Using a new monitoring framework for Regional Councils to assess the integrity of prioritised ecosystems in Hawke's Bay farmland

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The long-term sustainability of biodiversity in small native forest fragments on privately-owned lowland rural land is threatened by agricultural intensification, introduced pests, inadequate fencing, and species-area effects. Biodiversity monitoring of private land plays an essential role in protecting, and enhancing, the biodiversity that remains. This is important because these private forest fragments can contain rare and critically threatened species, and are reservoirs for indigenous species that are otherwise scarce in these landscapes. Regional Councils are increasingly required to work with landowners to monitor the biodiversity in these habitats to inform management and conservation. However, there has been no standard framework that councils follow for doing this. Regional councils contracted Manaaki Whenua–Landcare Research to develop a Tier 2 Monitoring Framework for standardised biodiversity monitoring in 2020. I have trialled this framework to survey vegetation, bird and mammal biodiversity at a selection of 10 ecologically prioritised native forest sites across Hawke's Bay. My research offers insights into the framework's ability to measure ecological change, by using it to assess the ecological differences between fenced and unfenced forests. Vascular plant species richness was affected by reserve fencing, with fewer natives and many more naturalised species present at unfenced sites. Within sites, there were much more native individuals across all height tiers. For birds, it was found that fencing determined species composition, whereby there was a greater species richness and detections of naturalised bird in unfenced areas, and greater detections of native birds in fenced sites. I also investigate autonomous recordings compared to 5-minute bird counts, concluding that although they both have their merits, greater detection and unbiasedness occurs using autonomous recording devices. I will offer my reflections on the suitability of this proposed framework for more widespread regional council use and offer recommendations. My findings document the status of lowland biodiversity at these sites and provide a valuable baseline for future monitoring of ecological change in lowland forest fragments in Hawke's Bay.

Keywords: Native biodiversity, forest fragments, fencing, ecological integrity, Regional Councils, birds, vascular plants, vegetation survey, autonomous recording devices, sample intensities, lowland farmland, Hawke's Bay.

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Chapter 1

Introduction

This chapter summarises the state of environment and biodiversity nationally, with particular attention to the current knowledge of the state of Hawke's Bay's biodiversity. I review New Zealand's (NZ) approach to biodiversity monitoring, including biodiversity monitoring requirements in international agreements or obligations. The chapter identifies the challenges of monitoring in NZ and goes on to introduce a new biodiversity monitoring framework.

1.1 State of the Environment

The Anthropocene epoch has resulted in substantial environmental sustainability challenges, led by the surging human population, and, as a result, has created an overload of resource consumption that is unstable and unsustainable (Harris & Roach, 2017). Finding solutions to mitigate these challenges is of great importance for every industry, culture, and community (Secretariat of the Convention on Biological Diversity, 2014; United Nations, 2019).

The ongoing extraction and use of fossil fuels and widespread deforestation have resulted in increased concentrations of atmospheric carbon dioxide, methane, and other gases, that now causing global climate change (Chapin et al., 2000). Industrial chemical applications and runoff of nutrients from agricultural and urban systems have caused ecological changes in waterways and inshore marine ecosystems. The transformation of land into agricultural and urban systems has enhanced productivity and utilisation of ecological services but reduced ecological integrity and ecosystem services. The mobility of people and cargo has transported organisms across geological barriers, disrupting ecological communities (Chapin et al., 2000).

There is a broad range of well-established evidence indicating that due to these alterations, there is an ongoing degradation of ecosystems and biodiversity decline (Arnold, 1994; Chapin et al., 2000; Coomes, Allen, et al., 2003; Harris & Roach, 2017; Latham et al., 2020; Loss et al., 2015; Norton & Roper-Lindsay, 2004; Purvis & Hector, 2000; Sullivan et al., 2005; The Cape Sanctuary, 2014; United Nations, 2019, 1992; Wilmshurst et al., 1997). This decline has been caused by the degradation of ecosystems across both local and global scales, yet, the details of which species have been most affected by these changes are often unknown, due to inadequate long-term ecological monitoring (Kuhl et al., 2020; Lee, McGlone et al., 2005; Lindenmayer et al., 2012a; McGlone et al., 2020; Schmeller et al., 2017).

1.1.1 Ecosystems

Ecosystems are the interrelationships between living organisms and the non-living environment in their natural place. Ecosystem functions help maintain Earth's balance by controlling the fluxes of energy, nutrients, and organic matter through an environment (Purcell, 2021), whereas an ecosystem service is a process created by these functions that fulfill the resource demands of human livelihoods (Sekercioglu et al., 2010). Human well-being is inextricably linked to the provision of these ecosystem services, as we indirectly and directly benefit through the production and manipulation of natural resources, cultural benefits, and environmental regulation (Yapp et al., 2010).

An ecosystem is made up of biodiversity. Biodiversity is the total collection of all biotic variations from the level of genes to entire ecosystems (Mace et al., 2012; Purvis & Hector, 2000; Wilson & Peter, 1988) or the abundance and diversity of species present (Binning, 2000; Lee, McGlone et al., 2005; Proença et al., 2017; Purvis & Hector, 2000). Biodiversity as a term was coined and popularised in 1988 by the book edited by Wilson & Peter (1988). Biodiversity was then deemed as a resource management concept in 1992, at the Rio Earth Summit. Following the Convention on Biological Diversity the same year, biodiversity was described as 'the variability among living organisms from all sources, inter alia, terrestrial, marine and aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and ecosystems (United Nations, 1992).

Biodiversity provides many valuable goods and services, therefore the maintenance of biodiversity is crucial for ecological services that humankind is directly dependent on (Harris & Roach, 2017; Mace et al., 2012). These services include climate regulation and control; carbon storage; filters air and water; mitigation of natural disasters; natural resources such as fish, and timber; freshwater systems; crops and medicines from plants; cultural identity; health benefits gained from connecting with nature (United Nations, 2019). Aside from anthropocentric interest, biodiversity has intrinsic values that provide for human well-being (Mace et al., 2012). This intrinsic value is due to the uniqueness of every living species, the irreversibility of extinction species, and the unity of life and its evolutionary journey (Ghilaro, 2000).

Concern and the importance of slowing down biodiversity decline have been expressed across multiple international agreements which emphasise sustainable resource usage whilst enhancing ecosystem functions. Such agreements have included the Antarctic Treaty System, Convention on Biological Diversity, The Convention on International Trade in Endangered Species, International Union for Conservation of Nature, Pacific Regional Environmental Programme, and the Ramsar Convention of Wetlands.

However, many of these targets within the agreements and conventions have not been met and collectively they have failed to reverse biodiversity decline (Nature, 2020; Tilman et al., 1994). Globally, there has been no clear progress towards realising the stated ambitions to slow and eventually reverse the loss of biological diversity. These targets have lacked detailed plans for implementation and they have often been contradicted by parallel efforts to increase economic development (Nature Editorial, 2020). Furthermore, much of the world's remaining biodiversity is found in poorer countries which are more focused on development than environmental protection.

Scientific research has shown without doubt that there is a global decline in biodiversity and it is accelerating therefore there must be an acceleration in progress towards further international biodiversity policy targets (Lee, McGlone, et al., 2005; McGlone et al., 2020; Nature Editorial, 2020; Tilman et al., 1994). Many of these concerns have been accepted and applied in national policy documents of countries globally.

1.1.2 Ecological Integrity

An ecosystem is deemed healthy when its "inherent potential is realised, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed" (Karr, 1991). Evaluating and maintaining ecosystem health is part of many national and international environmental goals (Miller et al., 2009; Ministry for the Environment, 2000; Mueller & Geist, 2016). These goals are strategic and complex as they are often intertwined with cultural, social, economic, and spiritual elements that impact decision-making processes (Maseyk et al., 2019; McMurdo Hamilton et al., 2021).

To meet the variation of legislative and obligative needs in regards to biodiversity and ecosystem health, the ecological integrity concept was created. Karr (1991) described ecological integrity as the ability to support and maintain "a balanced, integrated, adaptive community of organisms have a species composition, diversity, and functional organisation comparable to that of the natural habitat of the region" (Karr, 1991; McGlone et al., 2020). The major distinction between ecological integrity and ecosystem health is that an ecosystem may have good ecosystem health (i.e., it functions well, including stabile species composition with nutrient cycling occurring), however, its ecological integrity may still be poor (e.g., lack of representation of natural biota) (McGlone et al., 2020).

The concept of ecological integrity is thus central to 'naturalness' and overlaps with mauri (McGlone et al., 2020), whereby mauri is the Māori concept of wholeness of internal energy or life force derived from whakapapa, an essential essence or elements that sustain all forms of life (Harmsworth & Awatere, 2013).

Unambiguously assessing the ecological integrity of ecosystems is an essential foundation for a consistent approach to land management supported by private land owners, conservationists, and regional and national authorities (Lee, McGlone, et al., 2005). It has been recommended by McGlone et al. (2020), Schallenberg et al. (2019), and Tierney et al. (2009b) that ecosystems should be surveyed by measuring integrity or 'nativeness' and within this the connectivity, which refers to the dynamics of spatial and temporal interactions of and between species (Schallenberg et al., 2019). The indicators within surveys of ecological integrity include the native dominance of the cover of plants, the relative abundance of native and exotic birds, and the abundance of pest animals (Bellingham et al., 2021). Ecological integrity is further determined by measuring the species occurrence, the plant tier or size structure, and the distribution or presence of species.

1.1.3 National Review

Although geographically isolated and relatively recently colonised, NZ's environmental degradation from land usage, overexploitation, and ecosystem change patterns are not inconsistent with other parts of the world. Due to the rapid growth in population, demand for resources, and accompanying agricultural intensification, the clearance of natural habitats within NZ's short human history has been rapid and extensive (Davis & Cocklin, 2001).

Since the early settlement of Māori, wildlife has been harvested for cultural and food resource purposes (Ministry for Culture & Heritage, 2017), and this has negatively affected many species and ecosystems. For example, kererū (*Hemiphaga novaeseelandiae*) was once widely harvested and consequently went into decline due to a combination of mammal predation, harvesting, and habitat loss and degradation (Wright et al., 1995). This situation is shared by many other native species (Kingsford et al., 2009; Towns & Ballantine, 1993)

Land clearance has been accompanied by the change in land ownership and management practices, leading to radical changes in the relationship between Māori, settlers, and the land. In 1840, Māori had exclusive rights and ownership over 27 million hectares of land, now only about 1.4 million hectares of Māori freehold land today (plus small amounts of land returned through Treaty of Waitangi claims and agreements since 1975) (Ministry for Culture & Heritage, 2017; Ministry for the Environment & Stats NZ, 2019b).

Forests were commonly cleared through the widespread use of fire to reduce the total forest cover from 85-90% prior to human arrival to 25% by the mid-twentieth century (Baillie & Bayne, 2019). Māori are thought to have begun the clearance for travel routes and security, hunting tactics, and the maintenance of bracken beds and other cultivated vegetation (New Zealand Conservation Authority, 1997). By the time of European settlement around 1840, approximately 40% of New

Zealand had been deforested (Baillie & Bayne, 2019). However, it was the European settlers that established more of an intense means of clearance through burning for land conversion for agriculture, including pasture creation, fuel load reduction, and pest control (Baillie & Bayne, 2019). Additionally, the conversion of lowland landscapes for agriculture involved draining wetlands, channelising or piping streams, and felling trees for timber.

Today, around 30% of land in NZ is protected, although most of it is nutrient-poor with high elevation and rainfall, which biases the representation of ecosystem integrity (Kingsford et al., 2009; Towns & Ballantine, 1993). Of the protected land area within lowlands, the remnant ecosystems are often small, isolated, and modified (Norton, 2000). The decline of the state of the environment, the ecosystems that it supports, and the biodiversity within, has been escalated by land modification.

NZ has developed a large primary sector of natural resource cultivation, production, extraction, and manipulation (e.g., timber, sheep, beef, dairy, grapes) on private land (Greenhalgh & Hart, 2015). Subsequently, the NZ landscape has experienced significant landscape modifications to keep up with the global and national demand for agricultural produce through maintaining growth by intensifying and expanding production on land. NZ is now dominated by a range of landscape-intensive agricultural sectors, with 39.75% of our land area in 2018 being devoted to agricultural production (Trading Economics, 2021).

Collectively, agricultural successes, urban expansions, and the associated introduction of a considerable number of pests and weeds, NZ's natural ecosystems and biodiversity have declined and deteriorated in recent centuries (Bennett & Saunders, 2010; MacLeod et al., 2008). Ultimately, these changes have put immense pressure on our natural ecosystems and disrupted functions and services (eg carbon sequestration, soil creation, water purification) that we are so dependent on socially, spiritually, and economically (Kelly et al., 2010; Maseyk et al., 2018).

Fertilisers were introduced to keep up with the demand for agricultural products. After NZ's native forests were cleared for pastoral use, the initial soil nutrient supply was depleted with consequential reversion to low-producing species (Monaghan et al., 2007). Today in NZ, the annual application of nitrogen via fertiliser has increased 627% since the 1990s (from 59,000 tonnes in 1990 to 429,000 tonnes in 2015), due to the rise of irrigated dairy farming (Stats New Zealand, 2019). However, the long-term application of fertilisers has a disastrous effect on the stability of ecosystem services and functions as they block or disrupt ecological processes, such as the nutrient cycles and water purification (Melts et al., 2018; Savci, 2012). Furthermore, increased nitrogen application causes a simplification and loss of plant diversity from natural grassland ecosystems, resulting in few naturalised, fast-growing, nutrient-loving species dominating everything (Mitchley & Grubb, 1986; Savci, 2012), changing the ecological integrity.

Additionally, due to NZ's agricultural expansion, herbicides and pesticides are now commonly applied across landscapes. Widespread herbicide use across landscapes is generally bad for native flora, as application generally targets anything wild (Peterson et al., 2020) to control the competition effects of other vegetation for improved targeted cultivated vegetation performance. The application of pesticides removes or controls targeted pests' presence, however, the usage of pesticides have contributed to the loss of natural biodiversity in rural and urban landscapes (Barzman et al., 2015) as they reduce insect populations that provide services for species such as plants and birds (Mahmood et al., 2016). The application of insecticides also destroys soil microbiology and in some cases bioaccumulating in the food chain (Condron et al., 2010).

Furthermore, a great number of species have been introduced, both accidentally and on purpose, many of which are now destructive pests and weeds (Allen & William, 2006; Kelly & Sullivan, 2010a). Biological invasion is widespread throughout NZ in such an aggressive manner that it is rarely equaled elsewhere (Robert B. Allen & William, 2006). Impacts of biological invasions are complex (Robert B. Allen & William, 2006) as many species were introduced for a purpose and the threats that they pose are often countered by valued services. For example, due to the combined effects of the disappearance of vegetation on lowland hill country, historical use of fire, and pastoral grazing, environmental impacts of landslides, nutrient loss, and sediment erosion are of great concern (Baillie & Bayne, 2019). Seemingly appropriate, NZ has attempted to stabilise landscapes to restore sediment erosion and nutrient loss through planting tree species, such as contorta pine, across steep hill country (Pearson, 2020; Wilmshurst, 2012). However, this introduction for stabilisation and the forestry industry has resulted in the spread of wilding pines and has thus displaced native plants (Habib et al., 2016).

Despite this, NZ has a good environmental track record for remedying and adapting natural resource usage and practice within the agricultural landscapes. For example, NZ has set programs for restoring biodiversity through riparian planting, fencing waterways (Walker et al., 2021a), and introducing regenerative agriculture (Gwen Grelet et al., 2021). These actions have proven how environmental concerns and change can lead to the industry being better placed to act (Ministry for Primary Industries, 2017). There is a widespread and increasing appreciation of nature and the natural ecosystems services that are on offer in NZ (Ausseil et al., 2021), thus protection measures are becoming prominent now more than ever (Ministry for the Environment & Department of Conservation, 2007; Schneider et al., 2014; The Department of Conservation, 2020).

Biodiversity in NZ

NZ biodiversity makes a small but significant contribution to the overall global diversity, with an estimated 80,000 endemic species (Stats NZ, 2021) and is internationally recognised as a hotspot for biodiversity, defined as areas featuring exceptional concentrations of endemic species and experiencing exceptional loss of habitats (Myers et al., 2000). Although our biodiversity is unique with a high proportion of endemic species, NZ does not have many species on the global world stage (Lee, McGlone, 2005; McGlone et al., 2020; Walker et al., 2008). However, NZ's biodiversity and conservation strategies have much to offer to the world due to our unusual native ecology, our journey of human impacts on the native biota, and many innovative collective conservation initiatives to manage and protect our biodiversity (Kelly & Sullivan, 2010a).

NZ ecosystems are regarded as unique as a result of NZ's high proportion of endemic species and isolation from other land masses for around 80 million years, however, experience wild swings in climate and submergence (Cooper & Millener, 1993). Further, the dynamic of NZ's geological positioning astride a major tectonic plate boundary has resulted in our grand mountains and volcanos, due to much continental plate spreading and collision (Ballance, 2009) causing dramatic geological changes throughout the entire country, fuelling the evolution of many areas endemic flora and fauna. The combination of isolation and geological changes has resulted in high levels of unique flora and fauna throughout the landscape (Wiser & Stanley, 2007). The combination of varied geography and climate has led to a diverse range of unique ecosystems within the NZ landmass, some of which are naturally rare, while some ecosystems are uncommon nationally (Ministry for the Environment & Stats NZ, 2019a; Wiser et al., 2008; Wiser & Stanley, 2007).

NZ terrestrial biota is highly endemic at the species level: for instance, 83% of reptiles, 99% of millipedes, 100% of conifers, 85% of angiosperms 80% of vascular plants, 86% of mollusks, 81% of arthropods, 60% of vertebrates, 36% of mosses and liverworts, and 30% of lichens are native to the country (Brandt et al., 2021; Lee & Lee, 2015). However, while the native flora is highly endemic at the species level, there are very few genera and no plant families that are endemic to NZ (Brandt et al., 2021).

Before human settlement, three species of bats were the only mammals to resign on the NZ islands (Lord et al., 2009), whereby one became extinct on Big South Cape island when shop rats arrived (Bell et al., 2016). As a result, interesting evolution opportunities had arisen, whereby native birds evolved to fill ecological niches occupied elsewhere by mammals, known as functional gaps (Brandt et al., 2021). Many of our native species have evolved life history strategies with long lives and low annual reproductive output. It is suggested that this is because NZ flora and fauna have experienced relatively low frequency (i.e. long return time) of disturbance due to the absence of both humans and other terrestrial mammals, plus a mild oceanic climate (Kelly et al., 2010; Kelly & Sullivan, 2010a; Lord et al., 2009).

Nationally, biodiversity decline is the reduction of both species' populations and distribution. At least 75 native animal and plant species have become extinct since humans arrived in NZ and there are now currently almost 4,000 native species that are threatened or at risk of extinction (de Lange et al., 2013; Robertson et al., 2019). Of the 9,323 assessed native species (Stats NZ, 2019), 27 percent (2,519) have been classified as data deficient, which means we don't know enough about them to evaluate their conservation status. Among our native plants, 289 are threatened and 749 species (or 40%) are at risk, meaning that they will likely become extinct within the next century (de Lange et al., 2013; Hooker, 2011; Stats NZ, 2019). Suffering an even greater decline, 74% of our native freshwater fish are currently at risk or threatened of extinction. Of the species assessed (refer to (Stats NZ, 2019)), 79 percent (174 of 220 species) of native terrestrial vertebrate species (bats, terrestrial birds, frogs, and reptiles) were classified as either threatened with or at risk of extinction. Of the 417 bird species still present in NZ (56 are already extinct), over 40% are now threatened or at risk of extinction (Environment Guide, 2018; Robertson et al., 2017; Stewart, 2015). Indigenous lizards are also at serious risk, with approximately 85% threatened or at risk, however, there is insignificant data, and the true percentage is expected to be higher (Stats NZ, 2019). Combined with the substantial reduction and health of native ecosystems, these threatened species statistics indicate the uncertain state of much of our remaining indigenous biodiversity.

Conserving and sustaining our native biodiversity is primarily through the critical protection of public conservation lands (Norton et al., 2018). These areas represent ecosystems that have been least impacted by human activities, but do not represent the full scope of ecosystems throughout the country (Reid & Norton FLS, 2013; Wiser et al., 2011a). NZ is biased towards protecting ecosystems that are inland and upland regions that often exclude areas of higher productive ecosystem values as they have already been converted for human activities (Reid & Norton FLS, 2013; Wiser et al., 2011a). There is now a need to prioritise the protection of biodiversity that is situated in lowland privately owned land, non-conservation lands that are used for human activities such as pastural systems, cropping, horticulture, etc (Norton et al., 2018).

Our Forests

Once covering 80% of NZ's land, forest ecosystems contain diverse combinations of flora and fauna (New Zealand Statistics & New Zealand Statistics, 2015; Perry et al., 2014) with continuous changes over millennia due to climate cycles, species arrivals, and disturbance events. The most rapid and detrimental change forests have experienced has occurred over the last 700 years of human settlement (Baillie & Bayne, 2019; Kelly & Sullivan, 2010b) as actions have rescaled forests with fire regimes causing fragmentation (Perry et al., 2014) resulting in ecologically harmful consequences. native forests were cleared so rapidly that by 1913, some native species were threatened with extinction. To avoid this disaster, exports of native timber were restricted, and in 1925, incentives were introduced to create plantations of exotic species, such as the radiata pine (New Zealand Forest Service, 2021).

Today, NZ has a total of 10.1 million hectares of forests, covering 38% of the land (New Zealand Forest Service, 2021) and of this, native forests cover over 8 million hectares (30% of land area)(New Zealand Statistics & New Zealand Statistics, 2015). The Crown owns most of the native forests (5.2 million hectares), which are managed by the Department of Conservation (DOC) and are protected in national parks, scenic reserves, and other conservation areas (New Zealand Forest Service, 2021).

At present, native forests are more common in higher altitudes much of coastal and lowland landscapes have been cleared for a pastural landscape. The clearance of lowland forests has significantly reduced several forest types, resulting in many rare and sparse forest types. Of the native forests that remain, the threats and damage of invasive species, sparse fragmentation, and lack of forest buffers or connections are concerning (Wiser et al., 2011b; Norton et al., 2018).

There are two main types of native forest: beech, made up of 5 species of southern beech; and podocarp forest, including rimu (*Dacrydium cupressinum*), totara (*Podocarpus totara*), miro (*Prumnopitys ferruginea*), kahikatea (*Dacrycarpus dacrydioides*), and matai (*Prumnopitys taxifolia*) (Singers & Rogers, 2014; Wiser et al., 2011b), though forest types differ across the country.

The ecological integrity of native forest ecosystems differs across regions because of dissimilar environmental and ecological conditions thus several integrity factors are observed. Furthermore, contemporary disturbances (e.g. drought, cyclones, floods, earthquakes, fires) influence NZ forests and are likely to have enduring effects on the forests' vegetation and bird dynamics (Bellingham et al., 1999). Eternal and historic environmental elements therefore must be considered at the local level in order to obtain a sound understanding of site-specific ecological integrity which may be difficult as there are many drivers of biota compositional variation, including climate, history, local ecology, disturbance, soil fertility, and herbivory (Wiser et al., 2011a).

Biodiversity on Private Land

The protection of biodiversity is fundamentally reliant on the relationship between landowners, regional governance, and supporting authorities (Nahuelhual et al., 2018). Monitoring the health of forests on this private land can play a critical role in managing, and enhancing, the natural biodiversity that remains. (Hashiba et al., 2014; Hawke's Bay Biodiversity, 2015). An active example of this relationship is the collaboration between Queen Elizabeth II National Trust (QEII) and private landowners and Regional Councils to protect native biodiversity. QEII National Trust has established under Queen Elizabeth the Second National Trust Act in 1977 (QEII National Trust, 2022a). The purpose of this trust is to aid in conservation efforts on private land by encouraging and promoting the provision, protection, preservation, and enhancement of open spaces (QEII National Trust, 2022a).

QEII National Trust reported that 70% of New Zealand is privately owned land (approximately 19 million hectares), with coastal and lowland landscapes containing small native pockets of threatened habitats (QEII National Trust, 2021). Currently, QEII has over 4,912 covenants registered since 1977 (QEII National Trust, 2022b). The long-term integrity of these QEII covenant native forest fragment habitats that remain in lowland areas throughout farmland are threatened indirectly due to economic pressures to intensify agriculture in the surrounding land (Maseyk et al., 2021). These covenants themselves are protected, but the intensified land usage and practices of surrounding landscapes can have dire long-term effects on the forest, thus the protection and care by the QEII organisation help the covenant to help flourish native biota (QEII National Trust, 2021).

National Biodiversity Threats

The first settlers initiated an immediate and rapid biotic transformation across the landscape in NZ about 750 years ago and now with devastating ecological consequences of deforestation, flora and fauna declines or extinctions followed (Wilmshurst et al., 2008). The biodiversity decline in NZ has been evident due to the rapid consequences of pests, habitat loss, and landscape modifications.

Fragmentation causing habitat loss is a major cause of ongoing biodiversity loss (Burns et al., 2011; D. B. Lindenmayer & Fischer, 2006; Spiesman et al., 2018) as it creates negative influences including edge effects, prevention of migration, declines in effective population sizes, loss of genetic variability, and invasion of exotic species (Penariol & Madi-Ravazzi, 2013). Fragmentation prevents species dispersal and movement across the farmland matrix due to connectivity alterations (Bennett & Saunders, 2010) and also minimises some interior species richness and overall native biodiversity as it increases forest perimeter and area ratio (Young & Mitchell, 1994), increasing edge effects.

Another major main threat to native biodiversity is biological invasions, as they essentially reduce ingenious species abundance, and in some cases the extinction of resident biota (Norton, 2009). There are now 31 naturalised mammals established with wild or feral populations (Parkes & Murphy, 2003) having massive impacts on NZ native biodiversity and depriving ecosystem integrity. Mammals disrupt ecological processes and interactions through predation, competition, resource consumption, and compositional changes (Sweetapple et al., 2016; Towns & Ballantine, 1993). Mammalian pests as they damage forest compositions through heavy herbivory on sensitive, yet key vegetation (Allen et al., 2019). Herbivory pests reduce resource availability for native species, and can eventually compromise the forest structure (Brake, 1972; Coomes, Allen, et al., 2003), and larger mammalian pests trample vegetation that further alters the forest floor plant diversity and abundance. For example, 30 years of collective monitoring throughout NZ forests revealed significant changes in the composition of the forest itself, largely attributable to the selective browsing habits of Australian brushtail possums (*Trichosurus vulpecula*) (Clout, 2001), causing both direct and indirect impacts on many other native species.

Plant invasions also result in a number of negative effects, as they pose great threats through predation and competition for space and resources (Brake, 1972), such as smothering native plants and suppressing the natural regeneration of native understory and canopy species (Aviss & Roberts, 1994; Owen, 1998). It was found by Brandt et al. (2021), that there is a total of 1798 naturalised plant species, with an additional 1043 casual non-native species, occurring across NZ. Many of these exotic plant species are managed locally to prevent further naturalisation or spread. However, it is predicted that further spread is inevitable, but it is exacerbated by climate change, further transport networks and infrastructure, and the time lag of naturalisation (Brandt et al., 2021).

Moreover, biological invasions of parasites and pathogens have been the cause of much biodiversity extinction and declines for an array of biomes (Allen & William, 2006). Introduced species also pose hybridisation threats to our native species that dismantle populations and the integrity of communities (Norton, 2009).

NZ climate is mild but very variable like many islands, due to the strong environmental gradients of topography and coastal effects, there is much uncertainty surrounding the scope of environmental impacts that climate change will bestow on our native biodiversity (Macinnis-Ng et al., 2021). The changing climatic conditions will influence biota in multiple ways, depending on a given species' physiological tolerances, as well as through habitat shifts, migration, or even extinction (Dawson et al., 2011). With the combination of other influenced threats such as the discussed biological invasions and the fragmentation of forests throughout the country, biodiversity decline will magnify (Macinnis-Ng et al., 2021).

Although there is much momentum toward predator-free NZ, it is currently impractical to remove all predators using methods that are currently available (Norton et al., 2016). More so, these predator controls have primarily been focused on the control of targeted species (Norton et al., 2016). However, with the increasing efforts in predator control in response to our declining biodiversity, simultaneous multiple species eradication has become necessary and more tangible in recent years (Burge et al., 2021). Across the country, predator-proof fences and large-scale trapping grids to direct trapping focus has been used to create main-land islands or high predator-controlled managed areas.

NZ Ecosystem Value

The dependency we share in the relationship between the NZ people and natural biodiversity is significant as it provides cultural, regulating, provisioning, and supporting ecosystem services. These services are part of the NZ population’s livelihood and provide prosperity in both financial and identity terms (Meurk & Smith, 2007).

In NZ, our natural ecosystems, and the biodiversity within, is renowned as a treasure and taonga (Ministry for Culture & Heritage, 2017) as it is the core of our national identity (Department of Conservation, 2020; Maseyk et al., 2017). Healthy ecosystems have huge cultural value and significance in NZ as biodiversity within it enables Tangata Whenua to connect with the environment and with one another such as intrinsic value (Letica, 2020; New Zealand Conservation Authority, 1997). Nature and people are intertwined as we rely on sustenance, inspiration, and material for cultural practices and expressions such as waiata, karakia, and whakataukī (Ministry for Culture & Heritage, 2017), which supports national cultural identity and essence of life for NZer’s (Ministry for the Environment & Stats NZ, 2019a). Most New Zealanders have connections with nature and the land as De Bonnaire et al. (2019) describes how people enjoy open spaces and regard the natural environment as their most precious asset. It is understood that much of the population has great concern for our environment and that we are not doing enough to protect it (De Bonnaire et al., 2019).

Our diverse landscapes and ecosystems provide scenes and exposures for recreational staples and destination adventures that attract people from around the world, putting us on the map. Any decline in biodiversity that supports these functioning ecosystems will result in a deterioration within industries that relies on the image of the purity of international marketing (Kelly & Sullivan, 2010a), which is described as ecological integrity that can only be achieved through native biodiversity stability.

There is a lack of overall funding for monitoring biodiversity meaning there is minimal data on the effects of ecosystem functions and the status of biodiversity within (Patterson & Cole, 2013). However, it has been calculated in 2012 that land-based ecosystems contributed to producing $57 billion worth of services, about 27% of NZ's GDP for that year. Although this aggregated calculation may be indicative only, this calculation gives a visible and tangible economic value to these services for NZ (Patterson & Cole, 2013). Comparing this economic value with internationally measured ecosystem services is difficult as data is insufficient and inconsistent.

Within an aggregated calculation, the ecosystem which produces the highest net total value of ecosystem services was, unsurprisingly, the agriculture sector, accounting for $12,420 million (Patterson & Cole, 2013). Furthermore, it was estimated that the agriculture ecosystem contributed another $7,751 million of support services which are not accounted for in the net total. This is not surprising since the agricultural ecosystem covers 39% of NZ’s land surface (Patterson & Cole, 2013). NZ is currently economically dependent on intensive agricultural practices for its main exports (beef, lamb, and dairy products). Therefore, conserving our native biodiversity is essential for the long-term sustainability and resilience of farming systems and for sustaining the provision of the full range of ecosystem services (Maseyk et al., 2021; Reid & Norton FLS, 2013).

In recent literature focusing on the contribution and value of forest biodiversity within an agriculture landscape, Duriaux Chavarría et al. (2018a) demonstrated the positive influence of on-farm forest biodiversity on farming performance and yield. Duriaux Chavarría et al.(2018a) confirmed how forest fragments throughout farmland landscapes not only promote improved product performance or yield, but also improve farmland resilience, sustainability, and the distribution of essential ecological services or functions. The paper stated how neighboring forest fragments create energy and nutrients through the responding ecological functions that in turn support land use and provide ecosystem services. These local farming systems experienced environmental enhancements, in both agriculture productivity and ecological functions, with the two being mutual reinforcement. It can therefore be assumed that the forests on farmland benefit the farmland environmentally, but aid in delivering better products for people. However, there are limitations in understanding and propagating action for what is required to create and support changes in how native biodiversity is understood, valued, protected, and managed in agricultural practices (Maseyk et al., 2021)

Relationships between landowners and the environment are often varied across societies because of the differences in the property rights regime and the social, economic, political, legal, cultural, traditional, and religious influences (Rao, 2018). However, landowners often have a longstanding and deep interest in the integrity of the land they manage. This is because farms are traditionally passed down through families or within small local communities. Additionally, it is common for the identity of a farmer or landowner to relate to their land due to the intimate understanding of the land, animals, and the environment. Landowners' motivation for caring for their land is also a livelihood as it can make business sense to take responsibility for the management of their land, for the sake of the environment, on which they're so dependent (Baldwin et al., 2017).

1.1.4 State of Lowland Hawke's Bay Biodiversity

Hawkes's Bay

The Hawke's Bay region covers around 14,200km^2 (1.42 million hectares) on the central East Coast of the North Island, including the mountain ranges central north island, west from the coast and north, a span of 350km of diverse structured coastline, and productive plains and hill country (Hawke's Bay Regional Council, 2021).

Hawke's Bay is known as a sunny, vibrant, fresh produce-filled region (Smith et al., 2020) with 166,000 inhabitants (Statistics New Zealand, 2018). There are two main cities, Napier on the coast and Hastings 17km inland. The smaller main towns across the region are Wairoa, Waipawa, and Waipukurau. The region has seven major river catchments, (from the north); Wairoa, Mohaka, Esk, Ngaruroro, Tutaekuri, Tukituki, and Waipawa (Hawke's Bay Regional Council, 2013, 2021).

Hawke's Bay has a mild Mediterranean climate (Hawke's Bay Regional Council, 2021) with the lowland experiencing dry periods (defined as droughts when there is no measurable rain over 15 days) more commonly compared to the adjacent ranges (Wilmshurst et al., 1997). The short intensity of rainstorms occurs throughout the year but is more common in autumn. The winters are mild, although frosts and snow do occur at higher elevations (Wilmshurst et al., 1997). The high rainfall variability conjoined with a high frequency of dry periods means the region is prone to fire disturbances, often destroying forested landscapes. Hawke's Bay is also accustomed to serving storms of tropical origin from across the east coast and is renowned for damaging vegetation and eroding steep hill country (Wilmshurst et al., 1997).

Due to Hawke's Bay's ideal climatic and terrestrial conditions, farming, horticulture, wine, and tourism are key industries that rise the region's economy (Hawke's Bay Regional Council, 2021). As a result, the region has been highly modified through decades of agricultural, forestry, and urban development. In this region, approximately 50% of the landscape is used for pastural farming (Hawke's Bay Regional Council, 2013). Of this, it was reported in 2012 that 11% of the total NZ sheep and beef production was in Hawke's Bay (Ministry for Primary Industries, 2008). In 2018, it was found that there was a combined total of 12.4% of the region's population was employed within the agricultural, fishery, or forestry industries (New Zealand Statistics & New Zealand Statistics, 2015; Statistics New Zealand, 2018). Yet, Hawke's Bay is known for its iconic forests, such as on the northwest of the region's border, Te Urewera, joined with Lake Waikaremoana, and other natural reserves including White Pine Bush, Ruahine Forest Park, and Cape Kidnappers (Pollock, 2009).

History

Prior to any human settlement, Hawke's Bay was dominated by native forests and rich in bird life (Mc Glone, 1978; Pollock, 2009; Sheath et al., 1990). Ranges and foothills with high altitudes across the region were covered in beech forests and subalpine plants. Beech trees also coated the southern coast and end of the region, joining with the infamous 70-mile bush (Arnold, 1994). Through pollen assemblages, Wilmshurst et al. (1997) found that miro (*Prumnopitys taxifolia)* and rimu (*Dacrydium cupressinum)* were the dominant tall podocarps, and other podocarps included tōtara (*Podocarpus totara)* type, matai (*Prumnopitys ferruginea)*, and kahikatea (*Dacrycarpus dacrydioides)* in the northern area of lowland Hawke's Bay before human settlement destroyed the lowland forests. There was also evidence of well-dispersed pollen of *Nothofagus* subgenus *Fuscospora*. The study also accounted for smaller trees, shrubs, herbs, ferns, and climbers that are important in the sub-canopy and shrub layers, represented in the pollen spectra by low percentages of diverse taxa (Wilmshurst et al., 1997). Broadleaf-dominated forests and patches of grasslands also draped the lowland. Near the coast, kanuka (*Kunzea ericoides*) and manuka (*Leptospermum scoparium*) were prominent and pīngao (*Ficinia spiralis*) and *Spinifex sericeus* colonised the beaches (Pollock, 2009). These forests were frequently disturbed by lightning strikes, fire associated with droughts, and volcanic eruptions (Wilmshurst et al., 1997). Regardless of the frequency or intensity of these pre-human fires, the forest has always undergone rapid regeneration. This ability to recover quickly indicates that forest systems across the region have the potential to regenerate and restore well if protected (Perry et al., 2014; Wilmshurst et al., 1997)

It is widely recognised through pollen profiles that the major forest clearance was first made by Māori using fire (Perry et al., 2014; Wilmshurst, 1997; Wilmshurst et al., 1997). Māori made habitual and widespread use of fire to keep tracks clear, maintain their dwelling areas free of tall vegetation, site clearance for cultivation purposes, and encourage the growth of bracken (*Pteidium esculentum*) which provided the people with a major source of carbohydrates (Mcglone et al., 1999). The widespread burning by early Māori resulted in the loss of between one-third and one-half of the original forest cover by the time Europeans settled in the mid-19th century having a profound effect on the vegetation compositions and biodiversity across NZ (Mcglone et al., 1999).

Early European settler statements between the 1840s and the 1850s emphasised the lack of forest cover in lowland Hawke's Bay (Mc Glone, 1978) as it was observed that much of the low-lying country was occupied by extensive swamps and wetlands, mostly dominated by raupo (*Typha Orientalis*), flax (*Phormium tenax*), and toetoe (*Austroderia* spp.). The surrounding lowland hills were reported to mainly be covered with bracken (*Pteidium esculentum*) and varied scrub, commonly tutu (*Coriaria* spp.) (Mc Glone, 1978).

It was also observed by early European settlers that across the steeper ranges and covering the foothills of the Ruahine Ranges, and to the south of the region, large areas of dense forest remained. These dense forests had stands of matai and totara. In areas of poorly drained ground and swamp margins, kahikatea populations were observed (Mc Glone, 1978). Beech forest (*Nothofagus* spp.) were reported as absent from most lowland areas, and it was found that rimu occurred only inland towards the Ruahine Range and south towards Dannevirke (Mc Glone, 1978).

The large-scale European settlement across Hawke's Bay in the 1850s resulted in the rapid conversion of what little forest, scrub, and fern that was left from Māori clearance, into grassland. The forests were burned and milled, while the scrub and fern habitats were eliminated by burning and grazing from introduced stock (Mc Glone, 1978). The remaining forest and fern scrub in the lowland and hill country of Hawke’s Bay have since been steadily cleared away for the development of pastural farmland (Brown et al., 1999; Ministry for Primary Industries, 2008).

Hawke’s Bay Today

Little now remains of the original native forests which once covered the region containing habitats that supported rich biodiversity have since been greatly diminished and modified (van Noppen, 2020). However, despite ecological and environmental changes, Hawkes Bay still contains a great diversity of landscapes, habitats, flora, and fauna, and importantly supports great historic and natural value (Hashiba et al., 2014; Hawke’s Bay Biodiversity, 2015; Hawke’s Bay Regional Council, 2013). The region contains areas with a significant range of support for native biodiversity including the Kaweka Ranges, Lake Waikaremoana, Pekapeka Wetland, Whakaki Lagoon, Ahuriri Estuary, Blowhard Bush, Cape Kidnappers Sanctuary, and Boundary Stream, supporting the needs of many threatened and vulnerable species. (Hashiba et al., 2014; Pollock, 2009; Smith et al., 2020).

However, the present-day lowland landscape is almost entirely exotic grassland as the native vegetation has been steadily cleared away for the development of pastural farmland (Forbes, 2012; Sheath et al., 1990). The reduction of native forest has been particularly widespread in the areas that were found on alluvial plains, coastal and lowland hill country for prime grazing purposes for its fertile and accessible productive land (Hashiba et al., 2014; Hawke’s Bay Regional Council, 2013; Pollock, 2009; Smith et al., 2020). Much of these agricultural, horticultural, and forestry systems provide for regional biodiversity with areas of wetlands, riparian zones, and plantings for erosion and sediment control. These areas as such demonstrate how economic and environmental endeavors can combine to enhance biodiversity (Smith et al., 2020).

Consequently, there are now very few remnants of the pre-European settlement forests. It is believed that the current extent of native forest is now reduced to 23% within the Hawke’s Bay region (Walker et al., 2012), but within the lowland context, it is 6% (Pollock, 2009).

Local Biodiversity Threats

Conversion of land for agricultural and urban expansion and development, invasive species, browsing, sedimentation, and draining of waterways and wetlands are enlarging the effects of both habitats’ loss and fragmentation (Walker et al., 2012). Fragmentation of habitats across the landscape matrix has transformed forests into reduced and sparse smaller patches of habitats. Fragments are reduced in size and isolated in patches making them even more so susceptible to further habitat loss as it modifies the habitat configuration and changes the extent of functions (Hashiba et al., 2014). Connectivity is reduced thus excluding linkages of specialist species that are dependent on movement or less mobile or require larger forests, like tūī (*Prosthermadera novaeseelandiae*), or wētā (*Anostostomatidae* ssp.). The loss of habitats due to fragmentation of landscape has negatively affected a range of native population growth, reproductive and dispersal success, and other require ecological processes in the region (Hashiba et al., 2014).

Much of these forest fragments are isolated on private land, exposed to modification and possible eventual loss as they are subjected to the landowners' wishes for the extent of protection. However, even the forest fragments which are legally or voluntarily protected within the region are prone to threats from introduced and invasive animals and plant pests (Davis & Cocklin, 2001; Norton & Roper-Lindsay, 2004; Walker et al., 2012).

Further, many intact forest fragments are found at higher altitudes in the region but even these forests are under constant pressure from introduced and invasive species (Brandt et al., 2021; Kelly & Sullivan, 2010b). Introduced mammals have devastated the Hawke's Bay region's native biodiversity, and in some cases irreversibly altered ecological processes through direct and indirect pressures. Mammals pressure native biodiversity through predation, competition for food and habitats, and the modification of habitat through herbivory causing structure or composition change (Brandt et al., 2021; Innes et al., 2010; Kelly & Sullivan, 2010b; Parkes & Murphy, 2003; Tanentzap & Lloyd, 2017).

Biodiversity in Hawke's Bay is also impacted by several exotic plant pests that cause havoc on native ecosystems (Bourdôt et al., 2007; Scofield et al., 2011; Sullivan et al., 2005). For example, gorse (*Ulex* spp.), blackberry (*Rubus fruticosus*), bathurst burr (*Xanthium spinosum*), and ragwort (*Jacobaea vulgaris*) are common throughout the region (Hashiba et al., 2014; Pollock, 2009; Walls, 2014). Old man's beard (*Clematis vitalba*) is widespread, but actively controlled, but found in the north of the region and in drought-prone areas, nodding thistle (*Carduus nutans*) invade (Pollock, 2009). On the coast, marram grass (*Ammophila arenaria*) has colonized the sand dunes (Pollock, 2009).

Due to the disappearance of vegetation on lowland hill country, landslides and erosion are of great concern as it is now ranked one with one of the most highly erodible hill countries in NZ. There is currently much research for management and action plans to mitigate sediment loss and appropriate sustainable land practices (Davis & Cocklin, 2001; Dominati et al., 2019; Fitzgerald et al., 2019; Jay, 2005; Robertson et al., 2019). Through parts of NZ, pines (*Pinus* spp.) were planted to stop erosion across steep hill country but have since spread freely and displaced native plants (Pollock, 2009).

Hawke's Bay Regional Council's 'State of the Environment 2019' report (Smith et al., 2020), highlights how the surface water quality along with groundwater levels, riverbank erosion, and the management of wetlands represents the main environmental risks within the region. Freshwater quality across the entire region is declining due to the excess nutrients and sediments caused by intensified or poor land management depriving many natural ecosystems (Smith et al., 2020).

Without an adequate management plan for the protection, support, and enhancement of these threatened forest fragments, there is little hope for the restoration or regeneration of native terrestrial biodiversity.

Hawke’s Bay Regional Council, Iwi & Local Action

The Hawkes Bay Regional Council (HBRC) governs the Hawke’s Bay region with land management and environmental teams implementing policies Regional Councils have the responsibility for managing the use of natural resources; air; water; land; coast; and biodiversity. The Hawke’s Bay region is in a position of becoming a leader in restoring native biodiversity (Hawke’s Bay Biodiversity, 2015) as they are motivated by innovations towards the transition to sustainable agriculture in NZ. HBRC recognises prioritised ecosystems and seeks to protect and enhance what natural environment remains by focusing on environmental issues at the catchment scale, as well as on the private land scale. From this, the council can administer practices appropriate toward relationships with private landowners to benefit the natural environment and stakeholders.

Local Iwi and Hapū contain vital Mātauranga, which the region requires for reconnection and protection with the environment (Letica, 2020). In Hawke’s Bay, Iwi and Māori representatives’ partner with authorities such as HBRC to protect and respect when the partnership between the HBRC and Iwi significantly contributes to the region’s prosperity. This is achieved through local participation in the co-governance influence of kaitiaki in the conservation, preservation, and management of our natural resources (Hawke’s Bay Biodiversity, 2015; Smith et al., 2020).

Along with HBRC and local Iwi, there is huge public engagement and eagerness to restore and protect the region's natural environments and ecosystems, improve the water quality, and produce yields in alignment with climate change, and soil sustainability requirements. To list a few, these organisations include Biodiversity Hawke’s Bay, QEII, Forest and Bird, and Plant Hawke’s Bay.

Previous Studies on Hawke’s Bay Biodiversity

Like other regions, there is currently very little comprehensive understanding of the status of biodiversity and of many ecosystems’ integrity across lowland Hawke’s Bay. The taxonomic knowledge gaps attribute to the fact that many species are difficult to find or identify, are viewed as less attractive resulting in a lack of awareness or interest, or there has been an absence of expertise. Also, there is a lack of understanding of many species' ecological roles (Hashiba et al., 2014). In addition, grasping an understanding of many threatened species is difficult as they are often rare or sparsely distributed.

There have been few studies integrating the status of lowland biodiversity across Hawke's Bay, with the majority being specific studies, either undertaken by HBRC or contracted by HBRC. Few studies have been undertaken by governmental authorities or research establishments.

In Hawke's Bay, there have historically been many industries and livelihoods that correspond with the surrounding environment and the ecosystem services that are provided. The region has developed a history of monitoring environmental conditions which influence natural resource management and consumption. For example, rainfall monitoring was first accounted for in the 1930s, while river monitoring started in the 1950s, with groundwater monitoring in the 1960s (Hawke's Bay Regional Council, 2013). There is also an abundance of archive data reported by natural resource users and private landowners', outside of the regional council records (Hawke's Bay Regional Council, 2013). Despite this history of environmental data collection of water and soils, there have been little monitoring efforts by the region in terms of the status of lowland biodiversity (Hawke's Bay Regional Council, 2013; Smith et al., 2020).

In the interpretation of the Potential Vegetation of New Zealand, a portal on LRIS, it has been found that there is an identified sixteen different forest types and one scrub type across the region. Of these sixteen forest types, seven types are classified as 'Acutely Threatened', one type is 'Chronically Threatened', and one type is 'At Risk'. These forest types once dominated an extensive part of the region on the region's main alluvial plains (Heretaunga Plains) and podocarp forests covering coastal environments to low hill countries (Walker et al., 2012). The ecosystems or habitats for native species that remain are often highly modified and degraded from the original state as they are now dominated by pastural grassland, orchard, and cultivated crops.

Maanakai Whenua has data collection for reporting and monitoring from New Zealand's Vegetation Survey Bank. There are 878 NVS plots within the Hawke's Bay ecological region, and 347 plots within the lowland altitude of 300 m (200 m is described as lowland but one of my studied sites is situated at 300 m) (Manaaki Whenua Landcare Research, 2021b).

DOC routinely undertakes national-level surveys across the region. These surveys cover all three-tier reporting and monitoring regimes for data collection. Other local authorities, research institutes, and privately consulted research projects have contributed to the assortment of NVS data across the region (Manaaki Whenua Landcare Research, 2021a) providing a historical take on a uniform framework for monitoring the vegetation present in the region.

HBRC published a report, Hashiba et al., (2014), to understand the region's state and status of native biodiversity that identified missing links of knowledge. Hashiba et al., (2014) assessed the region's vegetation cover, based on historic surveys, and found that large and intact preserves of native forests across the region tend to be confined to the ranges on the west and at higher altitudes. Rolling hill country throughout the region has several significant, although not notably large, remnants. However, east on the plains of Hawke's Bay, forested areas are small and scattered (Hashiba et al., 2014; Hawke's Bay Biodiversity, 2015). Hashiba et al., (2014) discussed how native forest, scrub, and grasslands across the Hawke's Bay region are occupied by many different nationally threatened species, subsequently, many native species are reliant on these forest systems and are now confined to the forest-covered ranges, scattered remnants, regenerative pocket, waterways, and coastal habitats (Hashiba et al., 2014; Pollock, 2009).

Spatially limited remnant forests in the region support an array of species, for example, clusters of populations of the short-tailed bat in southern and central Hawke's Bay and throughout the Te Urewera rely on these forests. A diverse range of native birdlife is found in the ranges to the north and west edges of the region, including tūī (*Prosthermadera novaeseelandiae*), korimako (*Anthornis melanura*), kererū *(Hemiphaga novaeseelandiae*), riroriro (*Gerygone igata*), and pīwakawaka (*Rhipidura fuliginosa)*. Rare species include the blue duck (*Hymenolaimus malacorhynchosz*), North Island brown kiwi (*Apteryx mantelli*), and North Island kākā (*Nestor meridionalis*), and the New Zealand falcon (*Falco novaeseelandiae*) (Hashiba et al., 2014; Pollock, 2009). The ranges are also home to native skinks, geckos, bats, and large land snails. It has been found that regenerating forests and shrublands also provide a haven for native geckos (Hashiba et al., 2014; Pollock, 2009).

In addition, Hawke's Bay Biodiversity (2015) assessed the current status of the biodiversity in the region as a local organisation's response to the biodiversity decline crisis and envisions a new hope for biodiversity in Hawkes Bay. This report collaborates the biodiversity stock from an assortment of collected data and research from a range of authorities and other interested stakeholders or establishments.

1.2 Biodiversity Monitoring

We have a responsibility to safeguard biodiversity for future generations (The Department of Conservation, 2020). In order to manage and protect this biodiversity, we must be aware of conditions and threats. It is therefore necessary to monitor biodiversity beyond elements within ecosystem services that we utilise, but more so of ecological health or condition that correspond to the integrity of an ecosystem (Bellingham et al., 2021; McGlone et al., 2020; Tierney et al., 2009). Through doing so, we can better manage and intercept conservation efforts by focusing on native biodiversity directly, rather than the ecological services that biodiversity. Without monitoring specific targets of native biodiversity, NZ loses its unique sense of place and natural heritage (Norton et al., 2016).

Monitoring allows managers to understand biodiversity aspects of an area, site, or habitat and how it is changing over time through observing ecosystems, area, or targeted interactions of species providing an understanding of the status, threats, condition, and change over time (Lee, McGlone, & Wright, 2005; Lindenmayer et al., 2012a). The accumulation of collected data that underpins metrics reflects the status of biodiversity through finding quantifiable evidence of conservation successes (e.g. species recovery following appropriate management intervention as well as conservation failures) (Lindenmayer et al., 2012b). Monitoring delivers measurements on trends in key aspects of biodiversity (e.g. population changes), providing appropriate early warnings of problems or threats in need of mitigation, including management failures whereby action or re-assessment is required (Lindenmayer & Gibbons, 2012; Watson & Novelly, 2004). However, biodiversity is notoriously challenging to observe or measure as the structure and compositions are complex and changing across broad spatial and temporal scales (Lee, McGlone, et al., 2005).

For successful and effective monitoring of an ecosystem that builds a capacity of viable data of observed biodiversity, frameworks should be followed when implementing biodiversity monitoring (Schmeller et al., 2017). Firstly, managers must identify monitoring questions and aims that then allow identification of the key components, functions, and processes to monitor and the precision (e.g., sample size) required to answer the questions or conservational aims (Schmeller et al., 2017). Proper management of the interpretation of the data is of great importance to get a sound analysis of observed biodiversity trends (Schmeller et al., 2017).

Obligations toward biodiversity conservation can only be achieved through collective monitoring, which entitles combinations of citizen science, ongoing research endeavors, consultancy of developments, and governmental action (Bellingham et al., 2021; Kuhl et al., 2020; Lee, McGlone, et al., 2005). Effective monitoring should have an integrated approach, including intensive and extensive monitoring through the efforts of volunteers, scientists, and private landowners to enable a broad range of focuses (Chandler et al., 2017; Letica, 2020). Through this, it creates a larger scientific community for sharing biodiversity monitoring data, knowledge, and tools. This ensures that from a local to a global scale, accessibility, interoperability, and the reporting of biodiversity data are significantly ameliorated (Schmeller et al., 2017).

Challenges

A significant challenge for monitoring is the intervention timeline as the implementation of monitoring programs is often prolonged. Efforts may elapse between a given management intervention and the response of some elements of biodiversity, resulting in late action due to a lack of monitoring. The reality is that many of the funding programs for monitoring biodiversity across privately owned land are commonly short-term (Lee, McGlone, & Wright, 2005; McDonald-Madden et al., 2010). This is concerning as often intervention is applied without any initial idea of the status of biodiversity so managers end up not knowing how the intervention affected the biodiversity at the site, thus the effectiveness of intervention or management is unknown (Lee, McGlone, & Wright, 2005; McDonald-Madden et al., 2010).

Contractional, knowing how to and when to monitor biodiversity is simply not enough with respect to biodiversity management as information gained from adequate monitoring is not necessarily a conservation gain. In some instances, perhaps monitoring is less important than action, whereas for most species there is not enough knowledge about the threats or states of populations, so monitoring is essential (McDonald-Madden et al., 2010)

It is not possible to monitor all things, everywhere. It is not realistic to attempt to understand every single species within an ecosystem and the role they play, however, it is important to monitor as within remnants are overlooked (Lee, McGlone, & Wright, 2005; McDonald-Madden et al., 2010). Well-informed decisions must be calculated about situations when monitoring is deemed appropriate, and investigation is necessary. This can become a challenge when there is no observed or apparent strong ecological, environmental, or economical case to instigate monitoring programs that cater to appropriate decisions about when monitoring programs are and are not appropriate (McDonald-Madden et al., 2010).

Institutional monitoring has targets for monitoring and management, whereas biodiversity monitoring is driven by a curiosity about nature and aims simply to document how nature is changing. Much of the collective monitoring done globally is through citizen science, most of which is curiosity-driven (e.g. iNaturalist and eBird). This provides an understanding in the scope of trends in ecosystem or environmental changes (such as population size, species presence or disappearance, water quality, etc.) and assists in bridging the gaps of knowledge (Lindenmayer & Likens, 2010; Sparrow et al., 2020). Approaching biodiversity monitoring is challenging as institutional monitoring invests little funding into curiosity-driven monitoring due to no direct purpose, why it is largely done by interested volunteers through citizen science. Monitoring that is driven by curiosity often grasps the broad significant changes or trends in monitoring that otherwise institutionalised monitoring excludes (Lindenmayer & Likens, 2010).

Despite potentially improving conservation decisions and the awareness of an ecosystem's integrity through monitoring, the benefit of the information gained does not always outweigh the costs of acquiring it. In order to reduce monitoring expenditure, understanding the nature of monitoring costs and benefits in order to evaluate these factors in the context of financial limitations, therefore comparison against financial limitations and the urgency of conservation issues should be investigated (McDonald-Madden et al., 2010).

1.2.1 Biodiversity Monitoring in NZ

In NZ, environmental and biodiversity monitoring has previously been detached from policy, hardly used for assumptions surrounding pressures on biodiversity conditions, and often isolated or ignored from ongoing research (Lee, McGlone, & Wright, 2005). Biodiversity observations have historically been sparse and inconsistent, mainly from occasional local surveys by a range of stakeholders (McGlone et al., 2020). National inventory and monitoring systems have been rare, with exception of periodic bird censuses and native forest inventories (Lee, McGlone, & Wright, 2005). However, today there is mor incentive towards biodiversity monitoring.

Nationally there are significant, intensive, and long-term monitoring of a few selected local ecosystems by agencies and universities (McGlone et al., 2020). NZ has a robust system for surveying vegetation, the National Vegetation Survey (NVS) (Manaaki Whenua Landcare Research, 2021a), which has created a large physical and electronic archive databank. NVS provides records spanning over 70 years of native and exotic plants in NZ terrestrial ecosystems (Manaaki Whenua Landcare Research, 2021a). The physical archive includes plot sheets, maps, and photographs from many years of vegetation surveys. Original surveys were first conducted by the New Zealand Forest Service, Department of Lands & Survey, and the DSIR Botany Division (Manaaki Whenua Landcare Research, 2021a), but are now implemented by a range of organisations, authorities, and the public. This historic monitoring has significantly assisted NZ in addressing issues of current concern that were unforeseen at times of data collection (Hayman et al., 2021).

However, monitoring efforts have advanced in recent decades through international and national conservation pressures and new-aged motives. NZ's growing appreciation for resources, the large-scale involvement, and vulnerability of many sectors, cultures, communities, and individuals with biodiversity supports research and monitoring (Greig et al., 2018). This societal support and recognition for the natural environment have subsequently ordained national and international government initiatives toward sustainable resource knowledge and policies through biodiversity monitoring (Lyver et al., 2017).

NZ has been following international trends toward comprehensive reporting on government activities such as elements of efficient monitoring to gain greater attention to the outcomes of management activities. Additionally, the value and dependency of ecosystem integrity and the indigenous biodiversity within have led to better local monitoring and legislation incentives (Lee, McGlone, & Wright, 2005).

There is now much critique and debate (Costanza et al., 1998; Dominati et al., 2014; Greenhalgh & Hart, 2015; Maseyk et al., 2019; Neupane et al., 2020; Walker et al., 2021b) about the inclusion of ecological service concepts into decision-making and management relating to environmental ethics, human-nature relationship value, and intrinsic worth. Additionally, there is common uncertainty about the concept of biodiversity, ecological service valuation, and the classification of payments of these ecological services (Greenhalgh & Hart, 2015).

In NZ, monitoring is performed by a wide spectrum of agencies and groups, such as DOC, Regional Councils, Ministry for the Environment, Statistics NZ, Ministry for Primary Industries, Ministry for Affairs and Trade, Tangata Whenua, and nongovernmental organisations. Within each hierarchy of the NZ government and governmental authorities– central government, regional councils, territorial authorities, and governmental organisations – there are different natural resource management responsibilities (Lee, McGlone, et al., 2005; McGlone et al., 2020). Statistics New Zealand has contributed and initiated a collaboration of data sets for national environmental accounts for reporting internationally on the status of our biodiversity. These responsibilities coordinate appropriately with international and national context Act legislation obligations (Stats NZ, 2019).

NZ is a party and signatory to several international agreements that obligate government to protect biodiversity (Nature Editorial, 2020; United Nations, 1992). The international conventions or agreements do not hold the power to directly enforce the law through NZ courts unless protection requirements are incorporated into legislation. In the situation where legislation is unclear, a presumption of statutory interpretation may be incorporated when it is consistent with NZ's international obligations (Lee, McGlone, et al., 2005; McGlone et al., 2020). These obligations are binding to governmental agencies, such as DOC, and local governmental authorities, such as regional and district councils. These obligations are measured through efforts and outcomes of biodiversity monitoring.

International obligations and motivations can have significant moral force and persuasion within the country. The government, and NZ people alike, do not like to be perceived as acting in contravention of the country's international obligations (Lee, McGlone, et al., 2005; McGlone et al., 2020). Further, NZ has a history of reinforcing social and environmental strategies on a global scale. Therefore, NZ must keep up with the changing times and adhere to its obligations to do what is right globally, to better the country, the environment, the people, and the surrounding world (Lee, McGlone, et al., 2005; McGlone et al., 2020).

Monitoring the status and trends in NZ native forests is required for national legislation and international agreements, and can assist the DOC and local authorities with conservation performance and achievement (Bellingham et al., 2019). Currently, biodiversity monitoring schemes are largely devoted to significant biodiversity management issues or individual investigator interests, providing useful information to help address specific management concerns. However, these schemes often ignore the overall state of biodiversity (Lee, McGlone, & Wright, 2005).

NZ Biodiversity Monitoring Issue

There is a large absence of long-term biodiversity monitoring, consequently resulting in gaps invaluable data for many regions, meaning there is little active management for the protection and conservation of biodiversity (Willis, 2015). Subsequently, there is an inability to distinguish between natural variation over time and space, and the ecological aberrations, such as biological invasions induced by human activities (Bellingham et al., 2019) making it difficult in knowing what has been lost and how to therefore approach conservation intervention. Limitations of long-term biodiversity data have created uncertainty about the status and trends of our native biodiversity and thus the ecological integrity nationally (Willis, 2015).

Although there are regional and local authorities, there is currently no standard framework or processes which councils follow that allows them to aggregate and consolidate the data necessary for documenting ecosystem changes and assessing the successfulness of management practices (Bellingham et al., 2021). Instead, there is a wide variation of the monitoring efforts and methods used by Regional Councils on a national basis, with some councils developing monitoring methods using a combination of current best practices and have derived methods independently of one another (Bellingham et al., 2021). There is limited capacity to support evidence in reporting ecological status or management effectiveness and intervention approaches. Additionally, there is no means to aggregate data across sites or regional councils, resulting in uncertainty and minimal benefits for shared knowledge across the country (Bellingham et al., 2021). Additionally, constraints in methods, budgets, labor, and expertise result in a limited understanding of the actual ecosystem integrity and the effectiveness of applied management options. This current system (or lack of) often ignores specific and vulnerable ecosystem types and excludes synergies that would come from a national best practice method (Bellingham et al., 2021).

Regional Councils have classified areas of prioritised ecosystems as it is unrealistic to expect operational biodiversity to manage or monitor everything. However, there is also no consistent approach to the prioritisation amongst Regional Councils. The different approaches, values, criteria, and measurable scales that are applied for prioritising across the country have caused significant reliance on exterior professionals with subjective assessments (Willis, 2015).

The urgency to develop a framework that’s is able to measure local biodiversity and ecological integrity crucial as current, Regional Councils efforts for monitoring terrestrial biodiversity of different ecosystem types and thus its integrity across private land is not systematic and is often idiosyncratic (Bellingham et al., 2021). Therefore the current monitoring schemes limit the capacity to support evidence in reporting ecological status or management effectiveness and intervention approaches. Additionally, there is no means to aggregate data across sites or regional councils, resulting in uncertainty and minimal benefits for shared knowledge across the country (Bellingham et al., 2021).

NZ Biodiversity Monitoring Resolution

Accordingly in 2020 and 2021, the Tier 2 Terrestrial Biodiversity Monitoring Framework (T2BMF) Bellingham et al., (2021), project was developed by Manaaki Whenua Landcare Research, with the advice of Kevin Collins (Collins Consultancy). This project was led by the Hawke's Bay Regional Council on behalf of the Regional Councils' Biodiversity Working Group seeking to assess ecological integrity, in response to the management of sites of interest to Regional Councils (Bellingham et al., 2021).

The Tier 2 Biodiversity Monitoring Framework (T2BMF) takes a more systematic approach to measuring NZ's native biodiversity. It is derived from the DOC monitoring systems of information known as tiers, which operate at different scales with varying levels of detail and coverage mentioned above. All the Tier monitoring systems are needed to help predict and interpret national and local-scale trends (The Department of Conservation, 2021). These Tiers provide a consistent approach so that new and historic data can be combined and compared across regional council projects, and used to build an understanding of NZ's ecological integrity (McGlone & Dalley, 2015). The T2BMF coordinates and complements elements with this established monitoring and reporting system.

The T2BMF project objective was to develop a harmonised approach to ground-based monitoring across all the Regional Councils to correlate management effectiveness for targeted or prioritised ecosystem types (Bellingham et al., 2021). The framework uses recommended monitoring tools or methods aiming to assess and determine the ecological integrity of specified prioritised ecosystems of sites that are secured, enhanced, or maintained through active management (e.g. fencing, pest control). The framework approach standardises monitoring so Regional Councils can receive and exploit the greater synergy of benefits, including shared knowledge, regional biodiversity inventory and status, effective management interventions and adaptions, and improved communications and interactions with communities across the country (Bellingham et al., 2021).

The framework provides details regarding collected indicators of forests (and other ecosystems based on classification) ecological integrity: native dominance is measured and observed as the largest relative cover of native and non-native plants, the relative abundance of native and non-native birds within the forest, and the abundance of pest animals (Bellingham et al., 2021). The abundance of native and non-native species is measured through species occupancy which is the size or class structure of observed species and the distributions of widespread of these species or the species grouped throughout a typical natural forest (Bellingham et al., 2021).

1.3 Study

My research proposes to support agricultural industries' transformation into a more sustainable and biodiversity-conscious future through monitoring the biodiversity of native forest fragments. I have been fortunate enough to conduct this study within my homeland in the Hawke's Bay region, providing an assessment of the ecological integrity of lowland forests across farmland. My study aims to explicitly link the relationships between the abiotic environments of the site, the biotic components found within each plot, and the disturbances observed, such as browsing (Molina-Montenegro et al., 2019).

My study uses the T2BMF together with additional biodiversity data collection to survey across lowland Hawkes Bay the vegetation cover, birds, and pest mammals in a selection of 10 ecologically prioritised native forest sites. Ecological prioritisation is based on regional council and landowner interest with connections with the location, size, ecosystem type, and vulnerability of the native forest. The study investigates sites of similar age and with comparable environmental conditions, with a comparison of management regimes between fenced or conventionally fenced to determine the effects on biodiversity. Additional monitoring biodiversity data (e.g., bird audio recorders) we used to augment the T2BMF to measurable ecological patterns in more detail.

Together, the recommended T2BMF and additional monitoring aspects both allowed for an assessment of the suitability of T2BMF for measuring ecological trends. Overall, my study will justify whether the T2BMF is suitable for measuring the ecological change in response to different management regimes, such as the effects of fencing, by focusing on vegetation and bird composition. My assumption is that if ecological differences between fenced and unfenced sites can be detected and described in detail using the T2BMF, then the framework will also be suitable for detecting ecological changes over time of similar magnitude. Within the sites, I used different measurements to assess the variation of biodiversity, such as sites' age, size, slope, soils, exposure, facing direction, etc. – allowing consideration to related biodiversity impacts greater than just management. My data will be archived in the National Vegetation Survey database for future use.

My study assesses the ecological integrity of sites and provides the HBRC with baseline data for future monitoring at these sites. My study provides an understanding of the status of biodiversity in lowland forests on private land in the Hawke's Bay region. This is important as forest integrity in this area is largely unknown, highlighting the need for Regional Councils to implement a terrestrial biodiversity monitoring framework that will guide councils and landowners in understanding ecological integrity, such as the T2BMF.

My study demonstrates the effectiveness of monitoring regimes, such as the T2BMF to other councils that can therefore direct what councils should monitor and with what depth they need to document the status of their prioritised ecosystems. My results will open discussions about the effects of grazing, pest control, a perspective of invasive weeds, and the urgency to monitor native forests. I hope my research offers concepts for improving management practices that protect and enhance native biodiversity across farmland.

1.3.1 Study Objectives

I. Assess the effectiveness of fencing for protecting lowland forest fragments in the Hawkes Bay.

II. Collect baseline data of the vascular plant, bird, and mammal diversity of native forest fragments within Hawke’s Bay farmland

III. Trial and assess the proposed Tier 2 Biodiversity Monitoring Framework for the ecological monitoring of lowland forest.

Chapter 2

Study Sites & Methods

In this chapter, I explain the procedure for selecting study sites and describe each of these study sites in detail. This chapter then goes on to outline the monitoring methods implemented in this study.

2.1 Studied Sites

This study investigated 5 areas across the lowland Hawke's Bay region (Figure 2.1), with each area consisting of paired (2) native forest fragments sites (fenced and unfenced). Information for the description of the following studied sites has been sourced from the private landowners, mātauranga from affiliated communities or people, and related local literature.

All of the studied sites were described as a forest fragment as they were small in size (area ha) (being less than 27 ha) and with few native forested areas in the surrounding landscape matrix. It is disclosed that many of the fenced sites were larger (area ha) compared to unfenced sites. It is assumed this is because it is more purposeful to fence larger forested sites.

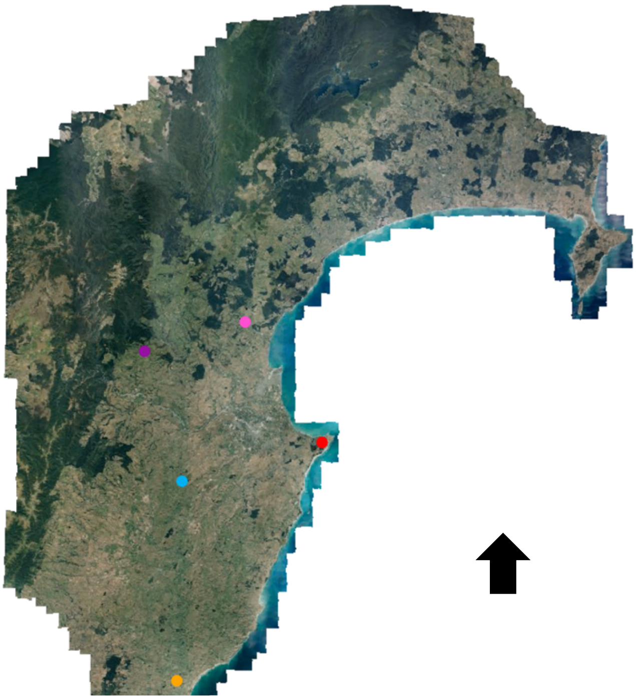


Figure 2.1: Study areas distributed across the Hawke's Bay region. Each dot represents a set of paired (fenced and unfenced) indigenous forest fragments (blue = Raukawa; red = Cape Sanctuary; purple = Waiwhare; orange = Pōrangahau; pink = Glengarry). Scale 1:1770561

2.1.1 Raukawa

Raukawa is situated in rural Hawke’s Bay on the East Coast of North Island, part of the Hastings District, 17 km south of Hastings city. It is a small settlement nestled west of the Kaweka Range, connecting to the Maraekakaho area and the Heretaunga Plains on the east. Raukawa settlement has the upper Patangata stream running through the valley, which connects through to Mangarotai stream and then finally the Waikorure river.

The pre-human settlement, this site was covered in lowland podocarp and hardwood forest (Wilmshurst, 2012). Now, there are very few old-established native fragments in the Raukawa landscape, primarily in retired farmland near streams or as shelters for stock across hilly pastures.

Once this area was part of the tribal territory of the Ngati Kahungunu, who were raided in the Musket Wars in the 1820s by war parties from the Waikato and Hauraki. During the 1870s, the Plains were targeted by the Europeans and leased off from Māori for modified pastural farmland investments (Boyd, 1984). The Plains and the backcountry ranges, like Raukawa, were suitable for stock grazing encouraging landscape modification and creating the current pastural matrix we see today in the Raukawa area (Boyd, 1984).

Much of the district’s landscape is now coated in vineyards, orchards, and lifestyle blocks. The association of temperate climate, fertile soil and availability of groundwater for irrigation enables the production of an annual harvest of fruit, vegetables, grapes, pastural grassland, and crops (Brown et al., 1999). Kaweka Ranges feature warm temperatures, high solar radiation, low annual water deficits, and very low monthly water balance ratios. The Raukawa range is formed by limestone and is known to be very fertile. The Anaroa valley has alluvial and peat soil characteristics (Forbes, 2012). native cover is very scarce and those sites which remain are isolated amongst exotic grassland (Forbes, 2012).

Prepared for HBRC, Forbes (2012) found that tītoki (*Alectryon excelsus*), cabbage tree (*Cordyline australis*), and tōtara (*Podocarpus totara*) were the most common tree species in the Raukawa area. Forbes (2012) also recorded karaka (*Corynocarpus laevigatus*), pigeonwood (*Hedycarya arborea*), kaikomako (*Pennantia corymbose*), tawa (*Beilschmiedia tawa*), ngaio (*Myoporum laetum*), rukawa (*Metrosideros robusta*), turepo (*Streblus heterophyllus*), lancewood (*Pseudopanax crassifolius*), and rewarewa (*Knightia excelsa*) can also be present in varying abundance (Forbes, 2012).

Native forests were rare within the Karamu catchment, and even more so within the Raukawa District. In this study, the paired sites (fenced and unfenced) are located over two different land holdings, 2 km from one another (Figure 2.2).

Mahana Bush

The Mahana Bush (Figure 2.2), being the largest protected QEII covenant for the catchment area, is one of the most ecologically spectacular forests across the entire Karamu catchment. Mahana bush par of Kirk family farm, with Carolyn and Duncan Kirk having owned this property since the early 2000s.

Carolyn Kirk's Grandfather, Bill Nelson, purchased a segment of the property in 1934. Over the century, Carolyn's father, and now herself and husband Duncan Kirk, have expanded the block into an area of 630 hectares of owned and leased land for sheep and beef. The surrounding landscape has experienced significant modification for stock grazing.

Donated to the Crown and fenced in 1998, the forests cover an area of 8.3 hectares. However, due to accessibility, the studied area is only 5.6 hectares. Carolyn's father, Forklin knew of its significant ecological importance and therefore drove the fencing of it. The Kirk family takes great pride in this forest as they have witnessed progressive ecological change throughout the years of protection and management. The forest looks over the Anaroa valley across the Raukawa settlement and is very aesthetically pleasing for locals.

Forbes (2012) surveyed the Mahana bush, recording 121 species of naturally occurring plants, 85 being either native or endemic. This was made up of three naturally occurring gymnosperms, 47 angiosperms, seven monocots, one orchid, and 23 fern and fern allied species (Forbes, 2012). The Mahana Forest has been described as a titoki-dominant broadleaf forest, including a few scattered podocarps and emergent rewarewa. Interestingly being so far from the coast, a sole nikau palm tree (*Rhopalostylis sapida*) was found (Forbes, 2012).

Forbes (2012) observed tūī, korimako, kererū, riroriro, and pīwakawaka, ruru (*Ninox novaeseelandiae)*, rifleman (*Acanthisitta chloris*), Australian harrier (*Circus approximans*), whitehead (*Mohoua albicilla*), tomtit (*Petroica macrocephala*), and silvereye (*Zosterops lateralis*). Many exotics were also observed, including California quail (*Callipepla California*), finches (*Fringillidae pp.*), sparrow (*Passer domesticus*), blackbird (*Turdus merula*), song thrush (*Turdus philomelos*), starling (*Sturnus vulgaris*), myna (*Acridotheres tristis*), and shining cuckoo (*Chrysococcyx lucidus*). Additionally, within the forest, Forbes (2012) found forest gecko (*Hoplydactylo ranulates*) and green gecko (*Naultinus elegans*).

Since being fenced, it has been observed by the Kirks, and findings from Forbes (2012) that the forest has been recovering well from previous decades of grazing as they have witnessed the denser growths of vegetation which has dramatically regenerated since it was fenced in the late '90s. Duncan remarked how within such a short time frame, the density of the understory has thickened significantly, and remembers when the understory was bare but now finds it a real struggle to walk through. Duncan has witnessed a phenomenal increase in birdlife and expressed how loud the birdsong emits from the forest. Along the fence line of the fragment, there have been some native plantings to help restore and protect the regenerative edge.

Within the forest, Duncan and Caroline Kirk have observed hundreds of hedgehogs, a large population of feral cats, which locals are trying to control. The farm organised bait stations established for possum and rat control. Historically, the forest experienced much browsing damage from possums in the ’90s. Duncan recalls shooting 90 possums in one night. In addition, the Kirks spray blackberry throughout the farm and on the edges of the fenced forest fragment every few years.

Dave Kaijser Bush

The unfenced forest site covers 4.07 hectares over a small gully surrounded by pastural grazing land (Figure 2), owned by Dave Kaijser. Dave converted a segment of his property into an apricot orchard. The orchard was adjacent to the forest fragment but has since been taken down back into grazing land. The property and the forest experience light lifestyle grazing of cattle, sheep, and donkeys but had previously been intensely grazed by stock for several decades. Dave Kaijser's property was once part of the grand Rautama station, which covered the southwest side of Raukawa, but since the station was carved into 7 lifestyle properties in 1995, of which Dave purchased 7.6 hectares 26 years ago. It is thought that this forest fragment is a remnant and direct regeneration of the Anaroa forest which was milled for tōtara in the early 1900s and turned into pastural landscapes for sheep and beef practices. The Anaroa Forest once connected Mahana Bush and the Kaijser forest.

Possum control was conducted 18 years ago, but since then there has been no need as possum populations are absent in the area. Dave did remark that since grazing intensity had diminished, the blackberry populations have grown and have become a problem in some areas, stating that he couldn’t miss year a of blackberry spraying.

Dave has noted that at current, the biodiversity status seems stable as there is constantly a lot of bird life. He has also observed that since the intensity of grazing lessened thee, regeneration of native vegetation has exploded. He has noticed that within the 26 years of ownership, the forest fragment has regrown and expanded noticeably more so. Dave remarks how he perceives the bush to be healthy and functioning. He is also mindful of the property's good water, having a tank and pumps at the bottom of the forest that taps into a spring.

The Kaijser family enjoys the forest fragment. Dave explains how they all love being present in nature and their bush is a good place to relax and hang out. They have placed several park benches overlooking the forest to appreciate its nature and have mountain biking and walkway tracks throughout the bush to enjoy.

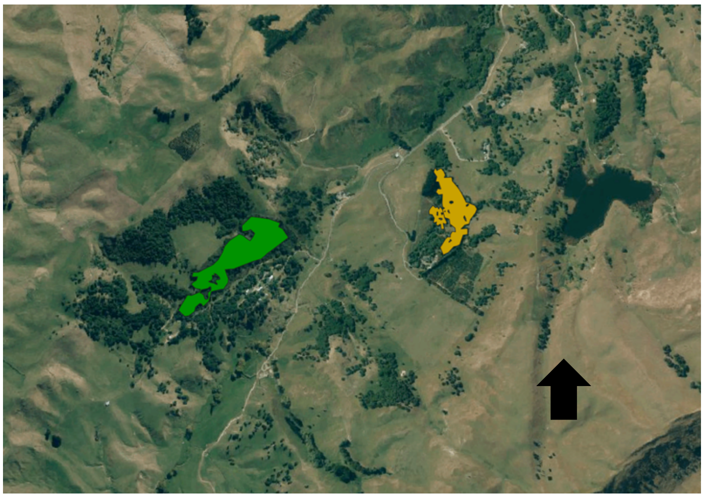


Figure 2.2: Raukawa study sites (green = fenced and yellow = unfenced) found at the end of Anaroa Road. Coordinates of these sites and the surveyed plots within are found in the appendix. Scale 1:20000

2.1.2 Cape Kidnappers

Cape Kidnappers is an extraordinary sandstone peninsula to the east of Hastings. To Māori, the peninsula was named Te Matau-a-Māui, after a fish hook with which Māui hauled up the North Island (Pollock, 2009). Cape Kidnappers was named by Captain Cook after an attempt by local Māori to abduct one of his crew. The Cape supports the largest and most accessible gannet colony in the southern hemisphere (Cockrem et al., 2016).

The Cape Sanctuary

The Cape Sanctuary is a privately owned conservation sanctuary with a pest-proof fence that cuts across the Cape Kidnappers Peninsula over 2500 hectares, covering a large land mass that offers an array of habitats within. The fenced area covers forest plantations, crops, pastural farmland, and rare, native prioritised ecosystems. The Sanctuary aims to be a modern model for sustainability, with human habitation, farming and forestry, and tourism activities co-existing and thriving with native and endangered species.

The Cape Sanctuary was founded by Andy Lowe, Julian Robertson, and their families in 2006. Andy had become worried by his surrounding silent native bush and coast systems, and he had observed how the local bird numbers had collapsed. Andy wanted to offer a model of conservation efforts for other concerned private landowners by implementing a large-scale conservation project to protect and enhance NZ's threatened species (The Cape Sanctuary, 2014).

Joining Andy was Julian Robertson, the owner of the Cape Kidnappers' golf course and a generous philanthropist in NZ, who supported Andy in the funding of the Sanctuary. This operation relies on private funding, commercial outputs, and mass efforts of committed volunteers (The Cape Sanctuary, 2014). Ongoing support from the HBRC and government agencies has helped the Sanctuaries accomplish. The Sanctuary employs a small team of dedicated wildlife staff who manage and control pest eradication, monitor biodiversity, and run conservation projects.

The fence is known as a 'leaky system' as it supports working farms, forestry, and tourism operations inside the wire. The barrier has serval gates around the perimeter with a wire gauge that mice can get through. There is also no fence along the beach waterfront to protect both ecological movements and the amenity values for beach users. This means that this fencing system will never be able to withstand an entirely predator-free environment (The Cape Sanctuary, 2014). Managers acknowledge that due to the Sanctuaries' commercial operations, a vermin-free environment is impossible, However, pest control has been operating since 2006 and this has proven to dramatically reduce the number of predators inside the sanctuary. The management practices use aggressive pest and predator controls such as trapping, spot shooting, and 1080 inside the wire. Using current technology for eradication, wildlife managers do not expect to completely remove these pests (The Cape Sanctuary, 2014; Walls, 2014).

The Sanctuary supports a large dynamic stretch of sandy coast dune systems known as Ocean Beach and Rangaiika. The Ocean Beach coast is recognised as an outstanding area in a national and regional context as it contains rare biodiversity. However, the Ocean Beach dune system had historically experienced intense land-use pressure or modification from human settlement. This depleted the system's ecological condition. The poor ecological state of the sandy dune system encouraged the initial pest-proof fencing across the base of the peninsula and covered the majority of the system (Walls, 2014). The fencing offered the dunes a massive restriction of public vehicle access, exclusion of farm stock, an intensive program for animal and plant pest control, and restorative plantings. The most evident success of ecological recovery from the pest-proof fence of the Sanctuary has been exhibited in the Ocean Beach Dunes and the Maire Raunui Forest. Other ecosystems within the fenced area have also benefited but have not recovered as much because of the continued presence of domesticated stock (Walls, 2014).

The Cape Sanctuary believes in open-source conservation whereby sharing knowledge efforts so that NZ's biodiversity can thrive, especially with other non-governmental conservation projects. The Sanctuary was founded with the intention of sustainable conservation through protecting endangered species and enhancing the natural environment while commercial operations were maintained inside the fence. This concept achieves responsible and ethical social, economic, and environmental or conservation purposes (The Cape Sanctuary, 2014).

The Maire Raunui Forest

My study focuses on the centre of the southern part of the Sanctuary (Figure 2.3), where there is an extensive bush-clad area at the head of the main catchment. This forest area is dissected by steep gullies that merge to form the flat gravelly central valley, draining northeast. The forest has the dignified name of Maire Raunui, in honor of the lone black maire (*Nestegis cunninghamii)* tree. This forest is primarily dominated by kanuka, with few ruminates from the prehistoric forest. In 2003 before fencing, the Maire Raunui Forest had historically experienced harsh ecological damage from feral goats (*Capra aegagrus*), wandering stock, and rabbits (*Oryctolagus cuniculus*), hares (*Lepus europaeus*), and possums (*Trichosurus vulpecula*). As a result, there was minimal understorey growth and regeneration, apart from inaccessible gullies.

Since the Sanctuary was fenced in 2006 and implemented pest management control, the Maire Raunui Forest has changed dramatically (The Cape Sanctuary, 2014; Walls, 2014). Contracted by the Cape Sanctuary owners and the Hawke's Bay Regional Council, Walls (2014) examined the status of the biodiversity and the observed pressures or changes within different ecosystems situations across the fenced peninsula. Walls (2014) Maire Raunui Forest vegetation survey shows a remarkable growth of dense undergrowth of ferns, saplings, seedlings, and shrubs beneath the old-standing adult tree canopy. Leading the regenerative forest growth was mahoe (*Melicytus ramiflorus*), rewarewa (*Knightia excelsa*), pigeonwood (*Hedycarya arborea*), cabbage tree (*Cordyline australis*), native broom (*Astragalinae* spp.), climbing rata (*Metrosideros fulgens*), rukawa (*Metrosideros robusta*), and various ferns. Surprisingly, this forest contained a richness of 28 native ferns indicating the area's reliable humidity, despite the arid landscape. Even in exposed and harsh open areas, manuka, kanuka tauhinu and rewarewa are regenerating apace (Walls, 2014).

Supplementary to this vegetation response and growth, Walls (2014) found a clear renaissance of birdlife in the forest. It was observed that the forest is now busy with tūī, korimako, kererū, riroriro, pīwakawaka, ruru, rifleman, toutouwai (*Petroica longipes*), Australian harrier, whitehead (*Mohoua albicilla*), tomtit (*Petroica macrocephala*), silvereye (*Zosterops lateralis*), kākāriki (*Cyanoramphus novaezelandiae*), North Island brown kiwi (*Apteryx mantelli*), North Island kākā (*Nestor meridionalis*), and saddlebacks (*Philesturnus rufusater*). Since fencing in 2006, wildlife managers have reintroduced several endangered or threatened bird species into the Sanctuary, where many have thrived. The reintroduction of kiwi, saddlebacks, whiteheads, kākāriki, toutouwai, rifleman, and tomtit has been successful in the absence of predators and due to the vegetation recovery (Walls, 2014). Walls (2014) concluded that within the forest, the invertebrate fauna is considerable with abundant tree wētā (*Hemideina* spp.), native spiders, and cicadas (Cicadoidea spp.), butterflies, and moths. It was suggested that native reptiles probably have also made a comeback with active conservation management (Walls, 2014).

The eradication of goats and possums, and the decimation of rabbits, rodents, and mustelids have allowed a massive surge of regeneration within the forest. However, the forest the number of these pests has since recovered and rabbits, rodents, and mustelids still endanger much of the native flora and fauna (Walls, 2014). Additionally, the biomass of *Vespula* wasps within the forest was frightening and there was a concern of competition for native fauna. These threats will be an ongoing battle for wildlife managers due to the leaky fence system and constraints on pest control (Walls, 2014).

Study areas

Much of the Maire Raunui Forest is difficult to farm and a significant area had appropriately been fenced off and retired in 2013. This research compares the biodiversity status of the conventionally fenced area against a non-fenced area within the Maire Raunui Forest (Figure 2.3). The studied conventionally fenced area is situated northwest of the forest and covers 24 hectares. The non-fenced forest studied area is east of the fenced site and follows along the flat valley. The non-fenced area was clipped to a size of 18.90 hectares to match the fenced. Like the survey, when Walls (2014) surveyed the area, the forest still held farm stock in the valley of the northern end of the block. While surveying, the unfenced area contained stock but shortly after the completion of my survey, the area has since been retired.

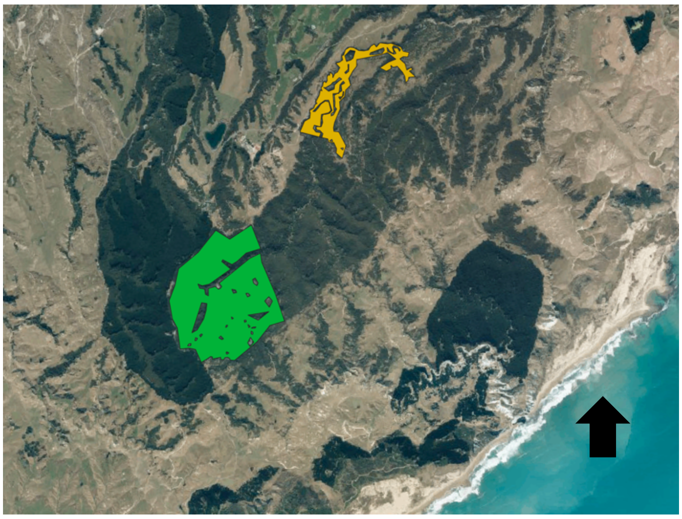


Figure 2.3: Cape Kidnappers study sites (green = fenced and yellow = unfenced) found within Cape Sanctuary. Coordinates, the area, the elevation of these sites, and the surveyed plots within are found in the appendix. Scale 1:40000

2.1.3 Waiwha

The most inland studied site for this research is located in Waiwhare, situated 50km west of Hastings on the Napier-Taihape Road. The Waiwhare landscape has experienced significant modification. The area has been burnt several times, both by Māori and European settlers in the 1800s. Much of the lowland and foothills of the Kaweka forest have been logged for timber (Wright, 1985).

From Kuripapngo to Waiwhare, the foothills of the Kaweka Ranges, the condition of the land was one of the prime factors which influenced farming in the area as from the early 1900s, the land was relatively clear, and it was possible to farm sheep without difficulty. However, manuka (*Leptospermum scoparium*) and kanuka (*Kunzea ericoides*) scrub slowly began to encroach and reduce the stocking capability over time becoming an ongoing battle of clearance for landowners. On the route to Taihape, the productive land creates a quiet rural community (Wright, 1985).

It is assumed that the forest fragments across the district were not cleared due to the steep contour and poor land and only parts were suitable for grazing but are left to act as a shelter for stock during harsh weather conditions. The area has high soil moisture and experiences persistent wind and has a rainfall of 1224 mm of rainfall per year (Wright, 1985).

My study focuses on a forest block located on Bryan Livesey's sheep and beef 267-hectare property, purchased in 2006. The farm runs 10 stocks per hectare across 210 hectares of productive land, with 30% of the farm being in tree, either of native scrub or pine plantations. There is a stream, that has a low flow but tends to flood on terraces with heavy rain and runs through the forest covering both the fenced and unfenced sites. The stream within this fragment ranges from deep, sheer banks to flat and broad and is predominantly shaded by the forest canopy.

There is a strong conservation attitude within the district. The surrounding area has undertaken intensive possum, rodent, and mustelid programs since 2003, conducted by Forest and Bird and members of the community. These programs run voluntarily to reduce pests by regularly servicing traplines and bait stations throughout the district. The local volunteers also maintain tracks through bush reserves, carry out restorative plantings, and have ongoing removal management of wildling pines (Forest and Bird, 2016).

Within the studied forests, Bryan reported that there are few possums within the forests, but bait stations are set to further reduce numbers. Both the study sites show evidence of browsing from ungulates, suggesting movement from the Ranges, across the reserves, and, throughout farmland. Bryan has observed occasional wild pigs.

This studied forest (Figure 2.4) has a neighbouring lowland forest giant, known as Blowhard Bush. This bush was donated in 1962 as it is recognised as an ecologically significant forest having survived deforestation. The bush has outstanding native vegetation, including a matai (*Prumnopitys taxifolia*) that is 25 m tall and thought to be 700-800 years old. This bush also contains precious and delicate orchids including the *Thelymitra "*comet*"*, found only in and around the Kaweka Ranges (Forest and Bird, 2016; Wright, 1985). It has been observed by Forest and Bird surveys that since intensified predator control, the birdlife within the district has benefitted immensely, with reports of tūī, korimako, kererū, riroriro, pīwakawaka, toutouwai, tomtit, and silvereye (Forest and Bird, 2016).

These sites are a vital part of this study as it relatively inland and at high altitudes. Additionally, because this lowland site is near the foothills of the Kaweka Range, it is an important connective corridor within the transition into a farmland matrix.

Livesey QEII Covenant

The QEII block (Figure 2.4) was conventionally fenced in 2009 with its fence running electric to stop the stock from entering, covering an area of 3.9 ha. The area has steep slopes but also contains a flat terrace within. The forest was encouraged into a QEII covenant as it contains several old remnants of mountain beech trees as they are rare within the region. The protection of this forest is vital as it is a large intact corridor offering support for species movement and acts as a regenerative seed source for other forests across the district.

The fencing of the QEII block also allows Bryan to have better management and access to his stock. The Livesey family feels proud of the QEII covenant, and Bryan feels a great accomplishment in seeing the difference in the understory within the fenced block. The family enjoys spending time in the covenant and often showcases the forest to visitors and friends. When the farm "runs dry" Bryan says he is sometimes tempted to put stock in the covenant, but quickly rejects the idea as he knows that the success of the block will diminish.

Livesey Unfenced Forest Fragment

The paired unfenced forest resigns south-east, right next to the QEII covenant, covering an area of 2.25 hectares (Figure 2.4). The unfenced forest holds much value for stock as it aids as a shelter and lambing area for protection against harsh environments. Being grazed, the forests have pastural grass to support the stock. Alike the QEII site, this unfenced forest offers species movement and opportunities for vegetation seeds to disperse.

In comparison with the fenced QEII covenant, this forest shape or format is slightly different, being much longer and slimmer as it follows the creek. However, the unfenced follows the site selection rules and its shape can be overlooked as within the study the surveyed sample points are randomly generated, regardless of interior or edge effects throughout all the sites.

This area has the prospect of becoming conventionally fenced soon through HBRC and QEII funding. Bryan has an ambition to the fence and retired the land to protect the creek running through and prevent erosion. Fencing will also assist in better management and access of the stock as they won't get lost or caught up in the bush.

Figure 2.4: Waiwhare study sites (green = fenced and yellow = unfenced) found along the Taihape Road. Coordinates, the area, the elevation of these sites, and the surveyed plots within are found in the appendix. Scale 1:10000

2.1.4 Pōrangahau

Pōrangahau is part of the rolling coastal hill country in southern Hawke's Bay. In Māori, Pōrangahau means mad winds, so named due to the prevalence of heavy westerly gales blowing from the seacoast. Pōrangahau is situated in the Patangata country, 120 km south of Napier (Cyclopedia Company Limited, 2006). The area receives low annual rainfall (700-900 mm) and soil moisture extremes are regularly experienced (Sheath et al., 1990). Although this area is very rugged and with harsh environmental conditions, it is hugely productive (Cyclopedia Company Limited, 2006).

The area enjoys significant Māori territorial history. At various times, the people of Pōrangahau had built and occupied at least 19 pā. The people of Pōrangahau were fortunate as the area provided a lagoon that provided an array of salt and freshwater species throughout the seasons. Up along the east coast, there were once fishing communities associated with Hapu from Pōrangahau (Ballara, 1992). The swampy grounds also offered the people a food source of birds, eels, and useful plants such as raupō (*Typha orientalis*). All across their territory, there was a braided network of streams that connected to old standing, podocarp, and hardwood forests that provided even more resources of timber, birds, berries, and wild vegetables (Ballara, 1992). The Pōrangahau area is now a matrix of farmed pastural systems, harvested forest plantations, and contains snippets of old remnants of the pre-European coastal forest. Pōrangahau estuary is the largest and least disturbed estuarine environment on the east coast, south of the Bay of Plenty (Cyclopedia Company Limited, 2006).

Pōrangahau is at the north-eastern edge of the Seventy Mile Bush, a dense native forested area extending from Wairarapa to Central Hawke's Bay and onto the East Coast. During the 1870s, this bush was cleared by slash and burning by the newly arrived Scandinavians who cleared areas for farmland settlement, from both south and north ends of the forest (Arnold, 1994). The bush was once known as a vast forest rich in birdlife. Now there are a few pockets of original bush remaining, most notably the 942-hectare Pukaha Mount Bruce National Wildlife Centre protected in 1888 (Arnold, 1994).

The Pōrangahau district has benefitted from significant local community pest management control. Since the 1950s, the site has had efforts of trapping, bait poisoning, and shootings from the surrounding landowners and community efforts. During the 1990s, the HBRC applied cyanide and poisoned approximately 8090 hectares across the Whangahu Valley to eradicate tuberculosis reactors. This was hugely labor-intensive. However, it was noted by locals that after this control, feral cats populated the valley in large numbers.

Across the site, there have also been community and private landowner plantings of native plants since the 1960s. There have been significant plantings of tree lucerne across the landscape in a targeted attempt for an additional food source for nectar-consuming fauna. Community efforts have been planting native vegetation across the banks on the Cook Tooth Road, where the study areas are located.

The Pōrangahau site has prioritised ecosystems because they are a remnant of the pre-European coastal forest, which there are now only a few examples left due to human settlement, clearing, and land modification. Thus, the necessity to monitor these sites of forest fragments is crucial. The Pōrangahau site was specifically chosen due to the rarity of the remnants and the need to get a gauge of the status of biodiversity. The site was essential to understand the importance of fencing management, especially for old and established forests. The studied Pōrangahau sites are found across two different properties as there were no suitable comparable forested fenced and unfenced forests within one property.

Motu Mokai

Te Ana farm was purchased by Guy and Caroline Stoddart in 1992 with additional block purchases in 1998. The land has been in the Stoddart family for 3 generations and the surrounding farm is owned by Guy's brother. The farm runs 6000 stock units of sheep, beef, and deer over 680 hectares.

A significant area of the previously forested landscape was cleared by Guy's grandfather for pastural production purposes. In doing so, the land was then bombarded with a dense growth of opportunist gorse. Guy reported that the gorse once stood 2.5 m high and has since established substantial spray regimes across the land every 2 years for 8 years and now receives spray every few years for control of gorse.

The studied fenced area is appropriately named Motu Mokai Forest, meaning favourite land or forest (Figure 2.5). This forest is situated within a gully, off a rolling pastural hill, with an area of 10.16. Several small slips are scaring the forested area. Along the northern side of the forest, an old track was once operational but has since been replaced by the Cooks Tooth Road. There are a significant number of surrounding properties that have been modified into pine forestry.

The Motu Mokai forest backs onto a 4.5-hectare 2015 fenced QEII covenant which has a younger established vegetation growth compared to the Motu Mokai forest, therefore was not used for this research. It is assumed that the younger covenant forest regenerated after a fire disturbance many years ago. Guy has observed that the QEII understory has significantly grown and become very dense since the block has been fenced.

The Motu Mokai Forest and the adjacent forest has a hugely diverse of establishment adult trees. Vegetation surveys performed by the Hawke's Bay QEII representative, Troy Duncan concluded that the forest contained tōtara, kahikatea, tawa (*Beilschmiedia tawa*), nikau, tītoki, northern rata (150 – 200yrs .5mm per year), rimu, rewarewa (*Knightia excelsa*), lacebark (*Hoheria sexstylosa)*, *Coprosma aereolata*, supplejack (*Ripogonum scandens*), kawakawa (*Piper excelsum*), hangehange (*Geniostoma ligustrifolium*), putaweta (*Carpodetus serratus*), white rata (*Metrosideros albiflora*), *Griselinia lucida*, bush lawyer (*Rubus australis, Rubus cissoides*), manuka (*Leptospermum scoparium*), lancewood (*Pseudopanax crassifolius*), *Coporsma rhamnoides*, ngaio (*Myoporum laetum*), ponga (*Cyathea dealbata*), pigeonwood (*Hedycarya arborea*), cabbage tree (*Cordyline australis*), mahoe (*Melicytus ramiflorus*), pukatea (*Laurelia novae-zelandiae*), karaka (*Corynocarpus laevigatus*), milk tree turepo (*Streblus heterophyllus*), miro, and matai (*Podocarpus spicatus*). Troy also noted how the unfenced area contains a high presence of pukatea, which is deemed to be uncommon in lowland Hawke's Bay.

During the biodiversity survey period, cattle were grazing under the forest. The landowner, Guy Stoddart stated that there was high deer presence and browsing pressure within the non-fenced forested area. He did mention that when possible, shooting takes place, but deer populations were still dense. In comparison, possum bait lines are strong and there is now little presence of possums within the Motu Mokai Forest. Guy has observed that since the eradication of possum populations, there has been an increase in the thickness of the canopy. He was also excited to discover the increase of native berries throughout the bush such as karaka, as for many years he saw little or no berries in the forest. Guy also commented on the occurrences of rata flowering which have now become more abundant. He suggests that these vegetation and biodiversity changes are due to the possum control and the fencing of the QEII block.

Due to this forest's significance, the QEII representative Troy Duncan, HBRC, and landowner, Guy Stoddart have developed a fencing proposal. This will allow protection for the forest to regenerate, so the forest can replenish itself. This is important to Guy as he has a great personal connection to the forest and has observed how the QEII covenant has regenerated and is on a mission to get the same ecological response from Motu Mokai. Guy enjoys learning about the ecological significance of the native species within the forest and feels a sense of pride walking through this bush and sharing the ecological significant stories with family and friends. He stated that the local community's passion for native restorations and protection is contagious, and it is easy to get others to join the excitement.

Te Ana (McLeans)

The Te Ana (formally known as McLeans) is a 27-hectare reserve in Pōrangahau, southern Hawke's Bay. The reserve is located between Ireland Road and Wangahauhi Road, about 16 km south of Pōrangahau and 45km south of Waipukurau via Cooks Tooth Road (Figure 2.5).

The reserve covers a small tributary, headwaters, and catchment area of the Whangaehu River. The reserve has a southeast aspect and is on a moderate slope. Since the landscape has changed into pastural systems, erosion processes are active both in and around the reserve. Farm management plans surrounding the reserve have developed a catchment plan which aims to stabilise the gullies and hillsides around the reserve to mitigate erosion effects. The stabilisation has been established through open block plantings of poplars to ensure the protection of the reserve (New Zealand Department of Lands and Survey, 1975). The areas underlying strata formation is comprised of crushed argillite, glauconitic sandstone, and bentonite clay (New Zealand Department of Lands and Survey, 1975)

The studied forest fragment was part of returned soldier blocks that were billeted off to the two McLean brothers in 1918 after the war. The idea of donating the bush as a reserve was first discussed and encouraged among the landowners by Rob McLean in the early 70s. Rob pitched the idea to his father after being inspired by the Cape Kidnappers reserve, which he had worked on at the time. The area studied was then gifted to the Crown by the McLean family in 1973. In 1975, the reserve was conventionally fenced, and funded equally between the landowners and the Department of Lands and Survey.

Although the reserve is governed by DOC, the forest's true Kaitiaki is Rob McLean. Rob has noted very fine biodiversity details and observations of the forest for the last 40 years. Within Rob's lifetime, the birdlife dynamic has changed as the landscape and vegetation have changed. He noted that there were more seabird and water birds along the banks growing up, but he suggested that they disappeared as possums invaded. In 1945, possums decimated the bird populations. Rob observed how the area used to have a lot of kererū, but they had disappeared. However, after HBRC's application of cyanide across the valley and surrounding hill land, Rob witnessed the birdlife return. He stated that in 1993, the bird life flourished again including the bellbird and tūi who were the first to arrive in the area with the Kererū following.

The farmland surrounding the reserve has been farmed very extensively, but not intensively over the last century. During Rob's father's time, the native trees were surprisingly not cleared. Rob's father remembers that the land used to be covered in toe toes and native rushes. However, Rob explains that since the valley has been ploughed and turned into pastoral land and grazed by cattle, toe toes growths have been destroyed.

Rob also recognises that there are more fenced reserves throughout the area than ever before. In 2021 there are 6 remnant bush reserves, compared to when Rob was young when there were none. There are similar stories to the McLeans / Te Ana reserve and inspired by the McLean family. In the 1980s two other private reserves within the Pōranghau region were created, Arataura and Tautane. Since the establishment of these reserves, they have notably supported species movement and have encouraged forest regeneration.

In 1995, DOC set out bait stations across the reserve. Since then Rob has observed the direct effects of local possum eradication on the thriving vegetation composition. However, within the reserve, he has noticed that some of the very old titokis and karaka is staying to die off. DOC acknowledges that this reserve also processes scientific and educational value. It is traditionally used by the Pōrangahau school for nature studies and supports the school's sense of ecological knowledge and identity.

The Te Ana reserve holds significant meaning and richness to the McLean family, especially Rob who has spent his life protecting the reserve and educating the community. He remarks that seeing natives and knowing their story and importance inspires people to protect them even more. He grew up loving the rata bloom and wants to continue seeing this. Much of Rob's memories and heritage are associated with the reserve and he has been a worthy guardian of it. The reserve is of great importance within the Hawke's Bay region as it represents a sample of the pre-European coastal forest part of the Seventy Mile Forest. This reserve is one of the few ancient forest fragments which holds a diverse range of species or compositions of species that are extremely rare now. Notably, the site contains northern rata (*Metrosideros robusta*), rarely seen on the east coastline (New Zealand Department of Lands and Survey, 1975).

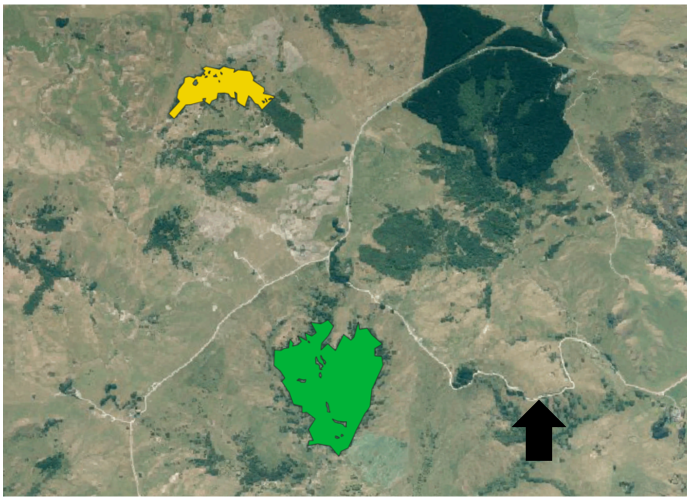


Figure 2.5: Pōrangahau study sites (green = fenced and yellow = unfenced) found along the Cooks Tooth Road. Coordinates, the area, the elevation of these sites, and the surveyed plots within are found in the appendix. Scale 1:30000

2.1.5 Glengarry / Eskdale

The most northern of the study sites are separated into two different properties, covering two districts. The areas are environmentally similar and are only around 8km apart. The rural settlements, Eskdale and Glengarry are 20km northwest of Napier on State Highway 5, heading toward Taupō. Eskdale, a farming settlement east of Glengarry, was one of the first settlements in Hawke's Bay. The area was the scene of several fierce conflicts between Māori and Europeans. This area has various land usages including an assortment of orchard types, vineyards, crops, pastoral land, and pine plantations. The Eskdale valley has taken full advantage of the production which is gained from the rich and flat areas for cropping and grazing (Pollock, 2009). The Esk river running through the valley has a history of flooding. The areas surrounding the forest also experienced fires throughout the late 19th century and were burnt before European arrival due to its dry landscape (Forbes, 2012; Pollock, 2009). The studied areas have experienced huge disturbance and there is now little native old-growth forest in the catchment.

The Mitchell Bush

The fenced study site (Figure 2.6) is in Glengarry on a traditional sheep, beef, and deer farm owned and operated by Mark and Frances Mitchell. Mark is the fifth generation, having his family farm on the land for approximately 100 years.

The forest fragment is sheltered from the north, east, and south boundaries by pine plantations. The fencing of the forest fragments includes these pines; however, these pines were clipped from the survey study. Below these pines, the forest is regenerated and dominated by kanuka, mahoe, and coprosmas. This forest also has a high density of blackberry in some places, grassy patches, and rock faces that have been clipped from the survey. The area's gullies merge and follow down through a stream. It is a diverse remnant, with hot dry ridges and cool moist streams with three large waterfalls. Many of the waterway banks are sheer and high drops. The survey within this area was treated with significant caution and advice from Mark about areas to avoid. The studied area only covered 12.36 hectares.

Knowing of the ecological importance of the gully, Mark talks about approaching his father, to have this remnant forest in the gully fenced and retired. Mark and his father reportedly “cut a deal” whereby the gully would be fenced and retired, on the basis that Mark planted the pines surrounding the gully and fenced the area himself. The deal was done, and the forest fragment has now been fenced for 15 years covering a 20-hectare area. Initially, 11 ha of this area was fenced to prevent stock from entering the bush. In 2020 the remaining gorge area plus additional adjoining bush (approx. 19ha total) was deer fenced due to the impact of feral deer. The deer fence is on the edge of the east, north, and south side of the fragment which is back onto farm paddocks. The studied remnant on the property is at least 70 years old, likely older, and is dominated by kanuka with a diverse dense understory (Figure 2.6). The forest is identified as an Ecosystem Prioritisation site (HB_EP_493) and once represented the titoki tōtara forest which is now acutely threatened in Hawke’s Bay. The forest has high densities of NZ smooth stick insects (*Clitarchus hookeri*) and a populations of whiteheads. *Scandia rosifolia* is present which is now classified as nationally critical. The stream is home to a large number of koura (*Paranephrops planifrons*) and red-finned bullies (*Gobiomorphus huttoni).*

This area has regenerated significantly as Mark has observed how dense the understory has restored since being fenced. He has noted that the regeneration he has observed such as the pepper tree returning, is likely to be because of kererū returning. Mark has also carried out some plantings of locally sourced secondary species around the forest across 2 hectares. These areas have been excluded from the survey as they are man-made. After intense eradication of possums throughout the area, whitehead populations have returned and Mark has noted that the populations have been increasing every year, The forest fragment has been monitored by Kay Griffiths, who has detected the existence of bats within the forest.

There have been several few clusters of old man's beard (*Clematis vitalba*) within the forest which Mark has controlled and keeps monitoring forest for any further growth. The site also had a small patch of African club moss (*Selaginella kraussiana*) that has been routinely sprayed. Key threats to this remnant are feral deer, possums, stoats, rats, feral cats, hedgehogs, cotoneaster, and old man's beard. All these pests are managed through fencing, annual pest and plant control, and predator control using bait stations (50x100m spacing) and traps, serviced four times a year.

While growing up on the farm, Mark talks about spending a lot of time running around in the bush remnant, building huts, looking at all the different native plants and insects, and hunting possums with the family foxy. He spent time catching and releasing koura and eels. 30 years on, Mark is now actively fencing and restoring this bush through pest control and growing and planting native trees which he collects from small surrounding remnants. Seeing the plants grow and the increase in numbers of invertebrates and birds due to pest control provides him with great satisfaction.

Eskdale Ridge

The unfenced forest within the Glengarry site (Figure 2.6) on a farm is known as the Eskdale Ridge, owned by Neil McLaren. Neil brought the 299-hectare sheep and beef farm in 2001. The property is located 5 km down Ellis Wallace Road off State Highway 5. The studied area is 11.20 hectares of shrubland forest dominated by old-standing kanuka, blackberry, and ferns, with pasture throughout. It sits on the west side of the property and is beside a pine plantation. The forest is grazed primarily by sheep, and occasionally cattle. There is a small creek running through the fragment, which stock used to drink. The area has some notable steep banks and slips where the vegetation has diminished, and erosion has occurred.

Historically, the land and the scrub or forests resigning have experienced significant disturbance of clearing. The farm was originally cleared for pastoral production in the 1900s and since then, there has been an ongoing battle between landowners and the regenerative scrub. The most recent and largest clearance of the land was undertaken in 1987, whereby the entire farm was sprayed and cleared to eradicate the undesired scrub. However, the studied scrub persisted and stayed as a remnant, and other areas regenerated to become just as dense. Once Neil settled onto the farm, a third of the farm was sprayed and felled into timber from 2001-2004. Again, the studied scrub has persisted and has grown back twice as thick, and Neil now periodically sprays across the farm, excluding the studied site, when and where possible.

Neil states that because there is so much scrub and forest around, there is significant wildlife. He said that the area has an abundance of deer, goats, feral cats, and hares, but no pigs. Neil notes that the land has a lot of birdlife and is vibrant with tūī. He also noted that there are also "pesky" birds such as peacocks and turkeys around. He has observed that this summer there has been less birdlife throughout the area. He suggests that this may be due to forestry harvests that had previously been home to possum populations across the landscape.

In the 11980sthe district received vast amounts of possum control to eradicate tuberculosis reactors. Neil remembered that as a child when he visited the farm, there used to be thousands of possums on the property. Before Neil took over the property, in 2000, the previous landowners shot 300 possums one night. Neil reports that the land now has no possums, and the bait stations are tended to regularly. However, due to pine harvesting mentioned earlier, Neil has observed that possums are coming back into the farmland, down the streams which are where the bait stations are getting them. Remarkably, 35 possums were killed in 2021, after years of absence. In response to the possum threat and to protect his QEII covenant from rising possum populations, Neil has also secured transects of 1080 bait running through the block.

Neil sees no future for pastoral production on his farm because of the ongoing battle of removing scrub with little return. He has revealed just how exhausting the battle of clearing the scrub has become and he is now seeking an alternative use for the land. He is wanting to invest much of the suits kanuka-dominated forest in carbon credits. With the land that is cleared or with unmatured scrub, he is considering planting pines for both harvest and credits. Another option for Neil is to sell the property as is, but he would rather see some of it stay in the native forest. Neil does appreciate and value the scrub in certain areas. He admits that it is very good for lambing and relies on it when a harsh season hits. He has noted that there is a higher percentage of lamb survival when there are storms and stock can shelter within the forest fragment. Neil appreciates the value of the area as he knows he is saving lambs when they're in it. He compares this to his neighbours who do not have as much shelter and lose a lot more lambs. The aesthetic and the value of having native scrub in desirable areas attract Neil. He dislikes clear farms and prefers to see texture and contrast across the farmland which natives provide. He enjoys seeing a thin scrub of forest across the farm as opposed to the dense and unmanageable forests or scrub.

Figure 2.6: Glengarry study sites (green = fenced and yellow = unfenced) found along the SH5. Coordinates, area, and the elevation of these sites and the surveyed plots within are found in the appendix. Scale 1:50000

2.2 Methods

Within this chapter section, I discuss the protocol used to determine the study sites. This section also describes the methods used to collect the biodiversity data following the T2BMF and the additional or adaption of data collection or methods.

2.2.1 Site Selection Process

A site selection process was set in place and followed thoroughly (Allen, 2007). The time taken to select the sites for my study was vital for the overall credibility of the collected data and results. Each of the studied sites was initially evaluated carefully, before beginning the plot surveys.

This research works with a total of five different sites situated across the lowland Hawke's Bay region, as detailed in the previous section of this Chapter. It was recommended by Manaaki Whenua – Landcare Research (Sarah Richardson and Peter Bellingham, pers. comm.) that this research study five paired sites. This recommendation was based on what could be achieved with the time and resource constraints, while still being suitable to test the effects of fencing on forest structure and diversity.

Care was taken to ensure, as much as possible, that each of the paired areas of each of the sites was environmentally identical. The aim was for the only difference between the selected paired sites to be the management of fencing: non-fenced versus conventionally fenced. This strict requirement between paired site areas was crucial as the research aimed to get a gauge of the effects of fencing rather than other environmental factors affecting biodiversity. Thus, each of the paired sites needed to be as environmentally identical as possible to separate this fencing factor and limiting variables.

Beyond the fencing management, there were additional strict parameters surrounding the site selection are outlined in the following rules which were followed in my study:

- Maximum of 30 m elevation difference between sites
- Site size must be a maximum of up to 30 ha and a minimum of 4 ha
- The area difference between paired sites must not exceed 20 ha
- Similar fragment history and establishment or maturity
- Similar coastal distance or exposure

From the HBRC perspective, sites were also selected from what the organisation classified as prioritised ecosystems. The prioritised ecosystems are ecosystems that are rare within the region or ecosystems where there have been little to no previous monitoring efforts. These prioritised ecosystems were therefore targeted as study sites so that the Council could get a better understanding of their ecological integrity.

The Council also wanted to get an understanding of practical resources necessary and any conceivable constraints, beyond the status of prioritised ecosystems biodiversity, or efforts required to monitor an arrange of ecosystems. HBRC sought to find the best means for efficiency and the reality of monitoring certain types of ecosystems, prioritised or not.

Following the site selection, collaboration, and consultation with HBRC advisor, Mark Mitchell, and QEII region Representative, Troy Duncan, identified a broad selection of potential study properties. Discussions between the organisations were based on their local knowledge of properties that contained significant or prioritised native forest fragments that had been fenced through their respective organisations. Potential site discussions considered the landowners' relationship with these organisations and the potential for additional opportunities for fencing fragments throughout the property.

With this collaboration, an assessment of suitability for the research was evaluated before any visits took place. An evaluation using the QGIS (QGIS Association, 2022) mapping system of a recommended 13 potential sites followed. These potential sites were assessed based on the site rules mentioned above and the observed site accessibility.

Another key factor that narrowed potential sites was the discussions with the landowners of the potential sites. In this discussion, both through email and phone calls, the research's objectives were disclosed, and site requirements, survey methods, and predicted results were discussed. Sites that were selected were effectively chosen based on how the landowner responded to the proposed research and their description of the native forest fragments at hand. When the landowner reacted positively to the prospect of a biodiversity survey, the site value increased as it was vital that these important stakeholders were prepared to allow fieldwork to take place and for results to be published.

After using spatial mapping evaluations and having discussions with landowners, the potential paired study site numbers had dropped to 8. Before any initial visits took place, a site selection sheet was made for an on-field assessment during the field and forest fragment tour with the landowner. The site selection assessment sheet contained the mentioned rules. Re-assessing the potential plots with these rules rather than basing a potential site on what advisors suggested or what was seen on the spatial mapping system, allowed for a realistic judgement. The ground-based assessments of the proposed sites-enabled comparability and representative visualisation of the potential sites. Also, these visits were important to plan the fieldwork accessibility for monitoring for both safety and efficiency.

After these final on-field assessments, a targeted 5 paired sites were confirmed, as detailed in the previous section of this thesis chapter. These 5 paired sites underwent further detailed assessments to ensure they contained the required characteristics for the trial.

There was a small pool of potential study site options due to the site selection rules and the scarcity of intact native forests throughout the region. Therefore, the sites that were chosen did vary in area. There is a much greater range of site areas in the fenced sites, and often the largest site of a pair was unavoidably the fenced site. This was since there is more of a motive to fence a larger area of forest as it is still more intact and ecologically viable. Also, for private landowners, it is assumed that it is easier to fence off larger forested areas rather than smaller ones for financial, accessibility, and productive measures.

2.2.2 Tier 2 Biodiversity Monitoring Framework & additional methods

My study implemented the Tier 2 Monitoring Framework (T2BMF), produced by Manaaki Whenua Landcare Research, and prepared for HBRC and other Regional Councils to adopt. Following the recommended T2BMF, the field protocols for this study were dissected and modified from existing DOC Tier 1 Inventory & Monitoring and Land Use and Carbon Analysis System (LUCAS) plots, all created by Manaaki Whenua Landcare Research. This new framework provides recommended ground-based methods of monitoring biodiversity elements to report changes in the ecological integrity of sites managed by regional councils.

My research studies 10 sites of native forests throughout farmland. Each of these sites has 3 permanent 20x20m plots. Plots within the sites will be examined using the Manaaki Whenua Landcare Research reconnaissance (RECCE) site and vegetation descriptions. Other factors within the plot data collection include non-vascular species collection, stem diameter and height measurements, sapling counts, understory subplots, soil samples, and photos.

Plots

This study acknowledges that the T2BMF, developed by Manaaki Whenua - Landcare Research, did not make explicit recommendations regarding the process for plotting sites. Decisions about how many plots should be established per site and the placement of these plots was needed and this study has therefore developed its protocol based on advice from Manaaki-Whenua Landcare Research members, Sarah Richardson, and Peter Bellingham.

Three plots within each of the studied forest fragment areas were considered an adequate replication, after consultation with members from Manaaki Whenua Landcare Research. Plot sampling intensity is later tested in Chapter 5 of this thesis.

It was decided that random permanent plot locations would be selected before fieldwork, in areas of intact forest canopy cover which were safe to sample, including areas facing edge effects. Plots were placed under the canopy with a requirement of trees present with a minimum of 5 metre (m) from the forest edge. Using the spatial mapping QGIS program (QGIS Association, 2022), each of the forest fragments was outlined and undesirable areas were excluded (cliffs, riverbeds, streams, grass areas). Plots were then located within a forest fragment area using 10 random sample points created by layering an incremented grid across each of the study areas. Using a randomised number generator, sample plot points were devised from this grid and then the coordinates were noted in the order.

The randomised sample point locations were then assessed based on suitability of implementation using evaluation from map contours, recommendations by the landowner who knew the land well, and discussions amongst the assisting field workers before visitation. The unsuitability of sample plot points is described as areas that were geographically or environmentally inappropriate to sample safely and have been avoided e.g., steep slopes, riverbeds, dense blackberry, and swamps. The 3 sampled plots were determined in an unbiased discussion between landowners and field staff to ensure that data collection has provided a correct stratified random model of the study area.

Plot layout procedure

Plots were located by inputting the assessed randomly generated coordinated plot points into a GPS. The plot layout procedure was a minimum 2-person job. The time that it took to lay the tapes out averaged 1 hour but could range from 30 minutes to 2 hours, depending on the number of fieldworkers, the terrain, and understory vegetation density.

Plot tapes on the surface of the ground defined the plot area and the gridding system of the subplots. The procedure for establishing and laying out new plots is tedious, but the layout must be precise for a true and comparable survey. The tapes lay as straight as possible and were pulled tight when on even ground. In situations whereby there was no flat or even ground, the tapes needed to follow the ground surface thus following the surface through a gully or over a ridge, and sometimes needed to be held down to follow this terrain. The tape layout sometimes needed to ignore small bumps or depressions, but when possible, tape followed under logs, or as close to the ground surface as possible. Trees on plot boundaries were included within the survey but only when the trunk is predominantly >50% rooted within the plot.

Once a sample plot point destination was identified, this then becomes the P station and was the basis of orientation (Figure 2.7). The corner P was marked using cruise tape or a waratah. To set up the plot, the bearing which runs along the predominant contour of the slope was identified. This was achieved by standing at corner P of the plot and determining the bearing by using a sighting compass to sight somebody standing 10–15 m away along the contour of the slope. The P–M boundary was then established along this contour by laying a 20-m tape along this bearing to form the lower boundary of the plot (P–M (Figure 2.7)). Then, taking 90° off the compass bearing of the P–M boundary to then determine the compass bearing of the P–A and M–D boundaries, two boundary tapes were laid out at right angles to the first. The open end along the A–D boundary is joined with a fourth boundary tape to form a square plot. When a newly established plot is located on flat terrain, the plot was established so that the M–P boundary lies in a north-south direction (i.e. corner M is north of corner P) (Department of Conservation, 2013a).

The plot was then subdivided into 16 5×5-m subplots by laying out six internal tapes at 5 m intervals (Figure 2.7). This grid system made by the tapes outlined the subplots. Subplots are ordered from A to P starting in the top left-hand corner (Figure 2.7) and labelled as such on all record sheets during the vegetation survey.

Throughout this process, fieldworkers continuously readjusted and checked the bearings and distancing of the tapes within the plot. In doing so, it mitigated confusion about the subplot placement and incorrect species counts. Additionally, the flagging tape was strategically tied around surrounding branches and trees often to ensure that position is identifiable during the monitoring period. These steps were vital to follow for efficiency and correct data collection.

Each of the four corners in every plot had a permanent waratah stake inserted into the ground for future locating. Near each corner of these waratahs, a tree outside the plot was selected on which to nail a strip of permolat o provides corner location information. Each permolat strip is labelled with the measured distance along the ground and the magnetic bearing from the base of the tree nearest to the corner peg and with the appropriate corner letter (e.g. 'Corner A 1.6 m @ 205°'). Adequate permolat marking near corners is invaluable when plots are to be remeasured, as waratahs may be lost over time. This information was recorded on the Plot Layout Record Sheet (Department of Conservation, 2013a).

The boundary tape plot dimension (i.e. tape distances), and internal tape distances were also recorded on the Plot Layout Record Sheet diagram. This is done because plot boundary and internal tape information will provide useful information when the plot is remeasured and resurveyed. At the end of the survey, GPS readings were waypoint marked and recorded for all four corners and the center of the plot.

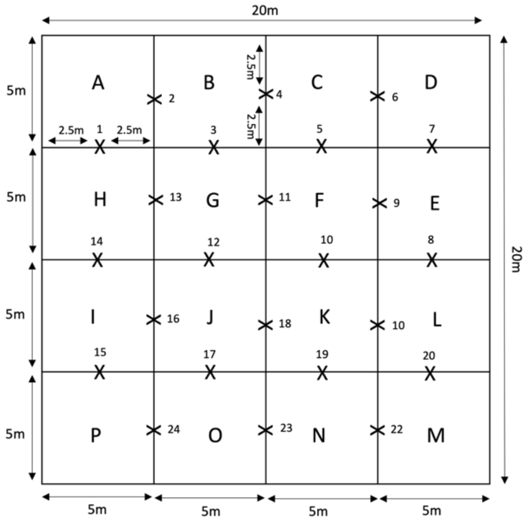


Figure 2.7: 20x20 m plot diagram with 5 m intercepts creating subplots. Each subplot is allocated an alphabet letter (A-P) and intercepts of 2.5 m of these subplots are numbered (1-24)

Field measurement within plots

The efficiency and the quality of collected data which a team can establish and measure each plot was determined to some extent by the allocation of people to tasks. The following division of labour worked well on the majority of plots but was adapted as necessary depending on the nature of the vegetation and the skills of the field staff.

On arrival at the plot, plot corners P was established, and boundary tapes were correctly laid out with the internal grided subplots. Firstly, a 5-minute bird count was conducted. Then the stem diameter and sapling count were performed, followed by the understory survey. Tasks on the vegetation description sheet were then debated amongst field workers and the RECCE site description sheet (Figure 8.1, Figure 8.2) was completed. Once the plot was packed up, the second 5-minute bird count was done allowing for more potential species detection to be recorded using a systematic and replicable method..

Vegetation Survey

Following the T2BMF, implementation protocols of the established National Vegetation Survey were conducted (Allen, 2007; Manaaki Whenua Landcare Research, 2021a), whereby for each of the study sites, three 20x20 m plots undertook this vegetation survey.

RECCE site description

A RECCE site description was completed for every plot providing essential covariate data for many analyses and comparisons between sites. Plot identification and descriptive data are recorded on this sheet. This sheet can be found on the National Vegetation Survey website of resources (Manaaki Whenua Landcare Research, 2022) (Appendix A Figure 8.1, Figure 8.2).

Plot axes were then measured and recorded. The bearing on all plots and the tight tape distance, tight tape slope, and horizontal distance was measured on the corner of the four plot axes (A–D, D–M, M–P, P–A (Figure 2.7). These measurements are important as it describes the position of the plot and also allow for precise remeasurements.

A diagram was illustrated of the orientation of the plot and markings of the position of the plot on the RECCE sheet was documented. The approach the fieldworkers took to get to the plot, concerning prominent features of the landscape or vegetation was also noted. Photographs of the plot were taken from each corner (A, D, M, and P (Figure 2.7)) looking inward towards the centre of the plot.

In the centre of the plot, records of the plot's physiography, drainage, slope, aspect, topographic index, and elevation were gathered. Site data collected within this sheet provides important information on the abiotic factors that influence the vegetation structure and composition of the plot. Site land-cover and land-use class was evaluated from the classifications provided.

Vegetation parameters were then estimated visually, and as such, they were relatively subjective. Here, there was much discussion between field workers about the plot's ground cover percentage. Over these observations across the plot, vegetation percentage, non-vascular vegetation, leaf litter, rock, and bare ground were discussed and calculated. The values did not have to sum up to 100%.

The average top height of the dominant vegetation was then estimated. The dominant vegetation is defined as all vegetation in the tallest tier (as recorded on the RECCE Vegetation Description Record Sheet (Appendix A Figure 8.2) with an overall cover of > 25% (i.e. overall cover class of ≥ 4). Height estimates were calibrated regularly using a builder's measuring tape and measuring pole.

Canopy cover percentage was then visually estimated. The total canopy cover of the plot above 1.35 m, to the nearest 10% was part of this percentage. Canopy cover is based upon the vertical projection of all vascular and non-vascular live or dead material (leaves, trunks, and branches) over the plot area. This is a measure of the cover of the canopy and reflects how much light is being blocked. The canopy cover scale was used (Appendix A) to help arrive at the estimate of the canopy percentage. Vegetation > 1.35 m above the ground was only included (Department of Conservation, 2013a).

RECCE vegetation description

On the RECCE Vegetation Description Record Sheet (Manaaki Whenua Landcare Research, 2022) (Appendix A) vegetation structure and composition were described in height tiers (strata) using cover classes (Appendix A) The RECCE vegetation description provides the most complete record of the species composition of the plot, including uncommon or epiphytic species that may not be captured in the stem diameter, sapling, or understorey data. This description also indicates the dominance of lianas in subcanopy and canopy tiers.

A complete observed vegetation description on the RECCE vegetation description was listed. At this point, a team of two was needed, one leading the measurements and one person recording and labelling any collected plant specimens. The data collection for the vegetation description was a whole team effort with communication from all field-party members essential to ensure that all species present on the plot are recorded. As the stems, saplings, and seedlings were measured first, the team was familiar with the vegetation on the plot.

The RECCE (Manaaki Whenua Landcare Research, 2022)(Appendix A) requires cover classes of species, which is the percentage of the coverage by a vertical projection split into different height tiers perimeter of an organism. The total coverage can exceed over 100 percent due to multiple layering.

Vegetation descriptions were achieved by recording all vascular plant species in each plot and allocating the percentage of live cover class to each species as a plot bounded by the plot edge. In each tier, the visualisation of plants of each species in corner of the plot, and then their estimation of what proportion of the plot they cover was discussed. The measurements used the required ordinal cover score for each species (<1%, 1–5%, 5–25%, 25–50%, 50–75%, 75–100%) within fixed height tiers (<0.3 m; 0.3–2 m; 2–5 m; 5–12 m; 12–25 m; 25 m+) and the presence, not cover, of epiphytes were recorded (Department of Conservation, 2013a).

Stem diameter & sapling data

Once the plot and subplots were set up, starting at subplot A and following subplots alphabetically, the stem diameter and sampling count survey was performed. A team of at least two people was needed to measure and record stem diameter and sapling data and recorded on the stem diameter & sampling data sheet (Manaaki Whenua Landcare Research, 2022)(Appendix A).

A tree was defined as an alive woody specimen with a diameter at breast height (DBH) greater than 2.5 cm and greater than 1.35 cm in height. These trees were permanently tagged using 200 mm nails, leaving a few centimetres for the tree to grow. A tree may have more than one tag as all stems ≥2.5 cm DBH at 1.35 cm in height above ground were recorded. It was noted when a specimen had multiple tags (multi-stemmed) and this was bracketed on the sheet and counted as an individual. Stem diameters were measured accurately, at the same place on the stem with each diameter measurement made 1 cm above the tree tag, which is placed 1.35 m along the tree stem.

On this sheet, each tagged tree was identified, and its diameter was measured to the nearest 0.01 cm against the subplot in which they occurred in. Tree ferns and palms were identified and tagged using long nails. The subplots in which they occur, and the measured height from ground to the point of the stem from which fronds or leaves emerge, to the nearest 0.1 m were measured. Liana stems were not tagged, nor were their diameters recorded (Department of Conservation, 2013a).

Measuring the diameter of the stem provides information on the size structure of the tree populations. Also, remeasurements of the plots of vegetation calculate tree populations recruitment, growth, and mortality rates.

While collecting stem data, and after a subplot tree abundance was recorded and measured, saplings were counted. This excluded lianas or tree ferns greater the 1.35 m but smaller than 2.4 cm in DBH. A sapling is defined as an alive woody specimen with a stem greater than 2.5 cm but smaller than 1.35 cm. In each of the 5x5 m subplots, saplings were counted, not tagged, into a tally by species in each subplot. This data was also recorded on the stem diameter and sapling sheet, below the stem measurements. To distinguish sapling records from stem diameter data, tallies were denoted using the symbols '< >' for each subplot.

The counts worked best when fieldworkers subdivided the subplot into smaller areas to count or mark counted specimens with chalk once they have been counted.

Seedling subplot data

Seedling subplot data collection was completed early in the plot measurement sequence so that the understorey was disturbed as little as possible. A team of two people was required to measure and record understorey subplot data and were careful not to tread inside seedling plots, and efforts were made to minimise damage to other understorey vegetation. Data is recorded on the Understorey Record Sheet. This survey records all species in each subplot by the maximum height of each individual, in height tiers (<0.15 m, 0.15–0.45 m, 0.46–0.75 m, 0.76–1.05 m, 1.06–1.35 m tall) (Manaaki Whenua Landcare Research, 2022)(Appendix A).

The established 24 seedling subplots were surveyed collecting counts of plant species frequency from the 24 subplots, each 0.75 m^2, located on the plot 20×20-m plot into the 16 5×5-m subplots (Figure 2.7). The data is collected across height tiers (0-1.35 m), allowing for an assessment of the structural change or description of the understory vegetation. When measuring, a piece of 0.49 cm string is used to define the area of the subplot. The sampled understory vegetation only measured vegetation that grows at the ground surface, within the specified circular area (Department of Conservation, 2013a).

For each plant rooted within the subplot, the measure of height is taken vertically from the base of the plant to the uppermost leaf foliage and measured as the plant naturally stands. All species measured in understorey subplots were rooted in the subplot area. The survey only recorded the presence of species <0.15 m, presence in height tiers for herbaceous species, and tallies in height tiers ≥0.15 m tall of different woody species (Department of Conservation, 2013a).

Samples, Collections, & Indices

Many samples and collections were taken from within plots for further investigation and assessment of environmental factors.

Unknown collections

Samples of unknown specimens were collected throughout the survey. Photographs or preferably standard collections material of specimens that could not be readily identified in the field were pressed for later determination. The collected samples were labelled and transcribed species onto the RECCE Vegetation Description Record Sheet and other sheets, as they were encountered.

Standard collections were taken whenever the identity of a species on a plot was unknown or uncertain. The collected specimen represented all occurrences of that species throughout the data (RECCE Vegetation Description Record Sheets, new Stem Diameter/Height/Sapling Record Sheets, and Understorey Record Sheet ((Manaaki Whenua Landcare Research, 2022)(Appendix A) Each standard collection had a unique provisional tag name that reflected the collection's notable feature or potential genus or species. On every sheet with a standard collection, the specimen's tag name was recorded with the symbol '©' to indicate that a specimen was collected.

Collections would ideally contain at least three full stems, with foliage and any seeds or fruit where possible. Collections were then stored in a plastic bag with a label containing information regarding the plot's identifier, subplot, and collection date. Once home, these samples were transferred into the newspaper to be pressed and dried out. The paper was changed every few days for the first 2 weeks. Samples needed to be linked to their collection information while they're drying out so there is no risk of material being mixed.

The gathering of photographs or preferably a collection material of plants which could not be readily determined in the field and pressed for later determination by myself or experts was vital. Additionally, the flowering or fruiting material (or otherwise fertile material, e.g. fern fronds bearing sori) should also be gathered wherever possible to facilitate identification.

Soil Samples

A soil sample from the top 100mm of mineral soil using a soil probe from every four quarters and aggregated into a single 500-g sample for each plot. The leaf litter layer and the horizon was removed for the soil sample. Fieldworkers used a trowel to scoop out and collect the top 10 cm of mineral soil. Used the trowel like a core to dig up a hole that was 10 cm deep with a diameter of 10 cm. Any non-soil objects > 2 cm (e.g. small rocks, roots, chunks of bark) were taken out of the core of the sample. The samples were collected into heavy-duty plastic bags with their plot identifier and sent away for processing. Samples were bagged and stored in a cool, dark, and dry environment until they were able to be processed. These samples were pooled for an analysis of the pH, total carbon, nitrogen, and phosphorus.

Mammal Monitoring

Due to time and resource constraints, mammal monitoring was restricted. The Tier 2 Monitoring Framework for forest ecosystems required a collection of data on the presence of fecal pellets of ungulates and lagomorphs and chew cards along four 150 m transects extending diagonally from each corner of each permanent 20m×20m vegetation plot. Given the studies' time frame and labour resources, and the difficulty of navigating through the weedy understorey of some sites, this was not achievable.

Instead, the presence of mammals (e.g. ungulate, lagomorphs, stock) was recorded based on a pellet count at the plot with a search time effort of 20 minutes. Using the following categories; None, Light (very little pellets, hard to find), medium (fair presence of pellets, not consistent), and heavy (noticeable presence pellets). Additionally, conspicuous browsing damage in all height tiers to plant species on the plot was observed. Using the following categories: Light (browse on 1 or 2 shoots only, on only a few of the plants of the species present), medium (browse on more than 1 or 2 shoots, but most plants of the species not browsed), and heavy (browse on most accessible shoots on most plants of the species).

As an additional monitoring method, foliage browsing cover through quantified crown dieback in palatable trees concerning whether or not they had visible mammal browsing. Pictures of the canopy were taken using a fish lens to determine each plot's browsing class (Sweetapple et al., 2016).

Bird Monitoring

Bird monitoring within the plots and across the site is a vital factor in assessing the integrity of an ecosystem. The T2BMF only suggested a five-minute bird count two times for each survey, one prior plot set up and another following the vegetation survey. This was considered potentially inadequate for rarer bird species therefore additional bird monitoring through autonomous sound recording was implemented.

Five Minute Bird Counts

After the plot has been set up and tapes have been laid, the first of the two five-minute bird counts (5MBC) at the centre of each plot was conducted to sample the bird community. At the end of the survey and before the plot is packed up, the second 5MBC was performed in the middle of the plot. This 5MBC was recorded on an iPhone and any unknown birdsong was noted with the time of the call for a follow-up identification. Observed and counted bird species were recorded though seen/heard tally per species (Manaaki Whenua Landcare Research, 2022)

5MBC constrained counts to ≤100 m of the recording point. These counts are performed in the centre of the plots rather than transects across the site as undertaken by DOC as it is assumed that the forests of interest to Regional Councils may often be small. The framework recommends that because bird abundance and conspicuousness vary with time of year, 5MBC should be undertaken at the same time of year to ensure comparability in repeated measures.

Autonomous Sound Recordings

In addition to the recommended 5MBC, this study also detected bird occupancy using autonomous sound recordings made with DOC AR4 recorders. Each site had an AR3 mounted to a tree branch, placed 1.8–2 m up the tree, positioned to minimise any disturbance by stock or blocking of sound, and secured using a cable tie. Each AR3 had 2 settings programmed, low frequency (for night time recordings) and forest frequency (for daytime recordings). However, the forest recording was mis-performed as the setting was unable to detect vocals of a high frequency (e.g. small high-pitched vocal birds). It is assumed that the forest setting needed to be updated or it had been altered. Further, recordings were truncated to fit the minimum time the shortest AR3 battery life lasted which was 12 days (both day and night protocols) for analysis.

These detection devices were left out for four weeks and were collected at the end of the field research or the end of the four weeks. The monitors were set for 23.55 hours, 11.55 hours of Low setting to measure diversity at night, and 11.55 hours in the Forest setting for daytime monitoring. The setting swap time was at 7 am and 7 pm and needed a 5-minute pause between the setting change.

Chapter 3

Lowland Hawke's Bay Vegetation Survey

This chapter will discuss the state of lowland forest vegetation in Hawke's Bay region. Data was collected using the recommended Tier 2 Biodiversity Monitoring Framework (T2BMF), but within this Chapter, only data from vegetation samples and observations and environmental variables are analysed. The surveyed vegetation dynamics are compared against different tiers, growth types, species biostatus, species richness, palatability index, succession, recruitment probabilities, and age. These elements are assessed by reviewing the three proposed hypotheses which assess the vegetation dynamics.

3.1 Introduction

Biological invasions have become normal across the globe as humans have colonized and spread throughout the world (Chapin et al., 2000). The introduction of agricultural practices has further aided in the naturalisation of many biological invasions. Invasions are detrimental as they lead to native ecosystem integrity collapses and failure to support and sustain native species (Aikio et al., 2012). Therefore, there is a global obligation for serious focus on conservation strategies for the protection and preservation of native biodiversity, specifically, vegetation, which is the foundation of greater biodiversity (Aikio et al., 2012; Purvis & Hector, 2000; Tierney et al., 2009a).

Naturalised species are exotic (non-native) species that have self-sustaining wild populations (Richardson et al., 2000). The success of the naturalisation of introduced species within a forest ecosystem is caused by the combination of rapid changes in land use and the landscape matrix (Hobbs et al., 2009). Novel ecosystems are blended that a permanent system has a substantial presence of exotic species as well as native species (McGlone et al., 2020; Sullivan et al., 2005)

Naturalised species have increased in NZ since European settlement and now today, nearly all types of NZ's native plant communities, including conservation land, have been invaded by some exotic plant species (Aikio et al., 2012; Hulme, 2020). Many of these naturalised plant species are known to have a substantial impact on native flora (D. M. Richardson et al., 2000), thereby threatening many forest ecosystems and processes (Wiser & De Cáceres, 2013).

Keeping stock and other exotic wildlife, such as ungulates, out of native forests is a conservation priority (Aviss & Roberts, 1994). Since the late 19th century it has been recognised that ungulates (e.g., deer, *Cervus* spp and feral goats, *Capra hircus*) have caused major changes to our native forest (Holloway, 1960; Husheer & Robertson, 2005; Lucas, n.d.). Within an native forest system, mammalian herbivory can threaten the forest integrity as they create unfavourable disturbances by compacting and eroding soil; browsing and trampling seedlings which reduce and even prevent natural regeneration; damaging tree bark and roots; introducing naturalised herbaceous species through depositing seeds through hooves and droppings; and can increasing soils fertility, of which is favoured by exotics rather than our natives (Dickie et al., 2010).

Gathering data regarding the different tiers or the placement of growth stages and forms describes the vertical forest structure in a way that gives insights into how a system is regenerating (Richardson et al., 2000). By monitoring a system's growth history, the variations in response to light environments can be assessed. Vegetation monitoring across height structures also identifies the competitive balance between and amongst species at a given light level exposure or growth tier. Greater light exposure percentage within a forest subcanopy has a positive relationship with growth and height tiers, which determines the forest composition and dynamics (Coomes & Allen, 2007b; Kunstler et al., 2009), therefore this percentage should be measured.

Cieraad et al. (2015) found that within the NZ landscape, areas of forest in flat, low, warm, and dry environments were the areas where the most forest has been felled and cleared for other land uses. Like much of NZ, lowland native forests within the Hawke's Bay region are scarce because of anthropogenic influences, biological invasion, species movement restrictions, and nutrient or synthetic contamination. Hashiba et al. (2014) explain how the ecological integrity of Hawke's Bays' native ecosystems is threatened due to the alteration of natural states.

It was also found by Allen et al. (1984), Coomes & Allen (2007a), and Wardle (1963), that there has been a reduction of subcanopy species that are part of the regeneration cycle of the canopy trees in altered ecosystems. Appropriate conservation management intervention of native forest fragments is critical for the stability and longevity of areas as aging populations of native canopy trees can die out without any natural replacements if processes like ungulate browsing prevent regeneration (Dodd et al., 2011).

3.1.1 Ecological Integrity & Vegetation

A forest ecosystem can be classified by forest type, as dictated by the dominant vegetation species composition and other occurring vegetation compositions, structures, and conditions over time; collectively known as the vegetation dynamics (Rose, 2012). Measuring or sampling vegetation trends and changes in populations are difficult to directly quantify in a single survey due to variable environmental controls, drivers, or factors for recruitment or establishment, growth, and the mortality rates of individuals (Peltzer et al., 2014), however, it is easy to quantify over time. While the combination of many environmental drivers and limiting factors makes it hard and complex to assess the integrity of a forest ecosystem, there are indicators for assessment that can withstand such limitations.

Several indicators of ecological integrity can be obtained through measuring vegetation dynamics, for example, the native dominance cover relative to the cover of exotic plants. Specifically, indicators of integrity within this study are determined using species occupancy within the plot (although much larger in the site area), and structural class measured in height tiers (Bellingham et al., 2021) The trajectory of vegetation is measurable through stem diameter measurements, seedling and sapling counts, is one of the most important indicators of forest condition (Innes, 1993). The measurement of the trajectory of vegetation can assess the overall ecosystem integrity as it determines a forest's likely future under the current conditions.

Evaluating the ecosystem health of a forest requires an assessment of the physical and biological state and foreseeable pressures on the ecosystem (McGlone et al., 2020). The ability to support ecosystem services and ecological functions relies on the population dynamics of vegetation and is therefore driven by both short-term and long-term biotic and abiotic processes that determine the ecosystem health (Coomes, Duncan, et al., 2003; Peltzer et al., 2014).

Assessment of a forest's ecological integrity is an essential process for describing the ecosystem's health, services, and functionality for private land owners, conservationists, and regional and national authorities. Measurements gathered for the assessment of ecological integrity include elements of the ecosystem vegetation dynamics. Through these assessments, conservationists and stakeholders can understand an ecosystem's integrity and how it is responding to factors of environmental or ecosystem change over time (Tierney et al., 2009b). As a result, proper intervention and wildlife management goals and practices can be put into place where appropriate.

3.1.2 Study Purpose

This research assesses the status of lowland vegetation diversity and dynamics within forest fragments across a farmland matrix in the Hawke's Bay region. In the Hawke's Bay region there has been minimal research on vegetation dynamics of lowland native forest fragments across farmland. There are currently few monitoring regimes for native forest remnants on private land, making it difficult for conservationists to know what exactly to manage or conserve.

The purpose of this research is to measure the success and effectiveness of the management programs. The initial process for conservation focus is to understand patterns and trends of biodiversity concerning underlying environmental variables, as a predication of assessing ecological conditions (Burns, 1995). Trends and patterns are monitored by measuring attributes of communities within an ecosystem, which relate to ecological processes, such as competition (Burns, 1995). These communities, and the processes that occur, are both directly and indirectly affected by environmental factors. Therefore, monitoring of both measurable variables of community dynamics such as induvial counts, in conjunction with environmental factors such as fencing management, gives wildlife managers a better appreciation of biodiversity status and ecological integrity.

My research seeks to assess the ecological changes caused by fencing forest fragments on private land. The success of fencing management will be determined through the monitoring of surveys using observations of biodiversity elements and environmental factors to ascertain the ecological integrity of forest fragments. Methods used to monitor vegetation are intended to determine the vegetation composition, structure, condition, and dynamics, and how these change over time in response to the management intervention of fencing. In doing so, the rate of the effectiveness of fencing management over the number of years the fence has been established can be determined through the analysis of species richness, palatability, and individual counts throughout height tiers. These measurable vegetation dynamics enable a better understanding of the forest ecosystem's integrity.

This research implements and tests T2BMF (Chapter two), examining the framework's performance by testing its ability to detect the effects of fencing on forest fragments' ecological integrity. The research comprises additional biodiversity monitoring at a vegetation dynamic scale and records additional environmental factors (other than management) so that an understanding of other elements affecting vegetation dynamics within the studied sites can be obtained.

This research tested the following four hypothesises:

1. *The composition of plant species will differ between unfenced and fenced sites,*
2. *There will be lower natives species richness in the understory of the unfenced sites compared with fenced sites,*
3. *There will be a higher proportion of palatable species, especially in lower height tiers, in fenced sites than in unfenced sites,*
4. *Site management will have significant relationships between the number of total individual counts across the different height tiers, with fenced sites having positive and unfenced sites having negative.*

3.2 Method

Three 20 m x 20 m plots were surveyed at 10 sites, one pair of fenced and unfenced sites in each of five different areas throughout the lowland Hawke's Bay region, for a total of 30 plots. Each pair of sites were chosen to be comparable to each other, with fencing management the only markedly different variable. The vegetation dynamics sampled were: species, adult tree stem diameter at breast height, sapling count, the tiers of which seedlings accumulate within subplot points, canopy coverage and tree height, and the overall coverage percentage of vegetation species presence from each of the 30 20 x 20 plots.

This research focuses on the composition of native and naturalised vegetation within plots between fenced and unfenced sites. The studied sites were dominated by broadleaf or podocarp trees or kanuka. The vertical stratification was most evident and distinguishable when separated and measured in different height tiers (For seedling surveys: 0.15 m, 0.15–0.45 m, 0.46–0.75 m, 0.76–1.05 m, 1.06–1.35 m, and for the RECCE survey: 0.3–2 m; 2–5 m; 5–12 m; 12–25 m; 25 m+). Seedlings were separated into different height tiers and growth forms within the results.

In addition, environmental factors were measured to get a better understanding of plot situations. These environmental factors included a collection of soil samples, weather conditions, slope and aspect position of the plot, and a browse index. For more information on the procedures of these measurements (Chapter 2).

Using the collected environmental and vegetation sample data, we merged the ungulate palatability data index (Forsyth et al., 2002) to get an indication of what plant species present within sites were susceptible to ungulate browsing. This index was devised from a collection of vegetation surveys of forests around the country that experienced different intensities of ungulates presence (Forsyth et al., 2002). IN this study, ungulate presence was determined through observations of droppings and browsing damage and were recorded using the categories of high, medium, low, or presence/absence. Search time effort was 30 minutes per plot after vegetation sampling.

The monitoring methods for this study were recommended by Maanaki Whenua Landcare Research and the National Vegetation Survey Databank (Manaaki Whenua Landcare Research, 2021a). Measurements were compared with environmental factors or other biotic variables considered statistically important. These variables were evaluated together based on the naturalness to determine the ecological integrity of a forest. The systematic vegetation surveys followed the methodology of site locations and plot points (Chapter 2) allowing for repetition.

3.2.1 Statistical & Data Analysis

I undertook a variety of analyses to describe the vegetation dynamics within the community, assess the environmental consequences of fencing, and reveal the effects of other underlying environmental gradients.

All of the observed species were pooled and separated based on biostatus, growth form, palatability, height tier, and primarily fencing management – fenced versus unfenced. The vegetative species found within the plot were assigned with bio status and life form based on the National Vegetation Survey Databank (New Zealand National Vegetation Survey Databank, 2022). Some species were not listed in the document or were unable to be identified and therefore classifications of bio status and life forms had to be assumed.

All analyses were performed in R version 4.1.0 (R Core Team, 2013). This research uses a variety of R packages depending on the analysis required, which are described when applied.

Hypothesis 1: The composition of plant species will differ between unfenced and fenced sites.
The data sets used for this analysis were first merged from data from the National Vegetation Survey current names document (refer to New Zealand National Vegetation Survey Databank, 2022) with our surveyed vegetation data. This combined vegetation dataset was then introduced with the environmental variables of each of the studied plots. Each of the plots was grouped into sites (fenced versa unfenced) and areas (paired fenced and unfenced sites). For adult trees with multiple stems, the DBH of the largest stem was used.

The vegan package (Oksanen et al., 2020) was used to analyse the vegetation composition using nonmetric multidimensional scaling (NMDS) and Adonis.

To accurately describe the pattern of species composition of the surveyed vegetation community with environmental factors, an indirect gradient analysis using NMDS was undertaken. NMDS collapses surveyed information from multiple dimensions into just a few so that they can be visualized and interpreted (Kent & Coker, 2012). It is an unconstrained method for aligning site dissimilarities, which makes it ideal for non-normal data (McCune et al., 2002). Sites are arranged in a sample space, known as an ordination (Kent & Coker, 2012). NMDS is considered to be the most competent ordination method for detecting vegetation gradients (McCune et al., 2002).

Samples within the ordinates are placed relative to the similarity of their species composition. This is done based on a rank of dissimilarity distances between the samples (McCune et al., 2002) so that the distances in ordination spaces illustrated the similarities between sample plots. I used the envfit function within the vegan package (Oksanen et al., 2020), to show how the environmental variables aligned with the NMDS ordinations. These graphs help to visualise the clusters of environmental variabilities between plot similarities. We used a simper to determine the taxa which are most driving the differences among fenced versus unfenced sites.

Permutational Multivariate Analysis of Variance, using the adonis function in the vegan package, was used to assess the significance of the effect of fencing, and the other measured environmental variables, on vegetation composition. For this and other PERMANOVA (below), a permutation of 9999 is used to calculate significant values of environmental variables.

I used Principal Components Analysis (PCA) using the prcomp function to reduce the ground cover and soil composition variables to simplify my statistical models. The ground cover PCA was on percentage cover of vascular vegetation, non-vascular vegetation, litter, rock, and bare ground. The first three PC axes explained 80.1% of the variation in cover and these PC variables were used in analyses (Table 3.1). Cover PC1 was positively correlated (0.60) with bare ground cover and negatively correlated (-0.63) with vascular vegetation cover. Cover PC2 was negatively correlated (-0.89) with litter cover. Cover PC3 was positively correlated (0.80) with rock and negatively correlated (-0.58) with non-vascular plant cover. The soil PCA was on soil pH, carbon, nitrogen, phosphorus, and volume:weight. The first two PC axes explained 62% of the and were included in the analyses (Table 3.2). Soil PC1 was negatively correlated with carbon (-0.54), nitrogen (-0.56), and phosphorus (-0.39) and is regarded as an (inverse) index of soil fertility. Soil PC2 was positively correlated (0.71) with pH and positively correlated (0.53) with phosphorus. Soil PC3, which explained an additional 20% of the variation, was negatively correlated (-0.63) with pH and positively correlated (0.54) with volume:weight. Soil PC3 was not clearly connected with any of our expectations for the factors driving species composition so we chose to only include the first two soil PC variables in our models.

Table 3.1: Ground cover of PCA

> pca_cover Standard deviations (1, .., p=5) [1] 1.380 1.101 1 0.88 0.322 Rotation (n x k) = (5 x 5):					
	PC1	PC2	PC3	PC4	PC5
Vegetation	-0.634	0.380	0.025	-0.123	0.661
Non-vascular	-0.383	0.124	-0.583	-0.067	0.191
Litter	-0.072	-0.891	-0.067	-0.021	0.441
Bare ground	0.603	0.212	0.006	0.531	0.555
Rock	0.285	0.001	0.808	-0.491	0.149
> summary (pca_cover) Importance of components:					
Standard Deviation	1.38	1.101	1	0.88	0.322
Proportion of Variance	0.381	0.242	0.2	0.154	0.02
Cumulative Proportion	0.381	0.624	0.824	0.979	1

Table 3.2: Soil PCA

> pca_soils Standard deviations (1, .., p=5) [1] 1.681 1.186 0.753 0.392 0.207 Rotation (n x k) = (5 x 5):					
	PC1	PC2	PC3	PC4	PC5
pH	-0.13	0.711	-0.631	0.276	-0.623
Carbon	-0.54	-0.293	-0.047	0.48	-0.691
Nitrogen	-0.565	0.026	0.324	0.309	0.691
Phosphorus	-0.393	0.536	0.447	0.498	-0.329
Volume.Weight	0.464	0.344	0.541	0.59	-0.151
> summary (pca_cover) Importance of components:					
Standard Deviation	1.68	1.187	0.753	00.392	0.204
Proportion of Variance	0.565	0.281	0.113	0.03	0.008
Cumulative Proportion	0.565	0.624	0.847	0.991	1

Hypothesis 2: There will be lower natives species richness in the understory of the unfenced sites compared with fenced sites.

For the analysis, my surveyed vegetation data were merged with the plant traits data from the National Vegetation Survey template spreadsheet (refer to New Zealand National Vegetation Survey Databank, 2022). This allowed the data to be separated by both height tiers and growth forms. Mixed models were then run using the package lme4 v1.1-26 (Bates et al., 2015) and the Anova function in the car package (Fox, John & Weisberg, 2011) within RStudio (R Core Team, 2013).

Plant species richness was calculated separately for native and naturalised species in each of the following growth forms and sizes: big adults (>10 cm diameter at breast height (DBH), small adults (<10cm stem DBH), woody saplings (>135 cm in height and <2.5 cm DBH), woody seedlings (16–135 cm), small woody seedlings (≤15 cm), and herbaceous ferns, forbs, and graminoids.

For each group, a mixed model was run with species richness as the response variable, the explanatory variables longitude, latitude, altitude, aspect, and fencing (either yes/no or the estimate of the years fenced), and location as a random effect.

Hypothesis 3: There will be a higher proportion of palatable species, especially in lower height tiers, in fenced sites than in unfenced sites.

This analysis focused on the effects of fencing on a subset of species with available palatability information. All species surveyed were matched against the ungulate palatability data index of Forsyth et al. (2002), and any species not in the index were removed (87 out of 110). Within the index, the palatability was ranked from preferred, not selected, and avoided. Palatability was simplified to palatable (preferred species) and not palatable species (species that were not selected or avoided by ungulates). Woody plants were separated into the same size/height categories as used in hypothesis 2: small seedlings, seedlings, saplings, adults, and big adults.

A binomial generalised linear model was used to analyse how the proportion of species in a given size/height category that is palatable is affected by mean plot canopy top height, longitude, latitude, fencing, and ungulate sign (presence/absence). The R function drop1 was used to assess the significance of each variable after all other variables were accounted for.

I created a table of the occurrence of palatable species and unpalatable species for fenced and unfenced sites across all height tiers. The seedling abundance of the palatable species richness will be portioned by how much presence out of the 24 subplots, rather than the adults and saplings which using individual counts. The total mean of these counts of avoided proportions were divided with the preferred proportion for each height tier to get the total proportion of palatability across height categories.

Hypothesis 4: There will be a significant relationship between species succession, recruitment, height, and stage between fenced and unfenced sites and other environmental factors.
To understand if size structure correlated with the years fenced of woody tree species across all sites, I removed all other growth forms from the dataset.

The height tier of the plots was split into the different sampled tiers, as described in hypothesis 2. However, the T2BMF method only required the smallest seedlings (≤15 cm) to be recorded as presence or absence, rather than a numerical count, so their abundance could not be compared with higher tiers. I, therefore, removed this smallest tier from the analysis and just compared the higher tiers. The larger seedlings were separated into two, mid-sized seedlings (>15 cm and ≤45 cm) and seedlings (>45 cm and ≤135 cm). These seedling height tiers were only sampled for each of the 24 subplot intercepts, each within a 0.49 cm radius, instead of throughout the entire plot like the counts of the higher tiers. Therefore, the full plot seedling count for each species was estimated by multiplying the seedling counts by 22.1, the ratio of the plot area to the area sampled.

Mixed models were created with the lme4 package with the log of total count as the response variable, the explanatory variables altitude, longitude, latitude, height tier, fenced (yes/no), the interaction of height tier and fences, and mean canopy top height, with the site as the random effect. This more complicated model was compared with a simpler model with just height tier, fenced, and their interaction, as the only explanatory variables.

3.3 Results

The results from this study in the lowland Hawke’s Bay region of native forest fragments and the vegetation dynamics within, both support and disagree with my four proposed hypotheses. These results are only based on data collected from environmental factors and vegetation survey results.

3.3.1 Hypothesis 1: The composition of species will differ between unfenced and fenced sites

Native and naturalised species varied in growth forms (Table 3.3).

Big adults

The NDMS ordination for big adult trees (stress 0.078), with envfit (Figure. 3.1A), shows that both the latitude and longitude positively affected plot species composition. The canopy height and the number of years that the forest had been fenced had similar positive effects and with altitude, low leaf litter levels, and bare ground having negative effects.

The NDMS ordination with hulls separating plots at fenced and unfenced sites (Figure 3.1B), shows that the big adult tree composition within fenced and unfenced plots was very similar. The NDMS illustrates that there was much overlapping of species composition, regardless of the fencing management and other environmental variables.

The variables that significantly explained the variation in big adult composition among plots were latitude, longitude, altitude, cover PC1 and PC2, and lagomorph presence/absence (PERMANOVA in Table 3.4A).

Small adults

The NDMS envfit graph (Figure 3.1C), shows that the latitude of the site plot placement has the largest positive effect on the species composition and plot similarities across both fenced and unfenced sites. The canopy height of the observed native trees and the number of years in which the forest had been fenced both shared similar positive effects on native adult species richness. The ordination shows a relationship between the presence of lagomorphs and fencing management. Bare ground and the altitude of the plots shared positive effect on the species composition. Low litter and longitude had an negative effect (Figure 3.1C).

The NDMS ordination graph (Figure 3.1D*)* illustrated that the fenced site species composition among the plots was similar to the unfenced sites. However, the unfenced sites had more of a broader composition concerning the environmental variables compared to the fenced. This shows that there was a greater difference in the composition of plots between unfenced and fenced sites.

The variables that significantly explained the variation in small adults composition among plots were latitude, longitude, altitude, top height mean, cover PC1 and PC2, and lagomorph presence/absence (PERMANOVA in Table 3.4B).

Saplings

The NDMS envfit graph of the sapling species composition amongst plots (Figure 3.1E), displays significant similarities for the longitude and latitude relationship with the placements of the surveyed plots. In addition, low leaf litter, with a strong relationship with the presence of observed lagomorphs had a significant negative effect on sapling composition. The ordination graph (Figure 3.1E) indicated how plots of greater fenced years with a high canopy, and with no observed lagomorph presence had similar positive effects in adult species composition.

Between the fenced and unfenced sites, there was a significant difference in the sampling composition (Figure 3.1F). It was found that in unfenced sites, sapling composition differed greatly amongst the plots, and in comparison to the fenced sites, sapling composition had similarities amongst plots.

The variables that significantly explained the variation in saplings composition among plots were latitude, longitude, altitude, top height mean, fenced years, cover PC1 and PC2, and lagomorph presence/absence (PERMANOVA in Table 3.4C).

Seedlings >135 cm

Seedlings were separated into different growth forms and height within the results.

Woody seedlings >15 cm – ≤135 cm

For woody seedlings, the envfit NDMS diagram (Figure 3.1G) illustrates how altitude has the most significant positive effect on woody seedlings' composition and site similarities. The age of the fence and the height of the canopy shared similar significant positive effects, alongside altitude. Additionally, bare ground and longitude both share similarities of significant negative effects of composition between plots. It was also found that plots with a high percentage of bare ground and with moderate lagomorph presence tend to have similar vegetation species composition. Low leaf litter variables also negatively impacted woody seedlings.

The NDMS ordination (Figure 3.1H), illustrates how the plots for all the woody seedling composition overlaps, but have both positive and negative effects on composition in different directions, suggesting that there are many strong environmental variables affecting the composition between fenced and unfenced sites. The variables that significantly explained the variation in seedlings composition among plots were latitude, longitude, altitude, top height mean, cover PC1 and PC2, and lagomorph presence/absence (PERMANOVA in Table 3.4D).

Woody seedlings ≤15 cm

The NDMS envfit diagram (Figure 3.1I), shows how fenced years had the largest positive effect on small woody seedlings for fenced plots. Additionally, longitude, latitude, and low litter all shared similarities of significant positive effect, having the greatest effect on unfenced plots. The variables that had a positive effect on the variation in small seedlings composition among plots were latitude, longitude, and top height mean (PERMANOVA in Table 3.4E).

Herbaceous species (<135 cm)

The composition of herbaceous species in plots was influenced by a variety of factors (Figure 3.1J). The largest but opposing in impacts were altitude and longitude. The ordination indicated how longitude and bare ground elements shared similarities of negative effect. The low litter had a positive effect similar to latitude but was found to be only an affecting factor in unfenced sites. The fenced years and the canopy height of plots shared similar negative effects on the composition of herbaceous vegetation for fenced sites (Figure 3.1J).

The environmental variables appear to have effected native and naturalised species composition within the fenced plots compared with the unfenced plots. The unfenced plots ordination is very compact compared with the fenced ordination. (Figure 3.1J).There were no significant environmental variables influencing herbaceous species (Table 3.4F)

The composition of naturalised herbaceous species in the unfenced sites had the largest positive effect with low litter levels, but similarities with altitude (Figure 3.1K*).* Fenced years and the categorised vegetation height tiers also had an negative effect within fenced plots. Bare ground and longitude shared similar, but less significant negative effects for both fenced and unfenced sites.

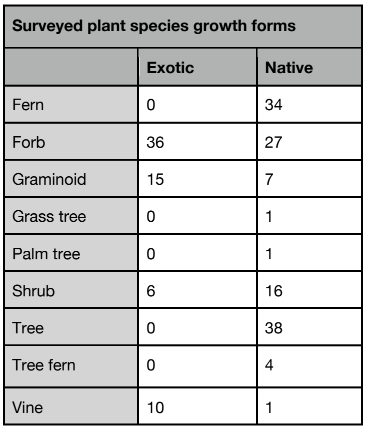
The composition of native herbaceous species was similar among plots. It was found that there was a negative relationship with altitude mainly in the unfenced sites. There were similar negative effects of the composition between the fenced age and height and also similar significant negative effects between the bare ground and longitude, but only for fenced sites. The composition of naturalised herbaceous species in the unfenced sites had the greatest positive effect with low litter levels and altitude (Figure 3.1L). In the fenced plots, fenced years positively influenced plant height. However, species composition showed negative effect from bare ground and longitude in both fenced and unfenced sites (Figure 3.1L).

Overall, it was found that the composition of fenced plots had a broader range of native herbaceous species compared to unfenced plots. Unfenced plots did however fit into the composition range of the fenced sites, reflecting a small part of the composition that sat within the fenced plots (Figure 3.1M)

The composition of naturalised herbaceous species had a negative effect with altitude and low litter (Figure 3.1N). There was a positive effect with fenced years, bare ground, longitude, latitude, and height. The variability of the composition of herbaceous species in unfenced sites was fairly broad (Figure 3.1O) but orientated in coverage in an opposite direction to fenced plots, indicating that fenced plots and unfenced plots were largely different in herbaceous species composition. However, there was much similarity of native herbaceous composition amongst the plots, as there is overlap between fenced and unfenced sites; however, there is a greater range and variability of environmental factors affecting the composition of the fenced sites (Figure 3.1O).

The variables that significantly explained the variation in seedlings composition among plots were latitude, longitude, altitude, top height mean, fenced years, cover PC1 and PC2 (PERMANOVA in Table 3.4G). Naturalised herbaceous species shared the same influence by significant environmental variables, but they also were influenced by lagomorph presence/absence (PERMANOVA in Table 3.4H).

Table 3.3: Number of exotic and native species growth forms derived from the NVS species growth form list (New Zealand National Vegetation Survey Databank, 2022). Species are separated as either native or exotic and further categorised based on growth form, according to growth habit or height.



Surveyed plant species growth forms		
	Exotic	Native
Fern	0	34
Forb	36	27
Graminoid	15	7
Grass tree	0	1
Palm tree	0	1
Shrub	6	16
Tree	0	38
Tree fern	0	4
Vine	10	1

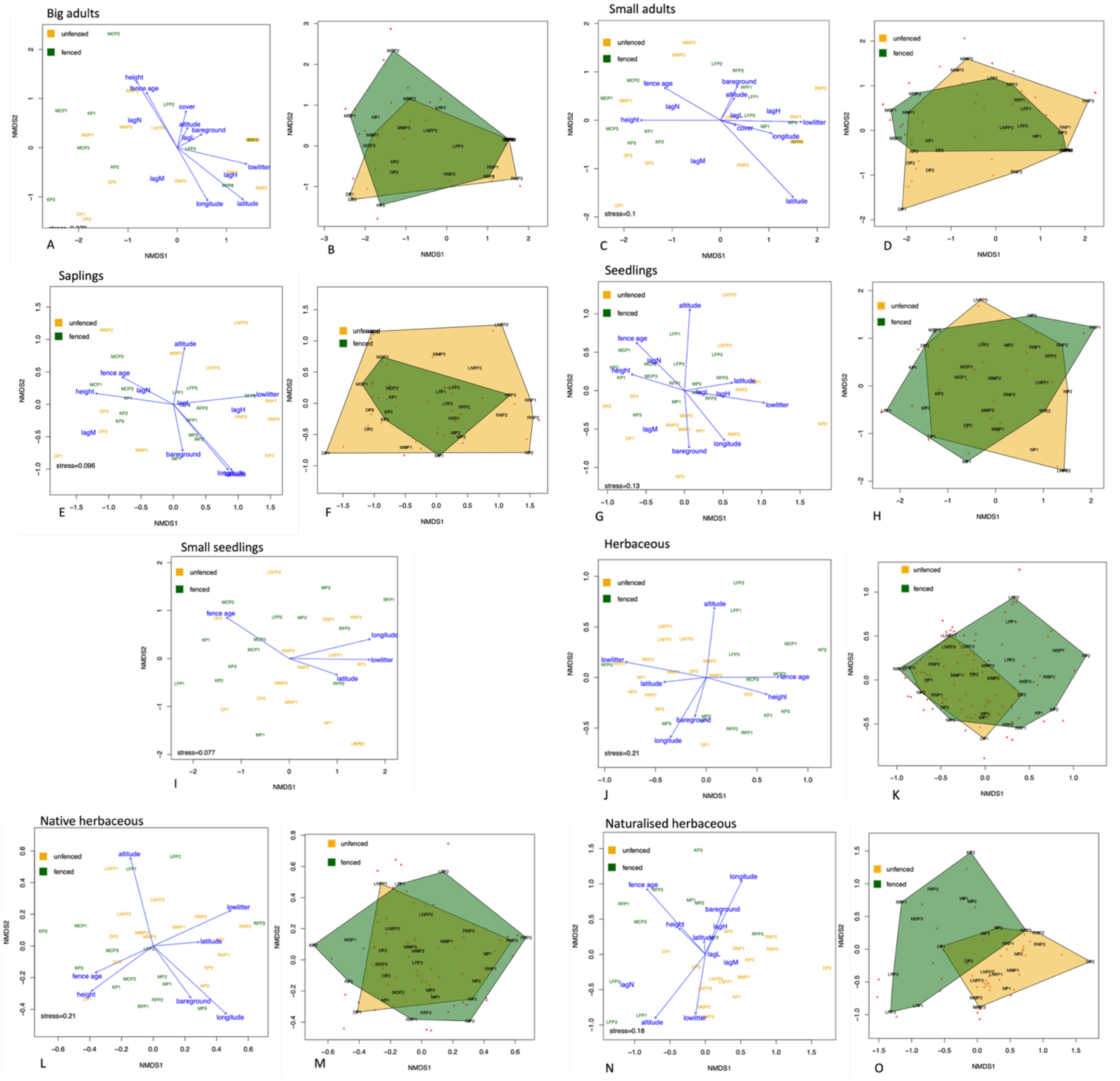
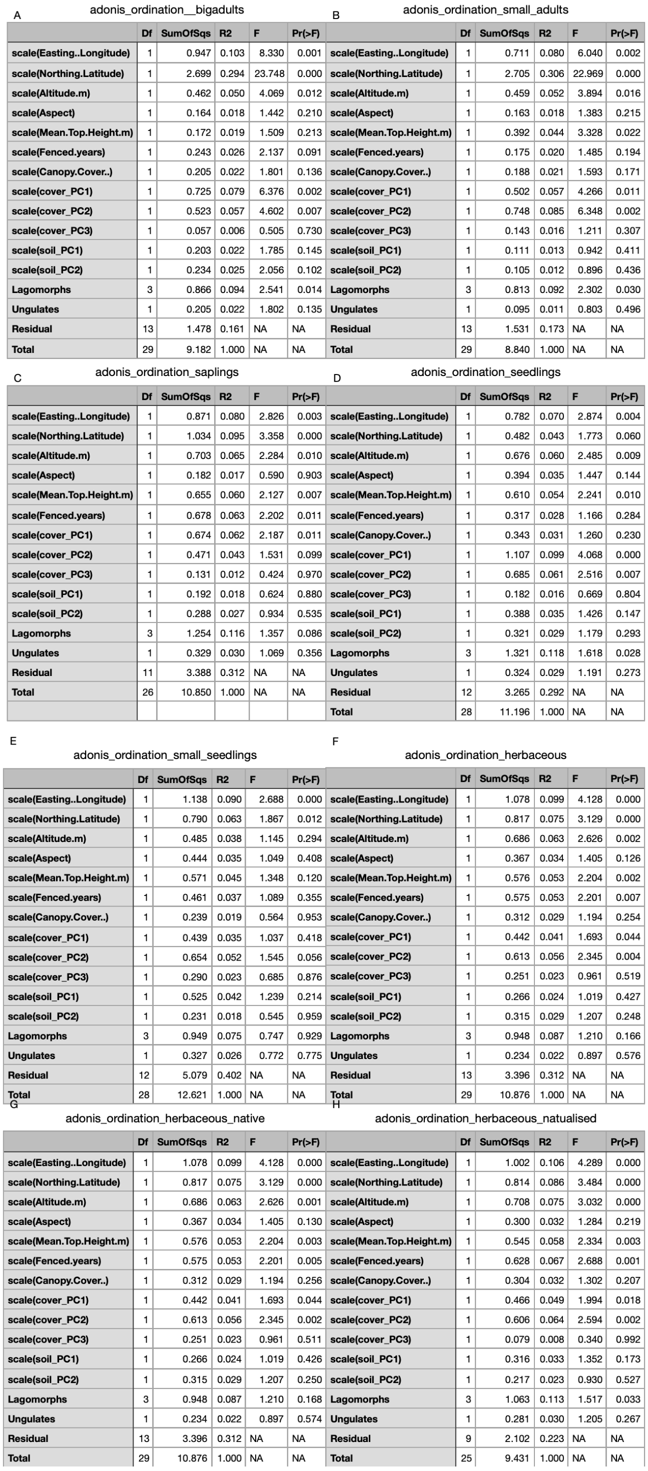


Figure 3.1: (A, C, E, G, I, J, L, N) NMDS ordinations of fenced and unfenced site plots displaying the environmental gradients that affected the big adult tree species composition. Plot IDs in green are fenced sites and plot IDs in yellow are unfenced sites. Figure 1: (B, D, F, H, K, M, O) NMDS ordination graph displaying the overall similarity of plot compositions between fenced and unfenced sites of adult tree composition. Fenced sites are in green and unfenced sites are in yellow.

Table 3.4: : PERMANOVA values for all height tiers and growth forms: big adults (A), small adults (B), saplings (C), seedlings (D), small seedlings (E), herbaceous (F), native herbaceous (G), and naturalised herbaceous (H). The significance of environmental values based on species composition is represented by the P-value.



A

adonis_ordination__bigadults

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.947	0.103	8.330	0.001
scale(Northing.Latitude)	1	2.699	0.294	23.748	0.000
scale(Altitude.m)	1	0.462	0.050	4.069	0.012
scale(Aspect)	1	0.164	0.018	1.442	0.210
scale(Mean.Top.Height.m)	1	0.172	0.019	1.509	0.213
scale(Fenced.years)	1	0.243	0.026	2.137	0.091
scale(Canopy.Cover..)	1	0.205	0.022	1.801	0.136
scale(cover_PC1)	1	0.725	0.079	6.376	0.002
scale(cover_PC2)	1	0.523	0.057	4.602	0.007
scale(cover_PC3)	1	0.057	0.006	0.505	0.730
scale(soil_PC1)	1	0.203	0.022	1.785	0.145
scale(soil_PC2)	1	0.234	0.025	2.056	0.102
Lagomorphs	3	0.866	0.094	2.541	0.014
Ungulates	1	0.205	0.022	1.802	0.135
Residual	13	1.478	0.161	NA	NA
Total	29	9.182	1.000	NA	NA

B

adonis_ordination_small_adults

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.711	0.080	6.040	0.002
scale(Northing.Latitude)	1	2.705	0.306	22.969	0.000
scale(Altitude.m)	1	0.459	0.052	3.894	0.016
scale(Aspect)	1	0.163	0.018	1.383	0.215
scale(Mean.Top.Height.m)	1	0.392	0.044	3.328	0.022
scale(Fenced.years)	1	0.175	0.020	1.485	0.194
scale(Canopy.Cover..)	1	0.188	0.021	1.593	0.171
scale(cover_PC1)	1	0.502	0.057	4.266	0.011
scale(cover_PC2)	1	0.748	0.085	6.348	0.002
scale(cover_PC3)	1	0.143	0.016	1.211	0.307
scale(soil_PC1)	1	0.111	0.013	0.942	0.411
scale(soil_PC2)	1	0.105	0.012	0.896	0.436
Lagomorphs	3	0.813	0.092	2.302	0.030
Ungulates	1	0.095	0.011	0.803	0.496
Residual	13	1.531	0.173	NA	NA
Total	29	8.840	1.000	NA	NA

C

adonis_ordination_saplings

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.871	0.080	2.826	0.003
scale(Northing.Latitude)	1	1.034	0.095	3.358	0.000
scale(Altitude.m)	1	0.703	0.065	2.284	0.010
scale(Aspect)	1	0.182	0.017	0.590	0.903
scale(Mean.Top.Height.m)	1	0.655	0.060	2.127	0.007
scale(Fenced.years)	1	0.678	0.063	2.202	0.011
scale(cover_PC1)	1	0.674	0.062	2.187	0.011
scale(cover_PC2)	1	0.471	0.043	1.531	0.099
scale(cover_PC3)	1	0.131	0.012	0.424	0.970
scale(soil_PC1)	1	0.192	0.018	0.624	0.880
scale(soil_PC2)	1	0.288	0.027	0.934	0.535
Lagomorphs	3	1.254	0.116	1.357	0.086
Ungulates	1	0.329	0.030	1.069	0.356
Residual	11	3.388	0.312	NA	NA
Total	26	10.850	1.000	NA	NA

D

adonis_ordination_seedlings

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.782	0.070	2.874	0.004
scale(Northing.Latitude)	1	0.482	0.043	1.773	0.060
scale(Altitude.m)	1	0.676	0.060	2.485	0.009
scale(Aspect)	1	0.394	0.035	1.447	0.144
scale(Mean.Top.Height.m)	1	0.610	0.054	2.241	0.010
scale(Fenced.years)	1	0.317	0.028	1.166	0.284
scale(Canopy.Cover..)	1	0.343	0.031	1.260	0.230
scale(cover_PC1)	1	1.107	0.099	4.068	0.000
scale(cover_PC2)	1	0.685	0.061	2.516	0.007
scale(cover_PC3)	1	0.182	0.016	0.669	0.804
scale(soil_PC1)	1	0.388	0.035	1.426	0.147
scale(soil_PC2)	1	0.321	0.029	1.179	0.293
Lagomorphs	3	1.321	0.118	1.618	0.028
Ungulates	1	0.324	0.029	1.191	0.273
Residual	12	3.265	0.292	NA	NA
Total	28	11.196	1.000	NA	NA

E

adonis_ordination_small_seedlings

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	1.138	0.090	2.688	0.000
scale(Northing.Latitude)	1	0.790	0.063	1.867	0.012
scale(Altitude.m)	1	0.485	0.038	1.145	0.294
scale(Aspect)	1	0.444	0.035	1.049	0.408
scale(Mean.Top.Height.m)	1	0.571	0.045	1.348	0.120
scale(Fenced.years)	1	0.461	0.037	1.089	0.355
scale(Canopy.Cover..)	1	0.239	0.019	0.564	0.953
scale(cover_PC1)	1	0.439	0.035	1.037	0.418
scale(cover_PC2)	1	0.654	0.052	1.545	0.056
scale(cover_PC3)	1	0.290	0.023	0.685	0.876
scale(soil_PC1)	1	0.525	0.042	1.239	0.214
scale(soil_PC2)	1	0.231	0.018	0.545	0.959
Lagomorphs	3	0.949	0.075	0.747	0.929
Ungulates	1	0.327	0.026	0.772	0.775
Residual	12	5.079	0.402	NA	NA
Total	28	12.621	1.000	NA	NA

F

adonis_ordination_herbaceous

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	1.078	0.099	4.128	0.000
scale(Northing.Latitude)	1	0.817	0.075	3.129	0.000
scale(Altitude.m)	1	0.686	0.063	2.626	0.002
scale(Aspect)	1	0.367	0.034	1.405	0.126
scale(Mean.Top.Height.m)	1	0.576	0.053	2.204	0.002
scale(Fenced.years)	1	0.575	0.053	2.201	0.007
scale(Canopy.Cover..)	1	0.312	0.029	1.194	0.254
scale(cover_PC1)	1	0.442	0.041	1.693	0.044
scale(cover_PC2)	1	0.613	0.056	2.345	0.004
scale(cover_PC3)	1	0.251	0.023	0.961	0.519
scale(soil_PC1)	1	0.266	0.024	1.019	0.427
scale(soil_PC2)	1	0.315	0.029	1.207	0.248
Lagomorphs	3	0.948	0.087	1.210	0.166
Ungulates	1	0.234	0.022	0.897	0.576
Residual	13	3.396	0.312	NA	NA
Total	29	10.876	1.000	NA	NA

G

adonis_ordination_herbaceous_native

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	1.078	0.099	4.128	0.000
scale(Northing.Latitude)	1	0.817	0.075	3.129	0.000
scale(Altitude.m)	1	0.686	0.063	2.626	0.001
scale(Aspect)	1	0.367	0.034	1.405	0.130
scale(Mean.Top.Height.m)	1	0.576	0.053	2.204	0.003
scale(Fenced.years)	1	0.575	0.053	2.201	0.005
scale(Canopy.Cover..)	1	0.312	0.029	1.194	0.256
scale(cover_PC1)	1	0.442	0.041	1.693	0.044
scale(cover_PC2)	1	0.613	0.056	2.345	0.002
scale(cover_PC3)	1	0.251	0.023	0.961	0.511
scale(soil_PC1)	1	0.266	0.024	1.019	0.426
scale(soil_PC2)	1	0.315	0.029	1.207	0.250
Lagomorphs	3	0.948	0.087	1.210	0.168
Ungulates	1	0.234	0.022	0.897	0.574
Residual	13	3.396	0.312	NA	NA
Total	29	10.876	1.000	NA	NA

H

adonis_ordination_herbaceous_natualised

	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	1.002	0.106	4.289	0.000
scale(Northing.Latitude)	1	0.814	0.086	3.484	0.000
scale(Altitude.m)	1	0.708	0.075	3.032	0.000
scale(Aspect)	1	0.300	0.032	1.284	0.219
scale(Mean.Top.Height.m)	1	0.545	0.058	2.334	0.003
scale(Fenced.years)	1	0.628	0.067	2.688	0.001
scale(Canopy.Cover..)	1	0.304	0.032	1.302	0.207
scale(cover_PC1)	1	0.466	0.049	1.994	0.018
scale(cover_PC2)	1	0.606	0.064	2.594	0.002
scale(cover_PC3)	1	0.079	0.008	0.340	0.992
scale(soil_PC1)	1	0.316	0.033	1.352	0.173
scale(soil_PC2)	1	0.217	0.023	0.930	0.527
Lagomorphs	3	1.063	0.113	1.517	0.033
Ungulates	1	0.281	0.030	1.205	0.267
Residual	9	2.102	0.223	NA	NA
Total	25	9.431	1.000	NA	NA

3.3.2 Hypothesis 2: There will be fewer natives species richness in the understory of the unfenced sites compared with fenced sites.

Big Adults

There was a positive association between native species richness and years fenced (Figure 3.2A, Table 3.5A), although note that most of these big adult trees will be older than the oldest fences. Latitude had a significant negative effect on big adults' species richness. There were no exotic trees in any plots.

Small adults

Small adult tree species richness increased with the years a forest fragment had been fenced (Figure 3.2B, Table 3.5A). The species richness of small adults is most pronounced after 20 year with the mean richness of small trees at sites fenced for 20 years was 17.8, which was greater than the richness at sites that were unfenced or had been fenced for <20 years with a mean of 10.6. There were no exotic trees in any plots.

Native saplings

Native sapling species richness was greater in the fenced sites than in the unfenced sites with a significant positive influence on species richness as time from fencing increased (Figure 3.2C, Table 3.5C). There were no exotic saplings found in forest fragments.

Woody seedlings >135 cm

There was an increase in the seedlings of woody species richness with the number of years that the forest fragment had been fenced (Figure 3.2D,Table 3.5D). There were no exotic woody species found in either the fenced or unfenced vegetation plot surveys.

Woody seedlings <15 cm

Woody seedlings under 15cm shared a relatively similar species richness across both fenced and unfenced sites (Figure 3.2E, Table 3.5E). Species richness appeared more concentrated with a richness of 4 seedlings richness per plot. There was a slight relationship of increase in small seedling richness per plot with fencing years. No exotic woody species under 15 cm were found in the survey.

Herbaceous vegetation >135 cm

Within this study, herbaceous vegetation was also surveyed with the seedling survey as a presence or absence in different height tiers. By measuring the pool of all native and non-native herbaceous species, it was found that unfenced sites had the greatest herbaceous species richness and there was a negative correlation between fenced years and species richness (Figure 3.2F, Table 3.5F)

Native herbaceous vegetation

Of these herbaceous species, native herbaceous species richness per plot was not significantly affected by fencing, or years fenced (Figure 3.2G and Table 3.5G).

Naturalised herbaceous vegetation

In contrast to the native species, there were significantly more exotic vegetation species within unfenced than in fenced sites (Figure 3.2H and Table 3.5H).

Herbaceous growth form

Herbaceous growth forms were separated into ferns, graminoids, and forbs of the observed naturalised species and native species and compared between fenced and unfenced sites.

Ferns

No exotic ferns were found. While there was a tendency for more fern species at fenced sites, this was not significant (Figure 3.2I and Table 3.5I).

Forbs

There were more naturalised forb species in unfenced sites (15 species) compared to fenced sites (4 species). There appears to be a negative association between the length of time since plots were fenced and the presence of naturalised forbs (Figure 3.2J and Table 3.5J). Similarly, there was a high species richness per plot of native forb species in unfenced sites (10 species) than in fenced sites (Figure 3.2K and Table 3.5K).

Graminoids

In the native graminoids (Figure 3.2L and Table 3.5L) there appear no differences in species richness irrespective of the treatment, but there were many more naturalised species than native species. There was significantly higher species richness per plot of naturalised grass species in the unfenced (7 species) than in fenced sites (4 species) (Figure.2M and Table 3.5M) Species richness of naturalised graminoids declined the longer a site was fenced.

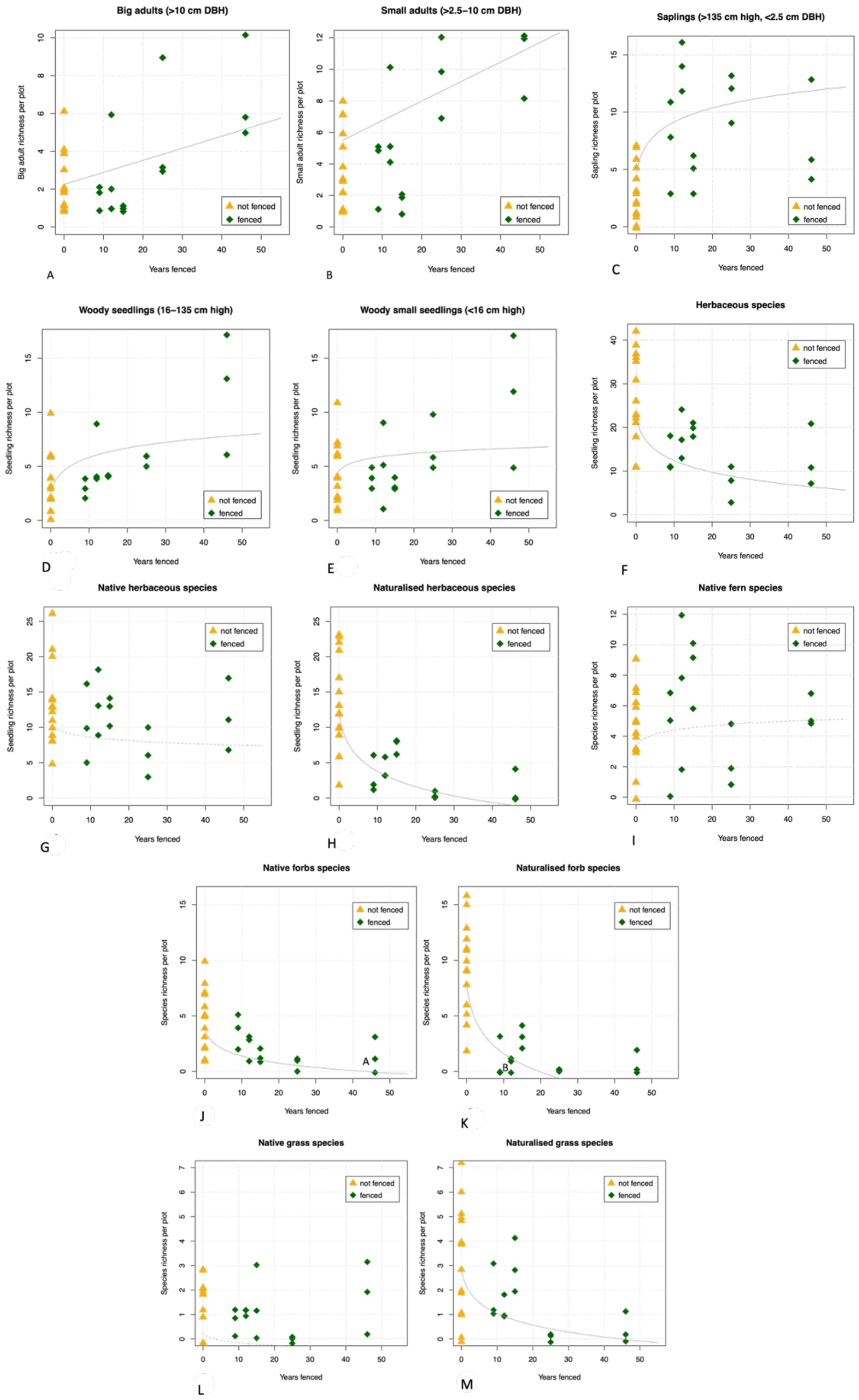
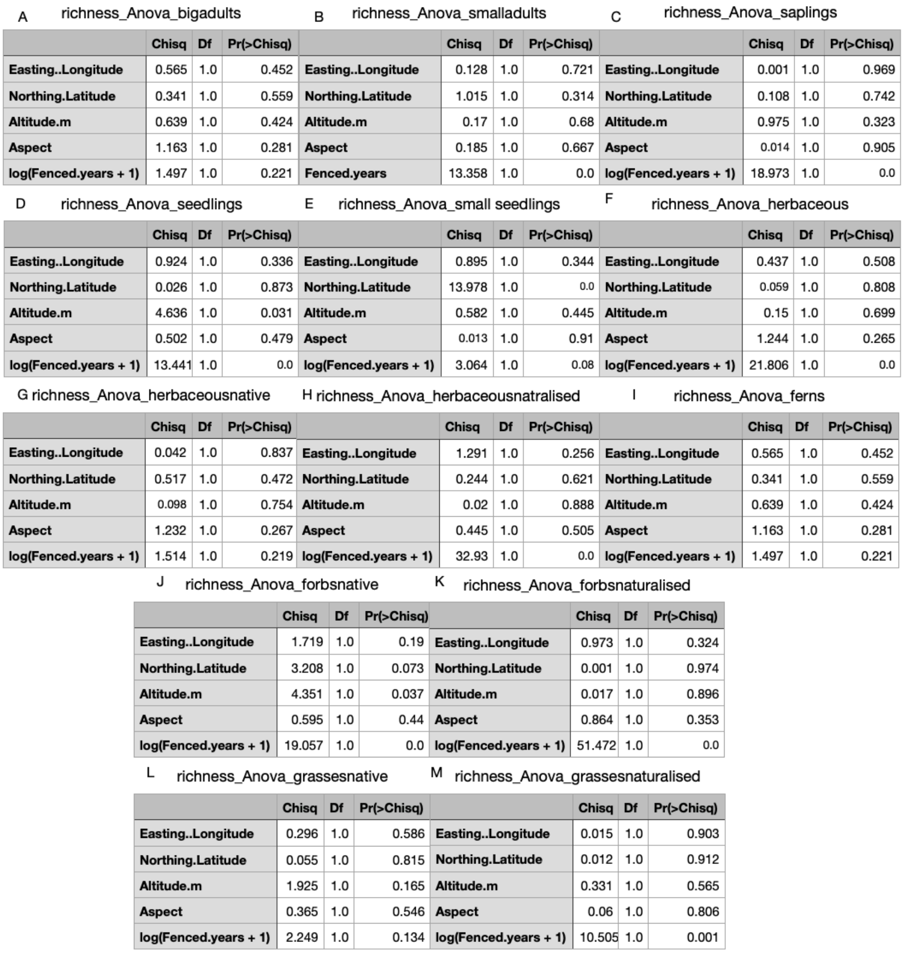


Figure 3.2: Mixed models regression displaying the relationship between big adult (A) and small adult (B) species richness and fencing years sites of adult tree composition. Fenced sites are in green and unfenced sites are in yellow. Curved regression displaying the relationship between saplings (C), seedlings (D), small seedlings (E), herbaceous (F), native herbaceous (G), naturalised herbaceous (H), ferns (I), forbs native (J), forbs naturalised (K), graminoids (grasses) native (L), and graminoids (grasses) native (M) species richness and fencing years.

Table 3.5: Analysis of Deviance Tables from GLMs of values of significant environmental variables that affect species richness across all height tiers are in. Within the table, significant values are separated into height tiers: big adults (A), small adults (B), saplings (C), seedlings (D), small seedlings (E), herbaceous (F), native herbaceous (G), naturalised herbaceous (H), ferns (I), forbs native (J), forbs naturalised (K), graminoids (grasses) native (L), and graminoids (grasses) native (M). The significance of environmental values based on species composition is represented by the P-value



A richness_Anova_bigadults

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.565	1.0	0.452
Northing.Latitude	0.341	1.0	0.559
Altitude.m	0.639	1.0	0.424
Aspect	1.163	1.0	0.281
log(Fenced.years + 1)	1.497	1.0	0.221

B richness_Anova_smalladults

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.128	1.0	0.721
Northing.Latitude	1.015	1.0	0.314
Altitude.m	0.17	1.0	0.68
Aspect	0.185	1.0	0.667
Fenced.years	13.358	1.0	0.0

C richness_Anova_saplings

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.001	1.0	0.969
Northing.Latitude	0.108	1.0	0.742
Altitude.m	0.975	1.0	0.323
Aspect	0.014	1.0	0.905
log(Fenced.years + 1)	18.973	1.0	0.0

D richness_Anova_seedlings

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.924	1.0	0.336
Northing.Latitude	0.026	1.0	0.873
Altitude.m	4.636	1.0	0.031
Aspect	0.502	1.0	0.479
log(Fenced.years + 1)	13.441	1.0	0.0

E richness_Anova_small seedlings

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.895	1.0	0.344
Northing.Latitude	13.978	1.0	0.0
Altitude.m	0.582	1.0	0.445
Aspect	0.013	1.0	0.91
log(Fenced.years + 1)	3.064	1.0	0.08

F richness_Anova_herbaceous

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.437	1.0	0.508
Northing.Latitude	0.059	1.0	0.808
Altitude.m	0.15	1.0	0.699
Aspect	1.244	1.0	0.265
log(Fenced.years + 1)	21.806	1.0	0.0

G richness_Anova_herbaceousnative

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.042	1.0	0.837
Northing.Latitude	0.517	1.0	0.472
Altitude.m	0.098	1.0	0.754
Aspect	1.232	1.0	0.267
log(Fenced.years + 1)	1.514	1.0	0.219

H richness_Anova_herbaceousnatralised

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	1.291	1.0	0.256
Northing.Latitude	0.244	1.0	0.621
Altitude.m	0.02	1.0	0.888
Aspect	0.445	1.0	0.505
log(Fenced.years + 1)	32.93	1.0	0.0

I richness_Anova_ferns

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.565	1.0	0.452
Northing.Latitude	0.341	1.0	0.559
Altitude.m	0.639	1.0	0.424
Aspect	1.163	1.0	0.281
log(Fenced.years + 1)	1.497	1.0	0.221

J richness_Anova_forbsnative

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	1.719	1.0	0.19
Northing.Latitude	3.208	1.0	0.073
Altitude.m	4.351	1.0	0.037
Aspect	0.595	1.0	0.44
log(Fenced.years + 1)	19.057	1.0	0.0

K richness_Anova_forbsnaturalised

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.973	1.0	0.324
Northing.Latitude	0.001	1.0	0.974
Altitude.m	0.017	1.0	0.896
Aspect	0.864	1.0	0.353
log(Fenced.years + 1)	51.472	1.0	0.0

L richness_Anova_grassesnative

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.296	1.0	0.586
Northing.Latitude	0.055	1.0	0.815
Altitude.m	1.925	1.0	0.165
Aspect	0.365	1.0	0.546
log(Fenced.years + 1)	2.249	1.0	0.134

M richness_Anova_grassesnaturalised

	Chisq	Df	Pr(>Chisq)
Easting..Longitude	0.015	1.0	0.903
Northing.Latitude	0.012	1.0	0.912
Altitude.m	0.331	1.0	0.565
Aspect	0.06	1.0	0.806
log(Fenced.years + 1)	10.505	1.0	0.001

3.3.3 Hypothesis 3: There will be a higher proportion of palatable species, especially in lower height tiers, in fenced sites than in unfenced sites

Vegetation palatability was highest in fenced sites, especially in the lower height tiers (Fig. 3.3, Table 3.6). Across all the stems that were of species with ungulate palatability data in Forsyth et al. (2002), fenced plots when compared with unfenced plots had 3.3 times more palatable small seedlings, 2.1 times more palatable seedlings, 1.8 times more palatable saplings, and 1.8 times more small adults (Fig. 3.3, Table 3.6). "Palatable" here refers to those species that ungulate prefers more than random. There was no difference in big adults. There were also significant interactions between palatability and fenced years, with the sites fenced longer generally having a higher proportion of palatable species, especially in lower height tiers (Table 3.7).

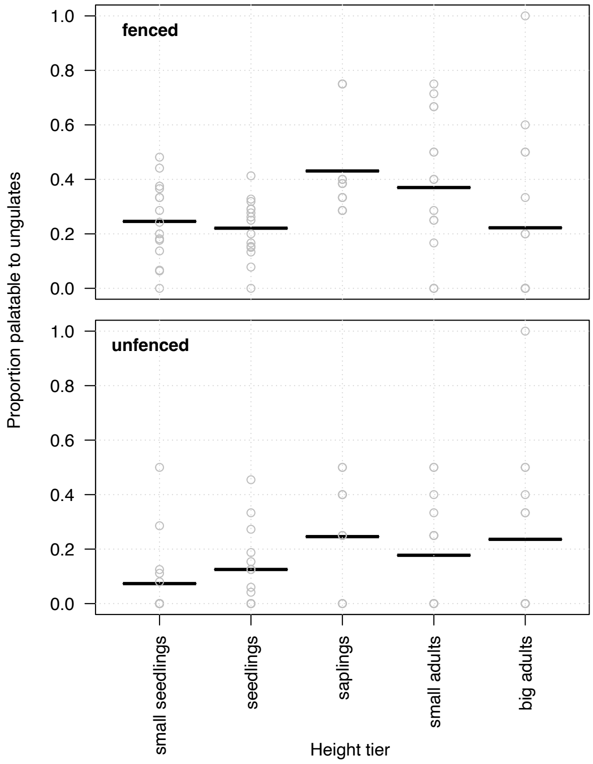


Figure 3.3: The proportion of total stems of species within a plot that are preferred by ungulates, across the vegetation height tiers between fenced and unfenced sites. The bars display the mean values and the grey circles are each plot.

Table 3.6: Ratios of the mean of the growth form abundance of palatable species present across fenced and unfenced sites. This table illustrates much variance between height tiers, represented across different rows, and within fencing management (Y/N).

Abundance of palatable species (mean)			
Height tiers & fencing (Y/N)	**avoided mean_abundance**	**Not selected mean_abundance**	**Preferred mean_abundance**
Small seedlings (Y)	3.129	4.659	4.413
Small seedlings (N)	2.888	2.931	1.428
Seedlings (Y)	4.542	5.578	4.727
Seedlings (N)	4.966	4.55	2.190
Saplings (Y)	2.023	15.333	25.980
Saplings (N)	32.478	1.333	1
Small adults (Y)	32.478	6	8.68
Small adults (N)	41.058	2.25	2.375
Big adults (Y)	17.888	5.571	2.5
Big adults (N)	18.785	2.75	1.571

Table 3.7: The analysis of the deviance table from the mixed models palatability; fenced_yesno; palatability:fenced_yesno.

Palatability (GLMER_anova) fenced yes/no			
Small seedlings	**Chisq**	**Df**	**Pr(>Chisq)**
Palatability	3.0075	2	0.2223
fenced_yesno	2.3110	1	0.1285
Palatability:fenced_yesno	8.7371	2	0.0127
Seedlings	**Chisq**	**Df**	**Pr(>Chisq)**
Palatability	23.9891	2	0.0000
fenced_yesno	2.2184	1	0.1364
Palatability:fenced_yesno	21.0763	2	0.0000
Saplings	**Chisq**	**Df**	**Pr(>Chisq)**
Palatability	392.1949	2	0.0000
fenced_yesno	0.0379	1	0.8457
Palatability:fenced_yesno	270.0579	2	0.0000
Small adults	**Chisq**	**Df**	**Pr(>Chisq)**
Palatability	426.2697	2	0.0000
fenced_yesno	1.0340	1	0.3092
Palatability:fenced_yesno	22.4940	2	0.0000
Big adults	**Chisq**	**Df**	**Pr(>Chisq)**
Palatability	103.8693	2	0.0000
fenced_yesno	1.2236	1	0.2686
Palatability:fenced_yesno	10.8389	2	0.0044

3.3.4 Hypothesis 4: There will be a significant relationship between species succession, recruitment, height, and stage between fenced and unfenced sites and other environmental factors.

While fenced and unfenced sites both had more stems in the smaller than larger height tiers, the fenced sites had many more small stems than the unfenced sites (Figure 3.4). There were on average 2.7 to 3.7 times more stems in the fenced than unfenced sites in the lower three height tiers (Table 3.8 and Table 3.9).

Figure 3.5 shows this difference in stems across the five different paired fenced and unfenced sites. While overall there are more stems in fenced than unfenced sites, especially in the lower height tiers, there was also considerable variation across the sites, and there were few differences between fenced and unfenced sites at the K/D site.

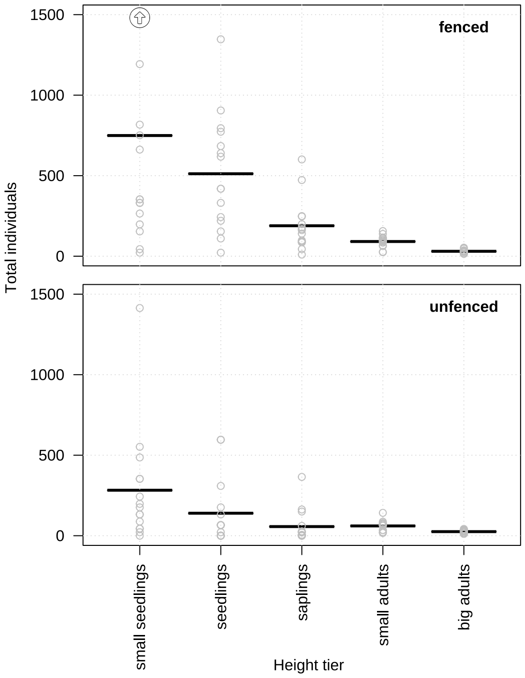


Figure 3.4: Separated into fenced versa unfence parallel to one another demonstrating the difference in total counts within height tiers within each of the 20 x 20 m sampled plots. The line indicates the mean of the total individuals per plot for each tier, with plotted points representing outliers of individual counts for each of the plots. Data on the y-axis are the counts of total individuals across tiers, with the height tiers on the x-axis. The arrow within the small seedling tier demonstrates that there are two additional total plot points within the study that have been excluded from the graph. The arrow indicates 2 small seedling plot means that are outliers (2615 and 3115) for small seedling fenced sites, (both found within the McLean's site).

Table 3.8: Stem count means and standard errors in the different height tiers in plots in fenced and unfenced sites. This is the proportion between the counts of height tiers and the total counts of fenced and unfenced sites.

Stem Count (means(s.e.))			
	Unfenced	Fenced	Proportion
Big adults	25.3(2.7)	30.5(2.7)	1.2
Small adults	60.7(8.3)	91.3(9.2)	1.5
Saplings	56.7(25.5)	189.1(41)	3.3
Seedlings	139.7(52.5)	512(92.4)	3.7
Small seedlings	282.5(91.8)	749.3(239.1)	2.7
Total	425.2	1572.2	3.7

Table 3.9: The Analysis of Deviance table for the mixed model of stem count.

Stem count model			
	Chisq	Df	Pr(>Chisq)
height.tier	4.877E+01	4	6.515E-10
fenced_yesno	1.334E+01	1	2.601E-04
height.tier:fenced_yesno	2.292E+01	4	1.312E-04

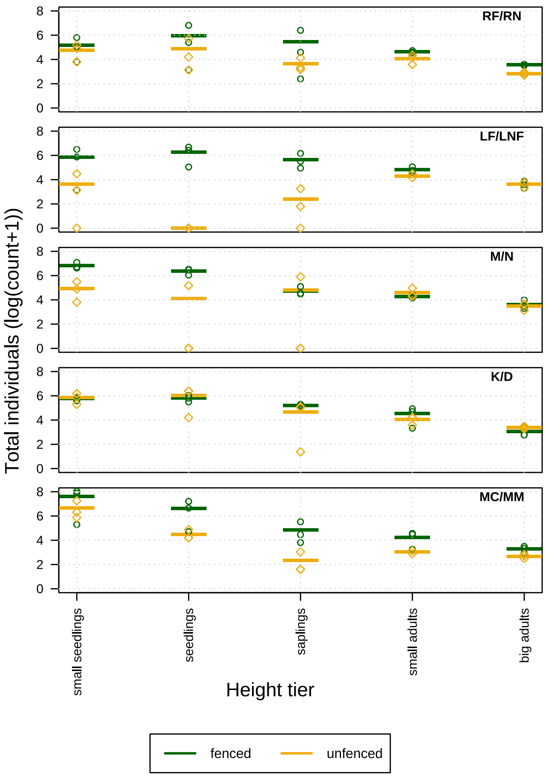


Figure 3.5: Total of individuals (log-transformed) for each site and each height tier including fencing management treatment (yellow = unfenced and green = fenced).

Discussion

3.3.5 Hypothesis 1: The composition of species will differ between unfenced and fenced sites

Within this study, the vegetation was affected by more than fencing management alone, with other factors including altitude, latitude, and canopy. The vegetation structure was measured by height tiers with characteristics of different species compositions determined by fencing management and environmental factors. It was found by Allen et al. (2019) that forests are undergoing both localised effects, such as altitude, and pervasive effects (e.g. climate change) that are changing the vegetation composition. This complements my study as it was found that there is large complexity of effects that determine forest compositions. Relationships between vegetation composition and environmental factors are expected as they control biological processes that give life to species and function in ecosystems (Burns,1995).

Effects of environmental factors were evident, with altitude being the most significant variable among all fenced and unfenced sites within the study. Altitude affects species richness for many native species in NZ (Coomes & Allen, 2007b; McGlone et al., 2010; Pausas & Austin, 2001), and it has indicated that although altitude itself doesn't have direct effects on vegetation dynamics, it had a direct impact on other environmental variables influencing vegetation dynamics. For example, as suggested by Allen et al., (1984), the transition of vegetation dynamics was a continuum in terms of changing species composition with altitude. Burns (1995) found that altitude was a determining factor of species composition as it causes gradients of environmental conditions that influence species composition at different locations, such as weather patterns and rainfall. Burns (1995) found vegetation growth forms, such as epiphytes and climbers, increased with altitude at a greater rate than tree species richness declined. My study had similar results, as herbaceous vegetation species richness increased with altitude. Coomes & Allen, (2007b); McGlone et al. (2010); Pausas & Austin, (2001), all stated (and found within this study) show that the effect of altitude shared strong relationships with other environmental variables, such as ground cover and the presence of lagomorphs.

Latitude appeared to play a significant role in the vegetation dynamics of forest composition. McGlone et al. (2010) suggested that latitude influences forest composition because coastal environments have higher disturbance regimes compared to inland areas. Disturbance creates environments susceptible for the invasion of exotic plants (Sullivan et al., 2005) and environmental extremes or disturbances for coastal forests (McGlone et al., 2010).

Fenced years was aligned with the canopy tree height, affecting species composition. This suggests that seedling and sapling species composition in fenced plots were determined by how long a site had been fenced and the canopy height, which created favourable conditions for shade-tolerant species. Burns (1995) and Coomes & Allen (2007b) both found that forests with greater canopy height reduce light levels to the sub-canopy, limiting sapling and seedling development and succession for shade-tolerant species. Tree growth and the canopy provided are determined by the factors affecting resource competition and environmental pressures or threats (Coomes & Allen, 2007b). Therefore, canopy height may determine the growth of vegetative ground cover in lower tiers because of resource availability such as reduced light which creates light competition for saplings and seedlings.

Difficulties exist when measuring potential species compositions across height tiers and the comparability of species composition between sites. Plant species may spend long periods in light-limited levels, followed by relatively brief periods of growth because of light openings from a variety of issues such as disturbance, seasonal changes, or biological invasions in the higher canopy levels (Smale, 2008; Wright et al., 2000). However, vegetation dynamics allow for a current understanding of the ecological condition, allowing for a prediction of forest dynamics through understanding the effects of species composition.

3.3.6 Hypothesis 2: There will be lower natives species richness in the understory of the unfenced sites compared with fenced sites

My findings are contradictory as evidence is both supportive and non-supportive of this hypothesis. Overall, native species richness reduces within lower height tiers within the unfenced plots, but species richness of native herbaceous species increased in unfenced sites. In contrast, surveyed woody vegetation has less native species richness in unfenced sites compared to fenced sites. Factors that hinder or enhance exotic vegetation species to naturalise within native forests complex issues, and my study only investigated the effect of fencing on species richness and height.

Scofield et al. (2011) and Tanentzap & Lloyd (2017) described how fenced forested areas support native species richness that could not otherwise survive in the surrounding landscape because of landscape matrix alterations, land usage, biological invasions, and other threats. In my study, surveying unfenced forests allowed me to compare the physical provision of fencing for native woody species across height tiers. My study found, with support from other studies (Aviss & Roberts, 1994; Scofield et al., 2011; Tanentzap & Lloyd, 2017), that vegetation dynamics were dependent on the fencing management. Fences create a physical barrier that excludes most herbivores, removing browsing pressure on vegetation allowing plant growth and succession, and therefore increasing species richness (Innes et al., 2012).

Fencing aids protecting forest growth by assisting in physically mitigating the effects that threaten susceptible native vegetation from herbivores such as lagomorphs. Both Aviss & Roberts (1994) and

Innes et al. (2012) concluded that without fencing, herbivory threatens the integrity of forests as they create unfavourable disturbance by compacting eroding soil; browse, and trampled seedlings which reduce and even prevent natural regeneration; damages tree bark and roots; introduces naturalised herbaceous species through depositing seeds through hooves and droppings; and can increase soils fertility, of which is favoured by exotics rather than natives (Aviss & Roberts, 1994). My findings demonstrate how fencing determines vegetation dynamics through enhancing favourable conditions (less browsing pressure) for woody native species and reducing the number of exotic plant species (creating favourable habitats for natives and unfavourable for exotics). Overall, species richness of woody vegetation was greater in fenced sites, where there was no grazing pressure or disruption (trampling, excess nutrients, herbivory), compared to unfenced sites, yet with no effect on adult trees.

McGlone et al. (2010) found that for the temperate region, alike the Hawke's Bay region, the flora is unusually dominated by trees, however, our climate is still very much suited for annuals and biennials (Kelly & Sullivan, 2010a). Large amounts of annuals and biennials were surveyed within my study, with significantly greater naturalised forb species in unfenced sites (15) compared to fenced sites (4), along with a relatively lower presence of exotic and naturalised herbaceous species. Lunt et al. (2007) shared findings whereby species richness correlates with stock presence, creating more dispersal opportunities and grazing exclusion led to changes in the abundance of common, most exotic species.

My study suggests that fencing management dictates woody seedlings and herbaceous species richness. (Diemer, 1998) explains how the life span of many herbaceous species is short-lived, following either annual or biannual seasonal rhythms, compared with long-lived woody species, suggesting that even when native herbaceous species are grazed, they will respond with increased growth rates (Hulme, 2020). Furthermore, herbaceous species have different root formations and can keep growing once grazed (Bourdôt et al., 2007; Dodd et al., 2011) making them more resistant to browsing and other disturbance meaning they have evolved with more resistant traits favouring higher levels of disturbance (Bourdôt et al., 2007). This is demonstrated in the unfenced forest fragments, where herbaceous species richness (both native and naturalised) was great as they are continuously replenished yet resilient by disturbances such as stock.

Remarkably, no exotic woody adults were observed within any fenced and unfenced plots, although sites were selected based on native dominant tree species (refer to Chapter 2). The presence of natives and the absence of exotic saplings and adult trees correlated with seed provision which dictates what species grow based on the surrounding landscape matrix that has a relativity devoid source of woody species (Estelle J. Dominati et al., 2019; Jay, 2005). Plus, the exclusion of grazing from fencing allows many of the native seedlings to transition into saplings and then adults (Lunt et al., 2007).

It is expected that there are many more naturalised plant species present within the surveyed forest fragments than were present in the plots, and naturalised richness is predicted to grow as more exotic pastural species invade the fragments (Aikio et al., 2012; Bourdôt et al., 2007; Hulme, 2020; Sullivan et al., 2005). Unfenced forest fragments provide a favourable environment for exotic naturalised species and native herbaceous species. In contrast, I found that for woody species, fencing provides favourable conditions for native woody species.

In support of my study, Aviss & Roberts (1994) and Burns et al. (2011) described how fenced years had the greatest effect on the native and naturalised vegetation species across different height tiers and growth forms. The fenced years had the greatest impact on species richness and composition similarities, with a shared effect with both saplings and seedlings, suggesting that the fenced years determines vegetation dynamics. Fenced sites had a significant increase in native sapling species richness, with this increase continuing with the fenced years. Fenced years substantially had the greatest positive effect on small woody seedlings and a decreasing effect on naturalised exotic herbaceous species richness. Correspondingly, as fenced years increased, exotic forb species decreased, demonstrating an increase in native species richness as conditions are more favourable and can then outcompete naturalised species, resulting in regeneration opportunities for native species as the fencing provides favourable and protected conditions for an increase in native species richness.

To better understand factors affecting the spread and establishment of naturalised species, extensive environmental variables, such as climate patterns, should be investigated in future studies. Primarily, methods provide a consistent approach for a broader suite for species detection across the site for monitoring cryptic or vegetation which is excluded from the plot (exterior) is necessary as the current T2BMF approach misses detection opportunities.

3.3.7 Hypothesis 3: The proportion of palatable species across the different height tiers will be dependent on fencing management

Findings support my third hypothesis as there was an increase in proportions of palatable species within the fenced sites, indicating that fencing provides a distinctive positive effect on population proportions of palatability across height tiers as fencing physically protects the palatable vegetation extensively more so than unfenced forests. It was found that there was a larger proportion of palatable species in fenced sites than in unfenced sites, especially in the lower tiers (height tiers that are more prone to grazing).

To maintain or enhance the ecological integrity of native forests, appropriate fencing management around the perimeter of these forests is highly encouraged and regarded as one of the most effective protection for vegetation by many studies (Aviss & Roberts, 1994; Dodd et al., 2011; Scofield et al., 2011; Tanentzap & Lloyd, 2017). Ungulate herbivory threatens forest integrity as they create unfavourable disturbance and composition changes by compact eroding soil; browse and trampled seedlings which reduce and even prevent natural regeneration; damages tree bark and roots; introduces naturalised herbaceous species through depositing seeds through hooves and droppings; and can increase soils fertility, of which is more so favoured by exotics rather than natives (Aviss & Roberts, 1994).

Fencing offers a mechanical strategy to separate vulnerable biota from threats in the surrounding environment and some herbivory pressure (e.g., (Aviss & Roberts, 1994; Burns et al., 2011; Tanentzap & Lloyd, 2017)). Herbivory had a significant effect on the proportion of palatable species as vegetation within fenced sites had a much greater proportion of palatable species compared with unfenced sites indicating that fenced sites contained a greater proportion of palatable species where ungulates (or stock alike) were unable to browse (Figure 3.4). In the absence of browsing, palatable species richness would increase over time, as seen in my study as in unfenced sites, herbivory pressure was apparent on palatable plant communities and therefore the manipulation of vegetation dynamics. This indicates that plots in sites that were fenced for longer had a higher proportion of palatable species compared with plots in the unfenced site.

NZ native flora has evolved in the absence of ungulate browsing, although the native flora has experienced a wide variety of avian herbivory (Bee et al., 2007). As characterised by Forsyth et al., (2002), New Zealand's native vegetation is relatively palatable to introduced ungulates. Native vegetation regeneration is largely determined by the browsing pressure of introduced mammals (Richardson et al., 2014). Richardson et al (2014) along with Lloyd et al. (2010) determined that the impact of ungulate herbivory results in a decreased abundance of palatable species. Augustine & McNaughton (1998) and Lloyd et al. (2010) found that the leaf traits that contribute to a palatability index, as defined by Forsyth et al., (2002), contribute to the palatability of vegetative biomass within a system, and therefore the leaf litter composition changes as a consequence demonstrating that native saplings and seedlings are sensitive to introduced herbivores, such as ungulates (Smale, 2008; Tanentzap & Lloyd, 2017; Wardle et al., 2001), and in this study seedlings and saplings palatability proportions varied the greatest (Table 3.6).

Allen et al. (1984) found that the exclusion of browsing mammals in the Urewera's resulted in plant species showing different response patterns based on their leaf traits. They found that some species, such as prickly heath, were found within and outside of the enclosures. Species that are avoided by ungulates, such as small-leaved Coprosmas, become dominant within a forest because of reduced spatial and resource competition (Forsyth et al. 2002; Smale et al. 1995). This is demonstrated in both our study and R. B. Allen et al. (1984), whereby more palatable species, such as hangehange (*Geniostoma ligustrifolium*) were found within fenced sites. The different proportions of these palatable species in the fenced or unfenced sites along with leaf functional traits could potentially predict the impact of ungulate herbivory in lower-tier species and change these dynamics.

Allen et al. (1984) found other common species (e.g. red māpou (*Myrsine australis*) and mahoe (*Melicytus ramiflorus*)) were more frequently inside the enclosures than outside compared with other species (e.g. large-leaved coprosma and hangehange (*Geniostoma ligustrifolium*)) which were virtually confined within enclosures. In the long-term, ungulate presence alters the vegetation composition through selective browsing as they sufficiently remove large-leaved shade-tolerant palatable species (Smale et al., 1995). This is a similar finding within our study, whereby in sites that were fenced for a shorter period (e.g. the Mark site), there was a high presence of short-leaved *Coprosma* spp. Allen et al. (1984) was a source within the Forsyth et al. (2002) diet palatability index, demonstrating variance of population proportions based on the exclusion of browsing effects on vegetation dynamics. Allen et al. (1984) was a great reflection in comparison for my study as both were conducted in the Hawke's Bay region whereby the vrgetation compositions are somewhat similar. Further, both my study and Allen et al. (1984) concludes that fencing has a significant effect on the proportion of palatable species.

Furthermore, palatability avoidance and feeding preference of herbivores can lead to altered nutrient and decomposition cycles (Lucas, n.d.; Wardle et al., 2001). Large-leaved plants are often more palatable to herbivores and also tend to provide the largest proportion of biomass to a forest ecosystem, as described by Allen et al., (2019), this then accounts for a larger basal area, which modifies the abilities of seedling presence and success for many other species. Therefore, many palatable species are keystone species within forests ecosystems (Allen et al., 2019; Burns et al., 2011; Smale et al., 1995)

Lloyd et al. (2010) suggested that there should be further research investigating levels of palatability across different ungulate species as Forsyth et al. (2002) only provide proportions of one palatability index for one taxon, further research should investigate the palatability of other taxa. For instance, such as other herbivory from introduced mammals such as possums (*Trichosurus vulpecula*) and feral cattle (*Bos taurus*), that browse seedlings, saplings, and trees require further studies (Smale, 2008). In addition, other plant taxa palatability should be investigated concerning recruitment processes within the regeneration cycle. Biotic and abiotic factors should also be investigated including wind exposure, drought, snow, and frost, along with the effects of climate change. These factors may contribute to microclimatic conditions, which can determine seedling success and seedling presence in the regeneration cycle (Smale, 2008).

3.3.8 Hypothesis 4: There will be a significant relationship between species succession, recruitment, height, and stage between fenced and unfenced sites and other environmental factors.

The total individual counts varied across height tiers, with lower tiers total individual counts were significantly greater within fenced sites compared with those in unfenced sites, as stated in the hypothesis.

Populations of native forests usually exhibit a history of large variations in vegetation dynamics that are significant to the forest structure and biomass (Coomes & Allen, 2007b). Determining a forest's integrity requires an understanding of the variation of dynamics amongst species such as the longevity, shade tolerance, size at reproduction, relative growth, and mortality rates at different life stages over time (McGlone et al., 2010; Peltzer et al., 2014). Many contributing factors determine vegetation dynamics and growth within NZ forest and plant communities, such as altitude, nutrient availability, light availability, competition, herbivory, and disease (Kunstler et al., 2009; McGlone et al., 2010; Wallace et al., 2018).

Adult trees offer important functional and supportive elements for subcanopy succession as they provide much of the light percentage, biomass, govern the vegetation ground cover in tropical or temperate regions, and can also have extensive and cascading effects on other interactions with biota (McGlone et al., 2010). Fenced and unfenced sites shared a similar negative trend of total individuals from small seedlings to big adult trees, with the initial total individual counts varying. The was no difference between the numbers of large individual trees as the fencing had not affected the succession of adult trees or no significant association between the fencing and small adults, as they were already present when the forest was fenced.

The growth of trees is determined by measuring diameter at breast height and multi-stemmed specimens (Russo et al., 2010). McGlone et al. (2010) and Russo et al. (2010) advocated measuring stem diameter as it gives an indication of woody plant densities in a forest community that influences the growth and mortality of other woody plants below the main canopy (Russo et al., 2010). My study concluded that stem counts were dependent on fencing management, as there were greater individual counts within fenced sites compared to unfenced sites. This finding was also consistent with Stern et al. (2002) as increased cattle grazing intensities impacted the total count and stem counts of trees, whereby there were significantly more small stemmed plants in grazed areas and significantly fewer large stemmed plants. My study suggests the structure of tree stems is dependent on fencing, whereby there was a greater number of tree stems in fenced sites compared with unfenced ones. Results in my study showed significant variation in stem and trunk measurements, suggesting that there was variation in vegetation dynamics amongst trees, within the different tree tiers as specifically small adults were dependent on the fencing management.

There were also large variations in the total number of individuals across the seedling and sapling height tiers suggesting that other than fencing, there are other significant environmental variables, such as light exposure and grazing, influencing the number of plants found within a plot. These environmental variables contribute to the growth of saplings and seedlings as plants grow and succeed in favourable conditions (Peltzer et al., 2014; Wright et al., 2000), and seedlings and saplings were still present in unfenced sites.

McGlone et al. (2010) found that vegetation dynamics are often restricted as plant communities have specialised niches with specific biota dynamics under trees. Larger trees interconnect small vegetation species through fundamental macroclimatic, light exposure, and soil conditions (McGlone et al., 2010). Additionally, both Bussell (1968) and Wardle (1963) concluded that gathering different tier data below adult trees is vital as much of NZ’s native vegetation is relatively slow-growing, relating growth to suppression and release dynamics. This involves frequent episodes of both suppression and release of growth spurts before reaching even subcanopy size, this suppression behaviour can persist for many decades, without a discernible growth (Smale, 2008; Wright et al., 2000). In this study, the surveyed seedlings may be an inadequate measurement of growth as native species of seedlings (and even saplings and trees) can prolong their existence by vegetative regrowth after dieback (Smale, 2008). However, survey counts are necessary to gauge forest integrity through understanding the succession and abundance of species across height tiers.

These growth/suppression episodes are often dictated by shade and light exposures and influence seedling and sapling species richness (Wright et al., 2000). Shade tolerant species have a relationship to past periods of suppression and release for sapling and seedling growth whereas species that are less shade-tolerant have varied responses to suppression and release of light exposure but can show a lag in response to this exposure (Wright et al., 2000). Lusk (2002) suggests that light dependant species (much of lowland native forest species) have significant growth rates and composition changes concerning light exposure or competition.

Due to my survey's large variability in the height tiers' total individual counts, it is assumed that light exposure was to play. The degree of light availability impacts native tree species' growth rates, as suggested by Coomes & Allen (2007b) and Muller-Landau et al. (2006) whereby competition regarding light availability and the stability of this light within a forest system determines the vegetation dynamics. Reduced light exposure, due to older trees and a greater canopy cover reduces light transmission, which is presumably responsible for the reduced development of larger seedlings, unlike sapling growth. In my study, there were lower seedling counts in plots that had been fenced for longer and therefore have greater canopy cover.

There were changes in individual tree counts across sites and it is assumed that this is associated with altitude. Other studies have indicated that altitude affects growth dynamics as temperature strongly influences the photosynthetic capacity of plants at higher altitudes (Coomes & Allen, 2007b; McGlone et al., 2010; Pausas & Austin, 2001) as altitude directly affects environmental variables, influencing dynamics.

There was a similar number of mature trees in the fenced and unfenced sites, implying that within unfenced sites that there are an inadequate number of seedlings and saplings to replenish the older more mature trees relative to the level of succession occurring in the fenced sites. Unfenced sites harbour fewer individuals in the lower tiers, suggesting little regeneration resulting in collapses in successional or recruitment processes. There was also a concerning negative trend in the number of individual seedlings across the unfenced sites. Major declines in established seedling populations are not to be expected within a regenerative cycle, as suggested by Smale (2008), because of the shortage of continuous seedling replacement. My study elucidates that even within fenced sites, the seedling counts decrease because of other issues such as competition and environmental variables. However, given that most seedlings and juveniles are destined to die before reaching a reproductive age or height tier, it is concerning that there are fewer individual seedlings in unfenced sites and therefore the percentage of survival is lessened initially. The reduced individual seedling count and seedling recruitment cycle have likely been influenced by a variety of factors, such as herbivory, habitat fragmentation, and environmental degradation and this has then affected one or more stages of the regeneration cycles: pollination, fruitful, fruit dispersal, seed predation, dispersal, germination, seedling establishment, and survival (Smale, 2008).

This scenario of failing recruitment for regeneration is not unique (Augustine & McNaughton, 1998; Coomes, Allen, et al., 2003; Enright & Watson, 1992; Grime, 2006; Richardson et al., 2014; van der Valk et al., 1992; Wiser et al., 2011). Determining the long-term success of establishment for forests is related to the extent to which species can regenerate underneath its canopy to replace the individual trees as they senesce (Grime, 2006; van der Valk et al., 1992) thereby enabling a forest to have a sustainable individuals counts. As discussed, this is largely driven by shade tolerance or light exposure and altitude, however, it is evident in this study that fencing management is the key driver.

3.4 Conclusion

My research indicates that within these ecosystems, there is still a high native vegetation presence and succession opportunities. The surveyed forests still display viable ecological integrity. However, this is dependent on the ecological pressures of the interactions between exotic and native species. NZ lowland forests support many threatened native species within many types of “at-risk” forest ecosystems. Anxiety is heightened for the future status of these species as biological invasions directly threaten native species and ecosystems (e.g., cats, rats, stoats, possums, blackberry). However, there are invasive naturalised species (most birds) that are tolerated by natives or managed in systems for recreation, food provision, or commercial endeavours (deer, thar, pigs, trout) (McGlone et al., 2020).

Monitoring vegetation dynamics improve our understanding of an ecosystem's integrity (McGlone et al., 2010). This allows for conservation opportunities surrounding biodiversity goals, detecting trends, and possible management interventions. This study's investigation of vegetation dynamics of forests is vital for the protection and enhancement initiatives for lowland forest ecological integrity.

Methods used in this study cater to repeated vegetation sampling, to determine the temporal trends in the vegetation communities (Rose, 2012). Although this research is locally focused, it contributes to the assessment of the status and trend of vegetation at higher levels, i.e. regionally, national, and international.

Future vegetation surveys must take into account that there are much more naturalised species on the edge of forest fragments, dispersed from the surrounding farmland landscape, thus it would be preferable to also put monitoring efforts into surveying the vegetation on the outskirts of the forest fragments. This would allow for better prediction of plant invasions rather than waiting for these invasions to reach the interior of the forests. In this study, the surveyed plots represent a small proportion of the site area and the surrounding pastural farmland landscape, so my findings of few naturalised plants in the plots should not be extrapolated too widely.

Fencing proved to be a protection for susceptible palatable species as it acts as a physical barrier to ungulates (and stock) from accessing the forest, which eliminates disturbances such as grazing of seedlings, allowing succession processes and species recruitment opportunities (Aviss & Roberts, 1994). This promotes forest restoration and understory recruitment for forest sustainability and integrity. In lowland Hawke’s Bay, fencing forest fragments is not only desirable for native vegetation species that reside within, but also for landowners as many wish to keep stock out of forests for better control and care. Also, many forested areas across lowland Hawke’s Bay are located across steep banks or gorges of non-productive land or semi-retired areas, and therefore fencing off is beneficial for the productiveness of the land-use business. However, fencing schemes in the Hawke’s Bay region have many challenges, with the major challenge being financial, as farm systems run as a business, and fencing native vegetation is an expenditure in both money and time, and not directly profitable to the business (Aviss & Roberts, 1994; Hashiba et al., 2014; Hawke's Bay Biodiversity, 2015; Maseyk et al., 2021; Tilman et al., 1994).

However, protecting and restoring vegetation dynamics can require more complex management interventions than fencing management, such as the reintroduction of native plant communities, pest and weed control, and advancement of networking opportunities across farmland matrix landscapes. Further investigations of factors affecting species richness of naturalised species and the hindering or enhancement of native species should be undertaken.

The overall aim of this thesis was to explore the implementation success of the new T2BMF. This chapter sought to find the factors affecting the vegetation structure, of which fencing was the most significant. Vegetation structure and composition was influenced by environmental variables, including fencing. This supports my study's overall purpose in testing the T2BMF in measuring the effectiveness and response of management of forest fragments.

Chapter 4: Birds

This chapter investigates the composition of native and naturalised bird species found within the study’s native forest fragments. In this chapter, I will describe the results of the five-minute point count and autonomous recording methods and provide comparisons between the two methods.

4.1 Introduction

The ecological roles that birds provide are critical for the health of many ecosystems (Whelan et al., 2015). Therefore, investigation of bird composition and their contribution or relevance to ecosystem services or processes and the overall integrity of an ecosystem has become immensely important. Birds are commonly used as bio-indicators to assess the quality or integrity of an ecosystem (Kułaga & Budka, 2019) as many bird species display characteristics of quick response to environmental or ecological changes (Kułaga & Budka, 2019). Birds respond to changing environments more readily as they engage in a mosaic of feeding, foraging, and associations with other organisms, making a change within that environment or ecosystem influence populations dramatically (Goodwin & Shriver, 2014).

Birds provide important ecological services within ecological integrity measures as they possess unique characteristics for a local habitat (Whelan et al., 2015). As many species of birds can fly, the ability allows for much dispersal and movement enhancing and enriching an ecosystem's diversity and providing vital functions and processes. Specifically, the relationships between vegetation and birds orientate ecosystem health, and in turn deem an ecosystem's integrity as much vegetation is dependent on birds' ecological provisions and contributions, such as dispersal mechanisms (Kearns et al., 1998; Whelan et al., 2015).

4.2 New Zealand Birds

Similar to many other oceanic islands, NZ has a long history of geographic isolation that has resulted in unique avifauna due to the absence of mammalian predators (Parlato et al., 2015; Spatz et al., 2017). Consequently, NZ native and endemic avifauna have experienced major extinction rates of 40-50 % of our total avifauna species (Parlato et al., 2015) since human settlement over 1000 years ago (Wilmshurst et al., 2008). Initially, avifauna species decline was driven by overexploitation by the early settlers for food, cultural resources, and deforestation (New Zealand Conservation Authority, 1997). With the arrival of humans to NZ, biological invasions and the colonisation of exotic species presented competition, parasites, diseases, or predation pressures causing native avifauna extinctions (Parlato et al., 2015), which continues today (Innes et al., 2010). Additionally, habitat reduction from anthropogenic activities has reduced avifauna area. This has reduced species perversion (Blake et al., 1984; Gwen Grelet et al., 2021), thus making it difficult to conserve or protect the diversity and the populations of what native avifauna is left.

Currently, different ecosystems across NZ support around 168 native bird species, of which 71 are at a high risk of extinction (Robertson et al., 2017, 2021). Many of our most threatened endemic species are large-bodied and flightless, and tend to have lower rates of population growth and generational growth (Owens & Bennett, 2000) making it difficult for many of NZ's bird species to recover from factors affecting their survival, such as habitat degradation, fragmentation, and predation.

NZ has experienced rapid landscape transformations of forest ecosystems into intensive agricultural landscapes, there is a lack of understanding of the widespread status or inventory of bird composition and dynamics nationally and regionally in forest fragments on farmland (MacLeod et al., 2008). Instead, NZ has abundant research and knowledge-based on understanding the specific ecology of targeted or interesting birds (MacLeod et al., 2008). Further, there is little knowledge on the effects of the changing environment on bird compositions or ecological changes impacting bird populations (Innes et al., 2010; MacLeod et al., 2008; Parlato et al., 2015), but it is widely accepted that NZ forest ecosystems are made up of both naturalised and native flora and fauna (Cassey, 2001; MacLeod et al., 2008; Robertson et al., 2017, 2021). Monitoring common and widespread bird species throughout forest habitats is vital as these assemblages can help maintain key ecosystem services and functions (Kelly et al., 2010; Macleod et al., 2012).

The extinction of NZ endemic birds has harmed the global genetic complement of biodiversity as it has removed a large proportion of evolutionary history and measures for ecological and economic resilience (Hoban et al., 2020). NZ conservation, therefore, has obligations to restore and protect our avifauna biodiversity (Garcia-R & Di Marco, 2020) and without further management strategies, predicted declines or extinctions in avifauna populations may result in a loss of ecosystem processes and services (Loss et al., 2015).

4.2.1 Ecological Services & Function

Birds provide significant cultural and spiritual services across many communities in NZ (New Zealand Conservation Authority, 1997) as they are many people's identity through religion, cultural customs, and sense of home or history. Our native birds are also an inspiration for much of our prized art, photography, and innovations for design, and contribute to popular outdoor recreational activities, such as bird watching and hunting (New Zealand Conservation Authority, 1997; Whelan et al., 2015).

Native birds were once part of Māori's traditional diet, with the hunting of birds being city controlled by the season, within specific tribal territories and sites (New Zealand Conservation Authority, 1997). Bird feathers provided people with bedding, insulation, and mana ornamentation (cloaks). Hunting or capture methods were derived from intimate Mātāuranaga's knowledge of the bird's ecology and the relationships they had with vegetation and the surrounding environment (New Zealand Conservation Authority, 1997).

The greatest ecological service that our native birds provide is the interactions between birds and vegetation (e.g., pollination, browsing, foraging, and seed dispersal) (Clout & Hay, 1989; Lord et al., 2009; MacLeod et al., 2008). Much of our native vegetation has evolved mutualistic or specialist relationships with bird species (Ladley & Kelly, 1996). For instance, species of native mistletoe have evolved a specialist relationship with tūī and korimako as the flowers require fine beaks to open the flower, which in turn splashes birds with pollen, to later disperse to other plants (Ladley & Kelly, 1996). Much of NZ’s flora has co-evolved flower shapes, colours, and following periods with birds' behaviour and morphological traits. For instance, Flax (*Phormium tenax*), kōwhai (*Sophora microphylla*), northern and southern rātā (*Metrosideros robusta* and *Metrosideros umbellate*), and tree fuchsia (*Fuchsia excorticata*) are all pollinated by birds with long tubed shaped flowers (Lord et al., 2009; Science Learning Hub – Pokapū Akoranga Pūtaiao, 2010, 2011).

Moreover, New Zealand’s fauna also plays a role in the pollination processes of our native vegetation offering broader pollination services due to their large-scale mobility. In New Zealand, birds are important pollinators within forest ecosystems. The honey eaters, such as tūī, korimako, and silvereyes (*Zosterops lateralis*) perform the majority of pollination by all birds in our native bush (M Clout & Hay, 1989; MacLeod et al., 2008). Exotic vegetation throughout farmland acts as a key food source for many honey eaters, such as tūī, feeding on shelter belts of *Eucalyptus species* (MacLeod et al., 2008).

The ecology of plant-bird relationships within a forest is through avian frugivory and seed dispersal. The majority of our native forest bird species many insectivores (70%) eat fruits (Clout & Hay, 1989; Lord et al., 2009). Much of NZ's native fruits are adapted for bird dispersal by several species of birds. These fruits have a protective coat that requires scarification by passing through an organism's digestive system so can only germinate with the aid of birds (Lord et al., 2009; Science Learning Hub – Pokapū Akoranga Pūtaiao, 2011). This process enables the seed to be dispersed away from the host tree's location in an attempt to colonise a new area (Lord et al., 2009; MacLeod et al., 2008). However, as a consequence of biological invasions and habitat loss, New Zealand now has very few native dispersers of large-fruited native plants following the extinction or the decline of several frugivorous species (Clout & Hay, 1989). Additionally, pollination processes are at risk of environmental and ecological degradation pressures compared to seed dispersal processes due to greater interactions with entomology, seasons, and landscape modification (Kelly et al., 2010).

Canopy-feeding birds, such as kererū and kokako (*Callaeas cinerea wilsoni*) had exerted significant browsing pressure on prehistoric forests and were capable of significant local defoliation. The decline of these bird species has also removed the browsing pressures on native flora (Clout & Hay, 1989). Removing avian herbivores' pressure changes greater ecological processes as they influence plant habitat rather than vegetation physiognomy or composition (Lee et al., 2010). Moreover, the extinction of many native birds has caused widespread ecological consequences. For example, the extinction of many native foraging birds such as the Moa *(Dinornithiformes* spp) has impacted natural ecosystem processes, such as the dispersal of ecologically important fungi (Boast et al., 2018).

4.3 Monitoring Birds

Estimates of bird abundance and richness using observational and targeted counts through acoustic surveys and sightings are fundamental for bird conservation (Shonfield & Bayne, 2017). Monitoring allows for interpretations of the changes in bird population size, gauging the environmental impact and ecological pressures experienced (Buckland et al., 2008). In NZ, the five-minute bird count is the main method for monitoring birds (Hartley, 2012), however as technological advancements develop and the urgency of monitoring bird composition or forest integrity increases, technologies and methods must progress with the times and the conservation urgency for describing how populations respond or a prediction of response to environmental changes (Kułaga & Budka, 2019)

Counts of birds that are seen, heard, or captured to provide an understanding of the relationship between a habitat and avifauna presence (Thompson, 2002). Detectability of birds is often variable between and amongst species, therefore monitoring methods should be considered based on detectability factors, such as the local ecology, the environment, and observational resources or experience (Kułaga & Budka, 2019). This enables investigation in the response of avifauna populations towards management and the environments promoting conservation interventions for targeted ecosystems, habitats, areas, avifauna, or other species. This chapter investigates two types of monitoring methods, 5MBC and Autonomous recording devices (ARD).

4.3.1 5 Minute Bird Count

The 5MBC was developed by Dawson and Bull (1975) and has been used as a point count method to estimate bird abundance and diversity since the early 1970s (Dawson & Bull, 1975; Hartley, 2012). It is now the most widely used method for measuring trends and status of bird populations within forested habitats (Greene, 2013), with versions of the point count method popular in Europe and the United States of America (Gibbons & Gregory, 2006; Johnson, 1995). The 5MBC is an established nationally accepted monitoring standard with a large body of existing data enabling researchers to compare hundreds of historical studies with their own (Greene, 2013).

The 5MBC uses point count sampling whereby an observer standing at one spot, counts all birds seen and heard across a radius of 100m within five minutes (Gibbons & Gregory, 2006; Greene, 2013; Johnson, 1995). The 5MBC is an efficient technique for assessing the presence of relatively common or vocal species as it relies on the assumption that numbers of individuals detected (seen, heard, or captured) represent a constant proportion of birds present within the area (Thompson, 2002). However, point count methods are prone to biases.

The 5MBC is less time-consuming and therefore less expensive with the five-minute time frame for detection being sufficient between the cost generated and the benefits of data gained (Bonthoux & Balent, 2012). The method is flexible yet efficient as large numbers of counts can be recorded and is less demanding than other formal estimation methods on offer (Greene, 2013). However, this approach is an index as not all recorded potential bird species or populations that are present are detected therefore can leave uncertainty about the number of bird species and populations due to coverage of surrounding vegetation, species conspicuousness, and the time of the day or season (Greene, 2013). Counts can be conducted over complex terrain and densely vegetated habitats in situations whereby movement is required for indices data collection (Greene, 2013).

The 5MBC method also has a range of indices providing calculation opportunities (e.g. mean number of birds per count proportion of counts of a species)(Greene, 2013). As this method provides relatively large bird counts, it can then reduce the impact of high variance increasing the statistical power. However, unlike the National Vegetation Survey Databank (Chapter 3), there is no singular or formal national collection of 5MBC data, making it hard for researchers to access historical data directly (Hartley, 2012).

4.3.2 Autonomous Recorders

Technological advancements have led to improved means of bird monitoring by removing human error and dependency, such as autonomous recording devices (ARD) which are now widely used sampling tools in ecological research and population monitoring (Kułaga & Budka, 2019). ARDs provide a permanent data record for reanalysis by independent observers and combined with microphone arrays, can yield spatial and behavioural information that is not available to an observer in the field (Digby et al., 2013). ARDs has the potential to expand the innovations of avifauna ecological research (and other taxa) beyond being a substitute for a human observer (Shonfield & Bayne, 2017).

Within NZ, some DOC, universities, and governmental authorities or local organisations monitoring programs use ARDs (Department of Conservation, 2013b, 2021). This relatively new technology has become increasingly available and reliable to record sound in the field. With restricted access for undertaking bird surveys, the ARDs are an attractive and agile tool because they can be deployed at a time that is convenient with the research resources or the targeted species. Devices can also be programmed to record at a suitable time for the detection of targeted species (Shonfield & Bayne, 2017) and follow the activity of seasonal and daily changes through initial manual setups of start time and end time of recording (Kułaga & Budka, 2019).

The analysis of the collected acoustic data using ARD poses several possibilities and challenges. ARDs offer unlimited point count potential, which makes density estimations problematic as the detection radius or the area which can be surveyed can vary across species, habitats, environmental factors, and the ARD models used (Yip et al., 2017). Additionally, because bird species vary in amplitudes (i.e. sound of bird calls and song), it is difficult to compare the abundance and distribution of different bird species from one ARD (Kułaga & Budka, 2019).

The initial purchase cost of ARDs, and the costs associated with ongoing maintenance; including battery replacement, and the fixing of damaged microphones is expensive in the context of other monitoring devices (Darras et al., 2019; Shonfield & Bayne, 2017). However, in the long term they are considerably cheaper in context to labour and travel costs for example fieldworkers may only need to be at the site at the start and end of a survey using ARDs rather that repeatedly over a couple of weeks using point-count methods. Using ARDs allows for intense analysis, and although this may initially seem desirable, the process of listening to or analysing recordings is expensive in time, energy, and money (Shonfield & Bayne, 2017). In addition, the sound quality from the device used by the operator to listen to the recordings (e.g. headphones, multiple listeners, repetitively) determines the efficiency of the analysing (Darras et al., 2019; Shonfield & Bayne, 2017).

4.4 Hawke’s Bay Context

The Hawke’s Bay region environment ranges from warm dry climates to winter frosts and high rainfall. This variation of environments sustains many habitats, from sheltered coastal plains to dense forests, that support many endemic, native, migratory, and exotic bird species (Hashiba et al., 2014). However, beyond recreational bird watching and historic traditional cultural cultivation, there has been little bird research in lowland Hawke’s Bay (Hashiba et al., 2014; Mclennan, 2016; River & Seaton, 2013; Worthy, 2000), primarily due to widespread farmland and small forest pockets across the landscape (MacLeod et al., 2008).

4.4.1 Cape to City

The Cape to City program is a multi-agency initiative that aims to assess the observed economic and biodiversity consequences of large-scale, low-cost predator control. in 2015 and 2016, the program was conducted across agricultural landscapes of coastal Hawke’s Bay to understand the control predators, including possums, cats, ferrets, stoats, and weasels (Mclennan, 2016). The program investigates the successfulness of predator management on bird populations by 5MBC in open farmland, in patches of both native and exotic dominated woody vegetation and wetlands across the Cape to City studied area (Mclennan, 2016).

Mclennan (2016) found that native birds were most abundant in native forests and introduced birds were most abundant in exotic forests. The variation of habitat quality between native forest types was apparent, whereby kanuka-dominated forests had fewer nectar and fruit-eating species of birds compared to the older and more diverse forests (Mclennan, 2016). It was observed that the translocations of toutouwai and tomtits in some targeted and protected forested areas were successful. The potential of dispersal for small insectivores, such as toutouwai, tomtit, and whitehead, is unlikely due to the unfavourable surrounding farmland landscape (Mclennan, 2016). Similarly, it is thought that threatened species, such as kākā and tīeke *(Philesturnus rufusater*), are also unlikely to disperse because of the restricted forest habitats throughout lowland coastal Hawke’s Bay (Mclennan, 2016).

4.5 Study

My study considers two hypotheses, following the recommended T2BMF using two 5MBC per plot and ARDS. Specifically, a comparison between the observed naturalised and native bird species occupancy abundance will be investigated against environmental variables. Environmental elements affecting detection at the species level are also investigated. My study proposes two hypothesises:

- *Hypothesises one: Fencing affects native and naturalised bird species composition in forest fragments.*
- *Hypothesises two: Autonomous recording devices will detect more bird species than five-minute bird counts.*

4.5.1 Methods

The T2BMF uses 2 5MBC conducted in the centre of each plot, carried out during fair weather conditions. Records of the time, date, and environmental plot details were noted for each count, and all species were tallied when heard or seen within a 100 m radius. No bird was knowingly counted twice within a single count (Department of Conservation, 2013b). These counts were performed by myself and a collection of field assistors; Myles Mackintosh, Grace MacGillivray, Louise MacGillivray, Lochie MacGillivray, Phoebe Laugesen, Sophie Parsons, Mark Mitchell, Francess Mitchell, Will Telford, Mathew Short, Keiko Hashiba, Anabel Beattie, Hetty McLennan, Pete Shaw, Mike Lusk, and Natalie de Burgh). The 5MBC were recorded on an iPhone and any bird species that were unidentified on the field were later identified using these recordings.

Developed by the Department of Conservation, AR3 acoustic recorders were used. The electronics team of DOC created a high-quality, robust field audio recorder called the AR3 which is specifically designed to be lightweight, weatherproof, small, and simple to use (Department of Conservation, 2013b, 2021).

In my study, each site had an AR3 mounted to a tree branch, placed 1.8–2 m up the tree, positioned to minimise any disturbance by stock or blocking of sound, and secured using a cable tie. Following a minimum two-week period, the ARDs were retrieved due to limited battery life. However, the battery life of ARD was uncertain as they're dependent on the number of recordings based on vocal species detections. Each AR3 had 2 settings programmed, low frequency (for night time recordings) and forest frequency (for daytime recordings). However, the forest recording mis-performed as the setting was unable to detect vocals of a high frequency (e.g. small high-pitched vocal birds). It is assumed that the forest setting needed to be updated or it had been altered. Further, recordings were truncated to fit the minimum time the shortest AR3 battery life lasted which was 12 days (both day and night protocols) for analysis.

Study sites

The study took place throughout the Hawke's Bay Region as illustrated in (Figure 2.1 Chapter 2), being placed in the centre of each of the studied sites.

4.5.2 Statistical analysis

All analyses were performed in R version 4.1.0 (R Core Team, 2013). This research uses a variety of R packages depending on the analysis required, which are described when applied.

To test the effects of fencing (Hypothesis 1), the 5MBC and the ARD data were pooled to create a species richness list for each site. This collection was then tested against environmental variables, including fencing management, fencing years, altitude; canopy height, ground cover, lagomorph presence, ungulate presence, longitude; latitude, site area, and predator control.

There was a much greater range of study site area (ha) for the fenced sites (with the largest area (ha) sites being fenced sites) making it difficult to include area (ha) and fencing in the same models as they tended to cancel each other out, thus within this analysis process I only used fencing (yes/no). Additionally, similar effects were happening with predator control and larger area (has) sites, which also tended to be fenced. I excluded latitude, longitude, and elevation, which simplified the models due to low levels of site replication for the hypothesis 2 analysis. Further canopy height and canopy cover values from the three plots were averaged for each site, with the fenced sites tending to be a little taller in their canopy than unfenced sites.

ARD data analysis

All recordings were processed using Kaleidoscope 5.4.7 (AcousticsWildlife, 2021) with the assistance of Laura Molles (Verum Group). Kaleidoscope searches for sounds within the user-set frequency and temporal parameters, using discrete cosine transform coefficients of spectra from sequential frames to build feature vectors representing each target sound (Canureci et al., 2021). Within the Kaleidoscope program, K-Means clustering, and Hidden Markov Models are then used to determine initial clusters among these feature vectors. This allows for the user to add manual labels that Kaleidoscope will use to refine the clusters and automatically label detections 7 (AcousticsWildlife, 2021; Canureci et al., 2021)

For this study, three separate analyses were used, each with a different set of time/frequency settings, to better detect different types of vocalisations. For two of the analyses, the search was refined using Kaleidoscope's clustering capabilities to reduce the time and effort required for confirmation of detections.

1. The first analysis searched for vocalisations between 650-3500Hz, with a duration of 5-40s and a maximum inter-syllable gap of 1.25s. Clustering used an FFT window of 10.67ms, with 12 maximum states and a maximum distance of 0.5 to the cluster centre for cluster-building. This analysis was intended primarily to detect kiwi (*Apteryx mantelli*) calls, but also detected kākā, kākāriki, and tīeke. All detections were visually checked and labelled as one of these species or as a non-target sound, with no refining of the detection algorithm.

2. The second analysis searched for vocalisations between 600-2500Hz, with a duration of 0.75-8s and a maximum inter-syllable gap of 0.75s. Clustering used an FFT window of 21.33ms, with 16 maximum states and a maximum distance of 0.5 to the cluster centre for cluster-building. This analysis was refined to search for tūī, korimako, and ruru vocalisations. In the first stage, detections with a distance <0.8 from their respective cluster centres were visually scanned; clusters, where one of the target species was predominant, were labelled accordingly. These revised labels were used to rescan the data, which created a refined classification algorithm. The resulting algorithm was then applied to the dataset and all detections classified by Kaleidoscope as one of the target species-at any distance from the cluster centre-were included in the output. All detections from night-time recordings were visually confirmed. Daytime detections were visually scanned to determine whether each of the three species was detected at least once within each one-minute daytime recording.

3. The third analysis searched for vocalisations between 3500-4000Hz, with a duration of 1-4s and a maximum inter-syllable gap of 0.5s. Clustering used an FFT window of 10.67ms, with 16 maximum states and a maximum distance of 0.5 to the cluster centre for cluster-building. The analysis was refined to search for riroriro, korimako, alarm calls, and likely toutouwai, In the first stage, detections with a distance <0.8 from their respective cluster centres were visually scanned; clusters, where one of the target species was predominant, were labelled accordingly. The revised labels were used to rescan the data, and the revised algorithm was applied to the dataset. All detections classified by Kaleidoscope as one of the target species-at any distance from the cluster centre-were included in the output. All detections from both night-time and daytime recordings were visually scanned and confirmed.

Hypothesises one: Fencing affects native and naturalised bird species composition in forest fragments.

To test whether bird community composition differed between fenced and unfenced sites, I just used the data from the 5MBC, since these included counts. Data were sorted into fencing regimes (yes/no) and the species detected were allocated a bio-status for each bird (naturalised/native). To find the difference in bird richness and abundance between fenced and unfenced sites, minimum, maximum, and mean spreads of the counts for each species' bio-status were investigated.

The variation in bird community composition among sites was described using nonmetric multidimensional scaling (NMDS) with the vegan R package (Oksanen et al., 2020) (Chapter three). To display the effects of environmental variables on bird composition, I used the envfit function within the vegan package (Oksanen et al., 2020), to create the NMDS ordinations. These graphs helped to visualise the clusters of environmental variabilities between survey and species similarities. A simpler analysis was used to determine taxa identified for driving differences among fenced or unfenced, based on relative similarity distances. Adonis was used to assess the statistical significance of the effects of the site environmental variables on bird composition. For all the PERMANOVA, a permutation of 9999 was used to calculate significant values of environmental variables.

A binomial general linear model was used to analyse how the proportion of species richness that was native was affected by the following environmental variables: mean top height, longitude, latitude, fencing, a sign of ungulates presence, predator control, area, and fencing. The R function drop1 assessed the significance of each variable after all other variables were accounted for. These analyses used the packages lme4 (Bates et al., 2015), car (Fox, John & Weisberg, 2011), and chron (James & Hornik, 2020) within R (R Core Team, 2013).

This was a smaller dataset, with only one species list per site, and so only simpler models could be fitted to the data. This dataset was retained within the model, variables of latitude, longitude, elevation, area (ha), fenced (yes/no), predator control (low, moderate, high), canopy height, and canopy cover.As there were similar effects between predator control as there is more predator control in the larger area sites, which also tend to be fenced. Further, canopy height and canopy cover values from the three plots were averaged for each site, with the fenced sites tending to be a little taller in their canopy than unfenced sites.

To detect certain bird species, such as korimako; tūī; riroriro; ruru; toutouwai; log transformations were required to meet the assumptions of normality.

Since the study design is paired, we used a linear mixed model with an area (ha) as a random effect. A Poisson model was used to analyse differences in bird song for each site explaining the total number of audio detections per site, with models being done for each of the investigated species: korimako, tūī, riroriro, toutouwai, and ruru. All data were log-transformed to meet the assumptions of normality, with site area (ha) included as a random effect. The initial analysis only used the simple fence model with the area as a random effect. The next model was another simple model with only predator control (yes/no) with the area as a random effect. Area and top height canopy were used as the third model. Finally, a full model was tested using all variable site area (ha); predator control (yes/no); top height canopy. The fencing:predator control interaction in the full model was removed when not significant at $P > 0.05$.

The models were used to explain the number of audio detections of endemic bird vocalisations from several mixed models, assessed using AICc. Compared were linear mixed models with the area as a random effect, with the simplest model being fencing (yes/no); then predator control; area and mean top canopy height; and finally the full model of all the environmental variables. The fencing:predator control interaction in the full model was dropped when not significant. Fencing (yes/no) was a simpler model compared to predator control as there were different intensities of predator control and this control varied regardless of fencing (yes/no) across sites.

The Analysis of Deviance tables for these mixed models was calculated using Type II Wald chi-square tests, which means that each P-value is calculated after all the other effects are assessed first. The marginal and conditional R^2 values were calculated with the r.squaredGLMM function. Within the analysis, R^2m is the marginal R-squared, which is the effect of the fixed effect(s) and R^2m is the conditional R^2, which is the fixed effect(s) and the random effect (area). The models were compared with AICc to see which is the better model with the data as estimates the expected discrepancy with less bias using a small-sample application.

Hypothesis two: Autonomous recording devices will detect more bird species than five-minute bird counts.

Manual edits were necessary for the analysis as the Kaleidoscope detections fell short for some detection measures due to the ARD settings (forest and high). Within the Kaleidoscope algorithm, any for kākā and toutouwai detected with an uncertain identification were classified as 'maybe' and were manually checked. I listened to the complete recordings from 7 am until 9 am over the 12 days for kākā and toutouwai only to check for any recordings of these species that Kaleidoscope might have missed. I focused on toutouwai and kākā because they are good indicators of a healthy native forest, plus some of the higher frequency vocalisations of toutouwai were difficult for Kaleidoscope to detect from the frequency-truncated AR3 recordings, and kākā has highly variable vocalisations.

I collated the confirmed species detections from the Kaleidoscope and the 5MBC species detected for each of the sites and produced a table. Proportions of species detectability were calculated in comparison between bird monitoring methods (total species detected/5MBC or ARD species detected) and proportions of species detected between fencing management (total species detected and fencing (yes/no)).

4.6 Results

4.6.1 Hypothesises one: Fencing affects native and naturalised bird species composition in forest fragments.

The mean species richness within fenced sites was 8.2 and 7.3 within unfenced sites. For native species, the mean was 1.1 in fenced and 0.9 in unfenced sites. Similarly, for endemic species, the mean was 4.7 for fenced and 3.7 for unfenced sites. Further, in opposing response, mean for naturalised species richness was 1.1 in fenced and 3 in unfenced sites.

Fencing significantly affected the composition of both native and naturalised birds in forest fragments using the 5MBC data (Table 4.1, Figure 4.1, Figure 4.2), although this effect explained little of the variation in composition (adonis R^2 0.03 for native birds and 0.05 for naturalised birds, Table 4.1). The stronger significant effects on native bird composition, in order of decreasing adonis R^2, were site longitude, altitude, lagomorph presence, area, latitude, and predator control (Table 4.1, Figure 4.1). Native bird composition was also significantly affected by soil PC1 and ungulates (Table 4.1, Figure 4.1). For naturalised bird composition, the stronger significant effects, in order of decreasing adonis R^2, were latitude and predator control. Naturalised bird composition was also significantly affected by longitude, altitude, area, canopy height, cover PC1, soil PC1, and ungulates (Table 4.1, Figure 4.1).

While bird composition differed between fenced and unfenced sites, no significant fencing effects were detected in species richness of native or naturalised birds (Figure 4.2, Table 4.2), nor the proportion of native to naturalised species. For native bird species, when the model selection was applied and all models with delta AICc ≥4 were removed, only site elevation, area, and predator control remained (Table 4.2). The richness of native birds was significantly positively affected by site area and predator control (Table 4.2). Note that the larger sites were more likely to have predator control, so the effects of these variables are partially confounded. For naturalised bird richness, all variables were retained in the averaged subset model but no variables were statistically significant and the model with the lowers AICc was the NULL model (Table 4.2). None of the measured site environmental variables had a strong effect on naturalised bird richness.

I used the ARD recordings to look closer at four species, korimako, tūī, riroriro, toutouwai, and ruru (Table 4.3). The interpretation of these findings acknowledges that the fenced effect has larger sites (see Chapter 2), which may itself affect habitat use by the birds.

There were more korimako detections in fenced sites than in unfenced sites (Table 4.3), and fencing alone was the best model for explaining the number of korimako detections (Table 4.5), although the model with just predator control as a variable was within 1 AICc value. In sites with predator control and no fencing, detections were still greater than in sites that were fenced with no predator control (Table 4.2). Detections were lowest in sites that were unfenced with no predator control (Table 4.3, 4.5).

There were more tūī detections in fenced sites than in unfenced sites (Table 4.3), and the best model explaining tūī detections was only fencing (the model with just predator control had an AICc 2.52 greater)(Table 4.2, Table 4.4). It was found that there were greater tūī detections in predator-controlled sites and, somewhat unexpectedly, fewer tūī in the sites with a higher mean top height (Table 4.3).

There were fewer riroriro detections in fenced sites than in unfenced sites and there was no discernible effect on the detection of this species between different predator control methods (Table 4.3). However, there appears to be a contradiction between the fencing regime and predator control, whereby lower detections were occurring in fenced sites with predator control, but the same effect was seen with no predator control and no fencing (Table 4.3). The results suggest that riroriro detection occurred at intermediate sites, such as those fenced or with predator control but not the combination (Table 4.4C).

There were relatively few detections of toutouwai (Table 4.3) with no more detection between unfenced than fenced sites. However, there is close to a significant interaction between fencing and predator control, whereby toutouwai numbers, after area effects are accounted for, were detected more in sites that are either fenced or predator controlled, but not both (Table 4.4D).

There were more ruru detections in fenced sites than unfenced sites and more detections in predator-controlled sites (Table 4.3). The fenced effect alone and predator control alone models are about equally strong (<2 AIC between them), meaning that the data alone cannot determine what effect is strongest (Table 4.2, Table 4.4E).

Results for this chapter investigated a variety of conditions and components, thus a few significant effect results may be emulated by chance.

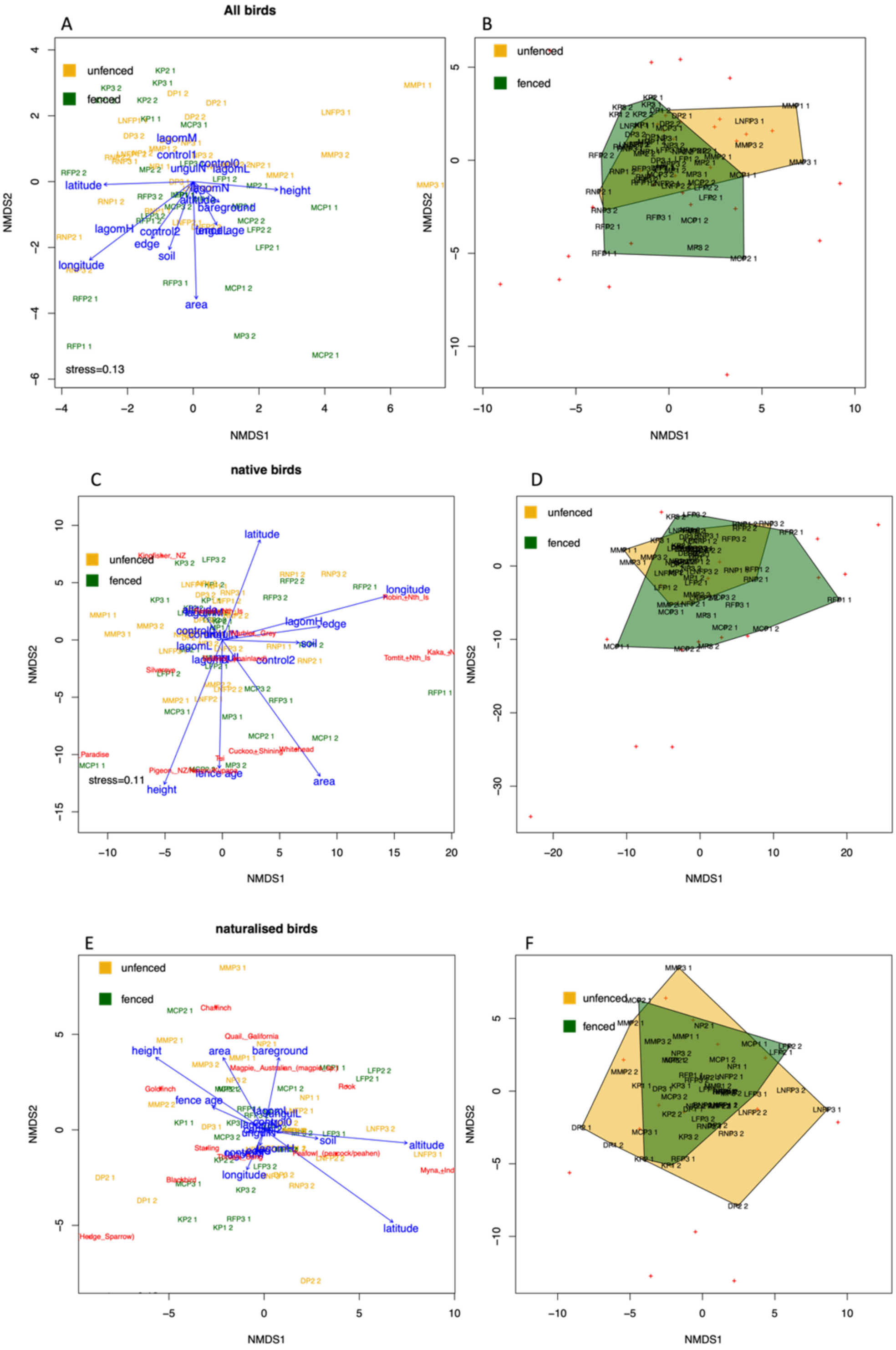


Figure 4.1: Figure 4.1: (A, C, E) NMDS ordination plots of fenced and unfenced site plots displaying similarities and the ratio of environmental variables affecting the bird species composition. Plot IDs in green are fenced sites and plot IDs in yellow are unfenced sites. Figure 1: (B, D, F) NMDS ordination graph displaying the overall similarity of plot compositions between fenced and unfenced sites of adult tree composition. Fenced sites are in green and unfenced sites are in yellow.

Table 4.1: **PERMANOVA for the effects of environmental variables on bird species composition. (A) all bird species, (B) native bird species, (C) naturalised bird species**

All birds:	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.7475	0.0836	8.5522	1E-04***
scale(Northing.Latitude)	1	0.6397	0.0716	7.3192	1E-04***
scale(Altitude.m)	1	0.5957	0.0666	6.8156	1E-04***
scale(Aspect)	1	0.1369	0.0153	1.5668	0.1196
scale(Area..ha.)	1	0.4266	0.0477	4.8807	1E-04***
scale(Plot.dist.edge.m)	1	0.1639	0.0183	1.8753	0.0546.
scale(Mean.Top.Height.m)	1	0.2885	0.0323	3.3012	6E-04***
scale(Fenced.years)	1	0.3447	0.0386	3.9441	1E-04***
scale(cover_PC1)	1	0.1744	0.0195	1.9958	0.0416*
scale(cover_PC2)	1	0.1425	0.0159	1.6302	0.1087
scale(cover_PC3)	1	0.1425	0.0159	1.6306	0.1107
scale(soil_PC1)	1	0.2762	0.0309	3.1605	0.0014**
scale(soil_PC2)	1	0.0916	0.0102	1.0479	0.3991
Lagomorphs	3	0.5621	0.0629	2.1437	9E-04***
Ungulates	1	0.2157	0.0241	2.4675	0.0108*
predatorcontrol	2	0.4944	0.0553	2.8283	3E-04***
Residual	40	3.4963	0.3911	NA	NA
Total	59	8.9395	1.0000	NA	NA

Native:	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.7267	0.0944	9.3087	1E-04***
scale(Northing.Latitude)	1	0.4533	0.0589	5.8065	1E-04***
scale(Altitude.m)	1	0.5597	0.0727	7.1688	1E-04***
scale(Aspect)	1	0.0936	0.0122	1.1988	0.325
scale(Area..ha.)	1	0.4685	0.0609	6.0008	1E-04***
scale(Plot.dist.edge.m)	1	0.0849	0.0110	1.0881	0.3813
scale(Mean.Top.Height.m)	1	0.1433	0.0186	1.8360	0.0942
scale(Fenced.years)	1	0.2637	0.0343	3.3774	0.003
scale(cover_PC1)	1	0.1063	0.0138	1.3610	0.2287**
scale(cover_PC2)	1	0.0934	0.0121	1.1958	0.3124
scale(cover_PC3)	1	0.1418	0.0184	1.8163	0.097.
scale(soil_PC1)	1	0.2502	0.0325	3.2052	0.0042**
scale(soil_PC2)	1	0.0864	0.0112	1.1065	0.3652
Lagomorphs	3	0.5478	0.0712	2.3391	0.0019**
Ungulates	1	0.1894	0.0246	2.4256	0.0258*
predatorcontrol	2	0.3649	0.0474	2.3373	0.0063**
Residual	40	3.1227	0.4057	NA	NA
Total	59	7.6966	1.0000	NA	NA

Naturalised:	Df	SumOfSqs	R2	F	Pr(>F)
scale(Easting..Longitude)	1	0.4933	0.0413	3.5898	0.0072**
scale(Northing.Latitude)	1	0.9388	0.0786	6.8321	2E-04***
scale(Altitude.m)	1	0.3529	0.0295	2.5680	0.0314*
scale(Aspect)	1	0.1987	0.0166	1.4459	0.2005
scale(Area..ha.)	1	0.4041	0.0338	2.9406	0.0143*
scale(Plot.dist.edge.m)	1	0.2363	0.0198	1.7197	0.1304
scale(Mean.Top.Height.m)	1	0.5007	0.0419	3.6438	0.0045**
scale(Fenced.years)	1	0.6264	0.0524	4.5585	0.0012**
scale(cover_PC1)	1	0.4905	0.0410	3.5700	0.0059**
scale(cover_PC2)	1	0.2677	0.0224	1.9480	0.0885.
scale(cover_PC3)	1	0.1055	0.0088	0.7677	0.5831*
scale(soil_PC1)	1	0.3301	0.0276	2.4020	0.0356*
scale(soil_PC2)	1	0.1040	0.0087	0.7566	0.59
Lagomorphs	3	0.7205	0.0603	1.7478	0.0501.
Ungulates	1	0.5799	0.0485	4.2206	0.002**
predatorcontrol	2	0.6548	0.0548	2.3827	0.0135*
Residual	36	4.9467	0.4139	NA	NA
Total	55	11.9508	1.0000	NA	NA

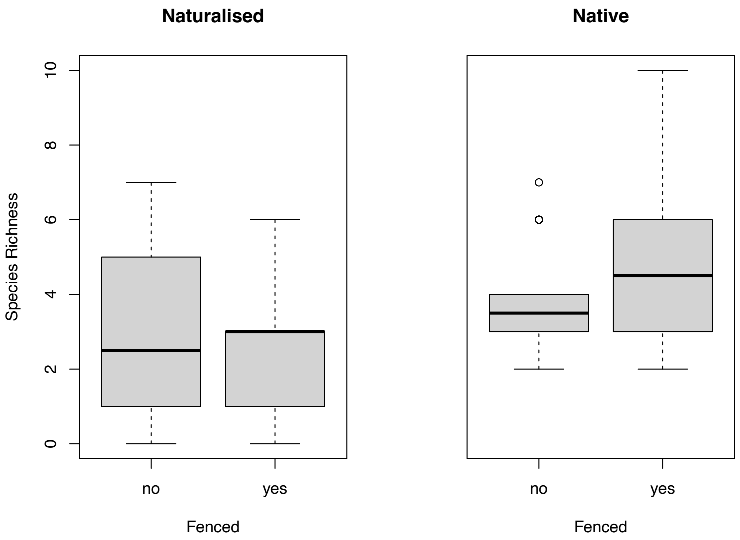


Figure 4.2: The effect of fencing regimes (yes/no) on naturalised (left) and native(right) bird species richness. The median is represented by the horizontal black line, the grey box spans the first and third quartile, and the whiskers extend up to 1.5 times the interquartile range with circles representing outliers. In total, there were 7 naturalised species and 10 native species (8 endemic and 2 indigenous). While there were significant differences in bird composition between fenced and unfenced sites, the differences in species richness was not significant (Table 4.2).

Table 4.2: The conditional model-averaged coefficients for the best models, within 4 AICc of the top model, of all subset models from the full model, species richness per site ~ latitude + longitude + elevation + area (ha) + fenced (yes/no) + predator control (low, moderate, high). The conditional averages are for just the models which contain each variable.

Native species richness					
	Estimate	Std. Error	Adjusted SE	z value	Pr(>|z|)
(Intercept)	2.27	0.31	0.34	6.74	<2E-016
Area (ha)	0.03	0.01	0.01	2.59	0.01
Elevation	0	0	0	1.74	0.08
Predator control L	0.38	0.13	0.15	2.52	0.01
Predator control Q	0.25	0.18	0.22	1.13	0.26

Naturalised species richness					
	Estimate	Std. Error	Adjusted SE	z value	Pr(>|z|)
(Intercept)	0.73	33.69	39.59	0.02	0.99
Elevation	0	0	0	0.88	0.38
Fenced (yes/no)	-0.13	0.25	0.29	0.43	0.67
Area (ha)	0	0.01	0.02	0.3	0.77
Latitude	-0.05	0.33	0.39	0.12	0.9
Longitude	0.06	0.62	0.73	0.08	0.93

Table 4.3: : Total ARD detections and environmental variables used for the models for each of the study sites. Paired sites are grouped by location, and separated by fencing regimes. Total detections are derived from the Kaleidoscope algorithmic process and additional manual edits for detections for korimako, tūī, riroriro, toutouwai, and ruru only. The total ARD detections for each of the sites, species type, fencing regime are displayed with the combined totals.

Total ARD detections:											
Sites	**Korimako**	**Tūi**	**Riroriro**	**Toutouwai**	**Ruru**	**Total ARD detections**	**Fenced (Y/N)**	**Predator control (Y/N)**	**Area (ha)**	**Top canopy height (m)**	**Location**
Dave	68	55	599	17	76	**815**	N	N	4.07	11	Raukawa
Kirk	416	657	163	10	192	**1438**	Y	Y	5.49	12	Raukawa
Livesey Nonfenced	285	13	39	4	8	**349**	N	N	2.25	8	Waiwhare
Livesey Fenced	393	64	125	17	11	**610**	Y	N	3.92	9	Waiwhare
Motumokai	483	890	79	6	67	**1525**	N	N	12.26	10	Pōrangahau
McLeans	610	888	72	17	238	**1825**	Y	Y	34.19	17	Pōrangahau
Neil	187	68	186	1	38	**480**	N	N	11.12	10	Glengarry
Mark	193	2	167	1	31	**394**	Y	Y	12.36	11	Glengarry
RoughBlock Nonfenced	394	10	192	147	109	**852**	N	Y	18.89	7	Cape Sanctuary
RoughBlock Fenced	964	53	57	86	164	**1324**	Y	Y	24.00	8	Cape Sanctuary
Fenced site detection total	**2576**	**1664**	**584**	**131**	**636**	**5591**					
Unfenced site detection total	**1417**	**1036**	**1095**	**175**	**298**	**4021**					
Total detections:	**3993**	**2700**	**1679**	**306**	**934**	**9612**					

Table 4.4: The top models for explaining the number of audio detections of endemic bird vocalisations from several mixed models, assessed using AICc.

A Korimako AICc models

Model:	AICc model value
AICc(fencemod)	**35.078**
AICc(predmod)	**35.519**
AICc(areamod)	45.154
AICc(fullmod)	58.542

B Tui AICc models

Model:	AICc model value
AICc(fencemod)	**48.622**
AICc(predmod)	51.138
AICc(areamod)	64.495
AICc(fullmod)	65.962

C Riroriro AICc models

Model:	AICc model value
AICc(fencemod)	**38.279**
AICc(predmod)	**38.314**
AICc(areamod)	52.746
AICc(fullmod)	87.243

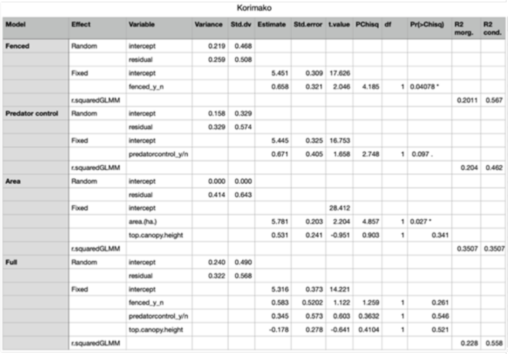
D Toutouwai AICc models

Model:	AICc model value
AICc(fencemod)	**45.342**
AICc(predmod)	**45.034**
AICc(areamod)	51.785
AICc(fullmod)	62.533

E Ruru AICc models

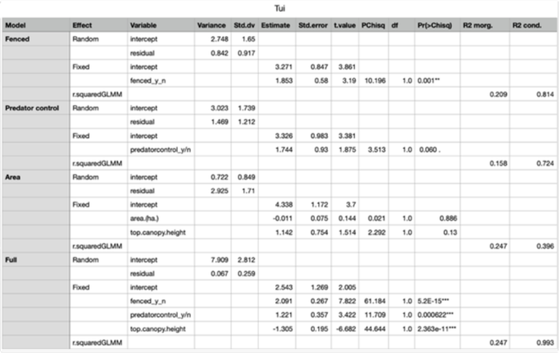
Model:	AICc model value
AICc(fencemod)	**38.647**
AICc(predmod)	**37.191**
AICc(areamod)	48.343
AICc(fullmod)	59.485

Table 4.5: Mixed models explaining the total number of audio detections per site for korimako, tūī, riroriro, toutouwai, and ruru. Total detections were log-transformed and the area was included as a random effect. The Analysis of Deviance table was calculated using Type II Wald chi-square tests. The marginal and conditional R^2 values were calculated with the r.squaredGLMM function. See Methods for details. The fencing:predator control interaction in the full model was dropped when not significant.



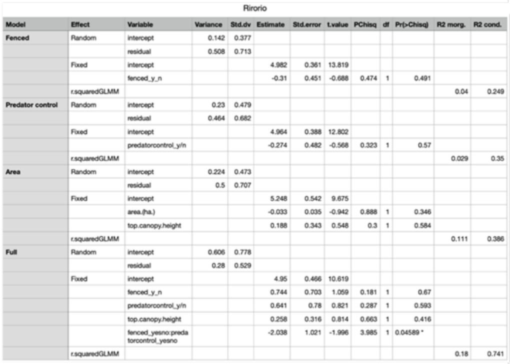
Korimako

Model	Effect	Variable	Variance	Std.dv	Estimate	Std.error	t.value	PChisq	df	Pr(>Chisq)	R2 morg.	R2 cond.
Fenced	Random	intercept	0.219	0.468								
		residual	0.259	0.508								
	Fixed	intercept			5.451	0.309	17.626					
		fenced_y_n			0.658	0.321	2.046	4.185	1	0.04078 *		
	r.squaredGLMM										0.2011	0.567
Predator control	Random	intercept	0.158	0.329								
		residual	0.329	0.574								
	Fixed	intercept			5.445	0.325	16.753					
		predatorcontrol_y/n			0.671	0.405	1.658	2.748	1	0.097 .		
	r.squaredGLMM										0.204	0.462
Area	Random	intercept	0.000	0.000								
		residual	0.414	0.643								
	Fixed	intercept					28.412					
		area.(ha.)			5.781	0.203	2.204	4.857	1	0.027 *		
		top.canopy.height			0.531	0.241	-0.951	0.903	1	0.341		
	r.squaredGLMM										0.3507	0.3507
Full	Random	intercept	0.240	0.490								
		residual	0.322	0.568								
	Fixed	intercept			5.316	0.373	14.221					
		fenced_y_n			0.583	0.5202	1.122	1.259	1	0.261		
		predatorcontrol_y/n			0.345	0.573	0.603	0.3632	1	0.546		
		top.canopy.height			-0.178	0.278	-0.641	0.4104	1	0.521		
	r.squaredGLMM										0.228	0.558



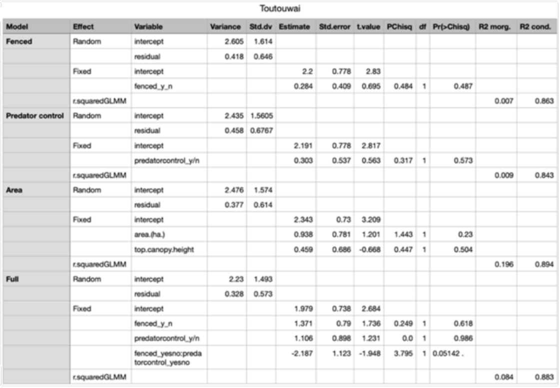
Tui

Model	Effect	Variable	Variance	Std.dv	Estimate	Std.error	t.value	PChisq	df	Pr(>Chisq)	R2 morg.	R2 cond.
Fenced	Random	intercept	2.748	1.65								
		residual	0.842	0.917								
	Fixed	intercept			3.271	0.847	3.861					
		fenced_y_n			1.853	0.58	3.19	10.196	1.0	0.001**		
	r.squaredGLMM										0.209	0.814
Predator control	Random	intercept	3.023	1.739								
		residual	1.469	1.212								
	Fixed	intercept			3.326	0.983	3.381					
		predatorcontrol_y/n			1.744	0.93	1.875	3.513	1.0	0.060 .		
	r.squaredGLMM										0.158	0.724
Area	Random	intercept	0.722	0.849								
		residual	2.925	1.71								
	Fixed	intercept			4.338	1.172	3.7					
		area.(ha.)			-0.011	0.075	0.144	0.021	1.0	0.886		
		top.canopy.height			1.142	0.754	1.514	2.292	1.0	0.13		
	r.squaredGLMM										0.247	0.396
Full	Random	intercept	7.909	2.812								
		residual	0.067	0.259								
	Fixed	intercept			2.543	1.269	2.005					
		fenced_y_n			2.091	0.267	7.822	61.184	1.0	5.2E-15***		
		predatorcontrol_y/n			1.221	0.357	3.422	11.709	1.0	0.000622***		
		top.canopy.height			-1.305	0.195	-6.682	44.644	1.0	2.363e-11***		
	r.squaredGLMM										0.247	0.993



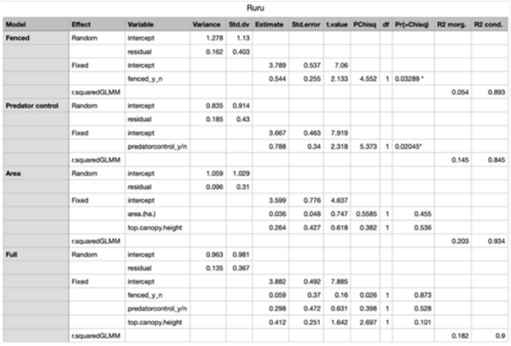
Rirorio

Model	Effect	Variable	Variance	Std.dv	Estimate	Std.error	t.value	PChisq	df	Pr(>Chisq)	R2 morg.	R2 cond.
Fenced	Random	intercept	0.142	0.377								
		residual	0.508	0.713								
	Fixed	intercept			4.982	0.361	13.819					
		fenced_y_n			-0.31	0.451	-0.688	0.474	1	0.491		
	r.squaredGLMM										0.04	0.249
Predator control	Random	intercept	0.23	0.479								
		residual	0.464	0.682								
	Fixed	intercept			4.964	0.388	12.802					
		predatorcontrol_y/n			-0.274	0.482	-0.568	0.323	1	0.57		
	r.squaredGLMM										0.029	0.35
Area	Random	intercept	0.224	0.473								
		residual	0.5	0.707								
	Fixed	intercept			5.248	0.542	9.675					
		area.(ha.)			-0.033	0.035	-0.942	0.888	1	0.346		
		top.canopy.height			0.188	0.343	0.548	0.3	1	0.584		
	r.squaredGLMM										0.111	0.386
Full	Random	intercept	0.606	0.778								
		residual	0.28	0.529								
	Fixed	intercept			4.95	0.466	10.619					
		fenced_y_n			0.744	0.703	1.059	0.181	1	0.67		
		predatorcontrol_y/n			0.641	0.78	0.821	0.287	1	0.593		
		top.canopy.height			0.258	0.316	0.814	0.663	1	0.416		
		fenced_yesno:predatorcontrol_yesno			-2.038	1.021	-1.996	3.985	1	0.04589 *		
	r.squaredGLMM										0.18	0.741



Toutouwai

Model	Effect	Variable	Variance	Std.dv	Estimate	Std.error	t.value	PChisq	df	Pr(>Chisq)	R2 morg.	R2 cond.
Fenced	Random	intercept	2.605	1.614								
		residual	0.418	0.646								
	Fixed	intercept			2.2	0.778	2.83					
		fenced_y_n			0.284	0.409	0.695	0.484	1	0.487		
	r.squaredGLMM										0.007	0.863
Predator control	Random	intercept	2.435	1.5605								
		residual	0.458	0.6767								
	Fixed	intercept			2.191	0.778	2.817					
		predatorcontrol_y/n			0.303	0.537	0.563	0.317	1	0.573		
	r.squaredGLMM										0.009	0.843
Area	Random	intercept	2.476	1.574								
		residual	0.377	0.614								
	Fixed	intercept			2.343	0.73	3.209					
		area.(ha.)			0.938	0.781	1.201	1.443	1	0.23		
		top.canopy.height			0.459	0.686	-0.668	0.447	1	0.504		
	r.squaredGLMM										0.196	0.894
Full	Random	intercept	2.23	1.493								
		residual	0.328	0.573								
	Fixed	intercept			1.979	0.738	2.684					
		fenced_y_n			1.371	0.79	1.736	0.249	1	0.618		
		predatorcontrol_y/n			1.106	0.898	1.231	0.0	1	0.986		
		fenced_yesno:predatorcontrol_yesno			-2.187	1.123	-1.948	3.795	1	0.05142 .		
	r.squaredGLMM										0.084	0.883



Ruru

Model	Effect	Variable	Variance	Std.dv	Estimate	Std.error	t.value	PChisq	df	Pr(>Chisq)	R2 morg.	R2 cond.
Fenced	Random	intercept	1.278	1.13								
		residual	0.162	0.403								
	Fixed	intercept			3.789	0.537	7.06					
		fenced_y_n			0.544	0.255	2.133	4.552	1	0.03289 *		
	r.squaredGLMM										0.054	0.893
Predator control	Random	intercept	0.835	0.914								
		residual	0.185	0.43								
	Fixed	intercept			3.667	0.463	7.919					
		predatorcontrol_y/n			0.788	0.34	2.318	5.373	1	0.02045*		
	r.squaredGLMM										0.145	0.845
Area	Random	intercept	1.059	1.029								
		residual	0.096	0.31								
	Fixed	intercept			3.599	0.776	4.637					
		area.(ha.)			0.036	0.048	0.747	0.5585	1	0.455		
		top.canopy.height			0.264	0.427	0.618	0.382	1	0.536		
	r.squaredGLMM										0.203	0.934
Full	Random	intercept	0.963	0.981								
		residual	0.135	0.367								
	Fixed	intercept			3.882	0.492	7.885					
		fenced_y_n			0.059	0.37	0.16	0.026	1	0.873		
		predatorcontrol_y/n			0.298	0.472	0.631	0.398	1	0.528		
		top.canopy.height			0.412	0.251	1.642	2.697	1	0.101		
	r.squaredGLMM										0.182	0.9

4.6.2 Audio picks up larger species richness and abundance over time compared to traditional 5MBC

ARD detected species at sites more consistently than the traditional 5MBC method. Only 12 bird species of the total detected 36 species were focused on when analysing the autonomous recordings, due to limitations in bird training sets available and the frequency band recorded by the recorder with the settings that were used.

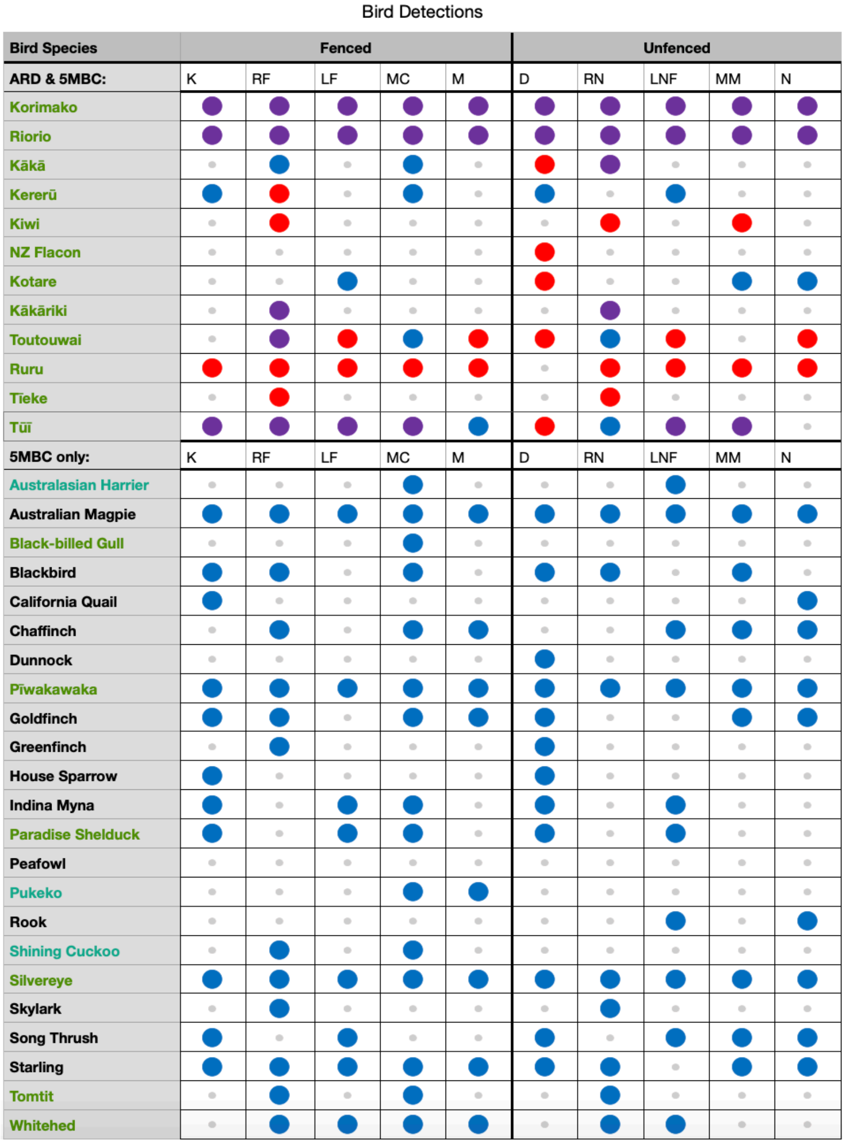
Of the total 36 species observed, 17 were endemic, three were native and 14 were naturalised (Table 4.6). The combined counts for both methods of detection (Table 4.6), there were 360 detection possibilities (36 species found over 10 sites). The detection methods of those 360 individuals showed that of these, 32.2% (116/360) were detected using the 5MBC methods, and 6.6% (24/360) used the ARD methods. In addition, 8.3% (30/360) detections were made using both ARD and 5MBC, with 52.7% (190/360) non-detections (species-site combinations absent from the data).

These 12 endemic species were surveyed over 10 sites creating 120 species-site detection combination possibilities. Of this, 10.8% (13/120) were only detected by 5MBC while 20% (24/120) were only detected by autonomous recordings. A further 25% (30/120) species were detected by both methods, and 44.2% (53/120) of possible species-site combinations were not detected by either method (likely an indication these species were absent from these sites during the surveyed period) (Table 4.6).

Between the detections for both ARD and 5MBC for fenced sites, the proportion of overall species detection with 60 detection possibilities across 5 fenced sites with 12 species, there were 16.6% (10/60) detections found only by ARD and 11.2% (7/60) found only using 5MBC and with 25.6% (16/60) detections for both 5MBC and ARD. There was 43.2% (27/60) no detection (Table 4.6). Comparatively, in the proportion of overall species detections for unfenced sites, there were detections of 43.3% (26/60), with detections of 23.3% (14/60) for both 5MBC and ARD. There were also only 23.3% (14/60) detections found ARD and 10% (6/60) found using 5MBC (Table 4.6).

The manual checking of recordings between 7 am and 9 am for toutouwai kaka indicates that Kaleidoscope was reliable at detecting kaka but missed some toutouwai (as was expected given the frequencies of the recordings). The proportion of missed detections from Kaleidoscope for kaka (these were programmed as maybes) was 0% over 12 days for all 10 sites (Table 4.6). The proportion of missed detections of toutouwai from Kaleidoscope was 37% between 7 am and 9 am over 12 days. While the 'maybes' derived and manually edited from Kaleidoscope were predominately correct, there were 7 recordings labelled as 'maybes' that did not sound like toutouwai when manually checked.

Table 4.6: Bird species detected throughout surveys across study sites. The table is separated into fenced and unfenced sites with bird species on the left column. The table is split based on the method (5MBC and ARD or 5MBC only) of detection used. There are 360 detection possibilities in total and 120 detection possibilities integrating ARDs.



Bird Detections

Bird Species	Fenced					Unfenced				
ARD & 5MBC:	K	RF	LF	MC	M	D	RN	LNF	MM	N
Korimako	🟣	🟣	🟣	🟣	🟣	🟣	🟣	🟣	🟣	🟣
Riorio	🟣	🟣	🟣	🟣	🟣	🟣	🟣	🟣	🟣	🟣
Kākā	·	🔵	·	🔵	·	🔴	🟣	·	·	·
Kererū	🔵	🔴	·	🔵	·	🔵	·	🔵	·	·
Kiwi	·	🔴	·	·	·	·	🔴	·	🔴	·
NZ Flacon	·	·	·	·	·	🔴	·	·	·	·
Kotare	·	·	🔵	·	·	🔴	·	·	🔵	🔵
Kākāriki	·	🟣	·	·	·	·	🟣	·	·	·
Toutouwai	·	🟣	🔴	🔵	🔴	🔴	🔵	🔴	·	🔴
Ruru	🔴	🔴	🔴	🔴	🔴	·	🔴	🔴	🔴	🔴
Tieke	·	🔴	·	·	·	·	🔴	·	·	·
Tūī	🟣	🟣	🟣	🟣	🔵	🔴	🔵	🟣	🟣	·
5MBC only:	K	RF	LF	MC	M	D	RN	LNF	MM	N
Australasian Harrier	·	·	·	🔵	·	·	·	🔵	·	·
Australian Magpie	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵
Black-billed Gull	·	·	·	🔵	·	·	·	·	·	·
Blackbird	🔵	🔵	·	🔵	·	🔵	🔵	·	🔵	·
California Quail	🔵	·	·	·	·	·	·	·	·	🔵
Chaffinch	·	🔵	·	🔵	🔵	·	·	🔵	🔵	🔵
Dunnock	·	·	·	·	·	🔵	·	·	·	·
Pīwakawaka	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵
Goldfinch	🔵	🔵	·	🔵	🔵	🔵	·	·	🔵	🔵
Greenfinch	·	🔵	·	·	·	🔵	·	·	·	·
House Sparrow	🔵	·	·	·	·	🔵	·	·	·	·
Indina Myna	🔵	·	🔵	🔵	·	🔵	·	🔵	·	·
Paradise Shelduck	🔵	·	🔵	🔵	·	🔵	·	🔵	·	·
Peafowl	·	·	·	·	·	·	·	·	·	·
Pukeko	·	·	·	🔵	🔵	·	·	·	·	·
Rook	·	·	·	·	·	·	·	🔵	·	🔵
Shining Cuckoo	·	🔵	·	🔵	·	·	·	·	·	·
Silvereye	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵	🔵
Skylark	·	🔵	·	·	·	·	🔵	·	·	·
Song Thrush	🔵	·	🔵	·	·	🔵	·	🔵	🔵	🔵
Starling	🔵	🔵	🔵	🔵	🔵	🔵	🔵	·	🔵	🔵
Tomtit	·	🔵	·	🔵	·	·	🔵	·	·	·
Whitehed	·	🔵	🔵	🔵	🔵	·	🔵	🔵	·	·

Legend:

No detection	·
Detection by 5MBC only	🔵
Detection by autonomous recorder only	🔴
Detection by both 5MBC & autonomous recorder	🟣
Naturalised / Introduced	
Native	
Endemic	

4.7 Discussion

4.7.1 Fencing determines species composition whereby there will be more native species in fenced sites and more naturalised species in unfenced sites

My study investigated the indirect and direct effects of fencing management on species composition, for native and naturalised bird species. The bird species composition was determined by the fencing management with species richness difference between fencing management based on bio status. Detection proportions between native and naturalised species across fencing management differed greatly whereby the overall species richness was constant across sites, but with naturalised species richness being greater in unfenced sites, while native species richness was greater in fenced sites. Although native species richness was constant between fencing managements, the abundance of native species detected significantly increased in fencing sites. Influential papers, Barnagaud et al. (2014), Cassey (2001), and MacLeod et al. (2009), shared similar findings whereby observations of bird species composition in response to fencing regimes in forests differed between native and naturalised avifauna.

The integrity of native vegetation and regeneration succession is fundamentally and directly related to fencing regimes (refer to Chapter 3) and is also therefore directly related to native bird species abundance and richness (Parker et al., 2022; Tanentzap & Lloyd, 2017) as my study suggests. Parker et al (2022) determined that fencing management influenced the population densities of several bird species as fencing around a forested area is an attractive habitat for many forest-dwelling songbird species as the exclusion of ungulate species through fencing, created better vegetation conditions making greater foraging availability for bird species. Likewise, within my study, bird species composition within the forests varied in response to fencing management, whereby fenced sites contained good vegetation integrity, the native bird richness and abundance were high, and correspondingly unfenced sites had poor vegetation integrity, the native bird species richness and abundance was low.

Blake et al. (1984) and Barbaro et al. (2012) found that declines in bird abundance are aligned with the reduction in size and isolation – fragmentation of forests as it decreases the frequency or magnitude of ecological processes, such as species interactions, connectivity, and dispersal opportunities, and increases edge effects (Barbaro et al., 2012; Parker et al., 2022). Consequently, fragmentation of forests often leads to the immediate loss of mobile species (birds) due to limitations for ecological processes such as dispersal, resource, food availability, and regeneration routine (Blake et al., 1984). My study witnesses these effects whereby there is less bird diversity in smaller and isolated forests. However, Ladin et al. (2016) found that bird composition is also largely driven by the surrounding regional land cover patterns, whereby there was a less varied composition in areas of extensively developed land cover (Ladin et al., 2016). Further investigation of the landscape matrix level is necessary to understand the spatial and temporal patterns and trends in bird composition in Hawke's Bay region.

The area (ha) of study sites was the most significant environmental variable on bird species composition, for both native and naturalised species. It is assumed that the bird composition between sites is dependent species' diet and habitat preference, which is ultimately the size of the forest and thus the vegetation dynamics catering (or not) to the bird's needs (Cassey, 2001;Clout & Hay, 1989; MacLeod et al., 2008; Parker et al., 2022; Robertson et al., 2017, 2021). Local vegetation dynamics varied based on a range of environmental variables (refer to Chapter 3) and significance differed across sites, but the area (ha) had a constant effect. Vegetation forms fundamental components sustaining terrestrial avian habitats and provides cues that guide habitat selection for a range of species, resources for food, and substrates used for shelter and foraging (Lee & Rotenberry, 2005), for both native and exotic bird species. Ladin et al. (2016) share findings similar to my study, in sites with greater area, there was a positive relationship with bird composition meaning that the larger the forest, the greater the bird species richness and abundance in areas it can support greater species compositions. The larger the area, the greater potential of vegetation compositions, thus as the area of a forest increase, biodiversity also increase (Cassey, 2001; Clout & Hay, 1989; MacLeod et al., 2008; Parker et al., 2022; Robertson et al., 2017, 2021).

As discussed (Barbaro et al., 2012, 2012; Burns et al., 2011; Parker et al., 2022; Penariol & Madi-Ravazzi, 2013a; van Heezik et al., 2008; Young & Mitchell, 1994), both large and small forest areas can support a range of bird compositions due to difference of habitat resource accessibility, preference, and traits because of edge effects and interior preference. Bird composition in response to forest area (ha) was different based on their bio status of native or exotic origin, which is directly linked to life histories and habitat presences (Cassey, 2001; Paul et al., 2021; van Heezik et al., 2008). My findings suggest that as smaller areas have greater edge effects, they support more naturalised species that prefer variable or open landscapes that edge supply. However, smaller or exposed forests may not be suitable for some species as they rely on microhabitats or interior habitats as they have a habitat or resource availability that is exposing systems. Rather, our forest-dwelling natives tend to occupy larger or dense native forests as it is preferable because of less disturbance, more resources, and provides metapopulation processes (Barnagaud et al., 2014; Case, 1996). Alternatively, large forests influence species composition, naturalised or native, because of the higher costs associated with competition for resources, lack of exposure to environmental disturbance, and predation, as found within my study and supported by others (Barbaro et al., 2012; Barnagaud et al., 2014; Lee & Rotenberry, 2005; Parlato et al., 2015).

Tūī detections across sites were foremostly dependent on fencing and aligned with the effects of fenced years, and areas with more detections in predator-controlled sites and, somewhat unexpectedly, fewer tūī in the sites with a higher mean top height of trees (Table 4.3). As stated in Chapter 2, sites that were fenced were often greater in size and are older with larger canopy, tūī response to fencing appears to be aligned with the size of the forest fragment. It is assumed that tūī detections are greater within larger fenced forest fragments as they provide better vegetative integrity (Chapter 3). Gravatt (1971) similarly found that tūī presence is largely determined by their dependence on food sources, whereby they are mainly nectareous throughout the year but forests providence also varies throughout the year based on the vegetation composition, thus what food source a forest ecosystem can provide throughout the seasons (Bergquist, 1985; Coomes, Allen, et al., 2003; Gravatt, 1971). Therefore, the larger the forest system, the more diverse vegetation compositions are and the forest may be able to support more particular species such as the tūī (Chapter 3). A larger forest fragment, which is fenced (Table 4.3, Table 4.4, Table 4.6) will thus allow for greater detection of tūī as there are more viable food sources throughout seasons compared to smaller forest fragments.

There were greater detections of naturalised bird species in unfenced sites as the forests were more often open with less vegetation across different height tiers (refer to Chapter 3), which naturalised bird species prefer, like habitats from their natural European range (MacLeod et al., 2008). In support of my findings, MacLeod et al. (2008) found that naturalised species do not favor dense vegetative forests that a fenced forest offers, such as magpie (*Gymnorhina tibicen*). Macleod et al. (2012) and MacLeod et al. (2009) found that in areas of high naturalised bird species, often in farmland or open forests, there were fewer native bird species detected. It is assumed that the combination of landscape and habitat alterations and removal, which favour exotic bird species, have strongly influenced native bird compositions.

MacLeod et al. (2009) suggest that naturalised birds have been widely successful within NZ landscapes relative to their homeland because of increased niche opportunities. MacLeod et al. (2009) proposed that many exotic species found within NZ are originally from Europe, whereby the landscape is primarily pastoral or open forests, whereas the altered farmland landscape we see throughout the country today, is better suited for these exotic species compared to our primarily forest-dwelling native species. Additionally, exotic species composition within specific habitats, such as forest fragments, may provide better resources compared to habitats in Europe as our native species do not prefer open landscapes. Barnagaud et al. (2014) also suggested that many forest-dependent exotic birds occupy both forest and adjacent farmland because of complementary resources that thus enhance their arability to thrive in smaller forest patches within mostly open matrix landscapes. To get a better understanding of native and naturalised bird compositions, further investigation of trends over time at local spatial and temporal levels is required.

Beyond deer-proof fences and conventional fences, predator control has significant effects on native species composition (Armstrong et al., 2006; Holdaway, 1999; Norton et al., 2016; Tanentzap & Lloyd, 2017; Tscharntke et al., 2016). Young et al (2013), and Boyer (2008) demonstrated similar findings whereby predator-controlled sites had a greater species richness, especially in native bird species. In my study, there were greater detections of forest birds within both the rough block sites (fenced and unfenced), but even though both sites were within a predator-proof fence (low populations of predators still within), there was is a very active regime for trapping and bait control.

As described by Innes et al. (2010), many of our native forest birds of declining populations are dependent on good habitats, with high integrity. These habitats are supported through management such as fencing, which provides physical protection against degradation of vegetation composition through browse (refer to chapter 3), and predator control in support of susceptible species (Holdaway, 1999; Innes et al., 2010). Predator control of forests results in improved life-histories species, such as the korimako, tūī, and ruru as the removal of predators results in increased hatchling success control of the omnivorous ship rats (*Rattus rattus*) and brushtail possums (*Trichosurus vulpecula*) whereby when predation was reduced, resources increased (Holdaway, 1999; J. Innes et al., 2010). Within my study, the frequency of detections for korimako, tūī, and ruru was determined by predator control (Table 4.4). Greater detections aligned with sites with predator control, however, the magnitude of this significance was dependent on species type and environmental variables. For both ruru and korimako, fencing and the predator control effects showed similar significant effects suggesting that they preferred good habitat. However, it was found that they weren't too picky in how a forest acquires good habitat, as they preferred habitats that were managed, whether this be fenced or predator controlled or ideally both.

Similarly, toutouwai detections were greatest in managed habitats, whether fenced or predator controlled. However, detections were lower in sites with both management, suggesting there is a goldy-locks habitat type for toutoutwai, whereby they preferred forest that had active management (fenced or predator control) but not with too much management (both fenced *and* predator control) nor too little (none). It is therefore assumed that forest habitats with high management (fenced *and* predator controlled) result in better quality habitats and thus more competition for resources between naturalised and other native forest-dwelling birds. In addition, in sites with only one source of management (fenced *or* predator control), toutouwai can have a higher occupancy of habitat as maybe other native birds rely on both forms of management as they prefer good habitat (Holdaway, 1999). In sites with both management, many native birds flourish and therefore can occupy greater niches that were unavailable to them in situations with no or little management to then apply pressure onto toutouwai populations.

Contradicting my toutouwai findings, Armstrong et al. (2006) found that intensive predator control within fenced sites saw toutouwai populations increase. However, it was suggested that within other locations throughout NZ, they can survive without predator control, but only in larger forested blocks. In my study, fenced forests were often larger in the area (ha) than the paired unfenced site (refer to Chapter 2), whereby toutouwai detections were greater, responsive to fencing, and thus able to support toutouwai populations. It is therefore assumed that the greater detections of toutoutwai in areas with no predator control but are fenced are attributed to the metapopulation dynamics (more habitat availability) associated with larger forests (Drechsler et al., 2003). Both Richard & Armstrong (2010) findings support my results as toutouwai detections were greater in fenced sites, whereby because the area (ha) of the forest fragment was larger (and fenced), it provided a better suited and supported habitat.

Controversially, in sites with poor habitats (unfenced and no predator control), rirorio detections were higher. This suggests that they prefer forests with poor habitats as they are less susceptible to predation and the benefits of fencing, therefore not affected by the presence of mammals. Correspondingly, poor habitats have less native species richness, which may provide less competition for resources, leading to greater populations of riroriro. Interactions with naturalised species may encourage rirorio populations as exotics species can occupy habitats other than the native forests (MacLeod et al., 2009) reducing resource competition in the surveyed poor habitats.

Longitude was another significant variable influencing species composition as it has similar effects as elevation and coastal exposure (Sullivan et al., 2005). It is assumed that this finding is directly related to the composition of the vegetation rather than a direct influence of longitude. Further research is needed to fully understand how longitude, for fencing and forest fragment size, might influence bird composition.

Bird species composition concerning other environmental variables is difficult to evaluate as understanding species abundance requires an examination of the species on an individual basis to get a gauge of resource ratio and habitat size preferability. Moreover, at a species level, persistence depends on many factors that fluctuate with the changing environmental and seasonal conditions and land usage in the surrounding landscape. Therefore, long-term monitoring of these environmental conditions should be included for a better idea of species-level and species composition trends.

Some of my results may be more an artefact of the design, than a result of current management (Chapter 2). For example, further analysis of the models of effect is required to investigate the direction of effects mean top height with fencing and predator control (Table 4.5). Further, the interpretation of these model findings should be investigated in greater detail as environmental variables affecting metapopulation factors may have biases within the analysis between the habitat and species detection. For example, consideration of habitat factors alone may lead to unjustified conclusions surrounding species habitat preferability, when rather it is habitat accessibility or occurrence due to forest fragmentation.

I further recommend that additional investigations need to be undertaken with comparisons between plant, entomology, and animal communities with their effect on avian populations. Bird species composition is subjected to differences in an ecological process such as dispersal capabilities, habitat enhancements, reproductive catering, and resource accessibility that other taxa provide thus wider investigation between bird composition and relative taxa should be sought.

4.7.2 Audio picks up larger species richness and abundance over time compared to traditional 5MBC

Findings of this study both support and contradict hypothesis two as species detection was higher using the ARD, however detection in species richness was greater using 5MBC. This is fundamental because the ARD misperformed as the protocol settings (described in methods of this Chapter) were unable to pick up high-frequency bird sounds (such as tomtit, rifleman, etc.), species calling at a distance, or non-vocal species, limiting the potential detection of species richness.

However, beyond the protocol error, ARDs can only detect vocally active birds within a close radius around the recorder, yet as my study and Kułaga & Budka (2019) found, ARDs have limitations in detectability based on species type, whereby as it is not suitable for silent species or species with the birdsong of high frequency. Kułaga & Budka (2019) confirmed that with the current technological state, some bird species can only be detected visually by an observer or through a camera and within these instances, the 5MBC is praised. For example, pīwakawaka were common throughout study sites using 5MBC but were not accurately detected with the ARD (Table 4.6). Moreover, tomtits (*P. macrocephala*) were only detected using 5MBC but not ARD, as small birds alike have high-frequency birdsong, which the ARD could not detect using the implemented protocol. This lack of detection using ARDs is a loss in the overall ecological assessment as small birds are often a better indicator species of ecological conditions due to their vulnerability to changing habitats and pressures of biological invasions (Holdaway, 1999), therefore 5MBC was an appreciated compensation.

The 5MBC method requires experienced observers skilled in the identification of different bird appearances and bird vocalisation (Digby et al., 2013; Greene, 2013). This method also has the potential for biases as the observer may produce disturbance, whereby birds will flee through sensory of sighting or sound or vegetation or habitat disturbance when observers are situating themselves (Buckland et al., 2008; Macleod et al., 2012). For instance, when conducting the second 5MBC, observers had been in the area for the entire day, probably scaring sensitive birdlife. Human conducted point counts, and other errors or limitations associated with the 5MBC method can be eliminated by using ARDs (Digby et al., 2013) as they records without this disruption, offering better accuracy of observations of vocal species abundance and richness.

Still, the 5MBC had limitations that affect the detection probabilities of individuals, as shared in my study and Thompson (2002). Thompson (2002) found that there was a variance of detectably species richness, suggesting that like all index methods, the measurement of species richness is weak because it relies on assumptions of observed relationships between the indices and actual abundance (Greene, 2013). Observation opportunities for relatively inconspicuous and cryptic species and maybe an inappropriate method for monitoring rare birds are often missed using the 5MBC. My study welcomed the benefits of ARD detection, whereby tīeke, ruru, kākā, and kiwi (*Apteryx mantelli*), were never detected at some sites using the 5MBC but were detected multiple times using the ARD. ARDs also increase the detectability of species as their recording ability enables the observer to listen multiple times (Shonfield & Bayne, 2017).

Furthermore, environmental factors, such as weather and time of day, also account for species detectability (Greene, 2013). Additionally, ARD provides longer counting times allowing for greater periods of opportunity to move within the survey area, and within closer proximity of the autonomous monitor where the bird species are likely to be detected. This also increases the opportunity for new birds to enter the surveys area and thus more individuals to be counted more than once (Cassey et al., 2007), for instance, the New Zealand Falcon (*Falco novaeseelandiae*) was detected in my study at one of the ten sites by the ARD. The ARDs were out for a total of 12 days, across 10 different sites, and five areas, yet only detected the species once (Table 4.6).

Estimating the occupancy of a species using ARDs is more feasible compared with estimating the density of a species, as suggested by MacKenzie et al. (2002) and Yip et al. (2017). This can be overcome by repeating observations with ARDs at sites to estimate detectability and account for imperfect detection when estimating the probability of a species occupying a site or patch (MacKenzie et al. (2002); Yip et al. (2017). My study's findings partially align with MacKenzie et al. (2002) and Yip et al. (2017), as I measure detection that represents habitat occupation. In Chapter 3, I indicated that vegetation provides support for other species, and is further seen in this section's findings whereby there are greater detections of habitat occupancy in fenced sites.

A logistical limitation with ARDs is that the devices pose a difficult trade-off decision for monitoring between the temporal coverage and spatial coverage as they require either a large number of expensive ARDs or moving the recorders to new locations which can be labour expensive (Digby et al., 2013; Shonfield & Bayne, 2017). Nevertheless, the power of the ARD programming system caters to surveying in remote locations and nocturnal surveys (Digby et al., 2013). The flexibility of this technology enables a broad spectrum of monitoring efforts or procedures for surveying a range of different bird species and habitats (Cotton, 2003). This has been largely demonstrated in my study where ruru detections were made using only ARDs at all but one site.

Greene (2013) described how the detection of small changes within bird population dynamics and species diversity can be problematic using 5MBC as it excludes an array of contributing factors such as spatial distribution, seasonal changes, and time. The 5MBC is insufficient for a true assessment of species richness and population trends (Cassey et al., 2007) as the time allocated for this count is a relatively small snapshot which neglects the greater effect of the diurnal rhythms of birdlife. Additionally, Scott & Ramsey (1981) concluded that count periods of different lengths are required as different species have different behaviours of movement and conspicuousness (Cassey et al., 2007). My study indicated similar issues to Greene (2013) whereby the small-scale 5MBC surveys were subjected to minimal recordings per site and limited site access to viable placement for further monitoring in the ecology of avifauna.

The 5MBC lacks consistency within or among different species and habitats making it difficult to consolidate and compare data, neglecting the probabilities of consistency across time. Future monitoring of bird composition requires authorities to transition from point count methods like 5MBC into the usage of technological tools, such as ARDs. ARDs provide permanent records of actual and accountable birdlife presence enabling peer-reviewing by multiple observers, thus reducing the potential for observational bias (Venier et al., 2012) allowing for reliable data collection and responsible results. The fact that ARD offers an easy setup for repeated sampling, including relatively long periods across spatial and temporal scales significantly simplifies data collection. Permanent records which ARDs can produce enable clear comparisons between both contemporary and historical vocalisations for further future analysing opportunities to investigate trends of changing species richness (Shonfield & Bayne, 2017).

When analysing the recorded birdsong, an observer’s ability or experience to detect the vocalisation of bird species can vary as observers’ have access to references. Here, the accuracy of species recorded by less experienced observers can match the similarities of more experienced bird observers (Celis-Murillo et al., 2009; Shonfield & Bayne, 2017). Within my study, manually reviewing the audio counts was essential for the output and statistical analysis of our data for results. Due to poor weather, high-frequency birdsong, locally changing birdsong, and other disturbances (e.g. stock licking microphone, dogs barking), I was able to revisit the recordings to establish a correct detection measurement.

In some situation, ARD detect fewer species or fewer individual birds compared to point counts as it has been suggested that humans can detect birds at greater distances (Hutto & Stutzman, 2009; Sedláček et al., 2015; Venier et al., 2012). Hutto & Stutzman (2009) suggested that although species detection is more consistent using ARDs, more species were attributed using visual detectability through human observations. Currently, ARDS is not always able to detect species' presence as far distance as humans can, but with technological advancements, the sensitivity of ARDs is improving. Klingbeil & Willig (2015) and Kułaga & Budka (2019) suggested that ARDs have difficulty detecting the differences in amplitudes amongst species and the various distances at which different species can be recorded. This finding was shared with my study as there were limitations of species detections found through the algorithm spectrogram, many species or individual counts were missed due to the variance of local birdsong and distance between bird and recorder for correct detection following the algorithm. However, for both human point counts and ARDs, the detectability of species is often determined by habitat type (Kułaga & Budka, 2019).

Automated species recognition matches recorded species from a template derived from previous recording data from devices such as ARDS, and then is registered with species heard (Shonfield & Bayne, 2017). Within this allorhythmia, the more recordings put into the system, the better the system becomes at recognising species within new recordings. The addition of recordings improves the allorhythmia for species recognition. Therefore, the sooner that authorities tap into ARDs and the automated species allorhythmia, there will be an efficient method in understand bird (and other vocal taxa) habitat occupancy. My study offers the starting process of local automated species recognition over recordings for HBRC to advance monitoring regimes.

Although automated species recognition is effective and efficient (Shonfield & Bayne, 2017), it is not a simple process. Within the system, complications arise with the disreputability of abiotic noise influencing the recordings and therefore the species recognition process, for both the automated systems and the observer, causing the potential of high amounts of both negative and positive counts (Buxton et al., 2016). My study experienced this, whereby there was a day of rain or often at some sites planes would fly overtop, removing any or limiting viable detections for some species. Additionally, the overlapping of species birdsong or calls can confuse both observer and automated species recognition systems, resulting in inconsistencies (Buxton et al., 2016).

4.8 Conclusion

The lack of refugia, such as forests, across the landscape, combined with impacts that threaten avifauna (e.g., predators) has resulted in the deterioration of species richness and abundance of NZ avifauna. My study examined how introduced naturalised avifauna species have acclimated to the changing pastoral landscape. Bird compositions were investigated between native and naturalised avifauna communities across HB farmland forests were surveyed. In addition, my findings suggested that the temporal and spatial patterns of bird composition patterns are related to environmental variables.

Although fencing was a significant determinator for bird species composition (Chapter 3), the composition of both native and exotic bird species was largely determined by other environmental variables, with the most influenceable being the area (ha) of the forest fragment. Although sites vary, I found that bird species composition was dependent on forest size (area ha), fencing, and predator control regimes. Effects of predator control and detected species were combined with the effect of the fencing regimes and forest fragment size. The significance of effects on abundance within a site, and thus habitat preference, was dependent on species type or bio status, and therefore behavioral and ecological traits concerning the habitat preference. Overall, bird species composition was determined by the preferred or tolerated environmental combinations of measurable environmental variables.

I found that despite the limitations, the traditional 5MBC method was still a valid option for measuring species compositions, however, there is a groundswell for a more accurate means of monitoring bird abundance and diversity through technological advancements, such as ARDs. As conservationists, acceptance of traditional methods of monitoring, such as counting technique based on the fact that it is commonly used or because there is a historical data set should not outweigh opportunities for updates techniques (MacLeod et al., 2008). ARD has proven to be simple to standardise while eliminating some bias and errors delivering more reliable results and data accumulation. When appropriate this system also increases the survey duration and durability of detectability (Kułaga & Budka, 2019).

Using ARDs to record birdsong allows for improved and repeatable analysis (Hutto & Stutzman, 2009), although within my study, species detections were limited to birdsong. This implementation error should act as catalysts in improving the tools for detection to transition into better monitoring and conservation strategies, such as fencing and monitoring regimes to enhance and sustain the integrity of species, habitats, and ecosystems. The usage of technological advancements such as ARD is fundamental to keep up with the changing environments and species trends. When combined with other technology such as camera traps, tracking tags, satellite remote sensing, and drones, the results would be highly beneficial to conservation strategies.

There is an urgent need for regional leadership within councils, governmental authorities, local organisations, and especially private landowners to provide for conservation management when addressing the decline of our native avifauna.

The findings and discussions within this Chapter fulfill the purpose of this thesis in supplying an assessment of monitoring tools, and through using these monitoring methods offering recommendations of implementation for further study or monitoring endeavors. This chapter also offers the current status and composition trends of lowland Hawke’s Bay farmland forest fragments.

Chapter 5

An assessment of sampling effort in Tier 2 Biodiversity Monitoring of Hawke's Bay Forests

This chapter reviews and offers recommendations for implementing the Tier 2 Biodiversity Monitoring Framework (T2BMF) in the Hawkes Bay. The vegetation survey data is used to assess sampling intensities for the seedling subplots and 20x20 m plots. I use this to recommend a minimum of intensity sampling. The chapter then describes some issues and disadvantages that were found using the T2BMF, and solutions are suggested.

5.1 Introduction

Forest fragments across the country, and specifically within the Hawkes Bay region, have increasingly become ecological assets for private land owners in landscapes of intensified farmland. The long-term health and ecological integrity of these forests relies on the presence of natural forest succession and vegetation dynamics (Standish et al., 2009). Species succession can be facilitated through management strategies such as fencing and predator control (refer to Chapters 3 and 4).

Ecological integrity is assessed through biodiversity monitoring, as this documents the state and changes in a site's species composition, and monitoring data help to understand, predict, and manage changes. Monitoring efforts allow wildlife managers to comprehend these changes in the composition, structure, and function of forest ecosystems over time (Bellingham et al., 2019). These efforts are important to gauge the overall changes in the vegetation dynamics by comparing the current ecosystem structure with what is expected from an equivalent natural forest ecosystem, and so assess a forest's ecological integrity (Bellingham et al., 2019).

Within an NZ context and at a regional scale, monitoring our local native forest integrity allows us to evaluate the current conservation performance of management at both a local and national scale (Lee, McGlone et al., 2005). It also allows for an inventory of our native biodiversity so we are aware of what to protect, thus providing an assessment to then prioritise conservation interventions (Lee, McGlone et al., 2005; D. B. Lindenmayer et al., 2012a; Watson & Novelly, 2004).

5.2 Implementation Process of the Tier 2 Biodiversity Monitoring Framework

The findings of this study demonstrate the success of the T2BMF, although there were some alterations made to best fit the study requirements and limitations (refer to Chapter 2).

In terms of plot logistics and expenditure, the survey time at each plot is important as it contributes to regional councils' prioritisation of data collection. Each of the plot's total sampling times consisted of the time it took to locate the plot and set up the 20x20m tapes including subplots; the time to survey the plot's vegetation, birds, mammal presence/browse, and abiotic site data; and a (rough) estimation of the time it took to curate the data, confirm identifications, and input data digitally. I do not include the travel time to and from the site.

On average, each plot setup time took on average two people around 2 hours, including the placement of waratahs and other plot features such as permolats (Table 5.1). Plots with dense vegetation and rugged terrain resulted in longer set-up and survey times. Fenced sites often had denser vegetation, meaning that the vegetation survey time increased due to a higher number of individuals, saplings, and multiple stemmed trees within a plot.

Tagging and measuring the diameter at breast height of every tree and multiple stemmed was a slow process, as care and coordination amongst people was required for both tagging stems (a process whereby the tag must be in the correct order, nailed at the correct location and height following strict procedure) and making diameter at breast height (DBH) measurements. However, Innes (1993) determined that the measurement and tagging of adult trees was crucial on a long-term scale for monitoring changes in a forest's ecological integrity.

Measurement of trees is necessary to document the forest composition and structure and to track tree growth and health; it is one of the most important indicators of forest condition, thus the overall ecosystem integrity (Innes, 1993). Both Innes (1993) and Dobbertin (2005) discussed how tagging provides wildlife managers with an idea of how a forest fragment is responding to management strategies. Tagging trees and the stems of trees allows for repetitive samples of plots for re-measurement, providing for a measurable assessment of ecological integrity in response to changing environment or management. The measurement of tree presence and growth through tagging individual trees and their stems provides managers with a sense of a tree's health, as it indicates the capacity for these forest fragments to survive stress and environmental or ecological changes. Therefore, repeated site assessments will be able to assess tree growth and mortality and predict changes in tree competition, tree size, tree health, and tree growth (Dobbertin, 2005).

Sub-plots that were densely populated by saplings were slow and challenging to correctly count. Fieldworkers were required to work together to separate saplings to gather correct counts and species identifications. Further, it was slow and demanding work to gather the seedling counts between subplots and in plots where there was dense vegetation due to leaf litter and low-growing vegetation. Seedling plots are where rare or cryptic species are often found (Chapter 3), which can have a big effect on estimates of the ecological integrity (Appendix B Table 8).

These sapling and seedling counts are necessary as their presence or absence determines the future vegetation and canopy composition (Chapter 3), which can only directly be assessed by measuring sapling sizes and seedling growth and presence repetitively within a plot. These counts provide estimates of the succession rate and the regeneration within a forest.

A large proportion of my time was spent identifying collected specimens and entering data from paper sheets into the National Vegetation Survey (NVS) Excel template (Appendix A)(Table 5.1). Specimen dentifications were at times confusing due to my inexperience, and I needed to seek the opinion of botanical experts. Having a botanical expert undertake the survey would ensure the data and species identifications are correct, making the process more efficient and reliable. However, combination of skilled but less experienced field workers with access to a botanist is likely the best approach.

A forests ecological integrity can be determined following the T2BMF as it provides an assessment of tree species growth and species succession, and its data can be fed into vegetation models to compare the differences between actual growth of measured vegetation against predicted growth (Dobbertin, 2005). This is only possible because total individual counts were recorded (Chapter 3). Deviations from the expected growth or succession of vegetation can reveal the effects of other environment or ecological variables, such as the advantages of fencing. For example, this study was able to determine how browsing affected vegetation composition and growth and anticipated species succession and therefore the ecosystem integrity (Chapter 3).

The vegetation survey provides an estimate of the overall percentage of biomass of species across height tiers, determined through a variety of indicators following these field protocols, including foliage or canopy percentages, and percentages of species cover amongst height tiers. By sampling individual trees over a size tiers, allometric equations can be developed using regression models for estimating the relationship between tree component biomass and one or more size parameters (Digby et al., 2013). In densely vegetated plots, it was often complicated to determine both the canopy percentage of the plot and also the percentage of species across different high tiers for the RECCE sheet assessment. In these situations, fieldworkers took time in communicating and discussing percentages to get accurate results following the protocol.

Following the Framework, even with additional monitoring methods, it was challenging to assess the full effectiveness of fencing (Chapter 3 and 4), as this site management occurs against a background of broader environmental variation. This underscore the importance of surveying many sites, as each site is different in idiosyncratic ways.

Even when using the Framework with constraints in resources, and within what was regarded as the minimum number of paired sites for testing the effects of fencing, the framework did successfully quantify the species composition and vegetation structure among sites (Chapter 3 and 4). The framework revealed some clear differences in species composition and forest structure, which showed clear links to environmental variables such as fencing (Chapter 3 and 4). I therefore conclude the sturdiness of the Frameworks measurement of ecological integrity of Hawkes Bay's native forest.

Site	Mean set up/locate time (hrs)	Mean sample time (hrs)	Mean number of people surveying	Mean processing collection time (hrs)
Kirk	2.3	7	2	4
Dave	2	6	2	3.3
Roughblock Fenced	2.5	6.3	2	4
Roughblock Unfenced	1.5	6	2	3.3
Livesey Fenced	2.75	7	2	5.4
Livesey Unfecned	2	6	2.5	4
McLeans	2.25	5	2	4.3
MotuMokai	2	5	2	4
Neil	1.5	4	2	5
Mark	2	6	2	4.3

Table 5.1: Recorded time estimates from survey, calculated as a mean value from the three surveyed plots per site

5.3 Sampling Intensities

In the Bellingham et al. (2021) report introducing the T2BMF, it was recommended that further research should investigate optimal sampling intensities and placement of samples for prioritised ecosystems. My study trialled the proposed framework at a sites across the Hawke's Bay region. However, the recommended framework focuses on the minimum requirements for monitoring change or trends rather than broader means for monitoring biodiversity that may allow for a greater grasp of ecological integrity. Therefore, this study implements the suggested methods, with few alterations and additions in the data collection (Chapter 2).

Further, the proposed T2BMF does not provide councils with direction as to how many samples or surveys are required to get a gauge of ecosystem integrity (Bellingham et al., 2021). Bellingham et al. (2021) focused on describing the details of the proposed sampling methods, and excluded a consideration of the sampling replication and frequency that would be required.

My research was an opportunity to assess this, by understanding the species richness accumulation with sampling effort. Also, due to the Regional Councils' limitations of resources and expenditure, I consider whether a mix of complete and simplified versions of the Tier 2 method could allow for more site replication. Therefore, this study seeks to determine the appropriate sample intensities to determine a forest fragment's ecological integrity so HBRC, and other councils, can implement the framework as efficiently and as informatively as possible.

5.3.1 Methods

The T2BMF methods that are discussed in detail in Chapters 2, 3, and 4. My study applies the T2BMF and trails modifications of survey methods due to labour and time constraints, such as the addition of automated recording devices and photo points (refer to Chapter 2). Monitoring methods are based on well-established methods which have a history of results in peer-reviewed publications (R B Allen, 2007; Department of Conservation, 2013a; Greene, 2013; Hartley, 2012; Manaaki Whenua Landcare Research, 2021a).

Statistical & Data Analysis

To analyse vegetation of dynamics within the community and to assess the intensity and frequency of surveys, I undertook two analyses. All vegetation analysis were performed using R (R Core Team, 2013).

Plot-Level Species Accumulation

An investigation of plot sampling intensities, with accumulated species richness found within each of the three plots across all sites, was undertaken. All of the observed species within the RECCE across all the height tiers were simplified into the presence or absence of species found within each of the sites' three plots.

Using this simplified dataset, total species were calculated in each site based on the species accumulation curve. The function “specaccum” in the package vegan was used to plot the accumulation of species richness with increasing sampling, and the function “specpool” was used to estimate the total species richness (the Chao estimate is used here).

Seedling Subplot Species Accumulation

Analysis of the accumulated species richness in the 24 seedling subplots per plot was then investigated. Observed species within the seedling subplot surveys were simplified into presence or absences within subplot. The same method as above was used to plot species accumulation and estimate total species richness.

5.3.2 Results

Plot-level Species Accumulation

The species accumulation curves (Figure 5.1) demonstrates the positive relationship between plot survey intensity and species richness. This graph illustrates the substantial increase in species richness between the first and second plots, and a smaller increase when adding the third plot. As the line does not plateau, this graph suggests that surveying another plot per site would further increase the documented species richness. There was generally a greater species richness in unfenced sites whereby the species richness collected is consistently steeper across sites compared to the fenced sites (Chapter 3). Collectively, the fenced sites have less species richness across the 3 surveyed plots (Figure 5.1)

The Chao estimates of the total species richness (Table 5.2) give an indication of approximately how many more species would be found with more plots. The Chao estimates indicate that three plots per site was sufficient to document 64–89% (mean 77.4%) of the species at a site.

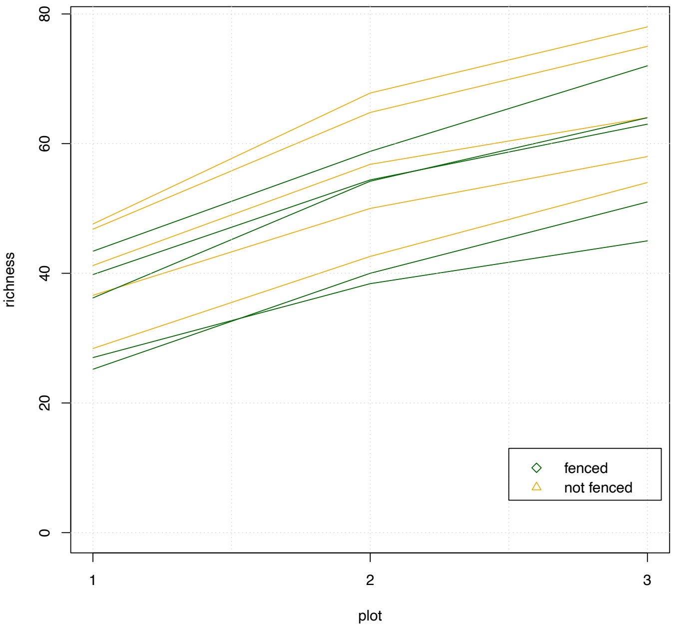


Figure 5.1: Species richness accumulated for each site over 3 surveyed plots. Each line represents a site's increase in species richness for every surveyed plot (green = fenced and yellow = unfenced)

Table 5.2: *Site species richness* at each site as sampled in *three* plots and the total richness estimated using Chao *estimates if there were enough plots to sample all of a site's species*

Site & Fenced (Y/N)	Species	chao	chao.se	3 plot chao estimate %
Kirk (Y)	45	61.1333	9.4597	73.6096
Dave (N)	64	78.5185	7.5061	81.5094
Roughblock (Y)	51	79.4444	14.0358	64.1958
Roughblock (N)	54	80.6944	13.3426	66.9191
Livesey (Y)	72	94.7368	10.3627	76.0000
Livesey (N)	78	92.5833	6.8179	84.2484
McLeans (Y)	63	79.4902	8.3662	79.2551
Motumokai (N)	75	83.7604	4.6631	89.5411
Mark (Y)	64	84.1667	9.5946	76.0396
Neil (N)	58	70.0000	6.7082	82.8571

Seedling Sub-plot Species Accumulation

The proportion of the species found in all 24 subplots, as they accumulate with added subplots. In general, there was more variation in how the seedling richness accumulated in the fenced sites compared with the unfenced sites (Figure 5.2). When only five subplots are done, this detected on average 51% (fenced sites) and 53% (unfenced sites) of all seedlings in a plot (Figure 5.2).

The total seedling species richness per plot, using Chao estimates (Table 5.3), varied considerably. The 24 subplots per plot are estimated to have recorded between 20.6–90.8% (mean 72.6%) of all the seedling species in each plot (Table 5.3), suggesting that the sampled 24 subplots per plot are a necessary minimum survey intensity.

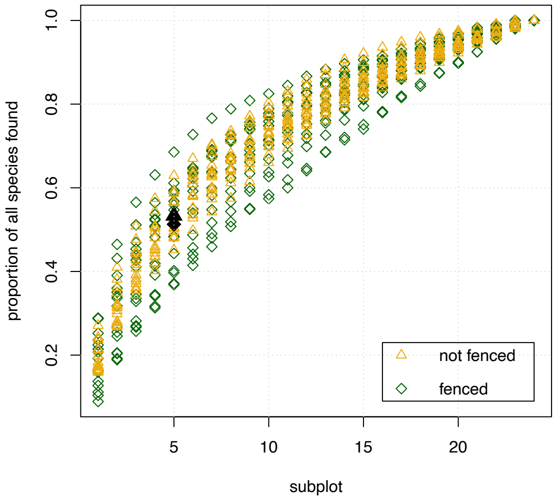


Figure 5.2: Curve demonstrating accumulation of the species richness per subplot per plot (green = fenced and yellow = unfenced). The black diamond (unfenced) and triangle (fenced) represent the average proportions of the species found in all 24 subplots that would have been detected in just 5 subplots.

Table 5.3: The recorded seedling species diversity in 24 subplots per plot, and the estimated total plot seedling richness using Chao estimates, for each plot at each sites

Site & Fenced (Y/N)	Species	chao	chao.se
Kirk (Y) P1	21	25.3810	4.0889
Kirk (Y) P2	13	43.6667	37.9649
Kirk (Y) P3	21	101.9792	93.0394
Dave (N) P1	27	33.4688	5.6604
Dave (N) P2	21	27.1333	5.6979
Dave (N) P3	48	63.4018	10.6088
Roughblock (Y) P1	17	20.8333	5.0867
Roughblock (Y) P2	24	33.7031	8.6652
Roughblock (Y) P3	14	19.7500	6.2300
Roughblock (N) P1	27	32.1111	4.7462
Roughblock (N) P2	25	32.8264	7.9527
Roughblock (N) P3	18	23.8698	5.8457
Livesey (Y) P1	18	25.6667	11.1903
Livesey (Y) P2	35	42.2474	5.7370
Livesey (Y) P3	22	25.4074	3.2408
Livesey (N) P1	33	48.9722	14.2497
Livesey (N) P2	39	47.0979	5.9019
Livesey (N) P3	46	73.6958	19.0156
McLean (Y) P1	14	19.9896	7.2545
McLean (Y) P2	30	34.3125	4.6217
McLean (Y) P3	36	49.7739	10.7219
Motumokai (N) P1	29	33.3125	4.6217
Motumokai (N) P2	52	67.3333	10.1735
Motumokai (N) P3	43	46.3542	3.3749
Mark (Y) P1	24	62.8125	47.0610
Mark (Y) P2	27	30.4500	3.7102
Mark (Y) P3	23	24.9928	2.4148
Neil (N) P1	27	46.4063	19.3671
Neil (N) P2	29	57.9896	27.4734
Neil (N) P3	25	28.9058	3.8943

5.3.3 Disscussion

Due to regional council restrictions such as funding for monitoring, assessments of the ecological integrity of forest ecosystems will always be limited in quantifying ecological complexity. Careful choices of energy and expenditure are therefore required and should be underpinned by a defensible rationale (McGlone et al., 2020). This can be achieved through simplifying collected data to get a measure of monitoring intensities of scales to best fit the available resources, as my study has attempted to do. Species accumulation results suggest that three plots per site and 24 seedling subplots per plot are a sensible minimum sampling effort, and less sampling per site and per plot would risk missing many species. The main purpose of the T25MBC is not so much to detect every species at a site but to detect enough species, and describe their composition in enough detail, to describe the ecological integrity of a site in a way that can be repeated over time to reveal trends.

Although the T2BMF is remarkable in the sense of the strategic and intense sampling regimes, a sampling intensity required for the initial assessments is potentially out of reach for Regional Councils if applied across many sites. In undertaking this intensity analysis, Regional Councils must therefore balance resource expenditure in time, monetary, and labour terms with the benefits of fully implementing the T2BMF for assessing the ecological integrity of prioritised forest ecosystems.

Plots per Site

The plots were a fixed sample area (20x20 m), as is standard for New Zealand forest vegetation monitoring. Due to the plot's square shape, they are slow to lay out, especially the first time, when ,markers are added for permanent relocation. Permanent plots are ideal for monitoring changes or trends of biodiversity over time as they are generally regarded as statistically efficient in estimating changes in measurable species abundance, as they typically provide the appropriate amount of cover observations and remeasurements (Allen, 2007; Bellingham et al., 2019).

As implemented within this study, plot locations were stratified at random points by randomly assigned across a clipped grid using QGIS, by way of a generalised random tessellation stratified sampling (Chapter 2). This system created an equal-probability design that allows for statistical inference while providing balanced spatial coverage (Tierney et al., 2009b). The question now is, how many plots are needed for a viable assessment of ecological integrity?

The findings of this study support the recommend three sample plot sizes per site made by the Maanaki Whenua representatives, Sarah Richardson and Peter Bellingham. It was found by simplifying the accumulative species richness of mean values across each of three plots for each of the sites, three was the bare minimum for a viable vegetation survey, shown through the percentage of species found (Fig 5.1 and Table 5.2).

As suggested by Lynch (2017), and in support of this study’s finding, the optimum sample size must be determined by the sample size needed to achieve a specified standard error of the mean whereby a specified error with a specified confidence level must be acknowledged. The results of this study suggest that the implemented sample intensities (Fig. 5.1), with three sampled plots within a site are a bare minimum.

There was a higher species richness within unfenced sites than fenced sites because unfenced sites contained more herbaceous species (Chapter 3). This meant that plot species accumulation was steeper in unfenced compared to the fenced sites. The fenced site's increase in species richness between plots is less steep as these sites are not as exposed to naturalised species which are commonly dispersed by the movement of stock (Chapter 3). It was found that for fenced sites, 3 plots were a viable survey intensity as there are fewer plant species within the forest, to begin with (as there are fewer exotics). Additionally, many of the species found within the unfenced sites are naturalised exotics, making surveying more complex in terms of identifying due to seasonal occurrence and cryptic tendencies because of predation awareness. For fenced sites, surveying 3 plots is more tangible for Regional Councils as native perennial species are often somewhat easier to identify. However, this is site-specific and dependent on management activities such as pest control or site area providing microhabitats.

Concerning Regional Councils limited access to resources, it is suggested that 3 surveyed plots are sufficient in terms of measuring ecological integrity. This is suggested following discussions within Chapter 3 where it was found that only three plots was sufficient to show that fenced forest fragments had higher ecological integrity than unfenced sites.

In this concept of sample redundancy, as described by Gauch & Gauch Jr (1982), sampling vegetation with greater intensity than necessary or required to meet objectives is often classed as inefficient. In contrast, the effects of oversampling, as in the generalised case of sampling ecologically interesting habitats more intensively, can provide managers with untargeted findings that often provide a new gauge of approach to management.

Further, as discussed by Cooper et al. (2006), sampling intensities needed for reliable landscape-scale ecological studies are usually high due to the ecological complexities over a larger matrix (or in this case, a region). This also corresponds with larger forested areas, as it is assumed that the area of a surveyed forest fragment will determine the plot sampling intensities. In support, my findings (Table 5.3) have demonstrated that the estimation of species diversity for each site exhibits a large difference from the actual count of species diversity for the Rough Block Fenced and Unfenced sites and also the McLean site. These sites had some of the greatest areas, supporting Cooper et al. (2006), concluding that the sample intensities were not intense enough to survey a broader range of species diversity within a larger area. Further analysis to optimise sampling intensities in larger areas would be beneficial.

As suggested, 3 plots in forest fragments of these sizes are viable in measuring ecological integrity, although it needs to be accepted that more plots will record more species (Figure 5.1). If one of the intentions in sampling is to understand the plant species diversity at a site, then more plots would be helpful, especially at larger sites. However, if instead, the main intention of sampling is to quantifying trends and patters across the larger landscape, then a smaller number of plots per site would be adequate. Sites may be spatially and/or temporally larger or diverse, meaning that 3 plots don’t offer a true representation of a site. Additionally, some sites may need to receive higher intensities of sampled plots in targeted monitoring schemes for specific species (e.g. status or protection of threatened native species).

Lynch (2017) also provided another approach for the determination of sample size by minimising the standard error of the mean for a fixed total cost or given sampling budget. Further investigation of the regional councils' time and budget would be required to understand this. Additionally, for the determination of sample size, a final approach was recommended by Lynch (2017) and it was suggested to minimise the total cost of a method and the loss in decisions, allowing a determination of the best sampling intensity for investors for multiple resource planning. Here, the cost-plus-loss determination offers different results than conventional sample size determination as this approach uses a set of desired confidence levels and desired error. Again, further investigation and calculations must be determined between the private landowners and the Regional Councils for a better understanding of this approach.

Seedling Subplot Intercepts

The National Vegetation Databank (Allen, 2007; Manaaki Whenua Landcare Research, 2021a) has provided legitimate national-scale assessments of the condition, trends, and status of vegetation composition. These well-established forest surveys require a consistent species and structural survey of 20 m by 20 m plots with 24 sub-plots within each plot. These surveys are vital in terms of assessment of ecological integrity and condition, and successional or regenerative processes (Burns et al., 2011; Coomes & Allen, 2007a; Kunstler et al., 2009; Smale, 2008).

Species compositions among the fenced sites differed significantly, but this was not shared with unfenced sites as the physical barrier of a fence ensured that all fenced sites were experiencing similar environmental effects (e.g., lack of browse, exposure, edge effects) (Chapter 2). Therefore, the proportions of species found within the subplot surveys differed more within fenced sites.

The 24-subplot vegetation survey provides an assessment of the successional or regenerative presence of many species; however, some species were inevitably missed with these subplots as seedling distributions can be patchy. As Chapter 3 touches on, there is a greater species richness in lower tiers (subplot surveys). Surveying subplots allow detections of extra species that were missed in the main plot survey. Due to the size of the plot, limited time or resources allocated, and the cryptic nature much vegetation species, many species were possibly excluded from the vegetation survey.

Within this study, the proportion of species collected for each additional subplot survey grew significantly initially and then this incline plateaued. The results of this study found that for a rapid or half survey of the plots (Chapter 6), 5 subplots detect about half of the seedling species collected in 24 subplots. This provided reassurance that although a survey of the implemented and well-established 24 seedling subplots provided the larger proportion of species richness, performing 5 subplot surveys provides a useful rapid assessment of just over a half of species found, and probably the most common of a plot’s overall species.

5.4 Conclusion

Although the T2BMF is remarkable in the sense of being a precise and repeatable sampling regime, each plot requires a lot of effort (and expertise) and regularly repeating a large grid of such sites across a region is likely beyond the budgets of many regional councils. In undertaking this intensity analysis, Regional Councils can therefore gauge resource expenditure in time, monetary, and labour terms to the implementation of the T2BMF for assessing the ecological integrity of prioritised forest ecosystems.

It is concluded that the implemented vegetation survey was sufficient to assess the ecological integrity of sites. Three plots per forest fragment was adequate for most sites to capture most of their plant species while describing species composition and vegetation structure. Larger sites may require more intense plot monitoring regimes to measure their greater and more variable species diversity. Further investigation of sampling intensities and resource allocation for Regional Councils would be recommended to implement these methods most efficiently across a whole region. It is expected that simplifying monitoring efforts may be necessary to meet regional council resource limitations, and modifications of the implemented survey method at some sites would be viable for a rapid, less-detailed surveys (Chapter 6). For example, analysis shows the seedling species richness with just 5 of 24 subplots is sufficient to detect about half the species.

It is necessary for Regional Councils to plan resources, time, labor, and monetary expenditure so that their monitoring efforts, and thus effective management can be sustainable. It is my hope that my assessment here will assist councils in their planning to begin efficient biodiversity monitoring. In doing so, Regional Councils will be able to address regional and national biodiversity crises and threats in a sense of controlled urgency needed for an understanding of the status of biodiversity and ecological integrity assessments. Using these results and discussion points, the next section of this chapter thesis provides recommendations for the Hawke’s Bay Regional Council to adopt a systematic regime for a manageable monitoring biodiversity program within the region.

Chapter 6

Recommendations

Recommendations are provided for the Hawkes Bay Regional Council (HBRC) for monitoring prioritised forest fragments. I provide a hierarchical monitoring framework for a network of flagship, community, and local sites, with more effort and resources devoted to monitoring flagship sites, and community partnerships used to monitor the other sites in less detail. It is my hope that my recommendations will help improve the protection of the Hawke's Bay region's forests by better understanding the trends in biodiversity and the benefits of management like improved fencing schemes.

6.1 Recommendation

The T2BMF provides an impressive framework for a survey of standardised measurements and assessments of prioritised ecosystems on private land. However, regardless of this impressive framework, there are limits to the intensity of large-scale biodiversity monitoring schemes (McGlone et al., 2020). The frequency and intensity of monitoring schemes are poorly defined and there are no detailed directions for how best to set up a programme to monitor the prioritised ecosystems across regions.

Therefore, developing a manageable biodiversity monitoring programme within a region is challenging, let alone coordinating this across all Regional Councils nationwide, yet, as discussed, doing this is essential. there are challenges such as the best timing and scope of tasks and working out how to best use the data to trigger interventions to achieve conservation or management goals. These complexities further increase when programs must align with private land holdings aspirations, which require the establishment of relationships and trust between landowners and councils, particularly if there is a change in land use. It is suggested that Regional Councils can use the recommendations within this study to provide attainable assurance to assess prioritised forest fragments' ecological integrity. The question is, where to begin?

6.2 Recommendation in Action

The deteriorating condition of NZ native biodiversity, the native forests, and the matrix networking between these forests, is concerning, and due to anthropogenetic activities and influences, such as the introduction of biological invasions, habitat destruction, and fragmentation (McGlone et al., 2010). Specifically, on a national scale, the Hawke's Bay region has significant biodiversity loss due to the extensive fragmentation of forest fragments at the lowland level due to the expansion of urbanisation and the intensification of agricultural and horticultural endeavours (Forbes, 2012; Hashiba et al., 2014; Hawke's Bay Biodiversity, 2015; Smith et al., 2020), not dissimilar to other regions across the country. The scattered and scarce native forests that remain across the lowland landscape (Hashiba et al., 2014) must therefore be actively managed for the preservation of endemic biodiversity for the protection of integrity.

As discussed in Chapters 3 and 4, the success of protecting and enhancing the integrity of prioritised forest ecosystems is determined by the management and interventions, such as fencing and predator control. These management regimes are largely dictated by private landowners, sometimes with assistance and support from regional councils.

A systematic and incorporated approach with landowners and local authorities to biodiversity monitoring is therefore highly desirable. Management activities or interventions are linked between private landowners and regional council representatives, such as fencing to exclude stock; wide-scale pest animal control, especially vegetation damaging pests such as goats and possums; local pest animal control (for this Hawke's Bay study example this control would be for deer); weed or invasive plant control; restoration planting or managing natural regeneration to augment forest area or enhance the connectivity of remnants across the landscape (Bellingham et al., 2021).

6.2.1 Zones & Connectivity

To overcome complexities within the Hawke's Bay region, and with aspirations for other regions to follow, my study firstly recommends dividing the region into different spatial sub-regions or zones across the lowland landscape. These zones would be based on similar spatial positioning and ecological or environmental variabilities (e.g. forest type, latitude, longitude) and geographically accessible for regional councils. It is appropriate that the Regional Councils selection of prioritised forest ecosystems are distributed throughout the region zones in an attempt to secure connectivity and regional representativeness. This purposely secures the coverage of prioritised ecosystems and therefore monitor efforts, across all the region.

6.2.2 Classification of Proritised Ecosystem Types

Any remanent or regenerated patch of native forests found in the Hawke's Bay region is important, since so little remains. However, for the management and protection of these forests on private land, classification of priority of fragments based on ecological importance is necessary, since there are limited resources to monitor and manage these sites. Only a few of the many forest fragments can be considered high priority due to the limitations of resources and available manpower. Prioritisation of ecosystems allows Regional Councils to engage in the protection and enhancement of our native biodiversity in bite-sized chunks, with the ambition over the years to increase the number of prioritised ecosystems across the region as expertise and resources become more equipped.

It is recommended that within the pool of prioritised ecosystems, (as determined by regional councils) each one is classified based on the prioritised value, the urgency for monitoring or its unknown status, community interest, and the foundation of connectivity within a zone. It is recommended that prioritised ecosystems are classified into three defined groups (flagship; community; local), which then determine monitoring regimes and monitoring procedures, as described below and outlined in (Table 6.1).

Flagship sites

It is proposed that within each zone, there should be at least one flagship forest fragment that well represents the other surrounding forest fragments. A forest fragment could also be classified as a flagship in situations whereby the forest is of interest, whether it be high in ecological values or strong community support (Table 6.1). The monitoring of these flagship sites will be led by the regional council. Although monitoring one current, flagship forest fragment per zone is viable, it would be expected that the number of flagship forest fragments within a zone would grow as resources, interests, and local knowledge expand.

Community sites

Due to the cost of a full flagship survey, a large proportion of prioritised forest ecosystems that have high ecological value and community interest are instead classified as community sites. These sites should be selected in a random manner within regional zones, stratified by sites with strong community interest and access, or sites of particular ecological importance (Table 6.1). It is suggested that the monitoring of these sites are facilitated by the Regional Councils but supported by communities and landowners.

Local sites

While it is appreciated that not all sites can experience full, frequent ecological integrity surveys due to limitations in resources and expenditure, many sites are not well known and would benefit from some monitoring. These sites can be classified as local sites when there is sufficient local landowner or community support to lead the monitoring (Table 6.1). A smaller proportion of forest ecosystems will be within this classification based on accessibility and relationships with the landowners. These sites have no public access and therefore little community accessibility making monitoring effort difficult but shall be maintained by Regional Council input, landowner input, and perhaps become QEII Trust led.

6.2.3 Recommended Monitoring Protocols & Regimes for Prioritised Sites

Fully implemented the T2BMF at a site, although it provides valuable assessments of ecological integrity, requires a considerable amount of time, resources, and expertise. This framework may seem daunting within the practical scheme for the regional council context, however, the assessment that it produces is invaluable in the sense of monitoring biodiversity, mitigating threats, applying appropriate interventions, and assessing the ecological integrity.

In this study, the T2BMF was assessed and it has been acknowledged that it is intense and expensive in time, resources, and money, and requires considerable botanical, avifauna, and ecological knowledge. Furthermore, due to limitations of resources and convenience on such a large regional spatial scale, regional council, and private landowners, this study recommends monitoring regimes that offer a more tangible approach to monitoring prioritised ecosystems.

When approaching conservation and monitoring endeavours, trade-offs between intensity and the cost of biological monitoring at large spatial scales is necessary. Therefore this study recommends developments in cost-effective monitoring techniques, including methods for rapid assessment of biodiversity across taxa, and community-based metrics of measurements (Ladin et al., 2016).

Therefore, using the foundations of the T2BMF, with the described additions of collected samples and data (Chapter 2), it is now recommended that monitoring regimes and intensities should be determined based on the classification of sites, as described above. Each categorised site has a different monitoring regime, frequency, and set of tasks to provide Regional Councils with an efficient, yet rational approach based on the current restrictions of time, resources, accessibility, and the urgency to acquire biodiversity data to assess the unknown status of ecological integrity.

The recommended monitoring frequencies and intensities are flexible and should ideally be focused on the urgency of the prioritised ecosystem at hand. This could be a subjective take on urgency, or if a site requires attention based on historical issues, or driven by landowner motivations. Furthermore, regimes of monitoring may be based on the accessibility of the location of sites which fits with other means of regimes within council protocols.

It is disclosed that each prioritised forest site will require initial efforts for the establishment of three permanent 20x20 m plots. By adopting the randomised plot point method in my study (refer to Chapter 2). It can take between 1–3 hours to set up, depending on the density of the vegetation and the geography (Chapter 5).

After a survey has been implemented and data collected, it is further recommended that regional council members provide a report of the status and condition of the prioritised ecosystem for the private landowner. The proposed monitoring regimes and intensities should produce measurements and reports that envision a story to communicate to private land owners. This process is essential in the sustainability of monitoring regimes and preservation strategies as it encourages and inspires private land owners to have a conservation focus within their practice. Providing reports gives a measurable gauge of the condition and integrity whilst providing a management plan that private land owners can relate to. When private land owners can relate to prioritised ecosystems, they are more motivated to action preservation and conservation strategies to maintain their connection with nature.

Flagship sites

The recommended monitoring approach to flagship sites should be surveyed fully, meaning that the T2BMF is implemented, with additional data collection (Table 6.1). This full survey involves a vegetation survey (Department of Conservation, 2013a; Manaaki Whenua Landcare Research, 2021a) of the three permanent plots with two 5MBC counts within the centre of each plot, and soil samples should be collected. For more detailed monitoring of the bird community, it is recommended that Regional Councils utilise the benefits of automatous recorders (Chapter 4). All sites should have an ARD, and pitfall traps for invertebrates, placed within the forest fragment every year.

It is recommended that these sites are fully surveyed once every five years, at a minimum, to maintain an understanding of an ecosystem’s integrity. In doing so, it allows regional council managers to follow trends or threats so that appropriate interventions can take place if needed.

Further, it is recommended that these flagship sites should be monitored every year undergoing the suggested half survey method (Table 6.1). This involves interior and exterior vegetation detection transect. Additionally, each of the three plots shall undertake monitoring consisting of the cover board method, seedling plot point pictures, browse index, and coordinated mammal monitoring or detection (Table 6.1).

Community sites

These sites will undergo full surveys, using the T2BMF, with additional biodiversity data collection, at least every 5 years (Table 6.1). Further, it is recommended that these community sites should be monitored using the recommended half survey every year if community volunteers are available. For each community site, they should be monitored every year undergoing the suggested half survey (Table 6.1). This involves interior and exterior vegetation detection transect. Additionally, each of the three plots shall undertake monitoring consisting of the cover board method, seedling plot point pictures, browse index, and coordinated mammal monitoring or detection (Table 6.1). It is further recommended that all community sites should have an ARD placed within the forest fragment every year, and pitfall traps every five years and later processing with eDNA methods.

Local sites

It is recommended that these local sites should be monitored every year if community volunteers are available (Chandler et al., 2017; Lee et al., 2005; Lindenmayer & Gibbons, 2012). For each local site, it is recommended that they be monitored every year undergoing the suggested half survey method (Table 3). This involves interior and exterior vegetation detection transect. Additionally, each of the three plots shall undertake monitoring consisting of the cover board method, seedling plot point pictures, browse index, 5MBC, and coordinated mammal monitoring (Table 6.1).

It is further recommended that all local sites should have an ARD placed within the forest fragment every year, and pitfall traps every five years. However, mammal monitoring, transects, browsing the index, and 5MBC are dependent on the expertise and available resources. Methods of monitoring are described within the following pages of this chapter.

Table 6.1: Proposed recommendations of monitoring scenarios for Hawke's Bay Regional Council to implement. Each site has set monitoring regimes based on their priority of monitoring and ecological importance or significance.

Tasks	Flagship sites	Community sites	Local sites	Time per task per site (h)
	Sites in each zone of the region with high ecological values and strong community support and interest. Regional council led.	Stratified random selection within regional zones, plus sites with strong community interest and access, plus sites of particular ecological importance. Regional council facilitated.	Sites with no public access or no community focus or limited regional ecological importance (but still of priority ecosystem). Regional council input, perhaps QEII Trust led.	
Within each 20x20 plot				
Cover board method	every year	every year (if community volunteers available)	every year	3h
full vegetation survey	every five years	every five years	no	15h
5MBC	every year at same time of year	every year at same time of year	no	30m
browse index in plot	every year	every year (if community volunteers available)	optional (depending on expertise)	1h
5 seedling plots	every year (species + photo)	every year (species + photo)	every year (photos only)	1h
soil	yes every five years	yes every five years (optional)	no	1h
Transect:				
Permanent periphery transects (min. 500 m total in 100 m sections, locations depends on site accessibility)	every year (council weeds, notable rare natives)	every year (council weeds, notable rare natives, with trained community volunteers)	optional frequency (depending on expertise)	2h
10 m wide transects between permanent plots inside core habitat	every year (selected vascular plants of importance)	every year (selected vascular plants of importance, with trained community volunteers)	optional frequency (depending on expertise)	1h
Contractor / arrangement with community or landowner				
residual trap catch	every year	every year (if trapping)	every year (if trapping)	-
chew cards/tracking tunnels	every year (alongside trapping contract work)	every 5 years during full survey	no	-
audio recorders	every year (at least)	every year (if community volunteers available)	every year if possible with collaboration with trapping contractors	1h
Pitfall trap	every year (at least)	every year (if community volunteers available)	every year if possible with collaboration with trapping contractors	30m
Predicted time every year per site (hours):	26	11	9	
Predicted time every 5 years per site (hours):	130	70	45	

6.2.4 Monitoring Task Descriptions

The recommended set of tasks is based on the prioritised ecosystem classifications, and monitoring regimes described above allowing for an assessment of ecological integrity while offering a cost-effective approach for the regional council. These tasks give Regional Councils an insight into the conditions and the urgency to which forests need further monitoring, management, or intervention through a concise manner of monitoring.

The following proposed monitoring protocols are led by the regional council but require ongoing coordination and communication with private landowners, local volunteers, interested communities and organisations, the QEII trust, and contractors. The HBRC is confident that contractors could assist in the mammal monitoring and tracking side of this scheme, and also assist in the collection of ARDs (Mark Mitchell, HBRC, pers. comm.).

Methods used for the full survey, as implemented in this study remain unchanged (Chapter 2). The following are the recommended add-on and reconstructed tasks for the monitoring protocols as discussed above and contained Table 6.1.

Within each site (flagship, community, and local):
Interior and exterior transect vascular plant checklist
The T2BMF species level survey is complex and requires a detailed knowledge of local botanical knowledge, which is crucial in determining a forest's ecological integrity. To assess the plant diversity at a lower intensity, it is recommended that for each site there is a checklist of vascular species found within.

As each of these sites is classified as prioritised based on their ecological value, a species list should contain vascular species of interest such as notable rare natives or of caution/threat such as local council-defined weeds. Each repeat survey may have additional species added to the list. Rapid surveys for vascular vegetation should be performed as fieldworkers enter the site (excluding edge) with a transect line directed towards the surveyed plot. Additionally, rapid surveys should be performed following 10 m wide transects between permanent plots inside the core habitat. These transect lines will differ for each survey but should be predominantly the same and represent the species diversity found within the forest.

As referred to earlier, the T2BMF excludes a broader suite for species detection across the site for monitoring cryptic or vegetation that is missed from the plot. Therefore, it is recommended that there is a permanent periphery transect (min. 500 m total in 100 m sections, locations depend on on-site accessibility), allowing for a consistent means for detecting species that are more cryptic or missing from plots. A list of caution/threat species such as council weeds of the site should be devised and detections for these species along this exterior transect should be targeted.

Initially, this list would need to be created by a skilled botanist for each site, which entails a brief survey of observed species by the transect to permanent plots or when permanent plots are set up whilst a botanist concisely surveys the surrounding area. Plots take a minimum of one hour to set up, and within this time, it allows a botanist to create a species list. In areas of more dense vegetation, it was found that there is a higher species richness (see Chapter 3), which will take longer for the plot to be set up, thus allowing the botanist to collect a more detailed species list.

Autonomous Recorders & 5-Minute Bird Counts

As discussed in Chapter 4, autonomous recorders are immensely beneficial in terms of efficiently detecting birds (and other vocal animals) to measure the integrity of a site. Chapter 2 outlines the recommended protocol by which this study conducted the ARD placement.

It is recommended that ARD are placed in late spring to early autumn so that birds will be recorded in locations where they breed. Application of ARD should avoid bad weather where possible. Each site should be chosen as close to the same time as possible each year (within 3 weeks). The closer together sites are recorded in time, the better. All sites at one Zone should not be recorded later in the season than all sites at another Zone. They should be mixed up so that Zones can be reliably compared.

DOC AR4s recorders, used within my study, are recommended as they are robust, weatherproof, and fully compatible with Tier 1. Although AudioMoth ARD are substantially cheaper, they will require the addition of a waterproof case and, preferably, a soldered-on higher quality microphone.

It is further recommended that when using the DOC AR4s ARD, the protocols for constant night time recording at 8KHz (AR4's Low mode) and constantly at 32KHz (AR4's High mode) for day time (between one hour before sunrise to one hour after sunset) should be used. Batteries should be sufficient for 7-day constant recording at these frequencies. A 32GB SD card will be sufficient for a week's constant recording on the above high-low setting, using the DOC models.

Regional Councils should anticipate a one-off up-front cost of 3–4 days per species to develop reliable species recognition algorithms from training data. For example, if 12 species were ready to process, then a one-week constant recording from each of 30 sites would, in total, take a trained person approximately 2 days to process and provide a report (Laura Molles, Verum Group, personal communication). This is a total of 2 days a year to process all audio recordings from all sites. The machine learning processing is now quick, and accurate when trained on good training data, but still requires a layer of manual checking of potential errors for some species. This will improve over time as the technology improves and the training sets expand. Each year, it is estimated that this process will generate about 800 GB of audio files which will need to be stored and backed up.

An additional recorder would need to be used for bats as they record at much higher frequencies and consequently use their batteries much faster. The same DOC AR4s could be used for this, and set out later, or earlier than the birds. Carol Bedoya (Verum Group), has developed a bat algorithm for DOC that uses the AR4 output with its bat setting. This would process a few nights of recording from 30 sites in much less than an hour. At sites where bats are of interest, it is recommended that a second recorder be set out, and set to the higher recording frequency.

Anyone can place the device in the field and collect the ARD with no barrier to who can collect or place the device, however, it is proposed that coordination with contractors and landowners will mitigate expenditure for the regional council in the placement and receiving process of the actual ARD.

It has been recommended in previous studies (Celis-Murillo et al., 2009; Digby et al., 2013b; Klingbeil & Willig, 2015; Tegeler et al., 2012), that conducting a combination of bird surveys by using both the 5MBC method point count technique, and ARDs may be more effective than using either one alone. ARDs efficiency of detection has a long way to go in terms of the algorithm for automated species recognition. Additionally, the practice and importance of human observation and detection will never diminish in respect to or be replaced by technological advancements such as ARD, as without human detection or certification, monitoring efforts can become weak.

This study recommends both methods of bird monitoring should be conducted providing compensation for each other's limitations and improving the overall assurance of species detectability. Although the integration of both methods increases cost, both in time and monetary terms for regional councils, it provides a robust approach to the current status of species richness and abundance. It is vital that this status understanding is achieved, as there is little prior knowledge.

Invertebrates

It is recommended that Regional Councils assess the entomology found within each of the sites as they play a vital role in various ecosystem functions and respond acutely to habitat manipulation (Maleque et al., 2006). Monitoring invertebrates within the recommended regime is important as there is high diversity, small body size, and high reproductive capacity provide for ease of sampling (Weaver, 1995). Additionally, invertebrates are acutely sensitive to environmental changes and sampling is easy to repeat, thus monitoring is hugely valuable in assessing the ecological integrity of a forest (Maleque et al., 2006; Weaver, 1995). Invertebrates occupy vital positions in food webs, dynamics of populations, and communities. Within ecosystem processes, invertebrates play various roles as predators, herbivores, decomposers, parasites, pollinators, and seed dispersers (Maleque et al., 2006).

As a start, it is recommended that Regional Councils place a pitfall trap beside the ARD. A pitfall trap is a simple device used to catch ground-dwelling invertebrates. In a basic form, a pitfall trap is a buried container, with its top-level to the ground, filled with water and ethanol, or glycol, to preserve specimens (Umair Sial et al., 2022). This trap captures any ground-dwelling invertebrate that wanders near the trap.

This trap, near the positioned ARD, should be placed at the same time as the ARD allowing for efficiency in the placement and retrieval of the pitfall trap. The collected specimens can then be stored and identified later when resources are available. It is recommended that the Regional Councils utilise citizen science platforms such as iNaturalist (Inaturalist, 2021) to identify species, although rapid advances in molecular technology will mean that these will soon be able to be processed with eDNA (as with the ARDs, the slow step is building the DNA barcode library of all the local species—this is something that some NZ Regional Councils are already contributing to).

Residual trap catch

Where possible, residual trap catches within a site should be documented and trends of catch should be monitored allowing for the readjustment and intervention of pest management plans that will then enhance the ecological integrity. Traps are common throughout forests in the Hawkes's Bay region and managed by contractors and/or private landowners. It is therefore recommended that trapping is coordinated by the regional council, but is actioned by either contractors or private landowners, and the trap catch status is then reported back to the regional council for appropriate interventions for an assessment of the integrity of the forest.

Within each of the threes plots per site:

Cover Board Method

To determine a forest integrity assessment of the structure of the vegetation is necessary. Therefore, it is recommended that Regional Councils undertake the cover board method efficiently in time and resources while still achieving a viable assessment of vegetation structure.

The cover board method is used to evaluate vertical cover and vegetation structure, with repeated surveys to measure how this has changed over time. A profile board or density board is used, which are generally narrow boards, usually 1.8 m with bands of alternating colours which define vertical bands or height tiers. The boards estimate the vertical area of the board covered by vegetation from a specified distance (15 m) (Karl & Coleson, 2012; Nudds, 1973). Reading the density or cover board at a uniform distance ensures that all habitats are sampled and compared by the same standard. A density score is then recorded, which corresponds to the mean value of the range of quintiles (percentage of visual of the board across the height tiers) (Nudds, 1973).

This method is fast and allows for easy repetition as is a simple procedure that doesn't require botanical expertise. This method is also ideal for the assessment of prioritised forest fragments as it is suited for a wide variety of vegetation types, but is particularly useful for vegetation types that show potential for change, such as management interventions of predator control and fencing (Karl & Coleson, 2012; Nudds, 1973).

Seedling plots surveys

The survey of seedlings is vital in terms of understanding the succession and regeneration processes of a forest system (Chapter 3). As analytically verified within the first section of this chapter, it is recommended that for an efficient yet valuable assessment, fieldworkers should take a photograph of 0.49 cm across 5 subplots (10 x 10 intercepts and the middle of the plot) (Fig. 6.1). Fieldworkers should take each picture, using a fish-eye lensed (small attachment to a phone and inexpensive) at 20 cm from the ground. This ensures that the picture can spread 0.49 cm of seedling cover, but at a height, that larger individuals or biomass won't block the seedlings which interests the survey. Species may be identified at a later time using these pictures.

This allows for an understanding of seedling composition, not necessarily at a species level, but will indicate regeneration succession. The photograph will provide a percentage of seedling regenerative growth or succession that will offer an understanding of the plot's seedling counts and thus ecological integrity.

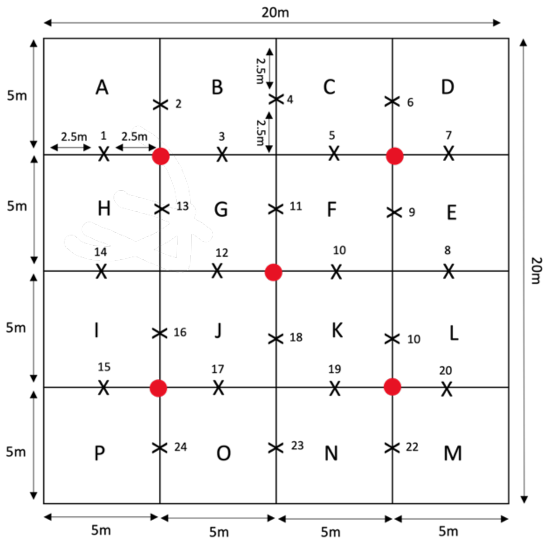


Figure 6.1: Plot diagram of the 5 recommended photo seedling points for a half survey

Browse index

Observing the browse index was an additional data collection within my study (refer to Chapter 2). It is recommended that this should be followed through with the suggested protocol as it gives an index of mammal presence and history of presence. This could be performed by either a council field worker, contractor or private landowners who are experienced in detecting browse on flora.

Pellet counts

Within my study, pellet counts were an adjusted protocol due to resource limitations. However, following the tier 2 monitoring framework, pellet counts should be performed following a 200 m transect line, with counts of a minute search effort time every 5 m, refer to (Department of Conservation, 2013b). Conducting a pellet count within each of the surveyed plots provides an index of mammal presence and history of presence. It is recommended that this task shall be performed by mammal predator contractors, while they lay out bait stations and traps thus this monitoring task should be implemented within their regime.

Chew card/tracking tunnels

As recommended by the T2BMF, but not implemented within this study, chew cards and tracking tunnels should be placed following the standard field protocols (see the T2BMF, Bellingham et al. (2021). This task should be coordinated with local contractors for pest management. It is proposed that as part of their management routine, contractors lay our chew cards or tracking tunnels following the transect line of placement disclosed within the field protocols. This has been proposed as contractors will be able to collect this data for the monitoring scheme in a sense that the mammal monitoring task isn't double-handled (since pest managers or contractors are performing similar routines of management, with or without the prioritised ecosystem project). In turn, this will better allow contractors to actively assess how their management programs are performing.

6.2.5 Engagement with Private Landowners

We all have a major relationship with nature, both in dependence on services and intrinsic values relating to well-being (Ausseil et al., 2021). These relationships are demonstrated in NZ especially as it accounts for many livelihoods as our county's primary sectors are revolved around land usage (Daigneault et al., 2017).

Nearly one-third of NZ land area is owned by private land owners, predominantly farmers in agricultural and horticultural sectors (A. W. Thompson et al., 2015). Particularly sheep and beef farmers support the greatest proportion of remaining woody native forests outside of public conservation land (Maseyk et al., 2021). Native biodiversity is important for the long-term sustainability and resilience of farm systems for ensuring the provision of ecosystem services (Reid &

Norton, 2013), which means that conservation management of native biodiversity on private land and in turn, strong relationships between conservationists and landowners alike is vital.

Consequently, how private land is managed has major implications for native and naturalised biodiversity nationally as decisions have the potential to alter the landscape and the environment, thus shaping the integrity and condition of ecosystems. Private land owners, therefore, have a critical role to play in the conservation of native biodiversity meaning that engaging appropriately with landowners is important as it leads to support for voluntary conservation (A. W. Thompson et al., 2015).

There has been much discussion regarding the need to implement conservation strategies and behaviours toward biodiversity preservation on private land in response to ecological degradation associated with agricultural practices (see (Ausseil et al., 2021; Chandler et al., 2017; Gwen Grelet et al., 2021; Liu et al., 2018; McMurdo Hamilton et al., 2021; Mills et al., 2017; Sheeder, 2009; Smith et al., 2020; A. W. Thompson et al., 2015; van Noppen, 2020)). The greatest quality of success in the conservation of ingenious biodiversity on private land is the engagement and involvement with landowners and the relationships which are formed between local or regional authorities and organisations in assistance. Unless landowners are willing to engage in the protection or preservation of biodiversity, then it won't happen, with or without regional council support.

To encourage the engagement of private landowners to action the conservation protection measures in monitoring regimes and interventions, a focus on the establishment of long-term, cooperative partnerships between landowners, regional councils, and organisations, such as Biodiversity Hawke's Bay and QEII are needed. These interactions endorse larger conservational goals and provide a better understanding of the social variables that motivate landowners to participate in conservation efforts with organisations and authorities that should be investigated.

However, it is difficult for both private landowners and Regional Councils to coordinate management or monitoring regimes. While farmers are responsible for their land, they are also contributing food or fibre production for NZ's largest economic sector, making it challenging to always balance environmental or ecological concerns (A. W. Thompson et al., 2015). Private land in NZ is fundamentally a business and monitoring environmental or ecological expenditures is not always a direct priority, whereas to protect regional biodiversity Regional Councils often need to incorporate private land for ecological or environmental assessments to achieve their conservation strategies. Although the described and proposed recommendations correspond with the HBRC goals, monitoring regimes require cooperation and decisions from private landowners whose attitudes and decisions in response to monitoring will rely on relationships between Regional Councils and their commitment to conservation.

Furthermore, monitoring efforts between regional organisations, such as QEII, are not without difficulty due to both parties having different approaches to monitoring and interactions with landowners. Landowners approach QEII for conservation and protection of forest fragments on land, often to receive financial assistance for fencing. In comparison Regional Councils more often than not approach landowners in an attempt to maintain or monitor the region's biodiversity that they're interested in. As both parties often have the same conservation aspirations, HBRC and other organisations coordinate strategies together well already, it is further recommended that efforts and regimes of monitoring across private stakeholders are needed and councils respond to the synergies in sharing knowledge with other organisations such as QEII.

It has been recommended within this chapter, following procedures like the QEII forest assessments, that communication between regional council, local or interested organisations and communities, and private landowners are required. Within each of the recommended monitoring regimes, council members should interact with the private land owners to get landowner perspectives on current and future biodiversity management. This participation and perspective should be included and highlighted within the survey as a reference to the landowners' observations of the ecological integrity or condition. This will allow councils to have greater scope and precise assessment of the prioritised ecosystem at hand as well as the interaction with encouraging local and personal knowledge.

By coordinating conservation management between private landowners, regional councils, and local organisations across private land, it is suggested that effort, financial resources, and focus should be allocated to integrate landscape preservation. The preservation of integrated landscapes includes sustainable agricultural practices and supporting ecosystem services in ways that can achieve the long-term conservation of biodiversity across the entire landscape (Maseyk et al., 2019; Norton, 2000). This includes the zoning approach proposed in this chapter's recommendations. This requires coordination and involvement with parties at a larger scale to local authorities and organisations, such as the natural resource and consumerism sectors as they delegate the dependency of landscape production by putting pressure on private landowners. Further investigation of the supply and demand relationships within the sector is required to understand the motives and aspirations of private land owners.

6.3 Conclusison

The more intense the sampling or surveys are, the more data is collected. Obtaining a high data collection allows for a more concentrated assessment of ecological integrity, thus allowing for an increased depth in the investigation of the trends and threats to native biodiversity. Collecting more biodiversity data through intense sampling or surveying allows managers to be more content with management or intervention strategies for protecting and enhancing native biodiversity.

This study has provided recommendations concerning regional councils' limited resources to reduce sampling intensities, while still following elements of the T2BMF. My recommendations also offer additional monitoring components in areas not covered by the T2BMF, providing quality and a broader suite of collections for surveys that provide a wider scope of measuring ecological integrity.

I believe that these basic recommendations are tangible and realistic for the HBRC to implement successfully. There is an urgency within the region to monitor and prioritise ecosystems due to the continuous threats that are deteriorating our environment and ecosystem integrity. Therefore, the recommended monitoring regimes of T2BMF should be thoroughly considered, as it provides the regional council with an efficient approach to monitoring the fragile native forest ecosystems throughout the region, so that appropriate management and intervention strategies can be implemented based on the surveys.

Within my study, I have been fortunate enough to work closely with private landowners that displayed a passion and commitment to native biodiversity in the Hawke’s Bay region. I am hopeful that my research’s interactions and findings trigger greater private landowner and local authorities relationships that encourage a further passion for conservation commitments.

Chapter 7
Conclusion

This chapter concludes my thesis and reviews section topics, study findings, and recommendations discussed in previous chapters. This chapter delivers a summary of the research purpose and provides a reflection on the research. I share what new knowledge my study has contributed to the scientific and local community. I also discuss the successfulness of my study in terms of assessing ecological integrity and recommendations for future work.

7.1 Final Disscussion

The overall aim of this thesis was to explore the implementation process of the new T2BMF and to evaluate its merit in delivering forest ecological integrity assessments. It has been found that the T2BMF is a robust systematic framework for assessment native forest ecological integrity, for plant and bird taxa. This study explores the potential to expand on reporting requirements of ecological integrity and biodiversity inventory to follow through with international agreements and obligations, using the T2BMF. The framework's protocol provided data to describe the structural and compositional vegetation of plots, which represented the study sites well.

My research compared naturalised and native species composition, as a high presence of naturalised species can contribute to the degradation of a native forest's ecological integrity. Some naturalised species change species composition in a way that compromises species interactions and the ecological functions or services provided by native species (Hobbs et al., 2009). I found that in forest fragments that were fenced, there were less naturalised species and more native species, indicating that fenced sites provide greater ecological integrity than unfenced sites.

In addition to fencing, forest fragment area can affect forest composition by exposing the species established near the forest edge to climatic or environmental extremes which affect succession and species composition (Innes et al., 2010; Robertson et al., 2019; Young & Mitchell, 1994). The size of forest fragments, and therefore the relative effects of edge and interior conditions, determines the long-term habitat suitability of sites for many native species. Many species exposed to these extremes of forest fragments are otherwise interior species but due to fragmentation and habitat degradation effects, the species are exposed to the edge effect conditions. It was found that the larger forest fragments (and more often than not, fenced), therefore, have the potential to support larger and more diverse populations of fauna and flora. Larger areas also usually means a greater variety of microhabitat conditions, which can also favour higher diversity, with the potential to support more native, specialist species, or interior dwelling species (Tierney et al., 2009b). Edge effects are particularly important for small fragments, but these other area effects are important for fragments of all sizes. Larger areas also support local populations which means more resilience in tough environmental or ecological times.

While I found that fencing benefited forests, even the fenced forest fragments still experienced some negative effects from browsing mammals. This is because the traditional farm fences are inadequate to exclude deer. The ecological integrity of forests will benefit from more effective fencing to exclude deer, and pest mammal control both at the site and in the vicinity.

In addition, the isolation of forest fragmentations reduces species dispersal between fragments, which reduces the ecological resilience of fragments. Reduced dispersal between fragments and makes populations in fragments more vulnerable to local extinction (Barbaro et al., 2012; Ladin et al., 2016; Spiesman et al., 2018), and increased inbreeding can result in reduced genetic diversity, which further puts populations at increased extinction risk (Spiesman et al., 2018). The recommendations made in Chapter 5 enables protection and support for forest fragments networking opportunities in ameliorating ecological integrity.

Forest fragments across Hawkes Bay have increasingly become an asset for private land owners across farmland matrixes. These forest fragments provide habitats for native species that are restricted areas of vegetation of an agricultural landscape. The fragments can assist in the recovery of degraded soils and prevent erosion, which is a frequent problem in dry Hawke's Bay, and they can also provide carbon credits to offset greenhouse gas emissions (Standish et al., 2009).

It is suggested however, that the scale of ecological services that these forest fragments provide, is reliant on the fragment's ecological integrity. Sustaining ecological integrity in these landscapes requires management strategies such as fencing and predator control. However, these management and interventions on private land are dependent on the relationships and the engagement between the landowner and local authorities or organisations.

A leading challenge for this study was the way in which government, multilateral environmental agreements such as the Biodiversity Strategy, QEll, private landowners, and I (as a researcher) negotiated and established culturally appropriate procedures. It's important to respect local cultural knowledge (Lyver et al., 2017). To achieve this, I communicated and involved local authorities (HBRC and QEII) and private landowners whenever possible and actively promoted the sharing of knowledge throughout this study. In doing so, the involvement of various stakeholders has enabled accountable and approachable monitoring incentives such as funds and motivations to fence forest fragments, bait station operations, and local community contributions which encourage appropriate management action for both landowners and Regional Councils.

Before I began my study, it was unclear what was the best sample replication (plots per site) and sample frequencies (plot resamples) needed for councils to monitor the ecological integrity of their prioritised forest fragments. My thesis provides an analysis of sampling intensities for plant and bird species compositions and provides recommendations on sampling intensities, monitoring efforts, and minimal surveys to obtain measurements to gauge a forest's condition. This process has been essential for Regional Councils to apply a monitoring regime in a way which caters to both the urgency of monitoring prioritised sites and the limited resources and expenditure available. I believe that the recommendations provided in this thesis provides the HBRC with a realistic solution to balance their resource restrictions with their need to monitor biodiversity and assess outcomes of biodiversity management.

My study offers an insight into the status, ecological integrity, and pressures on the studied sites. Findings of my study have provided the HBRC with baseline data for future monitoring and research and, I hope, will encourage other councils to follow this new protocol.

7.2 Future Work

In general, monitoring change in the ecological integrity of prioritised ecosystems sites in relation to management practices can be challenging. This is because management occurs against a background of pervasive changes over time, so it is essential that managed and unmanaged sites are monitored. The time frame over which I have monitored is brief. I can make robust statements about how fenced and unfenced sites differ. However, one survey at a time can only predict the successional trajectories of sites, using the species composition and stem densities of the different height tiers of the forest. Further monitoring at these sites will be needed to test those predictions, and to detect whether new threats, such as new pests and weeds or climate changes, are altering these trajectories.

Obtaining a high standard of ecological integrity for our lowland forest ecosystems is of great national significance. Nevertheless, there are concerns surrounding the definition and classifications of forest ecological integrity as it focuses on biodiversity conservation and the unattainable ideal of restoring previous ecosystems (McGlone et al., 2020). Ecological integrity is described as an ideal state of an ecosystem of historical form with indicators of ecological health. This illustrates the potential of the constituent species but can ignore the reality of the current novel ecosystem dynamics.

I believe that the concept of ecological integrity will develop over time as the reality is that a forest ecosystem's true integrity will slowly diminish as we move into a new chapter of ecosystems with everchanging biodiversity dynamics and interactions. We also are transforming as a society in the way we view and value our native biodiversity. Other environmental factors such as climate change and continued biological invasions will also affect how we classify ecological integrity as the world around us changes. Ecological integrity definitions will be forever changing but will always have a purpose for readily understanding an ecosystem's condition and help in providing a quantifiable goal for conservation protection and management.

My study results have the potential to open further discussions about the effects of grazing and or fencing, pest control, and plant invasions. These discussions will allow for improved management practices to help protect and enhance the native biodiversity across farmland.

While my study was confined to already fragmented ecosystems, further exploration of the effects of the fragmentation need be investigated. This might provide evidence of a lack of ecosystem resilience in lowland forest fragments to historical human impacts as suggested by Dodd et al. (2011) and Ewers et al. (2007), sharing similar findings to my study.

Three unfenced surveyed sites have since become, or will soon become, fenced. In 2022 Motumokai was deer fenced, the Rough Block unfenced site has been fully retired with a conventional fence around the perimeter beside the Rough Block fenced site, and the Livesey unfenced site has prospects to be fenced in the near future. Re-surveying these sites in years after fencing will provide important insights into the speed of forest recovery. Such evidence should further inspire private landowners to fence and thus protect native biodiversity on their land.

My recommendations in Chapter 6 offer additional methods for a wider range of taxa. However, some of these recommendations, such as the invertebrates, are speculative. Further work should investigate these methods and the effort necessary to acquire a robust collection of biodiversity data which will ensure an improved assessment of ecological integrity.

With expanding technological advancements and possibilities, it is hard to predict the ways in which we will monitor biodiversity in the future. However, innovators are looking for a transitional approach using technological advancements which are cheaper and involve more efficient methods of collecting biodiversity and ecological data to get a better understanding of an ecosystems integrity (Berger-Tal & Lahoz-Monfort, 2018).

There is debate over how much future biodiversity monitoring could be conducting through remote sensing. Numerous studies (Lawley et al., 2016; Reddy, 2021; Robertson et al., 2019; Schmeller et al., 2017), have found that many species populations can be monitored using remote sensing methods, and can also monitor such details as flowering phenology. However, for plants, this work is largely restricted to what is visible in the canopy, which in a forest can overlook important forest regeneration processes.

Digital spatial imagery for measuring vegetation cover represents an alternative to ground mapping and less biased means of sampling vegetation in terms of the potential for an observer error and observations (Lawley et al., 2016). However, there are ever advancements in technological methods for improving observer performance. Digital spatial imagery approach is also less time-consuming, less environmentally disturbing, and less expensive than vegetation plots. However, the benefits of an observer's visual estimation in vegetation sampling is not always comparable to digital spatial imagery. This is because observers have the advantage of being present in the actual forest studied and experienced observers can provide detailed site descriptions and important insights and through ground-based monitoring that are not possible from remote sensing.

While previous landscapes may have been spatially mapped for forest classification as mature, or well-established, forest fragments, it is possible that there are viable patches of native forests excluded from these surveys. As has occurred in the current study, discussions with landowners and councils are crucial to fully gauge the forest networking opportunities across the farmland matrix. It is suggested by Lawley et al. (2016) and Reddy (2021) that correct mapping of native forest systems and ecosystem types should be performed to better understand the greenspace matrix across the region to get a better understand how these fragments interact and the potential for connecting fragments through regeneration and restoration.

Vegetation sampling for assessing ecological integrity by field observers does come with a degree of error (Lynch, 2017; Morrison, 2016). The main error in vegetation sampling is often the subjectiveness of estimations of vegetation coverage and percentages by observers. Additionally, errors and potential bias can arise where the observer is inexperienced or there are different teams working at different sites (Morrison, 2016). With care, these errors and bias can be minimised and estimated. Further work on the field protocols should focus on workflows that ensure unbiased assessments. Additionally, education about local ecological knowledge should be encouraged among local authorities, organisations, and private landowners.

A large proportion of my time was spent entering my field data from paper sheets into the NVS spreadsheet template. This time could have easily been reduced by technological advancements as fieldworkers could input data digitally on site using a smart device system. Such a system has been developed for DOC's Tier 1 monitoring. This would remove human errors of double handling data and provide security of data as it can immediately become stored in the cloud as soon as internet service is reached. It also makes the data collection and input process quicker and efficient. However, it is suggested that in such cases, a botanical expert surveying the vegetation would be needed to ensure the data and species identification being input is correct. There should always be time after the field work to identify collected specimens.

7.3 Final Conclusions

The purpose of my study was to trial the new T2BMF for lowland Hawkes Bay forests and to evaluate its successfulness in delivering forests ecological integrity assessments. I have described and discussed the framework's components and outcomes for assessing forest ecological integrity, using additional data collected and the T2BMF protocols for both vegetation and birds. It was concluded that fencing does play a significant role in affecting both the vegetation and the birds in these forests, altering the forest structure and the ratio of naturalised and native plant species. Several other features of the sites, aside from fencing, also affected the plant and bird communities, such as the forest area, distance from the coast, elevation, aspect, and the surrounding landscape matrix.

The success of the Framework has been proven as it successfully revealed how site differences in fencing and other factors affected the biodiversity. It is therefore safe to conclude that the Framework, when used for long-term monitoring, will reliably detect and describe changes of similar magnitude that occur over time. The bird and vegetation composition at the different sites was successful in providing assessment of each forest's ecological integrity. The results of the study have identified how useful fences are in protecting the biodiversity at these sites.

The way in which this study implements data collection is adequate; however it needs to be emphasised that the T2BMF is a bare minimum for an assessment of ecological integrity. Overall, my study results show that the assessment in vegetation composition, and from this integrity, works well. However, further monitoring beyond two five minute bird counts per plot is recommended. This study has provided recommendations for a biodiversity monitoring approach for the Hawke's Bay Regional Councils that enables a future focused game plan. The recommendations won't always deliver detailed assessments of the ecological integrity of each forest site, but this is deemed prohibitively expensive. It does, however, offer as a substitute repeatable estimates of forest condition that can inform appropriate management and trigger more intensive Tier 2-type monitoring at focal sites.

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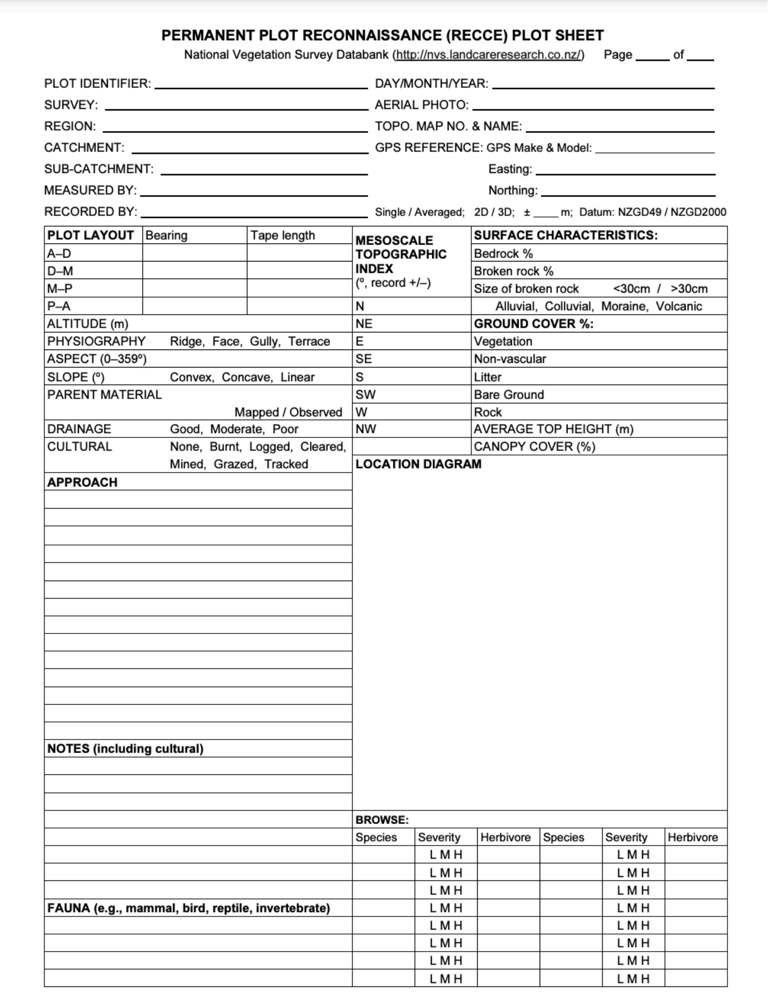
Appendix A

National Vegetation Survey Sheets

A.1 The REECE Sheet side A (blank)



PERMANENT PLOT RECONNAISSANCE (RECCE) PLOT SHEET

National Vegetation Survey Databank (http://nvs.landcareresearch.co.nz/) Page _____ of ____

PLOT IDENTIFIER: ______________________ DAY/MONTH/YEAR: ______________________
SURVEY: ______________________ AERIAL PHOTO: ______________________
REGION: ______________________ TOPO. MAP NO. & NAME: ______________________
CATCHMENT: ______________________ GPS REFERENCE: GPS Make & Model: ______________
SUB-CATCHMENT: ______________________ Easting: ______________________
MEASURED BY: ______________________ Northing: ______________________
RECORDED BY: ______________________ Single / Averaged; 2D / 3D; ± ____ m; Datum: NZGD49 / NZGD2000

PLOT LAYOUT	Bearing	Tape length	**MESOSCALE TOPOGRAPHIC INDEX** (°, record +/–)	**SURFACE CHARACTERISTICS:**
A–D				Bedrock %
D–M				Broken rock %
M–P				Size of broken rock <30cm / >30cm
P–A			N	Alluvial, Colluvial, Moraine, Volcanic
ALTITUDE (m)			NE	**GROUND COVER %:**
PHYSIOGRAPHY	Ridge, Face, Gully, Terrace		E	Vegetation
ASPECT (0–359°)			SE	Non-vascular
SLOPE (°)	Convex, Concave, Linear		S	Litter
PARENT MATERIAL	Mapped / Observed		SW	Bare Ground
			W	Rock
DRAINAGE	Good, Moderate, Poor		NW	AVERAGE TOP HEIGHT (m)
CULTURAL	None, Burnt, Logged, Cleared, Mined, Grazed, Tracked			CANOPY COVER (%)

APPROACH

LOCATION DIAGRAM

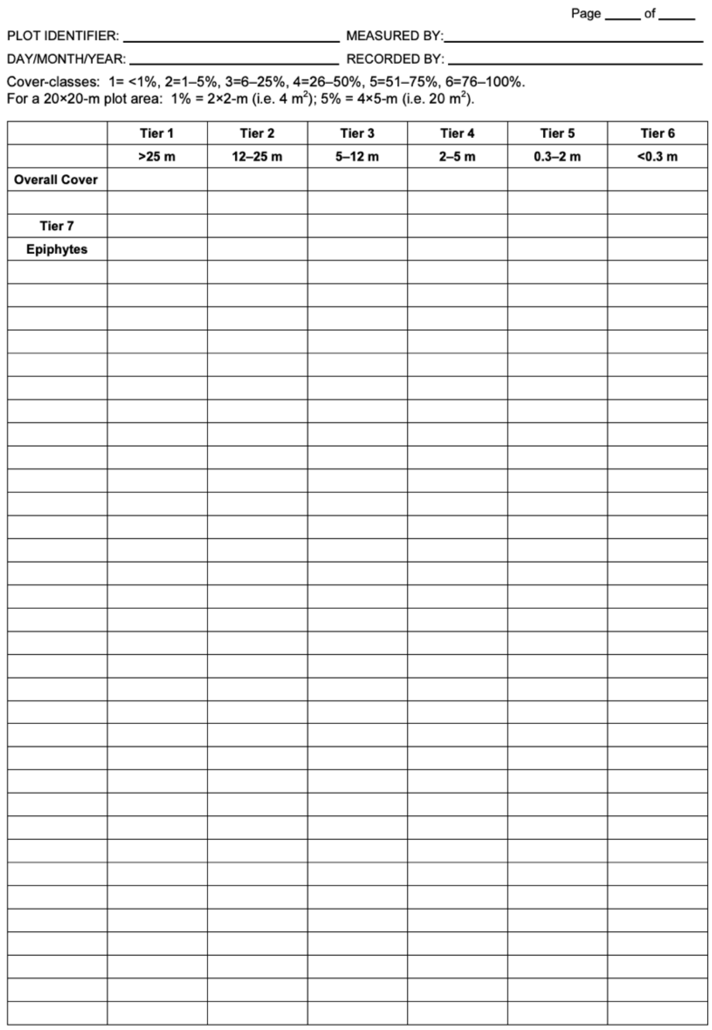
NOTES (including cultural)

FAUNA (e.g., mammal, bird, reptile, invertebrate)

BROWSE:

Species	Severity	Herbivore	Species	Severity	Herbivore
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	

A.2 The REECE Sheet side B (blank)



Page _____ of _____

PLOT IDENTIFIER: ____________________ MEASURED BY: ____________________

DAY/MONTH/YEAR: ____________________ RECORDED BY: ____________________

Cover-classes: 1= <1%, 2=1–5%, 3=6–25%, 4=26–50%, 5=51–75%, 6=76–100%.
For a 20×20-m plot area: 1% = 2×2-m (i.e. 4 m^2); 5% = 4×5-m (i.e. 20 m^2).

	Tier 1	**Tier 2**	**Tier 3**	**Tier 4**	**Tier 5**	**Tier 6**
	>25 m	**12–25 m**	**5–12 m**	**2–5 m**	**0.3–2 m**	**<0.3 m**
Overall Cover						
Tier 7						
Epiphytes						

A.3 The REECE Sheet side A (example of RECCE McLeans)

PERMANENT PLOT RECONNAISSANCE (RECCE) PLOT SHEET

National Vegetation Survey Databank (http://nvs.landcareresearch.co.nz/) Page _____ of ____

PLOT IDENTIFIER: MCP1
DAY/MONTH/YEAR: 2 May
SURVEY: Tier 2 Monitoring project
AERIAL PHOTO:
REGION: Porangahau
TOPO. MAP NO. & NAME:
CATCHMENT:
GPS REFERENCE: GPS Make & Model:
SUB-CATCHMENT:
Easting:
MEASURED BY: Ruby
Northing:
RECORDED BY: Ruby
Single / Averaged; 2D / 3D; ± ____ m; Datum: NZGD49 / NZGD2000

PLOT LAYOUT	Bearing	Tape length
A–D	72	20
D–M	145	20
M–P	234	20
P–A	332	20

ALTITUDE (m)	190 m
PHYSIOGRAPHY	Ridge, [✓ Face], Gully, Terrace
ASPECT (0–359°)	140
SLOPE (°)	[✓ Convex, Concave], Linear
PARENT MATERIAL	Mapped / Observed
DRAINAGE	Good, Moderate, Poor
CULTURAL	[✓ None], Burnt, Logged, Cleared, Mined, Grazed, Tracked

MESOSCALE TOPOGRAPHIC INDEX (°, record +/–)	
N	8
NE	5
E	-6
SE	0
S	6
SW	15
W	16
NW	17

SURFACE CHARACTERISTICS:	
Bedrock %	
Broken rock %	
Size of broken rock	<30cm / >30cm
	Alluvial, Colluvial, Moraine, Volcanic
GROUND COVER %:	
Vegetation	20%
Non-vascular	5
Litter	90
Bare Ground	5
Rock	2
AVERAGE TOP HEIGHT (m)	15m
CANOPY COVER (%)	85%

APPROACH

40 m from fence line (SE)

Sitting on a small face amongst supple jack.

N- lots DACDAC seedlings.

Moved P 5m NE to avoid cliff / bank

NOTES (including cultural)

Huge DACDAC fallen through
subplot between P & 1

Huge amounts of leaf litter.

FAUNA (e.g., mammal, bird, reptile, invertebrate)

Droppings found (P)

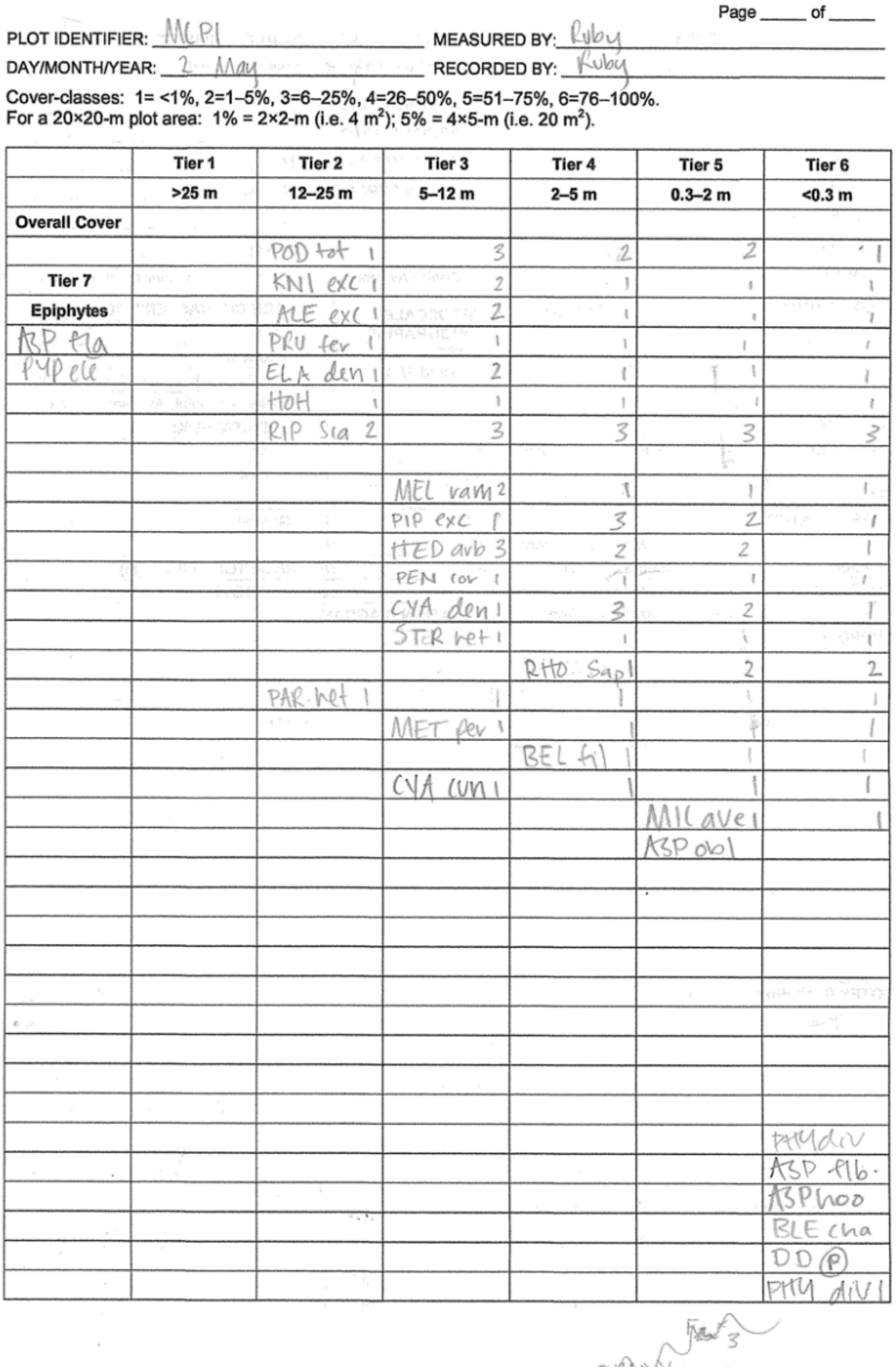
LOCATION DIAGRAM

grass terrace.
20m
Tree line.
20 m
A
D
P
M

BROWSE:

Species	Severity	Herbivore	Species	Severity	Herbivore
	[✓ L] M H		Deer	L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	
	L M H			L M H	

[illegible]



Page _____ of _____

PLOT IDENTIFIER: MCP1 MEASURED BY: Ruby

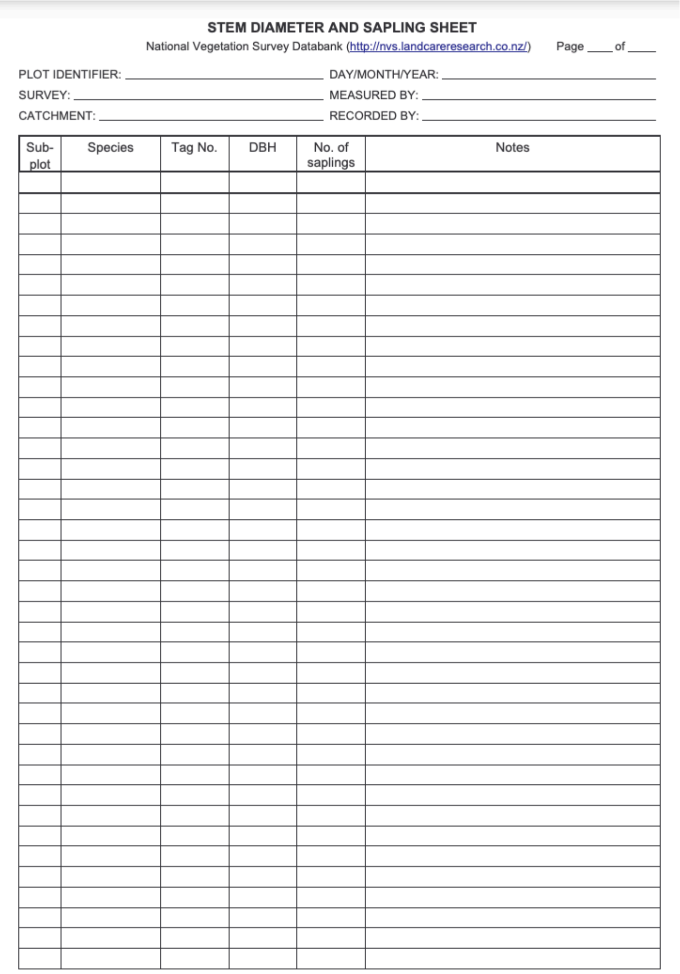
DAY/MONTH/YEAR: 2 May RECORDED BY: Ruby

Cover-classes: 1= <1%, 2=1–5%, 3=6–25%, 4=26–50%, 5=51–75%, 6=76–100%.
For a 20×20-m plot area: 1% = 2×2-m (i.e. 4 m²); 5% = 4×5-m (i.e. 20 m²).

	Tier 1	Tier 2	Tier 3	Tier 4	Tier 5	Tier 6
	>25 m	12–25 m	5–12 m	2–5 m	0.3–2 m	<0.3 m
Overall Cover						
		POD tot 1	3	2	2	1
Tier 7		KNI exc 1	2	1	1	1
Epiphytes		ALE exc 1	2	1	1	1
ASP fla		PRU fer 1	1	1	1	1
PYP ele		ELA den 1	2	1	1	1
		HoH 1	1	1	1	1
		RIP Sca 2	3	3	3	3
			MEL ram 2	1	1	1
			PIP exc 1	3	2	1
			HED arb 3	2	2	1
			PEN cor 1	1	1	1
			CYA den 1	3	2	1
			STER ret 1	1	1	1
				RHO Sap 1	2	2
		PAR ret 1	1	1	1	1
			MET per 1	1	1	1
				BEL fil 1	1	1
			CYA cun 1	1	1	1
					MIC ave 1	1
					ASP obl	
						PHY div
						ASP flb.
						ASP hoo
						BLE cha
						DD Ⓟ
						PHY div 1

A.4 The REECE Sheet side B (example of REECE McLeans)

A.5 Stem Diameter and Sapling Sheet (blank)



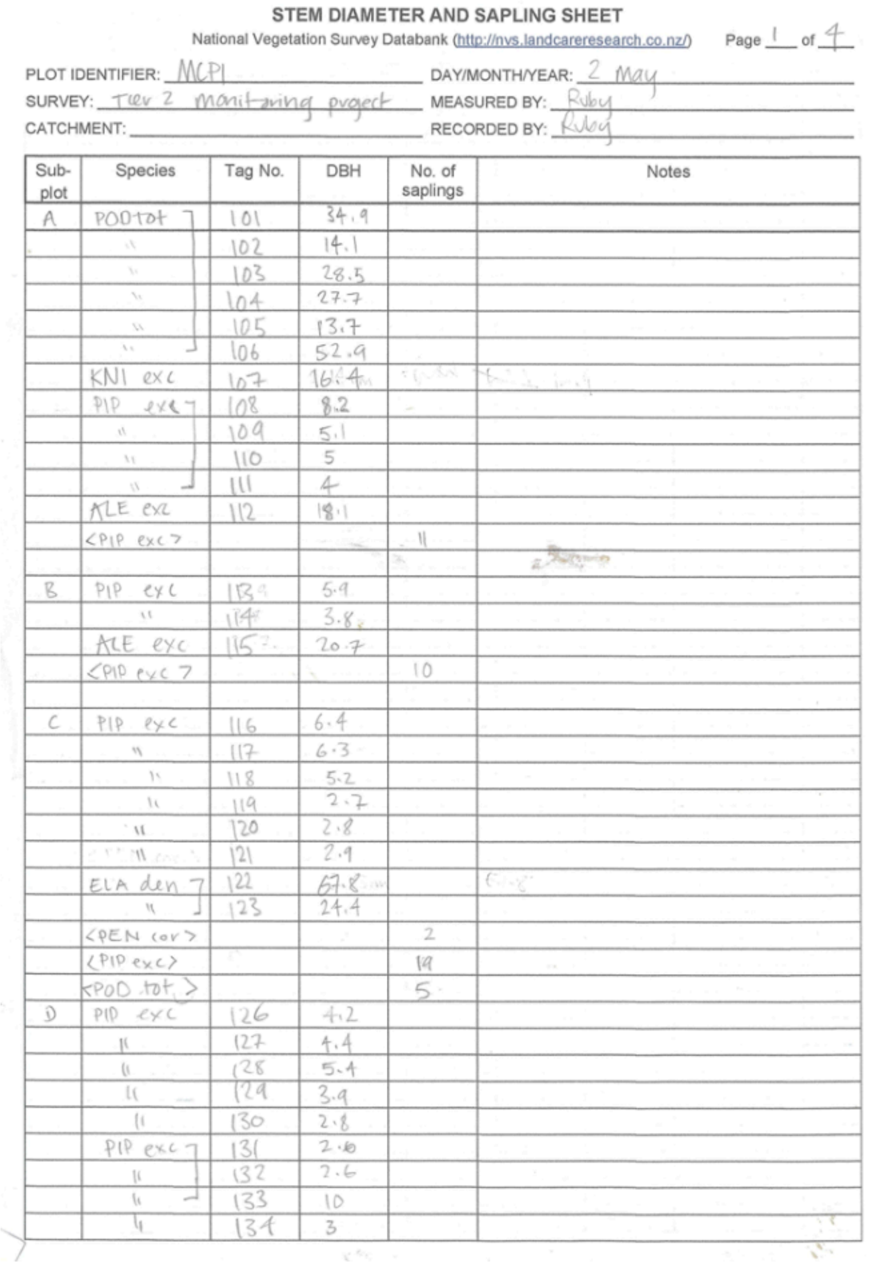
STEM DIAMETER AND SAPLING SHEET

National Vegetation Survey Databank (http://nvs.landcareresearch.co.nz/) Page ____ of ____

PLOT IDENTIFIER: ______________________ DAY/MONTH/YEAR: ______________________
SURVEY: ______________________ MEASURED BY: ______________________
CATCHMENT: ______________________ RECORDED BY: ______________________

Sub-plot	Species	Tag No.	DBH	No. of saplings	Notes

A.6 Stem Diameter and Sapling Sheet (example of McLeans)



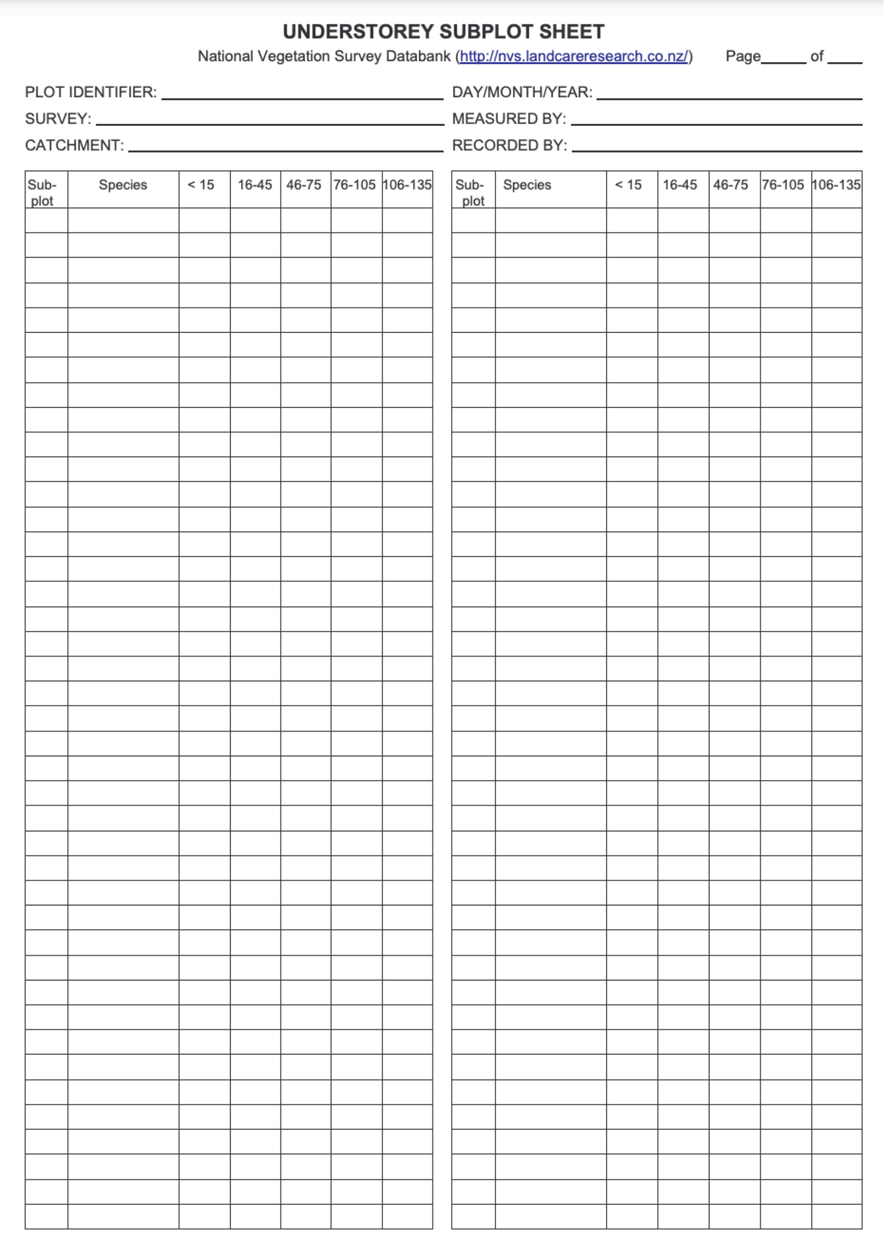
STEM DIAMETER AND SAPLING SHEET

National Vegetation Survey Databank (http://nvs.landcareresearch.co.nz/) Page 1 of 4

PLOT IDENTIFIER: MCP1 DAY/MONTH/YEAR: 2 May
SURVEY: Tier 2 monitoring project MEASURED BY: Ruby
CATCHMENT: RECORDED BY: Ruby

Sub-plot	Species	Tag No.	DBH	No. of saplings	Notes
A	PODtot	101	34.9		
	"	102	14.1		
	"	103	28.5		
	"	104	27.7		
	"	105	13.7		
	"	106	52.9		
	KNI exc	107	16.4		
	PIP exc	108	8.2		
	"	109	5.1		
	"	110	5		
	"	111	4		
	ALE exc	112	18.1		
	<PIP exc>			11	
B	PIP exc	113	5.9		
	"	114	3.8		
	ALE exc	115	20.7		
	<PIP exc>			10	
C	PIP exc	116	6.4		
	"	117	6.3		
	"	118	5.2		
	"	119	2.7		
	"	120	2.8		
	"	121	2.9		
	ELA den	122	67.8		
	"	123	24.4		
	<PEN cor>			2	
	<PIP exc>			19	
	<POD tot>			5	
D	PIP exc	126	4.2		
	"	127	4.4		
	"	128	5.4		
	"	129	3.9		
	"	130	2.8		
	PIP exc	131	2.6		
	"	132	2.6		
	"	133	10		
	"	134	3		

A.7 Understorey Subplot Seedlings Sheet (blank)



UNDERSTOREY SUBPLOT SHEET

National Vegetation Survey Databank (http://nvs.landcareresearch.co.nz/) Page_____ of ____

PLOT IDENTIFIER: ____________________

DAY/MONTH/YEAR: ____________________

SURVEY: ____________________

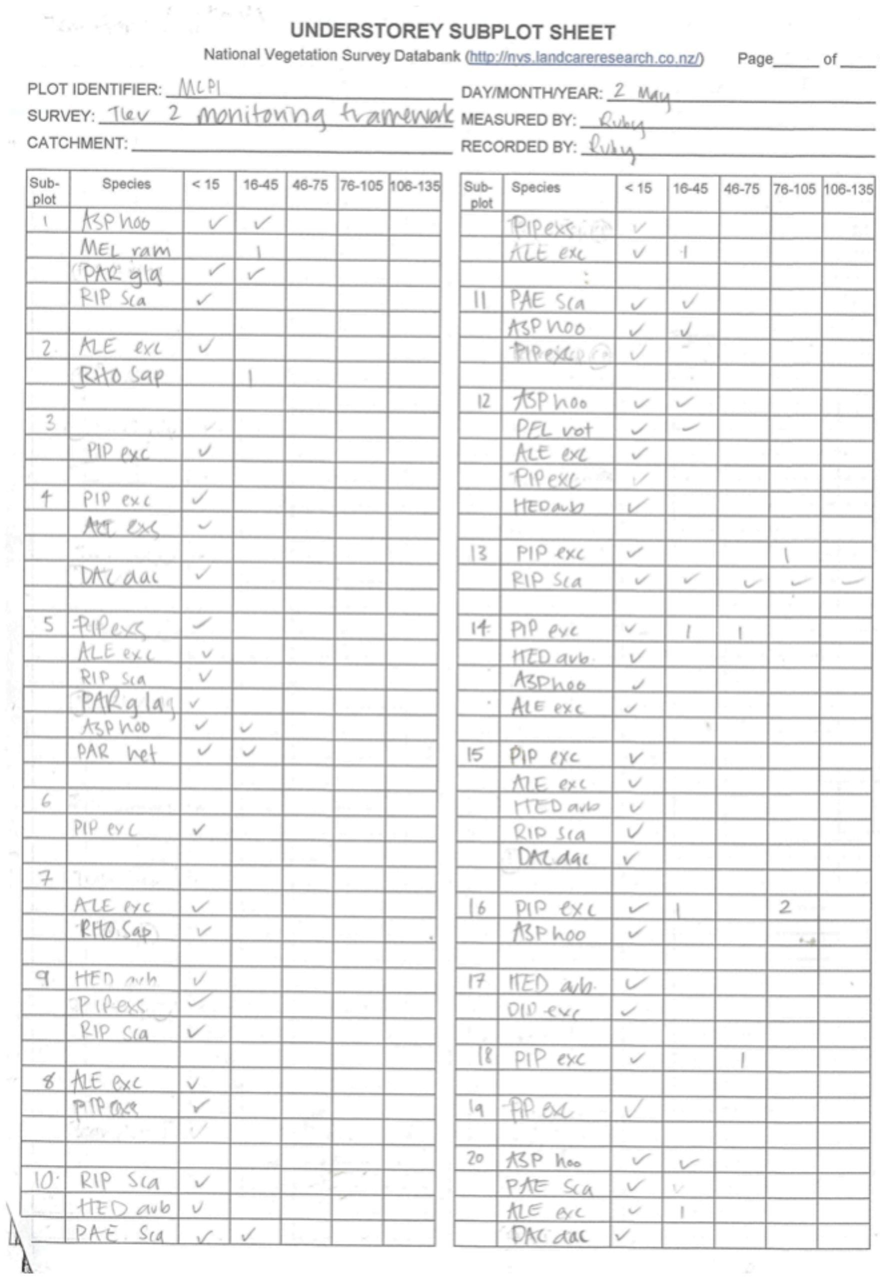
MEASURED BY: ____________________

CATCHMENT: ____________________

RECORDED BY: ____________________

Sub-plot	Species	< 15	16-45	46-75	76-105	106-135	Sub-plot	Species	< 15	16-45	46-75	76-105	106-135

A.8 Understorey Subplot Seedlings Sheet (example of Mcleans)



UNDERSTOREY SUBPLOT SHEET

National Vegetation Survey Databank (http://nvs.landcareresearch.co.nz/) Page_____ of ____

PLOT IDENTIFIER: MCP1

SURVEY: Tier 2 monitoring framework

CATCHMENT:

DAY/MONTH/YEAR: 2 May

MEASURED BY: Ruby

RECORDED BY: Ruby

Sub-plot	Species	< 15	16-45	46-75	76-105	106-135
1	ASP hoo	✓	✓			
	MEL ram		1			
	PAR gla	✓	✓			
	RIP Sca	✓				
2	ALE exc	✓				
	RHO Sap		1			
3						
	PIP exc	✓				
4	PIP exc	✓				
	ALE exc	✓				
	DAC dac	✓				
5	PIP exc	✓				
	ALE exc	✓				
	RIP Sca	✓				
	PAR gla	✓				
	ASP hoo	✓	✓			
	PAR het	✓	✓			
6						
	PIP exc	✓				
7						
	ALE exc	✓				
	RHO Sap	✓				
9	HED arb	✓				
	PIP exc	✓				
	RIP Sca	✓				
8	ALE exc	✓				
	PIP exc	✓				
10	RIP Sca	✓				
	HED arb	✓				
	PAE Sca	✓	✓			

Sub-plot	Species	< 15	16-45	46-75	76-105	106-135
	PIP exc	✓				
	ALE exc	✓	1			
11	PAE Sca	✓	✓			
	ASP hoo	✓	✓			
	PIP exc	✓				
12	ASP hoo	✓	✓			
	PEL vot	✓	✓			
	ALE exc	✓				
	PIP exc	✓				
	HED arb	✓				
13	PIP exc	✓			1	
	RIP Sca	✓	✓	✓	✓	✓
14	PIP exc	✓	1	1		
	HED arb	✓				
	ASP hoo	✓				
	ALE exc	✓				
15	PIP exc	✓				
	ALE exc	✓				
	HED arb	✓				
	RIP Sca	✓				
	DAC dac	✓				
16	PIP exc	✓	1		2	
	ASP hoo	✓				
17	HED arb	✓				
	PIP exc	✓				
18	PIP exc	✓		1		
19	PIP exc	✓				
20	ASP hoo	✓	✓			
	PAE Sca	✓	✓			
	ALE exc	✓	1			
	DAC dac	✓				

Appendix B

Site Species Lists

Table 7.1: Species detected for each studied site. Bio status and growth forms are included.

Species Name	Bio Status & Growth Form
Kirk (fenced)	
Alectryon excelsus	Native - Tree
Arctium minus	Endemic - Shrub
Asplenium hookerianum	Native - Fern
Australina pusilla	Native - Forb
Carex divulsa	Endemic - Graminoid
Cirsium vulgare	Endemic - Forb
Conyza sumatrensis	Endemic - Forb
Coprosma areolata	Native - Tree
Coprosma rhamnoides	Native - Shrub
Coprosma robusta	Native - Tree
Coprosma rotundifolia	Native - Shrub
Corynocarpus laevigatus	Native - Tree
Dacrycarpus dacrydioides	Native - Tree
Digitalis purpurea	Endemic - Forb
Ehrharta stipoides	Exotic - Graminoid
Hedycarya arborea	Native - Tree
Hoheria sexstylosa	Native - Tree
Lophomyrtus obcordata	Native - Tree
Melicytus ramiflorus	Native - Tree
Melicope simplex	Native - Tree
Metrosideros diffusa	Native - Vine
Metrosideros perforata	Native - Vine
Microsorum pustulatum	Native - Fern
Muehlenbeckia australis	Native - Vine
Mycelis muralis	Endemic - Forb
Myrsine australis	Native - Tree
Parietaria debilis	Native - Shrub
Parsonsia heterophylla	Native - Vine
Pellaea rotundifolia	Native - Fern
Pennantia corymbosa	Native - Tree
Piper excelsum	Native - Tree
Pittosporum eugenioides	Native - Tree
Pittosporum tenuifolium	Native - Tree
Podocarpus totara	Native - Tree
Polystichum neozelandicum	Native - Fern

Polystichum neozelandicum subsp. zerophyllum	Native - Fern
Pseudopanax crassifolius	Native - Tree
Pteris tremula	Native - Fern
Ranunculus repens	Endemic - Forb
Ripogonum scandens	Native - Vine
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Solanum pseudocapsicum	Endemic - Shrub
Streblus heterophyllus	Native - Tree
Urtica ferox	Native - Shrub
Dave (Unfenced)	
Adiantum cunninghamii	Native - Fern
Agrostis capillaris	Endemic - Graminoid
Alectryon excelsus	Native - Tree
Alopecurus pratensis	Endemic - Graminoid
Arthropteris tenella	Native - Fern
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Beilschmiedia tawa	Native - Tree
Bromus diandrus	Endemic - Graminoid
CARDAMINE SPP.	Exotic - Forb
Carex divulsa	Endemic - Graminoid
Carduus nutans	Endemic - Forb
Carpodetus serratus	Native - Tree
Cerastium fontanum	Endemic - Forb
Clematis vitalba	Endemic - Vine
Conyza sumatrensis	Endemic - Forb
Coprosma areolata	Native - Tree
Coprosma rhamnoides	Native - Shrub
Coprosma rotundifolia	Native - Shrub
Dactylis glomerata	Endemic - Graminoid
Dichondra repens	Native - Forb
Digitalis purpurea	Endemic - Forb
Echinopogon ovatus	Native - Graminoid
Ehrharta stipoides	- Graminoid
Fragaria vesca	Endemic - Forb
Geranium molle	Endemic - Forb
Holcus lanatus	Endemic - Graminoid
Hydrocotyle elongata	Native - Forb
Hydrocotyle microphylla	Native - Forb
Hypochaeris radicata	Endemic - Forb
Lapsana communis	Endemic - Forb
Lastreopsis glabella	Native - Fern
Lophomyrtus obcordata	Native - Tree
Melicytus micranthus	Native - Shrub

Melicytus ramiflorus	Native - Tree
Melicope simplex	Native - Tree
Metrosideros diffusa	Native - Vine
Muehlenbeckia australis	Native - Vine
Mycelis muralis	Endemic - Forb
Parietaria debilis	Native - Shrub
Parsonsia heterophylla	Native - Vine
Pellaea rotundifolia	Native - Fern
Pennantia corymbosa	Native - Tree
Piper excelsum	Native - Tree
Plantago lanceolata	Endemic - Forb
Podocarpus totara	Native - Tree
Polystichum neozelandicum	Native - Fern
Polystichum neozelandicum subsp. zerophyllum	Native - Fern
Prunella vulgaris	Endemic - Forb
Pteris tremula	Native - Fern
Pyrrosia eleagnifolia	Native - Fern
Ranunculus reflexus	Native - Forb
Ranunculus repens	Endemic - Forb
Ripogonum scandens	Native - Vine
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Solanum nigrum	Endemic - Shrub
Sophora microphylla	Native - Tree
Stellaria media	Endemic - Forb
Streblus heterophyllus	Native - Tree
Teucrium scorodonia	Endemic - Shrub
Trifolium repens	Endemic - Forb
Trifolium subterraneum	Endemic - Forb
Urtica ferox	Native - Shrub
Roughblock (fenced)	
Acianthus sinclairii	Native - Forb
Adiantum cunninghamii	Native - Fern
Agrostis capillaris	Endemic - Graminoid
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Brachyglottis repanda	Native - Tree
Carmichaelia australis	Native - Tree
Carex divulsa	Endemic - Graminoid
Carpodetus serratus	Native - Tree
Celmisia gracilenta	Native - Forb
Cirsium vulgare	Endemic - Forb
Clematis forsteri	Native - Vine
Conyza sumatrensis	Endemic - Forb
Coprosma rhamnoides	Native - Shrub

Dichondra repens	Native - Forb
Echinopogon ovatus	Native - Graminoid
Euchiton japonicus	Native - Forb
Euchiton sphaericus	Native - Forb
Galium propinquum	Native - Forb
Geniostoma ligustrifolium	Native - Shrub
Hedycarya arborea	Native - Tree
Helichrysum lanceolatum	Native - Shrub
Holcus lanatus	Endemic - Graminoid
Hydrocotyle elongata	Native - Forb
Hymenophyllum revolutum	Native - Fern
Hypochaeris radicata	Endemic - Forb
Knightia excelsa	Native - Tree
Kunzea ericoides	Native - Tree
Lapsana communis	Endemic - Forb
Leucopogon fasciculatus	Native - Shrub
Melicytus ramiflorus	Native - Tree
Metrosideros perforata	Native - Vine
Microlaena avenacea	Native - Graminoid
Microsorum pustulatum	Native - Fern
Olearia furfuracea	Native - Tree
Paesia scaberula	Native - Fern
Parsonsia heterophylla	Native - Vine
Pellaea rotundifolia	Native - Fern
Pennantia corymbosa	Native - Tree
Piper excelsum	Native - Tree
Pittosporum eugenioides	Native - Tree
Polystichum neozelandicum	Native - Fern
Pyrrosia eleagnifolia	Native - Fern
Ranunculus reflexus	Native - Forb
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Rytidosperma buchananii	Native - Graminoid
Senecio minimus	Native - Forb
Stellaria parviflora	Native - Forb
UNCINIA Spp.	Native - Forb
Uncinia uncinata	Native - Graminoid
Roughblock (unfenced)	
Acianthus sinclairii	Native - Forb
Agrostis capillaris	Endemic - Graminoid
Alectryon excelsus	Native - Tree
Asplenium flaccidum	Native - Fern
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Carmichaelia australis	Native - Tree
Carex divulsa	Endemic - Graminoid

Carduus nutans	Endemic - Forb
Cirsium vulgare	Endemic - Forb
Conyza sumatrensis	Endemic - Forb
Coprosma rhamnoides	Native - Shrub
Cordyline australis	Native, Gt]
Corynocarpus laevigatus	Native - Tree
DICONDRA Spp.	Native - Forb
Dichondra repens	Native - Forb
Digitalis purpurea	Endemic - Forb
Echinopogon ovatus	Native - Graminoid
Ehrharta stipoides	- Graminoid
Elytrigia repens	Endemic - Graminoid
Euchiton japonicus	Native - Forb
Euchiton sphaericus	Native - Forb
Galium propinquum	Native - Forb
Helminthotheca echioides	Endemic - Forb
Helichrysum lanceolatum	Native - Shrub
Holcus lanatus	Endemic - Graminoid
Hydrocotyle elongata	Native - Forb
Hypochaeris radicata	Endemic - Forb
Kunzea ericoides	Native - Tree
Lastreopsis hispida	Native - Fern
Leucopogon fasciculatus	Native - Shrub
Luzula multiflora	Endemic - Graminoid
Microlaena avenacea	Native - Graminoid
Microsorum pustulatum	Native - Fern
Muehlenbeckia australis	Native - Vine
Muehlenbeckia complexa	Native - Vine
Mycelis muralis	Endemic - Forb
Oxalis corniculata	Endemic - Forb
Ozothamnus leptophyllus	Native - Shrub
Pellaea rotundifolia	Native - Fern
Piper excelsum	Native - Tree
Polystichum neozelandicum	Native - Fern
Prunella vulgaris	Endemic - Forb
Pteris tremula	Native - Fern
Pyrrosia eleagnifolia	Native - Fern
Ranunculus reflexus	Native - Forb
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Rytidosperma buchananii	Native - Graminoid
Solanum nigrum	Endemic - Shrub
Sonchus oleraceus	Endemic - Forb
Sophora microphylla	Native - Tree
Stellaria parviflora	Native - Forb
Urtica ferox	Native - Shrub

Livesey (fenced)	
Adiantum cunninghamii	Native - Fern
Aristotelia serrata	Native - Tree
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Blechnum aggregatum	Native - Fern
Blechnum chambersii	Native - Fern
Blechnum discolor	Native - Fern
Blechnum fluviatile	Native - Fern
Blechnum minus	Native - Fern
Blechnum novae-zelandiae	Native - Fern
Blechnum penna-marina	Native - Fern
Blechnum procerum	Native - Fern
Brachyglottis repanda	Native - Tree
Carduus nutans	Endemic - Forb
Carpodetus serratus	Native - Tree
Centella uniflora	Native - Forb
Cirsium arvense	Endemic - Forb
Cirsium vulgare	Endemic - Forb
Clematis foetida	Native - Vine
Clematis paniculata	Native - Vine
Coprosma autumnalis	Native - Shrub
Coprosma lucida	Native - Tree
Coprosma rhamnoides	Native - Shrub
Coprosma rigida	Native - Shrub
Cyathea medullaris	Native - Tree fern
Dactylis glomerata	Endemic - Graminoid
Dichondra repens	Native - Forb
Dicksonia squarrosa	Native - Tree fern
Echinopogon ovatus	Native - Graminoid
Ehrharta stipoides	- Graminoid
Festuca rubra	Endemic - Graminoid
Fuchsia excorticata	Native - Tree
Galium propinquum	Native - Forb
Gaultheria antipoda	Native - Shrub
Geniostoma ligustrifolium	Native - Shrub
Hedycarya arborea	Native - Tree
Helichrysum lanceolatum	Native - Shrub
Hoheria sexstylosa	Native - Tree
Holcus lanatus	Endemic - Graminoid
Hydrocotyle elongata	Native - Forb
Hypolepis ambigua	Native - Fern
Kunzea ericoides	Native - Tree
Leptecophylla juniperina	Native - Shrub
Leucopogon fasciculatus	Native - Shrub
Melicytus ramiflorus	Native - Tree

Microlaena stipoides	Native - Graminoid
Muehlenbeckia australis	Native - Vine
Mycelis muralis	Endemic - Forb
Oxalis corniculata	Endemic - Forb
Oxalis incarnata	Endemic - Forb
Paesia scaberula	Native - Fern
Parsonsia heterophylla	Native - Vine
Passiflora tetrandra	Native - Vine
Pittosporum eugenioides	Native - Tree
Plagianthus regius	Native - Tree
Pneumatopteris pennigera	Native - Fern
Polystichum neozelandicum	Native - Fern
Polystichum vestitum	Native - Fern
Pseudopanax arboreus	Native - Tree
Pseudopanax crassifolius	Native - Tree
Pterostylis alobula	Native - Forb
Pyrrosia eleagnifolia	Native - Fern
Ranunculus repens	Endemic - Forb
Ripogonum scandens	Native - Vine
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Schefflera digitata	Native - Tree
Senecio bipinnatisectus	Native - Forb
Solanum nigrum	Endemic - Shrub
Stellaria parviflora	Native - Forb
Teucrium scorodonia	Endemic - Shrub
Urtica ferox	Native - Shrub
Livesey (unfenced)	
Acaena anserinifolia	Native - Forb
Acaena novae-zelandiae	Native - Forb
Agrostis capillaris	Endemic - Graminoid
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Blechnum fluviatile	Native - Fern
Blechnum penna-marina	Native - Fern
CARDAMINE SPP.	Exotic - Forb
Carex divulsa	Endemic - Graminoid
Carduus nutans	Endemic - Forb
Carpodetus serratus	Native - Tree
Cerastium fontanum	Endemic - Forb
Chenopodium album	Endemic - Forb
Cirsium arvense	Endemic - Forb
Cirsium vulgare	Endemic - Forb
Conyza sumatrensis	Endemic - Forb
Coprosma rhamnoides	Native - Shrub
Coprosma robusta	Native - Tree

Crepis capillaris	Endemic - Forb
Cynosurus cristatus	Endemic - Graminoid
Dactylis glomerata	Endemic - Graminoid
Dichondra repens	Native - Forb
Echinopogon ovatus	Native - Graminoid
Euchiton sphaericus	Native - Forb
Geranium molle	Endemic - Forb
Helminthotheca echioides	Endemic - Forb
Helichrysum lanceolatum	Native - Shrub
Hoheria sexstylosa	Native - Tree
Holcus lanatus	Endemic - Graminoid
Hordeum murinum	Endemic - Graminoid
Hydrocotyle elongata	Native - Forb
Hydrocotyle heteromeria	Native - Forb
Hydrocotyle microphylla	Native - Forb
Hydrocotyle moschata	Native - Forb
Hypolepis ambigua	Native - Fern
Hypochaeris radicata	Endemic - Forb
Jacobaea vulgaris	Endemic - Forb
Kunzea ericoides	Native - Tree
Lapsana communis	Endemic - Forb
Leontodon saxatilis	Endemic - Forb
Lobelia angulata	Native - Forb
Lolium perenne	Endemic - Graminoid
Luzula multiflora	Endemic - Graminoid
Melicytus ramiflorus	Native - Tree
Microlaena stipoides	Native - Graminoid
Muehlenbeckia australis	Native - Vine
Muehlenbeckia complexa	Native - Vine
Mycelis muralis	Endemic - Forb
Nertera villosa	Native - Shrub
Oxalis corniculata	Endemic - Forb
Oxalis exilis	Native - Forb
Oxalis incarnata	Endemic - Forb
Paesia scaberula	Native - Fern
Parsonsia heterophylla	Native - Vine
Passiflora tetrandra	Native - Vine
Pellaea rotundifolia	Native - Fern
Pittosporum eugenioides	Native - Tree
Polystichum neozelandicum	Native - Fern
Prunella vulgaris	Endemic - Forb
Pseudopanax crassifolius	Native - Tree
Pteris tremula	Native - Fern
Pyrrosia eleagnifolia	Native - Fern
Ranunculus repens	Endemic - Forb
Ripogonum scandens	Native - Vine

Rubus fruticosus	Endemic - Shrub
Rumex pulcher	Endemic - Forb
Rytidosperma buchananii	Native - Graminoid
Senecio bipinnatisectus	Native - Forb
Solanum nigrum	Endemic - Shrub
Sonchus oleraceus	Endemic - Forb
Stellaria media	Endemic - Forb
Stellaria parviflora	Native - Forb
Taraxacum officinale	Endemic - Forb
Teucrium scorodonia	Endemic - Shrub
Trifolium glomeratum	Endemic - Forb
Trifolium micranthum	Endemic - Forb
Trifolium repens	Endemic - Forb
Urtica ferox	Native - Shrub
McLeans (fenced)	
Alectryon excelsus	Native - Tree
Anthosachne solandri	Native - Graminoid
Asplenium flaccidum	Native - Fern
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Asplenium oblongifolium	Native - Fern
Beilschmiedia tawa	Native - Tree
Blechnum chambersii	Native - Fern
Blechnum filiforme	Native - Fern
Brachyglottis repanda	Native - Tree
Carex divulsa	Endemic - Graminoid
Carpodetus serratus	Native - Tree
Cirsium vulgare	Endemic - Forb
CARDAMINE SPP.	Native - Shrub
Coprosma areolata	Native - Tree
Coprosma australis	Native - Tree
Coprosma rhamnoides	Native - Shrub
Coprosma robusta	Native - Tree
Cordyline australis	Native - Grass tree
Cyathea cunninghamii	Native - Tree fern
Cyathea dealbata	Native - Tree fern
Dacrycarpus dacrydioides	Native - Tree
Dichondra repens	Native - Forb
Elaeocarpus dentatus	Native - Tree
Geniostoma ligustrifolium	Native - Shrub
Hedycarya arborea	Native - Tree
Hoheria sexstylosa	Native - Tree
Hydrocotyle microphylla	Native - Forb
Knightia excelsa	Native - Tree
Lastreopsis glabella	Native - Fern
Melicytus ramiflorus	Native - Tree

Melicope simplex	Native - Tree
Melicope ternata	Native - Tree
Metrosideros perforata	Native - Vine
Microlaena avenacea	Native - Graminoid
Microsorum pustulatum	Native - Fern
Muehlenbeckia australis	Native - Vine
Muehlenbeckia complexa	Native - Vine
Mycelis muralis	Endemic - Forb
Myrsine australis	Native - Tree
Oxalis exilis	Native - Forb
Paesia scaberula	Native - Fern
Parsonsia heterophylla	Native - Vine
Pellaea rotundifolia	Native - Fern
Pennantia corymbosa	Native - Tree
Piper excelsum	Native - Tree
Pittosporum eugenioides	Native - Tree
Pneumatopteris pennigera	Native - Fern
Podocarpus totara	Native - Tree
Polystichum neozelandicum	Native - Fern
Polystichum vestitum	Native - Fern
Prumnopitys ferruginea	Native - Tree
Pseudopanax crassifolius	Native - Tree
Pyrrosia eleagnifolia	Native - Fern
Ranunculus repens	Endemic - Forb
Rhopalostylis sapida	Native - Palm
Ripogonum scandens	Native - Vine
Rubus cissoides	Native - Shrub
Solanum nigrum	Endemic - Shrub
Stellaria parviflora	Native - Forb
Streblus heterophyllus	Native - Tree
Trifolium repens	Endemic - Forb
Uncinia uncinata	Native - Graminoid
Motu Mokai (unfenced)	
Acaena anserinifolia	Native - Forb
Adiantum cunninghamii	Native - Fern
Agrostis capillaris	Endemic - Graminoid
Alectryon excelsus	Native - Tree
Anthosachne solandri	Native - Graminoid
Asplenium flaccidum	Native - Fern
Asplenium hookerianum	Native - Fern
Azorella hookeri	Native - Fern
Bellis perennis	Endemic - Forb
Blechnum filiforme	Native - Fern
Blechnum fluviatile	Native - Fern
Carmichaelia australis	Native - Tree
CARDAMINE SPP.	Exotic - Forb

Carex divulsa	Endemic - Graminoid
Carpodetus serratus	Native - Tree
Cerastium fontanum	Endemic - Forb
Cirsium arvense	Endemic - Forb
Cirsium vulgare	Endemic - Forb
Conyza sumatrensis	Endemic - Forb
Coprosma areolata	Native - Tree
Coprosma australis	Native - Tree
Coprosma rhamnoides	Native - Shrub
Coprosma rigida	Native - Shrub
Cynosurus cristatus	Endemic - Graminoid
Dactylis glomerata	Endemic - Graminoid
DICONDRA Spp.	Native - Forb
Euchiton sphaericus	Native - Forb
Geniostoma ligustrifolium	Native - Shrub
Geranium molle	Endemic - Forb
Hedycarya arborea	Native - Tree
Hoheria sexstylosa	Native - Tree
Holcus lanatus	Endemic - Graminoid
Hydrocotyle elongata	Native - Forb
Hydrocotyle microphylla	Native - Forb
Hydrocotyle moschata	Native - Forb
Hypochaeris radicata	Endemic - Forb
Knightia excelsa	Native - Tree
Kunzea ericoides	Native - Tree
Lastreopsis glabella	Native - Fern
Melicytus micranthus	Native - Shrub
Melicytus ramiflorus	Native - Tree
Metrosideros perforata	Native - Vine
Microlaena avenacea	Native - Graminoid
Muehlenbeckia australis	Native - Vine
Muehlenbeckia complexa	Native - Vine
Oxalis exilis	Native - Forb
Paesia scaberula	Native - Fern
Parsonsia heterophylla	Native - Vine
Passiflora tetrandra	Native - Vine
Pellaea rotundifolia	Native - Fern
Piper excelsum	Native - Tree
Plantago lanceolata	Endemic - Forb
Poa pratensis	Endemic - Graminoid
Poa trivialis	Endemic - Graminoid
Podocarpus totara	Native - Tree
Polystichum neozelandicum	Native - Fern
Prunella vulgaris	Endemic - Forb
Pseudopanax crassifolius	Native - Tree
Pterostylis alobula	Native - Forb

Pyrrosia eleagnifolia	Native - Fern
Ranunculus reflexus	Native - Forb
Ranunculus repens	Endemic - Forb
Ripogonum scandens	Native - Vine
Rubus cissoides	Native - Shrub
Rytidosperma buchananii	Native - Graminoid
Senecio bipinnatisectus	Native - Forb
Sophora microphylla	Native - Tree
Stellaria media	Endemic - Forb
Stellaria parviflora	Native - Forb
Streblus heterophyllus	Native - Tree
Trifolium glomeratum	Endemic - Forb
Trifolium repens	Endemic - Forb
Ulex europaeus	Endemic - Shrub
Uncinia uncinata	Native - Graminoid
Veronica serpyllifolia	Endemic - Forb
Vicia cracca	Endemic - Forb
Mark (fenced)	
Adiantum cunninghamii	Native - Fern
Adiantum hispidulum	Native - Fern
Alectryon excelsus	Native - Tree
Anthosachne solandri	Native - Graminoid
Asplenium flaccidum	Native - Fern
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Asplenium polyodon	Native - Fern
Blechnum chambersii	Native - Fern
Blechnum filiforme	Native - Fern
Blechnum parrisiae	Native - Fern
Brachyglottis repanda	Native - Tree
Carex divulsa	Endemic - Graminoid
Carex inversa	Native - Graminoid
Cardamine uniflora	Native - Forb
Cirsium vulgare	Endemic - Forb
Clematis forsteri	Native - Vine
Conyza sumatrensis	Endemic - Forb
Coprosma australis	Native - Tree
Coprosma ciliata	Native - Shrub
Coprosma rhamnoides	Native - Shrub
Coprosma robusta	Native - Tree
Cynodon dactylon	Endemic - Graminoid
Dactylis glomerata	Endemic - Graminoid
DICONDRA Spp.	Native - Forb
Digitalis purpurea	Endemic - Forb
Doodia australis	Native - Fern
Erigeron karvinskianus	Endemic - Forb

Euchiton japonicus	Native - Forb
Griselinia littoralis	Native - Tree
Hedycarya arborea	Native - Tree
Helichrysum lanceolatum	Native - Shrub
Hoheria sexstylosa	Native - Tree
Holcus lanatus	Endemic - Graminoid
Hypochaeris radicata	Endemic - Forb
Kunzea ericoides	Native - Tree
Lastreopsis glabella	Native - Fern
Luzula multiflora	Endemic - Graminoid
Melicytus ramiflorus	Native - Tree
Microsorum pustulatum	Native - Fern
Muehlenbeckia australis	Native - Vine
Muehlenbeckia complexa	Native - Vine
Mycelis muralis	Endemic - Forb
Oxalis exilis	Native - Forb
Pellaea rotundifolia	Native - Fern
Pilosella officinarum	Endemic - Forb
Pittosporum ralphii	Native - Tree
Pittosporum tenuifolium	Native - Tree
Plantago lanceolata	Endemic - Forb
Podocarpus totara	Native - Tree
Polystichum neozelandicum	Native - Fern
Polystichum vestitum	Native - Fern
Prumnopitys taxifolia	Native - Tree
Pseudopanax crassifolius	Native - Tree
Pteris tremula	Native - Fern
Pyrrosia eleagnifolia	Native - Fern
Ripogonum scandens	Native - Vine
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Rytidosperma buchananii	Native - Graminoid
Senecio bipinnatisectus	Native - Forb
Solanum nigrum	Endemic - Shrub
Sonchus oleraceus	Endemic - Forb
Stellaria media	Endemic - Forb
Neil (unfenced)	
Adiantum cunninghamii	Native - Fern
Agrostis capillaris	Endemic - Graminoid
Asplenium flaccidum	Native - Fern
Asplenium flabellifolium	Native - Fern
Asplenium hookerianum	Native - Fern
Blechnum filiforme	Native - Fern
Carex divulsa	Endemic - Graminoid
Carex inversa	Native - Graminoid
Chenopodium album	Endemic - Forb

Cirsium arvense	Endemic - Forb
Cirsium vulgare	Endemic - Forb
Conyza sumatrensis	Endemic - Forb
Coprosma areolata	Native - Tree
Coprosma rhamnoides	Native - Shrub
Coprosma robusta	Native - Tree
Crepis capillaris	Endemic - Forb
DICONDRA Spp.	Native - Forb
Dichondra repens	Native - Forb
Digitalis purpurea	Endemic - Forb
Echinopogon ovatus	Native - Graminoid
Elytrigia repens	Endemic - Graminoid
Euchiton japonicus	Native - Forb
Helminthotheca echioides	Endemic - Forb
Helichrysum lanceolatum	Native - Shrub
Holcus lanatus	Endemic - Graminoid
Hydrocotyle moschata	Native - Forb
Hypochaeris radicata	Endemic - Forb
Kunzea ericoides	Native - Tree
Lotus pedunculatus	Endemic - Forb
Luzula multiflora	Endemic - Graminoid
Melicytus ramiflorus	Native - Tree
Microsorum pustulatum	Native - Fern
Mycelis muralis	Endemic - Forb
Nertera villosa	Native - Shrub
Notogrammitis heterophylla	Native - Fern
Oxalis exilis	Native - Forb
Pellaea rotundifolia	Native - Fern
Pilosella officinarum	Endemic - Forb
Pittosporum ralphii	Native - Tree
Plantago lanceolata	Endemic - Forb
Pneumatopteris pennigera	Native - Fern
Poa trivialis	Endemic - Graminoid
Polystichum neozelandicum	Native - Fern
Polystichum vestitum	Native - Fern
Prunella vulgaris	Endemic - Forb
Pteris tremula	Native - Fern
Pyrrosia eleagnifolia	Native - Fern
Rubus cissoides	Native - Shrub
Rubus fruticosus	Endemic - Shrub
Rumex acetosella	Endemic - Forb
Rytidosperma buchananii	Native - Graminoid
Senecio bipinnatisectus	Native - Forb
Solanum nigrum	Endemic - Shrub
Sonchus oleraceus	Endemic - Forb
Stellaria media	Endemic - Forb

Taraxacum officinale	Endemic - Forb
Trifolium repens	Endemic - Forb
Trifolium subterraneum	Endemic - Forb