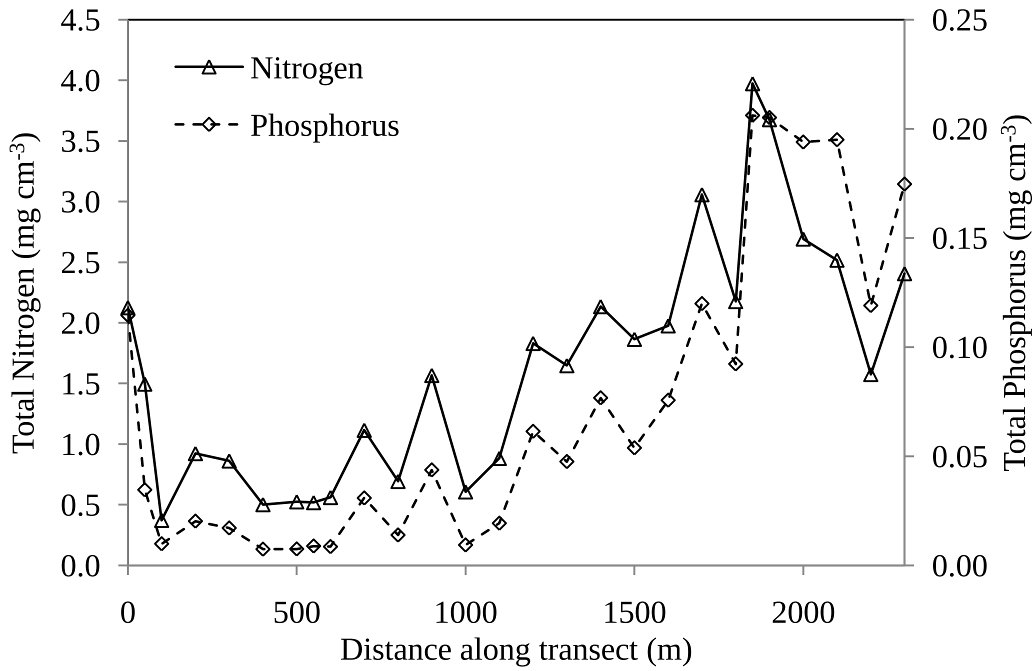


Figure 4.7: Total nitrogen (mg cm^{-3}) and phosphorus (mg cm^{-3}) along the Whangamarino transect line.

Total nitrogen in Whangamarino was 2.1 mg cm^{-3} on the farm/wetland fringe, dropped to 0.37–1.57 mg cm^{-3} from 50–1100 m and climbed from 0.88–1.87 (1100–1500) and 1.87–2.18 mg cm^{-3} (1500–1800 m). Concentrations rose significantly over the next 50 m, peaking at 4.0 mg cm^{-3} at 1850 m (Figure 4.7). From 1900–2300 m, TN ranged from 1.58–3.5 mg cm^{-3}.

TP concentrations (Figure 4.7) was higher near the farmland (0.11 mg cm^{-3}), while the lowest concentrations were in the zone from 50–1100 m (0.01 mg cm^{-3}). The effects of riverine flood inputs became apparent 1000 m from the farmland, with a greater increase in TP concentrations towards the river peaking at 1850 m (0.21 mg cm^{-3}), similar to nitrogen. The biggest increase was observed from 1500–1850 m (0.05–0.21 mg cm^{-3}).

C:N, N:P and C:P ratios are commonly used to assess the abundance of nutrients present in a sample. The lowest C:N ratio was measured near the Whangamarino River, reaching a lower limit of 11.3 (Figure 4.8). The maximum ratio of 48.7 was found at 100 m.

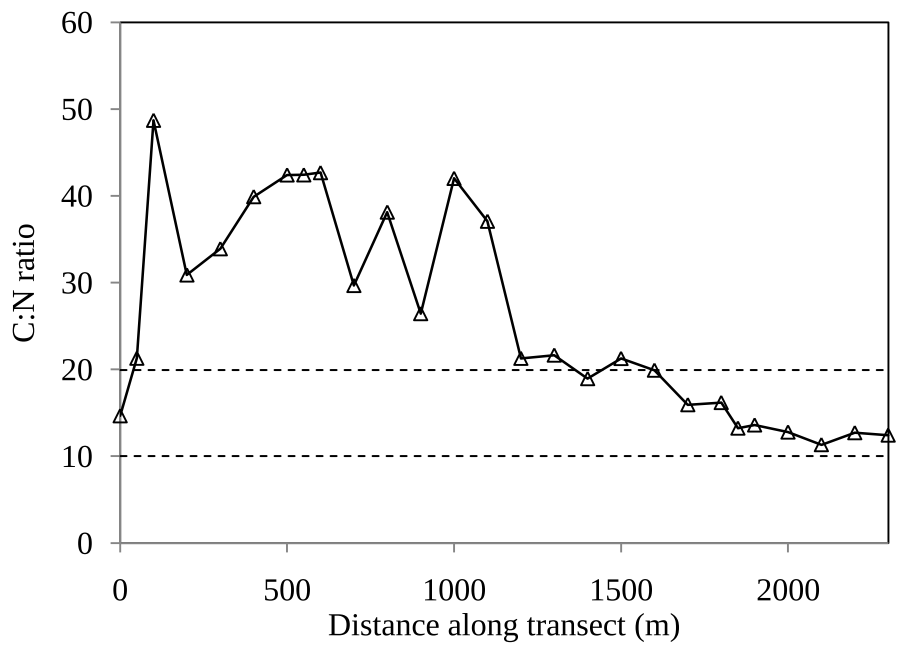


Figure 4.8: C:N ratio from peat and soil samples along the Whangamarino transect line. A ratio >20 indicates immobilisation is dominating, while below this (10–20) mineralisation is likely. A C:N ratio generally has a lower limit of 10 (refer to discussion) (Verhoeven *et al.* 1990; Schipper *et al.* 2004).

Nutrients N and P are important for plant growth and high ratios (N:P and C:P) can indicate a system is limited by one nutrient or the other (Figure 4.9). Highest N:P and C:P ratios were found from 50–1100 m, peaking at 600 m (70.1 and 2970). A substantial decrease was observed from 1100–1200 m. Lowest ratios were found from 1800–2300 m, with the minimum of 12.9 (N:P) and 146 (C:P) at 2100 m. An obvious decrease occurs from 1500 m.

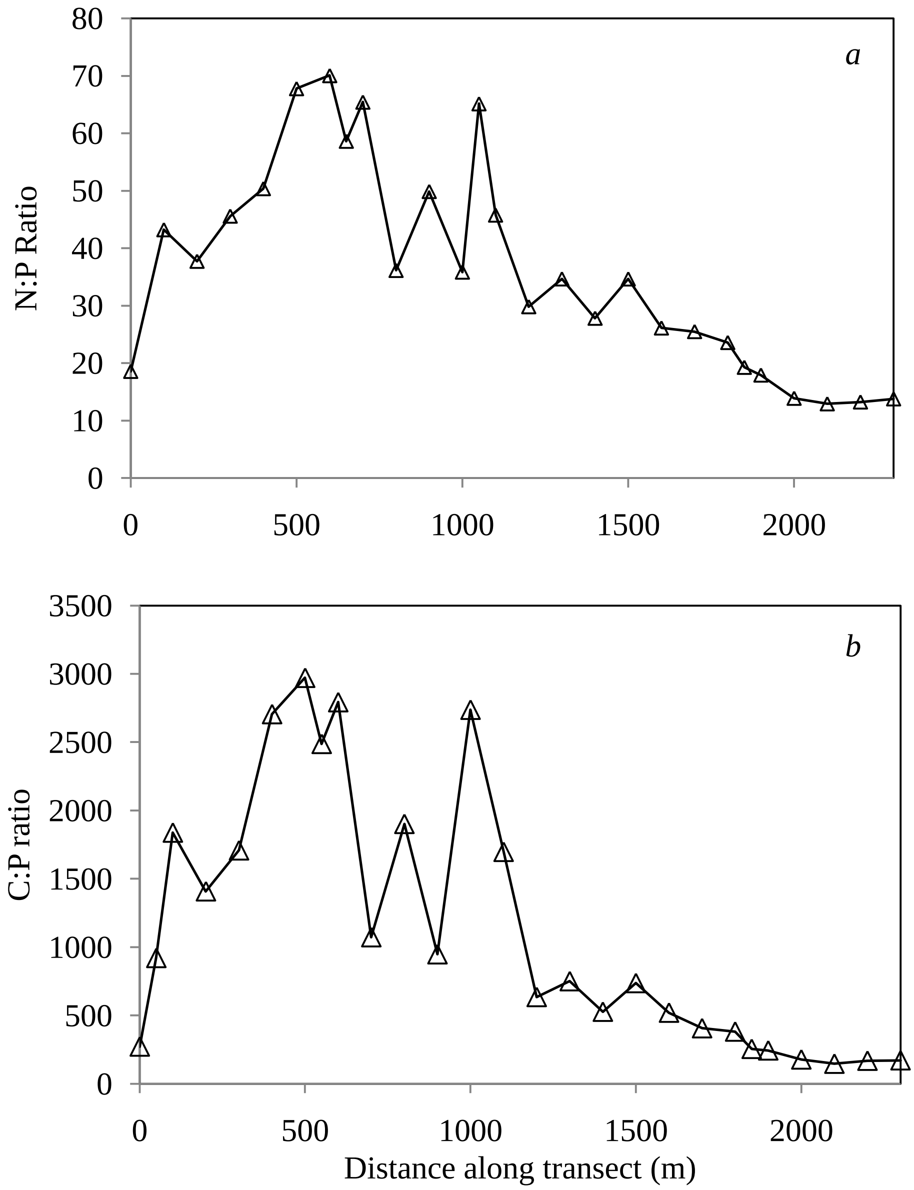
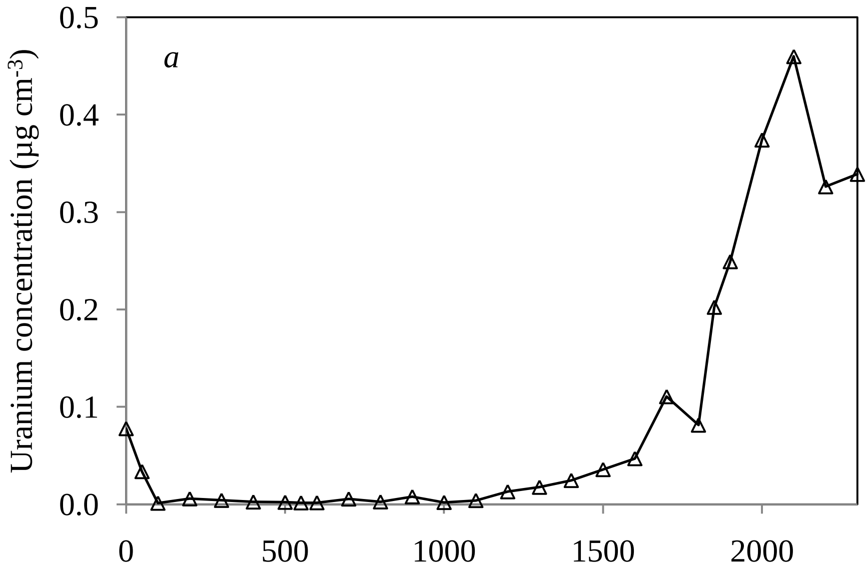


Figure 4.9: (*a*) N:P and (*b*) C:P ratio in peat and soil samples along the Whangamarino transect line.

4.3.4 Cadmium, uranium and δ ^{15}N

Cadmium (Cd), uranium (U) and the stable isotope $\delta^{15}N$ are potential indicators of anthropogenic inputs into a site, predominantly from agricultural practices.

Uranium increased from very low (background) levels (of around 0.004 µg cm^{-3}) at about 1100 m from farmland (Figure 4.10). Concentrations peaked at 0.46 µg cm^{-3} (2100 m), a 100 fold increase. Cadmium did not change from background levels (0.001 µg cm^{-3}) until around 1500 m from the farmland, with a large spike in concentration observed at 1700 m (0.11 µg cm^{-3}) (Figure 4.10). $\delta^{15}N$ increased from 1100 m (-1.22 ‰), with a noticeable change from negative abundance to positive, peaking at 2200 m (5.85 ‰) (Figure 4.11).



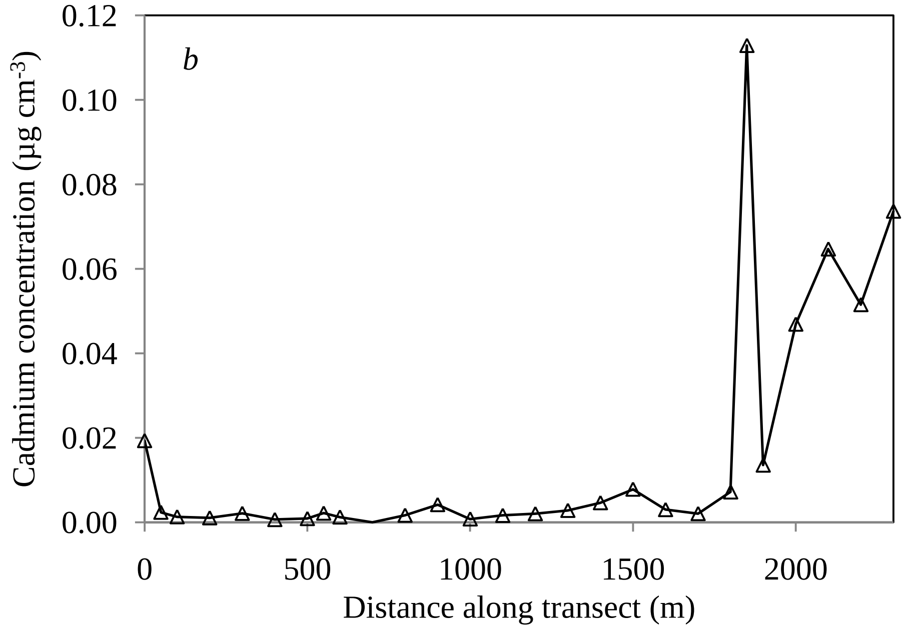


Figure 4.10: (*a*) Uranium and (*b*) cadmium concentrations (µg cm^{-3}) along the Whangamarino transect line.

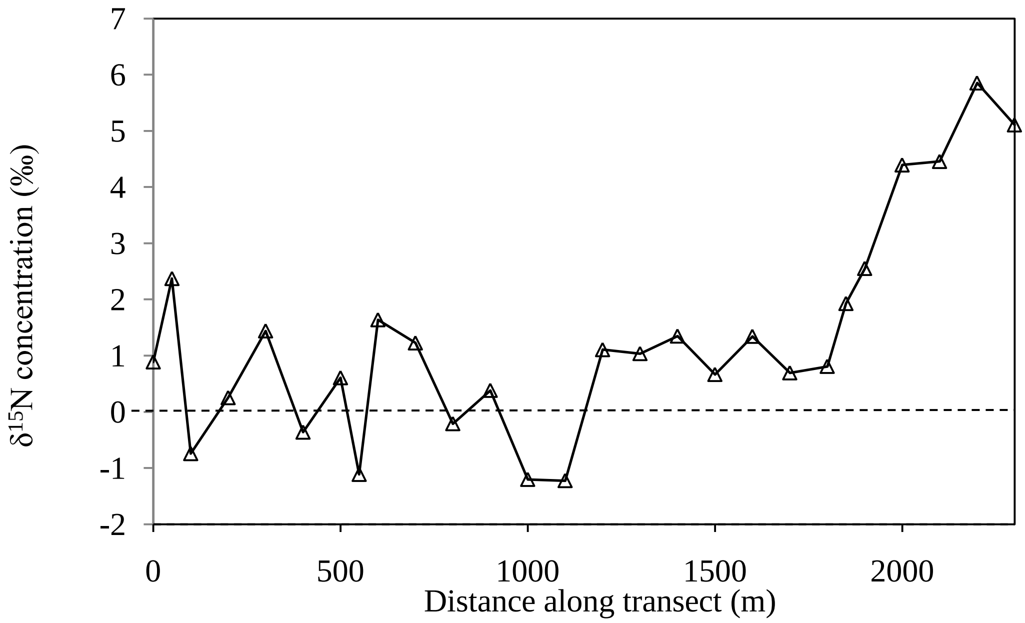


Figure 4.11: $\delta^{15}N$ (‰) abundance along the Whangamarino transect line.

4.3.5 Heavy metals and elements

In general, all elements exhibited the same trend and pattern along the transect. Initially, high concentrations were present near the farmland and then decreased rapidly 50–100 m from the farmland. A steady increase in concentrations was then observed from around 1100 m towards the river. Only common elements and some heavy metals will be presented.

Initial increases in concentrations were observed in Al, Fe, Zn and Cu near the farmland (Figure 4.12). Concentrations were very low in the zone from 50–1500 m. Aluminium and iron began to increase from 1100 m, with the biggest change occurring from 1500 m. Concentrations peaked at 2100 m (15.3 and 6.2 mg cm^{-3} respectively). Zinc and copper also peaked at 2100 m (0.02 and 0.004 mg cm^{-3} respectively). The manuka belt begins at approximately 1100 m (Chapter 5).

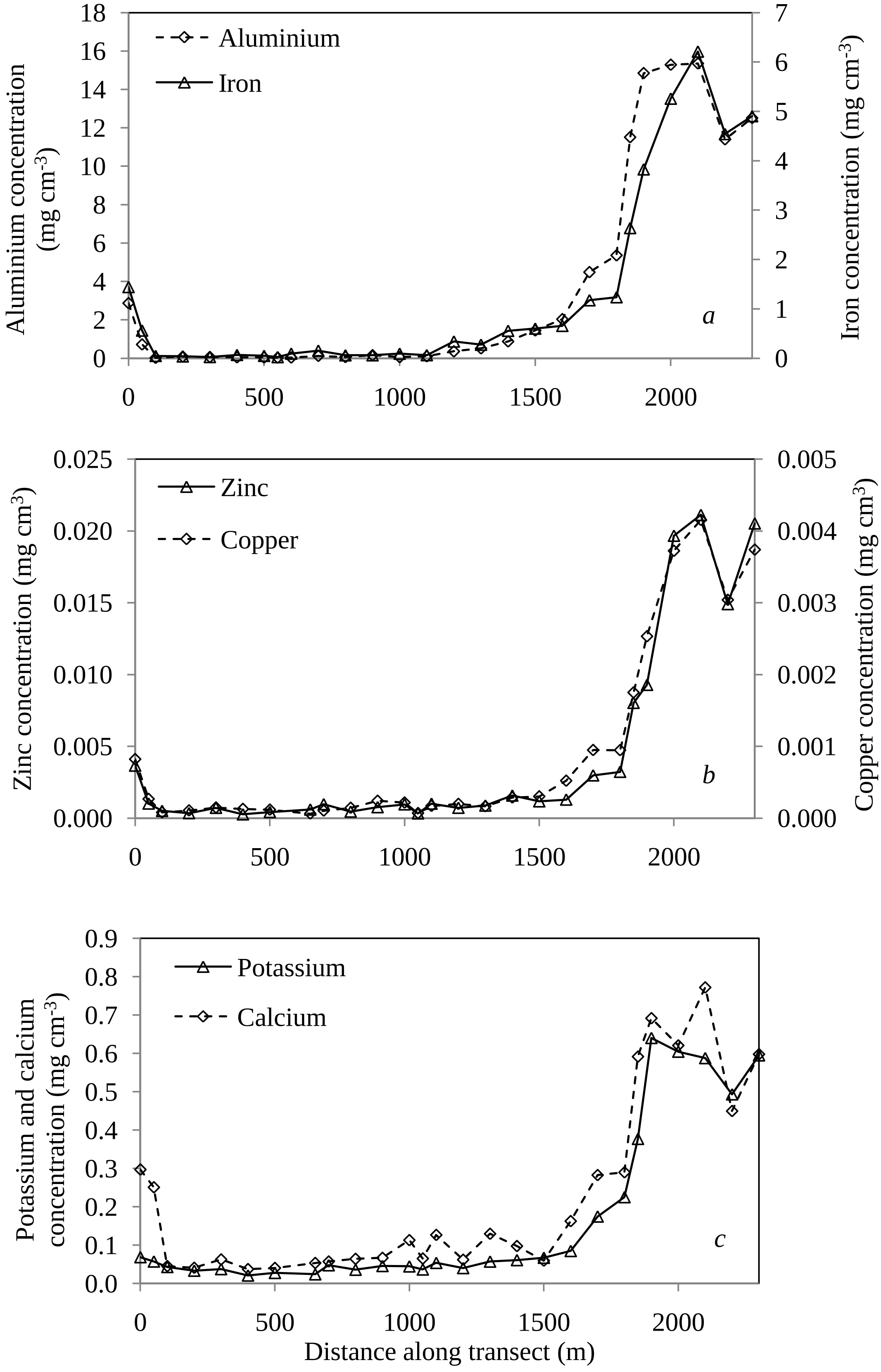


Figure 4.12: (*a*) Aluminium & iron, (*b*) zinc & copper and (*c*) potassium & calcium concentrations (mg cm^{-3}) along the Whangamarino transect (0–2300 m).

Concentrations of calcium were higher on the fringe of the farmland at 0.3 mg cm^{-3} (Figure 4.12). Potassium and calcium both increase significantly from around 1500 m with K peaking at 1900 m (0.63 mg cm^{-3}), and Ca peaking at 2100 m (0.78 mg cm^{-3}).

4.4 Discussion

Physical and chemical analyses of peat and soil along the Whangamarino transect provide an important insight into the current state of the wetland, due to samples being retrieved from the top 10 cm of the substrate (the most recent soil/peat).

Most of the results expressed a simple trend; higher levels near the farmland and then a decrease in the wetland to low levels, which was consistent from 50 m to around 1100 m from the farm edge. From 1100 m onwards, initial increases in degradation and peat chemistry were observed, becoming greater closer to the Whangamarino River. At around 1850–2000 m, levels of degradation and peat/soil chemistry peaked, then began to drop slightly over the last 300 m to the river. Results are from unreplicated samples, so are illustrative only.

4.4.1 Physical and field measurements

Bulk density, von Post decomposition, volumetric moisture content (VMC), pH and electrical conductivity provide an indication of wetland condition at a point in time, often without the need for additional chemical measurements.

Bulk density and VMC can be used as indicators of peat degradation and increased sedimentation in a wetland. Bogs and fens have low dry bulk density in the top 10 cm or acrotelm layer, often less than 0.09 g cm^{-3} (Verry & Boelter 1978). At Whangamarino, bulk density was generally lower than 0.10 g cm^{-3} for approximately 1500 m, from farmland into the wetland. There was a noticeable decrease in DBD from 0.10 g cm^{-3} next to the farmland to 0.05–0.08 g cm^{-3} approximately 50–1100 m into the wetland (Figure 4.4). A high VMC (Figure 4.4) of around 90% water per volume from 50–1100 m indicated a high water table common in bog conditions (Chapter 6), which is fundamental for peat development (Charman 2002). The region at Whangamarino with a low bulk density also had a low von Post decomposition index (Figure 4.4). Reduced decomposition was likely due to a high VMC and stable water table (Chapter 6), resulting in an accumulation of relatively undecomposed plant material, numerous pore spaces and hence a lower bulk density. The inability to undertake a von Post assessment from 1900–2300 m suggests a mineral substrate, supported with higher mineral contents (Figure 4.4). This is consistent with conditions present in a marshland (Johnson & Gerbeaux 2004).

DBD and von Post were consistent with other measurements of restiad bogs undertaken in New Zealand. Clarkson *et al.* (2004a) measured bulk density levels in lowland restiad bogs in the North Island between 0.03 g cm^3 – 0.15 g cm^3, depending on the dominant plant species present. In most cases, *E. minus* restiad bogs had the lowest bulk density (0.03–0.12 g cm^{-3}), while restiad bogs dominated with *Baumea spp* and *G. dicarpa* had a higher bulk density (0.6–0.15 g cm^{-3}).

Bulk density increased from 0.08 g cm^{-3} (1100 m) along the transect line, with the biggest change occurring from 1500–1850 m, where bulk density doubled from 0.11 g cm^{-3} to 0.25 g cm^{-3}, and peaked at 0.28 g cm^{-3} at 2100 m. Additionally, VMC decreased from 90% to 50% at 2100 m. The increase in bulk density and VMC was likely due to a dynamic water table (greater decomposition), coupled with flood events that reached up to 1400 m from the farmland, with the severity of the inundation distance and frequency increasing closer to the river (Chapter 6) (Charman 2002; Sorrell *et al.* 2007).

Higher sediment inputs (Figure 4.4) brought in from the river through flooding likely contributed to increasing bulk density from 1100–2300 m. Erosion from nearby farmland was probably the cause for the spike in mineral content on the wetland fringe (0–50 m). Sedimentation contributes substantial amounts of inorganic nutrients such as N and P to wetlands. Most of the sediment comes from the upper Whangamarino catchment of steep hills in the headwaters and alluvial plains before entering the wetland. This contributed over 80% of sediment inputs when the Pungarehu Canal flood gate closed. Highest proportion of sediment contributions from Lake Waikare (14%) were found near the river and canal confluence (Gibbs 2009). Changes of DBD, VMC and mineral content (Figure 4.4) along the transect line correspond with differing vegetation compositions which likely develop from an increasing nutrient concentration and hydrological variability towards the river (Chapter 5 and 6).

Field measurements of pH and electrical conductivity (EC) (Figure 4.5) support the proposition that peat degradation in the Whangamarino wetland was associated with the hydrological regime. Wetland class is an easy way to examine a site, where restiad bogs are often nutrient poor, older and have low pH and EC (Sorrell & Gerbeaux 2004). Swamps on the other hand, are often young wetlands with surface water inputs carrying nutrients and sediment which can lead to higher pH and EC measurements (Campbell & Jackson 2004). Clarkson *et al.* (2004b) used field pH as a guide to help classify a New Zealand wetland, along with other variables. Restiad bogs averaged a pH of 4.0 (3.7–4.4), while swamps averaged 5.2 (4.1–5.9). A significant proportion of the Whangamarino transect line fell within the criteria for a restiad bog, from 100 m to 1500 m the pH ranged from 3.5 to 4.3. Swampland became more abundant from 1500 m to 2300 m (based on pH, TC, TN, TP concentrations). EC increased significantly from 2000 m which corresponds with high mineral content (Figure 4.4) and a change in vegetation in this area (Chapter 5). High EC is consistent with the principal that high amounts of sediment and nutrients (coming in through flooding) increase the abundance of ions which lead to a high electrical conductivity (and generally a decrease in water quality) (Davies-Colley & Wilcock 2004).

4.4.2 Carbon, nitrogen and phosphorus

Total carbon (TC), nitrogen (TN) and phosphorus (TP) are three significant building blocks for life in wetlands and other ecosystems. Particularly, plant growth rates are primarily influenced by the abundance of N and P.

4.4.2.1 Carbon

Volumetric TC levels at Whangamarino (Figure 4.6) fall within the ranges outlined by Clarkson *et al.* (2004b) and reproduced in Table 2.3. Clarkson *et al.* (2004b) found bogs ranged from 24.1 to 239. 8 mg cm^{-3}, while swamps ranged from 5.2 to 100.6 mg cm^{-3}. The issue with this classification is the broad overlap in values between bogs and swamps. The values obtained in a bog could be entirely consistent with that obtained in a swamp, due to the conversion of gravimetric carbon (% per gram) to volumetric. This conversion incorporates bulk density, so while peat in a restiad bog has a high gravimetric abundance of carbon, it has a low density due to a lack of mineral component (sediment) or reduced consolidation from low decomposition. Therefore the mass of carbon per volume of peat is low. Hence using TC on its own would not be useful to classify zones along the Whangamarino transect line.

The ‘bulk density effect’ on volumetric carbon mass is supported by Bridgham *et al.* (1998) who studied 16 different wetlands in Minnesota. They looked at mineralisation rates in wetlands focusing on nutrients. They found bulk density alone explained 75% to 83% of the variation of carbon mineralisation. This was due to the fundamental distinction between nutrient turnover (mineralisation or immobilisation) and availability. To find mineralisation rate per volume of soil requires multiplication of the turnover rate with gravimetric nutrient content and bulk density. Ombrotrophic bogs and acidic fens (which possess a high turnover rate in the acrotelm, but low nutrient concentrations) ultimately have a low mineralisation rate due to the low bulk densities present in this horizon (see section 4.4.3.2). Hence there is lower volumetric (mg cm^{-3}) mass of nutrients for plant growth (Verhoeven *et al.* 1990).

4.4.2.2 Nitrogen and $\delta^{15}N$

Nitrogen is one of the most important nutrients in a wetland system that provide the required constituents for plant growth and in high concentrations can lead to eutrophication. The important trends in nitrogen and phosphorus are the obvious effect farmland and riverine inputs appear to be having along the transect line (Figure 4.7). Nutrient changes began at about 900 m (from the farmland) with an obvious increase in concentrations, while there were some apparent inputs from farmland over the first 50 m of wetland. The TN concentration of 0.37–1.57 mg cm^{-3} from 50–1100 in Whangamarino falls within the New Zealand bog average of 0.82 mg cm^{-3} (0.02-1.83) (Clarkson *et al.* 2004b). Nitrogen levels increased along the transect from 1100 (0.88–3.6 mg cm^{-3}), similar to concentrations in New Zealand swamps (TN = 2.12 mg cm^{-3} (1.15-3.24)) (Clarkson *et al.* 2004b). The first measurement (1100 m) on the fringe of restiad bog and manuka (0.88 mg cm^{-}

^{3}) is lower than the minimum for New Zealand ranges (1.15 mg cm^{-3}), the next sample site at 1200 m had a concentration of 1.85 mg cm^{-3} (falling within the desired New Zealand range). Changes in TN concentrations were consistent with bulk density along the transect, where the peak in concentration coincides with the peak in bulk density (1850–2000 m) (Figure 4.4).

Most of the nitrogen in a system is organic and unavailable to plants and requires mineralisation by microbes to create labile forms of N (Chapter 2). While TN concentration increased towards the river, potentially in areas with low mineralisation rates nitrogen availability to plants could still be low.

C:N ratios are often used to express the availability of nitrogen in a soil or peat system, where ratios above 20:1 would indicate immobilisation is dominating, and below this mineralisation is likely occurring (Verhoeven *et al.* 1990). As N accumulates (in mineral swamps such as at Whangamarino), the C:N ratio declines. This was likely due to N concentrations increasing (from river inputs) and increasing relative to carbon concentrations. Generally minimum C:N ratio is about 10:1, which exists because of the dominance of organic N forms in soil and because simple organic compounds are rapidly hydrolysed by soil enzymes or mineralized by soil microbes to inorganic forms (Schipper *et al.* 2004). C:N ratios (Figure 4.8) along the Whangamarino transect line suggests immobilisation dominated in the zone from 50–1100 m (restiad bog) and a proportion of the manuka zone (to around 1500 m from the farmland). From 1600 m, C:N was less than 20 and suggested mineralisation could be occurring which is consistent with the development of a mineralised swampland (and marshland from 1900–2300 m) with increasing nutrient concentrations and abundance of invasive plant species that grow in nutrient rich environments (such as *P. persicaria*; Chapter 5).

The C:N concentrations at Whangamarino were consistent with those found across New Zealand by Clarkson *et al.* (2004b) (see Table 2.3). The zone from 50–1100 m had a C:N ratio that averaged 38 (range of 27–49), while New Zealand restiad bog averaged 48.5 (ranging from 35.9–79.7). The Whangamarino C:N ratio for the restiad bog area (50–1100 m) are low compared to New Zealand bogs, and are likely due to this wetland and bog being relatively young and not having the same time period to accumulate deep peat layers and expansive bog areas. Clarkson *et al.* (2004a) identified the Whangamarino restiad bog as forming 1850 years BP, while other bogs in the Waikato, such as Moanatuatua and Kopuatai, were aged at 11,500 – 13,000 years BP. C:N ratios for the swampland area at Whangamarino (1200–1900 m) were around 14, (ranging from 12.8–21.2), while common C:N levels in swamps were 18 (ranging from 14-30.6) (Clarkson *et al.* 2004b). This is due to a large amount of nitrogen input from the Whangamarino River and mineral swampland as seen in the very low ratios present.

Using the stable isotope $\delta^{15}N$, external N inputs can also be identified (Chapter 2). In animal waste $\delta^{15}N$ is generally enriched between 10 and 20‰, and can end up directly in water ways. Fertilizer $\delta^{15}N$ ranges from 0.5 to 5‰ for oxidized N (NO_3^-); with lower values for the reduced form (i.e. NH_4^+) or urea (Bedard-Haughn *et al.* 2003; Xue *et al.* 2009). These enriched forms of nitrogen could eventually enter the Whangamarino wetland through flooding, as seen in the increasing $\delta^{15}N$ (‰) from 1100–2300 m (Figure 4.11) or through aerial deposition. This zone is the beginning of the manuka belt, and an area identified as receiving nutrient inputs during larger flood events (Chapter 6). While the overall increase in $\delta^{15}N$ was clear, and agricultural was an important contribution, other factors could contribute to $\delta^{15}N$ changes. Due to the variation in $\delta^{15}N$ concentrations through biological processes (such as nitrification and denitrification), and anthropogenic inputs from sewage (secondary treated water from Te Kauwhata enters Lake Waikare) and fertilisers, identifying the direct cause for the $\delta^{15}N$ increase can be difficult (Bedard-Haughn *et al.* 2003; Environment Waikato 2007). Enrichment occurs through fertiliser, manure and effluent application to soils, which is likely the cause for higher $\delta^{15}N$ levels next to the Whangamarino River.

Examining $\delta^{15}N$ in plant biomass can be a more useful approach to provide indications of nutrient limitation at a site, due to the strong correlation with phosphorus availability in both soil and plant biomass (Clarkson *et al.* 2005). This is covered in more detail in Chapter 5.

4.4.2.3 Phosphorus, cadmium and uranium

Phosphorus is required by plants for growth but only in small quantities, when compared to nitrogen (Sorrell & Gerbeaux 2004). Total phosphorus was 0.11 mg cm^{-3} on the farm/wetland fringe, dropped to 0.01 mg cm^{-3} (10 fold decrease) in the restiad bog, and rose to 0.21 mg cm^{-3} near the river (2000 m) (Figure 4.7). The concentrations from 50–1100 m (restiad bog) were around 0.01 mg cm^{-3}, on the lower end of New Zealand bog TP concentrations (averaging 0.08 mg cm^{-3} , ranging from 0.01–0.2 mg cm^{-3}) (Clarkson *et al.* 2004b). From 1500 m similar TP concentrations (of 0.05–0.21 mg cm^{-3}) were found when compared to New Zealand results (0.15–0.59 mg cm^{-3}), although on the low side (peaking at 0.21 mg cm^{-3}). Potentially this could be due to lower phosphate fertiliser inputs in the catchment (primarily sheep and beef), which would likely be higher if dairy farming was present. Phosphorus remained high from 1800–2300 m (0.12–0.21 mg cm^{-3}) indicating high nutrient inputs from the river and swamp conditions.

N:P and C:P ratios are often used in wetlands to determine if there is a limiting nutrient to plants. Phosphorus and nitrogen are often limited in bogs due to a lack of surface or groundwater inputs. Low N:P ratios occur when P is no longer coming in from surface waters, but small amounts of N come in from the atmosphere. Plants need P for growth, so generally are limited by phosphorus more so than nitrogen in most wetlands and particularly in bogs (Figure 4.9) (Sorrell & Gerbeaux 2004). Additionally, as surface water inputs increase (from the Whangamarino River) phosphorus usually becomes abundant and more accessible for plant growth, as it is transported in association with suspended sediment. In these situations, nitrogen often becomes the limiting growth nutrient (Sorrell & Gerbeaux 2004).

P mineralisation is faster in bogs (due to the same processes occurring with nitrogen), and often the labile pool is greater (up to 30% of total P) (Bridgham *et al.* 1998). Yet due to an overall low volumetric concentration of P and N, these nutrients still limit plant growth (Verhoeven *et al.* 1990; Bridgham *et al.* 1998). N:P and C:P ratios from 50–1100 m (bog) in Whangamarino wetland averaged around 50 and 1600 respectively, and ranged from 37–70 (N:P) and 947–2970 (C:P). These results are consistent with the values obtained by Clarkson *et al.*

(2004b) for New Zealand bogs, averaging 39 for N:P ratios (ranging from 20.6–81.6) and for C:P, averaging 1904 (ranging from 533-4221). The higher N:P ratio at Whangamarino suggested the bog could be limited by phosphorus.

A noticeable decrease in the N:P ratio occurred at 1100 m (start of manuka zone) (45.9) and at 1500 m (34.5) (start of swampland). The lower limit reached 12–13.5 from 1900–2300 m (marshland). While these values appeared to be very low, they were still higher than the average for New Zealand swamps, which is 9.1 and ranged from 4.0–20.6. Similar findings were observed for C:P ratios. This may indicate that while phosphorus concentrations were much greater than that found in the bog, the wetland may still be potentially P limited. As discussed in Chapter 2, the high sediment deposition and increasing levels of Al, Fe, K and Ca closer to the river could also potentially be sorbing P from solution, making it unavailable to plants.

Looking at N:P ratios in plant biomass is now accepted as a better approach in determining nutrient limitation in a wetland (Güsewell & Koerselman 2002; Tessier & Raynal 2003). This has been undertaken in Chapter 5, where a more detailed discussion can be found.

Phosphate rock commonly used in fertilisers is known to have relatively high concentrations of heavy metals such as arsenic, cadmium, chromium, lead, mercury, and uranium. Cd and U can be use as an indicator of P derived fertiliser (Roberts & Longhurst 2002). The increases observed in Cd concentrations near the farmland and from 1800 m (mineralised swampland) at Whangamarino wetland suggest there is some artificial input of phosphorus coming directly from fertiliser (Figure 4.10). This input is obviously much larger next to the river, which is likely due to the primary catchments consisting of pastoral farming (Gibbs 2009).

Uranium also can be used as an indicator of fertiliser inputs, as like cadmium, the technologies to remove it from superphosphate fertiliser are yet to be implemented worldwide (Roy 2001). A study in Rothamsted, UK and also a smaller study in New Zealand by Rothbaum *et al.* (1979) examined U accumulation with different application rates of superphosphate. They found most of the U was retained and concentrated in the surface layers of soils, which can subsequently be eroded. As uranium is naturally occurring, background concentrations would be expected to be found in the Whangamarino wetland. But due to large increases in U concentration when phosphate fertilisers are refined, any erosion or sediment coming in from the farmland (through river flooding) would bring in uranium that is strongly bound to organic matter and clay minerals (Takeda *et al.* 2006). An increase in U concentration was observed from 1100–1900 m, with the largest increase occurring from 1500–2100 m. This zone was likely swampland and received most of the mineral sediment input from the Whangamarino River (Figure 4.10). The small increase starting from 1100–1500 m is likely due to the larger flood events reaching further into the wetland and carrying finer sediments (Chapter 6).

4.4.3 Aluminium, iron, calcium and potassium

In the Whangamarino wetland sediment coming in from the river due to flood deposition (Figure 4.4) brought in significant amounts of fine particles which contained high levels of Al, Fe, Ca and K, with concentrations increasing from 1500 m (Figure 4.12) (Craft 1996; New Zealand Institute of Chemistry 2010). Al and Fe oxides brought in with sediment and deposited in the mineralised swampland (from 1100–2300 m) could potentially be sorping with P in solution, reducing the availability of phosphorus for plant growth (Chapter 2). This may be the reason for N:P ratios indicating the wetland could be phosphorus limited, even though higher abundances of P are present near the river.

The higher concentrations of Ca (Figure 4.12) on the wetland fringe and closer to the river were likely attributed to fertiliser use (lime) in surrounding farmland (Chapter 2) eventually being deposited in the wetland (Roy 2001; Fertiliser New Zealand Ltd 2010).

4.5 Summary

- Measurements of pH, conductivity (μS cm^{-1}), volumetric moisture content, bulk density and mineral content showed higher levels adjacent to the farmland, low levels between 50–1100 m and the greatest increase close to the river (from 1100 m to 2300 m).

- Volumetric carbon concentrations were highest in the zone from 1100–1800 m (manuka belt) and closer to the river (1900–2300 m), while gravimetric carbon was greatest in the zone from 50–1100 m (restiad bog). Values in the Whangamarino restiad bog and mineralised swampland fell within New Zealand ranges as described by Clarkson *et al.* (2004b).

- Volumetric nitrogen concentration was high next to the farmland, low from 50–1100 m (restiad bog) and increased significantly from 1100 m to the river (due to flood inputs). N concentrations (mg cm^{-3}) and C:N ratios for the restiad bog and mineralised swampland fell within the ranges for each wetland class relating to New Zealand (Clarkson *et al.* 2004b). C:N ratios indicated net immobilisation (and a loss of available N to plants) could be occurring from 50 m to 1300 m, while net mineralisation (and increased N availability) may be occurring from 1300 m to 2300 m (next to the Whangamarino River marshland and swampland).

- Volumetric phosphorus concentrations exhibited similar trends to nitrogen, with concentrations increasing from 1100 m (manuka belt) towards the river. Flood inputs likely contribute large amounts of P, bound in suspended sediment. P concentrations (mg cm^{-3}), N:P and C:P ratios lay within the ranges for a restiad bog (50–1100 m) and mineralised swampland (1500–2300 m) in New Zealand (Clarkson *et al.* 2004b).

- Concentrations of Al, Fe, Ca and K increased towards the river (from 1500 m) through deposited flood sediment. N:P ratios present in the soil suggested the wetland may be limited by phosphorus.

- Cadmium, uranium and the stable isotope $\delta^{15}N$ increase in abundance towards the Whangamarino River (from around 1500 m). Cd and U concentrations indicate fertiliser inputs (superphosphate) from pastoral farming in upper catchment were potentially having an effect on plants in Whangamarino wetland. $\delta^{15}N$ (‰) increased from 1100 m to the river (2300 m). The increase is attributed to agriculture in the upper catchment, likely from ruminant manure, human sewage and nitrate fertiliser applications.

Chapter 5: Vegetation composition and patterns

5.1 Introduction

Vegetation composition and chemistry was assessed along the transect line to determine the influences from the dynamic hydrological regime. Vegetation plots were established and assessed at 100 m intervals and in subjective plots near water level sites. There were a total of 27 plots along the 2.3 km transect line. At each site, peat cores (Chapter 4) and foliage samples were also retrieved for chemical analysis.

Plot structure and composition were assessed as per the Atkinson method while plot health was determined through die back and the presence of invasive species (Clarkson *et al.* 2004b). The Atkinson method assigns structure of each plot based on the dominant species first, and all other species with a cover abundance of 20% or greater as secondary (i.e. *L. scoparium/G. dicarpa*). When species were the same height they were separated with a hyphen (-), when species differed in heights they were separated with a slash (/) (Atkinson 1985).

These plots were later assessed through cluster analysis and ordination, with reference to the chemical composition of the peat samples as environmental variables that may be influencing the species of plants occurring along the transect.

5.2 Methods

A variety of methods were used throughout the vegetation analysis. These methods are summarised as follows.

5.2.1 Plot selection and establishment

Vegetation plots were established every 100 m along the transect line to ensure changes in species composition were not missed. When the 100 m interval fell close to a water level site (within 20 m) only one plot was set up (no additional plot for the nearby site). Plot establishment and vegetation assessment was undertaken using the technique developed in the Handbook for Monitoring Wetland Condition (Clarkson *et al.* 2004b).

4 x 4 m (16 m^2) quadrats were established at random distances (5–25 m) and direction (north or south) from the transect line was chosen using random number generation from a handheld calculator (Figure 5.1).

Figure 5.1: 4 x 4 m quadrat established 400 m from farmland. Vegetation species in this photo include *E. minus*, *G. dicarpa*, *Baumea spp.* and *L. scoparium.*

The percentage cover of species in the plot were assessed for the canopy and ground layers (only layers present in the wetland as no species heights were greater than 6 m), maximum heights were also recorded (Figure 5.2). The same observer was used to assess the cover rating for each species in the quadrat as it was a subjective procedure. All aspect of the wetland plot sheet were filled in including, pH, water table depth and GPS co-ordinates, of which can be found in Appendix D along with all raw chemistry results.

Figure 5.2: Maximum height (m) of *E. minus* measured in a plot.

5.2.2 Foliage sampling and chemical analysis

Foliage samples were collected from manuka (*L. scoparium*) within or nearby quadrats for the analysis of total N, total P and $\delta^{15}N$. Manuka was chosen as it was the most common plant species along the transect line. When manuka was unavailable, the next most common species was sampled. In order to be able to compare the chemistry results of different species, where one species was diminishing, a second sample was taken of the next most common species where they overlapped. For example, where manuka was diminishing, *C. tenuicaulis* (swamp coprosma) was sampled, as this species was the next dominant group. In total, three species were sampled; *L. scoparium*, *C. tenuicaulis* and *P. persicaria* (willow weed). When results were obtained, the difference between the various species and manuka was used to calculate the sites predicted foliage nutrient concentration relative to manuka (Clarkson *et al.* 2004b).

Samples were collected by stripping the leaves from the most elevated, young growth shoots of the various species. In some cases, manuka branches had to be broken down in order to retrieve the sample, as they were over 5 m high. Samples were stored in paper envelopes and later dried in an oven at 62 °C for 24 hours. These samples were then crushed using the same technique as used in the peat preparation. TN and $\delta^{15}N$ were analysed through Waikato University's stable isotope lab, while total P was determined by Hills Laboratories (Hamilton) where a microwave digestion was undertaken.

5.2.3 Cluster analysis and ordination

Using the wetland plot sheet field cover scores for vegetation types, statistical and computer analyses were undertaken using the program PATN, version 3.03 2004. This program analyses the abundance (cover %) of the various species present across the vegetation plots, and in doing so groups sites together based on commonality. Data analyses followed the method outlined in Clarkson *et al.* (2004a) and Sorrell *et al.* (2007).

Within PATN, statistical programmes were used to run a two dimensional (2d) ordination 1000 times for the 27 sites with varying species composition data. The ordination identified the best group clusters with the lowest stress value (0.1555), of which a tree dendrogram was then derived, displaying the relationship between the sites. Six main species groups were selected as the cutoff (all 27 plots fitted into these groups based on commonality). The cutoff value is selected by the ordination, based on the first instance 6 separate groups can be identified.

A Principal Component Correlation (a type of ordination) was also run. PCC uses multiple linear regression to 'add' data variables into the 'ordination space', such as extrinsic environmental variables (i.e. nutrient concentrations in mg cm^{-3} at each site) (Belbin 1995). The environmental variables were added over the ordination and an MCAO test (ordination equivalent to an ANOSIM) was undertaken to identify if these variables were significant to the group clusters (95% confidence) and a relationship was present. All variables that were not significant were removed (such as foliage $\delta^{15}N$) (Belbin 1995).

Once the cluster analysis and PCC ordination were completed, group statistics were run on PATN. A variety of statistics were derived for the various environmental variables over each of the six groups determined through the cluster analysis. These statistics included maximum, minimum, median, upper and lower quartiles. The group statistics were further analysed in the software program

STATISTICA, where box plots were created for each of the groups and various external variables.

5.3 Results

The following sections describe the results derived from the vegetation plots in Whangamarino wetland. A full vegetation species list (identified in the plots) can be found in the Appendix B**.**

5.3.1 Foliage chemistry

Foliage sampling started 50 m from the farmland (first area of *L. scoparium*) and continued to 2300 m (Whangamarino River).

Total nitrogen in foliage samples appeared to be relatively stable throughout the wetland. The major increase in predicted *L. scoparium* TN to double the previous levels began at around 1800 m (1.0 %) from the farmland, where it peaked at 2200 m (2.6%) (Figure 5.3). This zone was dominated by *P. persicaria* closer to the river and *C. tenuicaulis* from 1800 m to 1950 m.

The stable isotope $\delta^{15}N$ in the foliage samples showed less obvious trends. Initially there appeared to be higher $\delta^{15}N$ (‰) closer to the farmland, dropping dramatically in abundance as sampling moved into the zone from 50–1100 m. At approximately 1100 m, the abundance of $\delta^{15}N$ (‰) increased from -4.94‰ to 0.52‰ at 1200 m, which remained relatively constant all the way to the Whangamarino River (Figure 5.4).

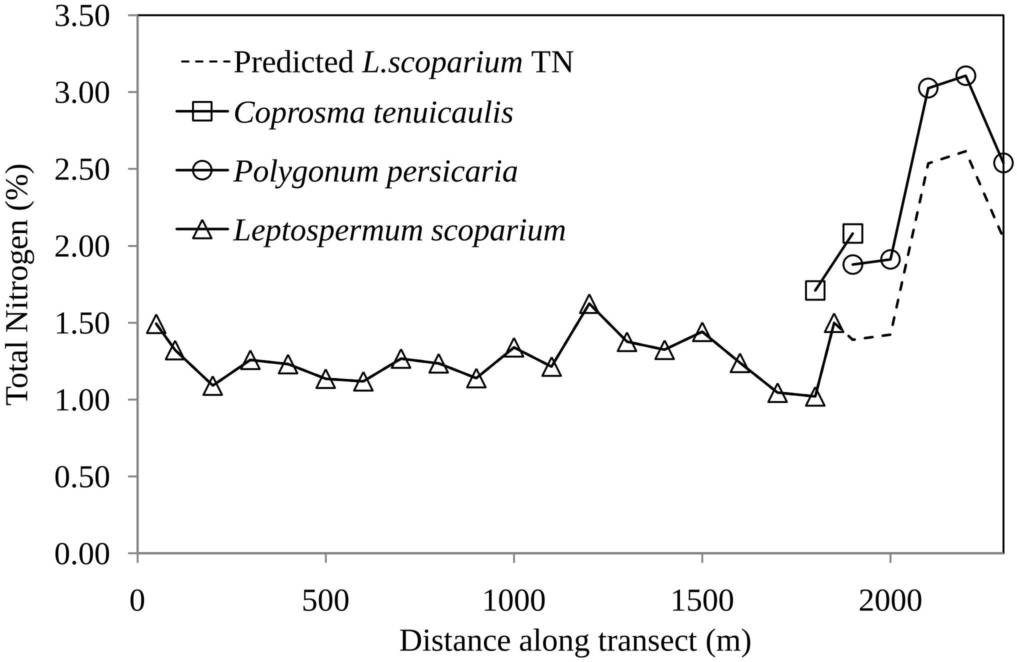


Figure 5.3: Total nitrogen (%) in *Leptospermum scoparium*, *Polygonum persicaria* and *Coprosma tenuicaulis* foliage samples and predicted *L.scoparium* TN (%) along the Whangamarino wetland transect from 50 m to 2300 m.

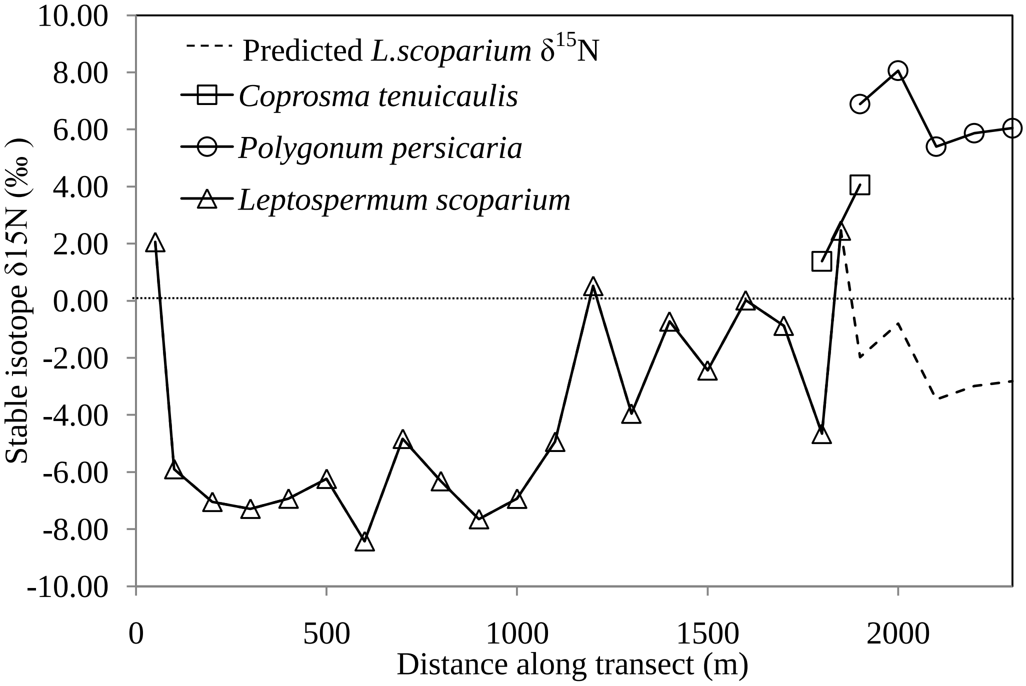


Figure 5.4: Isotope δ^{15}N (‰) abundance in *Leptospermum scoparium*, *Polygonum persicaria* and *Coprosma tenuicaulis* foliage samples and predicted *L.scoparium* δ^{15}N (‰) along the Whangamarino wetland transect from 50 m to 2300 m.

Foliage total phosphorus (Figure 5.5) measured by Hills Laboratory showed similar trends to the total nitrogen results. Both nitrogen and phosphorus had an initial increase (above the normal wetland foliage levels) at around 1800–2000 m. TP rose from a minimum of 0.04 % at 1700 m to a maximum of 0.145% at 2200 m.

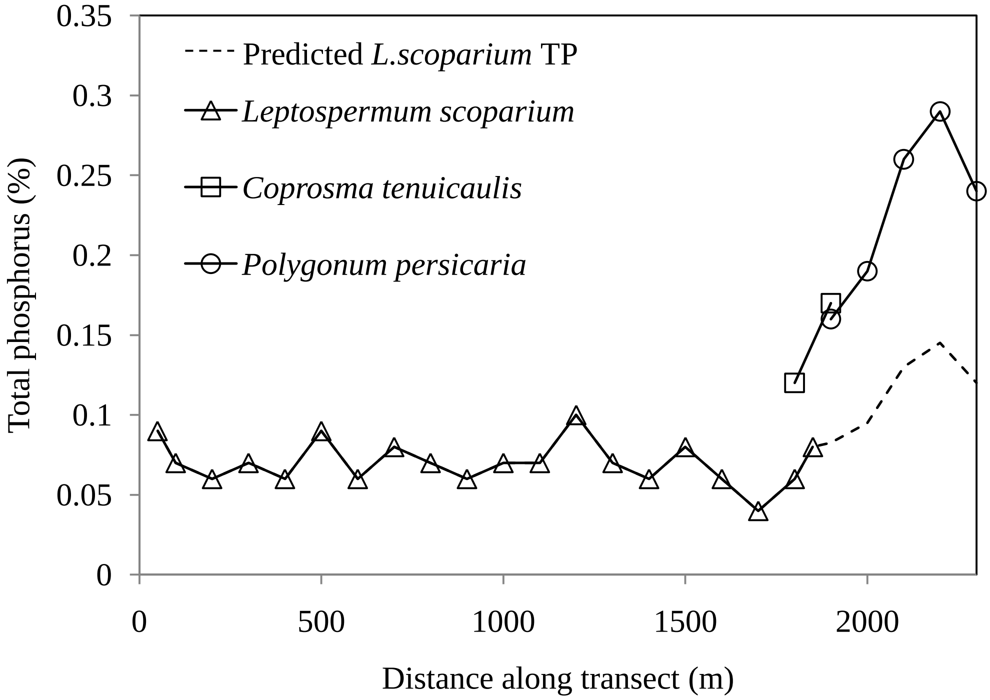


Figure 5.5: Total phosphorus abundance in *Leptospermum scoparium*, *Polygonum persicaria* and *Coprosma tenuicaulis* foliage samples and predicted *L.scoparium* TP (%) along the Whangamarino wetland transect from 50 m to 2300 m.

Foliage nitrogen and phosphorus results (Figure 5.3 & 5.5) peaked around 2000–2200 m and was consistent with the highest levels of mineral substrate. *P. arundinacea* (phalaris grass) and *P. persicaria* were the main species present in this zone. These field results from the wetland plot sheets can be found in the Appendix D.

N:P ratios in plant biomass are useful to determine whether a wetland is limited by certain nutrients (Figure 5.6). Ratios express the wetland is most likely limited by phosphorus.

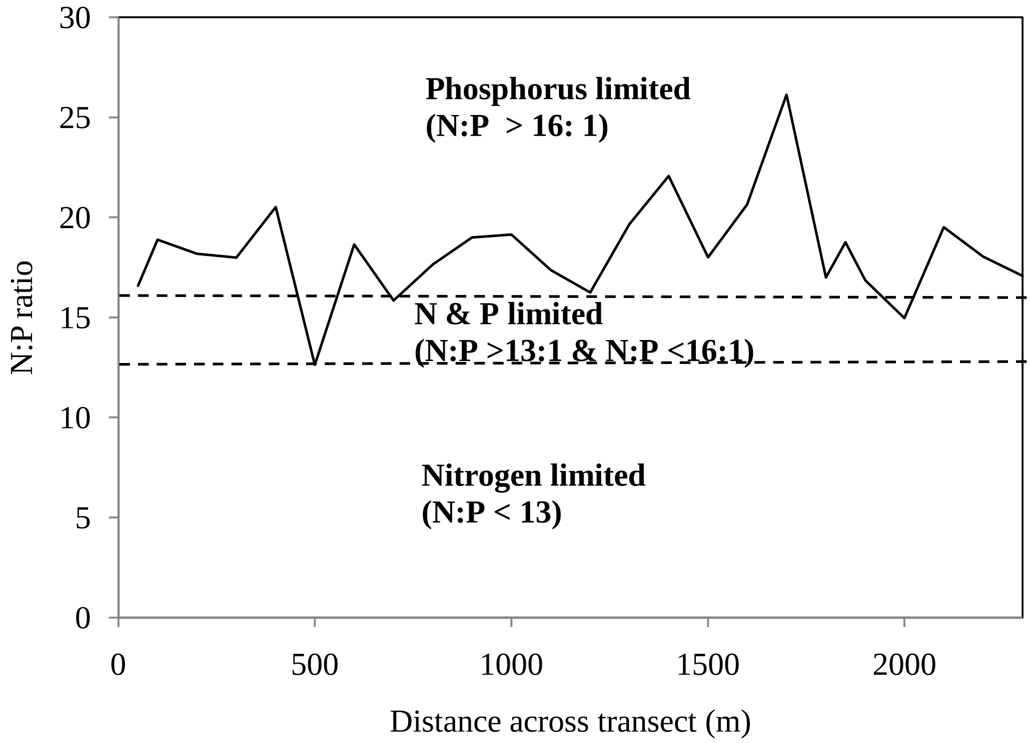


Figure 5.6: Plant biomass N:P ratios along the Whangamarino transect line from measured and predicted *L. scoparium* foliage chemistry results. Indications of phosphorus and or nitrogen limited conditions are based on Güsewell & Koerselman (2002) and Sorrell *et al.* (2004).

5.3.2 Vegetation ordination

An ordination was run to determine group clusters and relationships between species composition from the various sites. The output was an ordination plot and a tree dendrogram. The tree dendrogram (Figure 5.7) expressed the relationship between each of the sites, and the 6 final group clusters. These cluster group boundaries where then applied in an ordination plot of all the vegetation sites (Figure 5.8).

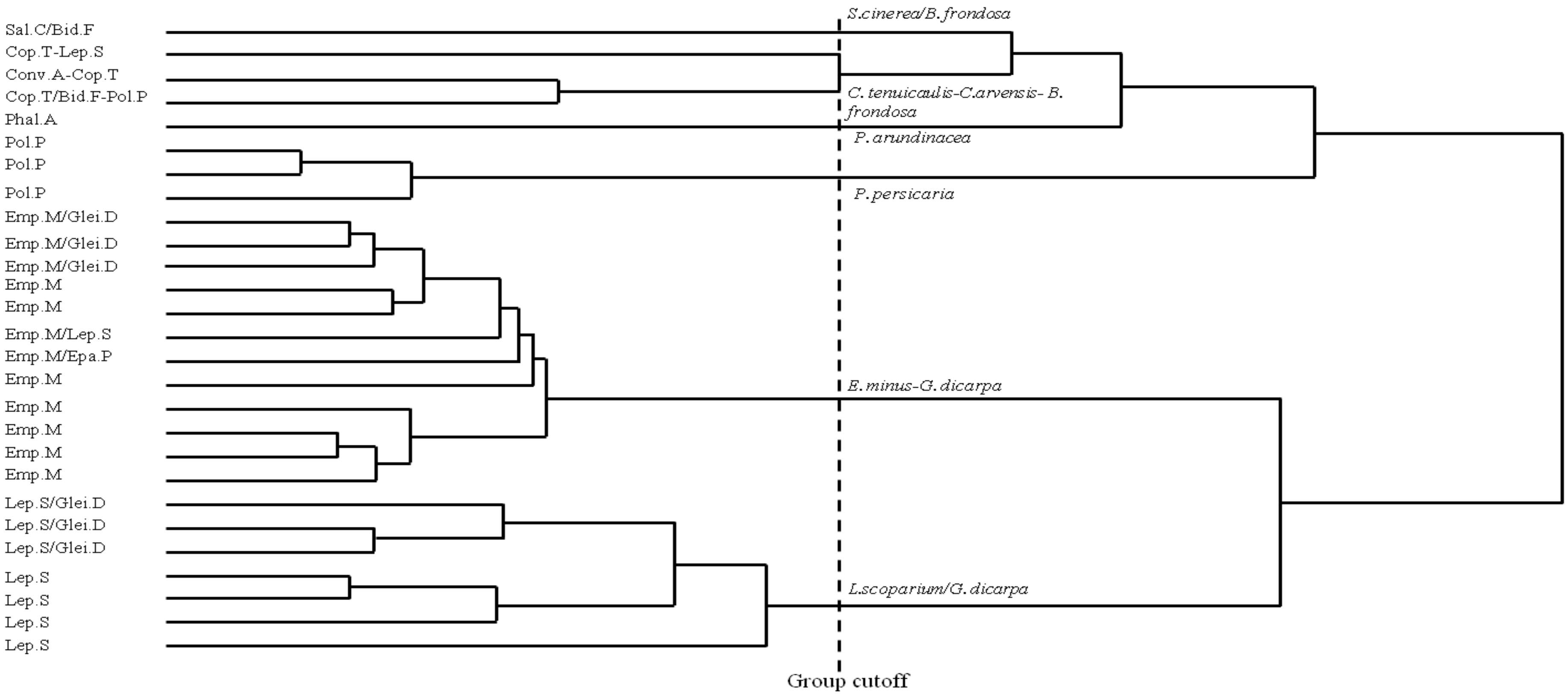


Figure 5.7: Tree dendrogram showing the relationships between the six clusters derived through ordination techniques. The six final groups are *Salix cinerea/Bidens frondosa*, *Coprosma tenuicaulis-Convolvulus arvensis-Bidens frondosa*, *Phalaris arundinacea*, *Polygonum persicaria*, *Empodisma minus-Gleichenia dicarpa* and *Leptospermum scoparium/Gleichenia dicarpa.*

The group clusters derived through ordination corresponded with field observations. Zones of vegetation that were obvious to the field observer ultimately ended up being classified as individual groups, such as the restiad bog characterised by *E. minus* and *G. dicarpa.*

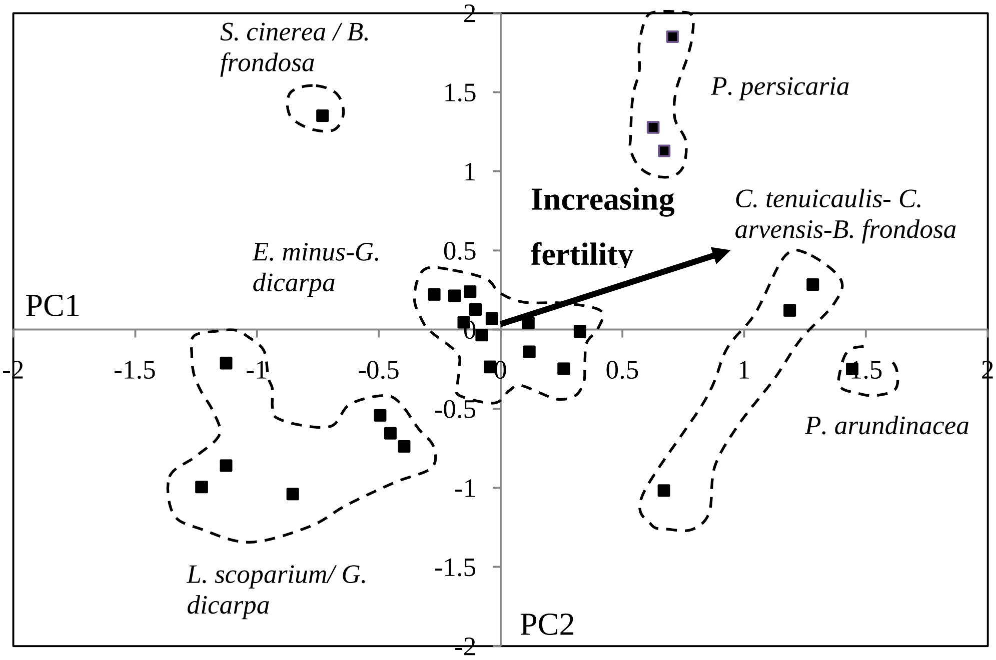


Figure 5.8: Ordination plot exhibiting clustering and relationships between the 27 vegetation sites along the Whangamarino wetland transect line. Dashed lines and names indicate the cluster groupings from Figure 5.6. The arrow indicating increasing plot fertility is derived from Figure 5.9.

Group clusters (Figure 5.8) were plotted in an arbitrary space, the axis having no meaning or units. The closer each data point was together, the greater the relationship. For example, the tight grouping of 12 vegetation sites for *E. minus – G. dicarpa* (50–1100 m) suggested those sites were very similar in species composition. Alternatively, the cluster dominated by *L. scoparium / G. dicarpa* had a poor relationship with the cluster of sites containing only *P. persicaria.* All sites were constrained by a 2d ordination, which could not necessarily show the nutrient gradient across the transect line as well as a 3d ordination would. The fertility arrow is an indication only, and the best fit possible through PCC based on the ordination (Figure 5.9). It would be relevant to assume that the *L. scoparium / G. dicarpa* group is not the lowest in nutrients, which is elaborated on in Figures 5.10–5.12 (B. Clarkson, personal communication October 21st 2010).

From the ordination plot (Figure 5.8) environmental variables were added in through PCC. Variables included peat and foliage chemical and physical data, and also species plot data such as maximum heights (Figure 5.9).

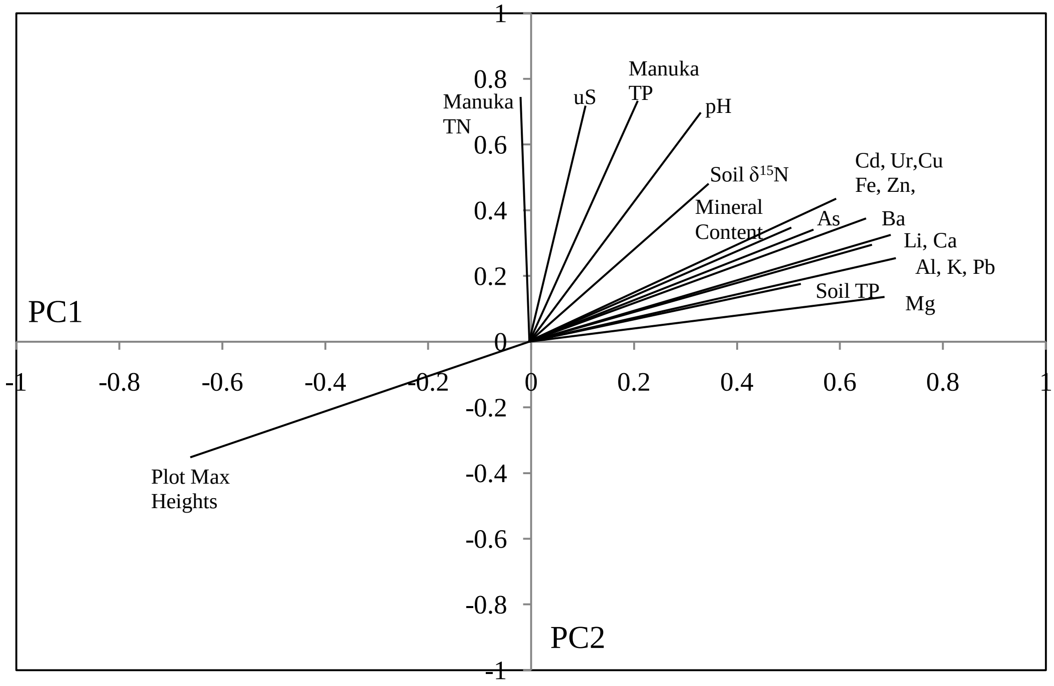


Figure 5.9: Vectors fitted on an ordination plot based on environmental variables relevant to species composition and determined through PCC and MCAO.

The broad scatter on the ordination plot (Figure 5.8) suggested a wide range of relationships between sites along the transect line. It appeared each of the zones were unique, with the three group clusters closest to the river being found on the right side of the plot, while the three group clusters closest to farmland were on the left of the plot. This is reinforced by the vectors derived by PCC (Figure 5.9) which showed an increased abundance of nutrients towards the groups closest to the river (*C. tenuicaulis*- *C. arvensis*- *B. frondosa*, *P. arundinacea* and *P. persicaria*). The group next to the farmland (*S. cinerea/B. frondosa*) was different from the rest of the plots, likely due to the fact it was high in nutrients from farmland inputs and has a diverse species composition of both invasive and natives.

5.3.3 Group statistics

Group clusters derived from ordination were used in box plots to explain possible relationships with environmental variables (i.e. soil nutrient concentrations) (Figure 5.10–5.12). Plots were ordered from the group nearest to the farmland (*S. cinerea/B. frondosa* at the top), through group divisions along the transect line to the Whangamarino River (*P. persicaria* at the bottom).

An initial high level of nutrients was present in the fringe willow belt (*S. cinerea/B. frondosa*) next to the farmland (0–50 m). Nutrient concentrations (such as TP, TN and elements like Mg) decreased into the wetland, and were lowest in the restiad bog (*E. minus –G. dicarpa* spanning 50–1100 m). Nutrients began to increase again in the *L. scoparium / G. dicarpa* group, where a tall *Leptospermum* canopy (up to 6 m) dominated over other species (1100–1800 m). The next group (*C. tenuicaulis- C. arvensis- B. frondosa*) appeared to have a very broad range of nutrient concentrations (1800–1950 m). Interestingly, this group was composed of three sites, only 150 m along the transect line. This zone was also the last area of dense woody and shrubby vegetation up to heights of 2.5–3.5 m. After this, the final two groups were dominated by vegetation less than 1.0–1.5 m high.

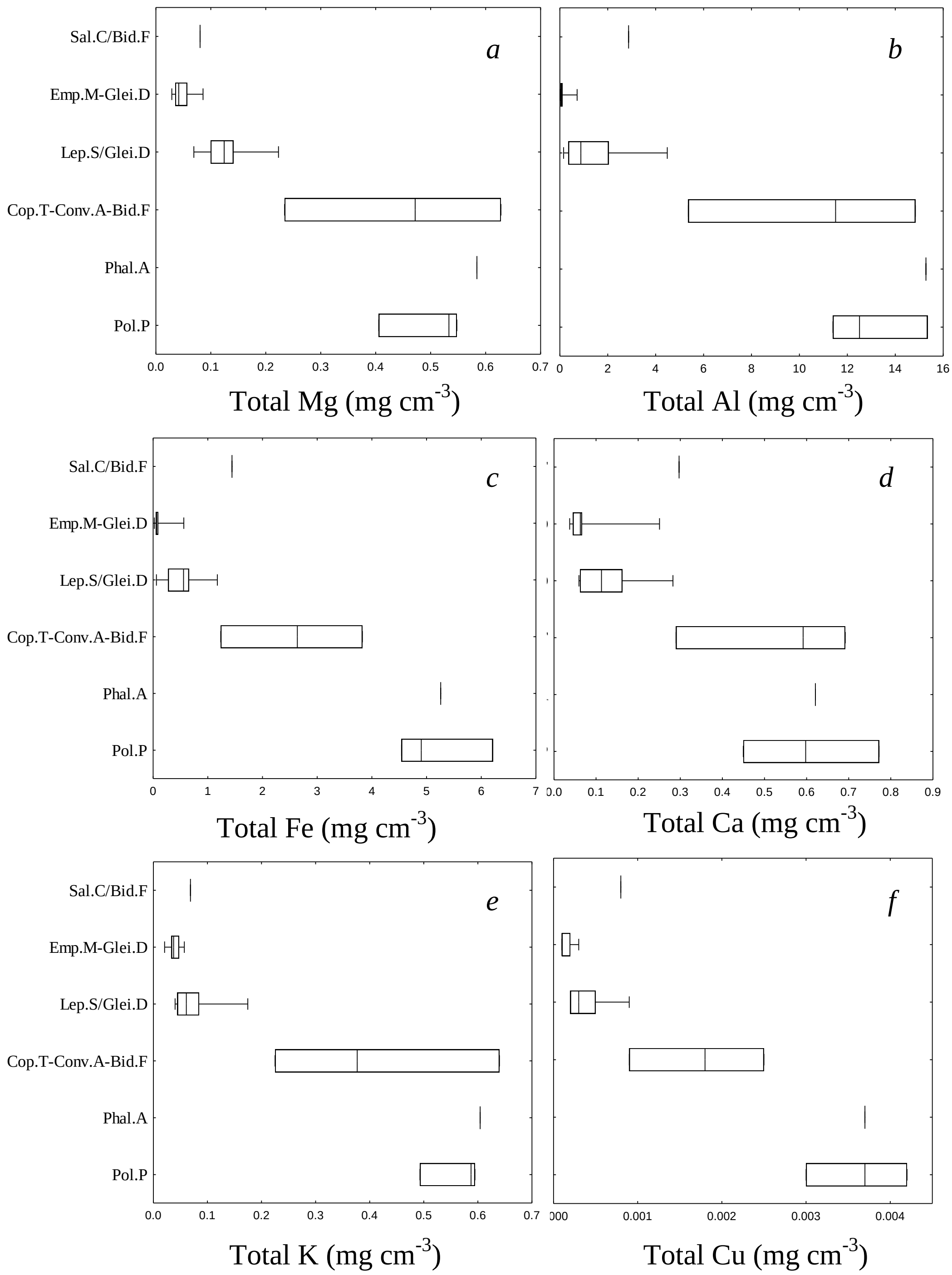


Figure 5.10: Total concentrations (mg cm^{-3}) of [*a*] magnesium (Mg), [*b*] aluminium (Al), [*c*] iron (Fe), [*d*] calcium (Ca), [*e*] potassium (K) and [*f*] copper (Cu) present in peat/soil samples taken along the Whangamarino transect line relative to each vegetation group derived from cluster analysis. *S. cinerea-B. frondosa* location is directly next to the farmland; while *P. persicaria* is located next to the Whangamarino River.

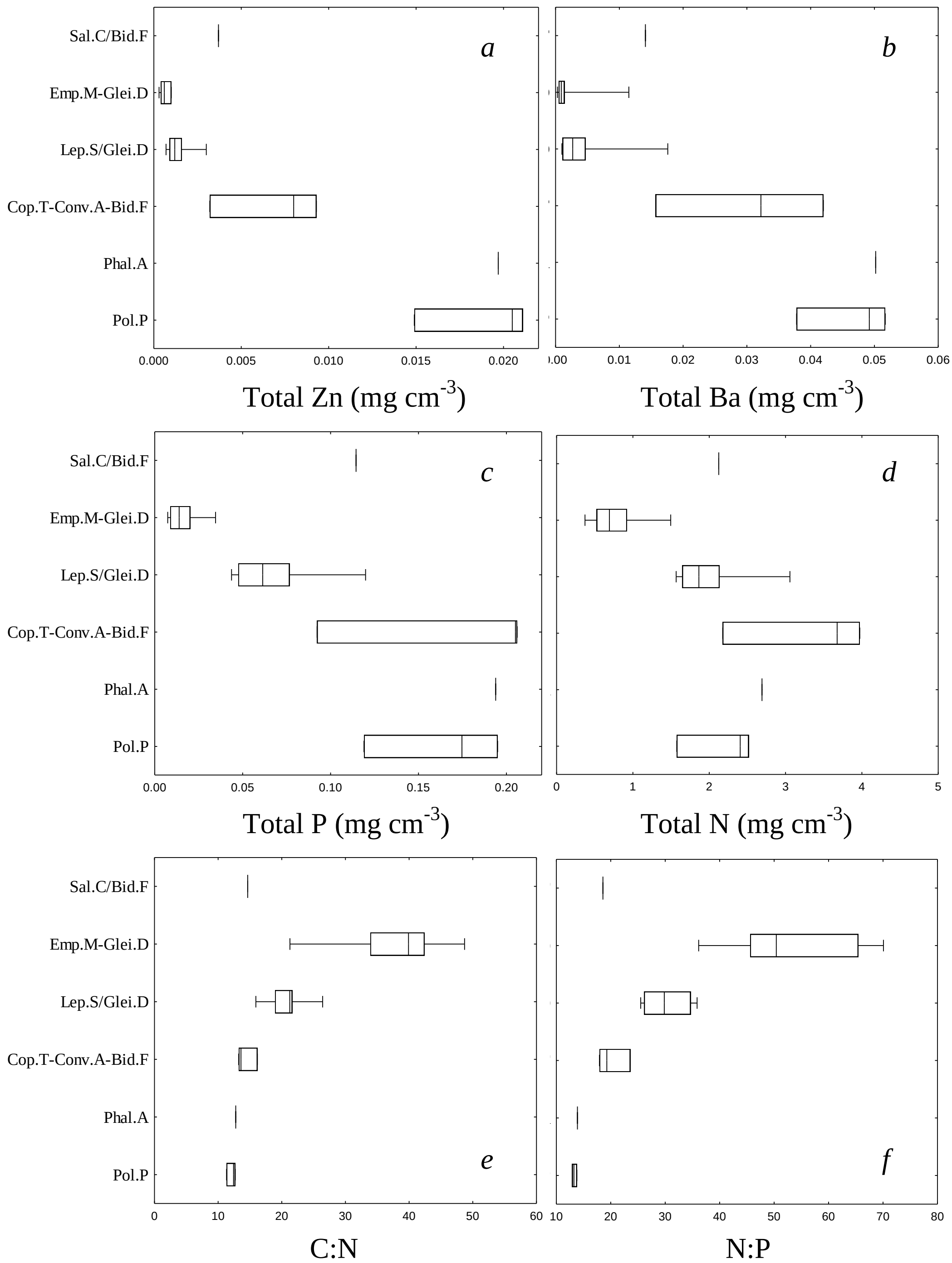


Figure 5.11: Total concentrations (mg cm^{-3}) of [*a*] phosphorus (P), [*b*] zinc (Zn), [*c*] barium (Ba) and [*d*] nitrogen (N) present in soil samples taken along the Whangamarino transect line relative to each vegetation group derived from cluster analysis. Additionally the peat [*e*] C:N and [*f*] N:P ratios are also included.

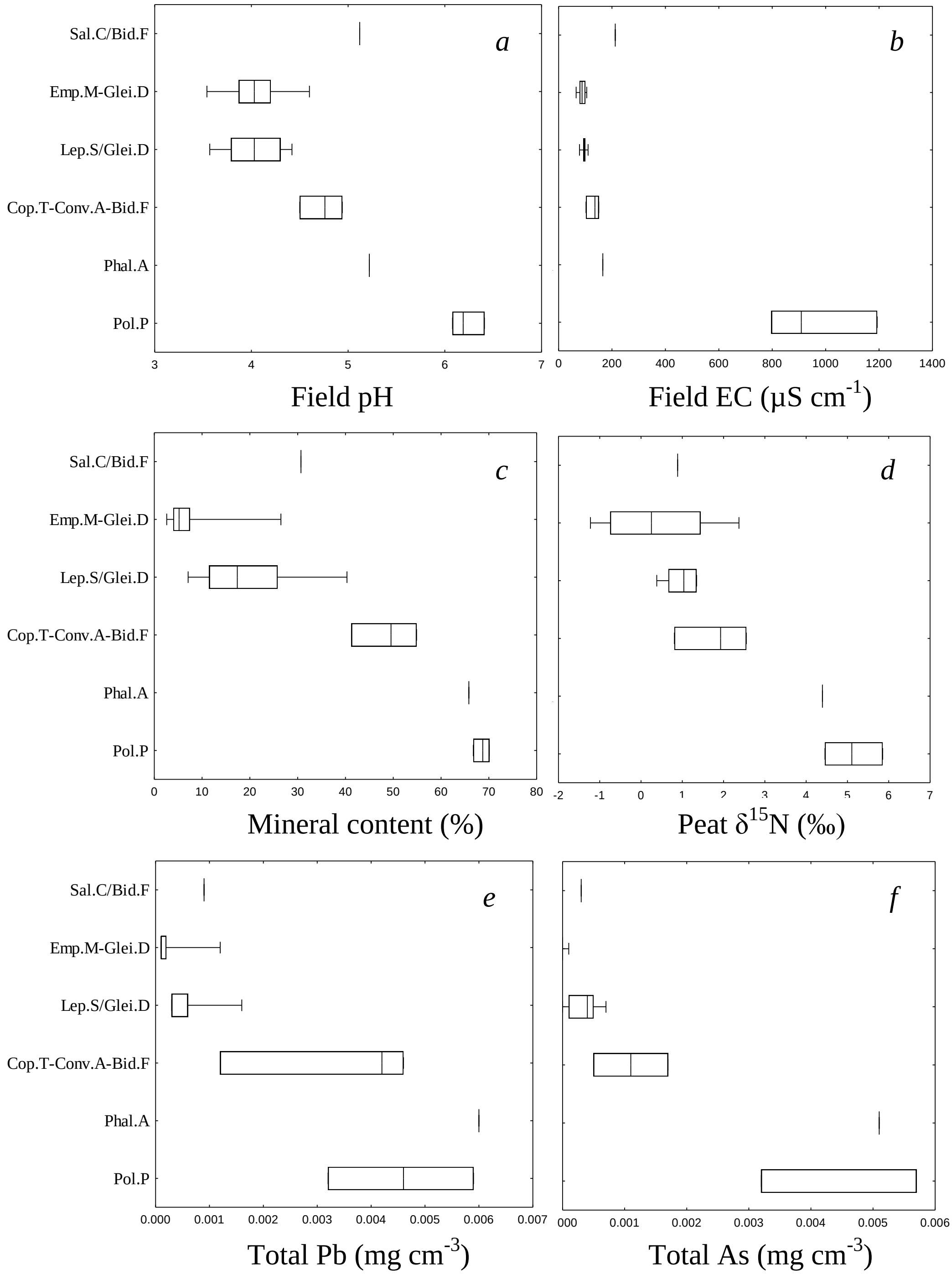


Figure 5.12: [*a*] Field pH and [*b*] conductivity (EC), [*c*] mineral content (%),[*d*] peat $\delta^{15}N$ and total concentrations (mg cm^{-3}) of [*e*] lead (Pb) & [*f*] arsenic (As) present in soil samples taken along the Whangamarino transect line.

The highest concentration of nutrients, mineral content and physical data (such as pH and field conductivity (μS cm^{-1})) were found in the last two vegetation groups. These groups (*P. arundinacea* and *P. persicaria*) were dominated by invasive, weedy plants that were less than 1.5 m tall and formed dense canopies with little ground layer, beside dead matter. The groups exhibited high levels of mineral content (Figure 5.12c) which was deposited from flood water coming from the Pungarehu Canal and Whangamarino River. At these sites, mineral content (rather than organic peat) was up to 70% of the soil sample by weight. The von Post assessment (Chapter 4) for peat decomposition was unable to be used in these last two groupings, because the entire core (from the ground surface) was made up of sediment and dead vegetation.

5.4 Discussion

Along the Whangamarino transect, six distinct vegetation groups were identified. These groups fell within three categories of wetland class, including bog, swamp and marshland.

5.4.1 Plant biomass N:P ratios

While C:N and N:P ratios (in soil) are good indicators of the wetland class and soil limitations (when compared with New Zealand finding by Clarkson *et al.* 2004b), they are not always the best indicator of plant biomass uptake which can vary due to factors such as available nutrients, low temperature, low pH, anoxia, microbial activity or competition (Güsewell & Koerselman 2002). Certain wetland species can solubilise 'unavailable' phosphorus, take up organic nitrogen compounds and can compete against microorganisms for mineralised nutrients which mean that soil concentrations of extractable nutrients may underestimate nutrient availability as experienced by plants (Güsewell & Koerselman 2002).
Some plant species such as sundew (*D. binata*) are even capable of trapping and digesting small invertebrates to augment their limited nitrogen supply (Johnson & Gerbeaux 2004).

Bogs are assumed to be nutrient poor and it would be appropriate to expect low available nutrients for plant species. Mineralisation in bogs (production of inorganic plant available N and P) has been shown to occur at much higher C:N and N:P ratios than described in swamp wetlands and the labile pool (although small) is usually rapidly up taken by plants (Damman 1988; Bridgham *et al.* 1998). Examining N:P ratios in plant biomass (tissue samples) is recognised as being a better indicator of plant available nutrients, including nitrogen saturation or phosphorus limitation (Güsewell & Koerselman 2002; Güsewell *et al.* 2003; Tessier & Raynal 2003).

Generally it has been found that foliage N:P ratios can explain nutrient limitation in at least 75% of worldwide wetlands, while nitrogen and phosphorus concentrations on their own can explain only 5% and 50% (Güsewell & Koerselman 2002; Venterink *et al.* 2002; Güsewell *et al.* 2003). The commonly accepted indicator of a phosphorus limited system is when N:P ratios are greater than 16:1, while a nitrogen limited system is when N:P in plant biomass is below 13:1 (Güsewell & Koerselman 2002; Sorrell *et al*. 2004). Figure 5.6 for *L. scoparium* (measured and predicted) indicated that the majority of this transect was limited by phosphorus, even closer to the river where large sediment and P inputs (Figure 5.11) were present. The wetland being limited by phosphorus may be due to P immobilisation or occlusion through the sorption in the sediment (Chapter 2). N:P ratios can provide indications of compositional and ecosystem functional changes if continued high N and P depositions occur at a site (Tessier & Raynal 2003). This may occur at Whangamarino, due to regular flood inundation and the likelihood of intensification of agriculture (bringing in fertilisers) in the upper catchments.

5.4.2 Foliage $\delta^{15}N$

The $\delta^{15}N$ (‰) found in *L. scoparium* measured and predicted foliage samples showed a depletion in this isotope from 50–1100 m (restiad bog) (Figure 5.4). *L. scoparium* is likely depleted in the restiad bog due to the very low availabilities of nitrogen for plant uptake. Generally the main form of uptake in the restiad bog is from nitrate, which is usually depleted in $\delta^{15}N$ due to fractionation occurring through the nitrification process (Falkengren-Grerup *et al.* 2004).

A study by Clarkson *et al.* (2005) of foliage δ^{15}N abundance in New Zealand bogs (focussing on Kopuatai) showed a positive correlation with foliage P concentrations, where increasing abundance of P in plant matter will result in enrichment in the heavier δ^{15}N isotope (Clarkson *et al.* 2005). Plant P did increase at Whangamarino, and so did δ^{15}N, although did not correlate well. This does not explain why the increase in phosphorus occurred at 1600 m, while δ^{15}N enrichment began at 1100 m (Figure 5.4 & 5.5). The increased δ^{15}N abundance (from 1100 m) was likely attributed to greater nutrient inputs from flood events in this manuka zone (Chapter 6).

Clarkson *et al.* (2005) found *L. scoparium* in New Zealand bogs was significantly depleted in δ^{15}N near the centre of these locations (around -14.96 ‰), while closer to the fringe of the bog (i.e. towards farmland) higher δ^{15}N (averaging around -3.71 ‰) was observed. This was consistent with the findings at Whangamarino wetland, although the *L. scoparium* in the centre of the bog was not as depleted (around -7 ‰) (Figure 5.4). This potentially may be due to the bog age and shallow peat at Whangamarino (when compared to Kopuatai). Restiad species such as *E. minus* in bogs usually have very stable foliage δ^{15}N abundances, due to their wiry stems which direct rainfall to a large cluster of roots capable of stripping out as much as possible of the limited nutrients available (Agnew *et al.* 1993; Clarkson *et al.* 2005). Hence taking *E. minus* foliage samples for δ^{15}N analysis would likely yield similar values to other New Zealand bogs (Clarkson *et al.* 2005).

5.4.3 Farmland swamp belt

At the start of the transect (0–50 m), adjacent to the farmland, swamp conditions appeared to dominate. This was likely due to the primary water inputs originating from surface, subsurface and groundwater flows from the farmland bringing in nutrients which are present in the farm soils due to fertilizers and agricultural practices (such as ruminant urine) (Sorrell *et al.* 2004; Parkyn & Wilcock 2004). Ammonia volatilisation off surrounding farmland was 1.0 kgN ha^{-1}yr^{-1} greater on this fringe belt and also could be linked to an increase in nutrients for plant growth (Appendix C). It is likely these nutrient inputs have led to the colonisation of a large number of invasive, fast growing plant species. This includes *S. cinerea* and *B. frondosa* (beggars tick). This degraded belt of vegetation is only present approximately 50 m (2 sites) into the wetland, from which point restiad bog species then dominate. Additionally the characteristics of this site correspond to similar observations at swamps across New Zealand, seen in Table 2.3 (Chapter 2). The soil pH on this fringe belt was around 5.1, while the mean for New Zealand swamps is 5.2. Additionally total N was 2.1 mg cm^{-3}, consistent with the mean of 2.12 mg cm^{-3} in mineralised swamps (Clarkson *et al.* 2004b). On the other hand, TP concentration in this fringe belt was low (0.11 mg cm^{-3}) compared to the concentrations found in sites receiving riverine inputs, and to New Zealand swamp ranges (0.15-0.59 mg cm^{-3}) (Clarkson *et al.* 2004b). The reason this site was lower in TP is likely due to the farming practices operating, where the low intensity dry stock (beef) farming will have reduced inputs of fertilizer such as superphosphate (Roy 2001). When looking at C:N (14:1) and N:P (18:1) ratios present in the soil (as an indicator of the availability of nutrients for plant growth) the values fall within the common New Zealand swamp readings (Table 2.3, Clarkson *et al.* 2004b), which further indicated the zone is a degraded mineralised swampland.

5.4.4 Restiad bog

The zone from 50–1100 m is dominated by restiad species, with the main group identified as *E. minus – G. dicarpa*, and the presence in all plots of *Baumea spp* (Figure 5.8). Species diversity in these areas were generally low, with plots usually having less than 6 different plant groups, likely due to the few native bog plants able to survive in nutrient poor, low pH and saturated conditions. *E. minus* is a bog species which has been shown to be tolerant of a wide range of environmental conditions, including broad pH, at all but the very highest von Post indices, and at low and medium levels of total P and total K (Clarkson *et al.* 2004a). This was consistent with field observations, where *E. minus* was found from 50–1300 m along the transect (including under a dense *L. scoparium* canopy), even though flood inputs (bringing in nutrients) were reaching up to 900 m from the farmland (Chapter 6). Hodges & Rapson (2010) identified *E. minus* as the dominant species involved in ecosystem engineering of fens changing to bogs, and the reduction of this species in Whangamarino wetland (possibly due to flooding and invasion of manuka) could lead to a reversal in the process (bog changing back to a fen or swamp).

The 12 plots (50–1100 m) classified through ordination as *E. minus – G. dicarpa* (Figure 5.8) was likely a restiad bog due to the similar characteristics for other New Zealand bogs seen in Table 2.3 (Johnson & Gerbeaux 2004; Clarkson *et al.* 2004b). The median pH in this group is 4.0 (acidic), total N and P is around 0.7–0.8 mg cm^{-3} and 0.01–0.02 mg cm^{-3} respectively, which was consistent with Clarkson *et al.* (2004b) average values of 0.82 mg cm^{-3} (TN) and 0.08 mg cm^{-3} (TP). Similar characteristics were seen in N:P and C:N ratios. This group had a median N:P of 50 and C:N of 39, which falls close to the averages of New Zealand bogs (39 for N:P and 48.5 for C:N). The low C:N ratio could be due to the low TC concentrations present through this zone (most likely as Whangamarino is only a young bog) (Clarkson *et al.* 2004a).

Interestingly, N:P in the *E. minus* bog was high, which suggested the site was limited by phosphorus while nitrogen may be present in greater abundance. Generally speaking, wetlands with low nutrient inputs from hydrological sources will be limited by phosphorus, more so than nitrogen. As a change in hydrological regime occurs and greater nutrient inputs occur, nitrogen limitation will become dominant. This is due to plants only requiring small amounts of P for growth of which will become more abundant when surface (river) hydrological inputs increase (Sorrell & Gerbeaux 2004). Whangamarino bog is only young, approximately 1850 years old (Clarkson *et al.* 2004a). Over time peat will potentially build up and nutrient availability will become increasingly limited. Eventually well established bog species such as *S. ferrugineus* would colonise and establish in the bog zone next to *E. minus* (Chapter 3) (Clarkson 2002).

5.4.5 Manuka swampland

Further into the wetland (approximately 1 km from the farmland), the hydrological impact of the Whangamarino River begins to influence vegetation composition. In this zone, a noticeable change occurs in nutrients, peat condition and hydrology (Chapters 4 and 6). A dominant canopy of *L. scoparium* prevails from 1100–1800 m. Closer to the river, there were greater hydrological impacts on vegetation composition. The largest flood event in 2010 reached the fringe of the restiad bog (1.4 km from the river), while smaller flood events regularly impact closer to the river. Hence understory bog species further into the *L. scoparium* belt (i.e. *Baumea spp.* and *E. minus*) become less abundant due to nutrient and hydrological changes along a gradient (disappearing from 1500 m). Additional vegetation present in this belt includes *G. dicarpa*, which becomes the dominant understory species from 1300–1800 m. Increased appearances of *P. tenax* (flax), *C. tenuicaulis*, *Blechnum minus* (swamp kiokio) and invasive species such as *Osmunda regalis* (royal fern) also occurred (from 1500 m). This belt cannot be defined as a fen, due to the fact the dominant water inputs are from surface water and rainwater (Johnson & Gerbeaux 2004). Fens are comprised mainly of groundwater inputs, of which would be minimal at this site. It appeared from 1100–1500 m a transition manuka zone had developed, while from 1500–2300 m foliage species and peat characteristics were similar to New Zealand swamp conditions as outlined in Chapter 4, when compared against Clarkson *et al.* (2004b), Table 2.3.

The changing flood regime and likely the historical low river water levels have led to a change in species composition over time in the southern portion of Whangamarino wetland. Reeves (1994) looked at the changes in vegetation composition in the Whangamarino wetland over 50 years. The manuka and willow invasion was observed 2 years (1963) after the control scheme was implemented, with a significant impact observed by 1977 (Figure 5.13) (Reeves 1994). Installation of a weir in 1993 (successfully completed in 2000) has reduced the impact of low water levels. By studying the hydrology on this transect we have been able to identify higher average water tables and flood inundation distance and frequency (Chapter 6). This is now potentially bringing in nutrients (Chapter 4) further into the wetland, and could be a cause for concern regarding continuous encroachment of the manuka belt into the remaining restiad bog.

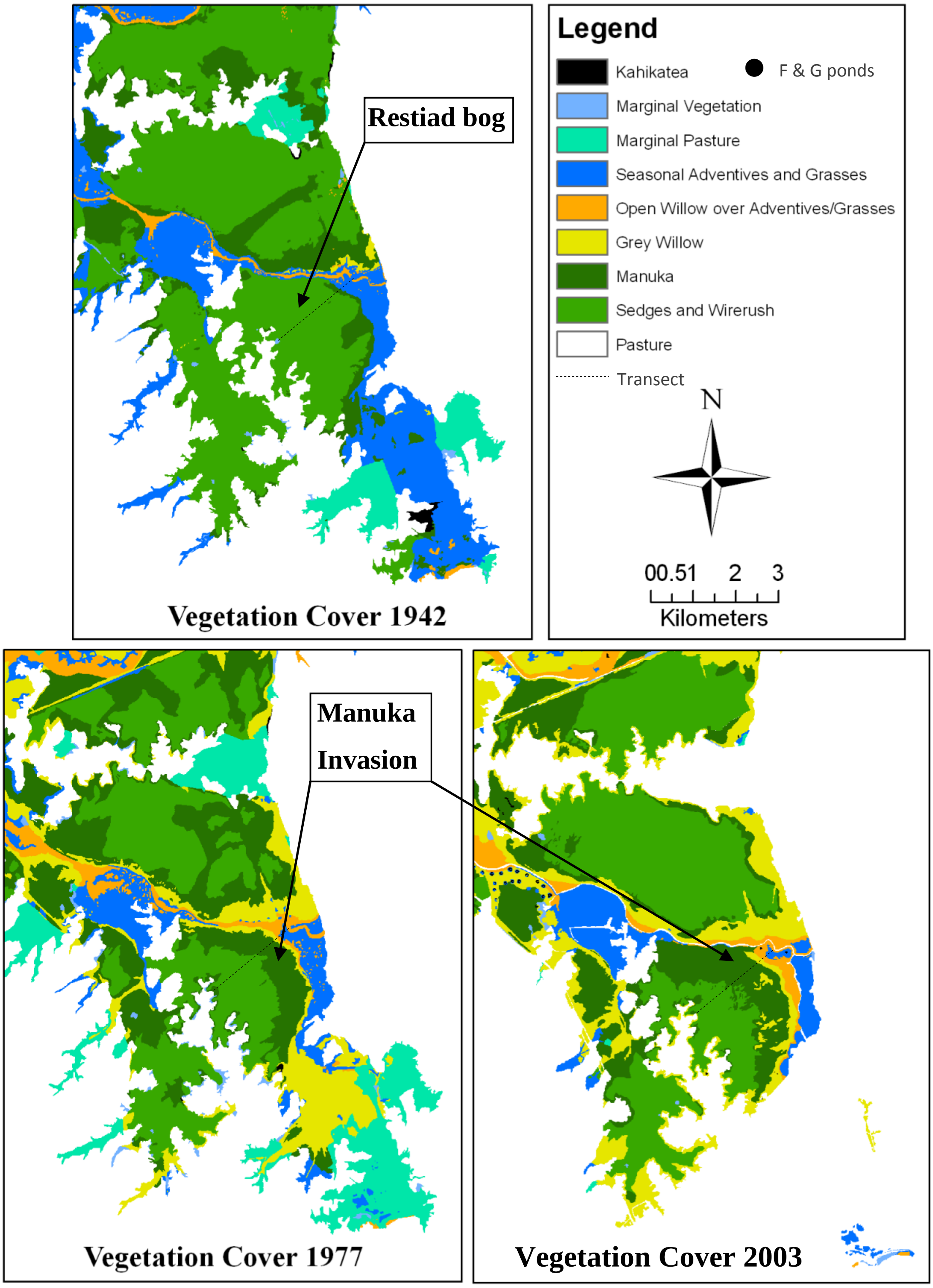


Figure 5.13: Manuka invasion in the southern portion of Whangamarino wetland from 1942 – 2003 (adapted from Reeves 1994). Data was sourced from Department of Conservation, including the 2003 vegetation updates. F & G= Fish and Game.

5.4.6 Mineralised marshland

Following the manuka belt, the vegetation group comprised of *C. tenuicaulis*, *C. arvensis* and *B. frondosa* dominated from 1800–1950 m (Figure 5.8). This zone was only 150 m wide and was a dense, shrubby belt with a wide range in various chemical and physical measurements (Figure 5.10, 5.11 & 5.12). This 2.0–3.0 m tall belt of vegetation was the first area next to the Whangamarino River where vegetation changed from *P. persicaria* and *P. arundinacea* groups (1.0 m tall). The highest nitrogen and phosphorus concentrations were also found in the peat in this area. Potentially this shrubby belt may be stripping a significant proportion of dissolved nutrients and suspended sediment, which would explain the large variation in abundances over a short distance.

Near the river, swamp conditions prevail with a more variable hydrological regime (Chapter 6) and the presence of a mineral soil with higher nutrient concentrations (Figure 5.10, 5.11 & 5.12). This led to the vegetation being dominated by invasive species which could survive in variable and sometimes harsh conditions (i.e. during flood events). *C. tenuicaulis*, *P. persicaria*, *P. arundinacea* and *B. frondosa* were common species present from 1800 – 2300 m along the transect line. Additionally *S. cinerea* and *S. fragilis* colonise many swampland areas around the Waikato Region and can invade further into fen and bogs due to their ability to establish in a variety of environments (Reeves 1994). However significant portions of willow have been removed from spraying operations along the Whangamarino River, which is the main reason annual plants such as willow weed now dominate these swampland fringes. As discussed in Chapter 4, swampland conditions at Whangamarino were directly comparable to New Zealand swamps (Clarkson *et al.* 2004b). Table 7.1 (Chapter 7) summarises Whangamarino wetland swamp and bog ranges compared to New Zealand (Clarkson *et al.* 2004b). N, P, C, C:N and N:P ratios and pH all fell within the ranges for a swamp classification. While peat measurements indicate swamp conditions, vegetation species (dominated by grasses and weeds) in this zone were types commonly found in marshlands. This zone also had a dynamic water table (Chapter 6) and entirely mineral substratum which are found in marshlands.

5.4.7 Water tracts

One area of particular interest along the transect line was the zone of change between the restiad bog and the manuka belt, which began at 900 m and was located next to water level site 3. Interestingly, 100 m on from this site the dominant vegetation composition reverted back to restiad bog (*E. minus*). After 1100 m, the manuka belt was entirely dominant (for the next 700 m). The reason for the change in vegetation composition is possibly due to nutrient and hydrological inputs from large flood events, or as hypothesised by Reeves (1994) the initial invasion may be from a lowered wetland water table due to sand abstraction. A spike in bulk density and decomposition (Figure 4.3 & Figure 4.4) at 900 m expressed the impact surface water inputs were having on peat, resulting in changing vegetation composition and increased degradation (likely due to aeration and a more variable hydrological regime). This was supported by a peak in nitrogen and phosphorus concentrations at the same site (Figure 4.8).

The reason for the change in vegetation composition in this area could be due to the surface elevation in this fringe zone. Areas of restiad bog usually have a raised surface, which can expand with a rising water table (Fritz *et al.* 2008). As large floods inundate the wetland and lose energy the further they travel, water will primarily follow the path of least resistance. Over time, flood events on this fringe belt may have created a crenulated edge of water tracts into the restiad bog, which likely have lower surface elevations, leading to the presence of scrub vegetation species such as manuka which could outcompete restiad species in a more degraded and nutrient rich environment (Figure 5.14).

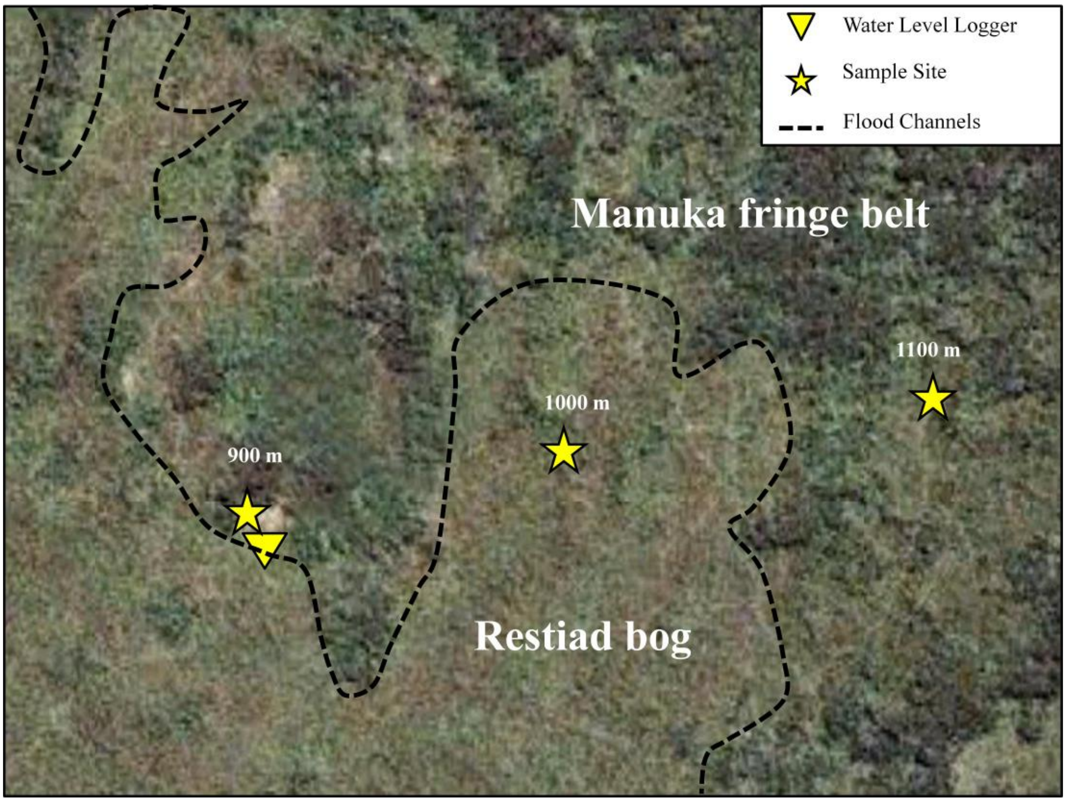


Figure 5.14: Flood water tracts altering vegetation (from restiad bog to manuka fringe) along the Whangamarino transect line (distance measured from farmland). Included in the image is water level site 3 and sample plots 11 (900 m), 12 (1000 m) and 13 (1100 m). Image courtesy of Waikato District Council (2010).

Similar findings of water tracts were present in Opuatia wetland, in the Waikato region. Browne & Campbell (2005) identified water tracks in the restiad bog area through a change in vegetation species from *E.minus* to *Baumea spp*. These water tracts were investigated further using detailed surface level surveys to identify changes in elevation. The results were consistent with what was observed in the vegetation composition, suggesting a depression area that allowed water to encroach further into the centre of the wetland. Ideally to investigate water tracts in the Whangamarino wetland, RTK-GPS should also be used to gain accurate elevation changes across the restiad bog-manuka fringe zone. Due to time constraints this was unable to be undertaken, but would be an important avenue for future research.

5.4.8 Comparison to other studies

The change in vegetation species along a nutrient (and varying hydrological) gradient is consistent with other studies around the world. Both plant growth and tissue nutrient concentration (per unit dry weight) tend to be positively correlated with nutrient supply when all other resources are sufficiently available (Güsewell & Koerselman 2002). Nutrients primarily enter a wetland though hydrological inputs, so a change in species composition at Whangamarino could be directly attributed to increased surface water inputs. Ombrotrophic-minerotrophic gradients (i.e. bog to swamp) are frequently considered to be related to an oligotrophic-eutrophic nutrient status. As bogs only receive atmospheric water input, ombrotrophic peatlands have low nutrient abundance compared to minerotrophic swamps. Most nutrient availability in bogs depends on internal cycling, and while overall nutrients are low, labile pools are rapidly up taken by plants (Bridgham *et al.* 1998). This is generally because these plants (such as *E. minus*) are adapted for nutrient deficient environments though mechanisms such as slow growth, ever-greenness, sclerophylly, high efficiency in nutrient use, high root: shoot ratio, and low optimal concentrations of nutrients (Bridgham *et al.* 1998; Sorrell *et al.* 2004).

A comparative case study was undertaken by Sorrell *et al.* (2007), looking at a New Zealand polje wetland affected by drainage and intense floods during winter (Chapter 2). Vegetation classification was similar to the techniques used in the present study at Whangamarino. Peat samples were taken and assessed for chemical and physical components. The results showed vegetation species-environment relationships were highly correlated with soil water content, aeration, nutrient content and bulk density. Lower and more dynamic water tables increase aeration and favour terrestrial plant species, while wetland plants use internal oxygen transport systems adapted to living in saturated soils. Increased aeration leads to greater nutrient enrichment through mineralisation (Venterink *et al.* 2002). Assessing environmental variables (chemical and physical data) revealed greater interquartile ranges for the invasive communities in the swampland fringe, and smaller variation in native bog species, which is linked to greater water table fluctuations (Sorrell *et al.* 2007). The same results were found at Whangamarino wetland (Figure 5.10, 5.11 & 5.12), where larger chemical fluctuations were observed in the mineralised swampland and marshland (vegetation groups *C. tenuicaulis- C. arvensis-B. frondosa*, *P. arundinacea* and *P. persicaria*).

In contrast to what would be expected in a wetland receiving greater nutrient inputs, lower species diversity was found in flood stressed areas (Sorrell *et al.* 2007). This was because native swampland species could not compete with the pastoral dryland species, and supports the idea abiotic factors are the main control on species composition and richness in flood habitats (Sorrell *et al.* 2007). Similar patterns were observed in Whangamarino wetland, where two groups with the highest nutrient concentrations and mineral abundances had only one plant species present (*P. persicaria* and *P. arundinacea*).

5.5 Summary

- Mineralised swampland dominated in a fringe belt around the farmland end of the transect (0–50 m). The dominant canopy was *S. cinerea* with an understory of *B. frondosa* (and various other invasive species). Chemical and physical measurements were in line with New Zealand swamp conditions as described by Clarkson *et al.* (2004b).

- Foliage N:P ratios provide a better understanding of wetland nutrient limitation than soil C:N and N:P ratios. The foliage results indicated this wetland is likely limited by phosphorus, even with significant P inputs coming in from the Whangamarino River. Phosphorus is possibly being immobilised due to high sediment inputs (Chapter 4).

- From 50–1100 m along the transect, vegetation species and nutrient conditions (matching levels across New Zealand) indicated the presence of a restiad bog. The dominant plant species were *E. minus*, *Baumea spp.* and *G. dicarpa*. Low C:N ratios and previous studies indicated this bog was relatively young (1850 years).

- A manuka belt dominated from 1100–1800 m. This area was characterised by a broad range in nutrient concentrations and physical conditions (such as pH) which increased towards the river (and was consistent with New Zealand swamp levels). The flood control scheme and lowered water tables from 1960–1990 were likely the cause for this manuka invasion into the restiad bog, spreading up to 1 km from the river. The main risk for further encroachment of this zone into the restiad bog is likely from continued flood inundation.

- Possible water tracts were evident on the crenulated fringe between the restiad bog and manuka (900-1100 m). Flood water appeared to be reaching into water level site 3 (900 m), and may have resulted in an increase in nutrient and heavy metal concentrations, and a change in species composition from *E. minus–Baumea spp* to *L. scoparium.* Water could be reaching in due to a decreased elevation at this site, while at 1000 m restiad bog was the dominant wetland class and would likely have a greater elevation (due to a build up in peat) that restricted flood peaks from overtopping the surface.

- Swampland was also present from 1500–2300 m (Whangamarino River). The zone from 1800–1950 m was characterised by a dense, tall (2.0–3.0 m) *C. tenuicaulis* belt with various understory species such as *C. arvensis* and *B. frondosa*. The soils in this belt had large ranges in chemical and physical parameters and were likely stripping out a portion of sediment and nutrients from flood water. From 1950–2300 m, vegetation species *P. arundinacea* and *P. persicaria* dominated (100 % canopy cover). This area was once colonised by grey willow and crack willow which has since been removed by the Department of Conservation. Due to the presence of grasses and weeds, high nutrient levels and entirely mineral substrate, this zone could be classified as a marshland.

- Vegetation composition is directly related to nutrient and hydrological inputs along the Whangamarino transect line. The main risk to a change in biodiversity and loss of pristine restiad bog areas is likely from flood inundation bringing in nutrients and encouraging the encroachment of the manuka belt.

- Changes in vegetation across an ombrotrophic (bog) – minerotrophic (swamp) gradient at Whangamarino are consistent with other studies around the world. Hydrological control is the main cause for vegetation change as it is linked with nutrient input and availability.

Chapter 6: Hydrological processes

6.1 Introduction

Hydrology is the defining feature in wetlands, driving ecology and influencing the community structure of many different plant species. The formation of different types of wetlands is predominantly related to the type of hydrological inputs (Chapter 2).

Along the Whangamarino wetland transect vegetation types and nutrient concentrations are likely to change relative to the different hydrological processes operating. This portion of the Whangamarino wetland has an altered hydrological regime, including inputs from Lake Waikare and the Whangamarino River, which led to enhanced flooding during winter. The flood control scheme and the artificial weir impact on maximum and minimum water levels respectively and play an important role in the wetlands current and future condition.

This chapter will describe the hydrological processes operating along the transect, particularly related to water table behaviour and surface water inundation from flooding. Water levels were measured continuously at representative sites to provide accurate information on the annual and seasonal changes along the transect line. Also elevation differences along the transect and peat surface oscillation (movement of the peat surface as a response to water levels) were identified (Chapter 2). Meteorological data were used to provide continuous records of climatic conditions which could influence the wetland and water levels.

Historical water level records obtained from Environment Waikato were used to identify the changes in river levels over time, and to compare these with the current hydrological trends occurring along the transect line.

Water quality samples were taken during a flood event to identify the potential contributions that nutrient and sediment loads of the Whangamarino River and Lake Waikare were having on the wetland. The impact of flooding along the transect line was assessed in more detail using the Type III extreme value probability distribution. This distribution used annual flood maximums from historical records at Falls Road to identify return periods for flood events of various stage heights (m). Using transect water level data and the relationships these sites had with Whangamarino River during a flood event, inundation distance along the transect could be estimated. This provides information on the distance from the river certain magnitude flood events will reach into the transect line, bringing in nutrients, sediment and altering the hydrology.

6.2 Methods

A variety of approaches were used to gain a better understanding of hydrological processes at Whangamarino wetland. These methods included:

- water level monitoring sites along the transect
- an automated meteorological station
- assessment of historical river water level records dating back 46 years
- flood water quality sampling
- flood return period analysis

6.2.1 Water level monitoring

Manual downloads of the water level and barometric pressure loggers (Chapter 2 and 3) were undertaken on a quarterly basis, although some locations were unable to be downloaded regularly due to the sites being under water through the winter. Raw water levels were compensated with barometric data (from the Barologger), to remove any atmospheric pressure changes that may be affecting the water levels (Solinst 2009). During downloads, manual readings of water level heights (using a tape measure) were also undertaken to ensure quality of data and to identify instrumental drift (Figure 6.1).

Figure 6.1: Matthew Brady (*a*) performing a manual check on water levels at site 1 and (*b*) Craig Hosking undertaking a water level download from the solinst probe at site 4, in early October after the large flood event.

The compensated water level data were stored in a custom database and retrieved for analysis using Matlab (Mathworks Inc., Massachusetts, USA). Quality of data was extremely good, with water level changes able to be observed to the scale of a few mm. There were no equipment failures, and unusual readings or 'data spikes' (possibly caused by manual downloads) were filtered out using Matlab.

The final download was undertaken on 14–Jan–2011. Water level data is available between 17–Mar–2009 to 14–Jan–2011 with the start date differing among sites (Table 3.3).

6.2.1.1 Peat oscillation

Absolute water levels (AWL) were determined through a reference peg (RP) fixed to mineral soil, and surveyed using RTK-GPS. A capacitance probe attached to the peg retrieved water level readings which were then calculated relative to the surveyed peg elevation (Figure 6.2). SE of the peat was determined by subtracting AWL from relative water level (RWL). Relative water level (RWL) is the depth of the water table below the peat surface (unsaturated zone) and was measured in the field by a capacitance probe connected to a wooden board that was attached to the peat surface. This ensured the probe measured RWL (water table depth) as the peat surface shrunk and swelled over the year (Figure 6.2).

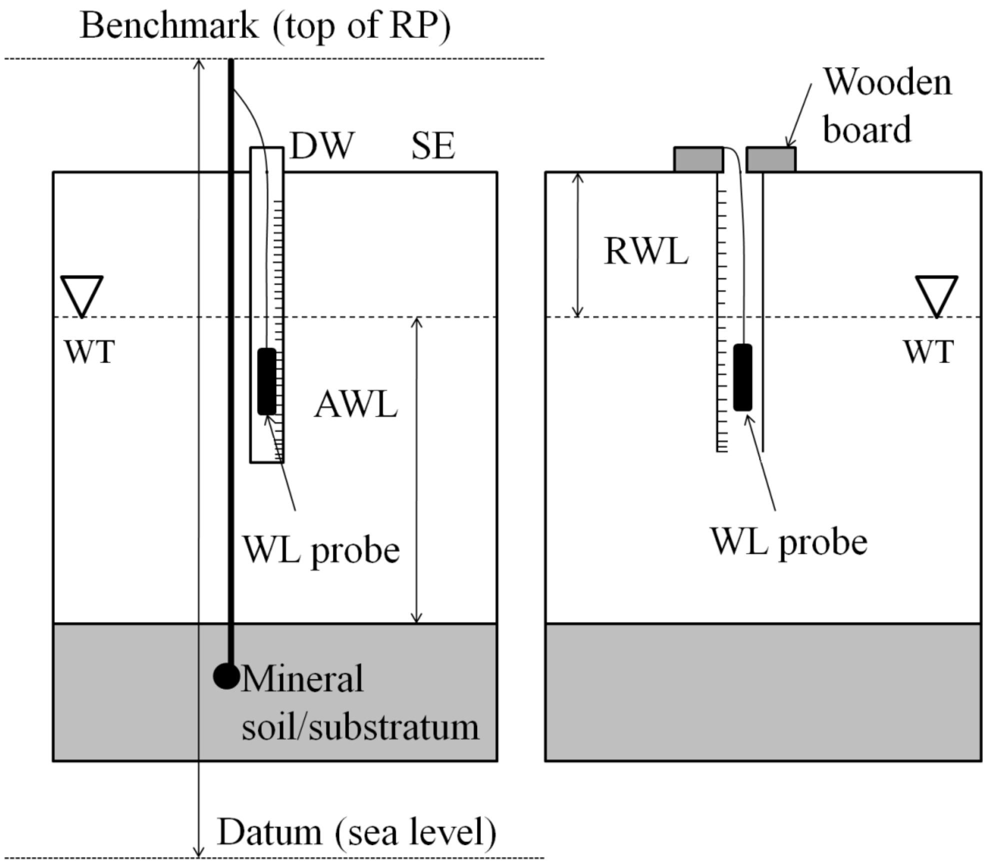


Figure 6.2: Diagram showing how water table (WT) was measured from a pressure transducer probe positioned in a slotted dipwell (DW) below the water table. Absolute water level was calculated from the probe readings by comparing with a surveyed reference peg (RP) anchored into a mineral substratum below peat layers. The top of the RP (fixed) is surveyed to the sea surface datum, from which AWL was then derived. Relative water level was derived from a probe 'floating' on the peat surface (attached to a wooden board). Surface elevation (SE) can be calculated from AWL and RWL.

6.2.2 Meteorological station

The meteorological station site had two water level recorders and a barologger probe. Stored data from all instruments were downloaded remotely using a cell phone modem, and saved in a database accessed by Matlab. The equivalent time vectors were used so all data could be plotted against one another.

6.2.3 Historical stage records

Historical Whangamarino River stage (m) and Lake Waikare records were provided by Environment Waikato, which had some gaps throughout the timeframe due to calibration checks and equipment malfunction. These historical records were added into the database and plotted with a similar time vector as the data retrieved from the Whangamarino transect line, so water levels could be compared. Historical records were also used in flood return period analysis based on annual maximums.

6.2.4 Flood Sampling

Water quality samples were collected during a flood event in August. Sampling started on 13-Aug-2010 and finished on 16-Aug-2010. Sample sites were at Whangamarino River, Falls Road Bridge and the Pungarehu Canal at the Waeranga Road Bridge. The samples were collected every 4–6 hours for 3 days, with a total of 28 samples being retrieved throughout the flood hydrograph. This flood event was relatively small compared to the larger events in June and September, and unfortunately had less than predicted rainfall which was also localised, resulting in a moderate flood response in the Whangamarino River, but only a small change in the Pungarehu Canal.

Samples were collected from the bridges using a 3 litre bucket, pre washing the bucket with flood water, then filling a 1 litre bottle with unfiltered sample water, which was later preserved with 2 ml of sulphuric acid. Samples were stored in a refrigerator and analysed for total suspended solids (TSS) using method 2540 D from Norweco (2010). 50 ml Falcon tubes were also filled with filtered sample water (0.45 µm filter), frozen as soon as possible, and later analysed for NH_4^+ (ammonium), NO_2 (nitrite), NO_x (Nitrous oxides NO and NO_2) DRP (dissolved reactive phosphorus) through a discrete analyser (Department of Biological Science). Measured concentrations could be considered the biologically available (dissolved) nutrients (Environmental Protection Agency 2010b).

Refer to Gibbs (2009) for more detail on the water quality sampling approach.

6.2.5 Flood return period and inundation distance

Annual maximum flood events were first identified through the historical records at Falls Road. The annual stage (m) maxima were plotted in a probability distribution to predict return periods. A Type III extreme value probability distribution was chosen, after trialling a type II (Gumbel) distribution which assumes a linear relationship between stage and return periods (Pearson & Henderson 2004).

Bardsley (1989) proposed a variety of functions to predict flood event return periods from river discharge which were adapted to fit river stage. Earl Bardsley provided a spreadsheet in Excel which predicted return periods based on annual stage maxima. Return periods were predicted from an equation, which corresponds to a Z value derived through applying the type III distribution equations (Bardsley 1989).

When each water level site first responded (in a 1:1 relationship) to the Whangamarino River during measured flood events, an estimate of the inundation distance relative to river stage could then derived (Figure 6.7). This was through the knowledge of the exact distance each of the water level sites was from the river. For example, site 4 (450 m from farmland) first responded along a 1:1 relationship at a river stage height of 3.96 m (Figure 6.7). The known river stage height when each of the water level sites first responded to a flood event was then plotted against inundation distance. A linear relationship was derived, from which extrapolation of the falls road stage height for various return periods (such as 10, 20, 50 and 100 year flood events) was then undertaken to identify the likely inundation distance into the wetland for various magnitude flood events (Table 6.2, Figure 6.14).

Wetland water levels at various river stages were determined by plotting falls road stage against sites 3–5 stages, when each of the water level sites first responded to the flood event. Extrapolation as per the inundation method was then undertaken to estimate wetland stage for other return periods (i.e. 50 and 100 year events).

6.3 Results

Hydrological processes affecting the wetland transect were evident at a range of timescales, from instantaneous (response to rainfall events) to multiyear (e.g. changes in river base levels). Results will focus on key trends throughout the calendar year of 2010, including important events (such as large floods) which impacted on the wetland.

6.3.1 Rainfall

Annual rainfall for 2010 totalled 1007 mm, below normal for the 30 year average (1280 mm). The seasonal pattern consisted of below average rainfall from January to April and October to December. May, June and August were wetter than normal (Figure 6.3).

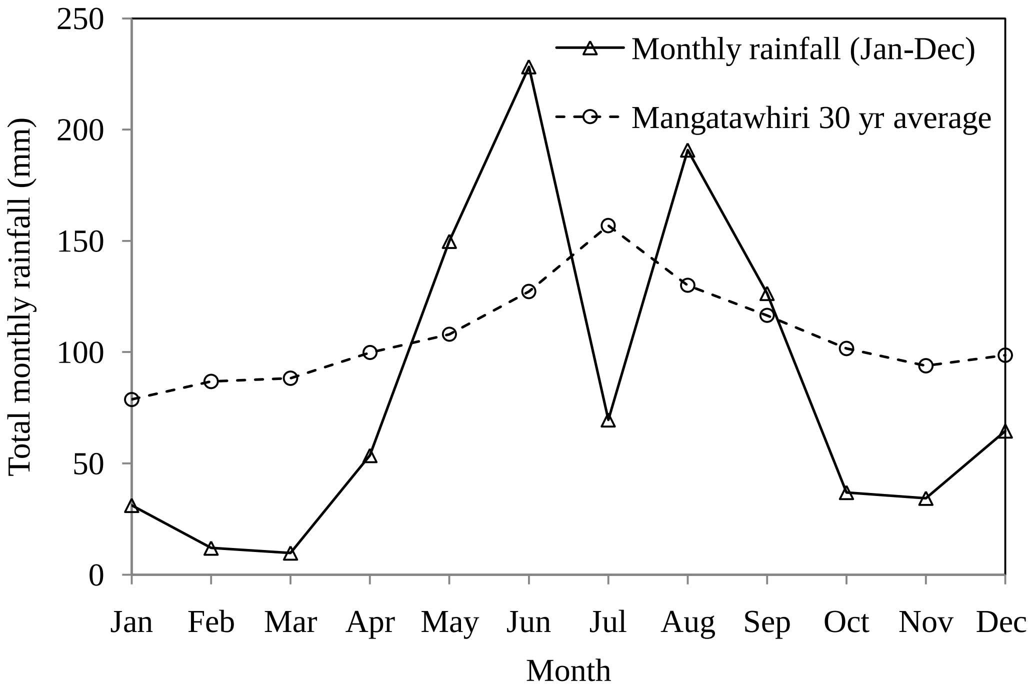


Figure 6.3: Monthly rainfall for Whangamarino wetland from 1-Jan-2010 to 31-Dec-2010 compared to 30 year rainfall averages (1979–2009) from NIWA's long term site at Mangatawhiri approximately 15 km north of the transect.

6.3.2 Absolute water levels

Water levels showed changes in elevation along the transect line due to the wetland surface elevation decreasing towards the river (Figure 6.4). Seasonal trends showed higher water levels in winter and spring (June to November) and lower water levels through summer and autumn (December to May). The lowest water levels were observed in April 2010, the highest in late September 2010.

Water levels followed rainfall patterns (Figure 6.5 & 6.6). Sites 1, 2 and 3 expressed a hydrological regime linked strongly to rainfall (Figure 6.5). Increases in water levels were due to rainfall, while decreases were due to prolonged periods without rainfall (January to April). Water level decreased along the transect as surface elevation decreased (Figure 6.4). Site 3.5 (manuka zone) expressed patterns similar to sites 1–3, being driven by rainfall for a large part of the year, but on two occasions, June and September 2010, water levels were controlled entirely by the river. Sites 4, 5 and 6 showed dynamic water tables driven by changes in the Whangamarino River (Figure 6.6). These sites were frequently affected by flood events during the winter and were underwater for several months in winter and spring (Figure 6.7 and 6.8).

As river levels declined after flood events, individual sites reacted according to their local controls. Above surface water levels in the mineralised zones closer to the river (site 4–6), had a rapid decline after rainfall events, likely due to surface overland flow. Sites 1–3.5 generally had a slower decline in water levels after rainfall events as water levels were below the surface and influenced by internal controls (such as peat hydraulics) (Figure 6.4, 6.5 & 6.6).

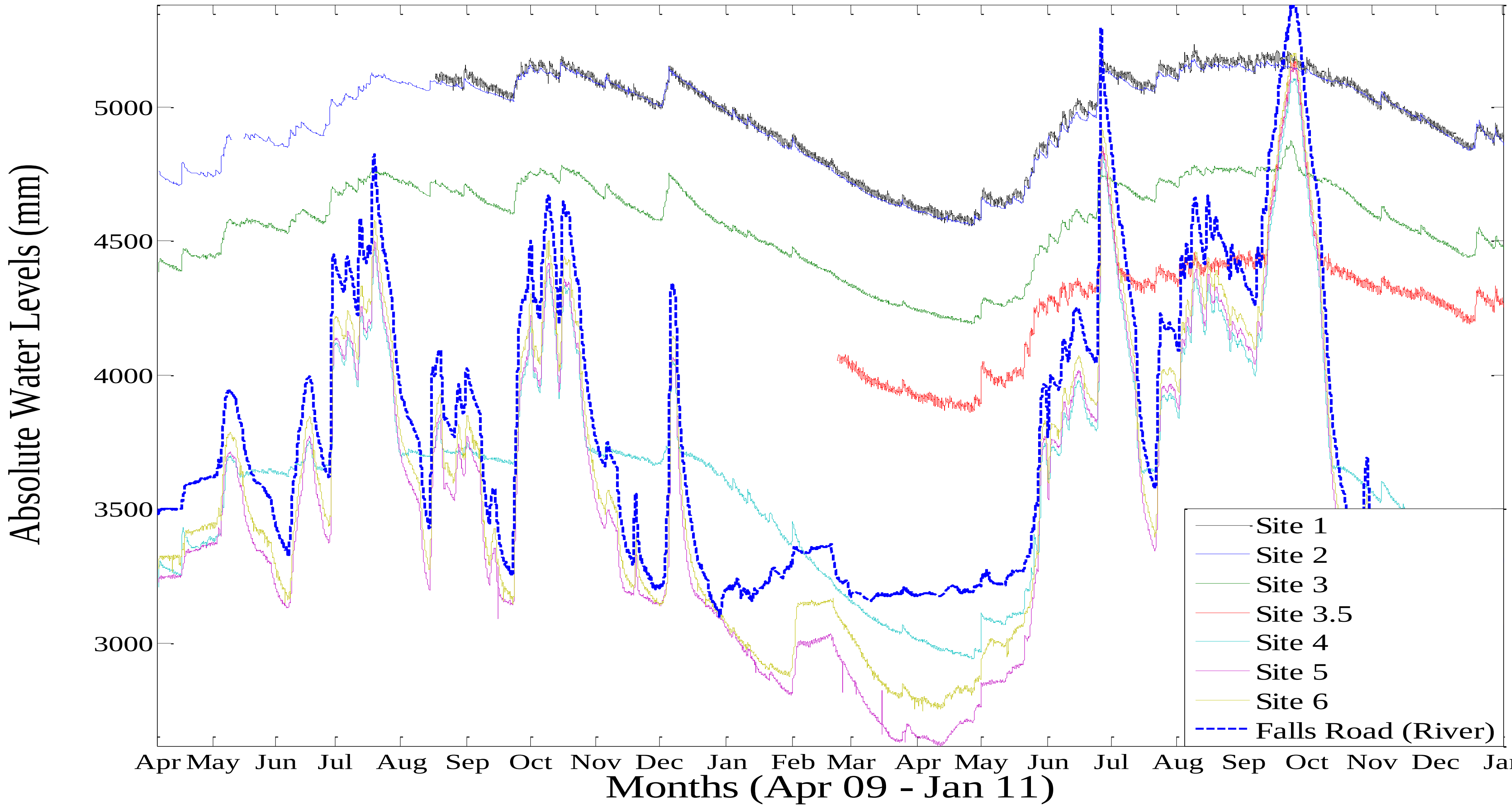


Figure 6.4: Absolute water levels (mm) for seven water level sites and the Whangamarino River along the transect from 9-Apr-2009 to 31-Dec-2010.

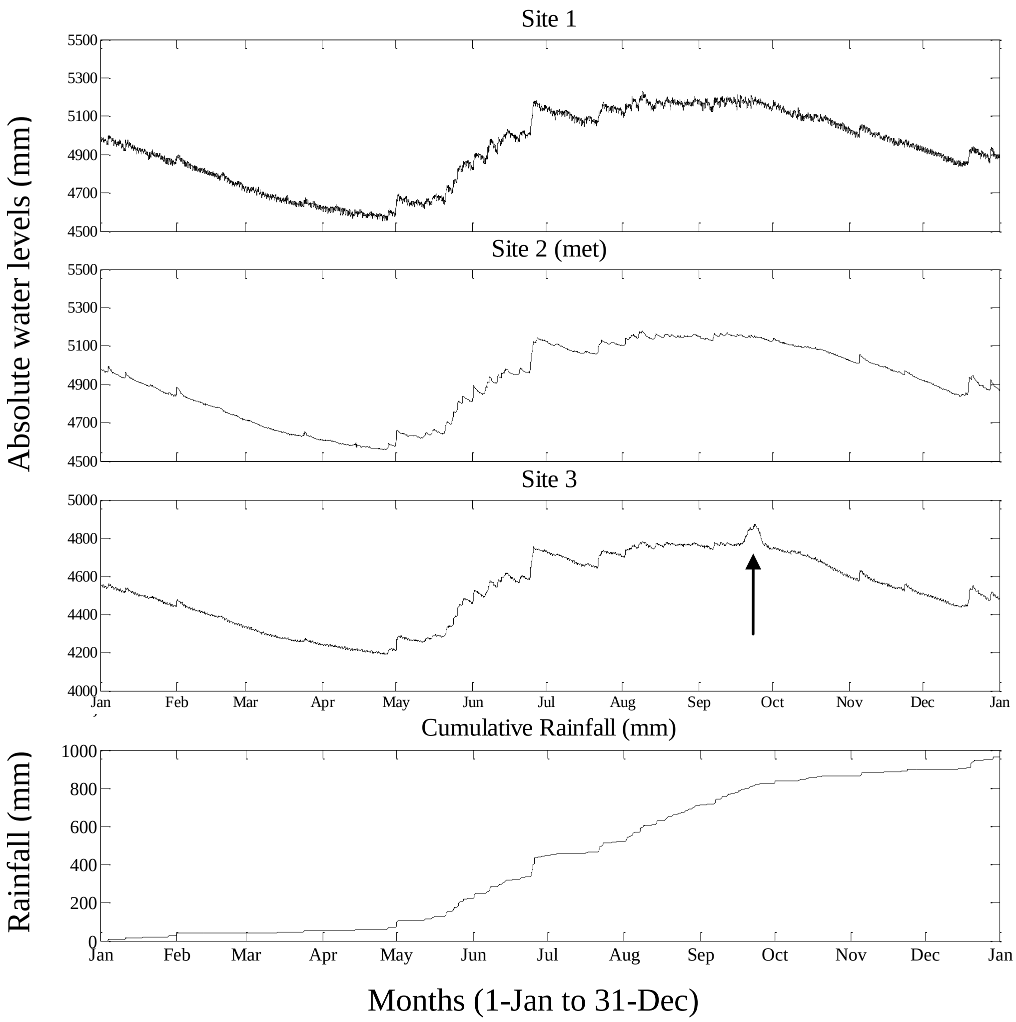


Figure 6.5: Absolute water levels and cumulative rainfall for sites 1, 2 and 3 from 1-Jan-2010 to 31-Dec-2010. Arrow indicates when inundation from Whangamarino River reached site 3.

Significant rainfall events occurred in late June and September, both of which caused an increase in water levels and large scale flooding in the Whangamarino wetland (Figure 6.5 & 6.6). The flood event in September reached into site 3 (Figure 6.5). Site 3.5, in the manuka zone, responded to the Whangamarino River flood stage in June and September 2010. Sites 4, 5 and 6 were inundated by river water from 20-May to 15-October (Figure 6.5 & 6.8).

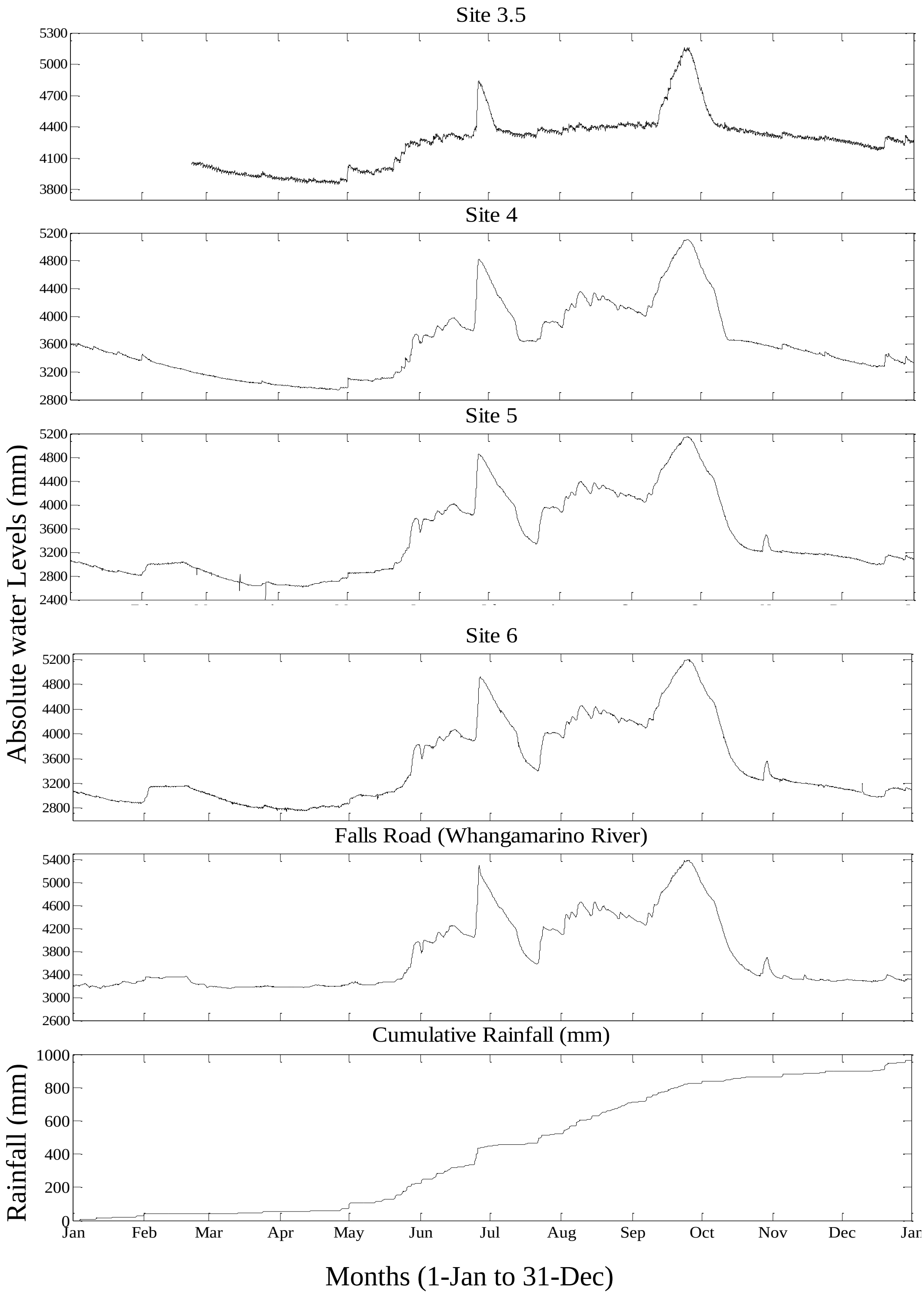


Figure 6.6: Water levels and cumulative rainfall for sites 3.5, 4, 5 and 6 and Whangamarino River (Falls Road) from 1-Jan-2010 to 31-Dec-2010.

When water levels along the transect were plotted against Whangamarino River stage, the elevation at which inundation first occurred is clearly evident because the response at each site is followed parallel to the 1:1 line (Figure 6.7).

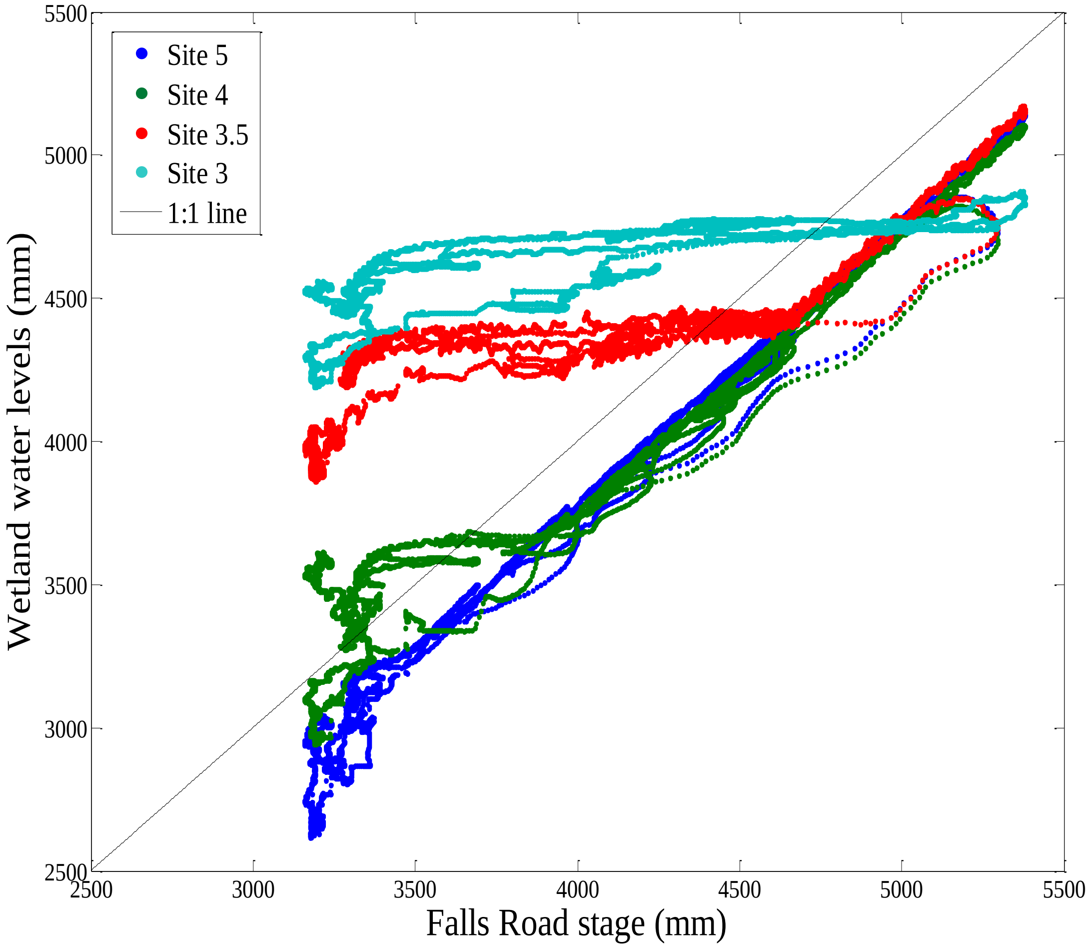


Figure 6.7: Absolute water levels at site 3, 3.5, 4 and 5 versus Whangamarino River stage (Falls Road) from 1-Jan-2010 to 31-Dec-2010. Response parallel the 1:1 line indicates the water table is being controlled by the river, due to flood inundation.

6.3.3 Relative water levels

Plotting RWL was useful to determine whether the water level overtopped the surface during the year, and how dynamic the water table regime was at each of the sites. RWL could only be derived where it was specifically measured (site 2) or where the wetland surface was assumed not to oscillate.

RWL became more dynamic towards the river. At site 2 the water table did not overtop the surface. Site 3.5 in the manuka belt had a similar water level pattern to site 2 over summer (up to June), but was inundated through winter by flood events and spent a large portion of winter under water (averaging around 10 mm, but peaking at around 800 mm). At site 5 water levels were around 600 mm below the surface during summer, but were influenced strongly by river flooding through winter, peaking at around 1800 mm during the September flood (Figure 6.8).

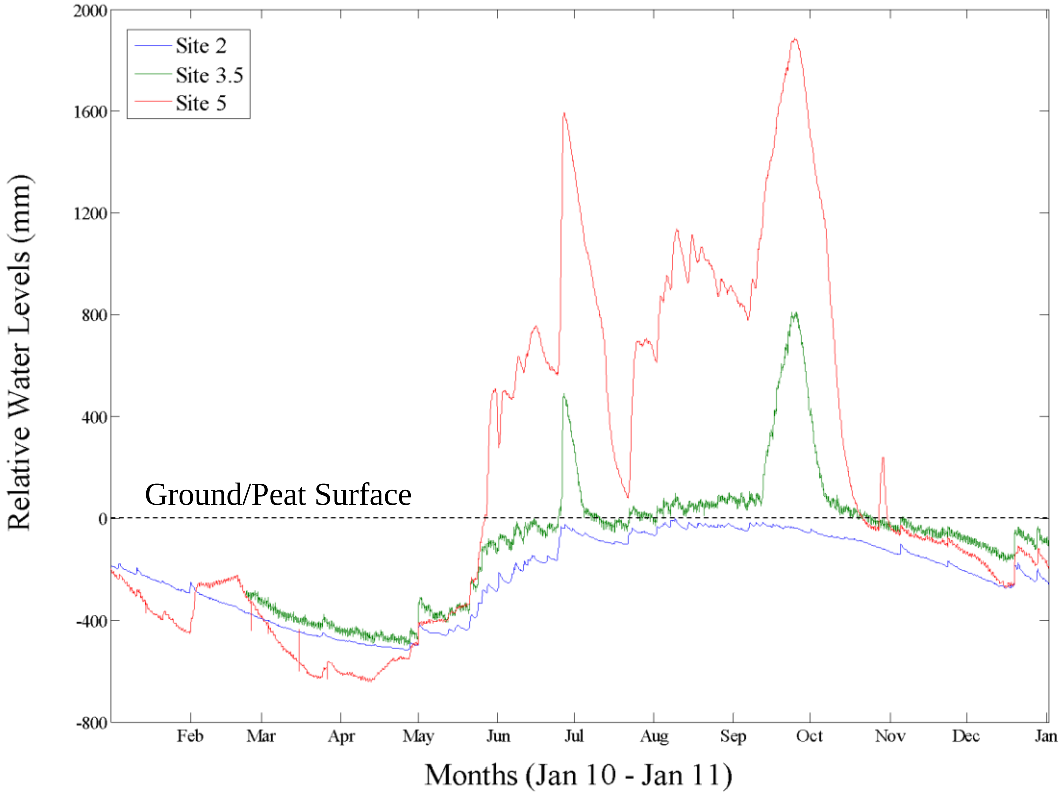


Figure 6.8: Relative water levels for site 2 (restiad bog), site 3. 5 (manuka belt) and site 5 (river marshland) on the Whangamarino transect from 1-Jan-2010 to 31-Dec-2010.

RWL and AWL ranges at water level sites along the transect line increased towards the river. The maximum range (2.45 m) was found at site 6 next to the river, while site 2 had the lowest ranges, 0.51 m and 0.62 m for RWL and AWL respectively (Table 6.1). RWL could not be calculated at sites 1 and 3 due to the likely occurrence of peat oscillation.

Table 6.1: Maximum, minimum and range (m) for RWL and AWL at water level sites along the Whangamarino transect from 1-Jan-2010 to 31-Dec-2010.

	RWL (m)			AWL (m)		
Site	Max	Min	Range	Max	Min	Range
Site 1	-	-	-	5.23	4.56	0.67
Site 2	-0.001	-0.52	0.51	5.18	4.56	0.62
Site 3	-	-	-	4.87	4.19	0.68
Site 3.5	0.81	-0.50	1.31	5.17	3.86	1.31
Site 4	1.37	-0.79	2.16	5.1	2.94	2.16
Site 5	1.89	-0.64	2.53	5.15	2.62	2.53
Site 6	2.3	-0.15	2.45	5.2	2.75	2.45
River	-	-	-	5.38	3.17	2.21

RWL and AWL ranges were identical at sites 3.5–6 as these sites have mineral soil, so oscillation will not occur. Site 2 in the restiad bog had an RWL range smaller than the AWL, because of peat surface oscillation.

6.3.4 September 2010 flood event

A large flood event occurred in late September 2010 and was primarily driven by a number of rain events. Land surfaces in the Waikato Region were saturated from the large amount of rainfall that occurred through August (191 mm). Consistent high rainfall (120 mm from 1-Sep-2010 to 25-Sep-2010) meant water levels in the local rivers and also in the Whangamarino wetland increased significantly (Figure 6.9).

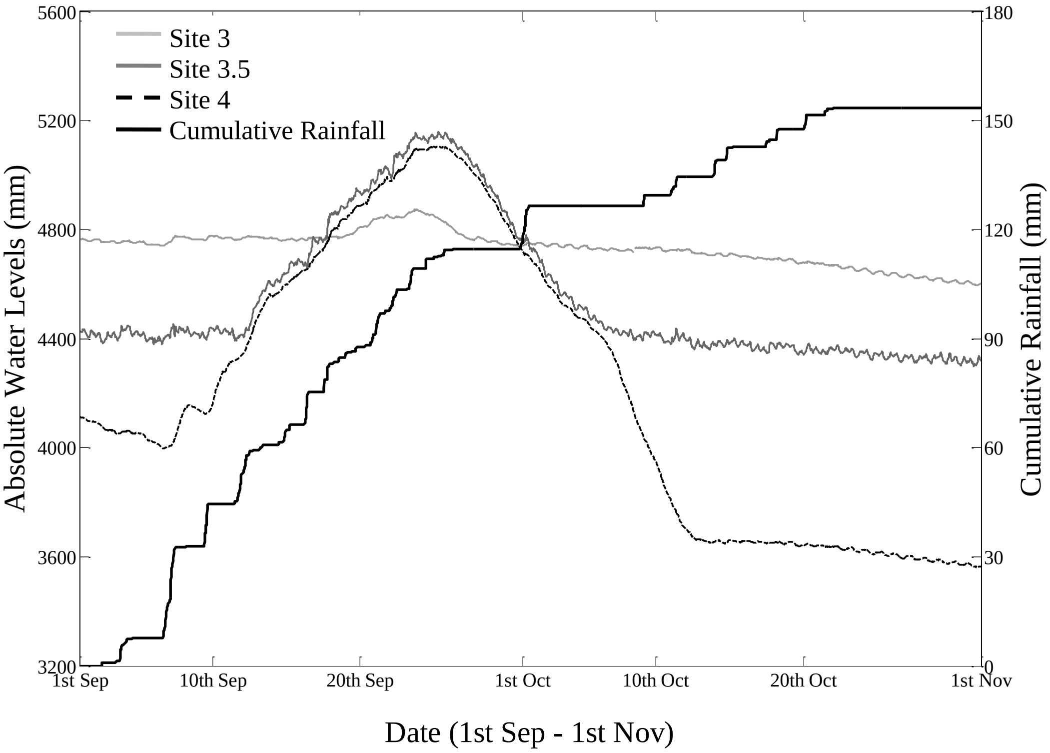


Figure 6.9: Absolute water levels for sites 3, 3.5 and 4 and cumulative rainfall during a flood event from 1-Sep-2010 to 1-Nov-2010.

Water level at site 4 responded to increasing rainfall from 6 September, while sites 3 and 3.5 were relatively unaffected by the consistent (but not heavy) rainfall (Figure 6.9). Site 3.5 responded to flood waters moving into the transect line from 13 September, while site 3 (1.4 km from Whangamarino River) did not respond until the flood peak arrived on 19 September (Figure 6.9). As rainfall rate decreased from 25 September, so did water levels in the Whangamarino River, which resulted in the flood peak subsiding, and on 6 October site 3.5 became independent from the Whangamarino River.

Sites 1 and 2 showed little response to the flood event, while sites 4 – 6 mimicked the river hydrograph, although with an offset in elevation when compared to the Whangamarino River (Figure 6.4).

6.3.5 Peat surface oscillation

Peat surface oscillation (PSO) has previously been described by Fritz *et al.* (2008). Water level measurements to determine PSO were obtained at site 2 from pressure transducers measuring RWL and AWL. Following Fritz *et al.* (2008) the relationship between peat surface elevation (SE) and water level (AWL) was examined (Figure 6.10 & 6.11).

SE responded to AWL, with the maximum SE in winter and minimum in autumn. In summer and autumn, AWL fell far below the surface and the surface elevation response was muted. Frequent rain events in the winter forced AWL close to the surface, but did not overtop the surface. Peat oscillation effectively kept the peat surface above winter water levels (Figure 6.10).

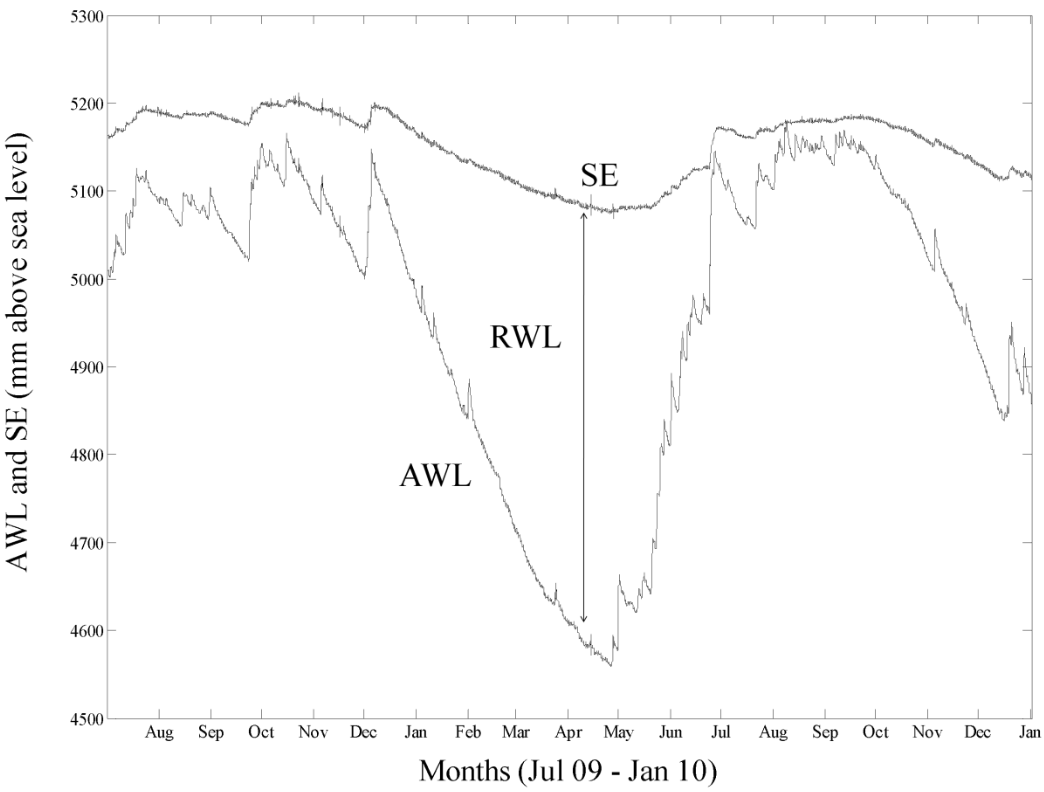


Figure 6.10: Time series of AWL and SE changes at site 2 (meteorological station) on the Whangamarino transect from 1-Jul-2009 to 31-Dec-2010. The zone between SE and AWL is the unsaturated zone or the water table depth (RWL).

The total range of surface elevation was approximately 125 mm from 1-Jan-2010 to 31-Dec-2010 (Figure 6.11).

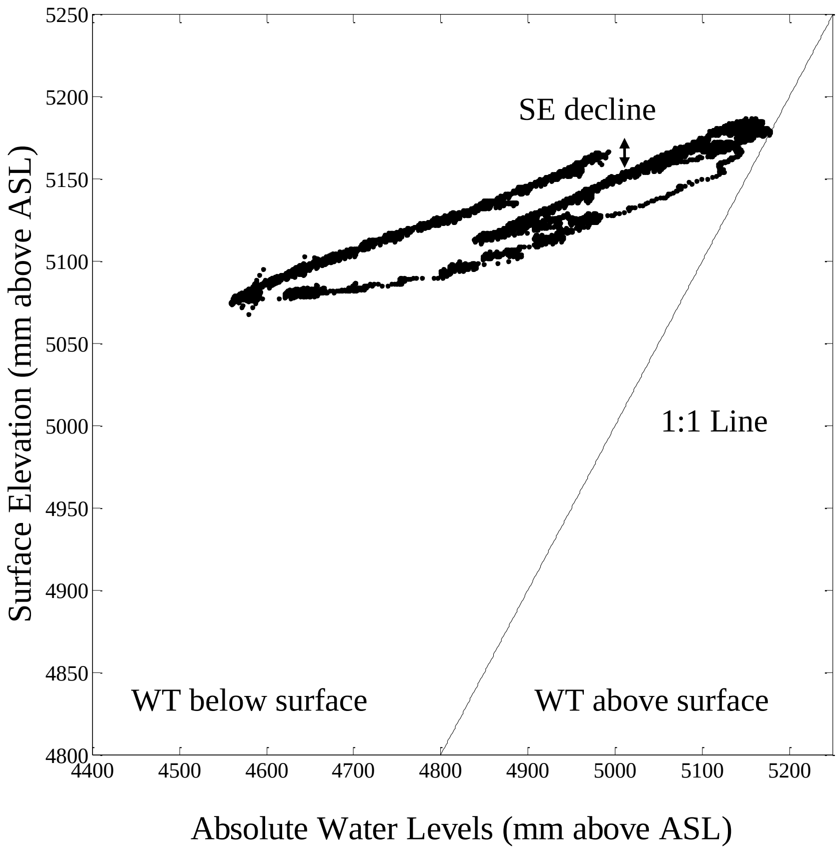


Figure 6.11: Peat surface oscillation at site 2, from 1-Jan-10 to 31-Dec-2010. Also shown is the 1:1 relationship between surface elevation and AWL.

When SE was plotted against AWL, a distinct non linear drying curve was observed (Figure 6.11) during the two summer periods. A rewetting curve and evidence of hysteresis occurred from winter rainfall increasing the water level. An oscillating peat surface responding to water levels should show a decline through summer and an increase through winter, with the pattern overlapping throughout consecutive years. A surface elevation decline was observed from late October-2010 through to January-2011, and may be suggesting a natural or anthropogenic degradation of the peat surface layers.

6.3.6 Historical records of minimum river water levels

Minimum river water levels are of particular importance due to the effect they can have on the wetland. A decline in water levels was observed from the 1960's to the 1990's. Engineering of the weir failed in 1995 and was evident with a subsequent decrease in minimum water levels (Figure 6.12). Following rebuilding and completion of the weir in 2000, minimum water levels at river sites Ropeway and Falls Road increased by >1 m.

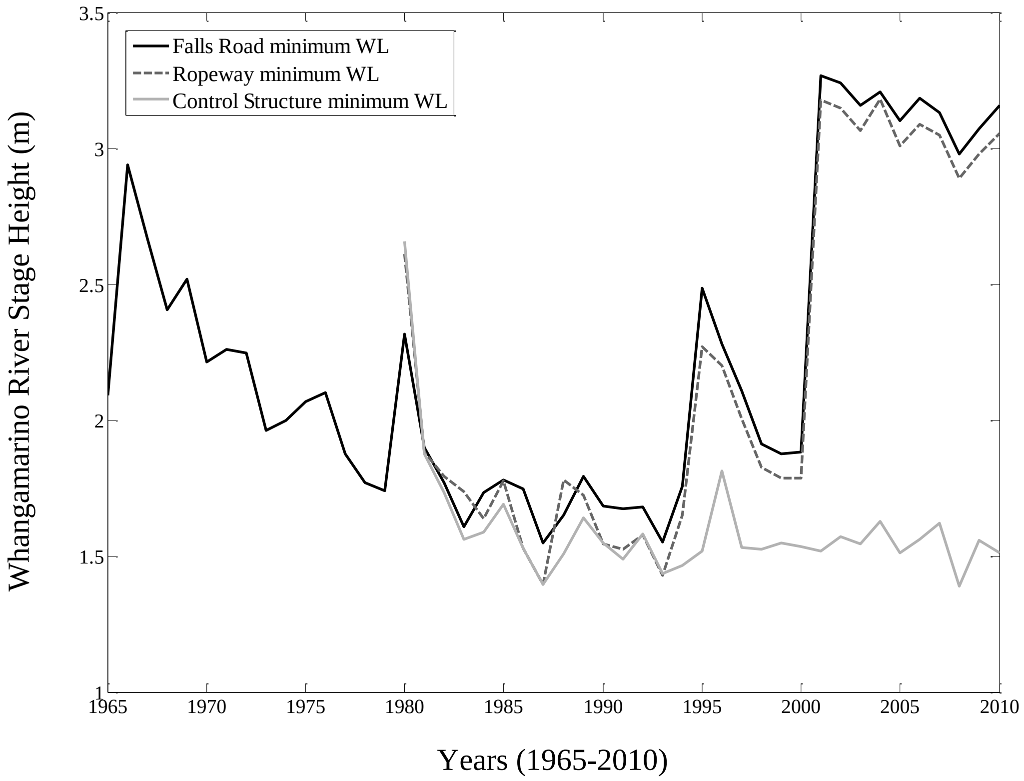


Figure 6.12: Minimum annual water levels for Whangamarino River stage records at Falls Road, Ropeway (upstream of the weir) and the Control Structure (downstream of the weir). Falls Road data set is from 1965-2010 while Ropeway and the Control Structure are from 1980-2010.

6.3.7 Flood return periods

The type III extreme value distribution was used to predict annual flood stage maxima return periods (years) based on Falls Road stage height (Figure 6.13). The Gumbel distribution (type II extreme value distribution) did not predict the return period (or stage) for Whangamarino records accurately enough to be used, as annual flood maximums of Whangamarino River did not lie along a straight line (Figure 6.13).

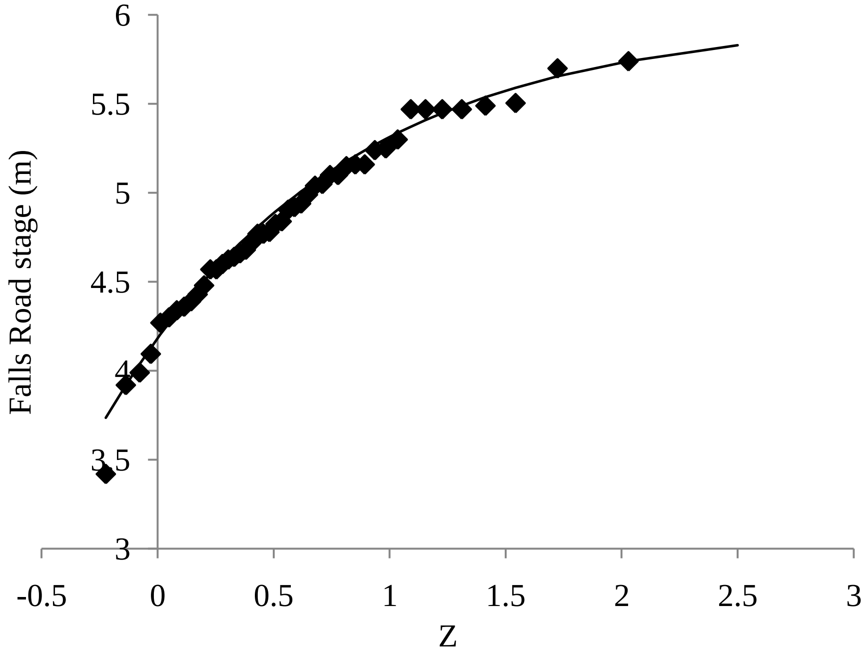


Figure 6.13: Type III extreme value distribution of floods based on Whangamarino River (Falls Road) stage height versus corresponding Z values. Data points indicate the annual maximum flood stage at Falls Road from 1965 to 2010. The trend line is the predicted flood stage based on equations from Bardsley (1989).

A formula was used to convert the Z values derived from Figure 6.13 into return periods following Bardsley (1989). The return periods corresponded to a stage height at Falls Road. Inundation distance of flood events were then identified by plotting water level data from sites 3–5 against Falls Road water level (Figure 6.7). When water levels at each site first responded along a relationship parallel to a 1:1 line, it was assumed that the river had inundated that part of the transect. Following method 6.2.5, river stage was plotted against inundation distances for the four water level sites, a trendline was fitted and extrapolation of inundation distances for various return periods was then undertaken (Figure 6.14).

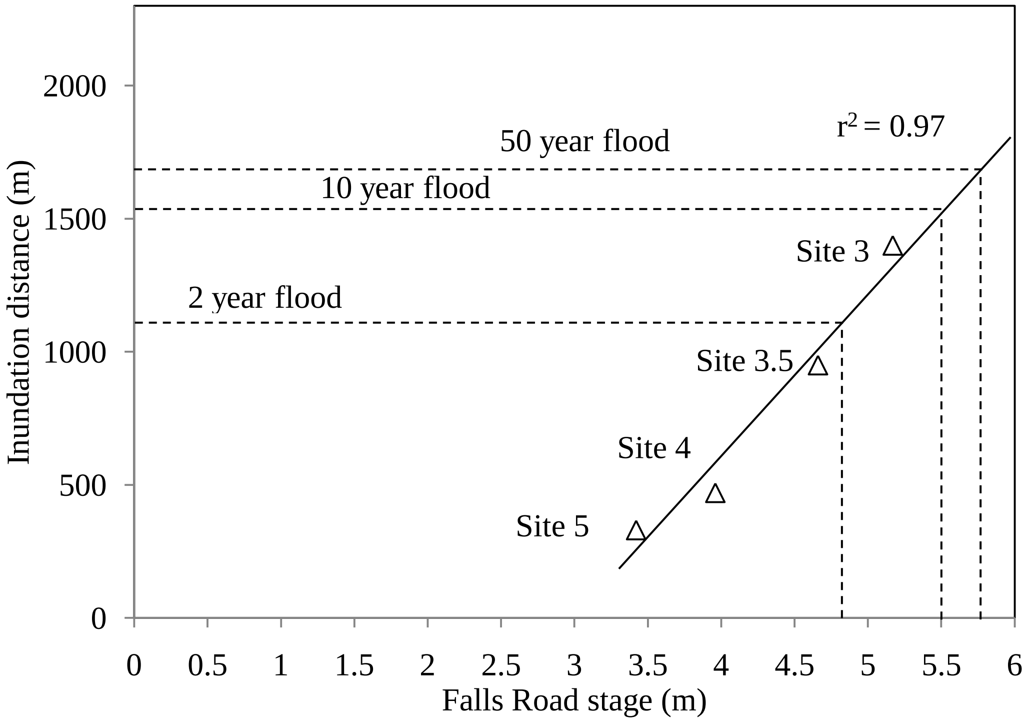


Figure 6.14: Estimation of inundation distances for various stage heights (corresponding to a return period) for Whangamarino River. 2, 10 and 50 year flood events are shown as an example. Extrapolation was from the known inundation distances corresponding to stage heights for water level sites 3–5 (marker points). Refer to method 6.2.5 for a more detailed description.

Transect water level stage was determined similar to Figure 6.14 with the known water level elevation from sites 3–5 when they first responded to a flood, plotted against the falls road stage for the same period. The transect water levels (Table 6.2) were approximately 0.2–0.4 m lower than the Falls Road stage. Transect water levels were also plotted against inundation distance, of which the output was identical to what was derived in Figure 6.14.

Return periods, stage heights and estimated inundation distances derived from Figures 6.13 and 6.14 are shown in Table 6.2. Additionally transect water level stage has been included. A flood event with a return period of 3.3 years would reach 1.4 km into the wetland, while a 100 year flood was estimated to reach only 1.75 km into the wetland.

Table 6.2: Predicted flood return periods and inundation distance along the Whangamarino transect based on stage heights for Falls Road river level site, derived from a type III extreme value distribution. Water level stage along the transect line is also presented. See Figure 6.7, 6.13 and 6.14 for method used to derive stage heights and inundation distances at each site.

Flood Event Return Period (years)	Z	Falls Road Stage (m)	Transect Stage (m)	Inundation distance (m)
Site 5 (Annual)	-0.36	3.42	3.23	330
Site 4 (1.1 yr)	-0.12	3.96	3.67	470
Site 3.5 (1.49 yr)	0.32	4.66	4.43	950
2	0.52	4.91	4.6	1160
Site 3 (3.3 yr)	0.80	5.17	4.79	1400
5	1.01	5.32	4.96	1450
10	1.34	5.5	5.13	1550
20	1.65	5.63	5.22	1640
50	2.06	5.75	5.35	1710
100	2.36	5.81	5.42	1750

6.3.8 Flood water quality

The flood event during August 2010 resulted in a small but distinct flood hydrograph for Whangamarino River, but almost no change in the Pungarehu Canal (Figure 6.15). The reason for the lack of change in the canal was due to the flood gates being opened before the rain event which allowed Lake Waikare to pond a significant portion of the inflow from surrounding catchments and evenly discharge this into the wetland.

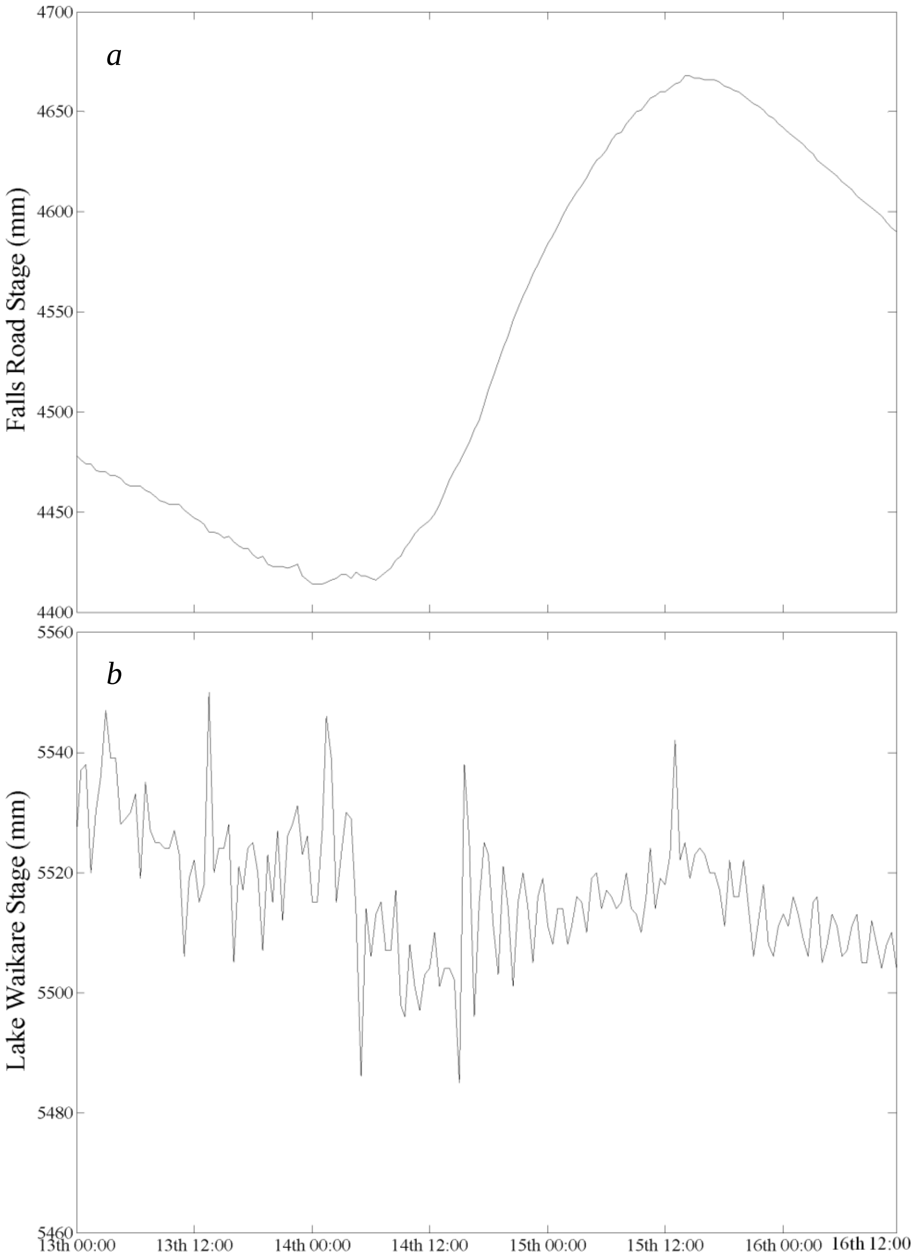


Figure 6.15: Flood stage hydrograph for Falls Road, Whangamarino River (*a*) and Lake Waikare, Pungarehu Canal (*b*) from 00:00 13-Aug-2010 to 12:00 noon 16-Aug-2010.

The time series of total suspended solids (TSS) resembled similar patterns to the flood hydrograph, which would be expected as higher volumes of water should be carrying greater amounts of eroded sediment from river banks and surrounding farmland (Figure 6.15). Higher concentrations of sediment were found in the Whangamarino River, nearly two fold that of Pungarehu Canal.

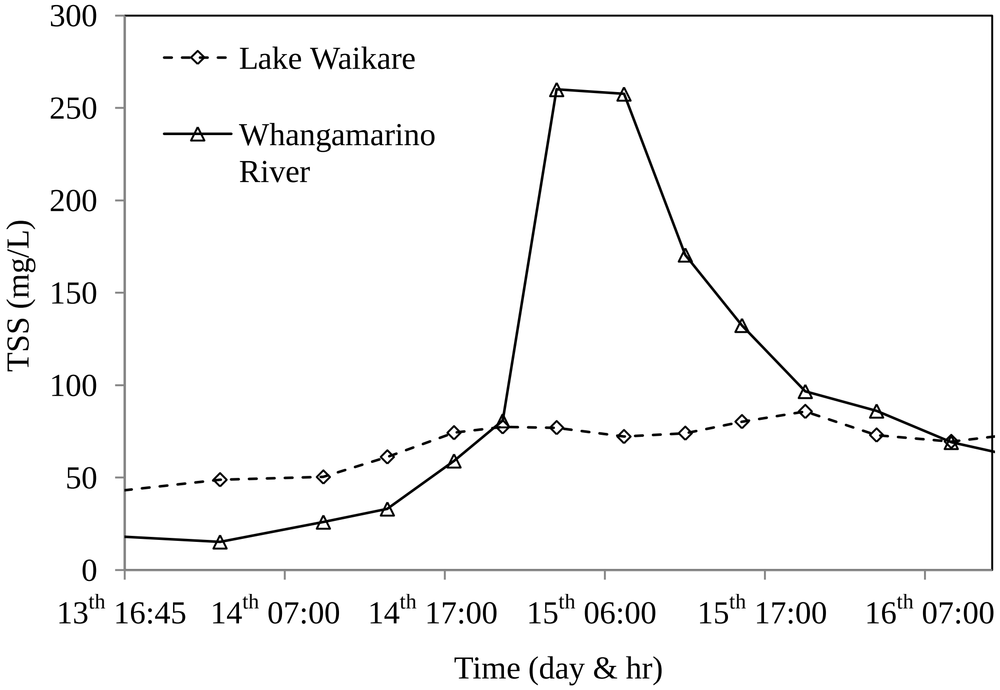


Figure 6.16: Total suspended solids (TSS) for the Whangamarino River and Pungarehu Canal (Lake Waikare) from 4:45 pm 13-Aug-2010 to 11:55 am 16-Aug-2010.

General trends exhibited in the nutrient analysis were an increase in NH_4^+ (ammonium) and DRP in the Whangamarino River as the flood proceeded. Increases in DRP were more pronounced, with the highest concentration observed at the maximum flood peak (Figure 6.17). NO_x increased significantly towards the tail of the hydrograph (as the flood peak decreased), but changed little throughout the majority of the flood event.

In the Pungarehu Canal NH_4^+ and DRP increased towards the tail of flood sampling, while NO_2 and NO_x followed a pattern similar to total suspended solids (Figure 6.15 and Figure 6.18).

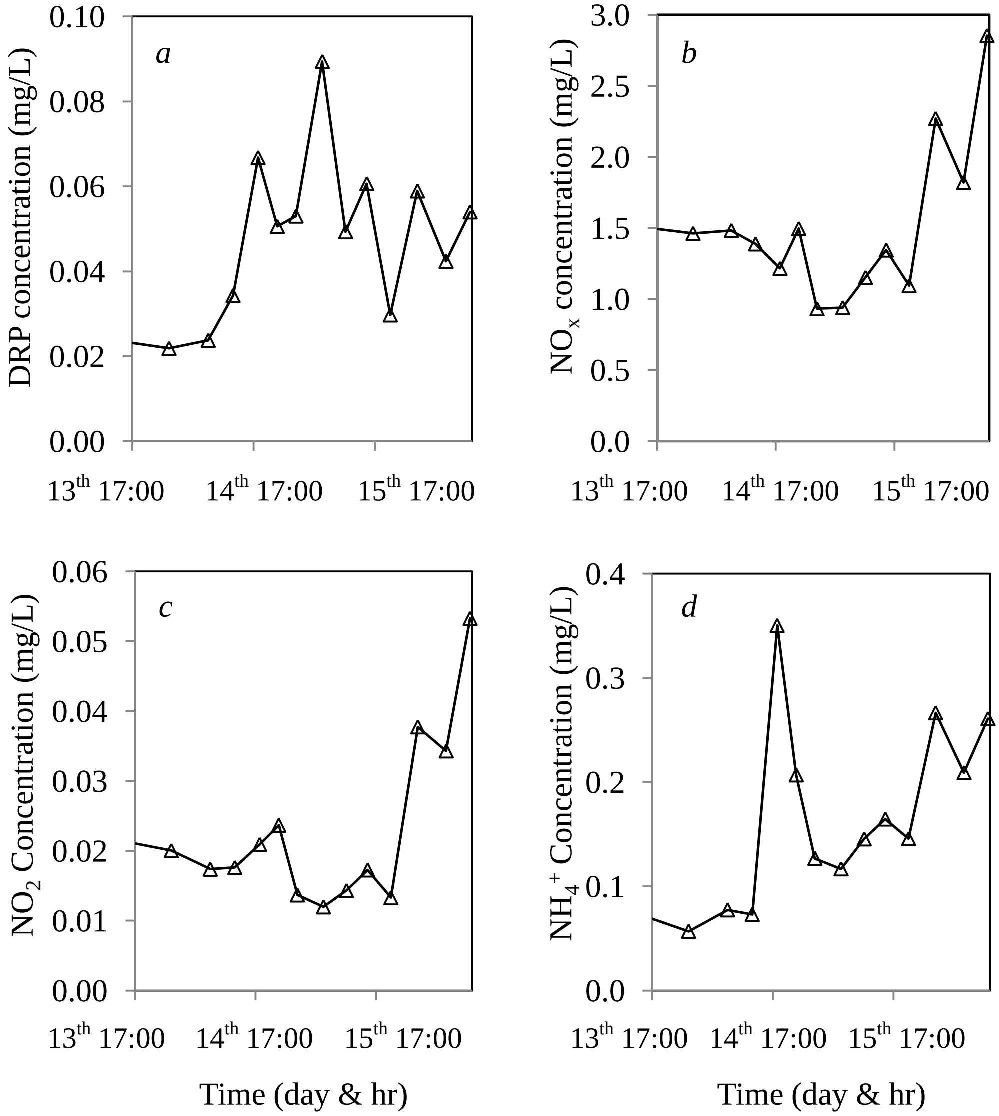


Figure 6.17: (*a*) DRP, (*b*) NO_x, (*c*) NO_2 and (*d*) NH_4^+ concentration (mg/L) in the Whangamarino River from 4:45 pm 13-Aug-2010 to 11:55 am 16-Aug-2010.

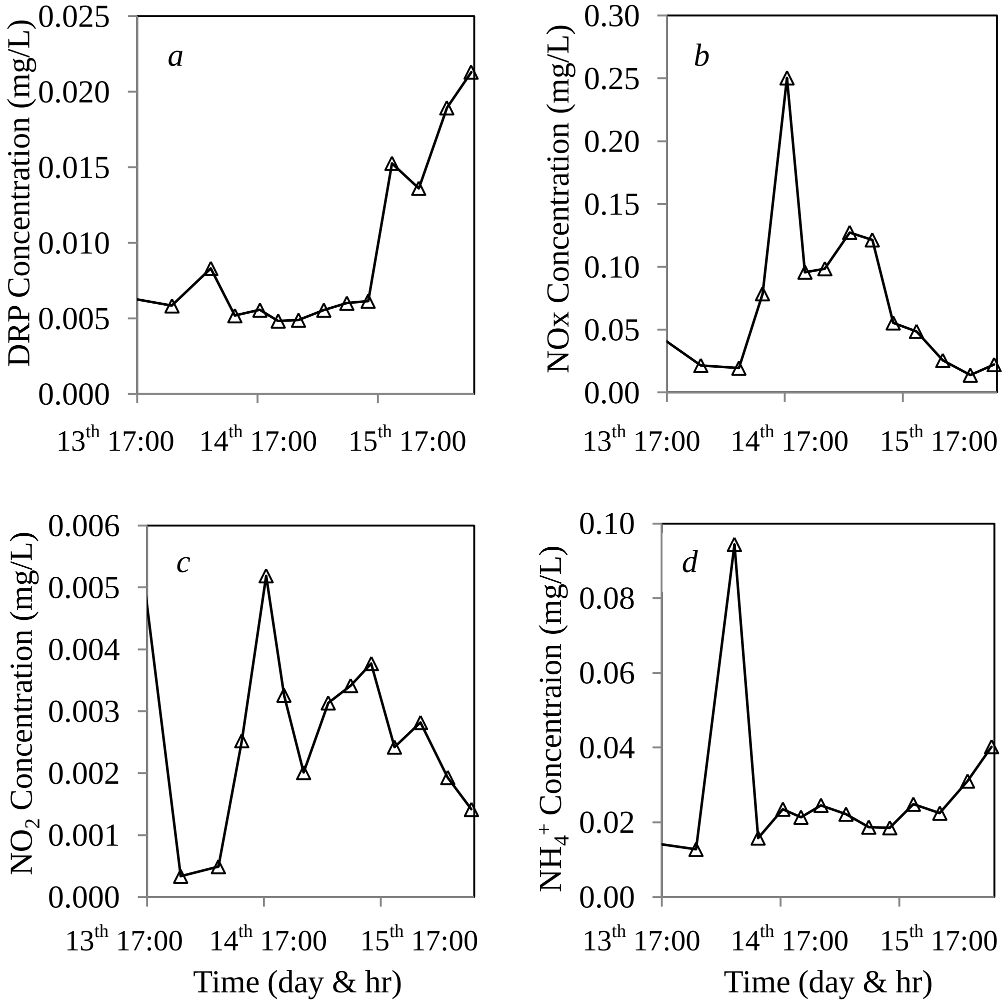


Figure 6.18: (*a*) DRP, (*b*) NO_x, (*c*) NO_2 and (*d*) NH_4^+ concentration (mg/L) in the Pungarehu Canal from 4:30 pm 13-Aug-2010 to 11:20 am 16-Aug-2010.

6.4 Discussion

Hydrological patterns along the Whangamarino transect line give an insight into the processes operating in the wetland and how these may influence vegetation composition, soil and peat development and overall wetland condition (discussed in Chapter 7).

6.4.1 Water level patterns

Rainfall was the predominant driver of water level change at sites 1–3. A dry summer and autumn from January–April 2010 (Figure 6.1) resulted in a declining water level at all monitoring sites (Figure 6.4, 6.5 and 6.6). The slow recession of water levels was similar at sites 1–3 after rainfall events (Figure 6.5 and 6.8). This was likely due to local controls at these sites, such as low hydraulic conductivity in peat layers deeper than 10 cm holding water through fluvic and humic acids (Flaig 1986; Devito *et al.* 1996) (Chapter 2). Limited rainfall events from January–April led to small increases in water levels at sites 1–3.5, while greater increases were observed at sites 4–6 (closer to the Whangamarino River). A stable water table at sites 1–3 indicated this portion of the transect (50–900 m) could potentially be a restiad bog (Campbell & Jackson 2004; Johnson & Gerbeaux 2004). Site 3 is evidently affected by river flood events > 3.3 yr return period. Sites 1 and 2 had the lowest range in absolute water levels (0.62–0.68 m) over the year (Table 6.1) and at site 2 the water level never exceed the surface elevation, which supports the principal of stable water levels in restiad bogs (Campbell & Jackson 2004).

Site 3.5 (1350 m from farmland) was independent from the river hydrological influence for around 60% of the year with the water level responding to rainfall events only, similar to sites 1–3 (Figure 6.7). During this period, site 3.5 exhibited a stable, slow decline in water levels consistent at sites 1–3 (Figure 6.6 & 6.8). Flood events during winter reached into this manuka belt on regular occasions, and controlled the water level directly for the remaining 40% of the year (Figure 6.7). The 1.31 m range in absolute water level at site 3.5 over the year was double the range at site 3 (450 m away) (Table 6.1). This difference is attributed to the increased flood inundation. After flood events, a rapid decline in water levels was observed as the flood peaks resided, although water levels were still above the peat surface (by around 0.1 m) for the remainder of the winter (Figure 6.8). This suggested the local controls on site 3.5 are similar to sites 1–3 during dry periods with little surface flooding, while over winter the Whangamarino River is the primary control on the manuka belt water levels. Even after large flood events have resided (with the primary movement of water occurring through surface overland flow), high water levels maintained in the Whangamarino River sustain water levels above the surface at site 3.5.

Near the river, sites 4, 5 and 6 (from 1850–2300 m) had water levels increasingly controlled by the Whangamarino River (Figure 6.4), with some independence during dry periods when river water levels declined and rainfall became the sole driver (Figure 6.6 and 6.7). Relative water level ranges (Figure 6.8 and Table 6.1) increased closer to the river, where site 5 was the most dynamic, with the greatest range in elevation (2.53 m) over the year (Table 6.1). The water level decline after flood events at sites 4–6 was rapid, where surface overland flow was likely the primary cause for rapid decline. Throughout the year sites 4 and 5 were controlled directly by the river for around 70% and 80% of the time respectively (Figure 6.7). During summer, when river water levels were low, sites 4 and 5 and 6 were independent and likely controlled by local processes such as rainfall (increasing water levels) and lateral drainage of groundwater towards the river.

At Bullock Creek wetland, Sorrell *et al.* (2007) found areas of *Sphagnum* peat accumulation in a fen were characterised by stable water tables and were independent from river flood events. Sites closer to the river and with a dominance of *Carex* and non-native plant species were subjected to periodic flooding and had a dynamic water table, experiencing brief periods of deep flooding and extreme drying. These characteristics are similar to what occurs in water levels along the Whangamarino transect.

The change in water level elevation along the transect was quite distinct. At site 1 (50 m) and site 2 (250 m), elevation was similar (around 4.56 m a.s.l during the lowest water level recordings in 2010). At site 3.5 (1350 m), elevation declined to around 3.86 m a.s.l (a vertical elevation change of 0.7 m over 1300 m). Finally at site 4 (1850 m from farmland) and 5 (1970 m from farmland) minimum water level elevations had decreased to 2.94 m and 2.62 m respectively (Table 6.1). This results in a change in vertical elevation of water levels by approximately 2.0 m over a horizontal distance of 2000 m (from farmland to the Whangamarino River).

6.4.2 Flood events and water quality

Numerous flood events occurred throughout winter in 2010. Two significant events were in June and late September. Flooding resulted from rain events ranging from quick moving fronts causing rapid but short lived rainfall, to sustained periods of rainfall that led to a slow rise of water levels. The September flood event (Figure 6.9) was caused by the latter pattern. This event had a maximum water level of 5.38 m at Falls Road (Whangamarino River) (Figure 6.4 and Table 6.1).

The September flood inundated five of the seven sites (3–6) (Figure 6.7 and 6.9). At the flood's maximum peak, water reached into site 3 (1400 m from the river) and overtopped the surface (Figure 6.7). The technique used to identify the water level sites response to a flood event (parallel to a 1:1 line) had been used in a wetland study by Browne & Campbell (2005) when describing the Opuatia River water level response compared to the Waikato River.

After the rainfall ceased (25 September), flood water levels declined (Figure 6.6). During the period 1 June to the 1 October site 4 (450 m from the river) was inundated by flood water for around 90% of this period (108 days). Site 3.5 (950 m from the river) was inundated approximately 50% of the period (60 days), while site 3 (1400 m from the river) was under flood water for around 10% of the time (12 days).

The difference in water level elevation from the Whangamarino River at Falls Road to the Ropeway site (12.3 km downstream) during flood flows averaged 0.357 m (from 10 years of maximum flow data). This equates to a difference of 38 mm in water level elevation from Falls Road to site 6 during flood flows (1.3 km downstream). The offset between the Whangamarino River and site 6 during flood peaks (Figure 6.4, Table 6.2) was on average greater than 30 mm, with the 2010 September flood having a difference of up to 180 mm (Table 6.1). This may be due to the Whangamarino River water levels being recorded upstream of the bridge at Falls Road which was not part of the wetland floodplain. This water level site was constrained by stop banks and hence the river level will be higher with a greater flow. As it joins the wetland, the stop banks end and the river can flood across the wetland, resulting in a reduced elevation of flood peaks at transect water level sites. While the flood peak is reduced by 180 mm (September 2010) from Falls Road to site 6, the spreading of the Whangamarino River exacerbates flood inundation distance, i.e. it reaches further into the wetland than if it was constrained by channels (which is the primary design for flood storage).

Flood events were analysed using probability distributions, to provide estimates of the return period (years) for a particular event size (based on stage) (Figure 6.13 and Table 6.2). Accuracy was determined based on the designed '100 year' flood event which could be contained within the wetland by the Whangamarino control gate, which has an RL of 5.85 m (Waugh 2007). When the predicted stage height did not fall close to the 100 year flood design (expected stage of 5.85 m) then a type III extreme value distribution was adopted. The type III distribution predicted a 100 year flood event would have a stage of 5.81 m at Falls Road (Table 6.2).

The reason for the inadequacy of the Gumbel distribution was due to attenuation of flood peaks within the wetland. Generally as river stage or discharge increases, so does the return period of each particular flood event. As the Whangamarino River floods across a large area into the wetland and is prevented from discharging into the Waikato River by a control gate, stage and return period increase linearly until a point. This point occurs when the wetland acts as a storage area similar to a lake, where larger flood events will increase greatly in horizontal inundation with only a small change in vertical stage. Hence the dataset fell away from the linear relationship with larger flood events (and also very small floods), which is why the type III distribution was adopted as it is designed for events that have an upper limit or maximum (i.e. a tailing off at the most extreme limits) (Pearson & Henderson 2004).

Due to a lack of annual flood maximums at the lowest stage heights (3.5–4.0 m), the type III distribution had reduced accuracy when predicting return periods for small flood events. Sites close to the river (i.e. sites 4, 5 and 6) that were regularly inundated by small flood events therefore had reduced accuracy when predicting return periods. For example, site 4 and 5 were predicted to be inundated every 1 and 1.1 yrs, when it is likely more frequent than this. A large number of maximum annual flood events over 4.2 m ensured the type III distribution can predict return periods for higher stages (>4.2 m) with greater accuracy.

The September 2010 flood event had a return period of 3.3 years (inundating 1400 m from the river), while an annual flood event would inundate at least 330 m of the wetland (Figure 6.14 and Table 6.2). The largest flood capable of being contained by the wetland is a 100 year event (Mullholland 1991; Waugh 2007), and was estimated to inundate approximately 1750 m into the wetland (Table 6.2 & Figure 6.18). This would still leave 0–550 m of the transect free from inundation.

The wetland water level elevations derived for various flood events appear to be slightly greater than what is expected (Table 6.2). Figure 6.10 showed the peat surface at site 2 did not exceed 5.2 m over an 18 month period. Yet a 100 year flood had predicted wetland water levels of 5.42 m, which would therefore cover lead to inundation of 2300 m, rather than 1750 m. The cause for this error is likely in the extrapolation technique, where we have no data to predict water level stage in the restiad bog as a response to a large flood. Hence we are estimating the response of water levels (and inundation distance) into the restiad bog based on the known inundation of a smaller flood event through the manuka zone. Potentially local controls such as peat hydraulics, a raised surface elevation (that would require surveying) or even PSO may influence the restiad bogs response to a flood event and therefore reduce the accuracy of this inundation estimate. Surveying of the restiad bog surface along the transect line during winter, when the peat is swelling as a response to PSO, would be useful to investigate the hypothesis that the bog area restrains floods from inundating the entire transect.

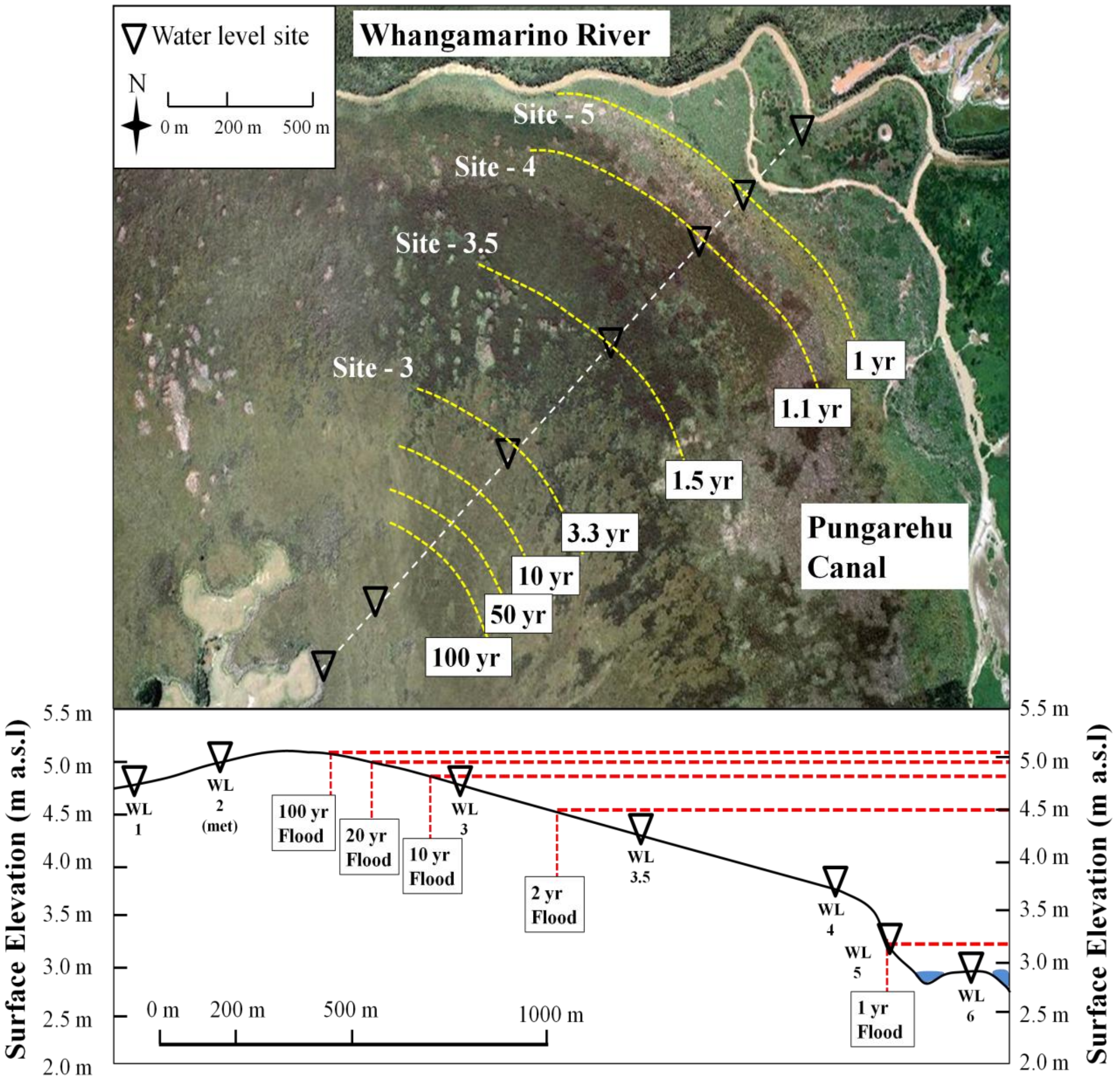


Figure 6.19: Aerial image and cross section (based on surveyed surface elevations) of predicted flood inundation return periods (years) along the Whangamarino transect line (from 0–2300 m).

Bogs generally form a raised dome of peat in the centre of a wetland (Johnson & Gerbeaux 2004; Campbell & Jackson 2004). Hence it is likely surface elevations are highest around 500 m from the farmland, and aid in stopping flood peaks from 100 year events inundating the entire wetland. The slight crest in surface elevations indicated in the cross section of Figure 6.19 is therefore hypothetical and would need to be substantiated through RTK-GPS ground surveying.

Water quality samples obtained during the small flood event in August 2010 indicated that high TSS loads come through the Whangamarino River and follow the pattern of a hydrograph, with highest concentrations of 260 mg L^{-1} found at the peak of the flood (Figure 6.14 & 6.15). These loads were over twice that of flows from Lake Waikare (maximum of 86 mg L^{-1}). This supported the initial findings by Gibbs (2009) who found sediment from Lake Waikare made up 14% of cores taken near sites 5 and 6, with the remainder coming from the Whangamarino River. Lake Waikare could contribute significant sediment loads as it is highly turbid, but to prove this is correct, flood sampling would need to be undertaken during pronounced hydrographs in the Pungarehu Canal. Dissolved nutrient patterns had some similarities to an increase in sediment load and water volume throughout the hydrograph, but were not entirely consistent (Figure 6.16 & 6.17). The higher dissolved nutrients in flood waters will ultimately enter the wetland transect line, and during larger flood events these nutrients will inundate further.

6.4.3 Minimum water levels

The decline in minimum water levels from 1965 to 2000 is due to sand abstraction and river training in the Waikato River (Reeves 1994). The weir was installed in 1993, but failed a few years later. When the weir was repaired in 2000 (to a level of 2.95 m) water levels increased by approximately 1.1 m (Department of Conservation 2007). Since 2000, minimum water levels have again declined due to erosion of the rock wall, and repairs in 2010 are set to bring minimum water levels back up towards the target of 3.0–3.4 m (Department of Conservation 2010c).

According to Reeves (1994) lowering of minimum water levels (coupled with increased nutrient input from the flood control regime) led to an invasion of manuka into the wetland, covering approximately 500 ha. Also, Shearer (1997) suggested these low water levels from 1965–2000 consequently caused drying and increased decomposition of peat adjacent to the rivers.

6.4.4 Peat surface oscillation

At site 2, the peat surface was found to oscillate by 0.125 m during 2010 (Figure 6.11). This PSO is similar to that measured by Fritz *et al.* (2008) in Opuatia wetland, who found that PSO ranged from 0.032 m to 0.28 m across 23 sites, with an average of 0.149 m. Hysteresis in the wetting and drying response of the peat surface elevation at Whangamarino was also observed by Fritz *et al.* (2008). This was observed when the surface level decreased during summer (through drying) and increased through winter (re wetting). The drying phase was distinct, where the unsaturated zone (RWL) reached a maximum depth of 0.52 m below the peat surface in late April 2010 (Figure 6.10 and Table 6.1). The rewetting phase can be observed through July to October where water levels nearly overtopped the surface (Figure 6.10 and 6.11).

Fritz *et al.* (2008) found PSO was controlled by the AWL, and reduced fluctuations of the water table below the surface (RWL) up to 30–50 %. Low RWL fluctuation in peatlands is an important control on plant composition, many species of which are specially adapted for a stable, low nutrient environment. Flooding increases RWL fluctuations, which could result in a possible change in plant species composition and restoration of peatlands being impeded.

There appeared to be a decline in surface elevation of peat from January 2010 to the January 2011(Figure 6.11). This may indicate peat degradation and enhanced decomposition, but requires more research and a longer data series to observe patterns.

6.5 Summary

- Water levels became increasingly dynamic closer to the Whangamarino River, with larger fluctuations in water level ranges caused by flood inundation.

- At sites 1–3 (50–900 m), water levels were relatively stable throughout the year, fed primarily by rainfall. The large September flood event reached into site 3. Water level ranges in these sites were less than 0.7 m over the year and the conditions indicated this zone is likely best classed as a restiad bog.

- Site 3.5 (manuka zone, 1350 m) exhibited water levels controlled by both rainfall (throughout summer and autumn) and the Whangamarino River (through winter flood events). Water level ranges doubled from site 3, to 1.38 m, and for 40% of the year water levels were controlled by river flooding.

- Water levels at sites 4–6 (1850–2300) were increasingly dynamic closer to the Whangamarino River. For a large part of the year (up to 80% at site 5), water level was controlled directly by the river level. Fluctuations in water levels ranged from 2.16–2.53 m over the year. These hydrological conditions are characteristic of swamp or marshland environments.

- The September flood event caused the Whangamarino River at Falls Road to reach a stage height of 5.38 m. This flood inundated 1.4 km into the wetland, encroaching on the edge of the restiad bog.

- Flood nutrient sampling showed TSS and dissolved nutrients (such as phosphorus) increased relative to the flood hydrograph. TSS peaked at 260 mg/L in Whangamarino River, over twice that of Pungarehu Canal (86 mg/L).

- Minimum water levels showed the installed weir has increased Whangamarino River water levels by >1 m which Reeves (1994) suggested was the cause for the encroachment of manuka into restiad bog.

- Through flood frequency analysis, inundation distances along the transect relative to Falls Road stage (Whangamarino River) were derived. Small floods (likely more than 1 a year) will inundate sites 4–6 (450 m from the river). A 1.5 yr flood will reach into site 3.5 (950 m from the river), while a 3.3 yr flood will reach into the restiad bog at site 3 (1400 m from the river). Due to the increase in surface elevation from the river to the restiad bog (of over 2.0 m), a 100 yr flood was estimated to inundate 1,750 m from the river and not cover the wetland.

- Peat surface oscillation at site 2 was 0.125 m over the year. RWL reached a mximum of 520 mm below the surface during summer, but due to PSO throughout winter, water levels never overtopped the surface. There was some indication of the peat surface declining by around 25 mm from January 2010 to January 2011 (not related to instrument drift), but the cause of this requires more research.

- Flood driven imports of nutrients and sediment are likely the main cause for higher nutrient concentrations in peat and foliage measurements.

Chapter 7: Discussion, recommendations and conclusions

7.1 Introduction

This study of Whangamarino wetland involved looking at peat or soil properties and vegetation composition along a hydrological gradient. These two significant components are primarily controlled by the hydrological processes operating in the wetland, which was also studied through a variety of approaches. The ecohydrological characterisation of Whangamarino requires linking these three major components together to gain an overall understanding of wetland condition and the various processes which have led to its present status.

7.2 Wetland classes

Chapter 4 described the peat and soil characteristics along the Whangamarino transect line. Coupled with vegetation composition and hydrological patterns at different zones on the transect, wetland class can be identified by comparing peat and soil chemistry with other New Zealand studies. Table 7.1 recreates Table 2.3, with the addition of Whangamarino wetland measurements for the zones identified as restiad bog (50–1100 m along the transect) and swampland (0–50 m and 1500–1900 m). The area from 1900–2300 m was described as a marshland, due to the significant mineral inputs and vegetation species present in this location, but for the purposes of Table 7.1 the sample sites from this zone has been included as swampland.

Swampland was identified from 0–50 m (2 sites) in a marginalised wetland fringe belt, dominated by invasive *S. cinerea* and species such as *B. frondosa*. Nutrient concentrations in this zone were consistent with those found across New Zealand for swamps (Table 7.1). Other characteristics such as relatively high levels of peat decomposition (von Post 6–7) and invasive plant species were consistent with classifications by Johnson & Gerbeaux (2004) (Table 2.1). The main factor influencing this zone was likely farming directly next to the wetland, increasing nutrient and sediment inputs through groundwater, surface runoff, atmospheric ammonia deposition (Appendix C) or directly via cattle entering the fringe.

Table 7.1: Means and ranges (in brackets) for soil parameters at 17 swamps and six bogs sampled in New Zealand (Clarkson *et al.* 2004b), 6 swamp and 22 bog sites in Opuatia wetland (Browne & Campbell 2005) and 11 swamp and 12 bog sites in Whangamarino wetland. TN= total nitrogen, TC= total carbon, TP= total phosphorus.

	New Zealand		Opuatia		Whangamarino	
	Bogs	Swamps	Bog	Swamp	Bog	Swamp
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 cm^{-3})	92.7 (24.1–239.8)	39.8 (5.2–100.6)	33.3 (24.2–43.29)	37.8 (29.8–47.4)	26.94 (18.1–41.4)	36.5 (20.0–52.6)
TN (mg cm^{-3})	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 cm^{-3})	0.08 (0.01–0.20)	0.28 (0.15–0.59)	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)

From 50–1100 m (12 sites) the dominant wetland class was restiad bog. The main vegetation throughout this zone was *E. minus*, *G. dicarpa* and *Baumea spp.* with small patches of *L. scoparium*. The vegetation types found were consistent with other bogs in the Waikato region, as discussed by Browne (2005) and Clarkson *et al.* (2004a). The chemical measurements taken from peat samples in this zone were similar to findings by Clarkson *et al.* (2004b) (Table 7.1). Average pH and TN were very similar to New Zealand bogs, while TC was lower than average but fell within the New Zealand ranges. Phosphorus appeared to be at low levels throughout the restiad bog, with averages on the very low side for New Zealand conditions. C:N and C:P ratios were similar to New Zealand bogs, although C:N was lower due to the low TC concentrations. While TC and TP are both relatively low in Whangamarino, their ratio (C:P) appears similar to other New Zealand bogs. Low TP concentrations were expressed through the N:P ratio, where a high N:P ratio suggested the restiad bog was limited by P. The presence of peat and a low-moderate decomposition state (von Post 2–5), coupled with surveyed surface elevations showed this area was the highest point in the wetland indicating a raised peat bog. This zone had a stable, high water table fed primarily by rainfall which was consistent with bog formation (Campbell & Jackson 2004; Johnson & Gerbeaux 2004). Flood events with a return period of greater than 5 years may bring surface water and nutrients into this area but rainwater is the dominant hydrological control.

From 1100–1800 m a manuka transition zone with low to intermediate nutrient levels was present. The zone ended at 1800 m where the most dramatic change in nutrient and metal concentrations occurred, along with a canopy change to *C. tenuicaulis*. Due to a lack of groundwater inputs the manuka zone cannot be strictly classified as a fen, although would be consistent with New Zealand swampland characteristics closer to the river (Johnson & Gerbeaux 2004). The start of this transition zone began when *L. scoparium* dominated the canopy cover, up to 100% in most cases (1100 m). Understory vegetation changed from *E. minus* and *G. dicarpa* near the restiad bog (1100–1500 m) to swampland species such as *P. tenax*, *C. australis* and *C. tenuicaulis* closer to the river. Nutrients, peat degradation (von Post 7–8) and pH (4–6.4) from 1500 m onwards fell within similar ranges for New Zealand swamps, which led to the area from 1500–2300 m being classified as swampland (Table 7.1). From the restiad bog/manuka fringe, nutrients increased towards the river, due to regular hydrological inputs from surface water flooding, including inundation from a flood event in September 2010 that reached 1.4 km into the wetland and had a return period of 3.3 years (Chapter 6). The water table in this zone was increasingly dynamic closer to the river, which is consistent with swamp conditions (Table 2.1, Johnson & Gerbeaux 2004). N:P ratios in peat and foliage indicated the swampland (from 1500 m) was also limited by phosphorus, with higher than average N:P ratios when compared to other New Zealand swamps (Figure 5.6). From 1100–1500 m was classified as a transition zone between bog and swamp, and possibly could be classed as a fen if groundwater inputs were significant (which is unlikely).

Closer to the river (1800–2300 m) swampland conditions were dominant, with the highest levels of nutrients, pH and mineral inputs. The inability to perform a von Post assessment (from 1900 m) due to the lack of peat suggested high rates of organic matter degradation with no peat accumulation. The hydrological regime was strongly driven by the Whangamarino River and consisted of a dynamic water table, which fluctuated over 2.0 m throughout 2010, with long periods of continuous inundation throughout winter. Regular flood events deposited sediment and nutrients, where a large portion appeared to be deposited within a dense fringe belt of *C. tenuicaulis* that was possibly acting as a buffer zone (stripping out the most sediment and nutrients from 1800–1950 m). The dominant species present were grasses and adventives, such as *P. persicaria* and *P. arundinacea* that rarely exceeded 1.0 m in height. *P. persicaria* (willow weed) is an invasive weed able to colonise in moist and wet areas and often outcompetes other plants due to its sprawling nature. This was the case in the field, where the final 300 m of the transect was dominated entirely by willow weed, with a 100% canopy cover in most situations. Willow weed is a summer annual (colonising from rootstock), which is likely the reason it inhabits this area close to the river that is flooded most of winter (Massey University 2010). The high levels of nutrients, a dynamic and fluctuating water table, mineral soil and presence of weeds and grasses indicate from 1900–2300 m could likely be classified as marshland (Table 2.1, Johnson & Gerbeaux 2004).

7.3 Ecohydrology of Whangamarino wetland

The wetland classes along the transect in the southern portion of Whangamarino are formed and maintained through different hydrological processes.

The restiad bog formed through plant species adapted to living in areas with rainwater as the sole input, indicated by stable water levels throughout the year. The manuka zone was inundated periodically by surface water flooding which brought in nutrients, and ultimately changed the vegetation composition. A greater fluctuation in water levels closer to the river led to enhanced peat degradation. Sediment deposited from flood waters increased from 1100 m and peaked in the marshland, resulting in a mineral soil with high pH, nutrients and the presence of a large number of invasive plants.

Overall there is a change in vegetation and peat characteristics along a hydrological gradient. This is consistent with other wetland ecohydrological studies, where hydrological processes have been found to be the main cause for vegetation and nutrient change. Sorrell *et al.* (2007) found species–environment relationships were strongly correlated with soil water content and aeration (oxidation), as well as soil nutrient content and bulk density. Nutrients increased and vegetation composition changed to invasives towards locations with higher flood inundation and water table variation. Similar results relating to bulk density, moisture content and nutrient availability were found in a study of 16 Minnesota wetlands by Bridgham *et al.* (1998). They studied the rates of carbon, nitrogen and phosphorus mineralisation across an ombrotophic (bog) – minerotophic (swamp) gradient. On a volumetric basis, C and N mineralization increased in a predictable manner across the gradient towards swampland with greater water level fluctuations. Ombrotrophic peatlands received only atmospheric nutrient inputs and had inherently low nutrient input rates compared to more hydrologically open (minerotrophic) sites. These findings are supported by Venterink *et al.* (2002) who found shifts in vegetation species along gradients in 44 different Northern Hemisphere wetlands were correlated to the availabilities of soil N, P and K brought in through increasing surface water and groundwater inputs.

A similar study to the one in Whangamarino was undertaken by Trepel and Kluge (2002) of the Eider Valley peatland in Northern Germany (Chapter 2). They found drainage and intensifying land use in an upland wetland catchment had resulted in increased eutrophication and colonisation of invasive plants species (in the wetland), particularly closer to the river. Reduced flooding had led to a drying of the wetland and reduction in peat and bog vegetation species. The change in species composition at Whangamarino is likely due to a similar cause, increased nutrient and sediment inputs from upper catchments.

Browne (2005) undertook an ecohydrological study of Opuatia wetland and found it was at risk from *S. cinerea* (grey willow), which had invaded the wetland margins over time. Nutrient runoff from surrounding agricultural land was also a cause for concern, with higher concentrations having entered the wetland and promoting weed invasion and degradation of wetland health. An integrative ecohydrological study was also undertaken by Eser & Rosen (1999) and focussed on Stump Bay wetland, part of the South Taupo wetland in New Zealand. The wetland was identified as being a zone of groundwater discharge which contributed to nutrient increases and changes in pH and conductivity that ultimately controlled the vegetation species. This is in contrast to Whangamarino, where changes in wetland condition were due to surface water inputs.

7.4 Impacts of artificial modification to Whangamarino wetland

Two significant impacts on the wetland from human modification have been the change in low flow regimes and the change in flood flow regimes.

7.4.1 Weir and minimum water levels

Low flow regimes have been affected in two parts. Firstly, sand abstraction, river training and lowering of the Waikato River bed resulted in lower minimum water levels in the wetland by over 1.0 m from 1960–1990. Secondly, the installation of an artificial weir (2000) had increased minimum water levels to around 3.0 m, in an attempt to restore wetland condition back to a more original state.

The initial lowering of the Whangamarino River water levels was the possible cause for a large scale invasion of manuka into the southern portion of the wetland, which occurred during the period when water levels were at their lowest on record (Reeves 1994). Reduced water levels resulting in succession from herbaceous to woody vegetation has been well documented in many overseas studies (Toner & Keddy 1997).

At this stage it is hard to identify the possible changes higher minimum water levels (due to the installation of the weir) may be having on wetland condition. The vegetation present closer to the Whangamarino River are flood tolerant species, capable of living in a dynamic hydrological environment. Hence it is likely the marshland will continue to be colonised by invasive species, regardless of the higher minimum water levels. The manuka belt is well established (with a canopy height of up to 6 m) and would take a significant amount of time to revert back to original restiad bog. Coupled with the knowledge that flood events regularly inundate up to the fringe of the restiad bog and bring in nutrients, it is likely the manuka belt will remain established (despite raising minimum water levels) and could potentially encroach further into the ombrotrophic centre of the wetland.

Reeves (1994), Shearer (1997) and Shearer and Clarkson (1998) identified that areas of the wetland have likely been modified from lower water levels and the development of the flood control regime, with increased peat degradation and changes in vegetation composition towards invasive species. Trepel and Kluge (2002) suggested restoring natural flood inundation (by removing drainage) and raising minimum water levels to enhance moisture levels and encourage favourable wetland vegetation growth. They simulated raising the water level in the wetland by 30 cm, but this did not reduce peat loss as too much of the area was undergoing oxidation. Unfortunately, due to the level of peat degradation, subsidence and loss of peat hydraulic conductivities, raising the water table in this location would likely have only resulted in a shallow lake system.

It is possible lower minimum water levels affected sites along this transect line closer to the river, which show strong controls to the river hydrological regime throughout the year. Yet sites further into the wetland (such as water level site 3.5

950 m from the river) show an independence from the river water level regime for over half of the year. The enhanced peat degradation, invasion of weeds and manuka encroachment further into the wetland can most likely be attributed to flood inundation, rather than minimum water levels.

7.4.2 Flood control scheme

The LWWFCS is having a significant impact on the southern portion of Whangamarino wetland. The transect line covers an area which likely has some of the highest amounts of nutrient and sediment deposition in the wetland, due to the nearby confluence of the two dominant surface water inputs (Whangamarino River and Pungarehu Canal). Gibbs (2009) confirmed a significant amount of sediment is coming in from Whangamarino River during flood events, while Lake Waikare only contributes a small proportion. Assessment of sediment build up from the Pungarehu Canal was undertaken by Environment Waikato (2009) as part of resource consent monitoring. The Whangamarino River and Pungarehu Canal (near their confluence) both showed sediment deposition increased over a three year period (2005–2008). The observed increase in nutrients and sediments along the transect line is likely directly related to flood inputs. Frequency and duration of flood inundation events increase towards the river, and so do nutrients and sediment concentrations. This has led to the invasion of plants which outcompete natives and extended the swampland and marshland into what was once a large expanse of restiad bog.

It is likely flood inundation distances have increased compared to natural flooding, since the outflow of the Whangamarino River is restricted through the control gate, which results in ponding of nutrient and sediment-rich water through the wetland. Trepel and Kluge (2002) looked into the creation of a more natural flood regime for the possible restoration of the Eider Valley peatland, but this was not implemented due to the debate over a flood regime's benefits, where flooding has been questioned as a source for eutrophication in wetlands with negative effects on species composition (Olde Venterink 2000).

Reid & Quin (2004) examined the response of wetlands from regulated flood events in the highly controlled Murray River in Australia. Up to 80% of divertible flow in this river was being used for agriculture. The study aimed to identify if restoration of wetlands, which were drying out through periodic controlled flood inundation, was possible. After flooding (over 2 years), vegetation species composition began to change and through statistical techniques was linked directly to hydrological inputs and environmental variables such as soil characteristics (nutrients, bulk density, structure). This is similar to what has been identified at Whangamarino, where vegetation corresponds with soil characteristics and hydrological inputs.

Galat *et al.* (1998) studied the significantly modified Missouri River, USA which has been channelized, leveed (stop banks) and the banks stabilised for flood control. Controlled and highly managed flooding was recommended to ensure wetland survival along the river floodplains, as some wetlands (bogs) required little nutrient inputs to maintain condition (and therefore reduced flooding), while floodplain swamps required greater inundation frequency.

This study indicates flooding can be used to restore wetland health in some areas, but has to be highly controlled. Nutrients and sediment from flooding can potentially alter wetland condition (from an ombrotrophic peatland to minerotrophic swampland). Hence the flood regime in the Whangamarino wetland reduces the primary risk which affects most wetlands worldwide (drainage), but it can still be altered significantly by a flood regime which is not managed for wetland health or biodiversity goals (i.e. through winter the wetland has unrestricted amounts of exacerbated flood inundation).

7.5 Recommendations

Whangamarino wetland requires ongoing monitoring to identify any further changes the hydrological regime may be having on wetland condition, such as vegetation composition and nutrient inputs. The controlled flood regime is established at Whangamarino wetland and further research needs to be undertaken to assess its impact over the next 5–10 years. A detailed study of flood water quality (during various flood events) would be useful to determine the possible sources and concentrations of nutrients and sediment coming into Whangamarino wetland from flood events. Sediments and nutrients brought in through flooding and deposited throughout the wetland have a primary impact on changing vegetation composition, and need to be mitigated.

This could involve riparian planting and restrictions on farming or cropping practices in the upper Whangamarino River catchment, to reduce sediment loads entering the wetland. Significant inputs coming into Lake Waikare (primarily from Matahura Stream) ultimately will enter the wetland, and therefore requires similar approaches to the Whangamarino catchment. Similar to the Whangamarino study, Trepel & Kluge (2002) recommended an extensive land use scheme incorporating reduced cropping and pastoral grazing in the upper catchment and farms neighbouring the wetlands in the Eider Valley. Fencing of farmland directly adjacent to the wetland is important to ensure cattle do not encroach into the pristine restiad bog and encourage weed invasion through disturbance and nutrient inputs.

Nutrient and sediment analysis could be assessed in other portions of the Whangamarino wetland (similar to the study by Gibbs 2009), where it is likely flood inundation distance would differ. Depth profiles (cores) along transects assessing nutrient and sediment concentrations may reveal the longer term impacts of the flood control regime, through greater sediment and nutrient abundance. Also yearly deposition rates could be identified and changes in inundation distance or peat degradation could also be elaborated on for these locations.

Vegetation composition is an important indicator of wetland class and implementation of regular monitoring locations or transect lines through portions of Whangamarino would ensure changes due to the complex hydrological regime could be identified. Further study of the possible encroachment of the manuka zone into the ombrotrophic restiad bog is important, and would be recommended to occur over a few years to identify change as a response directly to flooding. Protection of this bog is important as it takes many hundreds of years to form but could disappear in a much shorter time span.

As flooding is the primary impact on the wetland, mapping the potential inundation through the entire wetland would be beneficial. This could be undertaken through spatial modelling. The knowledge of zones that are regularly inundated with nutrient and sediment rich water (and compared with vegetation changes) could provide an insight into areas most likely to be degrading over time and most sensitive to weed invasion. Management and restoration could then look at targeting these areas that are most at risk.

Peat surface oscillation (PSO) is a natural function in the restiad bog. A detailed study of PSO at various sites along the transect line would give a better understanding of the locations surface changes is. PSO is a potential indicator of wetland health, where surface oscillation only occurs in areas of pristine wetland condition (ombrotrophic peat bogs). Indications of a possible decline in surface elevation over 1 year at Whangamarino wetland could be due to peat degradation, but requires a long term study to investigate if this is occurring.

7.6 Conclusions

Whangamarino is a large and significant wetland in the Waikato region and New Zealand, with a range of ecohydrological characteristics resulting from a mixture of wetland classes. Wetland class and environmental indicators change along a gradient, primarily driven by surface water inputs into each zone increasing closer to the Whangamarino River.

Farming directly next to the wetland has led to the presence of a mineralised swamp belt with higher nutrients and a range of vegetation species, but may be an important buffer zone to prevent further nutrient penetration and cattle invasion into the nearby restiad bog. Atmospheric ammonia deposition in the first 250 m of the wetland-farmland fringe may be contributing to higher nitrogen inputs.

The restiad bog has been reduced in extent since the development of the flood control regime and low water levels, primarily due to invasion of manuka. This zone was low in nutrients, had a stable water table and vegetation dominated by *E. minus*. The area is at risk from further encroachment of the manuka belt should flood events impact further into the bog zone, which would reduce the likelihood of eventual colonisation by *S. ferrugineus* seen in older Waikato restiad bogs (Clarkson *et al.* 2004a).

Swampland and marshland are present closer to the river and are affected increasingly so by a more dynamic water table. This is driven by river flood events inundating the wetland and in doing so alters nutrient abundance and vegetation composition. Spraying of willows near the Whangamarino River by the Department of Conservation has resulted in colonisation of annual plants such as *P. persicaria*. The stunted vegetation near the river meets a buffer belt comprised primarily of *C. tenuicaulis*, which strips out a large amount of sediment and nutrients from flood events over a short distance.

Increased flood inundation due to the flood control regime is the primary cause for nutrient and sediment increases (at the peat surface) and changes in vegetation composition along the transect. Whangamarino wetland is at risk from a loss of biodiversity and a decline in wetland condition. The wetland requires ongoing monitoring and implementation of mitigation methods to ensure any significant ecosystem changes, due to human influences, can be halted or reversed.

References

Agnew, A. D. Q., Rapson, G. L., Sykes, M. T. & Wilson, J. B. 1993, 'The Functional Ecology of Empodisma minus (Hook. f.) Johnson & Cutler in New Zealand Ombrotrophic Mires', *New Phytologist*, vol. 124, no. 4, pp. 703-710.

Asman, W. A. H., Sutton, M. A. & Schjorring, J. K. 1998, 'Ammonia: emission, atmospheric transport and deposition', *New Phytologist*, vol. 139, no. 1, pp. 27-48.

Atkinson, I. A. E. 1985, 'Derivation of vegetation mapping units for an ecological survey of Tongariro National Park, North Island, New Zealand', *New Zealand Journal of Botany, The Royal Society of New Zealand*, vol. 23, pp. 361-378.

Baisden, T. 2008, *Ammonia Concentrations within the Taupo Region during March-April 2008*, Draft Report, GNS Science, Lower Hutt.

Balek, J. 2003, 'Hydrology of wetlands in the headwaters of great African rivers', *NATO Advanced Research Workshop on Environmental Role of Wetlands in Headwaters*, Springer, Marienbad, Czech Republic, pp. 203-210.

Bambalov, N. N. 2005, 'Agrogenic evolution of drained peat soils', *Eurasian Soil Science*, vol. 38, no. 1, pp. 23-30.

Bardsley, W. E. 1989, 'Graphical Estimation of Extreme Value Prediction Functions', *Journal of Hydrology*, vol. 110, no. pp. 315-321.

Battye, R., Battye, W., Overcash, C. & Fudge, S. 1994, *Development and Selection of Ammonia Emission Factors.* , EPA/600/R-94/190, Research Triangle Park, NC.

Bedard-Haughn, A., van Groenigen, J. W. & van Kessel, C. 2003, 'Tracing 15N through landscapes: potential uses and precautions', *Journal of Hydrology*, vol. 272, no. 1-4, pp. 175-190.

Beheim, E. (ed) 2006, *The Effect of Peat Land Drainage and Afforestation on Runoff Dynamics*, Springer, Dordrecht, Nertherlands.

Belbin, L. 1995, 'PATN: Pattern analysis package. Technical reference', in (ed.), Division of Wildlife and Ecology: CSIRO, Canberra, pp.

Biester, H., Selimovic, D., Hemmerich, S. & Petri, M. 2006, 'Halogens in pore water of peat bogs - the role of peat decomposition and dissolved organic matter', *Biogeosciences*, vol. 3, no. 1, pp. 53-64.

Boeye, D. & Verheyen, R. F. 1992, 'The hydrological balance of a groundwater discharge fen', *Journal of Hydrology*, vol. 137, no. 1-4, pp. 149-163.

Bowden, W. B., Fahey, B. D., Ekanayake, J. & Murray, D. L. 2001, 'Hillslope and wetland hydrodynamics in a tussock grassland, South Island, New Zealand', *Hydrological Processes*, vol. 15, no. 10, pp. 1707-1730.

Bragg, O. M. 2002, 'Hydrology of peat-forming wetlands in Scotland', *The Science of The Total Environment*, vol. 294, no. 1-3, pp. 111-129.

Bragg, O. M. & Tallis, J. H. 2001, 'The sensitivity of peat-covered upland landscapes', *Catena*, vol. 42, no. 2-4, pp. 345-360.

Branfireun, B. A. & Roulet, N. T. 1998, 'The baseflow and storm flow hydrology of a precambrian shield headwater peatland', *Hydrological Processes*, vol. 12, no. 1, pp. 57-72.

Brassard, P., Waddington, J. M., Hill, A. R. & Roulet, N. T. 1999, 'Modelling groundwater-surface water mixing in a headwater wetland: implications for hydrograph separation', *50th Annual Hydrology Section Meeting of the Canadian Geophysical Union (CGU-HS)*, Banff, Canada, pp. 2697-2710.

Bridgham, S. D., Updegraff, K. & Pastor, J. 1998, 'Carbon, nitrogen and phosphorus mineralization in northern wetlands', *Ecology*, vol. 79, no. 5, pp. 1545-1561.

Brown, T. L., LeMay, H. E., Bursten, B. E. & Burdge, J. R. 2003, *Chemistry, The Central Science, 9th edition*, Prentice Hall, Upper Saddle River, New Jersey.

Browne, K. 2005, ‘Ecohydrological characterisation of Opuatia wetland complex’, MSc. Thesis, The University of Waikato.

Browne, K. & Campbell, D. 2005, *Ecohydrological characterisation of Opuatia Wetland and recommendations for future management*, Environment Waikato Technical Report, Waikato University, Hamilton, Waikato

Bullock, A. & Acreman, M. 2003, 'The role of wetlands in the hydrological cycle', *Hydrology and Earth System Sciences*, vol. 7, no. 3, pp. 358-389.

Burt, T. P. (ed) 1995, *The Role of Wetlands in Runoff Generation from Headwater Catchments*, John Wiley & Sons Inc. , West Sussex, England.

Campbell, D. & Jackson, R. 2004, 'Hydrology of wetlands', in J. Harding, P. Mosley, C. Pearson & B. Sorrell (ed.), *Freshwaters of New Zealand*, New Zealand Hydrological Society INC and New Zealand Limnological Society INC., Christchurch, New Zealand, pp. 20.1-20.14.

Campbell, D. I. & Williamson, J. L. 1997, 'Evaporation from a raised peat bog', *Journal of Hydrology*, vol. 193, no. 1-4, pp. 142-160.

Carter, V., Bedinger, M. S., Novitzki, R. P. & Wilen, W. O. 1978, 'Water Resources and Wetlands', In P. E. Greeson, J. R. Clark, & J. E. Clark (eds.), *Wetland functions and values: The state of our understanding* (pp. 344-376). Minneapolis, Minn.: American Water Resources Association.

Charman, D. 2002, *Peatlands and Environmental Change*, John Wiley & Sons Ltd, West Sussex, England.

Chimka, C. T., Galloway, J. N. & Cosby, B. J. 1997, *Ammonia and the Chesapeake Bay Airshed*. Chesapeake Bay Program, Scientific and Technical Advisory Committee, Publication 97-1, Annapolis, MD.

Clarkson, B., Schipper, L., Moyersoen, B. & Silvester, W. 2005, 'Foliar $\delta^{15}N$ natural abundance indicates phosphorus limitation of bog species', *Oecologia*, vol. 144, no. 4, pp. 550-557.

Clarkson, B. R. 1997, 'Vegetation recovery following fire in two Waikato peatlands at Whangamarino and Moanatuatua, New Zealand', *New Zealand Journal of Botany, The Royal Society of New Zealand*, vol. 35, pp. 167-179.

Clarkson, B. R. 2002, 'Swamps, fens and bogs.' in M. M. B. Clarkson, T. Downs (ed.), *Botany of the Waikato*, Waikato Botanical Society, pp. 49-57.

Clarkson, B. R., Schipper, L. A. & Lehmann, A. 2004a, 'Vegetation and peat characteristics in the development of lowland restiad peat bogs, North Island, New Zealand. ' *Wetlands*, vol. 24, no. 1, pp. 133-151.

Clarkson, B. R., Sorrell, B. K., Reeves, P. N., Champion, P. D., Partridge, T. R. & Clarkson, B. D. 2004b, *Handbook for monitoring wetland condition*, A Ministry for the Environment Sustainable Management Fund Project (5105).

Clevering, O. A., Blom, C. W. P. M. & Van Vierssen, W. 1996, 'Growth and morphology of *Scirpus lacustris* and *S. maritimus* seedlings as affected by water level and light availability', *Functional Ecology*, vol. 10, pp. 289-296.

Clymo, R. S. 1983, 'Ecosystems of the world 4A Mires: swamp, bog, fen and moor.' in A. J. P. Gore (ed.), Elsevier Scientific Co., Amsterdam, The Nertherlands, pp. 159-224.

Cooke, J. G. 1992, 'Phosphorus Removal Processes in a Wetland after a Decade of Receiving a Sewage Effluent', *J. Environ. Qual.*, vol. 21, no. 4, pp. 733-739.

Craft, C. B. 1996, 'Dynamics of nitrogen and phosphorus retention during wetland ecosystem succession', *Wetlands Ecology and Management*, vol. 4, no. 3, pp. 177-187.

Damman, A. W. H. 1988, 'Regulation of Nitrogen Removal and Retention in Sphagnum bogs and Other Peatlands', *Oikos*, vol. 51, no. 3, pp. 291-305.

Davies-Colley, R. & Wilcock, B. 2004, 'Water quality and chemistry in running waters', in P. M. J. Harding, C. Pearson, B. Sorrell (ed.), *Freshwaters of New Zealand*, New Zealand Hydrological Society INC and New Zealand Limnological Society INC., Christchurch, New Zealand, pp. 11.1-11.18.

Department of Conservation 2007, *Whangamarino Wetland 2007 Monitoring Report*, Tonkin and Taylor LTD, Hamilton.

Department of Conservation 2009, *Whangamarino Wetland*, viewed November 10th 2009, http://www.doc.govt.nz/conservation/land-and-freshwater/wetlands/wetlands-by-region/waikato/whangamarino/

Department of Conservation 2010a, *A Directory of Wetlands in New Zealand: Waikato Conservancy, Whangamarino Wetland*, Department of Conservation, 34-43, Wellington, New Zealand.

Department of Conservation 2010b, *Ramsar Convention on Wetlands 10th Conference of Parties: The Ramsar Convention*, Department of Conservation, Wellington, New Zealand.

Department of Conservation 2010c, *Management of Whangamarino wetland*, viewed February 1st 2011, http://www.doc.govt.nz/conservation/land-and-freshwater/wetlands/arawai-kakariki-wetland-restoration-programme/sites/whangamarino/management/

Devito, K. J., Hill, A. R. & Roulet, N. 1996, 'Groundwater-surface water interactions in headwater forested wetlands of the Canadian Shield', *Journal of Hydrology*, vol. 181, no. 1-4, pp. 127-147.

Di, H. J. & Cameron, K. C. 2008, 'Sources of nitrous oxide from 15N-labelled animal urine and urea fertiliser with and without a nitrification inhibitor, dicyandiamide (DCD)', *Australian Journal of Soil Research*, vol. 46, pp. 76-82.

Eaton, A. K., Rouse, W. R., Lafleur, P. M., Marsh, P. & Blanken, P. D. 2001, 'Surface energy balance of the western and central Canadian subarctic: Variations in the energy balance among five major terrain types', *Journal of Climate*, vol. 14, no. 17, pp. 3692-3703.

Emili, L. A. & Price, J. S. 2006, 'Hydrological processes controlling ground and surface water flow from a hypermaritime forest-peatland complex, Diana Lake Provincial Park, British Columbia, Canada', *Hydrological Processes*, vol. 20, no. 13, pp. 2819-2837.

Enviromental Protection Agency 2010b, *Phosphorous, All Forms (Colorimetric, Ascorbic Acid, Two Reagent) Method 365.3*, viewed http://water.epa.gov/scitech/swguidance/methods/bioindicators/upload/2007_07_10_methods_method_365_3.pdf

Environment Waikato 2007, *Lake Waikare Management Options*, Environment Waikato, Hamilton, New Zealand.

Environment Waikato 2009, *Resource Consent 101727 (Lake Waikare)-Sediment Deposition Survey Results*, Waikato Regional Council, Hamilton, New Zealand.

Environmental Protection Agency 2010a, *EPA Method 200.2 Digestion of sediments for ICP-MS total recoverable elements analysis Version 3*, viewed January 2010, http://www.caslab.com/EPA-Method-200_2/

Eser, P. & Rosen, M. R. 1999, 'The influence of groundwater hydrology and stratigraphy on the hydrochemistry of Stump Bay, South Taupo Wetland, New Zealand', *Journal of Hydrology*, vol. 220, no. 1-2, pp. 27-47.

Evans, M. G., Burt, T. P., Holden, J. & Adamson, J. K. 1999, 'Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995', *Journal of Hydrology*, vol. 221, no. 3-4, pp. 141-160.

Falkengren-Grerup, U., Michelsen, A., Olsson, M. O., Quarmby, C. & Sleep, D. 2004, 'Plant nitrate use in deciduous woodland: the relationship between leaf N, 15N natural abundance of forbs and soil N mineralisation', *Soil Biology and Biochemistry*, vol. 36, no. 11, pp. 1885-1891.

Fitzgerald, D. F., Price, J. S. & Gibson, J. J. 2003, 'Hillslope-swamp interactions and flow pathways in a hypermaritime rainforest, British Columbia', *Hydrological Processes*, vol. 17, no. 15, pp. 3005-3022.

Flaig, W. (ed) 1986, *Some Basic Principles of Water Management in Peat*, Elsevier Applied Science Publishers, New York.

Fowler, D., Cape, J. N., Unsworth, M. H., Mayer, H., Crowther, J. M., Jarvis, P. G., Gardiner, B. & Shuttleworth, W. J. 1989, 'Deposition of Atmospheric Pollutants on Forests [and Discussion]', *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, vol. 324, no. 1223, pp. 247-265.

Fritz, C., Campbell, D. I. & Schipper, L. A. 2008, 'Oscillating peat surface levels in a restiad peatland, New Zealand - magnitude and spatiotemporal variability', *Hydrological Processes*, vol. 22, no. 17, pp. 3264-3274.

Fuchsman, C. H. (ed) 1986, *Peat and Water: Aspects of Water Retention and Dewatering in Peat*, Elsevier Applied Science Publishers, Minnesota, United States of America.

Galat, D. L., Fredrickson, L. H., Humburg, D. D., Bataille, K. J., Bodie, J. R., Dohrenwend, J., Gelwicks, G. T., Havel, J. E., Helmers, D. L., Hooker, J. B., Jones, J. R., Knowlton, M. F., Kubisiak, J., Mazourek, J., McColpin, A. C., Renken, R. B. & Semlitsch, R. D. 1998, 'Flooding to Restore Connectivity of Regulated, Large-River Wetlands', *BioScience*, vol. 48, no. 9, pp. 721-733.

Gibbs, M. 2009, *Whangamarino Wetland pilot study: sediment sources*, Hamilton, New Zealand, 2-39. National Institute of Water & Atmospheric Research LTD.

Gibson, J. J., Price, J. S., Aravena, R., Fitzgerald, D. F. & Maloney, D. 2000, 'Runoff generation in a hypermaritime bog-forest upland', *Hydrological Processes*, vol. 14, no. 15, pp. 2711-2730.

Gronlund, A., Hauge, A., Hovde, A. & Rasse, D. P. 2008, 'Carbon loss estimates from cultivated peat soils in Norway: a comparison of three methods', *Nutrient Cycling in Agroecosystems,* vol. 81, no. 2, pp. 157-167.

Güsewell, S., Bollens, U., Ryser, P. & Klötzli, F. 2003, 'Contrasting effects of nitrogen, phosphorus and water regime on first- and second-year growth of 16 wetland plant species', *Functional Ecology*, vol. 17, no. 6, pp. 754-765.

Güsewell, S. & Koerselman, W. 2002, 'Variation in nitrogen and phosphorus concentrations of wetland plants', *Perspectives in Plant Ecology, Evolution and Systematics,* vol. 5, no. 1, pp. 37-61.

Haigh, M. (ed) 2006, *Environmental Change in Headwater Peat Wetlands, UK*, Springer, Dordrecht, Netherlands.

Haigh, M. & Kilmartin, M. 2003, 'Hydrology of disturbed peat-land, heads of the valleys, Wales', *NATO Advanced Research Workshop on Environmental Role of Wetlands in Headwaters*, Springer, Marienbad, Czech Republic, pp. 95-105.

Halabuk, A. 2006, 'Influence of different vegetation types on saturated hydraulic conductivity in alluvial topsoils', *BIOHYDROLOGY 2006 International Conference*, Slovak Academic Press Ltd, Prague, Czech Republic, pp. S266-S269.

Hodges, T. A. & Rapson, G. L. 2010, 'Is *Empodisma minus* the ecosystem engineer of the FBT (fen–bog transition zone) in New Zealand?' *Journal of the Royal Society of New Zealand*, vol. 40, no. 3, pp. 181 - 207.

Hogg, E. H., Lieffers, V. J. & Wein, R. W. 1992, 'Potential Carbon Losses from Peat Profiles: Effects of Temperature, Drought Cycles, and Fire', *Ecological Applications*, vol. 2, no. 3, pp. 298-306.

Holden, J. & Burt, T. P. 2003a, 'Hydrological Studies on Blanket Peat: The Significance of the Acrotelm-Catotelm Model', *Journal of Ecology*, vol. 91, no. 1, pp. 86-102.

Holden, J. & Burt, T. P. 2003b, 'Runoff production in blanket peat covered catchments', *Water Resources Research*, vol. 39, no. 7.

Holden, J., Shotbolt, L., Bonn, A., Burt, T. P., Chapman, P. J., Dougill, A. J., Fraser, E. D. G., Hubacek, K., Irvine, B., Kirkby, M. J., Reed, M. S., Prell, C., Stagl, S., Stringer, L. C., Turner, A. & Worrall, F. 2007, 'Environmental change in moorland landscapes', *Earth-Science Reviews*, vol. 82, no. 1-2, pp. 75-100.

Holland, E. A., Braswell, B. H., Sulzman, J. & Lamarque, J.-F. 2005, 'Nitrogen deposition onto the United States and Western Europe: Synthesis of observations and models', *Ecological Applications*, vol. 15, no. 1, pp. 38-57.

Ingram, H. A. P. 1978, 'Soil layers in Mires: Function and Terminology', *European Journal of Soil Science*, vol. 29, no. 2, pp. 224-227.

Ingram, H. A. P. 1983, *Hydrology: Ecosystems of the World 4A*, Elsevier, Oxford, Oxford.

Jiang, M., Lu, X. G., Xu, L. S., Chu, L. J. & Tong, S. Z. 2007, 'Flood mitigation benefit of wetland soil - A case study in momoge national nature reserve in China', *Ecological Economics*, vol. 61, no. 2-3, pp. 217-223.

Johnson, P. & Gerbeaux, P. 2004, *Wetland types in New Zealand*, Wellington, New Zealand.

Jolly, I. D., McEwan, K. L. & Holland, K. L. 2008, 'A review of groundwater-surface water interactions in arid/semi-arid wetlands and the consequences of salinity for wetland ecology', *Ecohydrology*, vol. 1, no. 1, pp. 43-58.

Kalbitz, K. & Geyer, S. 2000, 'Different effects of peat degradation on dissolved organic carbon and nitrogen', *10th International Meeting of the International-Humic-Substances-Society*, Pergamon-Elsevier Science Ltd, Toulouse, France, pp. 319-326.

Kechavarzi, C., Dawson, Q., Leeds-Harrison, P. B., SzatyLowicz, J. & Gnatowski, T. 2007, 'Water-table management in lowland UK peat soils and its potential impact on CO2 emission', *Soil Use and Management*, vol. 23, no. 4, pp. 359-367.

Kirchmann, H., Esala, M., Morken, J., Ferm, M., Bussink, W., Gustavsson, J. & Jakobsson, C. 1998, 'Ammonia emissions from agriculture.' *Nutrient Cycling in Agroecosystems*, vol. 51, no. pp. 1-3.

Kvaerner, J. & Klove, B. 2006, 'Tracing sources of summer streamflow in boreal headwaters using isotopic signatures and water geochemical components', *Journal of Hydrology*, vol. 331, no. 1-2, pp. 186-204.

Kvaerner, J. & Klove, B. 2008, 'Generation and regulation of summer runoff in a boreal flat fen', *Journal of Hydrology*, vol. 360, no. 1-4, pp. 15-30.

Ludden, A. P., Firnk, D. L. & Johnson, D. H. 1983, 'Water storage capacity of natural wetland depressions in the Devils Lake Basin of North Dakota', *Journal of Soil and Water Conservation*, vol. 38, no. 1, pp. 45-48.

Marani, M., Silvestri, S., Belluco, E., Ursino, N., Comerlati, A., Tosatto, O. & Putti, M. 2006, 'Spatial organization and ecohydrological interactions in oxygen-limited vegetation ecosystems', *Water resources research*, vol. 42, no. pp. 1-12.

Martini, I. P., Cortizas, A. M. & Chesworth, W. (ed) 2006, *Peatlands: Evolution and Records of Environmental and Climate Changes*, Development in Earth Surface Processes, Elseviers, New York.

Massey University 2010, *New Zealand Weeds: Willow Weed*, viewed August 2010, http://weeds.massey.ac.nz/weeds.asp?pid=41

McGill, W. B. & Cole, C. V. 1981, 'Comparative aspects of cycling of organic C, N, S and P through soil organic matter', *Geoderma*, vol. 26, no. 4, pp. 267-286.

Ministry of Environment 1999, *Ambient Water Quality Guidelines*, Overview report, Government of British Columbia,

Mitsch, W. J. & Gosselink, J. G. 2007, *Wetlands: Fourth edition*, John Wiley & Sons Inc. , Hoboken, New Jersey, U.S.A.

Mullholand, W. M. 1991, *Whangamarino Wetland: Hydrological issues associated with water level management*, Hamilton, New Zealand.

Murray-Hudson, M., Wolski, P. & Ringrose, S. 2006, 'Scenarios of the impact of local and upstream changes in climate and water use on hydro-ecology in the Okavango Delta, Botswana', *Journal of Hydrology*, vol. 331, no. 1-2, pp. 73-84.

National Wetland Trust 2009, *National Wetland Trust Whangamarino and Lake Waikare Trail Guide*, Department of Conservation, National Wetland Trust, Environment Waikato, Quarry Road, 1-2 Hamilton, New Zealand.

New Zealand Institute of Chemisty 2010, *The fate of phosphate fertiliser in soil*, viewed March 2010, http://nzic.org.nz/ChemProcesses/soils/

Norton, D. A. & De Lange, P. J. 2003, 'Fire and Vegetation in a Temperate Peat Bog: Implications for the Management of Threatened Species', *Conservation Biology*, vol. 17, no. 1, pp. 138-148.

Norweco 2010, *Method 2540 D. Total Suspended Solids Dried at 103–105°C,* viewed June 15th 2010, http://www.norweco.com/html/lab/test_methods/2540dfp.htm

Novitzki, R. P. 1978, 'Hydrologic Characteristics of Wisconsin 's Wetlands and Their Influence on Floods, Stream Flow, and Sediment ', *Wetland Functions and Values: The State of Our Understanding*, American Water Resources Association, Florida, pp. p 377-388.

Ogawa 2011, *Ogawa Passive Sampler Protocols*, viewed 10th February 2010, http://www.ogawausa.com/protocols.html

Olde Venterink, H. 2000, ‘Nitrogen, phosphorus and potassium flows controlling plant productivity and species richness’, PhD Thesis, Faculty of Geographical Science, Utrecht University.

Panno, S. V., Hackley, K. C., Hwang, H. H. & Kelly, W. R. 2001, 'Determination of the sources of nitrate contamination in karst springs using isotopic and chemical indicators', *Chemical Geology*, vol. 179, no. 1-4, pp. 113-128.

Parfitt, R., Schipper, L., Baisden, W. & Elliott, A. 2006, 'Nitrogen inputs and outputs for New Zealand in 2001 at national and regional scales', *Biogeochemistry*, vol. 80, no. 1, pp. 71-88.

Park, G. 2002, 'Swamps which might doubtless Easily be drained: Swamp drainage and its impact on the indigenous', in E. P. a. T. Brooking (ed.), *Environmental histories of New Zealand*, Oxford University Press, pp. 151-165.

Parkyn, S. & Wilcock, B. 2004, 'Impacts of agricultural land use', in P. M. J. Harding, C. Pearson, B. Sorrell (ed.), *Freshwaters of New Zealand,*, New Zealand Hydrological Society Inc. & New Zealand Limnological Society Inc., Chrustchurch, New Zealand, pp. 34.1-34.16.

Pearson, C. & Henderson, R. 2004, 'Flood and low flows', in J. Harding, P. Mosley, C. Pearson & B. Sorrell (ed.), *Freshwaters of New Zealand*, New Zealand Limnological Society INC and New Zealand Hydrological Society INC, Christchurch, pp. 10.1-10.16.

Price, J. S., Branfireun, B. A., Waddington, J. M. & Devito, K. J. 2003, 'Advances in Canadian wetland hydrology, 1999-2003', *Annual Hydrology Session Meeting of the Canadian-Geophysical-Union*, Banff, Canada, pp. 201-214.

Price, J. S., Cagampan, J. & Kellner, E. 2005, 'Assessment of peat compressibility: is there an easy way?' *Hydrological Processes*, vol. 19, no. 17, pp. 3469-3475.

Reeves, P. N. 1994, Fifty years of vegetation change in the Whangamarino wetland: past influences and future management options, MSc. Thesis, University of Auckland.

Reid, M. & Quinn, G. 2004, 'Hydrologic regime and macrophyte assemblages in temporary floodplain wetlands: Implications for detecting responses to environmental water allocations', *Wetlands*, vol. 24, no. 3, pp. 586-599.

Roa-García, C. 2009, ‘Wetland and water dynamics in small tropical headwater catchments of the Andes’, PhD Thesis, The University of British Columbia.

Roadman, M. J., Scudlark, J. R., Meisinger, J. J. & Ullman, W. J. 2003, 'Validation of Ogawa passive samplers for the determination of gaseous ammonia concentrations in agricultural settings', *Atmospheric Environment*, vol. 37, no. pp. 2317–2325.

Roberts, A. H. C. & Longhurst, R. D. 2002, 'Cadmium cycling in sheep-grazed hill-country pastures', *New Zealand Journal of Agricultural Research*, vol. 45, pp. 103-112.

Roberts, A. H. C., Longhurst, R. D. & Brown, M. W. 1994, 'Cadmium status of soils, plants, and grazing animals in New Zealand', *New Zealand Journal of Agricultural Research*, vol. 37, pp. 119-129.

Robinson, D. 2001, 'δ^{15}N as an integrator of the nitrogen cycle', *Trends in Ecology & Evolution*, vol. 16, no. 3, pp. 153-162.

Rothbaum, H. P., McGaveston, D. A., Wall, T., Johnston, A. E. & Mattingly, G. E. G. 1979, 'Uranium accumulation in soils from long-continued applications of superphosphate', *Journal of Soil Science*, vol. 30, no. 1, pp. 147-153.

Roy, A. H. 2001, 'Fertilizer feeds the world', *Fertilizer Industry Federation of Australia INC "Fertilizers in Focus"* pp. 1-13.

Schipper, L. A., Percival, H. J. & Sparling, G. P. 2004, 'An approach for estimating when soils will reach maximum nitrogen storage', *Soil Use and Management*, vol. 20, no. 3, pp. 281-286.

Scudlark, J. R. & Church, T. M. 1999, *A Comprehensive Reexamination of the Input of Atmospheric Nitrogen to the Rehoboth and Indian River Estuaries.*, Report submitted to the Centre for the Inland Bays and Delaware Department of Natural Resources and Environmental Control, Dover.

Scudlark, J. R., Jennings, J. A., Roadman, M. J., Savidge, K. B. & Ullman, W. J. 2005, 'Atmospheric nitrogen inputs to the Delaware Inland Bays: the role of ammonia', *Environmental Pollution*, vol. 135, no. 3, pp. 433-443.

Shearer, J. C. 1997, 'Natural and anthropogenic influences on peat development in Waikato/Hauraki Plains restiad bogs', *Journal of the Royal Society of New Zealand* vol. 27, no. 3, pp. 295-313.

Shearer, J. C. & Clarkson, B. R. 1998, 'Whangamarino Wetland: Effects of lowered river levels on peat and vegetation', *International Peat Journal*, vol. 8, no. pp. 52-65.

Smakhtin, V. U. & Batchelor, A. L. 2005, 'Evaluating wetland flow regulating functions using discharge time-series', *Hydrological Processes*, vol. 19, no. 6, pp. 1293-1305.

Smith, R. L. & Smith, T. M. 2000, *Ecology and Field Biology 6th Edition*, Prentice Hall, New York, USA.

Solinst 2009, *Solinst Levelogger Series: User Guide*, viewed 1st January 2010, http://www.solinst.com/Prod/3001/3001-data-loggers.html

Sorrell, B. & Gerbeaux, P. 2004, 'Wetland Ecosystems', in P. M. J. Harding, C. Pearson, B. Sorrell (ed.), *Freshwaters of New Zealand*, New Zealand

Hydrological Society Inc. and New Zealand Limnological Society Inc. , Christchurch, New Zealand, pp. 28.1-28.15.

Sorrell, B., Partridge, T., Clarkson, B., Jackson, R., Chagué-Goff, C., Ekanayake, J., Payne, J., Gerbeaux, P. & Grainger, N. 2007, 'Soil and vegetation responses to hydrological manipulation in a partially drained polje fen in New Zealand', *Wetlands Ecology and Management*, vol. 15, no. 5, pp. 361-383.

Sorrell, B., Reeves, P. N. & Clarkson, B. 2004b, 'Wetland management and restoration', in P. M. J. Harding, C. Pearson, B. Sorrell (ed.), *Freshwaters of New Zealand*, New Zealand Hydrological Society Inc. & New Zealand Limnological Society Inc. , Christchurch, New Zealand, pp. 40.1-40.12.

Stevenson, A. C., Jones, V. J. & Battarbee, R. W. 1990, 'The Cause of Peat Erosion: A Palaeolimnological Approach', *New Phytologist*, vol. 114, no. 4, pp. 727-735.

Stevenson, B. A. 2004, 'Changes in phosphorus availability and nutrient status of indigenous forest fragments in Pastoral New Zealand Hill country', *Plant and Soil*, vol. 262, no. 1, pp. 317-325.

Sutton, M. A., Dragostis, U., Tang, Y. S. & Fowler, D. 2000, 'Ammonia emissions from non-agricultural sources in the UK.' *Atmospheric Environment*, vol. 34, pp. 855-869.

Sutton, M. A., Fowler, D., Moncrieft, J. B. & Storeton-West, R. L. 1993, 'The exchange of atmospheric ammonia with vegetated surfaces. II: Fertilized vegetation', *Quarterly Journal of the Royal Meteorological Society*, vol. 119, no. 513, pp. 1047-1070.

Sutton, M. A., Moncrieff, J. B. & Fowler, D. 1992, 'Deposition of atmospheric ammonia to moorlands', *Environmental Pollution*, vol. 75, no. 1, pp. 15-24.

Sutton, M. A., Place, C. J., Eager, M., Fowler, D. & Smith, R. L. 1995, 'Assessment of the magnitude of ammonia emissions in the United Kingdom', *Atmospheric Environment*, vol. 29, pp. 1393–1411.

.

Takeda, A., Tsukada, H., Takaku, Y., Hisamatsu, S. i. & Nanzyo, M. 2006, 'Accumulation of uranium derived from long-term fertilizer applications in a cultivated Andisol', *Science of The Total Environment*, vol. 367, no. 2-3, pp. 924-931.

Tessier, J. T. & Raynal, D. J. 2003, 'Use of nitrogen to phosphorus ratios in plant tissue as an indicator of nutrient limitation and nitrogen saturation', *Journal of Applied Ecology*, vol. 40, no. 3, pp. 523-534.

Thompson, M. A., Campbell, D. I. & Spronken-Smith, R. A. 1999, 'Evaporation from natural and modified raised peat bogs in New Zealand', *Agricultural and Forest Meteorology*, vol. 95, no. 2, pp. 85-98.

Thornburrow, B., Williamson, J. & Outram, P. 2009, *Kopouatai Peat Dome Drainage and Desktop Hydrological Study (report prepared for Department of Conservation)*, Sinclair Knight Merz, Auckland.

Timmins, S. M. 1992, 'Wetland vegetation recovery after fire: Eweburn Bog, Te Anau, New Zealand', *New Zealand Journal of Botany, The Royal Society of New Zealand*, vol. 30, pp. 383-399.

Toner, M. & Keddy, P. 1997, 'River hydrology and riparian wetlands: a predictive model for ecological assembly', *Ecological Applications*, vol. 7, no. 1, pp. 236-246.

Trepel, M. & Kluge, W. 2002, 'Ecohydrological characterisation of a degenerated valley peatland in Northern Germany for use in restoration', *Journal for Nature Conservation*, vol. 10, no. 3, pp. 155-169.

Venterink, H. O., Pieterse, N. M., Belgers, J. D. M., Wassen, M. J. & de Ruiter, P. C. 2002, 'N, P and L budgets along nutrient availability and productivity gradients in wetlands', *Ecological Applications*, vol. 12, no. 4, pp. 1010-1026.

Verhoeven, J. T. A., Maltby, E. & Schmitz, M. B. 1990, 'Nitrogen and Phosphorus Mineralization in Fens and Bogs', *Journal of Ecology*, vol. 78, no. 3, pp. 713-726.

Verry, E. S. & Boelter, D. H. 1978, 'Peatland Hydrology', *Wetland Functions and Values: The State of Our Understanding*, American Water Resources Association, Forida, pp. 389-402.

von der Heyden, C. J. 2004, 'The hydrology and hydrogeology of dambos: a review', *Progress in Physical Geography*, vol. 28, no. 4, pp. 544-564.

Waugh, J. R. 2007, *Report on Whangamarino Wetland and its role in flood storage on the lower Waikato River*, Hamilton, New Zealand, 1-8.

Wheeler, B. D. & Shaw, S. C. 1995, 'Plants as hydrologists? An assessment of the value of planrs as indicators of water conditions in fens.' in J. M. R. H. a. A. L. Heathwaites (ed.), *Hydrology and hydrochemistry of British wetlands*, John Wiley & Sons Ltd, Chichester, UK, pp. 63-82.

Whittington, P. N. & Price, J. S. 2005, 'The effects of water table draw-down (as a surrogate for climate change) on the hydrology of a fen peatland, Canada', *31st Annual Meeting of the Hydrology Section of the Canadian-Geophysical-Union*, John Wiley & Sons Ltd, Banff, CANADA, pp. 3589-3600.

Williams, R. M. 1982, 'A model for the dry deposition of particles to natural water surfaces', *Atmospheric Environment*, vol. 16, pp. 1933-1938.

Worrall, F., Burt, T. & Adamson, J. 2004, 'Can climate change explain increases in DOC flux from upland peat catchments?' *Science of the Total Environment*, vol. 326, no. 1-3, pp. 95-112.

Xue, D., Botte, J., De Baets, B., Accoe, F., Nestler, A., Taylor, P., Van Cleemput, O., Berglund, M. & Boeckx, P. 2009, 'Present limitations and future prospects of stable isotope methods for nitrate source identification in surface- and groundwater', *Water Research*, vol. 43, no. 5, pp. 1159-1170.

Yafa, C. & Farmer, J. G. 2006, 'A comparative study of acid-extractable and total digestion methods for the determination of inorganic elements in peat material by inductively coupled plasma-optical emission spectrometry', *Analytica Chimica Acta*, vol. 557, no. 1-2, pp. 296-303.

Zalewski, M. 2002, 'Ecohydrology: the use of ecological and hydrological processes for sustainable management of water resources', *Hydrological Sciences Journal*, vol. 47, no. 5, pp. 823 - 832.

Zalewski, M. 2007, *Ecohydrology as a Concept and Management Tool*, International Centre for Ecology, Polish Academy of Sciences, Poland.

Appendix A

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

Degree of decomposition	Description
H1	Undecomposed: Plant structure unaltered. Yields only clear colourless water.
H2	Almost undecomposed: Plant structure distinct. Yields only clear water coloured light yellow-brown
H3	Very weakly decomposed: Plant structure distinct. Yields distinctly turbid brown water; no peat substance passes between fingers, residue not mushy.
H4	Weakly decomposed: plant structure distinct. Yields strongly turbid water; no peat substance passes between fingers, residue rather mushy.
H5	Moderately decomposed: Plant structure still clear but becoming indistinct. Yields much turbid brown water; some peat escapes between the fingers; residue very mushy
H6	Strongly decomposed: Plant structure somewhat indistinct but clearer in the squeezed residue than in the undisturbed peat. About half the peat escapes between the fingers, residue storngly mushy
H7	Strongly decomposed: Plant structure indistinct but still recognisable. About half the peat escapes between the fingers.
H8	Very strongly decomposed: Plant structures very indistinct. About two-thirds of the peat escapes between the fingers; residue consists almost entirely of resistant remnants such as root fibres and wood.
H9	Almost completely decomposed: Plant structure almost unrecognisable. Almost all the peat escapes between the fingers.
H10	Completely decomposed: Plant structure unrecognisable. All peat escapes between the fingers.

Appendix B

Table B.1: Species list (alphabetical order) for plants found in vegetation plots along the Whangamarino transect line.

Vegetation species
Alternanthera sessilis
Baumea
Bidens frondosa
Blechnum minus
Carex maorica
Convolvulus arvensis
Coprosma tenuicaulis
Cordyline australis
Dianella nigra
Drosera binata
Empodisma minus
Epacris pauciflora
Gleichenia dicarpa
Juncus effusus
Leptospermum scoparium
Lotus penduncululatus
Ludwigia palustris
Muehlenbeckia australis
Nertera scapanoides
Osmunda regalis
Phalaris arundinacea
Phalaris arundinacea
Phormium tenax
Polygonum persicaria
Polygonum salicifolium
Ranunculus repens
Rubus fruticosus
Rumex obtusifolius
Salix cinerea
Schoenus brevifolius

Appendix C: Atmospheric ammonia (NH_3) deposition

Introduction

Atmospheric deposition of nitrogen can occur through wet deposition (rainfall) and dry deposition (volatilisation of NH_3 particles). Volatilisation occurs off local sources, primarily agriculture and is highly influenced by meteorological conditions (such as temperature and wind direction). Volatilised NH_3 is rapidly deposited onto surrounding (downwind) ecosystems (Holland *et al.* 2005). Wet deposition includes NH_4^+ (ammonium) created in the atmosphere from reactions with gaseous NH_3 and dry particulates of NH_3 which is scrubbed out and converted to NH_4^+ (aqueous form) (Asman *et al.* 1998). Dry deposition in New Zealand occurs for a duration of 66.6% of the year, with the remainder of the year being wet deposition (33.3%), although higher total N deposition may occur through wet deposition (even with a shorter timeframe) (Scudlark & Church 1999; Baisden 2008). A small change in NH_3 inputs to nutrient poor wetlands (such as peat bogs) is important, as the by product of NH_3 reacting with water is NH_4^+, the latter of which is directly available to plants (Asman *et al.* 1998; Holland *et al.* 2005). There are very few studies on NH_3 deposition in New Zealand, and appear to be no studies on deposition rates into wetlands.

Measurements of dry deposition include direct chemiluminescence, filtration packs, denuders and gas scrubbers of which all show good agreement with one another (Roadman *et al.* 2003). Passive ammonia samplers are being used more frequently and involve NH_3 diffusing into a reactive surface which chemically traps the gas. Passive samplers are portable, less expensive (than fixed gas scrubbers) and can be deployed for long periods of time. The Ogawa sampler was designed in Japan and has been used to determine concentrations of gaseous species like NO, NO_x, O_3 and NH_3 (Figure C.1) (Roadman *et al.* 2003). The methodology for this sampler has been verified and proven to yield precise concentration compared to reference methods (Roadman et al. 2003).

Six Ogawa samplers and consumables such as collection pads (Ogawa & Co. Inc., Florida, USA) were purchased as a trial to determine atmospheric dry deposition coming into Whangamarino wetland from surrounding farmland.

Figure C.1: Ogawa passive sampler ready for deployment in the field (image from Ogawa USA Inc.).

Deployment and collection

Three of the samplers (attached to stakes with protective rain shelters) were initially positioned directly on the farmland-wetland fringe, while the remaining three were positioned 100 m into the wetland (Figure C.2). This preliminary study aimed to identify if similar NH_3 loads were being measured from the three samplers at each of the sites (i.e. if the measurements were reproducible), relative to prevailing westerly winds.

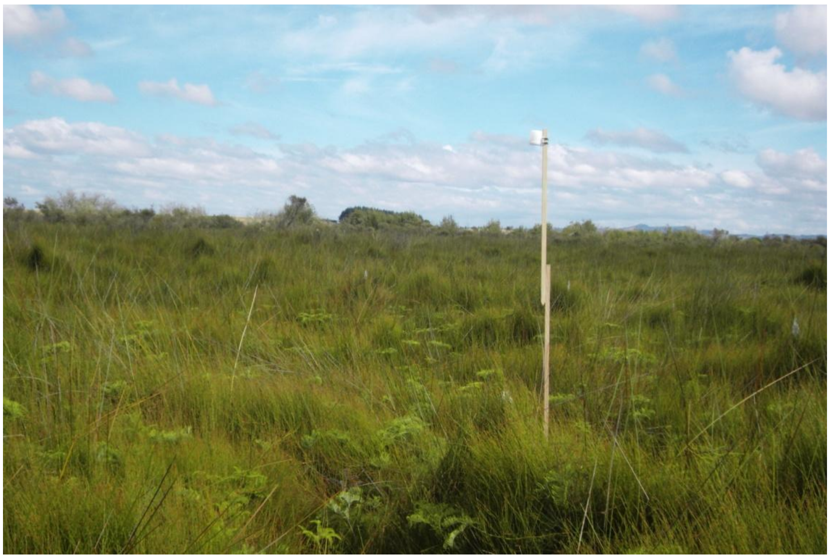


Figure C.2: Ammonia sampler attached to a stake and covered with a rain shelter, positioned 100 m into the restiad bog.

Three months of consistent measurements were taken, where the three samplers in each zone (0, 100 m) had similar concentrations (± 5–10%), with variations likely dependent on meteorological conditions (such as differing wind directions or local temperatures). The samplers were then repositioned to determine the possible input of ammonia into the wetland. Samplers were located at 0, 100, 250, 500, 750, and 1000 m along the established transect (Figure C.3).

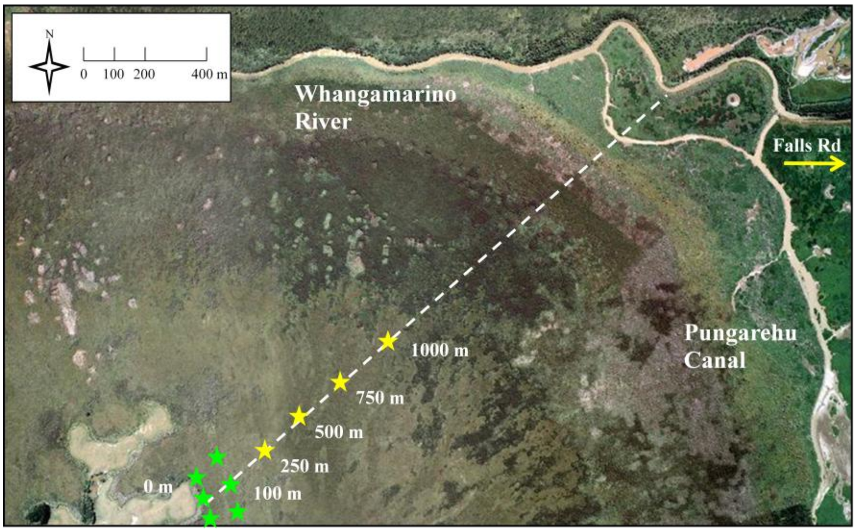


Figure C.3: Preliminary layout of ammonia samplers (green stars) and final layout into the wetland (yellow stars). The two samplers on the transect line nearest the farmland did not change, and were included as 0 and 100 m for the final layout.

The citric collection pads in the passive samplers were retrieved on roughly a monthly basis after exposure in the field (exact duration was recorded). The pads were stored in labelled, sealable glass vials that had been prewashed with de-ionised water. In the field, samplers (and mesh filters) were washed with de-ionised water to prevent cross contamination, dried and then new blank collection pads were installed before being re-deployed.

More details on the method can be found in the NH_3 sampling protocol, version 2.0 (Ogawa 2010), Baisden (2008), Roadman *et al.* (2003) and Scudlark *et al.* (2005).

Analysis

The collected samples were stored in a refrigerator for a maximum of 14 days before being analysed through The University of Waikato Biological Departments discrete analyser. Analysis was undertaken 30–60 minutes after adding 8 ml of ultra pure deionised water to each of the vials (to convert adsorbed NH_3 to NH_4^+). When samples could not be analysed in under 14 days, de-ionised (ultra pure) water was added to each of the vials, which were then frozen to preserve NH_4^+. A blank collection pad was also included in each batch of analyses for quality control (which was subtracted from the final concentrations for each sampler).

Concentrations (mg/L) were converted to deposition rates using a deposition velocity, which is vegetation specific. The deposition velocity (2 cm s^{-1}) was derived from numerous studies looking at short vegetation (present in moorlands and wetlands) (refer to Sutton *et al.* 1992, Sutton *et al.* 1993, Fowler *et al.* 1989 and Holland *et al.* 2005). Numerous calculations to derive the final concentration of nitrogen input (kg ha^{-1} yr^{-1}) relative to duration of deployment, temperature and gas flux can be found in the Appendix D.

Wet deposition strips ammonia from the air and deposits on the land surface. This deposition was not captured by the Ogawa passive samplers (hence needs to be accounted for). For each day where rainfall was >1 mm, the day was assumed to have negligible dry NH_3 deposition (Scudlark *et al.* 2005; Baisden 2008). In total, there were 116 days of rain (33%) from the sampling period of 3-Feb-2010 to 14-Jan-2011, which is consistent with long term means for New Zealand (Baisden 2008). Therefore a correction has been made to determine final concentrations of total N deposition (including both wet and dry deposition).

A complete annual dry deposition load could not be determined because samplers were first deployed on the 3-Feb-2010 and final removal was on the 14-Jan-2011. To overcome this data gap, the average concentrations derived for 0 m and 100 m from the preliminary test period (around three months) were included in the final estimation of yearly total N deposition. The four remaining samplers (250, 500, 750 and 1000 m) were in the field for over half a year (3-Jun-2010 to 14-Jan-2011). This period encompassed summer and winter climate patterns that affect ammonia deposition, so extrapolation to an entire year was possible (although with some uncertainty).

Greater detail on the analysis method can be found in Baisden (2008), Ogawa (2010) and Scudlark *et al.* (2005).

Results

Dry deposition and total N deposition (wet and dry) was greatest on the farmland fringe (0 m) with values of 2.87 and 4.0 kgN ha^{-1} yr^{-1}. Deposition rates decreased into the wetland to background levels from around 500–1000 m, with dry deposition around 1.8–2.0 kgN ha^{-1} yr^{-1} and total deposition around 2.4–2.7 kgN ha^{-1} yr^{-1} (Table C.1, Figure C.4).

Table C.1: Dry deposition N inputs (kg ha^{-1} yr^{-1}) and total N deposition (wet and dry) (kg ha^{-1} yr^{-1}) for Whangamarino wetland, extrapolated for the sampling period of 3-Feb-2010 to 3-Feb-2011.

Distance from farmland (m)	Deployment time (days)	Dry N (kg ha^{-1})	Dry N (kg ha^{-1} yr^{-1})	Total N (kg ha^{-1} yr^{-1})
0	348.9	2.87	3.0	4.0
100	349.0	2.41	2.5	3.4
250	228.9	1.4	2.2	3.0
500	229.0	1.12	1.8	2.4
750	229.0	1.25	2.0	2.7
1000	229.0	1.18	1.9	2.5

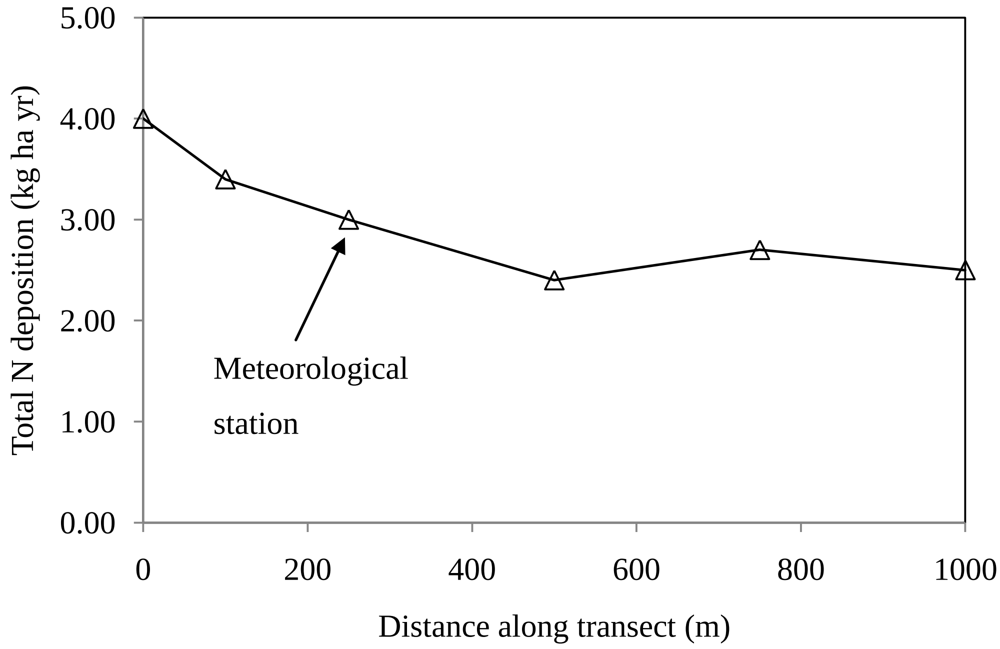


Figure C.4: Total nitrogen inputs (wet and dry) (kg ha^{-1} yr^{-1}) to Whangamarino wetland, extrapolated for the sampling period of 3-Feb-2010 to 3-Feb-2011.

Discussion

Nitrogen deposition was greatest on the wetland fringe likely due to the prevailing westerly winds (Chapter 3) carrying higher amounts of ammonia volatilised from the pasture soils. Ammonia has a relatively short time span in the atmosphere (from a few hours to 5 days) and can be rapidly deposited through dry deposition (Chimka *et al.* 1997; Scudlark & Church 1999).

Most of the total deposition (dry and wet) occurred at the willow belt fringing the farmland (4.0 kgN ha^{-1} yr^{-1}). Deposition decreased in the centre of the restiad bog to around 2.4–2.7 kgN ha^{-1} yr^{-1} and was likely close to background levels (Figure C.4). Parfitt *et al.* (2006) estimated wet and dry deposition in New Zealand to be approximately 1.5 kgN ha^{-1} yr^{-1}, but higher values (>5 kgN ha^{-1} yr^{-1}) were measured in areas of the Manawatu. Higher deposition was predicted in regions with more intensive agriculture (including higher stocking rates and fertiliser application), such as Waikato and the Bay of Plenty. The greater N input on the wetland fringe was likely due to rapid dry deposition of NH_3, which decreased further into the wetland as distance from the agricultural ammonia source increased (Chimka *et al.* 1997; Scudlark & Church 1999). This measured deposition was similar to findings by Baisden (2008) for two farms bordering forestry at Kinloch, New Zealand where values of 2.8 and 3.6 kgN ha^{-1} yr^{-1} were obtained. These levels were 50% and 75% greater than pristine forestry sites (near the coastline), where the increase was attributed to the presence of agriculture.

Deposition at the wetland fringe was about 1.4 kgN ha^{-1} yr^{-1} greater than inputs measured in the centre of the restiad bog. Agricultural activities are the primary source for NH_3 deposition into ecosystems (Roadman *et al.* 2003) and the greater deposition on the fringe of the wetland was presumably derived from adjacent agriculture. This increase of ammonia deposition at the farmland fringe would be providing more nitrogen directly available to plants (as NH_3 is rapidly solubilised to NH_4^+ which can be taken up by plants). The rapid decrease into the wetland suggests only the first 500 m was likely to have greater N inputs from dry deposition.

Wet deposition was not measured in this study (through sampling rainwater), and extrapolated nitrogen inputs are based only on the % of time it was raining. Aerosols in the atmosphere carrying ammonium (NH_4^+) have a longer residence time (> 5 days), can travel further than NH_3 and are removed through wet deposition (Chimka *et al.* 1997: Roadman *et al.* 2003). Wet deposition can have NH_x (NH_4^+ and NH_3) inputs from 50–90%, which therefore can contribute greater impacts to an ecosystem on a wider scale than the locally dry deposited NH_3 (Scudlark and Church 1999). Scudlark and Church (1999) found at Delaware's Inland Bays (USA), dry deposition accounted for only 20% of NH_4^+. If this was applied to Whangamarino wetland, inputs of atmospheric total N (plant available) would be 15 kgN ha^{-1} yr^{-1} on the wetland fringe, 11.2 kgN ha^{-1} yr^{-1} (250 m) and 9.5 kgN ha^{-1} yr^{-1} (in the restiad bog). This substantial increase of plant available nitrogen could be a possible source for future eutrophication, and therefore further studies of wet deposition should be undertaken.

Conclusion and recommendations

Dry deposition on the farmland fringe was 3.0 kgN ha^{-1} yr^{-1} and decreased to 1.8–2.0 kgN ha^{-1} yr^{-1} approximately 500 m into the wetland. Total N inputs (including wet and dry deposition) were estimated to be around 4.0 and 2.5 kgN ha^{-1} yr^{-1} at 0 and 500 m. Higher nitrogen inputs could contribute to greater plant growth and promote colonisation by invasive plant species. Further study is required to determine N inputs into wetlands (near drystock farms) including wet deposition, which may be a significant nutrient source. Intensive dairy farming such as adjacent to Moanatuatua wetland would likely support greater NH_3 volatilisation and local deposition. The affects of dairy farming adjacent to sensitive native ecosystems warrants further investigation.

Appendix D

A CD-ROM is enclosed containing the contents of Appendix D. For a full list of the material on the CD-ROM refer to the ‘Context.txt’ in the root directory.