

A paleolimnological investigation of agricultural  
intensification, water quality and ecosystem change  
at Lake Nganoke, southern Wairarapa, NZ

By

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## **ABSTRACT**

Decreasing water quality of lakes as a result of anthropogenic landuse and specifically agricultural intensification is well documented in New Zealand. However, monitoring records of lake health are typically short, only commencing once signs of lake deterioration are observed. The shortness of the instrumental record precludes a detailed understanding of the relationship between landuse change, lake ecosystem trajectories and the effectiveness of mitigation strategies such as riparian planting. Paleolimnological reconstruction from sediment cores has the potential to develop high-resolution time series that may extend lake monitoring centuries into the past. This thesis uses paleoenvironmental reconstruction to investigate lake ecosystem change and water quality in Lake Nganoke, Wairarapa, New Zealand as a result of landuse intensification. The primary aim of this thesis is to reconstruct the past environment of Lake Nganoke from a pre-human reference state to the current day to assess: 1) how increased nutrient fluxes associated with landuse intensification have impacted the lake ecosystem; and 2) the ability of riparian zones to buffer these fluxes. The reconstruction was achieved using a multi proxy approach with pre and post-human environments of Lake Nganoke characterised using Palynology, geochemistry, eDNA and hyperspectral scanning.

Māori land clearance was identified at ~AD 1450 (95% CI: AD 1417-1551). The appearance of *Pinus* pollen and increases in fertilisation and stocking rates placed European arrival at ~AD 1850 (95% CI: 1809 - 1870), while intensification of agricultural landuse occurred post ~AD 1950 (95% CI: 1948 - 1964). The prehuman environment of Lake Nganoke experienced little change, with the catchment dominated by tall trees and likely heavily forested. The lake ecosystem and water quality during this time showed little to no change, with algal productivity likely driven by a constant input of natural nutrients. Post Māori arrival, algal productivity was reduced suggesting an increase in water quality likely driven by added lake marginal plants providing a riparian buffer to terrestrially derived nutrients. Lake productivity increased dramatically post European arrival ~AD 1850, coeval with an increase in sediment Cd, suggesting that fertilisation may have driven a decline in water quality. Further increases in fertilisation and stocking rates indicate additional agricultural nutrient fluxes entering Lake Nganoke in AD 1950 when agriculture intensified. Abundances in denitrifying Gammaproteobacteria indicate increases in nutrient loading while bloom forming Cyanobacteria peak ~AD 2000 before declining till present. Riparian planting following Māori arrival appears sufficient to buffer the lake against increased terrestrial nutrient fluxes associated with land clearing.

However, a riparian zone that covers the majority of the catchment post European settlement was inadequate in altering the lake's degrading ecosystem and water quality trajectory.

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# **1 CHAPTER 1: INTRODUCTION**

## **1.1 MOTIVATION AND RATIONALE**

The health of lakes in Aotearoa, New Zealand is central to our environmental, economic and cultural wellbeing. In recent years there has been increasing concerns over the declining quality of freshwater (Environment, 2012). Despite these concerns, there has been no robust assessment of the water quality or ecological health of New Zealand's 3,800 (>1 hectares) freshwater lakes and over 95% are un-monitored (Verburg et al., 2010). Furthermore, comprehensive assessment of lake ecosystems and water quality can be time consuming and costly, limiting monitoring to a select few sites. The small number of programmes that do exist are often short, spanning less than 10 years and are only implemented after the target lake has degraded (Ward et al., 1986; Burns et al., 1999a; Burns et al., 2005, 2009). Hence, the true extent of the decline from baseline natural conditions within New Zealand freshwater lakes is largely unknown. This is partly because current monitoring of lake water quality does not provide any information on baseline reference states which can be used to help set informed restoration goals (Bennion et al., 2011; Battarbee et al., 2012).

Constantly deposited over time, sediments preserve measures of historic lake health, landuse and water quality, equivalent to centuries of monitoring (Bennion et al., 2011; Battarbee et al., 2012). These data can therefore provide a continuous and high-resolution historical understanding of the lake environment and ecosystem change that is not possible with conventional methods. Consequently, the “paleo” approach offers otherwise inaccessible insights into New Zealand's freshwater issues.

Rapid expansion and improvements of the agricultural sector are seen as one of the biggest contributions to declining freshwater (Moller et al., 2008; Haggerty et al., 2009; Morgenstern & Daughney, 2012; Foote et al., 2015; Scarsbrook & Melland, 2015). Furthermore, nutrient overloading from increased stocking and fertilisation are directly recognised as drivers of in lake algal blooms and eutrophication (Abell, Özkundakci, et al., 2011). Improving strategies against further pollution is therefore a necessity for addressing declining water quality.

Riparian buffer zones are thought to intercept and mitigate potentially harmful nutrients from reaching fresh-water (Parkyn et al., 2003). Assessing the effectiveness of riparian

zones, however, is often problematic, as plants can take upwards of 5 to 10 years to reach maturity, requiring years of monitoring; during which a multitude of variables must also be accounted for (Anbumozhi et al., 2005). Paleoenvironmental techniques may provide insights into pre-riparian baselines, and how lake health has changed as the vegetation has matured.

The aim of this thesis was to use paleo-based approaches and environmental reconstructions to monitor changes in lake health over a 1,000 year period, assessing the effectiveness of relatively recent (post ~AD 1944) riparian planting. This study focuses on Lake Nganoke, which is a small lake currently surrounded by low intensity farming in the Wairarapa region. The chronology of changes in landuse will be constructed to provide insight into prehuman environmental conditions, Polynesian subsistence practices, agricultural intensification methods and the establishment and development of riparian planting. Using the above identified phases (European agriculture and agricultural intensification), a record of lake water quality and ecosystem evolution will be generated. A comprehensive reconstruction of Lake Nganoke's landuse, ecosystem and water quality change will offer insights into the ability of paleoenvironmental reconstruction to assess the effectiveness of riparian buffer zones in mitigating agriculturally rich run-off.

## **1.2 THESIS OVERVIEW**

The thesis has been set out in 7 Chapters. Chapter 2 presents a comprehensive review of historical landuse and water quality degradation within New Zealand. The importance of riparian zones in mitigating agriculturally rich nutrients is discussed, while the use of paleolimnology and the rationale for applying these methods is also explored. Chapter 3 presents the hypothesis and objectives. Chapter 4 provides background information on the Wairarapa region and Lake Nganoke, specifically geology, climate, vegetation, landuse and human occupation. This section will also outline the chronology of surrounding landuse change, informed by relevant literature and historical accounts. Chapter 5 outlines the methodology and methods and the rationale for why each was used. Chapter 6 highlights the findings of this research. Chapter 7 uses the results of the research to test the hypotheses identified in chapter three and discusses the findings in relation to existing literature. Chapter 8 summarises this study and provides general conclusions.

## **2 CHAPTER 2: LITERATURE REVIEW**

### **2.1 FRAMING THE PROBLEM**

Degradation of water resources as a consequence of agricultural land development is not restricted to New Zealand. Internationally, advancements in agricultural practices has led to greater disparities in water quality and availability (Stoate et al., 2002; Wolfe et al., 2004; Brodie & Mitchell, 2005). Looking to the future, increasing demand to feed a rapidly growing population will only exert greater pressure on the global environment, with increasing landuse intensification required to support this deficit (Giovannucci et al., 2012).

The agricultural sector is and has been a vital part of the New Zealand economy and culture (Quinn et al., 1993; Brown & Stone, 2007). In the year ended June 2016, the agriculture sector alone provided approximately 6.2% of New Zealand's real gross domestic product (GDP) and \$28 billion worth of exports (Government, 2016). However, growing consumer concerns and government restrictions on pollution is forcing the agricultural industry to adopt more sustainable farming practices in order to reduce further environmental degradation and live up to NZ's clean and green reputation (Collier et al., 1995; Environment, 2004).

A leader in agricultural systems, New Zealand has the ability to develop a sustainable framework that addresses the current and future management of natural resources and agricultural production. The need therefore becomes apparent to investigate past responses to changes in agricultural intensification in order to better develop efficient agricultural systems that focus on minimising future environmental degradation (Di & Cameron, 2002).

### **2.2 CHANGING LANDUSE**

Establishing the consequences and effects of historic human influence on lake health is crucial to informing future landuse best practice within localised environments (Groffman et al., 2006; Moore et al., 2008; Bennion et al., 2011; Carmichael & Boyer, 2016). In the case of New Zealand, three distinct landuse phases have occurred; the arrival of Polynesians/Māori (~AD 1300), the arrival of Europeans (~AD 1820) and the intensification of agriculture (~AD 1950).



Dated roughly (AD 1250 ~1200 – 1282), Polynesian settlement saw the introduction of subsistence horticulture and deforestation practices within New Zealand (McIntyre, 2002; Wilmshurst et al., 2011). Whilst the timing of initial occupancy is largely debated within academia, general consensus can be reached on the detection and validation of large-scale colonisation in ~AD 1250. Archaeology and radiocarbon dating of historic sites containing both moa bones and east tropical Polynesian artefacts date settlement no earlier than ~AD 1270 at Wairau Bar in the top of the South Island, (Higham et al., 1999). In the case of the Wellington and the wider Wairarapa region, proof of settlement can be identified in the first few decades of the 14<sup>th</sup> century (Sutton, 1987; McFadgen et al., 1994; Walter et al., 2017). Historical records and oral histories place importance on the modification of tropical subsistence systems to exploit the resource base of a new climate and ecology (McFadgen et al., 1994; Walter et al., 2017).

Further evidence regarding the timing of human arrival in New Zealand is provided through paleoenvironmental records of pollen and charcoal preserved by lake sediments. Lacking metal implements, Māori primarily used fire to keep tracks clear, maintain dwellings free of tall vegetation, clear sites for cultivation, and to encourage bracken (*Pteridium esculentum*): a major source of carbohydrates (McGlone, 1983b, 1989b; McGlone & Wilmshurst, 1999a). While New Zealand is rich in fuel, high moisture levels make naturally occurring fires rare (McWethy et al., 2013). Therefore, periods of intense burning likely indicate the presence of people on the landscape. These events are often reflected in abrupt increases in charcoal particle influx into lake basins, departing from low and irregular baselines (Newnham, Lowe, McGlone, et al., 1998; Wilmshurst et al., 2004; Ogden et al., 2006; Abraham et al., 2013). At the same time, a sharp decrease in tall tree pollen is coeval with an increase in light-demanding shrubs, scrub, bracken and grassland pollen (Elliot et al., 1995; Ogden et al., 2003). Furthermore, lake sediments record increases of detrital silt and sand from fire-disturbed catchments, with lake and swamp flora often changing from oligotrophic sedges (grass-like, wetland plants) to nutrient-demanding wetland plants such as raupo (*Typha*) (Deng et al., 2006). As a result of anthropogenic and natural burning fires, only 50 % of lowland forests remained by the 1800s (Salmon, 1975; McGlone, 1989b; Ewers et al., 2006; Stevens et al., 2013).

The next identifiable landuse change, early European settlement occurred 180 years ago and saw large-scale deforestation practices for the establishment of low-density agriculture. Well documented landscape changes are easily identified in

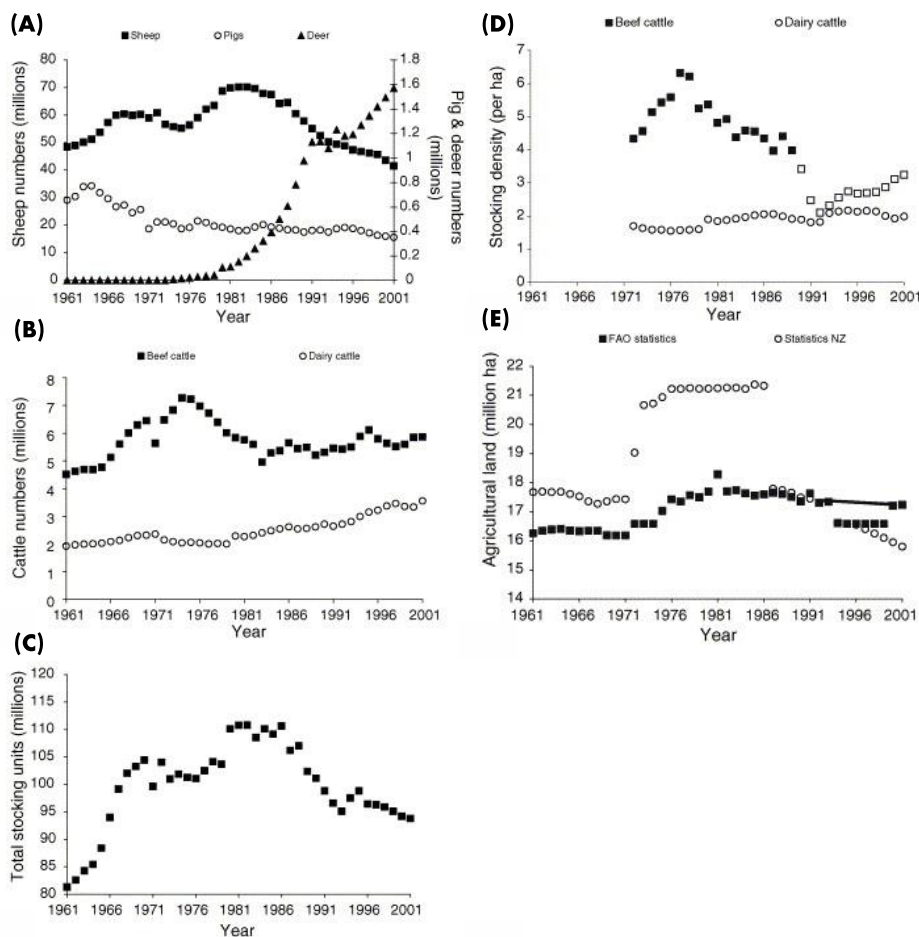
paleoenvironmental records through both the wide range of introduced exotic plant species and the decline of bracken, ferns and native Podocarps (Elliot et al., 1997; Elliot et al., 1998; Byrami et al., 2002; Ogden et al., 2006). Indicative of European settlement, increases in grasses (Poaceae), willow (*Salix*), and sheep sorrel (*Rumex acetosella*) are characteristic of landuse clearance for agriculture. The most significant indicator, however, is the introduction and presence of pine (*Pinus*) (McWethy et al., 2010)

Although present in New Zealand from the late eighteenth century, the generally accepted date for European impact on the terrestrial environment is ~AD 1840 (Wendelken, 1976; Newsome, 1987; Ogden et al., 2006). Following this date and until the mid-1950s, forested and swamp land was rapidly cleared in favour of low-density agriculture. The sheep industry grew from 2,760,000 in AD 1861 to more than 20,230,000 by AD 1901. By the turn of the 20<sup>th</sup> century, 335,000 dairy cows produced a total of 9,000 tons of butter and over 5,000 tons of cheese to be exported to Europe (McLintock, 1966a). Production grew again in the 1920s, with increasing accessibility of superphosphate and the beginning of organised agricultural research paving the way for the future of New Zealand's agriculture (Wood & Pawson, 2008).

Post AD 1950, the expansion of European agriculture as a result of technological innovations and increasing prices for farm products greatly influenced agricultural intensification within New Zealand (McLintock, 1966a; McGlone & Wilmshurst, 1999c; McFadgen, 2003b; Moller et al., 2008). Arguably, the invention of aerial topdressing in the late 1940s was one of the largest factors responsible for this development, including the spreading of superphosphate fertiliser, pastoral seed and poisoned bait from low flying aircrafts (Pike; & Hanmore, 2017). This progression in aerial topdressing enabled applications of fertiliser and seed at greater rates per hectare. Consequently, commercial top-dressing fertilisation increased from 0 tonnes in AD 1950 to 900,000 tonnes by AD 1965, encompassing over 3.5 million hectares. Unsurprisingly, between AD 1950 and AD 1960, sheep numbers within NZ alone rose by 40% (McLintock, 1966b; Peden, 2008). Rising prices and greater returns further intensified agricultural production as farmers had increased capital to invest in fertilisation, seeding, new machinery and the clearing of more land (MacLeod & Moller, 2006a).

By 2015, more than 14 million hectares of land had been cleared and converted to agricultural farming (Statistics NZ, 2019), equating to roughly 50% of New Zealand's

current total land surface (28 million hectares). Intensification during the European era saw the growth of agricultural land from 16 million hectares in AD 1961 to 17 million hectares in AD 2001 (Figure 2.1E), before dropping to 14 million hectares by AD 2015 (Statistics NZ, 2019). Unsurprisingly, stocking rates of cattle and sheep have also dramatically changed during this period, from 82 million in AD 1961 to 95 million in AD 2001 (Figure 2.1A & B), dipping to 38.4 million by AD 2015 (Statistics NZ, 2019). Although sheep numbers have dropped from 50 to 27 million during this period (1961-2015), dairy numbers have risen from 4.5 to 10.1 million, mainly due to the strong international demand for dairy products (Statistics NZ, 2019). Specifically, in the last 30 years there has been a rapid push for intensification with a 627% increase in nitrogen fertiliser applied, a 94% increase in irrigated agricultural land and a 70% increase in dairy cattle numbers (Statistics NZ, 2019).



**Figure 2.1: Statistics for agricultural diversification within New Zealand from 1961 – 2001** Highlighting sheep numbers (A), cattle numbers (B), total stocking units (C), stocking density (D) and agricultural land (E). Adapted from (MacLeod & Moller, 2006a)

Based on the above literature, four clear periods of landuse within New Zealand are identified: pre-human, Māori, European settlement and agricultural intensification. Due

to the large nature of changes within a short time period, the pressure exerted on localised environments and freshwater systems is unsurprising. It is important to note, that while these zones have been established at a national scale, variations in timescales may occur regionally. Further exploration of historical landuse changes specific to the Wairarapa Valley and Lake Nganoke are explored in Section 4.2.

### **2.3 FRESH WATER DEGRADATION**

The last 180 years of European landuse intensification have resulted in a trend towards more eutrophic conditions in New Zealand's freshwater environments, greatly affecting water quality, aquatic habitats and macroinvertebrates communities (Barmuta, 2001; Collier et al., 2014). Eutrophication can be defined as excessive growth of phytoplankton caused by nutrient over-enrichment leading to imbalances in the primary and secondary productivity, and is a well-known problem in lakes (Khan & Ansari, 2005; Yang, Wu, et al., 2008; Conley et al., 2009). While it is a naturally occurring process, eutrophication can be greatly accelerated by human activities primarily through increased rates of nutrients entering a water body.

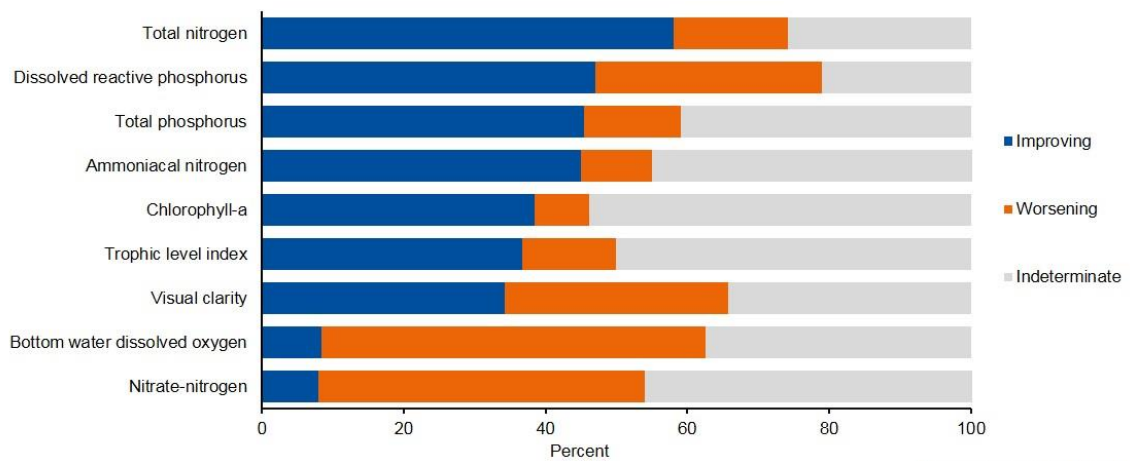
High nutrients within a watershed can stress lake ecosystems, consequently decreasing biodiversity as dominant species and functional groups are overrun by nutrient responsive algae (Song et al., 2019). For instance, an increased input of limiting nutrients (such as nitrogen and phosphorus) into a nutrient poor lake can cause a critical transition to a eutrophic state, where nutrients and organic productivity are high and non-nutrient dependant taxa are depleted (Hall & Smol, 2010; Lowe & Walker, 2014). Low resistance to high nutrient turnover within sediments can also promote growths of toxic bloom forming algae such as Cyanobacteria (Paerl et al., 2001). Dramatic increases in productivity can promote anoxia in the hypolimnion, which in turn diminishes lake ecosystem function and water quality further (Liu & Qiu, 2007; Verburg et al., 2010). The transition to an alternative stable state is often difficult and expensive to reverse or, in some cases, irreversible due to large changes in the physical or ecological properties of the system (Biggs et al., 2012; Scheffer et al., 2012).

The quality of a lake's water is a key indicator of its overall health, with a general consensus highlighting the importance of chemical, physical and microbiocidal elements (Hooda et al., 2000). More specifically, these characteristics include: pH, temperature, dissolved oxygen, conductivity, nitrogen and phosphorous concentrations. All lakes

within New Zealand are rated using the trophic level index (TLI), which is designed to rate bodies of water based on biological activity, water clarity and the amount of nutrients present (Burns et al., 1999b). A parameter within TLI, Chlorophyll a is often an indicator of eutrophication related to algae bloom proliferation and lake biological productivity (Burns et al., 1999b).

While there is agreement on what aspects of water quality to measure, differences on the accepted standards and the techniques are often a point of contention. Disagreement often arises over equipment, sampling strategies and lake health standards (Burns et al., 2009). Similarly, aspects that are valuable indicators for one catchments health may not be as relevant for another (Medema et al., 2003). The need therefore becomes apparent for localised understandings of the relationship between landuse and water quality when considering a lake's health (Bartram et al., 1996; Howard-Williams & Kelly, 2003).

Despite these understandings, it is widely acknowledged that New Zealand's lake health records is vastly short and incomplete (Environment, 2006). As such, comprehensive data of lake health within New Zealand is extremely limited with only 5% having greater than 1 years' worth of monitoring (Verburg et al., 2010). Lake Nganoke is no exception to this, having no current data on present or historic water quality. Nevertheless, there are indications that freshwater health is degrading. Observations over a 10-year monitoring programme highlight a large number of water quality indicators within 72 agriculturally dominated lakes as worsening (Figure 2.2) (Larned, 2015). While not conclusive evidence of a decline from baseline conditions, these indications of lake degradation only serve to emphasise the need for further investigations into lake health within New Zealand; research into Lake Nganoke supports this mandate.



**Figure 2.2: Trends in 72 lakes water quality measures over a 10 year period (2004-2013)**

Lake water quality is quantified as improving (blue) or worsening (orange) based on parameters (total nitrogen, dissolved reactive phosphorous, total phosphorous, ammoniacal nitrogen, chlorophyll-a, trophic level index, visual clarity, bottom water dissolved oxygen and nitrate-nitrogen). Retrieved from Larned (2015)

Globally, further research has linked the expansion and intensification of agricultural production with the rise of nitrogen (N) and inorganic phosphorous (P) nutrients within lake waterbodies (Hall & Schreier, 1996; Hamilton & Schladow, 1997; Elser et al., 2007). Specifically, Gillon et al. (2015) argue the need for baseline assessments in understanding how long-term changes in nutrient overloading and agricultural intensification act as non-stationary drivers of lake water quality. Nationally, studies conducted by Vant and Huser (2000) and Vincent et al. (1984) highlight the lack of timeframe covered by conventional monitoring. Crucially, measurements of baseline lake health provide a reference point for how lake ecosystems respond to landuse change, better informing future water quality management.

## 2.4 AGRICULTURAL POLLUTANT MITIGATION: RIPARIAN PLANTING

Two sources of agricultural pollution impacting freshwater bodies have been internationally recognised: point and non-point (diffuse) sources (Grant et al., 1996; Kronvang et al., 1997; Ulén & Mattsson, 2003; Chapman et al., 2005; Heathwaite et al., 2005; Nelson et al., 2005; Chardon & Schoumans, 2007). Point source pollution refers to identifiable locations of pipe or drain discharge of wastewater and/or sewage. Conversely, non-point pollution arises from over or under land flow, where landuse pollutants and nutrients leach into freshwater bodies (Peirce, 1998).

The majority of agricultural nutrients enter water systems through surplus runoff (non-point source) as it flows over or under farmland and hence is a non-point source pollutant

(Schoumans et al., 2014). One of the most significant pathways of P and N loss in hilly and mountain areas occurs when erosion detaches soil particulates during overland flow. Conversely, leaching and artificial drainage are key nutrient pathways in flatter or less hilly areas (Chapman et al., 2005; Chardon & Schoumans, 2007; Schoumans et al., 2014). Until the early 1990s, the majority of attention was focused on easily identifiable point-source pollutants. Recently, research and investigations have shifted to diffuse sources, which have proven to be much more problematic to control and mitigate (Environment, 2004; Environment, 2013)

Due to a nutrient's ability to travel vast distances, the largest limitation lies in identifying and containing the source of both N, P and/or both pollutants. A total of 83 mitigation strategies were identified from literature (Schoumans et al., 2014) and grouped into the following eight categories:

Farming System:

1. Nutrient Management
2. Livestock Management

Field System:

3. Soil Management
4. Crop Management

Landscape and Hydrological System

5. Water Management within Agricultural Land
6. Landuse Change
7. Landscape Management

Ecological Management

8. Surface Water Management

Riparian buffer zones fall within the Landscape Management category and are a preferred mitigation strategy because of their low cost and natural approach. The concept of riparian restoration has appeared within New Zealand for over 30 years, with a core focus on buffering aquatic systems from agricultural landuse (Quinn et al., 1993). Typically, the riparian zone is recognised as the vegetated strip of land ranging along streams, rivers, lakes and wetlands (Gregory et al., 1991; Martin et al., 1999; Parkyn, 2000). Holistically, it is the interface between terrestrial and aquatic environments, providing a disproportionately sizeable impact on water quality relative to its quantity of surface area (Kauffman & Krueger, 1984; Osborne & Kovacic, 1993; Harding, 2009).

Two types of zones can be distinguished: 1) unsaturated, vegetated buffer zones, and 2) saturated, riparian wetlands (Schoumans et al., 2014; Hénault-Ethier et al., 2019). While academics cannot agree on the most effective composition (width and types of plant species) of riparian zones, general consensus can be found on the overall functions provided. These include: bank stabilisation, flood control, reductions in peak flow during floods, stream temperature regulation, stock exclusion (reducing bank trampling, defaecation in-stream, stock losses and waterborne illness), filtration of surface runoff reducing in-stream sedimentation and nitrification, provision of organic matter in-stream as a food source, and provision of habitat for fish spawning and adult phases of aquatic invertebrates (Kauffman & Krueger, 1984; Lowrance et al., 1984; Osborne & Kovacic, 1993; Collier et al., 1995; Fennessy & Cronk, 1997; Jorgensen et al., 2000; Parkyn, 2000).

Typically, riparian zones are considered to offer efficient protection against P bound particles, between 41-92% retention (Hoffman et al., 2009). However, a riparian buffer zone's effectiveness depends on many factors, such as the concentration of pollutants, slope angle, soil type and vegetation (Schoumans et al., 2014). Unsurprisingly, high concentrations of nutrient pollutants potentially stress and overload a riparian buffer zone's ability to absorb agricultural run-off (Anbumozhi et al., 2005). Likewise, steeper slope angles in lake catchments can lead to faster groundwater and overland flow movement, further exacerbating stress. The type of soil directly affects how quickly water can be absorbed (Parkyn et al., 2003). Soils that are high in clay are less permeable, while soils that are made up of large particles such as sand may drain water too rapidly for roots to effectively trap pollutants. Furthermore, soils that are more acidic have a better capacity to take up nitrogen through denitrification (Hawes & Smith, 2005).

Concerning vegetation, plant type, density and riparian width are crucial elements to consider when implementing a vegetated buffer zone (Parkyn, 2000). Indigenous plants typically reach maturity at a later stage, providing better options in terms of future sustainability as plants require and absorb more nutrients whilst they grow. Additionally, a higher plant density and diversity is more effective at capturing pollutants (Zhang et al., 2010). Concerning width, there is very little known about the optimal size for a buffer zone to be successful. Multiple studies, however, have shown that increased buffer width results in greater sediment and phosphate elimination (Parkyn, 2004; Chow, 2012; Weaver & Summers, 2014; Miller et al., 2016). Identifying the minimum width required for a zone to be self-sustaining is inconclusive from literature with examples ranging from



10 to 50 m (Davis & Meurk, 2001; Parkyn, 2004). Evidentially, a ‘one size fits all’ approach is insufficient with sites needing to be considered on a case by case basis (Quinn et al., 2001). As a result, riparian approaches should consider the aims of the planting, channel width, slope, vegetation type and the local surrounding environmental landuse that addresses the immediacy and overall cause of the pollution (Collier et al., 1995; Quinn et al., 2001; Reeves et al., 2004).

Timescales are also imperative when contemplating riparian mitigation (Collier et al., 1995). The results of which are time consuming and never immediate, requiring a long-term perspective and investment (Schoumans et al., 2014). In particular, water quality has been observed to worsen prior to improvements (Williamson et al., 1992; Ellison et al., 2009), with a need for realistic expectations and targets to avoid disappointment and unbiased assessment. There is, however, uncertainty about the true effectiveness of riparian planting mitigation as the majority of data comprises short-term studies that do not cover the full timescale of riparian growth or water quality degradation (Schmitt et al., 1999; Liu, Zhang, et al., 2008). Historically, the majority of riparian based studies within New Zealand focus on comparisons between streams or lakes with already established riparian zones against those where there are none. Focus needs to be shifted to consider the changes in lake ecosystems pre, post and during riparian establishment to truly understand the temporal relationship in a riparian zone’s ability to mitigate pollutants. Through paleoenvironmental reconstruction it may be possible to assess the efficacy of riparian planting in promoting lake health at a singular site across the pre- and post-planting spectrum.

## **2.5 USE OF PALEOLIMNOLOGY**

Paleolimnology and paleoenvironmental reconstructions are recognised tools that inform future conservation management by quantifying how current lake ecosystem and water quality diverge from pre-human baselines and environments (Zhao et al., 2006; Rawlence et al., 2014; Lyver et al., 2015; Waters, 2018). Constantly deposited over time, sediments preserve measures of lake life, landuse and water quality, equivalent to centuries of monitoring (Smol, 1992). Examining these sediments can potentially provide detailed documentation and chronology of a lake’s history, offering insight into the relationship between drivers of change (both natural and anthropogenic) and measures of lake ecosystem response simply not possible through current monitoring (Davidson & Jeppesen, 2013; Hilt et al., 2017).

### 2.5.1 DRIVERS OF LAKE ECOSYSTEM AND WATER QUALITY CHANGE

The use of paleolimnological data to identify baseline conditions largely depends on the extent to which past human landuse can be separated from natural forcing within lake sediments (Leavitt et al., 2009; Bennion et al., 2011; Mills et al., 2017). Pollen grains and plant macrofossils are one example of a proxy within lake sediments that provide insight into anthropogenic impacts on the land. Typically, pollen and spores are dispersed by wind or insects, with the surplus that is not used in reproduction accumulating in environments such as lake sediments (Seppä & Bennett, 2003). The overall taxonomic assemblage of pollen is therefore often reflective of the surrounding vegetation of a lake, with specific indicator taxa likely representative of anthropogenic influence (Birks et al., 1976; Manny et al., 1977; Sanger, 1988; Chen & Wu, 1999; McGlone & Wilmshurst, 1999d).

Furthermore, charcoal peaks potentially distinguish between periods of natural and anthropogenic burning for land clearance, complemented by changes observed in pollen records (taxa such as *Pteridium*) (Patterson III et al., 1987). An increase in the magnitude and frequency of charcoal in stratigraphic records linked with changes in recognised indicator pollen (such as exotic taxa *Pinus spp* and *Salix*) are considered strong evidence for both Polynesian and European anthropogenic influence within New Zealand (McGlone, 1983a; Bussell, 1988; Page et al., 1994; Elliot et al., 1998; McGlone & Wilmshurst, 1999d).

Charcoal fragments also provide insight into the proximity of landuse through the identification of past fire regimes at local and regional scales (Charles et al., 1987; Empson et al., 2002; Cohen et al., 2005). Charcoal is typically split into two groups based on particle size: microscopic (<125 µm) or macroscopic (>125 µm) charcoal (Turner et al., 2004). Microscopic charcoal travels farther due to its smaller size and is therefore used as a proxy for regional fires (McGlone & Wilmshurst, 1999d; Turner et al., 2004), while macroscopic charcoal travels shorter distances due to its larger size and is used as a proxy for local fires (Turner et al., 2004). The size of charcoal fragments offers a unique insight into the likely proximity of nutrient and sediment flux into lake systems, consequently driving lake ecosystem change.

Further insight into the nature of sedimentation and erosional material entering a lake can be provided through chemical and mineralogical analyses such as carbon to nitrogen

ratios (C:N), sediment grainsize and core density measurements. C:N ratios indicate the origin of organic material within a lake, whether it be terrigenous in nature or the result of in-lake productivity (Thornton & McManus, 1994; Kaushal & Binford, 1999). Changes in grainsize and density (often informed by computerized tomography (CT)) are indicative of changes in the relative abundance of terrigenous and allogenic sediment and consequently can provide a proxy for erosion in the lake catchment (King et al., 1982; Håkanson & Jansson, 1983; Kashiwaya et al., 1987). Largely recognised as potential drivers of lake ecosystem and water quality change, changes in sedimentation rate are most often associated with the abundance of terrigenous sediment (Heathwaite, 1994; Davies et al., 2005). Sedimentation rates increase when there is more erosion in the lake catchment. Alternatively, if sediment remains primarily autogenic then increases in sedimentation rate may reflect an increase in productivity.

Another element found within lake sediments, cadmium bioaccumulates in soils as a result of super phosphate application and hence may provide a proxy for fertilisation, and agricultural intensification. While increases of cadmium within soil as a result of superphosphate fertilisation is widely documented (Taylor, 1997; Gray et al., 1998; McDowell et al., 2013), within lakes, however, cadmium measurements are primarily interpreted as contamination of heavy metals within surface sediments (Roberts, 1981; Sindayigaya et al., 1994; Sander et al., 2013; Foote, 2014; Salmanzadeh, 2017; Taylor et al., 2017). This thesis therefore aims to investigate the use of cadmium as a proxy for superphosphate application downcore, evaluating the element's capability to characterise historical intensification of agriculture within Lake Nganoke's catchment.

### 2.5.2 MEASURES OF LAKE WATER QUALITY AND ECOSYSTEM CHANGE

Measures of lake ecosystem and water quality within paleo records have traditionally involved diatoms and chironomid analyses, as opposed to the novel approach of Environmental DNA (eDNA) proposed within this thesis (Hall et al., 1999; Sayer et al., 1999; Langdon et al., 2006). Whilst time consuming to conduct, chironomid surveys are considered one of the best indicators of a lake's overall ecosystem health. Chironomid assemblages can be used to assess salinity, water depth, aquatic productivity, oxygen levels, lake acidification, pollution and temperature (Svensonn & Leonardson, 1996; Lotter et al., 1998; Rosén et al., 2001). Similarly, diatoms are also micro-organisms that can be used to reconstruct changes within a lake's internal pH and nutrient loading

(Bradbury, 1971; Smol, 1985; Frey, 1988; Lotter et al., 1998; Shumate et al., 2002; Bjerring et al., 2009).

A relatively new technique, eDNA from sediment has been used to quantify historical abundance of organisms, including bacterial communities, microbes, aquatic macrophytes, and invertebrates. As certain groups react differently to outside factors such as light, temperature and nutrient conditions, the abundance of bacteria, for instance, can be indicative of environmental conditions and changes within a lake or catchment (Pedersen et al., 2013; Thomsen & Willerslev, 2015; Lim et al., 2016). Traditionally, bacteria have been identified using time consuming microscope techniques not suitable for paleo-reconstruction from sediment cores. Recent advancements, however, in high-throughput DNA sequencing (HTS) technology and the establishment of taxonomic reference databases now allows eDNA to be used to quantify bacterial communities from sediment cores (Parducci et al., 2017). The 16S ribosomal RNA (16S rRNA) gene is now a widely used genetic marker for bacterial taxonomic identification and is perfect for classifying ecosystem changes archived by lake sediments (Shaw et al., 1999; Domaizon et al., 2017; Tse et al., 2018).

Bacterial taxa also have the potential to reflect the biotic or abiotic state of lake ecosystems and therefore provide evidence for the impacts of environmental change (De Cáceres et al., 2016). Often defined as ‘indicator taxa’, these subsets of the bacterial community provides an alternative to sampling the entire community and are particularly useful in long term monitoring (De Cáceres et al., 2012). Issues arise over how to assess and define taxa most representative a lake ecosystem, as each lake environment has the potential to be completely different. Traditionally, paleolimnology studies target taxa for which well-established responses to certain conditions or clear functions have been established by previous research.

Often limiting the habitat for other organisms, cyanobacteria are an example of a recognised indicator taxa for poor water quality (Catherine et al., 2013; Chrapusta et al., 2015). Members Cyanobacteria are known bloom formers, which live in lake surface waters often blocking light penetration to other organisms that depend on photosynthesis. Furthermore, Cyanobacteria produce dangerous toxins that are measured attributes in the Ministry for the Environment National Policy freshwater management scheme (NPS-FM) for human health and recreation; with high levels of toxins often signifying a degradation

of water quality (Greenfield et al., 2014). Cyanobacteria as an indicator taxon are therefore often representative of freshwater degradation, likely the result of increasing nutrients stimulating growth (Fay, 1992).

Other documented indicator taxa include Atribacteria, Nitrospirae and Proteobacteria. Atribacteria thrive in anaerobic environments and can therefore be indicative of increased stratification within a lake and reduced oxygen via increasing productivity (Nobu et al., 2016). Nitrospirae have known nitrate oxidising and sulphate reducing members and may provide a nursery effect for other nitrogen-limited bacteria (Kachiuru, 2013; Schneider et al., 2013). Proteobacteria include a wide variety of pathogenic genera, many of which are free living and/or are responsible for nitrogen fixation (Young, 1992; Van Dommelen & Vanderleyden, 2007). Examples of these genera include the class Alphaproteobacteria that can grow at very low levels of nutrients (Smith et al., 2013), Deltaproteobacteria that predate on other bacteria (Schwarz et al., 2008) and Gammaproteobacteria that have important denitrifying capabilities (Cramer et al., 2011).

The ability of eDNA reconstruction also has the potential to provide insight into historic stocking rates. Traditionally, reconstructions of faecal pollutants within lake sediments have been restricted to Bacteroides markers associated with ruminant animals (ruminant bacteria) such as sheep and cows. While ruminant presence through the use of polymerase chain reaction is recognised, quantification of stocking rates are not (Bofill-Mas et al., 2006; Hamza et al., 2011; Rusiñol et al., 2016). Whilst relatively unstudied, there is potential for Digital Droplet Polymerase Chain Reaction (ddPCR) to quantify concentrations of ruminant gut bacteria found within lake sediment (Hindson et al., 2011). Through the use of Poisson distribution-based statistics, the target DNA's concentration (in this case ruminant faecal biomarkers) can be calculated within a sample (Hindson et al., 2011). This provides a quantified 'number of copies' per sample that could potentially demonstrate changes in DNA concentration downcore. The potential therefore is to use replications of ruminant gut flora DNA as a proxy for changes in stocking rates within Lake Nganoke's catchment.

Another significant component of lake ecosystems and water quality, phytoplankton are the foundation of aquatic productivity, providing integral environmental services through carbon/nitrogen fixation and oxygen production (Yackulic, 2017). Chlorophyll *a* exists in almost all photosynthetic organisms as the primary pigment used to absorb light in the

violet, blue and red wavelengths of the visible portions of the photoelectric spectrum (Liu, Lauterbach, et al., 2008). Consequently, colour spectrometry from hyperspectral imaging of sediment cores have been widely used to provide a high resolution for chlorophyll a in sediments. Provided that the chlorophyll a can be attributed to algae then the scanning provides a high resolution proxy for autogenic productivity. Evidence within literature can be seen most notably in measurements of the reflectance trough related to Chlorophyll a and its diagenetic products at 660-670 nm (Rein & Sirocko, 2002; Grosjean et al., 2014; Butz et al., 2017).

Internal nutrient loading plays a vital role in promoting excess biological growth within lake systems (Burger et al., 2008). Primary productivity in lake systems is strongly limited by nutrient availability, with the role of phosphorous (P) in controlling lake water quality and ecosystem growth well established (Giresse et al., 1994; Wetzel & . 2001; Mankin et al., 2007; Zhang et al., 2015; Smith et al., 2016). P loading within lakes can be split into six fractions (*NH<sub>4</sub>Cl-P*, *BD-P*, *NaOH-rP* and *NaOH-nrP*, *HCl-P*, *Res-P*, *Total P*), all of which have the potential to be mobile during different conditions (Penn et al., 1995). However, the assessment of P fluctuations within a lake system is often complicated as the mobility of fractions within sediments may obscure the role nutrient loading plays in controlling lake health. While measurements of P loading within surface sediments is well documented (Jensen et al., 1992; Christophoridis & Fytianos, 2006; Waters, 2016), the degree to which P fractions can be applied down core as an indicator of paleo nutrient cycling is largely unknown (Zhu et al., 2013; Lü et al., 2016). Generally considered the least mobile fraction, residual P (Res-P) is the most likely fraction to be indicative of nutrient cycling at the time of sediment deposition because it is unlikely to be mobile within the sediment pile. As such, this thesis pioneers the novel approach of applying the phosphorous residual fraction down core as a proxy for paleo-primary production.

### 2.5.3 THE IMPORTANCE OF DATING LAKE SEDIMENTS

The strength of a paleo reconstruction greatly depends on the accuracy of sediment chronology (Zolitschka, 1991; McGeehin et al., 2001; Gamrod, 2009). Several methods can be used to establish a time scale, with the most common including the radioactive decay of lead (<sup>210</sup>Pb) (half-life of 22.3 yr.) (Appleby & Oldfield, 1978a; Von Gunten & Moser, 1993) and conventional radioactive decay dating of Carbon (<sup>14</sup>C) (t<sub>1/2</sub> = 5730 yr.) (Hajdas et al., 1995). Sedimentary records can therefore be correlated by authenticating

sediment profiles with absolute dates, with further synchronisation provided through records of pollen stratigraphy.

$^{210}\text{Pb}$  is widely used to provide chronology over the last century (Oldfield & Appleby, 1984; Evans, 1991; Arnaud et al., 2006). The Constant Rate of Supply (CRS) model is the most commonly used method of  $^{210}\text{Pb}$  dating because its assumptions are met in the widest variety of depositional settings (Appleby, 2008). While the CRS model assumes a constant rate supply of fallout  $^{210}\text{Pb}$ , allowing for sedimentation rate to vary through time. Isochrons derived from  $^{137}\text{Cs}$  or biostratigraphy are used to validate  $^{210}\text{Pb}$  dating models.

Radiocarbon dating ( $^{14}\text{C}$ ) is the most widely used dating technique for samples younger than c.50,000 years that contain sufficient organic matter (Björck & Wohlfarth, 2002; Howarth et al., 2013). Dating using this technique measures the ratio of remaining  $^{14}\text{C}$  to the unchanged  $^{12}\text{C}$  to see how long it has been since the material's source died.  $^{14}\text{C}$  samples require calibration because concentrations of  $^{14}\text{C}$  in the atmosphere have varied through time. Samples post AD 1950 often referred to as 'bomb spike carbon' due to increases in  $^{14}\text{C}$  concentrations in the atmosphere post nuclear testing (Broecker et al., 2001). Conventional and bomb spike  $^{14}\text{C}$  therefore require separate calibration standards as outlined further in Section 5.3

### **3 CHAPTER 3: RESEARCH HYPOTHESES AND OBJECTIVES**

#### **3.1 RESEARCH HYPOTHESES**

This thesis aims to reconstruct environmental change at Lake Nganoke using paleolimnological methods and examine anthropogenic impact on the lake ecosystem and water quality over the last 1000 years. Specific attention is given to the ability of riparian zones to buffer lake ecosystems against the deleterious impacts of nutrient rich runoff from land use intensification. To achieve this aim, sediment cores from Lake Nganoke were studied to address the following research hypotheses:

- 1) Anthropogenic landuse intensification in Lake Nganoke's catchment has resulted in ecosystem change and a stepwise decrease in water quality from a pre-human baseline, through Māori subsistence, European agricultural and intensification.
- 2) The riparian zone around Lake Nganoke has buffered lake water quality against intensification of agriculture.

#### **3.2 OBJECTIVES**

The testing of these hypotheses require four objectives to be achieved. These are:

1. Use historical information and paleo environmental reconstruction to develop a chronology of pre-human, Māori, European and agricultural intensification in Lake Nganoke's catchment. Specifically, this objective will include a high-resolution (sub-decadal) record of agricultural intensification over the last 180 years.

*Objective one will involve pollen, charcoal, ICP AES, ddPCR and radiometric dating approaches.*

2. Develop a paleo reconstruction of lake water quality, ecosystem evolution and nutrient loading during the above identified phases within Lake Nganoke.

*Objective two will involve eDNA, hyperspectral scanning, phosphorus chemistry and ICP-MS approaches*



3. Assess the relationship between drivers of landuse change and measures of lake water quality and ecosystem integrity over the history of Lake Nganoke.

*Objective three will involve a nonmetric multidimensional scaling (NMDS) based on Bray-Curtis distance and applied in RStudio*

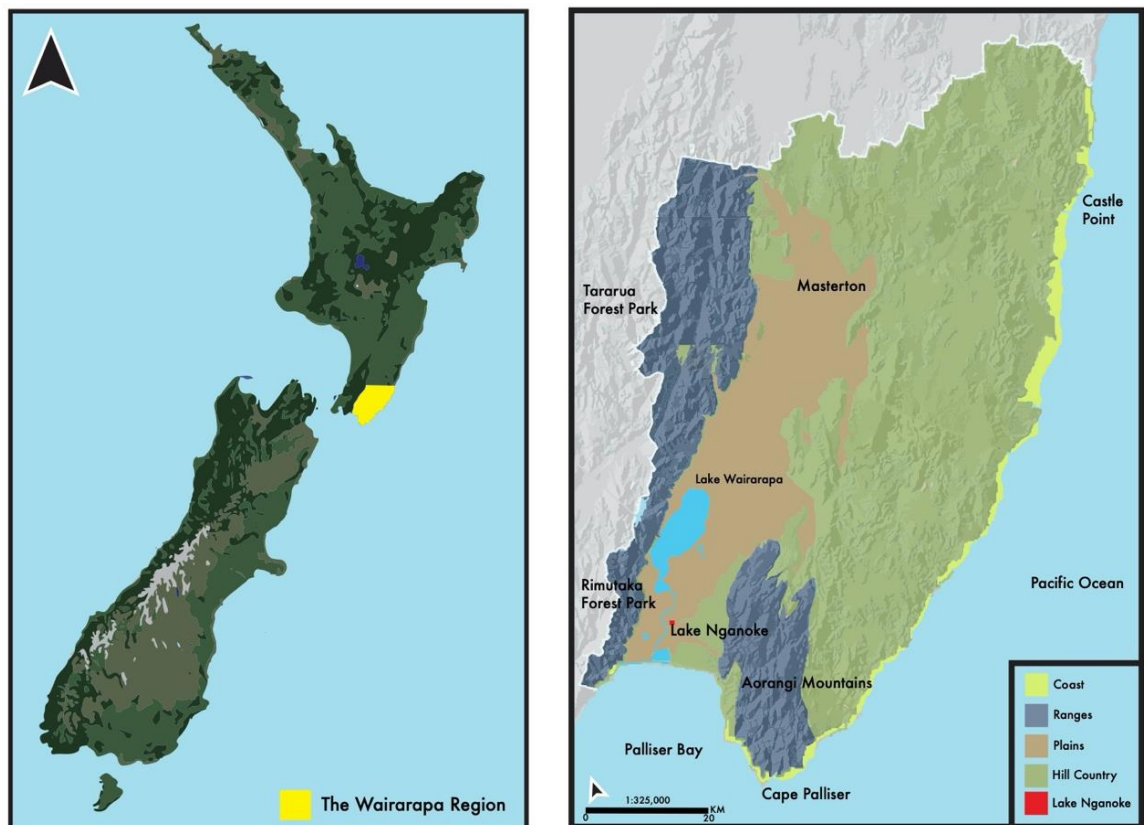
4. Investigate the establishment and development of riparian planting as a mitigation strategy against agriculturally rich nutrients.

*Objective four will use historical aerial photographs and records of water quality identified within objective two*

## 4 CHAPTER 4: STUDY SITE

### 4.1 PHYSIOGRAPHIC PROPERTIES

Encompassing an area of 8,423 km<sup>2</sup>, the Wairarapa is a region known for its wide valleys, rolling hill country and rugged coastline. The region is located on the South Eastern Corner of the North Island, bordered by the Pacific Ocean in the East, the Tararua district in the North and the Rimutakas and Tararua Ranges in the west (Figure 3). The term ‘Wai’ra’rapa’ translates to “Land of Glistening Waters”, with the region aptly named after the waters of Lake Wairarapa, situated at the bottom of the Wairarapa Plain (Fallon, 2015).



**Figure 4.1: Location of the Wairarapa region in relation to New Zealand (left) and associated landforms (right).**

Adapted from Miskell (2010)

Historically, the southern end of the region was inundated by water with Lake Wairarapa once a shallow marine extension of Palliser Bay (Leach & Anderson, 1974; Kamp, 1992; Hines et al., 2013). At the end of the postglacial marine transgression, maximum infiltration inshore of the extension was reached sometime during ~7 kyr BP (Leach & Anderson, 1974; Heath, 1979). Subsequently, the lower Wairarapa valley developed into an extensive estuary with the Ruamahanga river prograding south until 3.5-3.1 kyr. BP. At this time, the river mouth moved north-west, settling into the main water body that

would eventually become Lake Wairarapa, a freshwater riverine-lacustrine complex completely separate from the sea (Leach & Anderson, 1974; Leach, 1984; Trodahl et al., 2016b). Lake Wairarapa is the largest wetland system in the lower North Island, and combined with its associated wetlands they are now considered to be of national and international importance (Myers et al., 2013; Aotearoa, 2019).

The Wairarapa area has four distinct landforms: Torlesse greywacke ranges (the Tararua/Rimutaka ranges on the western periphery and the Aorangi ranges on the south-eastern). The ranges form the highest relief in the area, measuring some 1500 m in height above sea level (Guggenmos et al., 2011). Separated from the Wairarapa Coast by hill country, extensive plains run from the north of Masterton to Palliser Bay (Figure 4.2). Occupying roughly 20% of the Wairarapa, the plains consist of thick late Quaternary alluvial deposits originating from the surrounding hills and ranges (Begg et al., 1996). Broadly subdivided into two, the northern plain comprises flat, gently undulating to rolling land, while the southern plains are dominated by the lake and surrounding wetland (Miskell, 2010). The majority of Wairarapa townships (Featherston, Greytown, Carterton, Masterton) are located within the northern plain. In comparison to the ranges, the northern and eastern hills reach heights of 1,000 m above sea level and consist of an assortment of Cretaceous and Cenozoic sandstone, mudstone and limestone (Lee et al., 2002).

The area is tectonically active with predominantly strike slip faulting in the region, including the Wairarapa fault running along the western side of the plains. The Wellington and Wairarapa region are two of the most earthquake-prone regions in New Zealand with four recorded quakes greater than magnitude 7 since record keeping began (Rafferty, 2009; King, 2015). Furthermore, the Wairarapa fault has produced the largest earthquake (magnitude 8.2 in 1855) in New Zealand history. This greatly affected the region; the land rose up to 6 m in most places within the Wairarapa, while lateral movement was up to 18 m in some places (Lee & Begg, 2002).

The soils of the Wairarapa can be classified according to their potential for land use capability (Table 1) (A.K. Booth, 1969; Fallon, 2015). Class 1 soils are defined as free draining, easily cultivated, silt loam soils on river flats and terraces. These versatile soils are mainly used for intensive agriculture and only have limitations of flooding near major rivers. Class 2 soils are limited by drainage, occur predominantly on terraces and plains

and are highly suited for agricultural landuse. Class 3 are soils either Orthic Brown or Sand (low water holding capacity) or Rendzic Melanic (limestone) soils all of which are highly permeable. Class 4 and 5 are the predominant land type within pastoral hill country. Formed from siltstones, Class 4 are typically defined as brown silt and sand in regions of moderate to well distributed rainfall. Class 6 soils are described as steep land and, mainly occurring in the foothills of the Aorangi and Tararua Ranges. Highly erosive, these soils are not suitable for any form of agricultural landuse. Within the region encompassing Lake Nganoke, the landuse is predominantly Class 2, and supports extensive sheep and beef farming.

**Table 1: Landuse classification of the Wairarapa region.**

Adapted from (Fallon, 2015)

<b>Landuse Capability Class</b>	<b>Major Limitation</b>	<b>Landform</b>	<b>Major Landuse</b>	<b>Soil Order</b>	<b>Area (Hectares)</b>
1	Flooding	Plain	Horticulture, cropping, dairying	Recent	36 000
2	Drainage	Plain and terrace	Dairying, dairy support, and lamb cattle, finishing, viticulture	Grey Pallic	140 000
3	Moisture	Terrace, rolling land, dune	Dairying, dairy support, and lamb cattle, finishing, cropping, viticulture	Orthic Brown, Sand, Rendzic Melanic	24 000
4	Slight erosion and moisture	Rolling to easy hill	Dairy support, Sheep and cattle breeding and finishing	Brown	240 000
5	Moderate erosion and serious moisture	Easy to steep hill	Sheep and cattle breeding	Brown	220 000
6	Serious erosion	Very steep hill	Exotic forestry, native bush	Raw	160 000

Prior to human arrival, the Wairarapa vegetation mostly consisted of conifer-broadleaf and beech forest (Hill, 1963b; Wardle, 1991), with elevations above the tree line dominated by scrub and tussock (McGlone, 1989b). Burnoff from Polynesian settlers and natural fires left large areas of grass, fern and scrubland in the south and east (McGlone,

1989b; McFadgen, 2003a). European settlement, especially in the early years post AD 1844 saw large parts of open country cleared by hand for agriculture. As a result, vast areas of the landscape were converted to exotic pasture (*Poaceae family*), with plants such as sweet vernal grass (*Anthoxanthum odoratum*), timothy (*Phleum pratense*), yorkshire fog (*Holcus lanatus*), cocksfoot (*Dactylis glomerata*) and couch (*Poa spp.*) introduced (Hill, 1963b). Rapid intensification periods from early to mid 1900s resulted in further deforestation of lowlands and eastern uplands pastures. A large majority of this consisted of Australian sheep burrs, *Acaena ovina* and *Xanthium spinosum* as well as the common thistle (*Cnicus lanceolatus*) and sheep sorrel (*Rumex spp.*) (Wardle, 1991). The land surrounding Lake Nganoke currently consists of agricultural pasture and a mix of exotic and native tall trees.

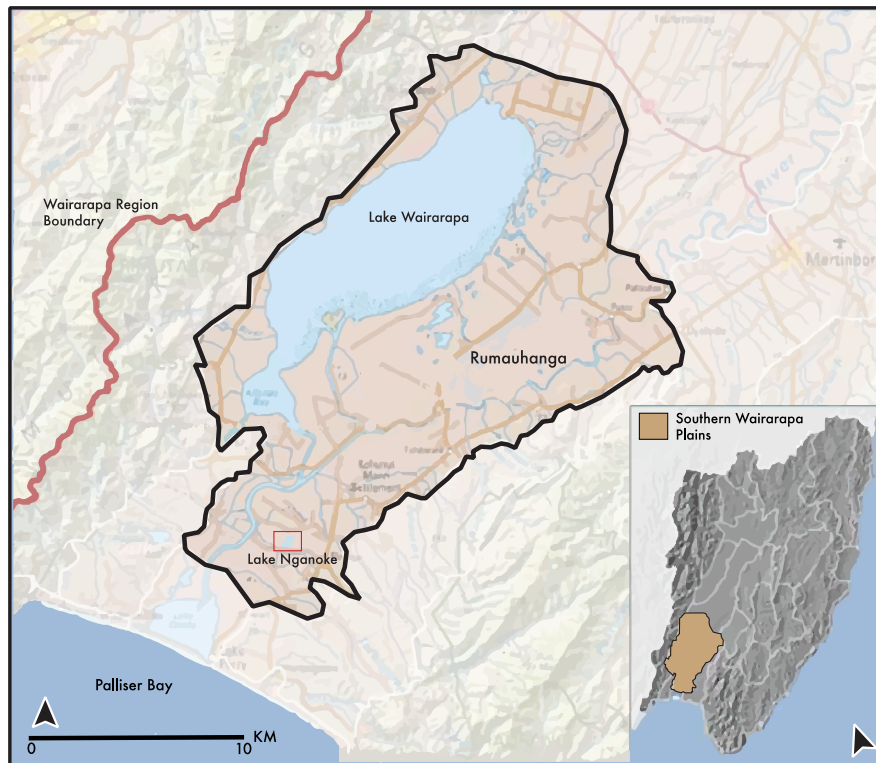
Typically, the Wairarapa is categorised as having cool wet winters and stable dry summers, with typically moderate to high winds experienced during spring and summer. Average temperatures range from 20-28°C in summer, keeping consistently below 10°C in winter (A.K. Booth, 1969; Hawke & Summers, 2003). Annual rainfall varies from 800-1,200 mm, with the rainfall more variable during the summer months and generally of higher intensity throughout winter (Guggenmos et al., 2011). Regionally, the western areas are wetter than the east due to orographic lift from the Rimutakas and Tararua ranges.

The Wairarapa is made up of many catchments, with headwaters in the low to mid altitude of Rimutakas, Tararua and Eastern Wairarapa hills. Generally speaking, these are all small river systems with high flows in winter and low flows in summer. Water is largely provided to these systems by rainfall and snowmelt, draining towards the coast. A small majority of these catchments are also supplied from underlying groundwater systems, subsurface and soil flows.

## **4.2 SOUTHERN PLAINS**

The Southern Plains (which Lake Nganoke is located within) are characterised by low-lying flood plains, the Ruamahanga River, Lake Wairarapa and the Kahutara dunes. The area is bordered by Lake Onoke to the south, Lake Ferry road to the east, and SH53 to the north (Figure 4.2). The general land around Lake Wairarapa is periodically flooded. Originally permanently water-logged, this land once supported an extensive wetland;

however this was significantly diminished after the Ruamahanga River diversion (1968) and the flood control barrage (1974) (Miskell, 2010).



**Figure 4.2: Location of the Southern Plains, New Zealand**

Adapted from Miskell (2010)

The surrounding area comprises fertile plains and river terraces. The predominant land type is classified as a plain, with the main landuse being horticulture, cropping and dairying. The soil type is fine grain silt, sand and gravel deposits which have likely travelled downstream, originating in the ranges. The prevalent land cover is pasture, with frequent plantings of poplar, dense plantings of willow and the scattered remnants of kahikatea and cabbage trees (Miskell, 2010). The land experiences moderate rainfall (1000 mL) and is dominated by sheep and beef farming and large-scale dairying. Typically, the Southern Plains have a low density of settlement and are extensively grazed by large farming systems (Miskell, 2010).

### **4.3 HUMAN SETTLEMENT AND AGRICULTURAL DEVELOPMENT**

This section reviews evidence and past research concerning Polynesian and European settlement within the Wairarapa region. Specifically, four zones are defined in order to provide greater insight into findings within Chapter 6. While not directly indicative of settlement at a local scale, the following evidence is likely to be representative of Lake Nganoke's catchment.

In the Wairarapa region, anthropogenic modifications likely began with moderate deforestation of the lowlands by Polynesian settlers. While it has been established that Polynesian settlement occurred within New Zealand c. 700-800 calibrated years before AD 1950 there is evidence that the settlement of Wairarapa may have not occurred until later (~ AD 1350 - 95% CI: AD 1280 - 1420)(Anderson, 1991; McFadgen, 2003b; McIntyre, 2012). Dates produced through radiocarbon techniques consist of a charcoal tree root buried beneath a garden at Okoropunga (eastern coast of the Wairarapa). Furthermore, paleo reconstruction of Lake Pounui (west of Lake Wairarapa) uses pollen records to further date Māori arrival inland, estimating a later period of AD 1500 to 1600 (Cochrane, 2017). Polynesian arrival within the Wairarapa is therefore defined as a range between the maximum radiocarbon dates (AD 1300) and minimum pollen stratigraphy (AD 1600) i.e. ~AD 1450 (95% CI: 1156 – 1744); pre-human baselines are recognised as any date prior.

First European arrival within the Wairarapa (Palliser Bay) is estimated at ~AD 1770. Widespread settlement of the Wairarapa region did not commence until AD 1844 with the establishment of two of New Zealand's earliest sheep stations (A.K. Booth, 1969; Mair, 1972; Leach, 1981; Trodahl et al., 2016a). By 1854, Masterton, Carterton, Greytown and Featherston were all settled, (Hill, 1963a; Leach, 1981; Fallon, 2015). Cochrane (2017) further supports these findings, with pollen records indicating formal settlement of the Wairarapa valley in AD ~ AD 1825 (95% CI: 1789 – 2068). Large scale European settlement within the Wairarapa region is consequently accepted as 1850 which fits with historical evidence of formal settlement of the Wairarapa Valley in ~AD 1853 (Leach, 1981; Hill, 1963).

The fertility and ease of development for agriculture meant that the plains and lowlands were readily developed for agriculture and consequently extensively modified. In 1949, advancements in aerial topdressing saw the first arrangement for aerial fertilisation within NZ. A Wairarapa farmer, L.T Daniell, arranged for 127 tonnes of fertiliser (the first large operation within NZ) to be dropped on 11 properties (Wendelken, 1976). As a result, agriculture rapidly intensified with stocking rates lifted from 1-2 ewes per acre to a minimum of 3 (A.K. Booth, 1969). By AD 1969 2.75 million sheep and 50,000 milking cows were located within the Wairarapa. In current times, the Wairarapa is home to three million sheep, 230,000 dairy cattle and 257,000 beef cattle (Anastasiadis & Kerr, 2013a).

Agricultural intensification within Wairarapa is therefore defined by advancements in technology and dated AD 1950 to present.

Based on literature and historical accounts the timing of landuse zones within the Wairarapa are defined as:

**Pre-human:** pre 1450

**Māori:** 1450 to 1850

**European:** 1850 to 1950

**Intensification:** 1950 to present

These *a priori* zones will be used throughout the thesis and their validity for Lake Nganoke examined in the Sections 7.1.1, 7.2.1, 7.3.1 and 7.4.1.

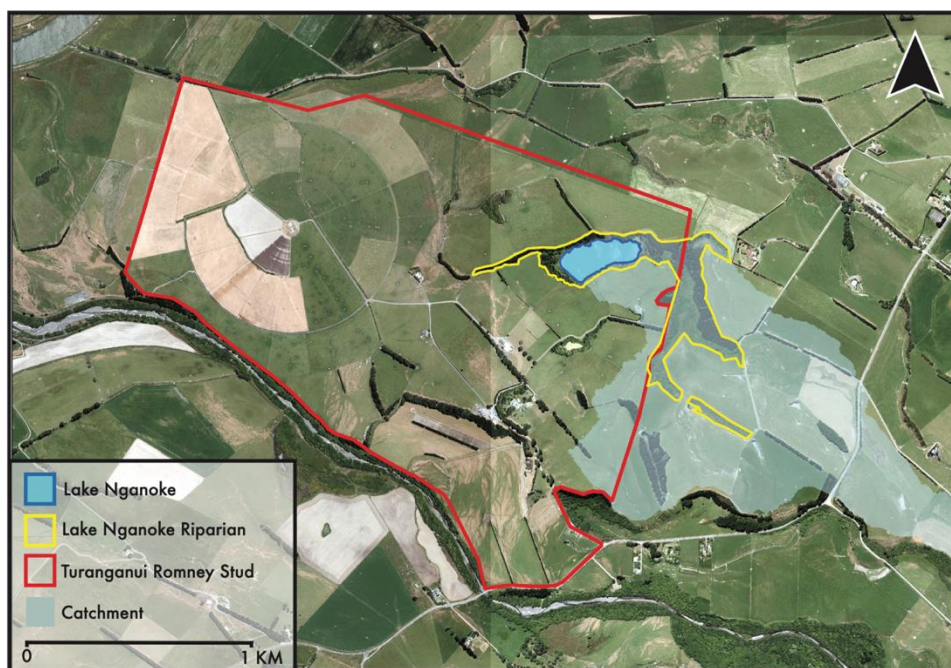
#### **4.4 LAKE NGANOKE**

Lake Nganoke is a small shallow lake on private land located in the southern end of the Southern Plains (41° 21' 20" S, 175° 11' 10" E) (Figure 4.3). It has a surface area of approximately 0.049 km<sup>2</sup> and a depth of no more than 2 m. The catchment of the lake largely consists of agricultural pasture used for farmland, covering approximately 1.5 km<sup>2</sup> to the south east of the lake. Currently, the lake is surrounded by a compact 5 m riparian buffer zone on the northern, eastern and western side. This is somewhat diminished on the southern side, however the catchment located to the east of the lake is extensively planted, with riparian thickness minimum of 20 m in some parts. To date, limited water quality analysis has been measured from the lake, making it an ideal candidate for the paleo approach.

Dominant species surrounding the lake include: *Cordyline australis*, *Phormium tenax*, *Carex secta*, *Typha orientalis*, *Cyperus ustulatus*, *Melicytus ramiflorus*, *Corynocarpus laevigatus*, *Pinus radiata* and *Salix sp.* (Department of Conservation, 2000).

The Turanganui Romney Stud is a fourth generational sheep and beef farm surrounding Lake Nganoke. Little is known of the land prior to the purchase on April 18<sup>th</sup>, 1907. Whilst the sale originally involved 6.84 km<sup>2</sup>, 4 km<sup>2</sup> was subdivided with a brother giving the Stud farm 2.84 km<sup>2</sup> (Figure 4.3).





**Figure 4.3: Lake Nganoke and surrounding land.**

Displayed is Lake Nganoke (blue), the riparian zone (yellow), Turanganui Romney Stud (Red) and the catchment (grey).

## **5 CHAPTER 5: METHODOLOGY AND METHODS**

### **5.1 METHODOLOGY**

Historical baselines can be used as the conceptual benchmark for lake ecosystem and water quality assessment and management (Battarbee, 1999; Duarte et al., 2009; Bennion et al., 2011). Furthermore, the assessment of landuse change prior to and after human arrival potentially provides a greater understanding of observed historical lake ecosystem and changes in water quality. Reconstructing the past environment of Lake Nganoke will therefore allow for the assessment of anthropogenic impact with respect to natural baseline conditions, producing information crucial to informing future mitigation strategies.

Paleolimnological methods are fast becoming recognised as an essential tool for evaluating past lake health internationally (Korsman & Birks, 1996; Battarbee, 1999; Bennion & Battarbee, 2007; Ruhland et al., 2008; Bennion et al., 2011), and nationally (Newnham, Lowe, & Matthews, 1998; McGlone & Wilmshurst, 1999d; Augustinus et al., 2006; Augustinus et al., 2012; Perry et al., 2012). This thesis will therefore make use of a multiproxy paleo-based approach to reconstructs the past environment of Lake Nganoke. The use of multiple proxies provides correlation of evidence concerning the history of landuse, water quality and ecosystem change. In this respect, the greater the consensus between a wide range of proxies, the stronger the argument.

To investigate research hypothesis one “Anthropogenic landuse intensification in Lake Nganoke’s catchment has resulted in ecosystem change and a sequential stepwise decrease in water quality from a prehuman baseline, through Māori subsistence, European agricultural and intensification” indicators of ecological health, catchment disturbance and human arrival will be evaluated. Specifically, a high resolution chronology will be created through the use of key radiometric dating techniques:  $^{210}\text{Pb}$ ,  $^{137}\text{Cs}$ , bomb spike  $^{14}\text{C}$ , conventional  $^{14}\text{C}$  and bio stratigraphic age constraints. Past landuse reconstruction will be informed through environmental Digital Droplet Polymerase Chain Reaction, geochemical analysis and palynology. Previous lake ecosystem and water quality change will be characterised through the use of 16s ribosomal Polymerase Chain Reaction, hyperspectral imaging and residual phosphorous fractionations.

Research hypothesis two “The planting of a riparian zone around the lake has buffered lake water quality against intensification of agriculture” will be investigated using

indicators of water quality and lake ecosystem change through periods of identified growth and maturity of riparian planting. The timing of planting and growth to maturity will be constrained using historical aerial photographs and compared to measures of water quality and ecosystem change.

## **5.2 CORE RETRIEVAL AND SUBSAMPLING**

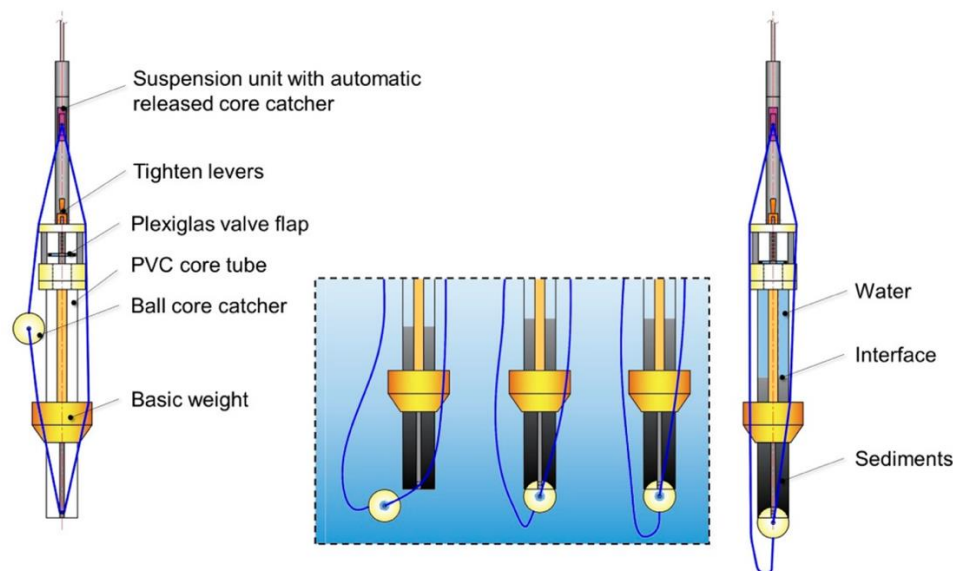
### **5.2.1 CORE RETRIEVAL**

Five sediment cores were retrieved from Lake Nganoke using a Uwitech gravity corer with a 2-m-long, 65-mm diameter polyvinyl chloride barrel (Figure 5.1) (UWITECH, 2019). The Uwitech corer was used because it is operated by hand, is easily transportable, and captures the upper 2 m of sediment within a lakebed, making it perfect for sampling relatively small and shallow lakes. Individual cores are cut to  $\leq 1$ -m subsections within the field for logistical reasons (handling, transport, splitter/scanner rails, etc.)

Cores were taken at the depocenter of the lake's basin (i.e. the southern end: 41°21'21.6"S, and 175°11'07.7"E : ~2m deep), as this should provide the most complete record (Figure 6). Lengths were collected in 2019 and are as follows:

- LC1U: 77 cm
- LC2U: 84c m
- LC3U: 93 cm
- LC4U: 89 cm
- LC6U: 92 cm

After collection the cores were chilled at 4°C in complete darkness until subsampling. Important to note, while LC5U was collected, this core was archived and therefore will not be included within this study.



**Figure 5.1: Structure and sealing sketch of the conventional UWITEC gravity corer.**  
Retrieved from Gong et al. (2019)

### 5.2.2 SUBSAMPLING

All five cores were sliced along the longitudinal plane, with the top two to three millimetres of one half carefully removed with a sterile spatula. Stratigraphic units within the cores were determined using a combination of visual logging (colour, texture, structure and core deformation) and computed tomography (CT) scanning. In order to minimise the effects of oxidation on the core, visual logs were completed directly after core splitting, with photographs also taken for later reference. All five cores were run through a medical grade CT scanner at Pacific Radiology Boulcott Hospital. The raw data was imported and processed using the ImageJ software platform (Schneider et al., 2012). Hounsfield units were converted into  $\text{g/cm}^3$  following the approach outlined in Reilly et al. (2017).

As the longest core, theoretically providing the greatest temporal extent, LC3U was selected as the master core to be sampled. However, due to the large amount of sediment required for the multiproxy approach, LC1U and LC6U were also sampled in order to complete the full range of proxy analyses. CT scans and hyperspectral scans were used to produce high resolution correlations between the sediment cores. Sample depths were transformed into master core depths using polynomial regressions developed using the inter-core correlations.

## 5.3 AGE-DEPTH MODEL

### 5.3.1 DATING

The sampling resolution for  $^{14}\text{C}$  and  $^{210}\text{Pb}$  was completed on LC1U and comprised two approaches in order to gain both a precise and broader resolution of dating within the lake. The first included a targeted approach at 1-cm intervals from 17-27 cm while the second broader approach included 2-cm intervals from 8-16 cm and 5-cm intervals from 28-84 cm. Macros used for dating were visually identified after core splitting or during proxy sub-sampling. Only three of the samples retained sufficient mass for dating after pre-treatment.

#### *Radiocarbon ( $^{14}\text{C}$ ) dating*

Accelerator mass spectrometry (AMS) radiocarbon dating was carried out at the Rafter Radiocarbon Laboratory, GNS Science, on one seed capsule, and two leaf fragments collected from Lake Nganoke cores (Table 2). Once collected, samples were rinsed in a 90- $\mu\text{m}$  and 150- $\mu\text{m}$  sieve stack using distilled water, before being inspected under a microscope to identify the sample and remove contaminants. Samples were then placed in an ultrasonic bath to remove any remaining contaminants, dried in a 50°C oven overnight and weighed. Acid/alkali/acid pre-treatment, using 0.5 M HCl and 1.0 M NaOH, was carried out on plant material to isolate the humin fraction.

While sample pre-treatment, chemical treatment, graphitisation and measurement are the same for both conventional  $^{14}\text{C}$  dating and bomb spike dating, calibration is not. Conventional  $^{14}\text{C}$  dates were calibrated using the SHcal13 curve (Reimer et al., 2013), whereas bomb spike dates were calibrated with SH1-2 (Hua et al., 2013).

#### *Lead $^{210}$ and Caesium $^{137}$*

Dating of  $^{210}\text{Pb}$  relies on the estimation of the residual radioactivity arising from the presence in the sediments of unsupported  $^{210}\text{Pb}$  derived from 'rain-out' onto the surface of the water body beneath which the sediments have accumulated (Appleby & Oldfield, 1978b; Fitzpatrick et al., 2003).

The sampling strategy comprised two approaches in order to gain both a precise and coarser resolution of dating within the lake. The first included a targeted approach with the aim of definitively identifying the 1954 Caesium $^{137}$  ( $\text{Cs}^{137}$ ) horizon within the same samples as  $^{210}\text{Pb}$ . The second takes a more complete approach capturing measurements

throughout the core. Particular depths of interest were informed through a previous bomb spike date of 1958 and palynology giving a rough estimate of European settlement around 1840 (defined by the presence of *Pinus*).

Once samples were collected, radionuclide measurements were made using gamma spectrometry at the Institute of Environmental Science and Research (ESR), Christchurch, New Zealand. Activities were quantified using a gamma counter with a high-purity germanium well detector. Activities are reported in Bq.kg<sup>-1</sup> and the uncertainties are based on the combined standard uncertainty (uc) multiplied by a coverage factor (k) = 2 (providing a level confidence of 95%) as described by WHO (1995). Core chronology from <sup>210</sup>Pb were calculated using the constant rate of supply (CRS) model in R by means of the paleolimbot package (Dunnington, 2019). Assuming that the supply of unsupported <sup>210</sup>Pb to the sediment is the same for each time interval, the CRS model is calculated as follows:

#### Equation 1

$$t_z = \lambda^{-1} - 1 \ln(A_0 A_z^{-1})$$

Where (t) is the age of any interval (z), ( $\lambda$ ) is the <sup>210</sup>Pb decay constant (0.031 year<sup>-1</sup>), ( $A_0$ ) is the integrated activity from the surface and ( $A_z$ ) is the integrated activity to ( $A_0$ ) (Appleby & Oldfield, 1978b).

### 5.3.2 AGE MODELS

An age-depth model was constructed using the P Sequence prior model in OxCal 4.3 (Ramsey, 2000; Ramsey, 2008). The P Sequence prior treats sediment deposition as a Poisson process, where the prior permits accumulation of a sedimentary sequence to fluctuate around a constant rate (Vandergoes et al., 2018). The model was parametrised using a variable event thickness constant (k), which is a loosely defined prior that ranges uniformly over two orders of magnitude above and below 1 cm<sup>-1</sup> (Bronk Ramsey & Lee, 2013).

## 5.4 LANDUSE RECONSTRUCTION

### 5.4.1 PALYNOLOGY

The pollen stratigraphy was used to identify changes in regional and local vegetation as well as fire histories through time (Section 5.4.2) Palynology sampling was completed prior to this thesis and consisted of 16 samples from LC3U. Pollen was provided by Xun

Li (GNS), who followed an adapted preparation method outlined in Faegri et al. (1989). A relative pollen diagram was produced by normalising taxa counts by the dryland pollen sum, defined as the sum of pollen grains from tall trees, small trees & shrubs and herbs. Fern spores and wetland taxa are not included but the percentage of fern spores and wetland taxa were calculated against the dryland pollen sum.

The main processing steps undertaken included: acid wash (remove calcium), acetolysis, heavy liquid flotation (remove mineral fraction), filtering through 6- $\mu\text{m}$  nylon cloth meshes (remove fine materials), and slide mounting. Every sample weighed roughly 0.25 and 0.5 CC ml. Minimum dryland pollen sum was approximately 150 grains but usually ranged between 150 and 200 grains. Charcoal particles were enumerated from pollen slides.

#### 5.4.2 CHARCOAL

Charcoal analysis was also quantified as part of the palynology sampling and presented alongside pollen data by Xun Li. Charcoal counts were converted to concentrations (charcoal particles  $\text{cm}^{-3}$ ) by dividing counts by volume of sediment sieved (Perry et al., 2012). Charcoal counts on pollen slides are used to reconstruct charcoal abundance, with microscopic equal to  $<100\ \mu\text{m}$  and macroscopic  $>100\ \mu\text{m}$ .

#### 5.4.3 ANALYSIS OF ENVIRONMENTAL DNA USING DROPLET DIGITAL PCR

The sampling strategy for environmental DNA comprised two approaches in order to gain both a detailed and broader understanding of environmental change within the lake. The first included a targeted approach on core LC3U at 1-cm intervals over three key areas of interest (pre-Polynesian settlement, European arrival/intensification and riparian planting). The second provided a more holistic approach at 2-, 3- and 5-cm intervals outside of these periods of interest. Particular areas of interest were identified through a combination of hyperspectral scanning, radiocarbon dating, palynology and historical reconstruction and included 12-25 cm, 38-42 cm and 57-72 cm.

Droplet Polymerase Digital Chain Reaction (ddPCR) was used to quantify concentrations of bacteria associated with ruminants within the samples in order to provide a proxy for changes in stocking rates in Lake Nganoke. This was completed Cawthron Institute in Nelson.

#### *DNA extraction*

All steps of the molecular analysis (DNA extraction, Polymerase Chain Reaction (PCR) or ddPCR) were conducted in a separate sterile laboratory, with chronological workflow to ensure no cross contamination. Ultra-violet sterilisation was available in all rooms dedicated to DNA extraction, amplification set up and template addition, and was used for a minimum of 15 min before and after each use. The ddPCR set-up and template addition were undertaken in laminar flow cabinets with high-efficiency particulate air filtration. Aerosol barrier tips (Eppendorf, 2019) were used throughout to avoid contamination between samples.

Sediment sub-samples were kept frozen (-20°C) and in the dark until DNA extraction. Operating in a laminar flow hood, approximately 0.25 g of sediment from each sample was weighed into the first tube of a DNEasy® PowerSoil® Soil DNA Isolation Kit (Qiagen, 2019). DNA was then extracted with DNA eluted in 100 µL. Extraction of DNA was performed in batches of twelve including a negative control every second batch. This negative control contained all reagents but no sediment, providing a means of assessing whether contamination had occurred. Following extraction, DNA concentrations were measured with the DNA protein ratio estimated for each sample using a Bio Spectrophotometer (Eppendorf, 2019). Samples were stored in the freezer (-20°C) until further analysis.

#### *Droplet digital PCR amplification*

Absolute concentrations of a 16S rRNA gene specific to ruminants was measured in all samples using a BioRad QX200 Droplet Digital PCR system and Bacteroides Ruminant specific primers and a probe (BacR\_f'-GCGTATCCAACCTTCCCG', BacR\_r'-CATCCCCATCCGTTACCG', BacR\_p'-FAM-CTTCCGAAAGGGAGATT-MGBNFQ', (Reischer et al., 2006). Each ddPCR reaction included 450 nM of each primer and probe, 10 µL of 1×BioRad ddPCR Supermix for probes, 1 µL of DNA, and sterile water for a total reaction volume of 22 µL. The BioRad QX200 droplet generator divided all reaction mixtures into nanodroplets by mixing 20 µL of the reaction mixture with 70 µL of BioRad droplet oil. Following processing, a total nanodroplet volume of 40 µL was transferred to a PCR plate for amplification. The following cycling protocol (40 cycles of 94°C for 30 s, 60°C for 60 s) was used and then analysed on the QX200 instrument held at 95°C for 10 min, with a final enzyme deactivation step at 98°C for 10 min.



For each ddPCR plate run, at least one negative control and one positive control were included. When inhibition was observed or samples were too concentrated these were diluted with RNA/DNA-free water (ThermoFisher, 2019) and reanalysed. The results were then converted to copies g<sup>-1</sup> using the following formula:

**Equation 2**

$$eDNA \text{ sediment samples} = \frac{(\text{Number of copies per } \mu L \times 22 \mu L \times 100 \mu L)}{(\text{weight of lyophilized starting material (in g)})}$$

#### 5.4.4 GEO-CHEMICAL ANALYSIS – CADMIUM, TOTAL NITROGEN AND TOTAL CARBON

A two-tiered sampling resolution was undertaken in the sampling of LC6U for both geochemical analysis and P fractionation. Funding constraints and sediment volumes dictated a maximum of 15 samples. A 2-cm sampling resolution was used over the transition between the European and Intensification period (samples at 8, 12, 16, 20, 22, and 24 cm). Coarser resolution was used over the remaining core with an emphasis on identifying periods of landuse change (samples at 28, 34, 40, 50, 60, 70 cm).

Chemical analysis of cadmium was conducted as part of the P fractionation method at Analytica Hamilton. Sample extracts were diluted and analysed on a Perkin Elmer 300D ICP-MS using collision mode (KED) for all elements and scandium as the internal standard. Detection limits (mg L<sup>-1</sup>) for the machine are Cd <0.02 and Pb <0.04.

Chemical analysis of total nitrogen and carbon was carried out on 15 samples as part of the P fractionation method at Analytica, Hamilton. The reporting limit was 0.02 g/100 g dry weight and 0.1 g/100 g dry weight for total nitrogen and total carbon, respectively. Specifically, samples were air-dried and sieved to < 2 mm then analysed by dumas combustion analysis with infrared detection, based on ISO 10694:1995, IDT. Total organic carbon is measured after first removing carbonates by hydrochloric acid treatment of the air dried sample. Samples were prepared and analysed at Analytica Laboratories. C:N ratios were calculated to infer the source of organic matter accumulating in the sediment.

#### 5.4.5 AERIAL PHOTOGRAPHY

Historical aerial photographs were sourced under ‘The Local Government Geospatial Alliance’ and used to establish and monitor the growth of Lake Nganoke’s riparian zone (LGGA, 2019). Using the software ArcGIS Pro, photographs were geo-referenced and annotated, with changes in Lake Nganoke and riparian growth outlined.

### 5.5 LAKE ECOSYSTEM AND WATER QUALITY RECONSTRUCTION

#### 5.5.1 HYPERSPECTRAL IMAGING

Hyperspectral imaging was used to derive the Relative absorption band depth (RABD) at 660-670 nm as a proxy for Chlorophyll a and in-lake productivity. All three cores LC1U, LC3U and LC6U were scanned. Cores were prepared by scraping the surface sediment ensuring a smooth and flat surface. This guaranteed there were no pockets or deviations within the surface, as these can create false readings within the hyperspectral data. Hyperspectral imaging scans were then collected for each of the four sediment cores using a Specim SISU single core scanner with a sCMOS-50-V10E spectral camera. Image pre-processing and normalization followed protocols from Butz et al. (2017). Spatial resolution was set to 1 mm/pixel at a spectral resolution of 660-670 nm. Spectral indices were calculated in R version 3.4.0, with the RABD ratio R670/R680 providing a simple estimate of changes of in-lake productivity downcore.

#### 5.5.2 BACTERIAL METABARCODING (16S rRNA)

Bacterial metabarcoding targeting the 16S rRNA gene was undertaken on 42 samples. The sampling resolution is the same as that identified in Section 5.4.3.

#### *Amplification, sequencing and bioinformatics*

The V3-V4 region of the bacterial 16S rRNA gene was amplified by Polymerase Chain Reaction (PCR), using the bacterial-specific primers 341F: 5’-CCT ACG GGN GGC WGC AG-3’ and 805R: 5’-GAC TAC HVG GGT ATC TAA TCC-3 (Herlemann et al., 2011; Klindworth et al., 2013). All batches included a negative and positive control. PCRs (48 µL) containing 47 µL of reagents and 1 µL of extracted sediment DNA were amplified, with the reagent mix for one tube containing; 25 µL of AmpliTaq Gold® 360 Master Mix (Technologies, 2019), 10 µL of DNA/RNA free water (Technologies, 2019), 5 µL of 360 GC enhancer (Technologies, 2019), 5 µl of BSA (Sigma) and 1 µL of each primer at 10 µM. The cycling profile was; denaturation at 95°C for 10 min, followed by

35 cycles at 95°C for 30 s, annealing at 52°C for 30 s, extension at 72°C for 1 min and a final extension step at 72°C for 7 min.

Final products, including all negative extraction and PCR controls were visualized on a 1.5% agarose gel electrophoresis stained with Red Safe DNA Loading Dye (WORLD, 2019). UV illumination of the product also guaranteed amplifications of a single 400 bp product. Purification and quantification of amplicons were performed with the 96-well SequalPrep™ Normalization Plate Kit (Harris et al., 2010), normalizing DNA concentrations up to 1.25 ng  $\mu\text{L}^{-1}$  per sample. Libraries were sent to New Zealand Genomics Limited at the University of Auckland for sequencing and paired-end sequences ( $2 \times 250$ ) generated on a MiSeq instrument using the TruSeq™ SBS kit (Illumina, 2019).

Sequence data were automatically demultiplexed using MiSeq Reporter (v2), and forward and reverse reads were assigned to samples. General sequence quality was assessed with FastQC, unwanted sequences were removed with Cutadapt. The DADA2 package (R / R Studio) was used to quality filter the sequences and remove sequences of poor quality. Amplicons were pooled into a single library and paired-end sequences ( $2 \times 250$ ) were generated on a MiSeq instrument using the TruSeq™ SBS kit (Illumina, 2019).

Due to time constraints, the bioinformatics were completed by John Pearman at Cawthron Institute. The R software version 3.4.2 (Team, 2017), RStudio software version 1.1.442 Studio (2019), and the phyloseq package (McMurdie & Holmes, 2013) were used for merging data from all sequencing runs, normalising the samples at the same sequencing depth, and carrying out statistical analyses and plots design. Based on the rarefaction curves, samples were rarefied at 14800 reads through random subsampling with no replacement.

### 5.5.3 STATISTICAL TESTING – NONMETRIC MULTIDIMENSIONAL SCALING

In order to provide greater insight into the composition change of taxa within Lake Nganoke, a nonmetric multidimensional scaling (NMDS) based on Bray-Curtis distance was used to summarize the pattern of species occurrence between zones identified in Section 4.3 (Legendre & Legendre, 2012). In order to test for statistical differences between these zones, an Adonis function of the R vegan package was applied, with values

of  $p$  less than 0.05 indicating a significant difference between the taxonomic composition of the zones (Oksanen et al., 2013).

#### 5.5.4 PHOSPHOROUS FRACTIONATION

This thesis pioneers the innovative approach of applying the phosphorous fractionation downcore. The sampling strategy is the same as that already identified in Section 5.4.4. In order to calculate the residual value, a complex process and assessment of all the P fractions must be completed. The methods used within this thesis, as described below, are based on Waters (2016).

##### Pore water

Prior to analysis, pore water was immediately separated from the sediment by centrifuging at 4000 rpm for 40 min, filtered through 0.45- $\mu$ m Millipore membrane and frozen until analysis. The remaining sediment was freeze-dried at -18 °C for 48 hr using the CHRIST Freeze Dryer (Fissore et al., 2018). Pore water samples were analysed for dissolved phosphorous by inductively coupled plasma optical emission spectrometry (ICP-OES). The limit of detection for P on this instrument is 0.00455 mg/L<sup>-1</sup>.

##### Sequential chemical extractions

Extraction of phosphorous is based on Psenner et al. (1988) and adapted slightly from that described in Rydin (2000) (Figure 5.2).

The P forms targeted by the extraction steps are as follows;

1.  $NH_4Cl$ -P;

For each sample approximately 0.5 g of sediment were placed into a Falcon tube, weighed and recorded. 25 mL of  $NH_4Cl$  solution was pipetted into the sample, weighed and placed on the shaker for 2 hr. Once thoroughly mixed, samples were centrifuged at 3900 rpm for 15 min before filtering (0.45  $\mu$ m) to waste, keeping the remaining filtered sample plus sediment. The extracts were sent to Analytica for dissolved reactive phosphorous (DRP) analysis.

2. BD-P;

The BD step (involving bicarbonate/dithionite ( $NaHCO_3/Na_2S_2O_4$ )) is a reducing reaction. 25 mL of BD solution was pipetted into solution, weighed and placed on the shaker for 1 hr. Samples were then centrifuged, decanted and

filtered (as in step 1), with the extracts sent to Cawthron for analysis using an inductively coupled plasma mass spectrometer (ICP-MS).

3. NaOH-tot-P;

The NaOH-reactive P (rP) step extracts remaining P forms which are sensitive to pH change. 25 mL of 0.1 M NaOH was pipetted into the sediment and weighed, before being placed on the shaker for 16 hr. The sample was then centrifuged, decanted and filtered (as in step 1) with the extracts sent to Cawthron for ICP-MS analysis.

*NaOH-rp*

1 mL of 0.1 M NaOH was extracted from the sample and placed within a 50-mL tube. The sample was then topped up with Milli-Q and weighed before being sent to Analytica for DRP analysis.

4. HCl-P;

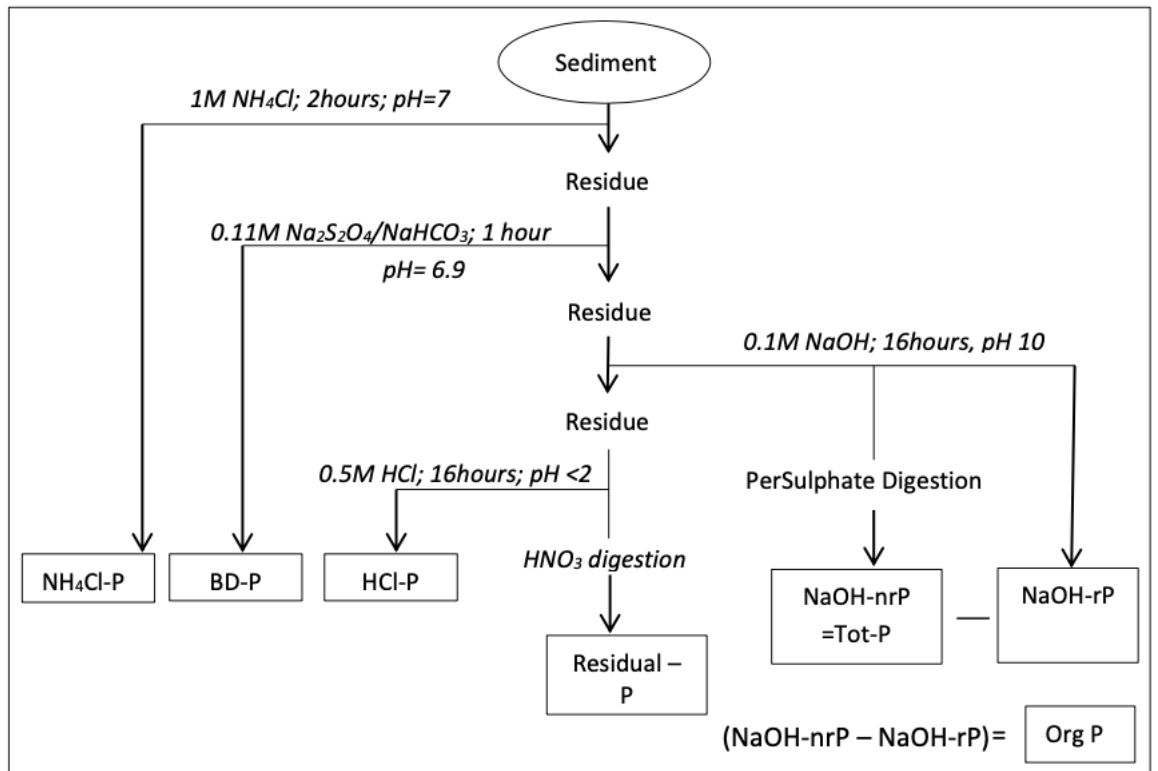
The HCl step targets remaining P forms, which are mobile at low pH. 25 mL of HCl concentrate were pipetted into the remaining sediment from Step 3 and weighed, before being placed on a shaker for 16 hr. The sample was then centrifuged, decanted and filtered (as in step 1) with the extracts sent to Cawthron for ICP-MS analysis.

5. Res-P;

Residual P is extracted with a strong acid digestion ( $\text{HNO}_3$ ) and is considered to include refractory organic and inert inorganic forms. The residual sediment was sent to Analytical Labs for Total Recoverable P analysis.

6. Total *P;*

Total sediment P is considered to be the sum of the extracted P forms (SOE).



**Figure 5.2: Sequential extraction scheme used on Lake Nganoke sediments.**  
Waters (2016)

#### 5.5.5 LOSS ON IGNITION (LOI)

LOI sampling was completed as part of the geochemical analysis and P fractionation outlined in Section 5.4.4. Wet weight to dry weight (DW) ratios give an estimation of water content in the sediment and were determined by drying a sediment sample at 105°C for 24 hr. The sample was weighed prior to and after the drying process. These samples were then combusted at 550°C for 4 hr. The weight lost during ignition (LOI) gives a percentage of organic material by the following equations:

##### Equation 3

$$LOI_{550} = \frac{(DW_{105} - DW_{550})}{DW_{105}} \times 100$$

Further ignition at 950°C for 2 hr gives an estimate of carbonate in the sediment and was calculated with the following equation:

##### Equation 4

$$LOI_{950} = \frac{DW_{550} - DW_{950}}{DW_{105}} \times 100$$

The weight of carbonate in the original sample is the weight lost due to ignition at 950°C (Heiri et al., 2001).





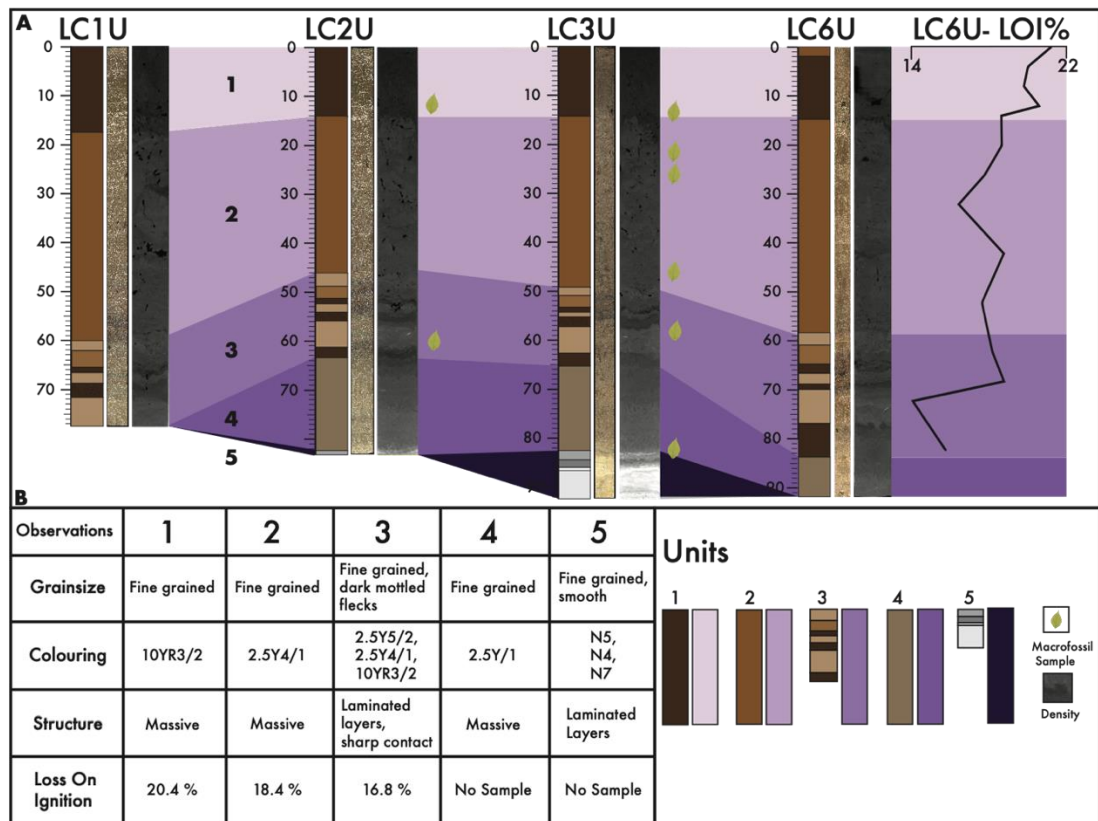
## **6 CHAPTER 6: RESULTS**

Chapter 6 describes the results of proxy analyses that inform changes in landuse, lake ecosystem and water quality within Lake Nganoke and its surrounding catchment. This chapter begins by characterising the sedimentology and producing a chronology of four cores (LC1U, LC2U, LC3U, LC6U) from Lake Nganoke (Section 6.1). The results for landuse change proxies, including palynology, ruminant ddPCR, geo-chemical analysis, historical photography, and stocking/ fertilisation rates are described in Section 6.2. Measures of lake ecosystem and water quality change are described through the proxies, bacteria metabarcoding, hyperspectral imaging, phosphorous fractions and geochemical analyses (Section 6.3). All proxies described are classified into Pre-human, Māori, European and intensification zones; the timing of which are defined based on historic and previously published records germane to Lake Nganoke (see Chapter 2, Section 4.3). Finally, aerial photography of riparian establishment and growth are characterised and described. Figures within Sections 6.2 and 6.3 plotted against depth are displayed within appendix B.

### **6.1 STRATIGRAPHY AND CHRONOLOGY**

#### **6.1.1 STRATIGRAPHY**

Defined on the basis of colour, texture, structure, organic content and CT scanning, cores LC1U, LC2U, LC3U and LC6U are characterised by five lithofacies that can be correlated accurately across all four cores. The exception is unit 5, which is not observed in LC6U (Figure 6.1). Unit 1 is a dark brown (10YR3/2) silt that has massive structure and an LOI average (proxy for organic content) of 20.4%  $\pm 1$  Sigma  $\pm 1.06\%$ . Similarly, unit 2 is an orange brown (2.5Y4/1) silt that has massive structure and an LOI average of 18.4%  $\pm 1$  Sigma  $\pm 1.2\%$ . Consisting of a laminated-layer structure with sharp contacts, unit 3 is a mixture of dark, orange and pale brown colouring (2.5Y5/2, 2.5Y4/1, 10YR3/2) with an average LOI of 16.8%  $\pm 1$  Sigma  $\pm 0.9\%$ . While unit 4 is an olive brown (2.5Y/1) silt with a massive structure, unit 5 is a greyish brown (N5, N4 and N7) silt with a laminated structure. No loss-on-ignition samples were available for either unit. The five lithofacies are easily distinguished in CT densitometry, with clear laminations found within units 3 and 5. Macrofossil samples for dating were taken from LC2U at depths of 12 and 60.5 cm and LC3U at depths of 13, 20.5, 26, 46, 57.5 and 83 cm.

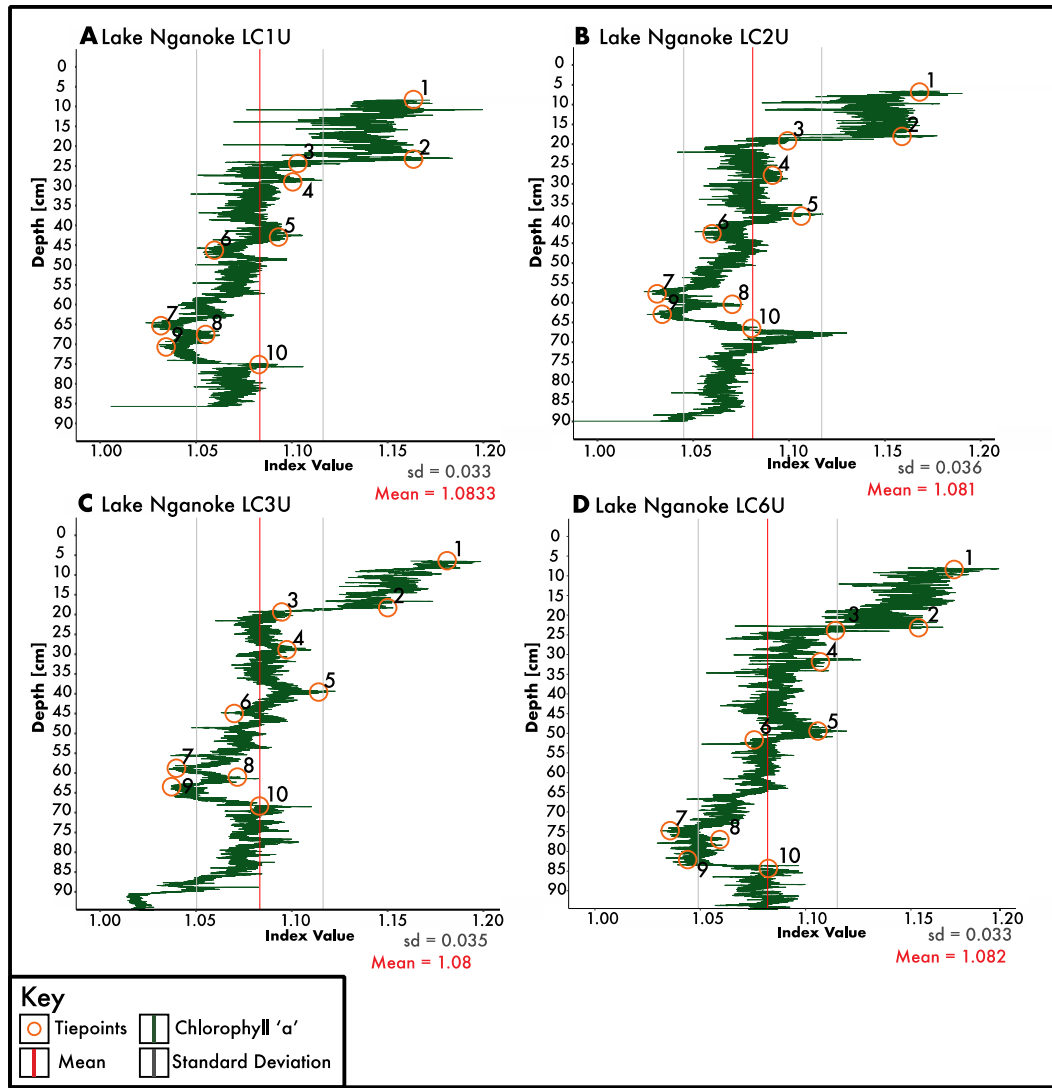


**Figure 6.1: Stratigraphic characterisation (A) and observations (B) of cores LC1U, LC2U, LC3U and LC6U.**

Cores are characterised from visual logs, scanned imaging and internal structures (CT). Lithofacies are described by colouring and grouped into five distinct units, with each of the five units characterised by grainsize, colouring, structure and averages of loss on ignition. Measurements of LOI% are representative of LC6U only and averaged for units 1, 2 and 3.

### 6.1.2 CORE CORRELATIONS TO MASTER STRATIGRAPHY

Ten tie points were identified in each core (LC1U, LC2U, LC3U and LC6U) from hyperspectral imaging (RABD 660-670 nm) in order to correlate all data to a master stratigraphy (Figure 6.2).



**Figure 6.2: Correlations of cores LC1U (A), LC2U (B), LC3U (C) and LC6U (D).**

Where correlated points are depicted in orange, with Chlorophyll *a* abundance (RABD at 660-670 nm) in green, mean in red and standard deviation in grey. All points are defined by a clear change in Chlorophyll *a* throughout all four cores.

A master stratigraphy was created by regressing LC1U, LC2U and LC6U against LC3U using points identified in Figure 6.2. This resulted in the following polynomial regression curves (Figure 6.3):

#### Equation 5

$$LC3U = -2.9525 + (1.0006 \times LC1U) - (-0.0007 \times LC1U^2)$$

$$r^2 = 0.9981$$

#### Equation 6

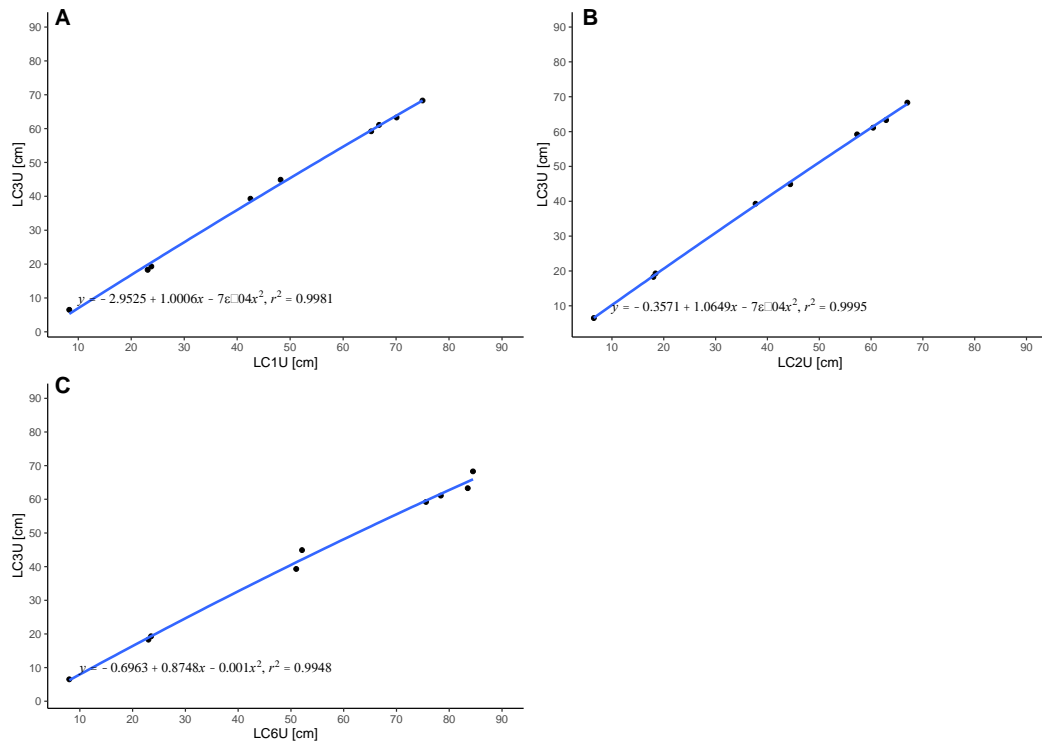
$$LC3U = -0.3752 + (1.0649 \times LC2U) - (-0.0007 \times LC2U^2)$$

$$r^2 = 0.9995$$

### Equation 7

$$LC3U = -0.6963 + (0.8748 \times LC6U) - (-0.0001 \times LC6U^2)$$

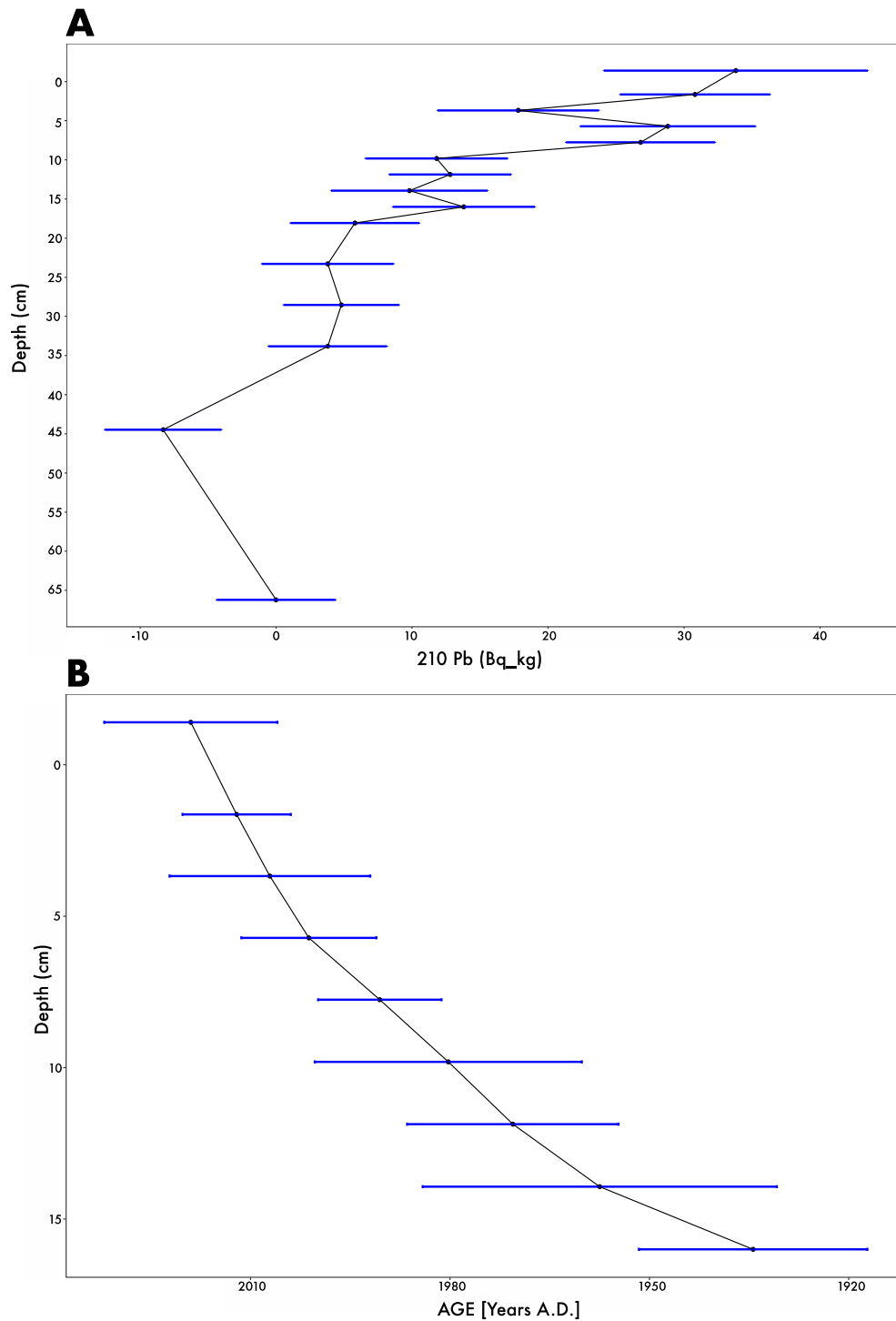
$$r^2 = 0.9948$$



**Figure 6.3: Polynomial regressions of cores LC1U (A), LC2U (B) and LC6U (C)**  
Correlations (blue) are calculated by regressing depths (black dots) on one core against the master core (LC3U). Error is displayed as grey

### 6.1.3 AGE DEPTH MODEL

Sediment within Lake Nganoke was dated using  $^{210}\text{Pb}$ , bomb spike radiocarbon, biostratigraphy (pollen) and conventional  $^{14}\text{C}$  techniques (See appendix A). Fourteen  $^{210}\text{Pb}$  samples were collected from Lake Nganoke, ranging from 0 to 66 cm (LC3U correlated depth) (Figure 6.4A). The CRS model returned a total of nine dates, ranging from AD 2019 (95% Confidence Interval (CI): AD 2032 – 2005) to ~ AD 1934 (95% CI: AD 1951 – 1917) (Figure 6.4B). Bomb spike dates returned fraction modern values of 1.063314885 (+/-0.00277113) (95.4 % AD 1958) and 1.043163663 (+/-0.002498979) (90% AD 1957). Bio-stratigraphic ages were based on known taxa arrival dates within literature, and as such, detections of willow at 38.5 cm were dated to AD 1870 (95% CI 1850 – 1889) pine at 51.5cm to AD 1865 (95% CI 1829 – 1904) and Māori arrival based on bracken fern at 58.5cm to ~AD 1450 (95% CI 1255 – 1645). The radiocarbon sample from 62.8 cm depth provided a calendar age once calibrated of AD 1431 (95% CI AD 1415 – 1446).



**Figure 6.4:**  $^{210}\text{Pb}$  excess plotted against depths (A) and ages calculated using the CRS model (B).

Where 95% uncertainties are depicted in blue.

Figure 6.5 depicts the age-depth model for Lake Nganoke produced using the P sequence prior and independent age information described above. The model covers a range from AD 1150 (95% CI: AD 759 – 1366) to AD 2019 (95% CI: AD 2018 – 2019) with maximum precision located at depth 0 cm and minimum at 93 cm. The agreement index provides a measure of model fit to the chronological data and is the degree to which the

posterior probability density functions (PDFs) overlaps with the likelihood PDFs. Good model fit is defined as a model agreement index greater than 60% (Bronk Ramsey, 2008). The agreement index for Lake Nganoke is 132%, demonstrating the robustness of the age model. The model also demonstrates that sedimentation rates have not been constant over the 1-kyr duration of the record. Fluctuations in sedimentation are observed between pre-human ( $0.087 \text{ cm.yr}^{-1}$ : 95% CI:  $0.07 - 0.087$ ) and Māori zones ( $0.03 \text{ cm/year}$ : 95% CI:  $-0.02 - 0.31$ ). The European zone experiences the highest sedimentation rate ( $0.301 \text{ cm/year}$ : 95% CI:  $0.297 - 0.304$ ) before declining during agricultural intensification ( $0.219 \text{ cm/year}$ : 95% CI:  $0.183 - 0.255$ ).

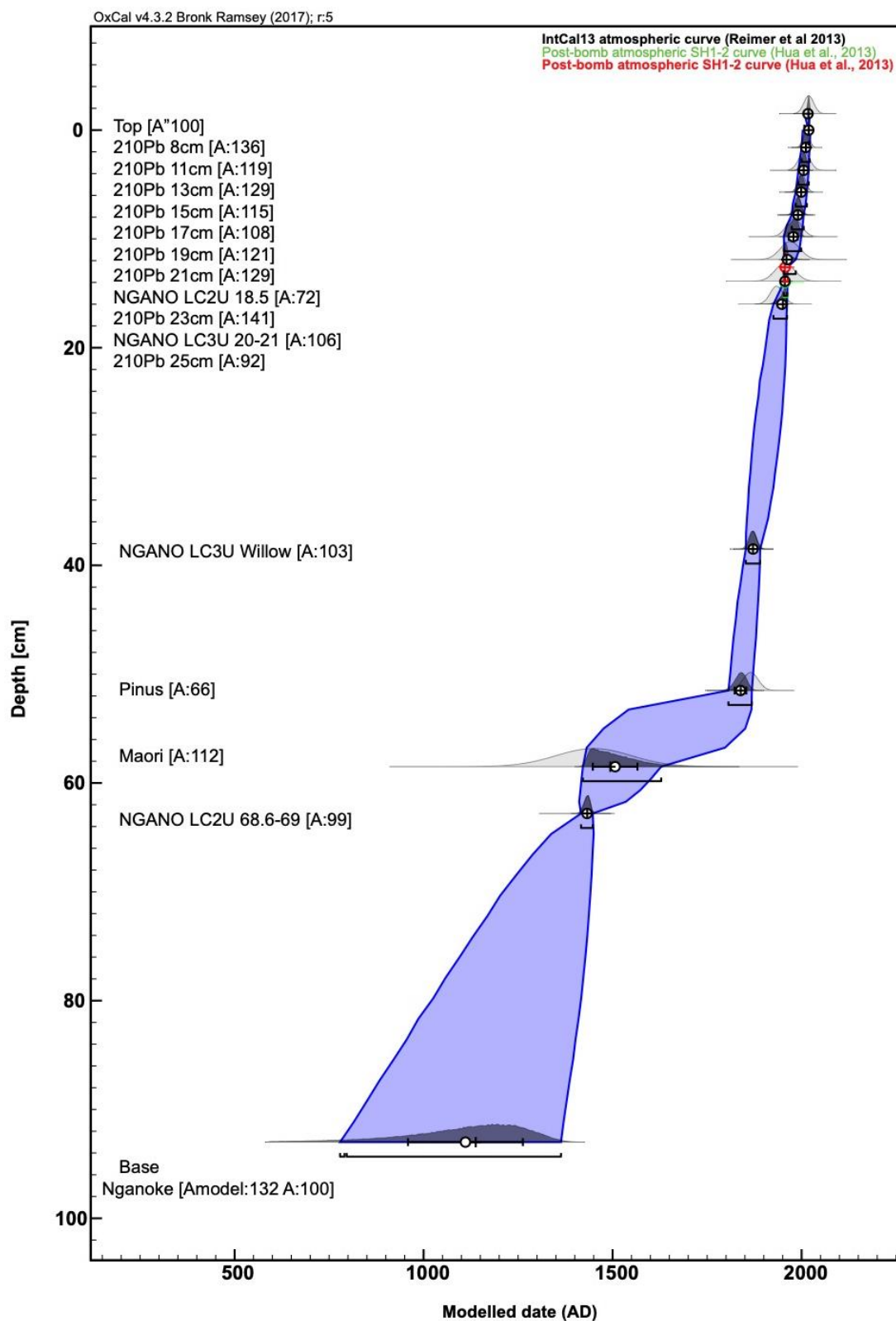


Figure 6.5: OxCal age model of Lake Nganoke sediment down core using depths correlated to a master core (LC3U).

## 6.2 LANDUSE RECONSTRUCTION

### 6.2.1 PALYNOLOGY

A total of 16 samples were submitted for pollen analyses. Summarised in Figure 6.6, pollen records are described by zones and displayed as a percentage of the dryland pollen sum.

#### **Pre-Human:** Pre AD ~1450

During the pre-human period (< AD 1450), terrestrial pollen is dominated by high abundances of *Fuscospora* (red beech, black beech, mountain beech and hard beech: 55.2%), *Dacrydium cupressinum* Spp. (rimu: 21%) and other Podocarps such as: *Prumnopitys taxifolia* (mataī) ; *Podocarpus* spp. (totora); *Dacrycarpus dacrydioides* (kahikatea) and *Prumnopitys ferruginea* (miro: all totalling 17.2%). Elevated values of *Cyathea Smithii* & *Cyathea dealbata* (fern pollen: 13.8% and 8.6%) are also recorded. Holistically, tall trees remain the dominant vegetation (>95%) until AD 1431, where the composition changes to 80% tall trees, 12.8% small trees and 6.8% herbs. A small charcoal spike (50-100 µm: 4.6%) is observed early in AD 1100s.

#### **Māori:** AD ~1450 – AD ~1850

Representative of Māori arrival and occupancy on the landscape, this period is characterised by several noteworthy changes in pollen spectra. *Fuscospora*, *D. cupressinum* and other Podocarp pollen show a general decline accompanied by a marked arrival in AD 1490 of Poaceae (Gramineae: 6.8%), *Cyperaceae* (Sedges: 18.2%) and *Pteridium esculentum* (Bracken: 126.4%). In contrast, small tree and shrub pollen concentrations do not decline post AD 1490 but rather increase (12.8 % to 20.3% by AD 1700). *Typha* (Raupo) shows a distinct increase (by 30%) towards the top of this zone (14.2% to 37.7% by AD 1700). Furthermore, herb abundance increases from 8.2% to 49.4% during the latter half of this zone (AD 1700-1850), with tall trees declining from 71.5% to 41%. A sharp decline in charcoal abundance is also observed by AD 1700 (68% to 27.5%), before decreasing to 19.3% by AD 1849.

#### **European:** AD ~1850 – AD ~1950

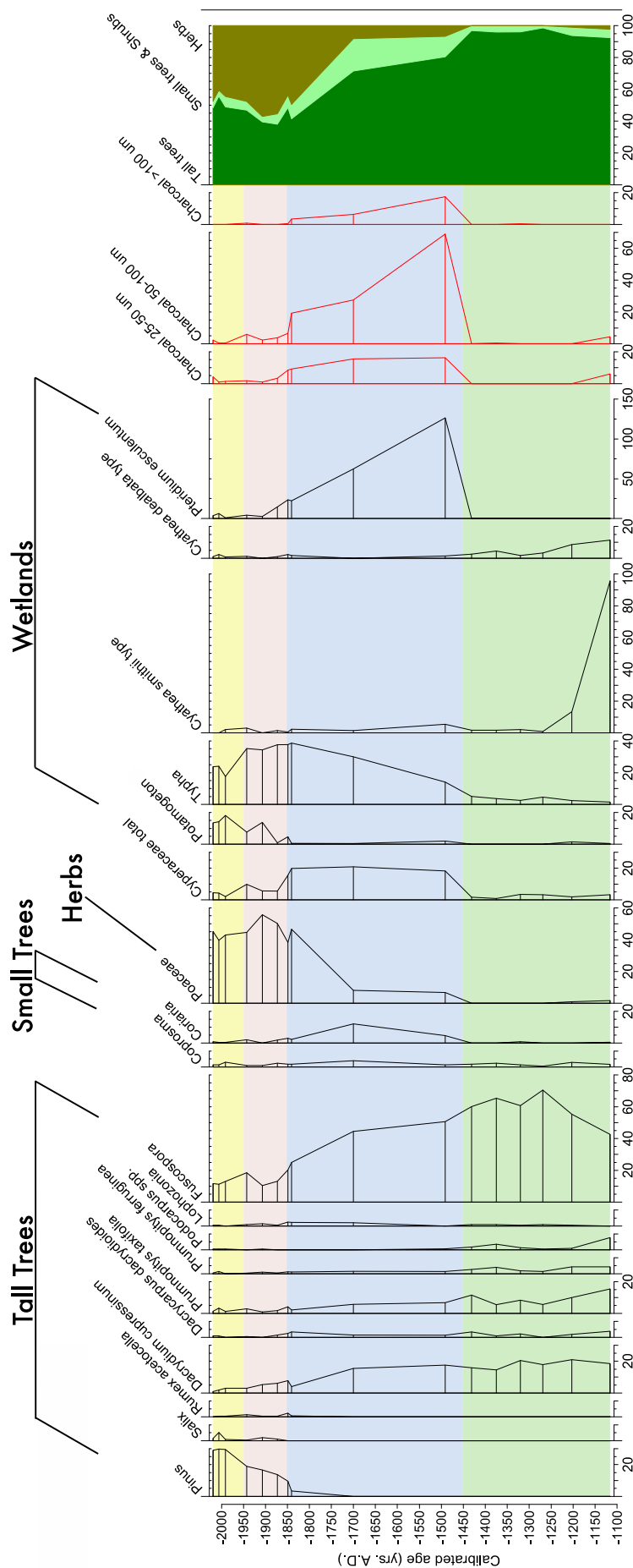
The European zone (AD 1850-1950) is characterised by the appearance and increase of exotic pollen types such as *Pinus* (9.6% to 18.9%), *Rumex acetosella* (0.5% to 1.5%) and the detection of *Salix* (willow: 0.9%) in AD 1873. Increasing abundancies in Poaceae pollen are identified between AD 1850 to 1873, combined with declining values of



*Pteridium esculentum* (23.4% to 4.3% by AD 1941) and the arrival of *Potamogeton* (Pondweed: 4.8%) in AD 1850. Overall, tall and small trees begin to increase towards the end of the European period (rising 8% from AD 1870 to 1941). Herb abundance begins to decrease in the second half of the zone (AD 1900 to 1950). A sharp decline in charcoal (charcoal 50-100  $\mu\text{m}$ : 6.8%) occurs at the start of the zone and there is a trend of decreasing abundance through the zone although a minor peak to 13.8% occurs at AD 1950

**Intensification:** AD ~1950 - present

Exotic taxa, such as pine (28.8%: AD 2019) continue to rise during the intensification period. Small fluctuations of *Potamogeton* (18.2% peak in AD 1991, 13.5% by AD 2019), Poaceae (trough of 39.6% in AD 2006 and 45.1% by AD 2019) *Typha*, *Cyperaceae* and *Macrocarpa dacrydiodes* are also observed with *Typha* decreasing in abundance across the zone boundary. A sharp peak of *Salix* (5.2%) is detected in 2006. The intensification period displays a continued trend of increasing tall trees (55.7% peak in AD 2006), consistent abundance of small trees/shrubs and declining herbs. Charcoal abundance continues to decline.

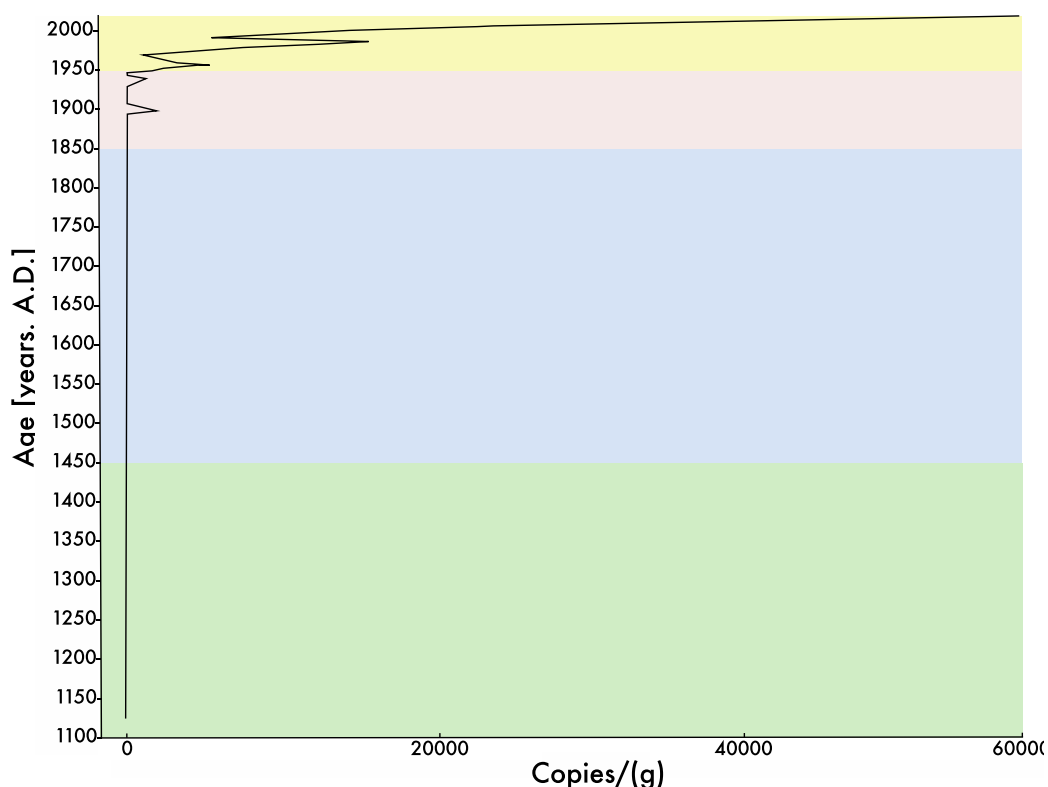


**Figure 6.6: Stratigraphy of key pollen taxa (see Appendix 1 for detailed stratigraphy).**

The y-axis constrains age, while the x-axis illustrates the abundance of different taxa, with the panel on the far right showing cumulative percentages based on forest and grassland vegetation type. The intensification zone is labelled yellow, European labelled pink, Māori blue and pre-human green.

### 6.2.2 ENVIRONMENTAL DNA – DDPCR

The detection of bacteria associated with ruminants was used as an indicator of their abundance within the catchment. High gene copies (1900 and 1200 copies per gram) were observed at ages AD 1900 and 1946 (Figure 6.7). The trend dramatically increases post 1950, rising to 60,000 copies per gram (a measure of abundance) by the end of AD 2019.



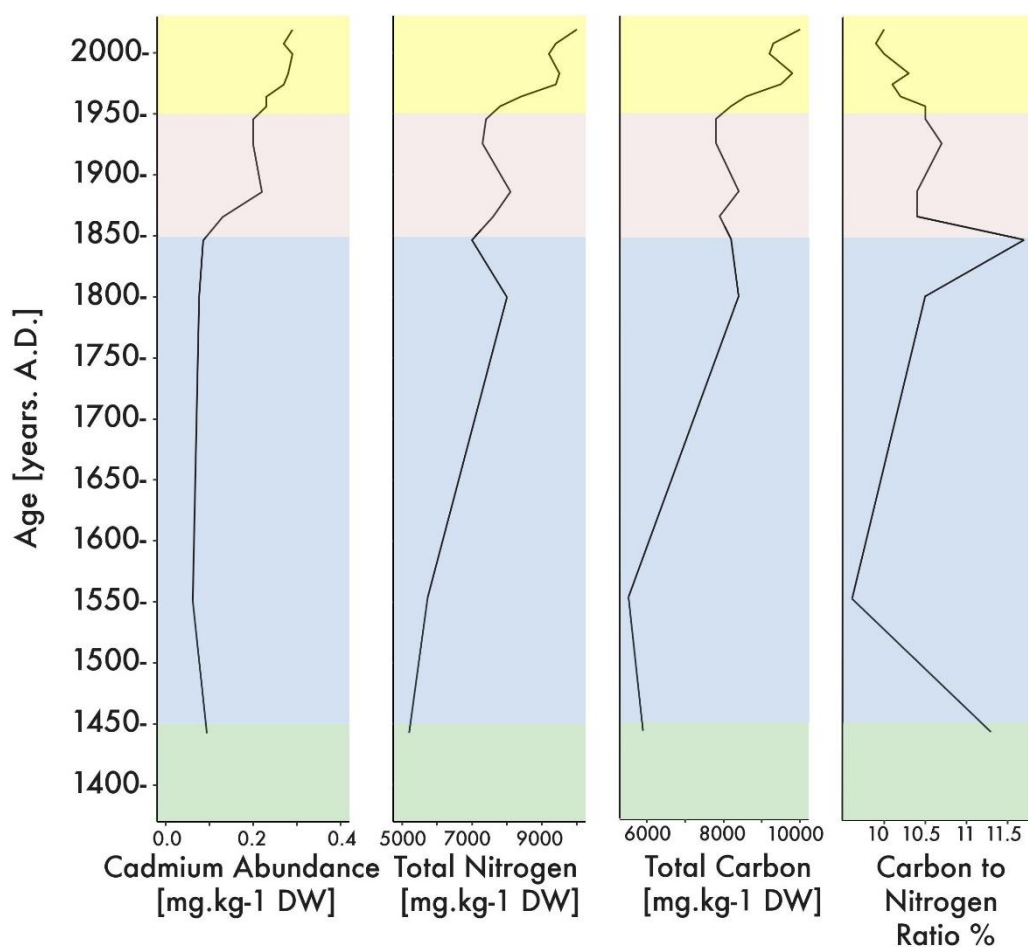
**Figure 6.7: Digital droplet ruminant PCR of Lake Nganoke samples.**

### 6.2.3 GEO-CHEMICAL ANALYSIS

A proxy for fertilisation application, cadmium abundance shows a slow incline from 0.08 to 0.12 mg kg<sup>-1</sup> DW (dry weight) during pre-human and Māori occupation. Cadmium increases throughout European landuse and again during intensification to more than double the values of previous zones (0.3 mg kg<sup>-1</sup> DW). A small decrease is highlighted at ~AD 1995. Total nitrogen rises relatively steadily during Māori occupancy to 8000 mg kg<sup>-1</sup> DW prior to fluctuations within the European Zone (7500 – 8200 mg kg<sup>-1</sup> DW). Increasing post AD 1950, total nitrogen reaches levels of 10,000 mg kg<sup>-1</sup> DW by AD 2019.

Following a similar trend, total carbon flux is relatively low (<6000 mg kg<sup>-1</sup> DW) during pre-human landuse before rising dramatically post AD 1550 from 5800 mg kg<sup>-1</sup> DW to more than 8000 mg kg<sup>-1</sup> DW by AD 1800. During the European zone, carbon flux follows a similar trend to cadmium and total nitrogen with a sharp peak in the mid to late 1800s.

Carbon flux increases from 7800 mg kg<sup>-1</sup> DW to 95000 mg kg<sup>-1</sup> DW during the first half of the intensification period (AD 1950 to AD 1965). Post AD 1970, carbon flux declines to 9000 by AD 1980 before increasing to 10,000 mg kg<sup>-1</sup> DW by AD 2019. A measure of organic matter provenance, carbon to nitrogen ratios decline (by 1.2) from AD 1450 to AD 1550. Ratios rise within the Māori zone to reach 11.5 by the start of the European period (AD 1850) before dropping to 10.5 in AD 1860. A general trend of decline is observed after AD 1930, for which ratios of carbon to nitrogen reach 10 by AD 2019.



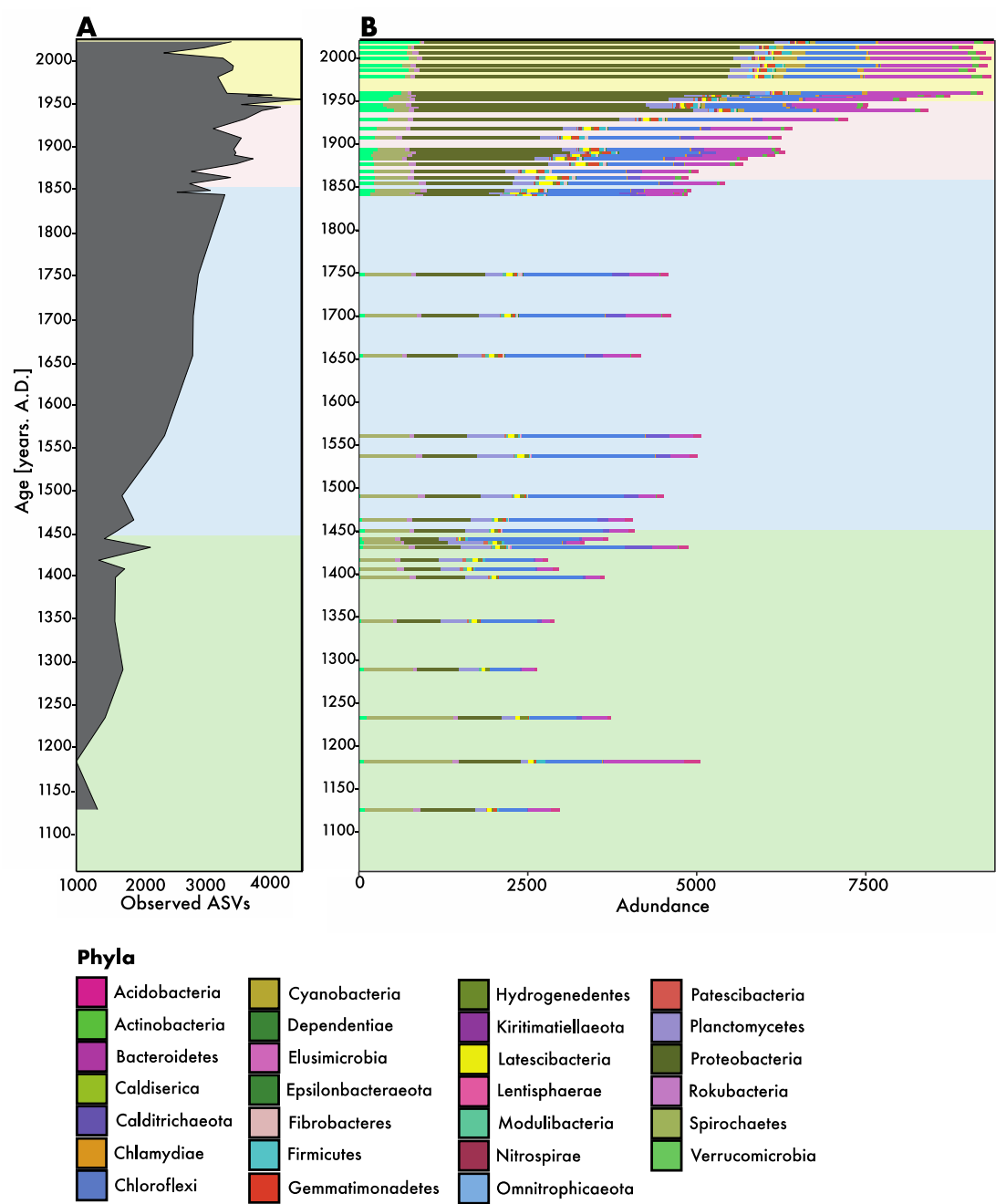
**Figure 6.8: Geo-chemical analysis of cadmium, total nitrogen, total carbon and carbon to nitrogen ratio (C:N ratio).**

## 6.3 LAKE ECOSYSTEM AND WATER QUALITY RECONSTRUCTION

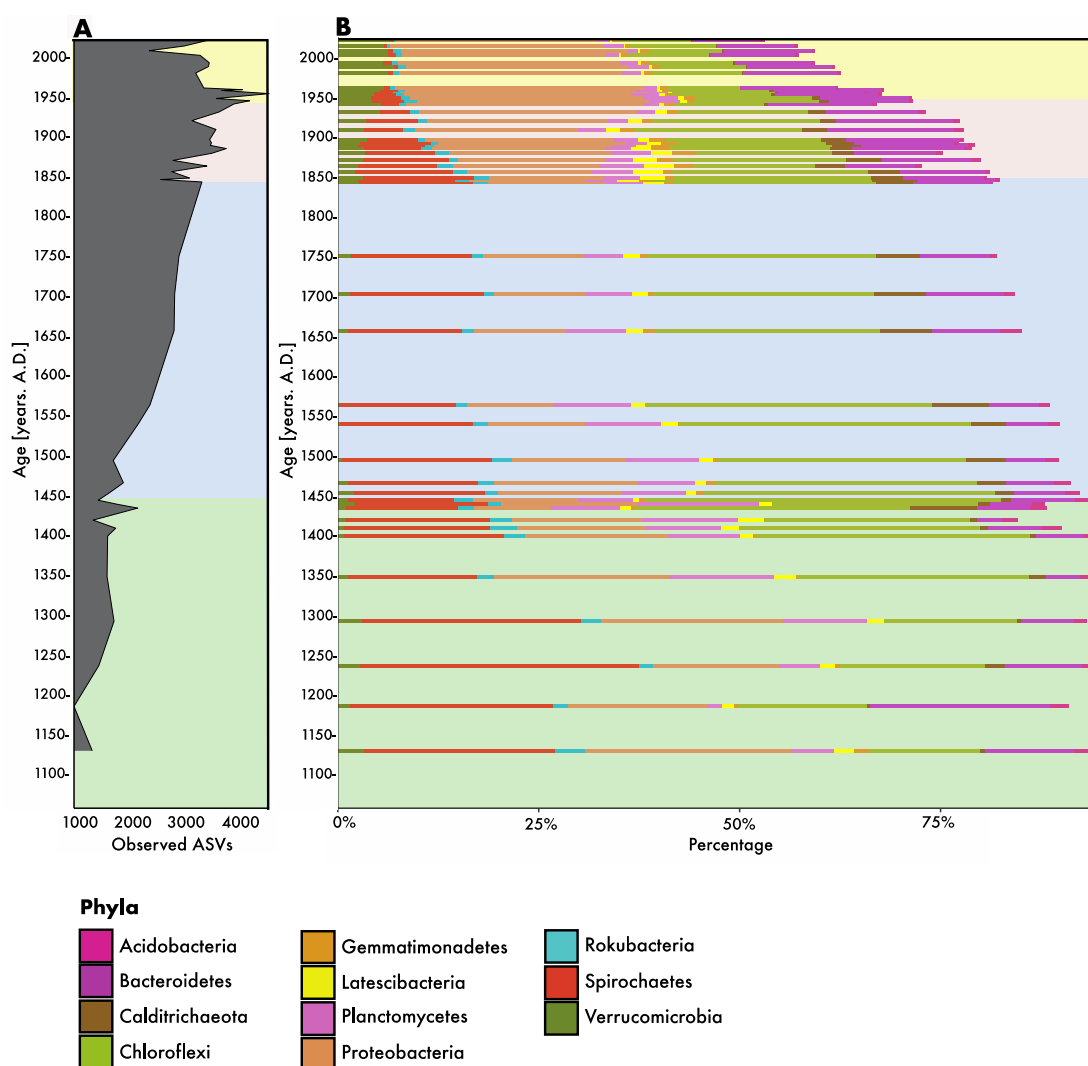
### 6.3.1 BACTERIAL METABARCODING – (16s rRNA)

Amplicon sequence variants (ASV's) refer to the individual DNA sequences recovered from a sample. A total of 44,040 bacterial ASV's were recovered from 47 samples within Lake Nganoke. A total of 27 bacterial phyla were identified (Figure 6.9), with Proteobacteria, Chloroflexi and Acidobacteria displaying the highest total number of ASV's. A total of 11 phyla were identified with mean percent abundances greater than

one (Figure 6.10). Chloroflexi, Proteobacteria and Spirochetes had the largest changes in percentage (>20%).



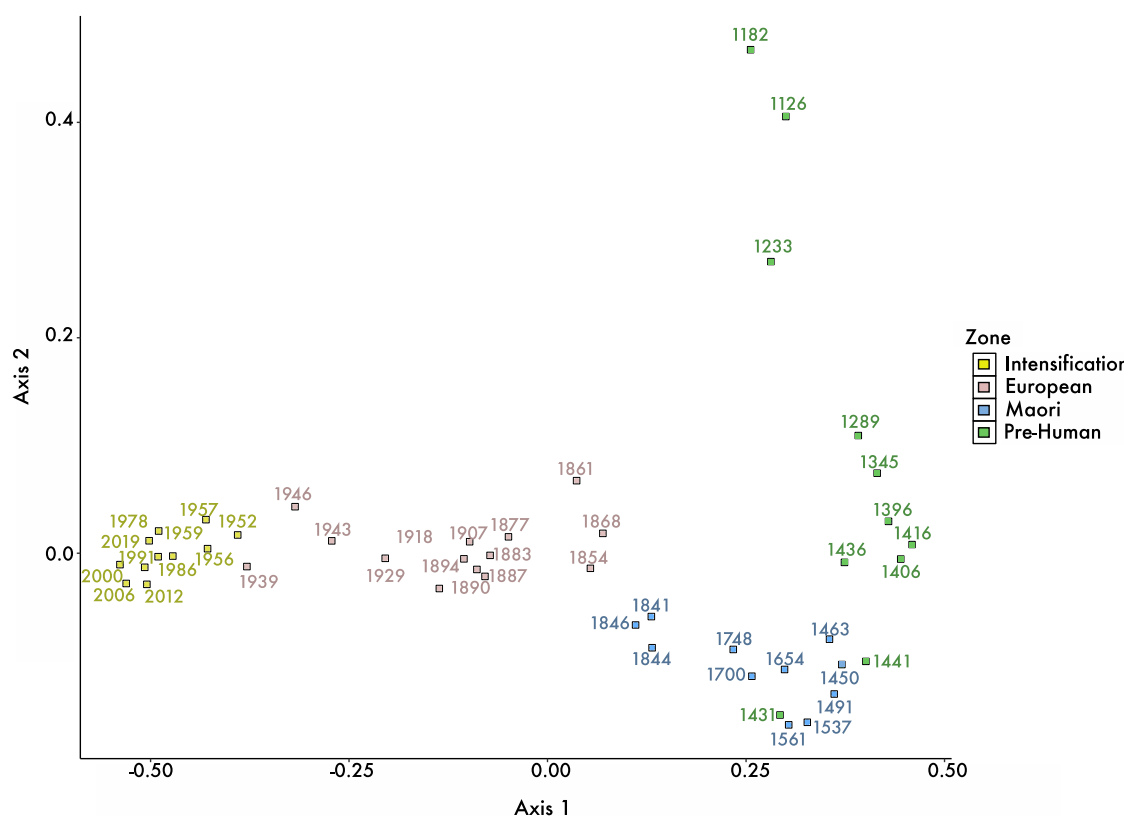
**Figure 6.9: Taxonomic richness (Observed ASV's) (A) and composition (reads per samples) (B) of Lake Nganoke at Phylum level.**



**Figure 6.10: Taxonomic percentage richness (Observed ASV's) and abundance (reads per samples) (B) of Lake Nganoke at Phylum level.**

This figure only includes phyla that have a mean greater than one.

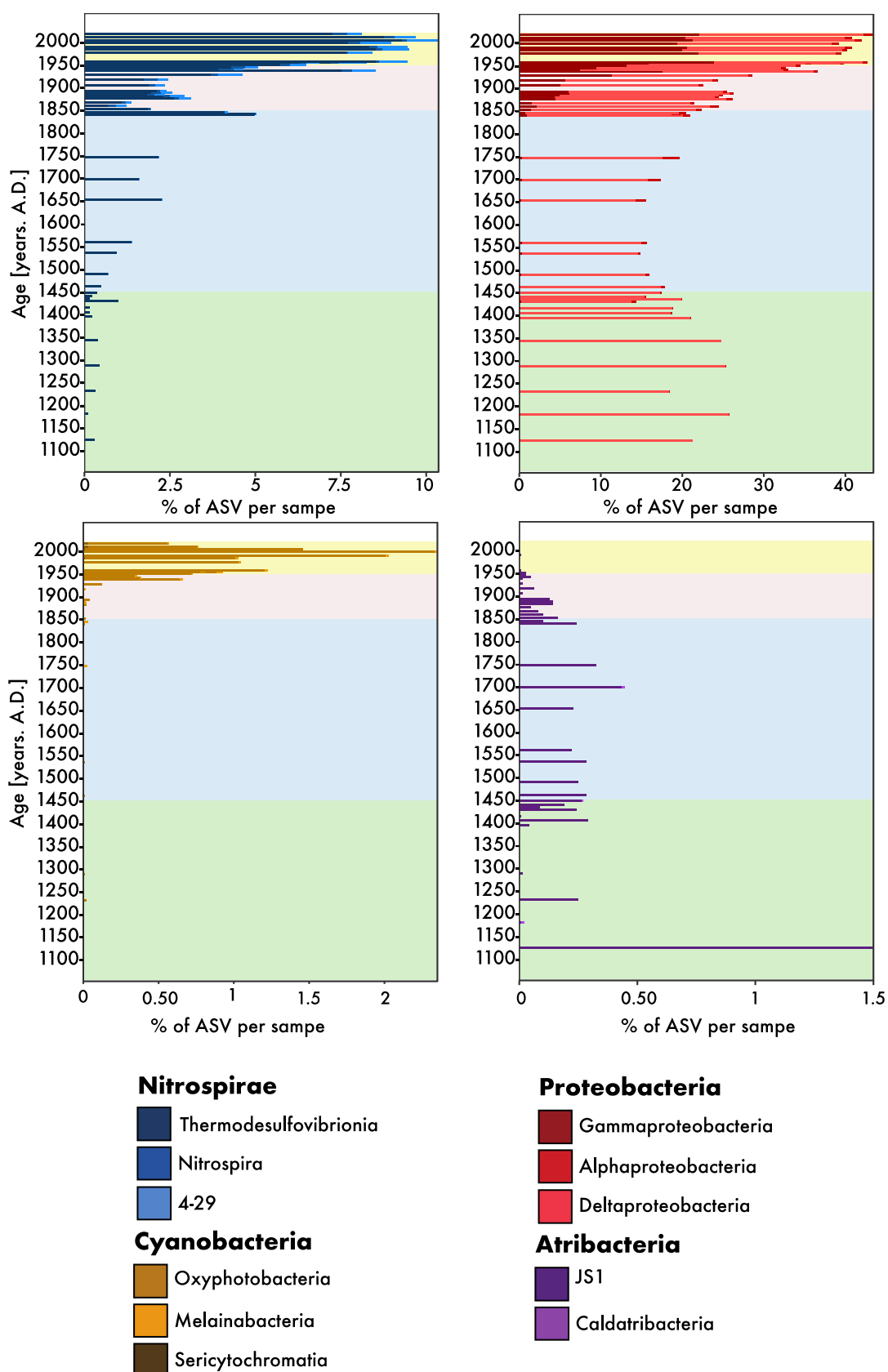
A non-metric multidimensional scaling plot was used to illustrate the difference in taxonomic composition between zones. Figure 6.11 displays a clear dissimilarity in species composition between intensification, European, Māori and pre-human, as no two zones cluster together. Two major trends are highlighted: axis one where intensification, European, and Māori are separate from each other and axis two where Māori and pre-human are separated. This may imply that the Māori zone is different from the rest. The *Adonis* statistical test, with a p-value of 0.001, confirms all four zones are significantly different from one another.



**Figure 6.11: Two-dimensional non-metric multidimensional scaling plot displaying taxonomic differences between landuse zones .**

Dissimilarity matrix was made using the Bray-Curtis dissimilarity distance, before being plotted on the NMDS.

As identified within Section 2.5.2 Nitrospirae, Proteobacteria, Cyanobacteria and Atribacteria have the potential to provide insight into historical changes within lake water quality and bacterial ecosystems. All four phyla are plotted within Figure 6.12 at class level, with Nitrospirae consisting primarily of Thermodesulfobacteria, with lower overall concentrations of the classes Nitrospira and 4-29 (<1% ASVs per sample) (4-29 ASV currently unidentified at class level). Further descriptions of the observed trends are explored below and classified within the zones established in Section 4.2.



**Figure 6.12: Taxa indicative of changes in lake water quality and ecosystem health identified through literature.**

Identified taxa consist of Phyla Nitrospirae (classes Thermodesulfobibrionia, Nitrospira and 4-29 - Blues), Proteobacteria (classes Gammaproteobacteria, Alphaproteobacteria and Deltaproteobacteria - Reds), Cyanobacterian (classes Oxyphotobacteria, Melainabacteria and Sericytochromatia - Golds) and Atribacteria (classes JS1 and Caldatribacteria - Purples)



**Pre-Human:** Taxonomic composition is relatively low in diversity(max < 5000 reads) during the pre-human zone (Figure 6.9). Three taxa (Spirochetes, Proteobacteria and Bacteroides) dominate Lake Nganoke's lake bacteria community, remaining relatively stable throughout (Figure 6.10). Chloroflexi increases during this zone from 15% to 37.5%. Indicator taxa Nitrospirae and Atribacteria display consistent ASV's per sample throughout this zone with the exception of a peak (>1.5%) at approximately AD 1125 (Figure 6.12). Proteobacteria shows a slight decline (26% to 22%), with the class Alphaproteobacteria the main taxa present. Of particular importance, there is a small abundance of ASVs for Cyanobacteria and no proteobacteria class Gammaproteobacteria are seen within the pre-human zone.

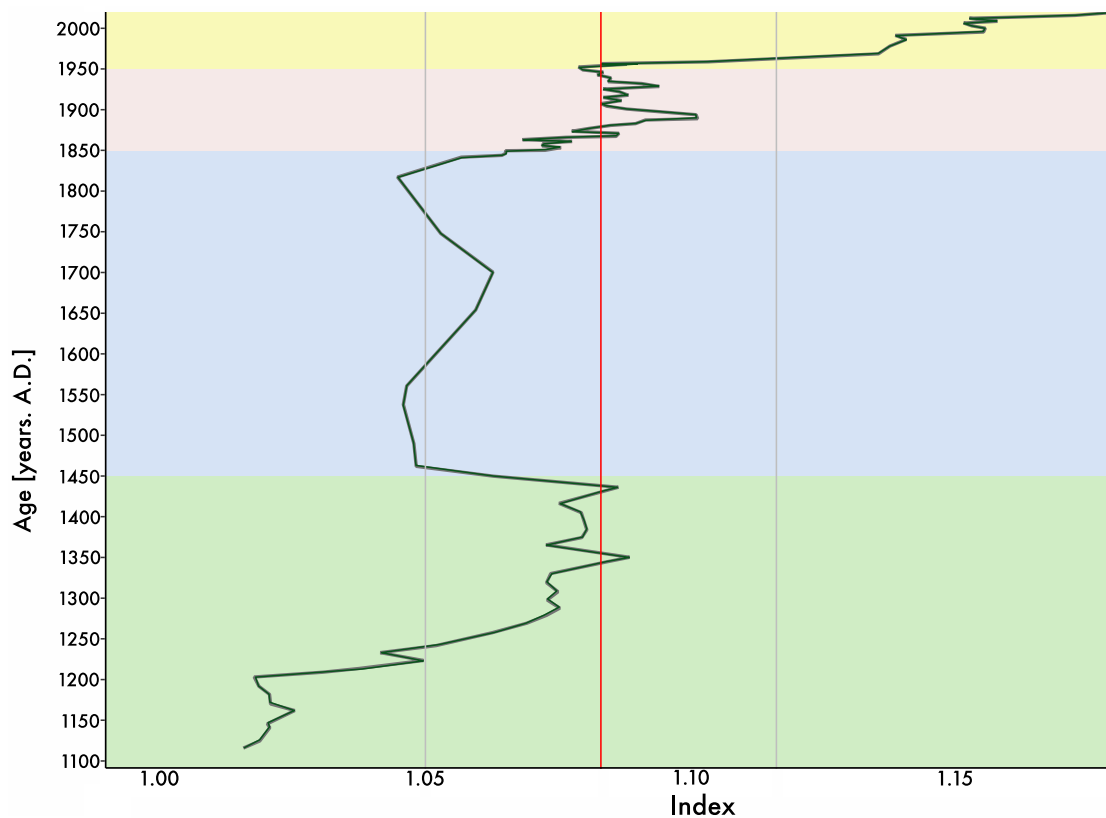
**Māori:** Taxonomic composition during the Māori zone is relatively low diversity and consistent with reads generally below 5,000 (Figure 6.9). Percentages of taxa largely stay the same across the zone with the exception of Chloroflexi, which decreases slightly (35% to 29%) from AD 1450 to AD 1500) (Figure 6.10). Indicator taxa Thermodesulfobirionia increases from 0.3% to 5% by mid AD 1800, while Alphaproteobacteria remains relatively constant ~17% within the zone (Figure 6.12). No ASV's of Cyanobacteria are present from AD 1450 to 1850.

**European:** The European zone is characterised by a noticeable increase in taxonomic composition (5,000 to 7,600 reads), with Proteobacteria displaying a largest increase (1,500 reads to 4,300; Figure 6.9). Verrucomicrobia (2% to 6%) and Proteobacteria (18% to 30%) change dramatically, exhibiting increases within the zone. Conversely, Spirochetes (17% to 2%) and Chloroflexi (25% to 18%) display declines in percentage per sample from AD 1850 to AD 1950 (Figure 6.10). Concerning indicator taxa, Thermodesulfobirionia experiences a rapid decline post AD 1850 (~1%) before increasing to 7.5% by the start of AD 1950 (Figure 6.12). Both Gammaproteobacteria and Alphaproteobacteria display increases throughout this zone (Alphaproteobacteria from 20% to 40% and Gammaproteobacteria from 2% to 20%). Of significance, this is the first time Gammaproteobacteria is observed within Lake Nganoke's sedimentary records. While percentages of Cyanobacteria are relatively low (>0.01) pre-AD 1930, percentages of Oxyphotobacteria increase to 0.75% by the start of AD 1950. Conversely, measurements of JS1 decrease post AD 1850 to approximately 0.03%.

**Intensification:** Taxonomic composition remains high (>7,500 reads) throughout the intensification period, with little change in the abundance (reads) of species (Figure 16). Observed ASV's drop rapidly following AD 1950, from 4,000 to 3,500, falling again in the early 2,000s to 2,200 ASVs per sample (Figure 6.9). Percentages of Verrucomicrobia, Proteobacteria, Chloroflexi and Actinobacteria remain relatively constant throughout the timeframe, with Spirochetes declining to 0 reads per sample by 2019 (Figure 6.10). Indicator taxa *Thermodesulfobrionia* remains relatively stable (~8.5%) throughout the intensification zone, as does Gammaproteobacteria (~20%) and Alphaproteobacteria (~40%) (Figure 6.12). Oxyphotobacteria increases post AD 1950, peaking at ~2.2% in AD 2000 before declining to 0.5% by 2019. No values of JS1 are presented within the intensification zone.

### 6.3.2 HYPERSPECTRAL IMAGING

The 660/670 RABD averages are well below the mean for the duration of Māori zones with an index range of 1.02 to 1.06 RABD at ~680 nm (Figure 6.13). Peaks are recorded at AD 1340 and AD 1420, reaching levels over the 1.803. During the European period, the levels increase above the mean to a peak above 1.10, rising sharply during the Intensification Zone, to end at 1.18 by 2019.

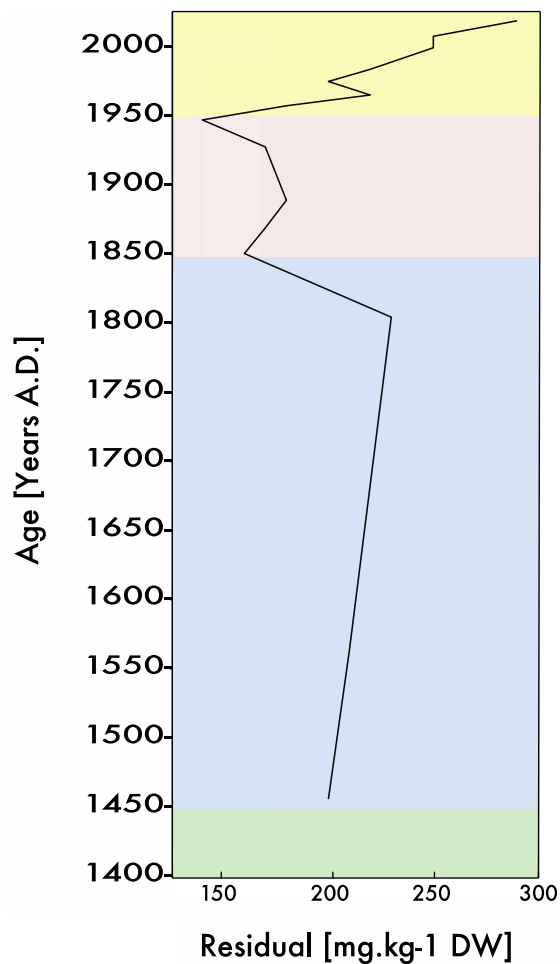


**Figure 6.13 Hyperspectral ratio of the 660/670 nm band (green) in Lake Nganoke with mean value of 1.803 (red) and standard deviation of 0.033 (grey).**

In order to estimate chlorophyll content, measurements are taken of the reflective absorption band at ~660/670 nm, with the data averaged at each depth (1 cm) to minimise noise.

### 6.3.3 PHOSPHOROUS FRACTIONATION

The residual phosphorous fraction profiles with age are presented as this is the least mobile fraction (Figure 6.14). The residual fraction trends upwards ( $>200 \text{ mg kg}^{-1} \text{ DW}$ ) until AD 1800 before dropping dramatically at the start of the European period. A sharp increase in the intensification period is observed post AD 1950 rising from  $150 \text{ mg kg}^{-1} \text{ DW}$  to  $290 \text{ mg kg}^{-1} \text{ DW}$ .



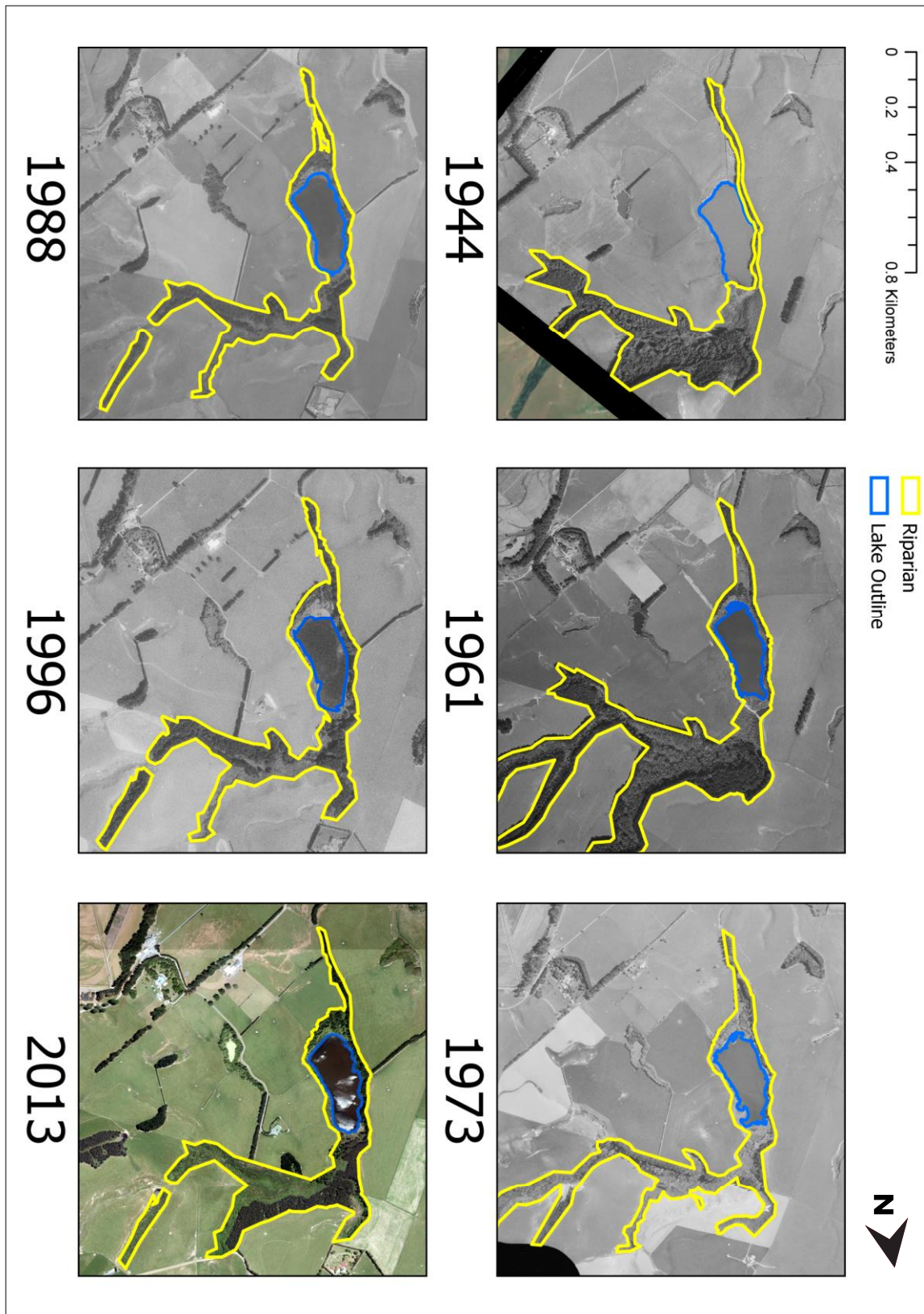
**Figure 6.14: Fraction of residual P**

#### **6.4 AERIAL PHOTOGRAPHY**

Historical air photography provides evidence of two strips of riparian zones in the Lake Nganoke catchment (Figure 6.15). The first, lake marginal riparian is located at the southern end of the lake and is the major area to receive planting and growth during the timeframe captured by available aerial photography. A lack of vegetation indicates the riparian as absent in AD 1944 and present in AD 1962. Growth of the lake marginal riparian is hence defined as post AD 1944 with maturity established by AD 1973, after which its size remains constant through to ~AD 2013. Average width is defined as 5 m.

The second, stream marginal riparian is present prior to ~AD 1944, and located to the south east of Lake Nganoke and covers the entire fluvial catchment. The composition of the riparian is mainly native with some plantation pine. The catchment riparian reached maximum size in AD 1961, with an average width of 30 m. Large amounts of deforestation from ~AD 1961 to 1973 lower the number of tall trees within the catchment

and therefore size. Changes in *Pinus* pollen abundance can provide some insight into the timing of maturity reached within this zone.



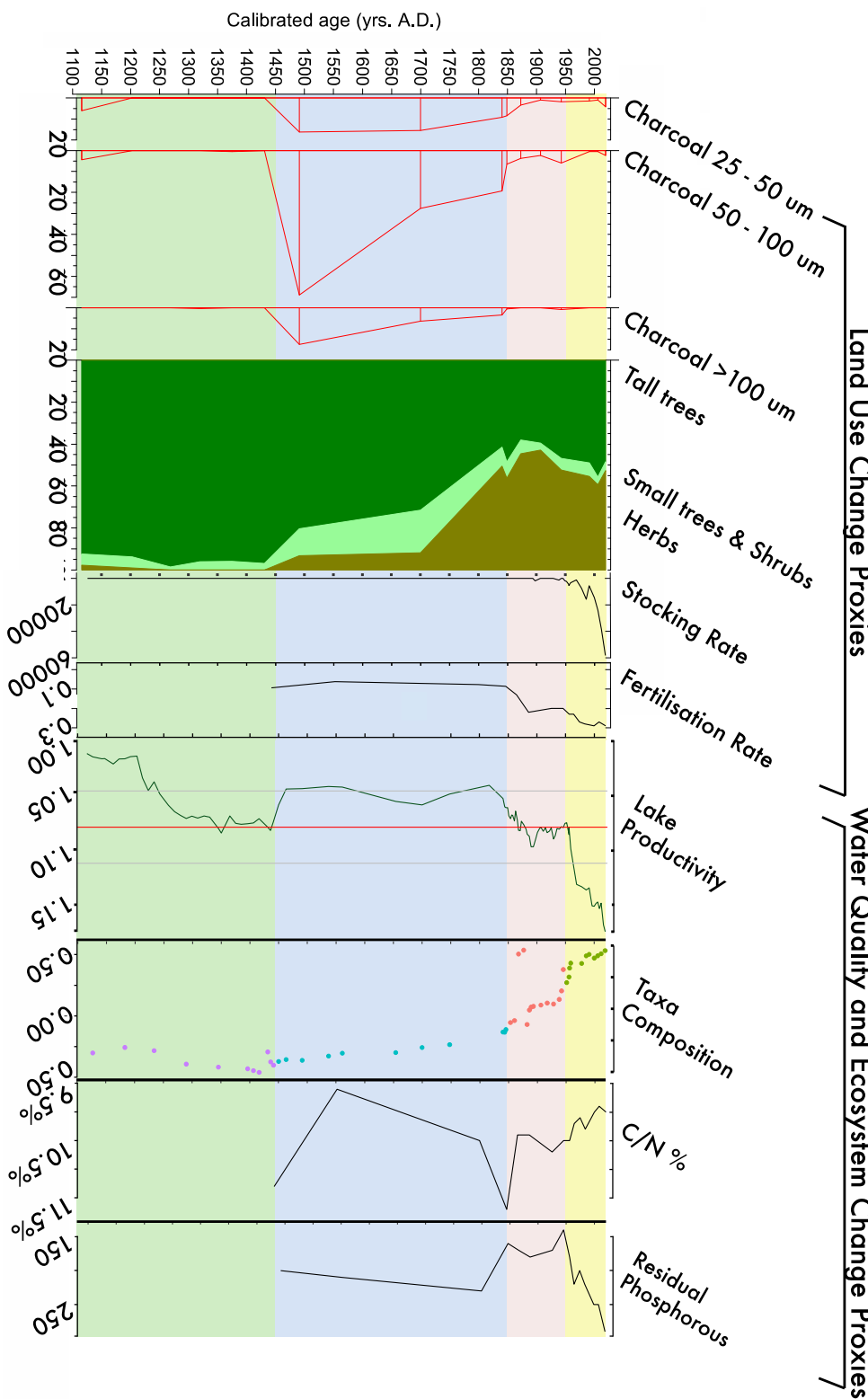
**Figure 6.15: Historical air photographs of Lake Nganoke (blue) and riparian change (yellow) from 1944 to 2013**



## **7 CHAPTER 7: DISCUSSION**

Chapter 7 aims to reconstruct environmental change at Lake Nganoke, to examine the impacts of landuse intensification on lake ecosystems and water quality. Two hypotheses are tested, 1) whether anthropogenic landuse intensification in the Lake Nganoke catchment has resulted in ecosystem change and a decrease in water quality from pre-human baselines through to Māori subsistence, European agriculture and intensification; 2) whether the planting of a riparian zone around the lake has buffered lake water quality against agricultural intensification.

In order to achieve this aim, a chronology of pre-human, Māori, European and intensification landuse regimes within Lake Nganoke's catchment is generated, with specific reference to a high-resolution record of agricultural intensification over the last 180 years (Sections 7.1.1, 7.2.1, 7.3.1, 7.4.1). Furthermore, a record of lake water quality, ecosystem evolution and nutrient loading as a response to drivers of landuse change is investigated (Sections 7.1.2, 7.2.2, 7.3.2, 7.4.2). Following environmental reconstruction, the effectiveness of riparian zones in mitigating agriculturally enriched nutrients fluxes is assessed (Section 7.5). The discussion will relate to the summary Figures 7.1 on page 66.



**Figure 7.1: Summary figure of landuse, water quality and ecosystem change within Lake Nganoke**

From left to right, landuse proxies consisting of charcoal records, overall vegetation composition, ddPCR of ruminant faecal DNA a proxy for stocking rates and cadmium as a proxy for fertilisation. From middle left to right, water and ecosystem proxies including of the RABD 660-670nm as a proxy for lake productivity, taxonomic composition change, total nitrogen, total carbon, carbon to nitrogen ratios and residual phosphorous. Taxonomic composition is coloured based on zones, pre-human, Māori, European and intensification and calculated using the y axis variable from NMDS plot in Figure 6.11



## 7.1 PRE-HUMAN VEGETATION: PRE AD 1450

### 7.1.1 LANDUSE

Consistent with findings by Hill (1963b) and Wardle (1991), the pollen record of pre-human vegetation largely consists of tall trees. Primarily *Fuscospora* and *Dacrydium cupressinum*; this indicates the majority of the catchment was forested. *Fuscospora* is typically a light demanding canopy tree that prefers deep, fertile, well-drained soils and low elevation (Stewart & Rose, 1990). While these preferences are true for red beech, mountain beech (which are also included within *Fuscospora* pollen types) can grow to forest tree lines (Okuda et al., 2002). However, due to the land surrounding Lake Nganoke being primarily lowland plains, the predominant beech within *Fuscospora* pollen is therefore likely red beech situated at low elevation, although it is important to note that beech pollen is relatively mobile. Podocarps, of which rimu is the most common, are historically categorized as emergent, forming pure stands to the exclusion of *Fuscospora* (Wardle, 1967). The catchment during this period is therefore likely well-vegetated, with *Fuscospora* located on dry spurs and ridges while Podocarps dominate the lower hills and plains.

### 7.1.2 RESPONSE OF LAKE ECOSYSTEM AND WATER QUALITY CHANGE

Measurements of RABD 660-670 nm provide a proxy for sediment chlorophyll a that is usually interpreted as an autochthonous productivity signal (Butz et al., 2015). Productivity pre-human arrival at Lake Nganoke is relatively high for the record, on par with the lowest values observed within the European era. The single measurement of C:N ratio of ~10 at ~AD 1444 demonstrates that organic material was primarily algae in origin, with values >20 from terrigenous origin while values <10 signifying algae origin (Herczeg et al., 2001). Therefore, measurements of RABD are likely autogenic (derived from algae). Measurements of RABD 660-670 are therefore likely the result of in-lake productivity which demonstrates average rates of organic biomass production from 1300 to 1440 AD. Caution however must be taken when considering measurements of C:N and residual P as they only have one sample, located near the end of the zone boundary (AD 1440). Low productivity observed at the bottom of the core is likely related to a different depositional environment (i.e. light grey laminated silt), potentially explained through the shallow marine embayment historically observed as occupying most of the Southern Wairarapa pre 3ka. This was outlined earlier in Section 4.1.

Taxonomic composition of bacterial flora within Lake Nganoke remains constant throughout the pre-human zone, suggesting a stable environment. Indicator taxa, like Deltaproteobacteria and Atribacteria, remain constant throughout the zone. While Deltaproteobacteria's continuous relative abundance suggests nitrifying nutrient input, Atribacteria points towards anerobic conditions. It is hard to imagine a 3-m deep lake would thermally stratify; however, this could have been encouraged by wind protection from a closed-canopy forest. More likely, however, oxygen depletion was driven by in-lake productivity and algal blooms (Schindler, 2006). The nutrient source driving these blooms prompts further investigation, but, from the C:N ratios, nutrients are not observed to originate from terrestrial vegetation. One potential source could include waterfowl nutrient loading, as bird life was prolific prior to human arrival in New Zealand (Worthy et al., 2017; Cole & Wood, 2018). Higher rates of algal productivity could therefore be driven by high abundances of bird populations inhabiting Lake Nganoke, further supporting internal lake nutrient loading. Additional research into molecular markers for historic bird life would better quantify this hypothesis. Finally, higher rates of algal productivity have serious implications for the NPS-FW and restoration efforts, as not all lakes necessarily have oligotrophic baseline states (Ministry for the Environment, 2014).

## **7.2 MĀORI: ~AD 1450 – 1850**

### **7.2.1 LANDUSE CHANGE**

This zone is characterised by large increases in the influx of charcoal (>100 um), a decline in tall trees, and an increase in grasses, shrubs, *Pteridium* and *Typha*. Similar transition from closed-canopy forests to *Pteridium* and early-successive shrubs and grasses is well documented within New Zealand palynology records and interpreted to be the result of Polynesian land clearance (McGlone, 1983b; Sutton, 1987; McGlone, 1989b; Elliot et al., 1995; Newnham, Lowe, & Matthews, 1998; Byrami et al., 2002; Ogden et al., 2006; McWethy et al., 2009; Abraham et al., 2013; Walter et al., 2017). Increases in *Pteridium* linked with influxes of > 100 um charcoal found within Lake Nganoke sediments indicate localised land clearance for subsistence practices (McWethy et al., 2009; McWethy et al., 2014; Trodahl et al., 2016a). *Pteridium* is an important source of carbohydrate for Polynesian settlers. Interestingly, a further increase in deforestation can be observed post AD 1700, with an additional decline in tall trees and a continuation of charcoal burning. This may be indicative of further Māori landuse, or resettlement within Lake Nganoke's catchment or possibly the product of a sampling artefact. A higher

resolution pollen analysis on the Lake Nganoke record is required to determine the most appropriate explanation of the substantial decline in tall trees post AD 1700.

Historic Māori land clearance, however, was typically small scale and localised, intended to clear undergrowth for the cultivation of food related taxa (Wardle, 1985). Within Lake Nganoke's catchment, evidence of small localised fires are reinforced through relatively high abundances of > 100- $\mu$ m charcoal suggesting burning of local origin (Higuera et al., 2007; Oris et al., 2014) and the continued presence of tall trees which precludes large tracks of regional deforestation. Interestingly, while Podocarps are inferred to most likely have dominated the forest on the plains surrounding Lake Nganoke, they only decline marginally through the pre-human Māori transition; further supporting the argument for clearing being localised.

Paleolimnological evidence therefore suggests that Māori arrival within the Lake Nganoke catchment occurred between ~AD 1417-1551 (95% confidence interval). Although this is younger than the established date for settlement in New Zealand: ~AD 1150-1250 (McFadgen, 2003b, 2003a; Wilmshurst et al., 2004; McWethy et al., 2010; McWethy et al., 2013), it agrees with the approximate date of arrival at AD 1450 prescribed in Section 4.2. The location of relatively early radiocarbon samples southeast of Lake Nganoke (Palliser Bay: AD 1350  $\pm$  70)(McFadgen, 2003b, 2003a) and later dates northwest (Lake Pounui: AD 1500 – 1600)(Cochrane, 2017) consequently make a strong case for the progression of Māori inland from Palliser bay to the Wairarapa Valley during this period.

Trodahl et al. (2016a), however, identify an increase in charcoal in Lake Wairarapa at ~AD 1230 not seen within Lake Nganoke record. This may indicate that Māori settled further inland around Lake Wairarapa prior to the area surrounding Lake Nganoke, contradictory to the hypothesis of progression inland from Palliser bay. Whilst evidence of regional burning (one charcoal sample at <100  $\mu$ m) can be seen at AD 1130-1200 within Lake Nganoke, neither local burning, a decline in tall trees, or a growth in grasses, shrubs and bracken fern are observed (for which Trodahl et al. (2016a) have no measurements). Currently, there is no evidence (charcoal <100 microns) in Lake Nganoke for this early landscape disturbance signal. The single sample at AD 1130-1200 is therefore more likely a part of the natural fire regime.

The validity of chronology within Trodahl et al. (2016a) however must be questioned. Trodahl et al. (2016a) chronology is modelled on one radiocarbon sample of a freshwater mussel, which has the potential to move within lake sediments. Furthermore, in the case where radiocarbon ages from lake sediments have been used to date Polynesian arrival, ages have the potential to be contaminated by old carbon (McGlone & Wilmshurst, 1999d; Sutton et al., 2008; Striewski et al., 2009). The fact that the date was on a freshwater mussel is particularly problematic, as such radiocarbon targets suffer from local reservoir effects as they fix their carbon from the water column of the lake.

Dating of Polynesian arrival within Lake Wairarapa by Trodahl et al. (2016a) is therefore at best speculative. While uncertainties within Polynesian arrival within Lake Nganoke are rather large (95%: AD 1418 to 1555), the arrival in ~ AD 1250 is observed well outside this. The chronology of Lake Nganoke is developed from a range of dating techniques, and therefore provide a more parsimonious chronology of surrounding landuse change, subsequently providing a superior estimate on Polynesian arrival. Furthermore, Cochrane (2017) found a similar timing of palynological evidence in Lake Pounui identified in Lake Nganoke. Located some ~7km west of Lake Wairarapa, the lack of data concerning Polynesian settlement at AD 1230 further questions previous dating of Polynesian arrival within Lake Wairarapa.

#### 7.2.2 RESPONSE OF LAKE ECOSYSTEM AND QUALITY CHANGE

Following Polynesian arrival ~AD 1450, Lake Nganoke's ecosystem and water quality is characterised by a relatively stable bacterial taxonomic composition and a decline in algae productivity within the lake. The NMDS plots for changes in bacterial taxa composition indicates a continuous change in taxa composition across the pre-human and Māori boundary. A clear overlap between the zone boundaries implies that taxonomic composition during this timeframe could simply be a record of gradual environmental shifts. More likely however, the two oldest pre-human samples overlap within the Māori zone and could therefore be a simple issue of sampling resolution. A higher resolution throughout this period would address this.

While increases of sedimentation and in-lake productivity post Polynesian land clearance are expected (McGlone, 1989a; McGlone & Wilmshurst, 1999a), the opposite is observed within Lake Nganoke. Riparian planting by Māori may explain both the decline in sedimentation rate and in-lake productivity decline

*Cyperaceae* and *Typha* growth on the fringes of the lake post-Māori arrival (AD 1450) is likely a local response to either increased light availability at the lake's margins due to forest removal and/or intentional planting as these taxa were used by Māori. Increased plantings in lake marginal taxa limit both sediment and nutrient inputs and in turn reduce productivity. Furthermore, restrictions in nutrients entering the waterways limits algae growth and consequently the productivity of benthic organisms (Carey & Migliaccio, 2009). Additional evidence is highlighted in AD 1450 - 1550 with a clear decrease in lake productivity at the same time as terrigenous organic material and sedimentation decreases within the lake. Restrictions in the temporal resolution of TN, residual P and C:N ratios samples restrict understandings of nutrient loading pre Māori arrival (~AD 1450), limiting an informed understanding of water quality and ecosystem change post Māori arrival.

The question arises as to why marginal lake plants such as *Typha* and *Cyperaceae* increase in abundance post land clearance AD 1450. Two potential explanations exist: clearing trees from the lake edge allows more light to reach the lake, consequently promoting these plants to grow and become established, and second these taxa have cultural value to Māori, and as a result, are intentionally cultivated. The pollen record suggest that tall trees decline in the Lake Nganoke post AD 1450, which could increase light availability for the growth of lake marginal plants. However, an increase in light availability would also increase within lake productivity, which does not occur in the Lake Nganoke record. Lake marginal taxa growth is therefore probably largely driven through cultivated planting by Māori, with increased light as a result of forest clearance playing a minor role in promoting plant growth.

Further evidence for Māori planting post settlement AD 1450, is observed through indicator taxon Atribacteria remaining relatively stable. Consisting of strictly anaerobic taxa (Frank et al., 2016; Liu et al., 2019), although nitrogen fixing *Thermodesulfobacteria* are non-photosynthetic organisms. Therefore, an increase in this taxon is unlikely the result of changes in the tree canopy surrounding the lake, but rather evidence of atmospheric nitrogen fixation. The lack in change of other indicator taxa, like Deltaproteobacteria and Atribacteria, combined with productivity (hyperspectral and residual P) further suggests that external nutrient loading within this zone is relatively low and constant, the result of nitrifying bacteria providing a small nursery effect for nutrient dependant taxa. Declines in lake productivity (hyperspectral) are therefore more

likely the result of decreases in nutrient rich sediment entering lake Nganoke; further supporting the hypothesis of buffering provided by the growth of marginal vegetative taxa.

The ecosystem within lake Nganoke during the Māori zone is therefore likely driven by a combination of in-lake nutrient cycling, with small increases in external nutrients post AD 1550. Decreases in sedimentation directly post settlement are likely the result of increases in marginal lake plant taxa, while low levels of algae productivity indicate healthier water quality levels than recorded during the pre-human era. This suggests that apart from the initial impact of land clearance, Māori intentionally or unintentionally “managed” nutrient loads entering the lake relatively well.

### **7.3 EUROPEAN: ~AD 1850-1950**

#### **7.3.1 LANDUSE**

Peaks in *Pinus* abundance and further declines in *Pteridium* and *Cyperaceae* occur in the early to mid 19<sup>th</sup> century and are consistent with European land clearance shortly after settlement in New Zealand ~AD 1840 (Elliot et al., 1998; Elliot, 1998; McGlone & Wilmshurst, 1999b; Horrocks et al., 2007). Historical evidence of European arrival within the Wairarapa region is well documented, consisting of first arrival at Palliser Bay in AD 1770, sheep farming production in AD 1844 and regional settlement of the Wairarapa Valley by AD 1853 (Hill, 1963b; Leach & Anderson, 1974). Considered a strong indicator for European arrival (Byrami et al., 2002; Ogden et al., 2006; Abraham et al., 2013), *Pinus* is first observed within Lake Nganoke at ~AD 1846. However, due to the resolution of pollen chronology, the exact introduction of the taxon cannot be constrained further than post ~AD 1841 (95% CI: 1700-1932).

Surprisingly, macroscopic charcoal flux >100 µm diminishes rapidly following European arrival, indicating low amounts of land clearance by fire. Regionally, charcoal influxes <100 µm show less of a reduction, likely influenced by a low sampling resolution in the palynology over the Māori to European transition which precludes any ability to draw conclusions about exactly when the last phase of deforestation occurred. Despite this limitation, further evidence of European arrival on the regional landscape is observed in additional changes in vegetation post ~AD 1850. Declines in *Fuscospora* are likely further evidence of deforestation for agriculture. Increases in Poaceae (made up of both native and exotic species) and exotic pollen *Rumex* and *acetosella* are suggestive of land

clearance for cultivated pastoral development and most probably regional (Kissling et al., 2005). With this in mind, it is estimated that while first European settlement occurred ~AD 1840, large scale European arrival is slightly later at approximately ~AD 1850 (95% CI: AD 1809-1870), which fits with the date of regional settlement prescribed from the literature (Cochrane, 2017; Waters, 2018).

Previously identified as a major problem associated with anthropogenic landuse change, land clearance destabilises the catchment, increasing erosion. Moreover, excess available nutrients can be found within the surrounding soils as they are not absorbed by diminishing vegetation (Fahey & Jackson, 1997). The effect is even greater when extra nutrients are added to the soil via animal effluent and fertiliser additions (Harper et al., 1993). Well documented, excess nutrients in the water body can cause eutrophication and internal nutrient loading, consequently degrading lake health (Canfield Jr & Hoyer, 1988; Correll, 1998; Smith et al., 1999; Smith & Schindler, 2009).

Reconstructions of agricultural stock around Lake Nganoke detect external nutrient loading post ~AD 1900 and again in ~AD 1940 via ruminant gut flora. While increases in stocking rates within the European zone are consistent with historical records for the Wairarapa region (for further information see Chapter 2, Section 2.2), declines are not (Anastasiadis & Kerr, 2013a). An explanation for fluctuations in stocking rate (ruminant gut flora) not matching historical stocking increases involves the taphonomy of faecal material. While faecal material traveling vast distances through a watercourse is well documented (Oliver et al., 2005), the resultant concentration deposited in a sedimentary basin is poorly understood (Vogel et al., 2007). Increases in ruminant detection are therefore most likely a local signal of stock that have come into direct contact with the lake edge or body. A comprehensive history of the catchments stocking rate would provide a separate chronology, and potentially a check on the validity of ddPCR as a proxy for stocking rates.

Levels of cadmium within Lake Nganoke sediments increase post arrival in AD 1850, remaining stable from AD 1890 to 1948. Another form of agricultural nutrient fertilisation, elevated concentrations of cadmium as a result of excess superphosphate application is well documented within New Zealand (Bramley, 1990; Ma & Rao, 1997; Taylor et al., 2017). Superphosphate was not imported into New Zealand until AD 1867 and national manufacturing only commenced in AD 1881, which seems at odds with the

increase in cadmium from AD 1850. However, when considering uncertainty in the chronology, the first increase of cadmium overlaps with the date of importation. Furthermore, low levels of cadmium pre AD 1950s are likely the result of fertilisation accessibility restricted to flatter areas. Early increases in cadmium levels post European arrival are likely the result of first superphosphate fertilisation in and around the low-lying areas of Lake Nganoke's catchment. Similar to the application of ruminant faecal DNA as a proxy for stocking rates, further validation of cadmium could be examined with a thorough history of fertilisation within the lake's catchment.

### 7.3.2 RESPONSE OF LAKE ECOSYSTEM AND QUALITY CHANGE TO DRIVERS OF LANDUSE CHANGE

Increases in lake productivity are often linked to increased fertilisation and stocking rates (Burns, 1991; Abell et al., 2010; Finlay et al., 2013; Nielsen et al., 2014). Rising levels of cadmium suggest increasing rates of fertilisation and higher nutrient loads entering Lake Nganoke post AD ~1850. Furthermore, fertilisation levels correlate with increases in algae productivity and Res P, suggestive of a close relationship throughout the entirety of the European zone. Lake Nganoke algal productivity within the European zone is therefore likely driven by the advent of increased agricultural landuse post-European settlement.

Within Lake Nganoke dramatic increases of the macrophyte *Potamogeton* can also be observed post European arrival. Two explanations arise for the appearance of the taxon: first, increased nutrients drive in-lake productivity, promoting macrophyte growth; and second, increased water clarity promotes suitable habitat for the taxon. While there is no evidence for increased water clarity (such as decreases in bloom-forming bacteria), highly productive lakes tend to have increased phytoplankton growth consequently blocking light from reaching lake macrophytes (Kane et al., 2014). Although improved water clarity cannot be ruled out, it is more likely that *Potamogeton* growth within Lake Nganoke is the result of agricultural nutrient loading and therefore supports the argument for nutrient driven in-lake productivity.

Clear changes in bacterial taxonomic composition (NMDS axis 1) are highlighted at the start of European arrival in ~AD 1850, followed by a period of stability from AD 1880 to AD 1940. The trend in bacterial composition correlates well with Cd concentrations, providing a strong argument for a link between fertilisation rates, lake productivity and



bacterial community change. Further evidence of Lake Nganoke's response to European landuse can be observed through increases of indicator taxa Gammaproteobacteria and decreases in Atribacteria. Known denitrifying bacteria, Gammaproteobacteria respond to increases in bioavailable nutrients entering a water body. This response provides strong evidence of the lake ecosystem responding to changes in nutrient flux because increased nutrient loading within Lake Nganoke has already been demonstrated through proxies such as cadmium.

Furthermore, bacterial taxa observed at ~AD 1865 indicate similar configuration to those found within the intensification zone suggesting a brief phase of degraded water quality that the system recovered from. Interestingly, both minor peak in productivity and the end of the increase in Cd correlate to this taxonomic change. This is the first time indicator bloom-forming Oxyphotobacteria are observed within Lake Nganoke. Consequently, it is speculated that initial fertilisation pushed the lake towards a more eutrophic state from which it recovered when the fertilisation rate plateaued.

Atribacteria live in anaerobic environments, typically indicative of a highly productive ecosystem promoting algae growth (Nobu et al., 2016). Diminishing levels of Atribacteria therefore likely represent increasing oxygen within Lake Nganoke, contrary to the low oxygen levels associated with high algal productivity. Two reasons as to why are speculated: first, nitrogen-fixing taxa could encourage oxygen-dependant taxa through oxidised nitrification; and second Lake Nganoke's water column is easily mixed. Given that Lake Nganoke is relatively shallow (>2 m), its water column would be easily mixed by wind-generated waves on the lake surface. The final clearance of lake marginal trees during European settlement would only have promoted this process.

Increases in productivity and changing taxa composition is opposite to the that found within Māori and pre-human zones. Anthropogenic forcing within the European zone is therefore identified as having far more of an impact on the productivity and composition of organisms within Lake Nganoke than Māori and pre-human zones. This is likely the result of the agricultural landuse and increased nutrient fluxes into the lake.

## **7.4 INTENSIFICATION: ~AD 1950 - 2019**

### **7.4.1 LANDUSE**

The intensification period within Lake Nganoke is characterised by further increases

in *Pinus*, a peak of *Salix* at ~AD 2000 and little to no changes of Poaceae (native and exotic grasses). Periods of burning indicated through charcoal abundance are observed to drop post AD 1950, remaining relatively low. This suggests that the majority of land within Lake Nganoke's catchment has already been cleared for agricultural purposes with any increases in excess nutrient loads likely not the result of deforestation or land clearance. Increases in sediment Cd are indicative of both increased superphosphate application rates in the Lake Nganoke catchment. These increases are consistent with trends in fertilizer application in the wider Wairarapa and New Zealand more generally (Wendelken, 1976; Vant & Huser, 2000; MacLeod & Moller, 2006b; Moller et al., 2008; Anastasiadis & Kerr, 2013b).

Industry driven reductions in cadmium concentrations in AD 1997 potentially provide an independent way of assessing the validity of cadmium as a proxy for fertilisation. At this time the source of superphosphate was changed from cadmium rich Nauru to Western Saharan that has lower cadmium concentrations. Cadmium levels within Lake Nganoke decline in AD 1999 (95% CI: 1992 – 2013), which correlates well with the change in superphosphate source. The correlation may suggest that cadmium is a reliable proxy for reconstructing historical fertilisation. While this decrease could be representative of fertilisation decreases at a local scale, the timing and size make it more likely to be representative of changes in cadmium abundance. Large increases of cadmium in the surface sediment could be evidence of a degradation signal, as values begin to rise contrary to the evidence of stabilising fertilisation use post AD 2000.

Further evidence of excess nutrients as a result of agricultural intensification are highlighted through increases in ruminant gut flora post AD 1950. Detections of ruminant faecal DNA within Lake Nganoke are consistent with records of increasing stocking rates post AD 1950 (Awarua-o-Porirua, 2018; Drewry et al., 2019). Numbers of stock within New Zealand is generally recognised as rapidly increasing post AD 1950 (Winder, 2009). While ddPCR and independent evidence are consistent with one another, further work that compares the DNA proxy with actually local stocking rates are required before conclusions about the robustness of the proxy can be drawn. Escalating numbers within the catchment are therefore more likely representative of over-saturation within the catchment, accounting for low levels identified pre ~AD 1950. This indicates there is probably some threshold stocking rate above which the proxy becomes reliable.

Therefore, the general consensus and evidence provided by this reconstruction aligns with the theory of increased stocking and fertilisation rates as the main cause of lake ecosystem and water quality degradation (Abell, Hamilton, et al., 2011; Dymond et al., 2013; Nielsen et al., 2014).

#### 7.4.2 RESPONSE OF LAKE ECOSYSTEM AND QUALITY CHANGE TO DRIVERS OF LANDUSE CHANGE

Cadmium (a proxy for superphosphate fertilisation) and ruminant faecal DNA follow a comparable trend to that of lake productivity, residual phosphorous, and bacterial community composition (NMDS). The importance of fertilisation as a driver for lake ecosystem and water quality change becomes apparent, as increases in external loading (fertilisation) can directly be linked to the growth of in-lake algae and subsequently productivity (Dodson et al., 2000; Salmaso, 2010; Norton et al., 2012; Thackeray et al., 2013). Furthermore, increased productivity has increased bioavailable P in the surface sediments and initiated a positive feedback that is driving the lake towards a more eutrophic state (Woodward, 2013).

Thriving in anaerobic conditions, the decline of Atribacteria to undetectable levels post AD 1950 further supports increasing oxygen within the water column. This is reinforced by ASVs of denitrifying Gammaproteobacteria increasing dramatically at the start of the zone before plateauing. Surges in the abundance of bloom forming Cyanobacteria along with increases in the productivity is an unequivocal indication of decreased water quality (Ministry for the Environment, 2014). Furthermore, observations of Oxyphotobacteria, a denitrifying bloom-forming cyanobacteria, increase substantially post AD 1950 before decreasing post AD 1990. Interestingly, both *Potamogeton* and Oxyphotobacteria peaks match nicely with stocking rates inferred from ruminant DNA (AD 1960). Potentially, this supports the argument that increased fertilizer application and stocking rates have amplified nutrient loading in the lake, with nutrient loading resulting in a wholesale shift in the ecosystem. The implication is that without immediate action, Lake Nganoke may continue to degrade into the future.

Declines of cyanobacteria abundance post AD 2000, however, indicate potential improvements in the degradation of lake quality. Issues, therefore, arise over the status of lake health over the last 20 years. High levels of lake productivity indicate further degradation to eutrophic conditions, yet Cyanobacteria which thrive in these conditions

decreases. Additionally, nitrogen-fixing taxa *Thermodesulfobrionia* increase post AD 1950 despite increases in external nutrient fluxes. Further studies into known water quality markers such as chironomids or diatoms could potentially shed some light on this puzzling phenomenon.

## **7.5 ASSESSMENT OF THE EFFECTIVENESS OF RIPARIAN ZONES IN MITIGATING AGRICULTURALLY RICH NUTRIENTS**

While the documentation of riparian zones is well investigated within literature (Ilhardt et al., 2000; Gold et al., 2001; Johansen & Phinn, 2006), factors influencing their ability to mitigate agricultural nutrients is often a point of contention (Rabeni & Smale, 1995; Thenya et al., 2006; Wondie, 2010). Furthermore, the ever changing composition of riparian zones makes understandings of buffering nutrients complex as a wide range of factors play a role in influencing the success in mitigating the negative effects of excess agricultural nutrients entering water bodies (Sabo et al., 2005). Riparian classification of Lake Nganoke through aerial images is therefore a rather simplistic assessment of riparian buffering, providing little understandings of the dynamics concerning taxa composition (Rowlinson et al., 1999; Baker et al., 2006; Johansen et al., 2010). Assessing the growth and maturity of Lake Nganoke's riparian are consequently limited to a presence or absence based approach, with maturity established by a lack of change over the remaining imagery. None the less, the aerial photo record allows to map change in the riparian zone both around the lake and in the catchment over decadal timescales. This is an unrivalled perspective.

### **7.5.1 THE EFFECTIVENESS OF RIPARIAN GROWTH IN MITIGATING AGRICULTURALLY RICH NUTRIENTS WITHIN LAKE NGANOKE:**

Riparian growth within Lake Nganoke is observed to have no impact on the trajectory of lake water quality or ecosystem change. Dramatic increases in fertilisation (Cd) and in-lake productivity (Chl a) are associated with rapid changes in the bacterial community composition (MDS axis 1) between AD 1950 and 1970. The riparian zone reaches its maximum aerial extent in ~AD 1961, with both the average stream (30 m) and lake marginal (5 m) riparian width remaining constant from ~AD 1970. Lake Nganoke experiences an ongoing eutrophication trend, despite riparian reaching its maximum extent and remaining constant from ~AD 1961.. Therefore, the alternate hypothesis outlined in Section 3.1 is rejected.

While riparian zones within Lake Nganoke have not been found to be an effective buffer of agriculturally rich nutrients, multiple studies concerning fresh water lake degradation indicate otherwise (Choi, 1970; Addiscott, 1996; George, 2009). Timescales over which these studies occur, however, are extremely small covering no more than 3-5 years of monitoring. As highlighted by variations in lake ecosystem and productivity within Lake Nganoke, changes within lake systems can take decades if not longer to manifest, with the effects of agricultural nutrients not always immediate within resilient lakes. As such, further research is needed to investigate the lag times in responses to riparian mitigation. The result implies that conventional monitoring studies that only represent a short snapshot of time may not be representative of long term trends. The paleo record within this thesis does however capture the full time frame of riparian growth and maturity, providing a temporal extent far beyond current monitoring methods.

One potential explanation for the poor performance of riparian planting at Lake Nganoke's the narrow extent of the lake marginal riparian (less than 5 m in some places with a maximum of 15 m). Dodson et al. (2005) highlight changes in 73 small and shallow lakes within Wisconsin, USA, finding that agricultural sites with wide riparian buffer strips greater than 30 m had significantly healthier lake ecosystems than those with narrower strips. Aguiar Jr et al. (2015) highlight a minimum width of 36 m as successful in reducing agriculturally rich diffuse source contaminants entering rivers. They measured a range of widths from 12 to 60m. Within New Zealand, investigations into riparian effectiveness are almost all exclusively river focused, with Baillie and Neary (2015) finding 42% of buffer zones were >5-2 m wide in a degrading stream while Greenwood et al. (2012) identified 62% as 5 m or less. This is consistent with findings by Parkyn et al. (2000) who suggested that areas of planting less than 5 m wide are not likely to mitigate agriculturally rich landuse.

Generally, the wider the planting the greater the filtration ability (Fennessy & Cronk, 1997), and although the minimum width is often a point of contention, 30 m is commonly referred to. Reasons for this primarily include the difficulty in discerning the distinction between the riparian zone and the adjacent landscape, for which each case is site dependant (Hansen et al., 2010). In the case of Lake Nganoke, the stream marginal riparian width likely meets best practice specifications (>30 m) while the small part of lake marginal riparian does not. The insufficient buffer width seen within the lake margin is therefore likely the main contributing factor in the increased water degradation

experienced within Lake Nganoke. The nature of Lake Nganoke's data suggests the importance of continuous minimum width standards for riparian planting because a minimum of the recommended width seems to have had an impact while the recent smaller riparian has not.

Riparian mitigation of terrestrial nutrient fluxes appeared to function much more effectively during the Māori zone compared to the European and intensification zones. Evidence of riparian mitigation following Māori settlement is observed through increases of marginal wetland taxa *Typha* being co-temporal with a decrease in lake productivity and sedimentation rate. Furthermore, decreases in abundances of *Cyperaceae* and *Typha* post European arrival imply a reduction in riparian composition, while increases in *Potamogeton* as lakebed dwelling macrophytes are unlikely to provide nutrient mitigation functions. Although riparian established during the Māori zone is observed to buffer nutrient run-off from the subsistence cultivation, riparian buffering capabilities post European settlement cannot cope with the combined agriculture/intensification and apparent decrease in riparian size. This further supports the hypothesis of riparian planting during the European and intensification zone being insufficient in size to capture and mitigate nutrient rich run-off.

Evidence of increased lake quality during Māori landuse is not restricted to this thesis alone, with a recent publication by Cochrane (2017) providing further support of the buffering capabilities observed. The pollen record from Lake Pounui in the Wairarapa indicated that the aquatic plant *Isoetes* increases when Māori settled the area. *Isoetes* is an indicator for high water clarity and hence good water quality (Clayton & Edwards, 2006). Further investigation into the success of riparian planting during Māori settlement and the taxa involved using a network of pollen records from shallow lowland lakes could potentially shed some light into the use for future mitigation schemes.

## **7.6 VALIDITY OF PALEO-BASED METHODS**

Generally speaking, a higher sampling resolution creates a more robust reconstruction. Time and funding, however, plays a huge role in defining sampling resolution and number of proxies. This thesis was fortunate enough to be supported by the Lakes380 project, able to draw on a wide range of skills and resources, reducing time as well as costs. Higher resolution radiocarbon and lead<sup>210</sup> dating would create a more robust age-depth model. Furthermore, additional sampling of all proxies at a greater resolution during key

identified areas (Māori/European and the prehuman/Māori transition) would provide greater insight. A greater suite of proxies such as diatom and chironomid analysis would have complemented findings providing supplementary evidence. These proxies should form the basis of any further work.

While indicators of landuse change are investigated within lake sediments, little to no research has been conducted on the identification and quantification of agriculturally intensified landuse (Gaillard et al., 1992; Enters et al., 2008; Yang, Anderson, et al., 2008; McWethy et al., 2010). Quantitative measures of agricultural intensification derived from lake records could provide insights into the complex relationships between excess nutrient loading and declining water health within New Zealand.

The proxy ddPCR provides a quantitative amplification of ruminant gut flora simply not possible through traditional techniques. While some studies have reported an increased sensitivity of the ddPCR to detect low quantities of target DNA (Strain et al., 2013; Sze et al., 2014; Doi et al., 2015), little to low readings of ruminant DNA were found in timeframes where stocking rates were known to increase (AD 1850 - 1950). As the majority of these studies were conducted from aquatic water samples, sediment may play a different role in storing target DNA. This raises the concept of ruminant detection within sediment relying heavily on localised stock access (Giguët-Covex et al., 2019), with readings requiring high saturation in order to increase amplifications of DNA. Furthermore, the rapid increase post AD 2000 could be indicative of a degradation signal as the DNA breaks down throughout the core. Further research is required to assess whether direct livestock interaction with the lake or thresholds for detection drive the stocking signal within sediments derived from ruminant faecal DNA. This validation is crucial before the proxy is used more widely within literature.

Concerning lake sediments, cadmium is primarily analysed for the purpose of heavy metal contamination (Sander et al., 2013), with little thought to the use as a proxy for fertilisation rates. Further authentication of cadmium was corroborated with known declines in nationwide cadmium within superphosphate, both as a result of changing the rock origin and industry reductions. It was concluded that cadmium levels within sediment cores of Lake Nganoke were a fairly accurate representation of fertilisation rates within the catchment. Further research into the movement of cadmium from soil to lake sediments would provide a greater understanding of the pathway process.

While total concentrations of phosphorus in sediments are limited in predicting ecological responses to nutrient loading, fractions of phosphorus provide an integral insight into internal nutrient cycling (Kleeberg & Dudel, 1997; House & Denison, 2000; Thirunavukkarasu et al., 2000). Although this thesis predominantly focused on fractions of residual phosphorous, other forms such as reductant P and metal bound P potentially provide further insight into the complex dynamics of phosphorus loading within lakes. Caution, however, must be taken when these forms as they are known to be more mobile throughout sediment cores, and as such may not provide accurate measurements.

Within the context of Lake Nganoke, readings of residual phosphorous replicate similar trends to that of lake productivity (RABD 660-670 nm). Unsurprisingly, productivity correlates nicely with residual P and therefore provides a useful validation of the productivity signal. This relationship also has the potential to provide useful insights into internal nutrient cycling.



## **8 CHAPTER 8: CONCLUSION**

With increasing pressure to improve national water quality, it is vital to have informed understandings concerning the components of lake ecosystem resilience, drivers of degradation and anthropogenic impact. The use of paleolimnology in evaluating historical landuse, lake quality and ecosystem change, provides a temporal extent simply not possible through conventional monitoring techniques. To investigate the response of lake quality and ecosystem response to changes in landuse, this study aimed to reconstruct the past environment of Lake Nganoke from a pre-human reference state to current day. Specifically, the role of rising nutrient fluxes as a consequence of agricultural intensification was addressed and the ability of riparian zones to buffer these nutrients examined.

### **8.1 RECONSTRUCTION OF HISTORIC LANDUSE AND WATER QAULITY WITHIN LAKE NGANOKE**

To reconstruct the past environment of Lake Nganoke, an age-depth model was created. The model placed the base of the sediment cores at ~AD 1150 (95%: AD 769-1366). Human impact on the landscape was documented using charcoal, pollen, environmental DNA and geo-chemical analyses. Based on these proxies, three changes of landuse were identified within Lake Nganoke's catchment, these being Māori arrival ~AD 1450 (95% CI: AD 1417-1551), European settlement ~AD 1850 (95% CI: 1809 - 1870) and agricultural intensification ~AD 1950 (95% CI: 1948 - 1964).

Lake Nganoke's water quality and ecosystem pre AD 1450 is presumably a shallow environment with a constant catchment input of natural nutrients and dominated by tall trees and likely heavily forested. Lake algal productivity during this zone is defined as average, while bacterial composition similarly shows little to no change. Large abundances in Atribacteria indicate Lake Nganoke as anoxic, with lake marginal vegetation limiting wind mixing, promoting lake stratification. A likely explanation for high rates of algal productivity is high bird numbers providing an alternative nutrient source within the lake.

As a result of land clearance for subsistence horticulture, tall trees within Lake Nganoke's catchment diminish post Māori arrival ~AD 1450. Algal productivity significantly decreases post settlement, most likely the result of increased lake marginal taxa buffering nutrient rich sediment from entering the water column. This is further supported by no

change in anoxic thriving Atribacteria. Apart from the initial impact of land clearance and further evidence of burning at ~AD 1700, Māori “managed” nutrients entering the lake relatively well, with changes in productivity indicating improvements to Lake Nganoke’s water quality post AD 1450. The healthier water quality and ecosystem during this period is thought to be the result of purposeful lake marginal planting of plant taxa such as *Typha*.

European arrival AD 1850 was characterised by increases in exotic taxa such as *Pinus* and *Salix*, a decline in *Pteridium* and an increase in fertilisation and stocking rates. Lake Nganoke algal productivity rapidly increases post settlement, likely the result of increased nutrients from further land clearance and increased agriculture. Further evidence of increased nutrients entering the water column are provided through increases of Gammaproteobacteria. The result of nutrients from fertilizer application, in-lake productivity and changes in bacterial composition, Lake Nganoke water quality likely declines post European settlement. Furthermore, a strong relationship between increases in fertilisation rates and lake productivity indicate fertilisation as a strong driver of change.

Agricultural intensification post AD 1950 further increases the amount of agricultural nutrients entering Lake Nganoke’s water column. Overall, vegetation changes very little during this period, likely indicative that the majority of Lake Nganoke’s catchment had already been converted for agricultural landuse. Increases in fertilisation and stocking rates provide further evidence that increased nutrient run-off as a result of landuse intensification degraded water quality. Algal productivity and bacterial composition continue to change dramatically post AD 1950, likely the result of excess nutrients promoting algal growth. The absence of Atribacteria suggest Lake Nganoke is oxygenated, while constant abundances in Gammaproteobacteria indicate high nutrient loading. Bloom forming Cyanobacteria peak in abundance during this zone, consistent with nutrient loading and direct evidence of a decline in water quality.

Changes in Lake Nganoke water quality and ecosystem from pre-human reference states to agriculturally intensified landuse is obvious in the Lake Nganoke record. The alternate hypothesis is accepted.

*“Anthropogenic landuse intensification in the Lake Nganoke catchment has resulted in ecosystem change and a stepwise decrease in water quality from a prehuman baseline, through Māori subsistence, European agricultural and intensification”*

While water quality does decrease from pre-human states, the decline is not immediate, largely starting from European arrival. Furthermore, water quality is observed to improve post Māori settlement, likely the result of lake marginal planting. This indicates European landuse and intensification as potentially having a greater impact on water quality within Lake Nganoke.

## **8.2 THE ABILITY OF RIPARIAN ZONES TO BUFFER NUTRIENT RICH AGRICULTURAL RUN-OFF**

A degradation is observed in Lake Nganoke water quality post riparian growth and maturity, likely the result of an insufficient buffer width. The alternate hypothesis is therefore rejected.

*“The planting of a riparian zone around the lake has buffered lake water quality against intensification of agriculture”*

The absence of robust assessment in riparian composition however makes other factors (such as nutrient flow paths, retention rates and taxa composition) a possible reason for riparian failure. A more in-depth assessment of the riparian zone within Lake Nganoke’s catchment would address this.

Regardless, the information gained through paleolimnological analysis of Lake Nganoke’s sediment has provided a better understanding of the natural lake environment. This information will enable more informed decisions on restoration targets and the future use of paleo proxies in reconstructing agricultural drivers of lake water quality and ecosystem change. It is recommended that lake ecosystem and water quality targets within Lake Nganoke be centred towards the low rates of algal productivity and sedimentation observed during Māori landuse. Finally, it is concluded that present riparian planting is insufficient to cope with nutrient loading from the current stocking and fertilisation regime.



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## 10 APPENDIX

An online digital data file has been created at:

<https://www.dropbox.com/sh/tkudy5uh8ok7qb7/AADbT9M9M4ZUIsJhUIveapzNa?dl=0>

### 10.1 A – Lake Nganoke Master Record

**Table 2: Pb210, Bomb spike, conventional 14C and bio stratigraphic age constraints for the Lake Nganoke master record.**

Calibrated radiocarbon dates (AD), minimum 95% (CI), maximum 95% CI and mean calibrated with SHCal13 (Hogg et al., 2013) and SH1-2 using (Hua et al., 2013) OxCal online.

Master Depth (cm)	Macro Type	Sample ID	CRA Ye BP	Uncertainty	Min 95% CI	Max 95% CI	Mean
1.6	Pb <sup>210</sup>	L380_NGAN O_LC2U_11 cm	Modern	8	2003	2020	2012
3.7	Pb <sup>210</sup>	L380_NGAN O_LC2U_13 cm	Modern	15	1992	2022	2007
5.7	Pb <sup>210</sup>	L380_NGAN O_LC2U_15 cm	Modern	10	1991	2011	2001
7.8	Pb <sup>210</sup>	L380_NGAN O_LC2U_17 cm	Modern	9	1981	1999	1990
9.8	Pb <sup>210</sup>	L380_NGAN O_LC2U_19 cm	Modern	20	1960	2000	1980
11.9	Pb <sup>210</sup>	L380_NGAN O_LC2U_21 cm	Modern	15	1954	1986	1970
12.6	Seed Capsule	L380_NGAN O_LC2U_1A_RC 18.5cm	Modern	NA	1953	1963	1957
13.9	Pb <sup>210</sup>	L380_NGAN O_LC2U_23 cm	Modern	26	1930	1984	1956
14	Manuka/Kanuka	L380_NGAN O_LC3U_1B_RC_20.5cm	Modern	NA	1953	1962	1955
16	Pb <sup>210</sup>	L380_NGAN O_LC2U_25 cm	Modern	17	1917	1951	1934
38.5	Pollen	Willow	80	10	1853	1890	1871
51.5	Pollen	Pine	85	20	1809	1870	1840

62.8	Leaf Fragme nts	L380_NGAN O_LC2U 68- 69	481	20	1415	1446	1431
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## 10.2 B – Full Pollen Taxonomy

	7-8	11-12	15-16	25-26	35-36	45-46	55-56	58-59	61-62	65-66	70-71	75-76	80-81	85-86	91-92	99-100
Pinus	T	28.8	29.7	29.4	18.9	13.6	9.6	3.4	0	0	0	0	0	0	0	0
Corylus	T	1.9	0.5	0.9	0	0.5	0	0	0	0	0	0	0	0	0	0
Salix	T	1.4	5.2	0.9	0.4	1.9	0	0	0	0	0	0	0	0	0	0
Rumex acetocella	H	0	0.5	0.4	1.3	0.5	2.4	0.6	0	0	0	0	0	0	0	0
Dactyldium cupressinum	T	0.9	1.9	3	3	5.2	7.8	4	15.5	17.6	15.9	14.6	20.5	17.8	21	18.6
Dactyrcarpus dactyloides	T	0.9	0.9	0	0.4	0	1.4	2.4	3.4	1.4	3.4	0.9	2.2	0	1.9	3.8
Prunopitys taxifolia	T	1.4	3.3	1.3	3	1	1.9	4.2	2.3	5.8	11.6	5.5	8.3	5.6	10	15.3
Prunopitys ferruginea	T	0.5	1.4	0	0.4	1	1.2	1.1	1.4	1.4	2.6	4.1	1.7	1.4	4.3	4.4
Podocarpus spp.	T	0.5	0.5	0.4	0	0.5	0	0	0	0.7	1.7	3.7	1.3	0.5	1	7.7
Lophozonia	T	0.5	0.5	0	0.9	1.4	2.4	2.3	1.9	0	0.9	0.9	0.4	0.9	0.5	0
Fusospora	T	11.6	11.3	13	18.5	10.5	20.4	25	44.4	50.7	60.1	65.3	60.7	70.4	55.2	42.6
Alectryon	T	0	0	0	0.4	0	0	0	0	0	0.4	0.5	0	0	0	0
Corynocarpus	T	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Elaeocarpus	T	0	0	0	0	0	0	0	0	0.7	0	0	0	0	0	0
Plagianthus	T	0	0	0	0	0	0	0	0	0	0	0	0	0.5	0	0
Pittosporum	T	0	0	0	0.4	0	0	0	0	0	0	0	0	0	0	0
Metrosideros	T	0	0	0	0	1	0	0	0.5	0	0	0	0	0	0	0
Nestegis	T	0	0	0	0	0	0	0	0	1.4	0.4	0.5	0.9	1.4	0	0
Knightia	T	0	0.5	0	0.4	0	0	0	0	0	0	0	0	0	0	0
Weinmannia	T	0	0	0	0	0	0.6	0	0.5	0	0	0	0	0	0	0
Halocarpus	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.5
Phyllocladus	S	0	0	0	0.4	0	0	0	0	0	0	0	0	0	0	1
Coprosma	S	1.4	1.4	3	0.9	1	1.8	1.7	3.9	1.4	1.7	2.3	1.3	0.5	2.9	1.6
Coriaria	S	0.9	0.5	0.4	2.1	0	1.9	3	12.1	4.7	0	0	0.9	0	0	0.5
Clematis type	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Discaria	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dodonaea viscosa	S	0	0	0	0	0.5	0	0	0	0.7	0.4	0.9	0	0.5	0	0
Acacia lucida	S	0	0	0	0	0	0	0	0.5	1.4	0.4	0	0	0	0	0
Fuchsia	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Griselinia	S	0	0	0	0	0	0	0	0.5	0	0	0	0.4	0	0	0
Hebe	S	0	0	0	0	0	0	0.6	0	0	0	0	0	0	0	0
Hedycarya	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Macropter	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Muehlenbeckia	S	0.9	1.4	2.2	1.7	1	0	0	0	0	0	0	0.4	0	0.5	0
Leptospermum type	S	0	0	0.4	0	0	0	0	0.5	0	0	0	0	0	0	0
Leucopogon	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lophomyrtus	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pseudowintera	S	0	0	0	0	0.5	0	0	0	0	0	0	0	0	0	0
Ranunculaceae	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rosaceae	S	0	0	0.4	0	0	0.6	0	0	0	0	0.5	0	0	0	1.6
Schefflera	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Myrsine	S	0	0	0	0	0.5	0	0.6	0.5	2.7	0	0.5	0.9	0.5	0.5	0
Petraxilla	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimelea	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pseudopanax	S	0	0	0	0	0	1.7	0	1.4	2	0	0	0	0	0	0.5
Cordylone	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.5
Ericaceae	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Freycinetia	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tupelia	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Quintinia	S	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asteraceae total	S	0.9	0.5	0	0.4	0	2.4	2.3	1	0	0.4	0	0	0	0	0

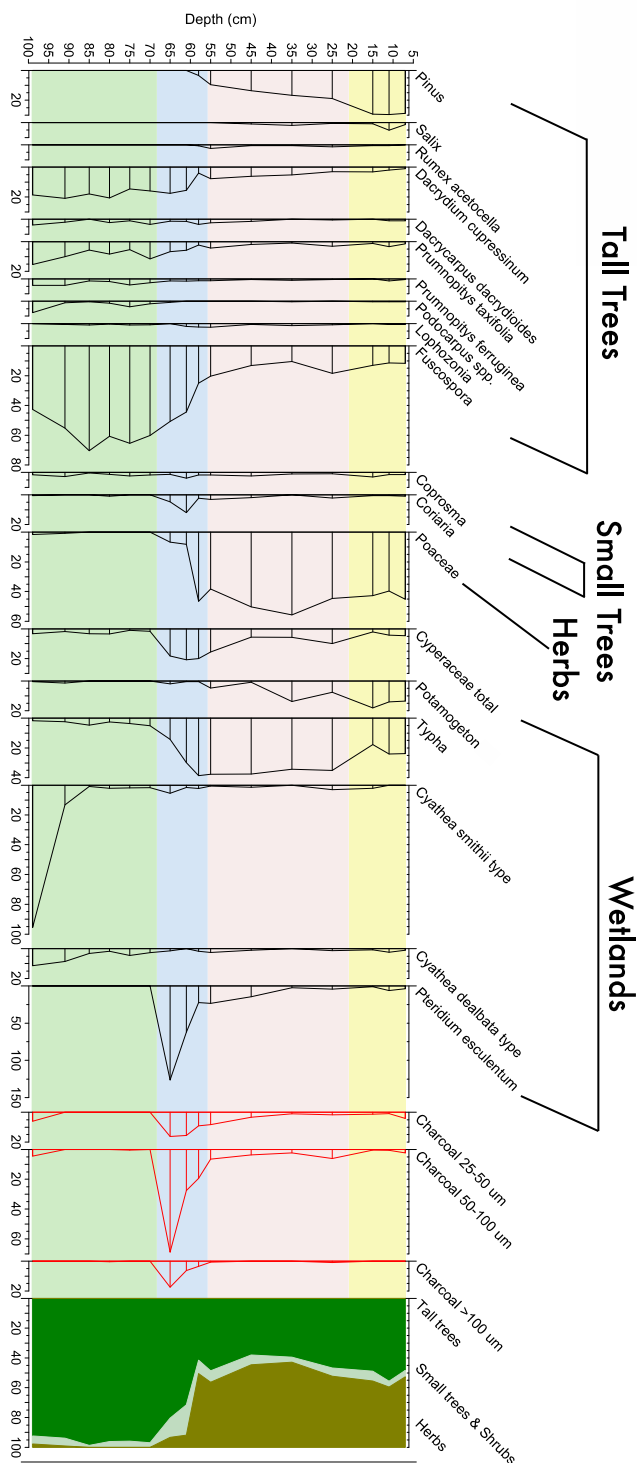


Name	Group	7-8	7	11-12	11	15-16	15	25-26	25	35-36	35	45-46	45	55-56	55	58-59	58	61-62	61	65-66	65	70-71	70	75-76	75	80-81	80	85-86	85	91-92	91	99-100	99
Charcoal 25-5°C			4.2		0.9		1.3		1.7		1		3.3		8.4		9.1		15.5		16.2		0		0		0		0		0		6
Charcoal 50-1°C			2.3		0.5		0.4		6		2.4		3.8		6.6		19.3		27.5		68.9		0		0.5		0		0		0		4.4
>100 um	C		0		0		0		0.9		0		0		0.6		3.4		6.3		17.6		0		0		0.4		0		0		0
Tall trees	P		48.4		55.7		48.9		46.8		39.5		38		48.5		41.5		71.5		80.4		97		95.9		96.1		98.6		93.8		92.3
Herbs	P		47.4		40.6		44.6		47.6		57.1		55.4		43.7		49.4		8.2		6.8		0		0		0		0		1		2.2
Small trees&	P		4.2		3.8		6.5		5.6		3.3		6.6		7.8		9.1		20.3		12.8		3		4.1		3.9		1.4		5.2		5.5
Wetlands	P		44.7		44.8		40.7		54.9		55.2		46.5		58.7		62.5		53.6		35.8		7.3		4.6		6.1		8		5.7		5.5
Ferns&ferna	P		6		9.4		1.7		6.4		4.8		17.4		28.7		31.8		77.3		145.3		3.9		0		1.7		2.3		7.6		8.2
Treeferns	P		0.9		2.4		3		4.3		0.5		3.8		3.6		5.1		1.4		6.8		4.3		6.8		4.4		4.2		21.9		107.7
Fungal	P		0		0		0		0.9		0		0		0		0		0		0.7		0		0		0		0		0		0
Charcoals	P		6.5		1.4		1.7		8.6		3.3		7		15.6		31.8		49.3		102.7		0		0.5		0.4		0		0		10.4
Pollen sum	T,H,S		215		212		231		233		210		213		167		176		207		148		233		219		229		213		210		183

**Figure 10.1 Full pollen taxonomy with depth along the top**

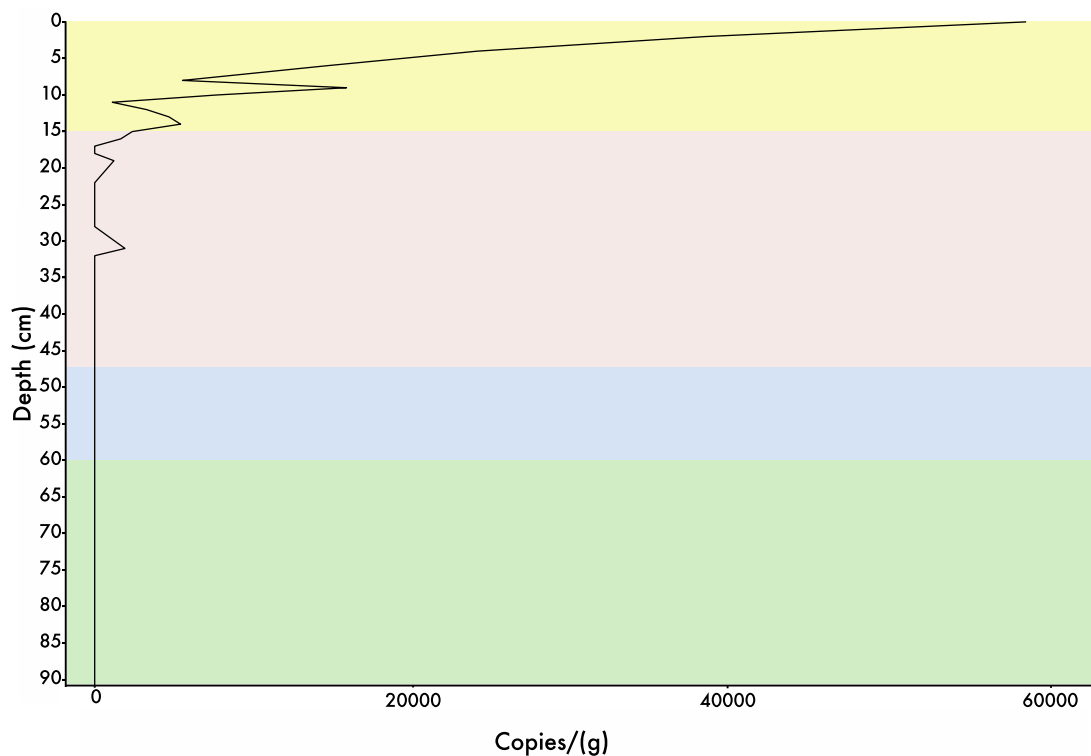


### 10.3 C – RESULTS SECTION 6.2 AND 6.3 PLOTTED ON DEPTH

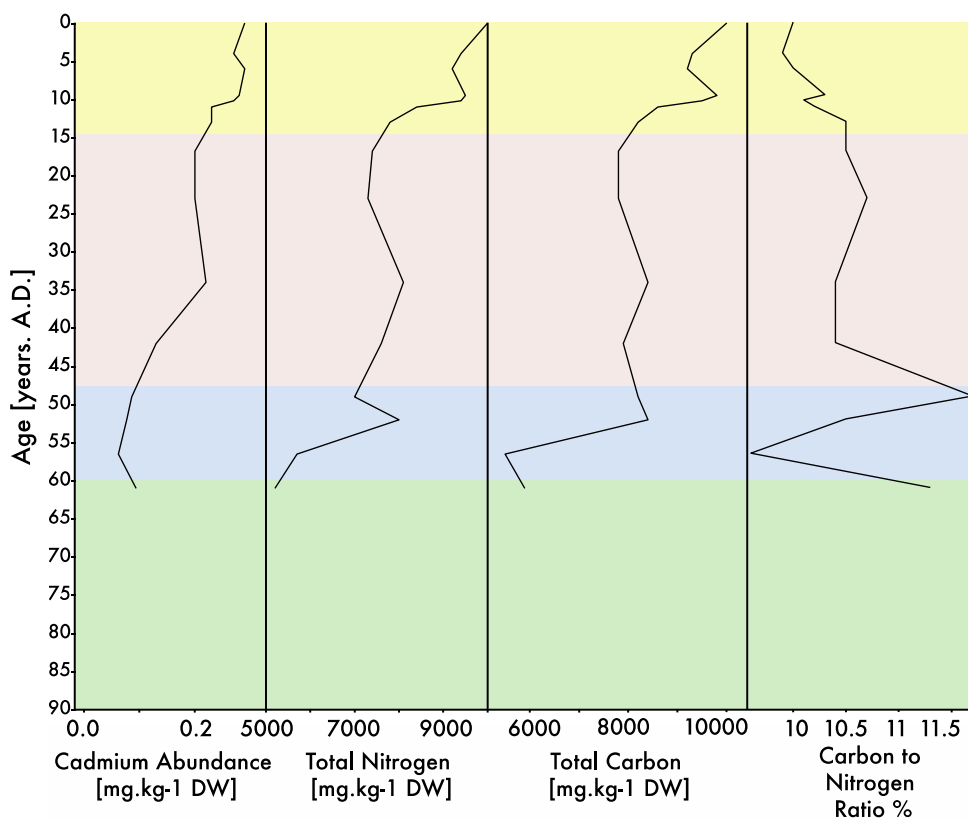


**Figure 10.2 Stratigraphy of key pollen taxa plotted on depth.**

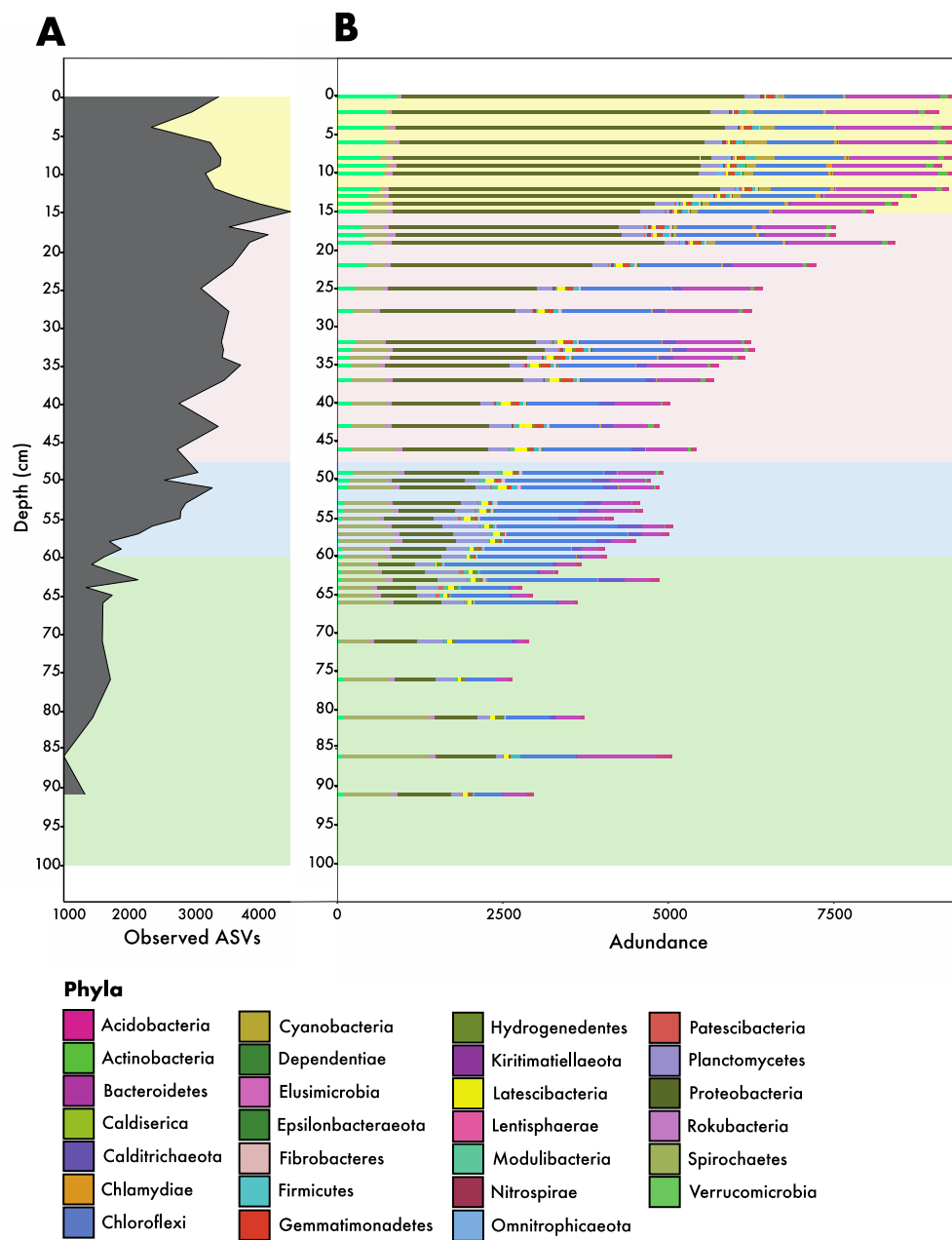
The y-axis constrains depth, while the x-axis illustrates the abundance of different taxa, with the panel on the far right showing cumulative percentages based on forest and grassland vegetation type.



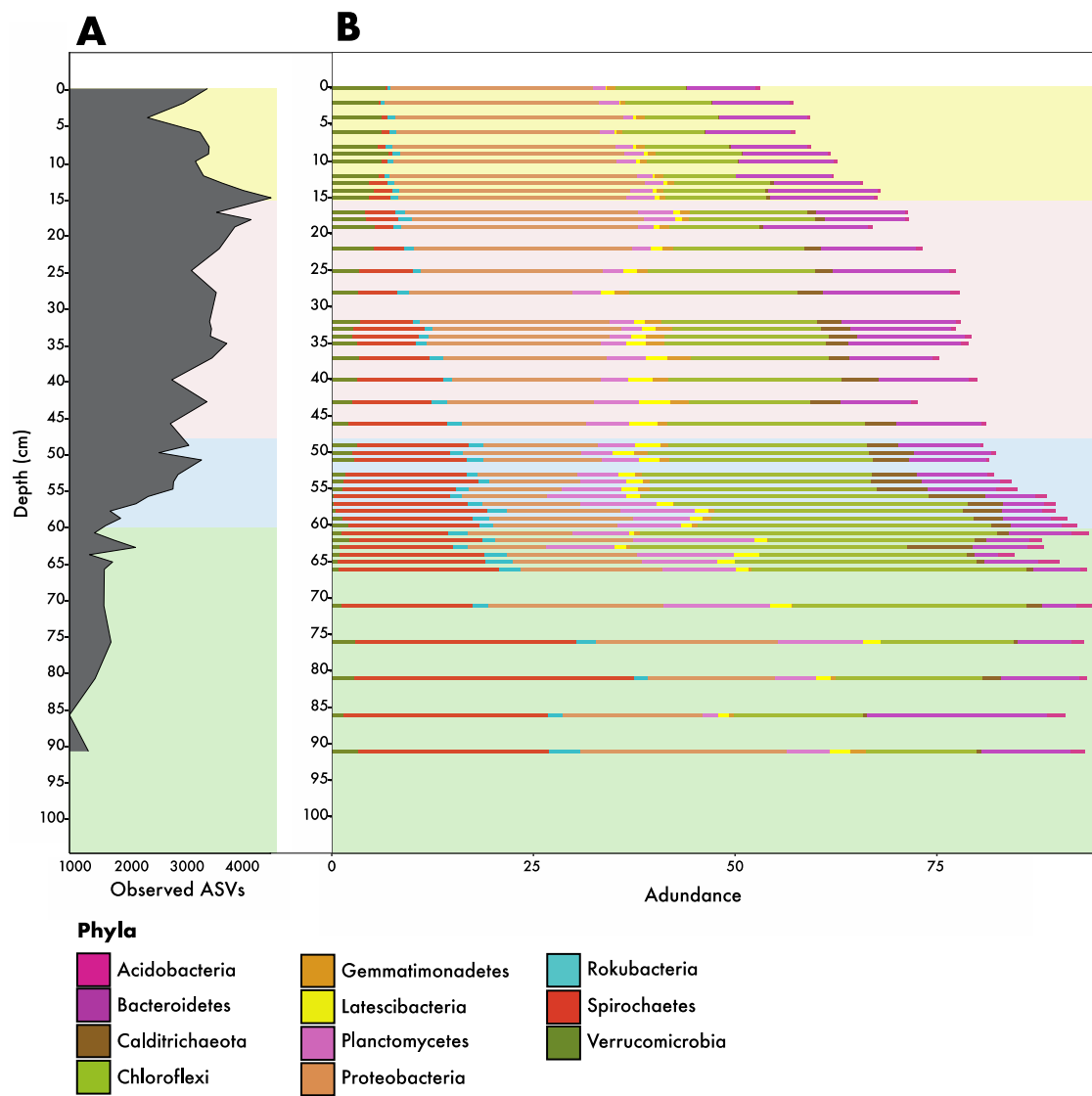
**Figure 10.3 Digital droplet ruminant PCR of Lake Nganoke plotted on depth**



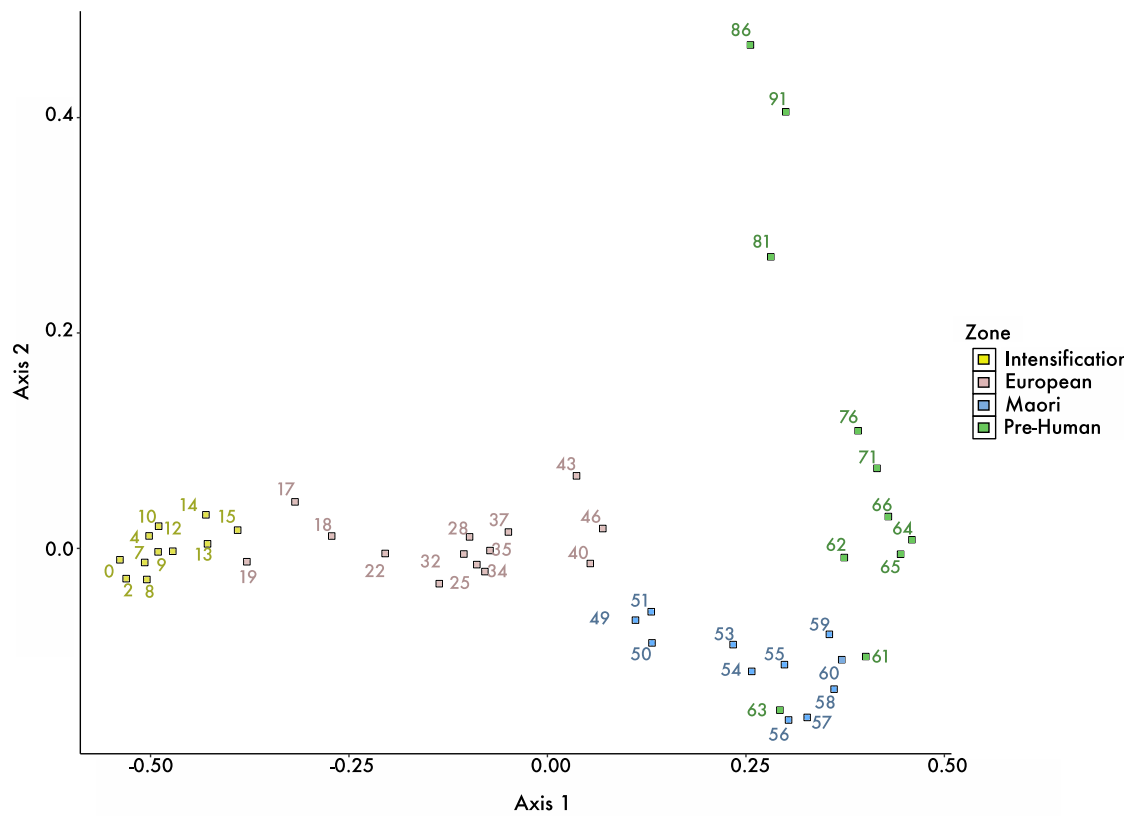
**Figure 10.4 Geo-chemical analysis of cadmium, total nitrogen (TN), total carbon (TC) and carbon to nitrogen ratio (C:N ratio) plotted on depth**



**Figure 10.5** Taxonomic composition (reads per samples) (B) and richness (Observed ASV's) (A) of Lake Nganoke at Phylum level plotted on depth

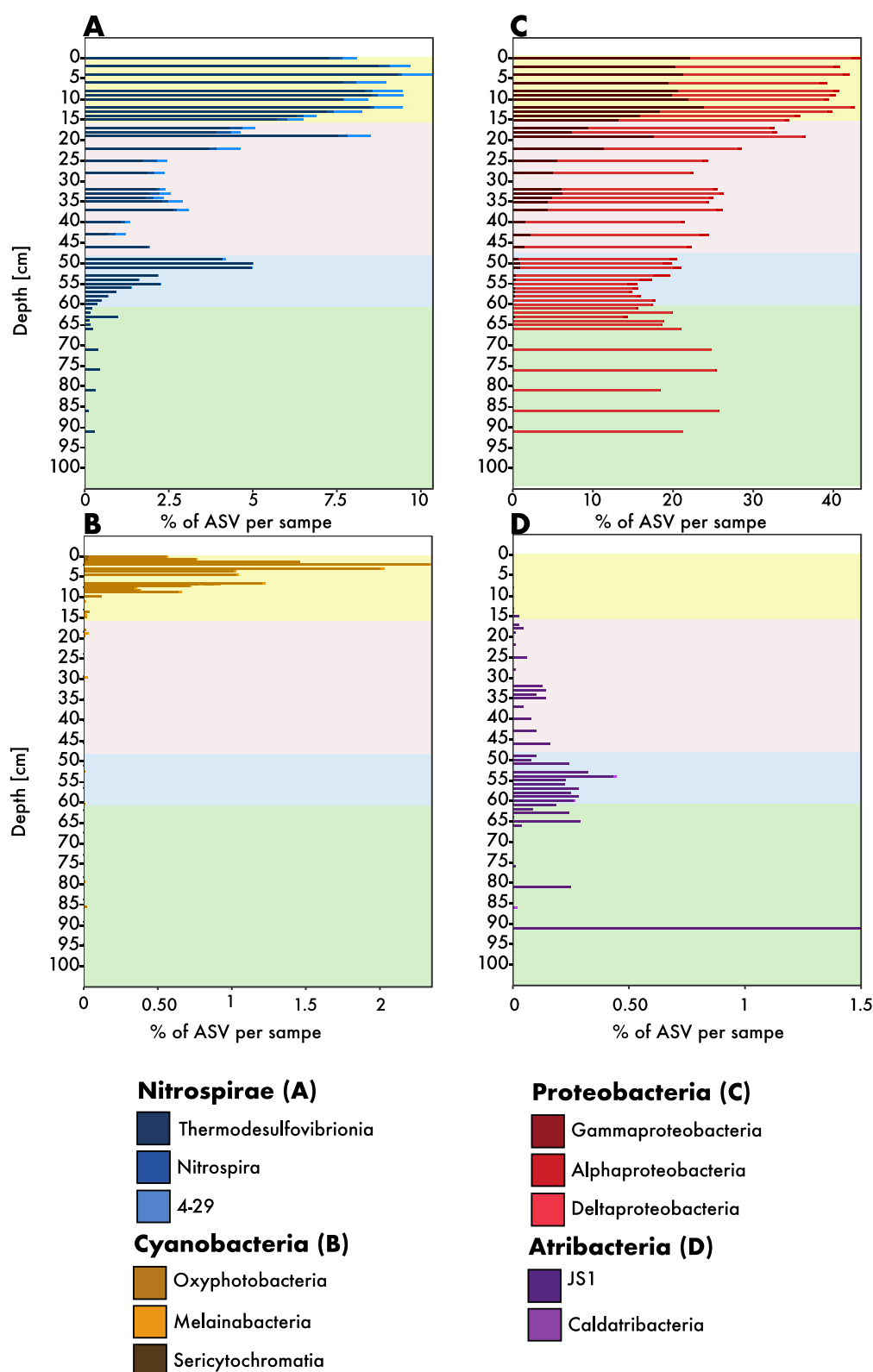


**Figure 10.6 Taxonomic percentage abundance (percentage of reads per sample) (B) and richness (observed ASV's) (A) of Lake Nganoke at Phylum level plotted on depth**  
 Important to note, only includes phyla that have a mean greater than one are included within this figure



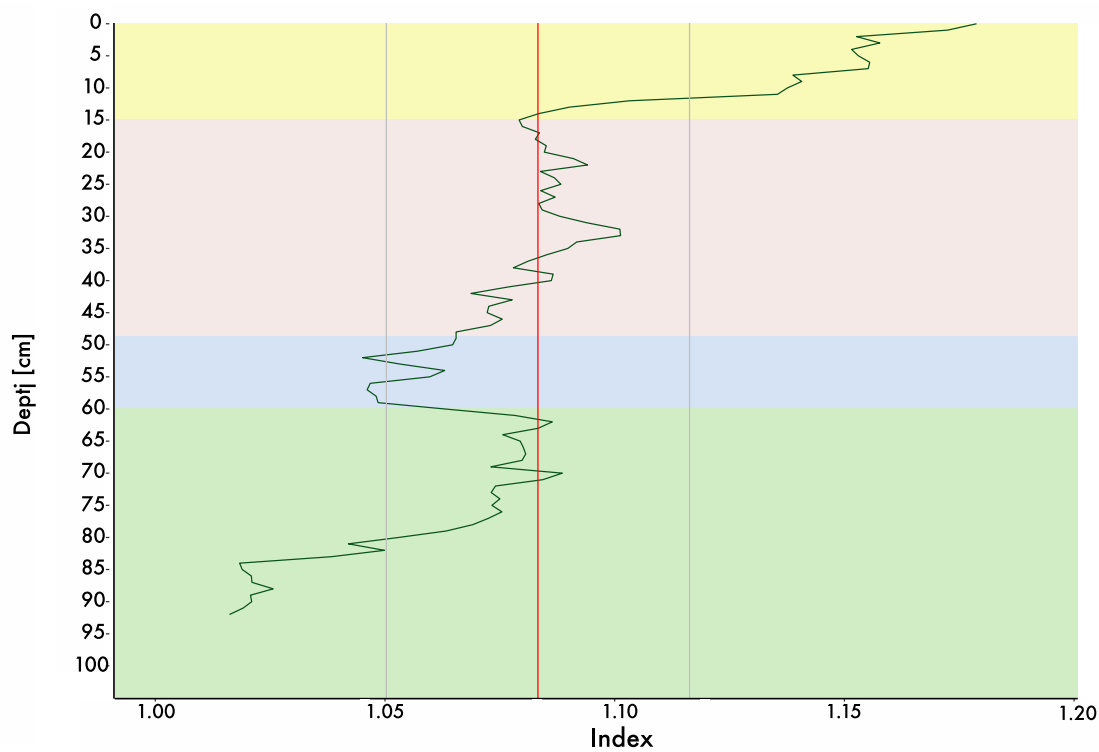
**Figure 10.7 Two-dimensional non-metric multidimensional scaling plot displaying taxonomic differences between landuse zones plotted on depth**

Intensification: yellow, European: pink. Māori: blue and Pre-Human: green. Data was transformed prior to ordination calculated using the Bray-Curtis dissimilarity distance



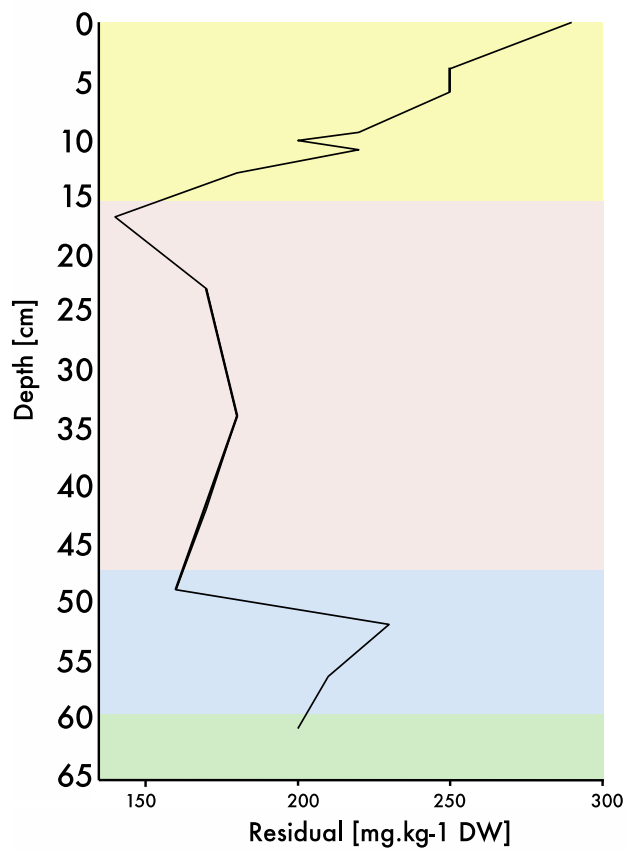
**Figure 10.8 Taxa indicative of changes in lake water quality and ecosystem identified through literature plotted on depth**

Identified taxa consist of Phylum Nitrospirae (classes Thermodesulfovibrionia, Nitrospira and 4-29 - Blues), Phylum (Proteobacteria and classes Gammaproteobacteria, Alphaproteobacteria and Deltaproteobacteria - Reds), Phylum (Cyanobacterian (classes Oxyphotobacteria, Melainabacteria and Sericytochromatia - Golds) and Phylum Atribacteria (classes JS1 and Caldatribacteria - Purples)



**Figure 10.9** Hyperspectral ratio of the 660/670 nm band (green) in Lake Nganoke with mean value of 1.803 (red) and standard deviation of 0.033 (grey) plotted on depth

In order to calculate abundance, measurements are taken of the reflective absorption band at ~680 nm, with the data averaged at each depth (1cm) to minimise noise.



**Figure 10.10** Fractions of residual phosphorous calculated by sequential extraction plotted on depth

