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Ecological Restoration of Dryland Kānuka Communities in an Irrigated Agricultural Landscape

A thesis
submitted in partial fulfilment
of the requirements for the Degree of
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by
Rebecca Dollery

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Rebecca Dollery

The Canterbury Low Plains ecological district exists as a highly modified, productive landscape containing little of its original indigenous vegetation. The residual native plant communities comprise scattered fragments of dryland shrubland with varying degrees of legal protection. One such area, the Eyrewell Forest, comprised a matrix of small dryland kānuka (*Kunzea serotina*) communities subsumed within an exotic pine plantation (6,764 ha) and pastoral land, and was described as the largest kānuka shrubland stand on the Canterbury Plains. Since 2010, the plantation has been progressively removed for conversion to precision-irrigated dairy farming. Within this landscape matrix, the current project focused on the establishment of ecological restoration areas and corridors to re-construct, protect and enhance indigenous species and habitats. The community structure and ecology of plant assemblages and associated edaphic conditions within existing remnants were investigated to assist in design, implementation and management of 18 ecological restoration blocks encompassing approximately 150 ha. A particular focus was the importance of moss carpets within natural kānuka-dominant ecosystems. I investigated the responses of the habitat and its plant species to nutrient spillover from adjacent farmland and evaluated restoration techniques suitable for the task.

Study of the remnant kānuka habitats revealed that the native vegetation is characterised by low-nutrient soils and the dry climate, supporting assemblages of plants with obvious xerophytic traits. Small-scale disturbance was found to be critical for regeneration of the single canopy species, kānuka, and an understory shrub, *Pomaderris amoena*. The dominant ground cover moss species, *Hypnum cupressiforme*, and two woody associates, *Leptecophylla juniperina* and *Carmichaelia australis*, regenerated in areas of low disturbance and high canopy cover. Germination trials had some difficulties; *P. amoena* seeds required heat treatment and naturally followed fires, but *L. juniperina* could not be germinated artificially. *H. cupressiforme* exerted significant influence over the functioning of the ecosystem with regards to hydrology, nutrient cycling and biotic interactions. Extremes of soil

temperature and moisture were mediated by the moss carpet; soil nutrients, particularly mineral nitrogen, were reduced. The moss layer also had species-specific effects on flora but generally decreased germination and establishment of native species. Glasshouse trials showed that *K. serotina*, *P. amoena* and *Kunzea robusta* grew better with soil additions of nitrogen. *C. australis* and *H. cupressiforme* responded positively to lime additions, but *L. juniperina* showed no response. In the field, fertilisation had either negligible or detrimental effects on native species. Increased soil nutrients from organic amendments elicited exotic weed encroachment. Nutrient spillover from farmland was observed at least 30m into remnant kānuka ecosystems and was positively correlated with exotic weeds but negatively related to moss cover.

Successful introduction of planted native species required protection from desiccation and competition from weeds. Tree guards successfully and cost-effectively sheltered young plants and moss. Weeds were best controlled by herbicide application and, moderately with moss and pine litter mulch. This research has contributed to the restoration of 150 ha of landscape restoration, generating practical applications for similar projects elsewhere. A major finding is the importance of integrating moss into dryland kānuka restoration projects.

Keywords: Ecological restoration, kānuka shrubland, *Kunzea serotina*, *Pomaderris amoena*, *Hypnum cupressiforme*, bryophyte, nutrient spillover, tree guard, glyphosate, direct seeding, fertiliser tablet.

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Chapter 1

Introduction

1.1 General Introduction

Global human population is increasing and estimated to reach 9 billion by 2050, placing increased demands on food production (Foresight 2011). A growing population requires not only adequate calorific energy production but also adequate nutrition, such as essential amino acids; in many situations this comes from animal proteins such as meat and dairy (Alexander et al. 2015; Consultation 2011; World Health Organization 2007). The requirement for increased meat and dairy has been reflected by the exported goods of many countries and New Zealand is the eighth largest producer of dairy products, with the top eight countries combined supplying 55% of the world's dairy needs (Fonterra 2016). In order to meet the global requirement and to remain economically viable, inputs into the dairy sector have increased to gain greater outputs, exemplified by intensive farming (Moller et al. 2008). As such, the production of animal products was responsible for 65% of land use change worldwide from 1961-2011 (Alexander et al. 2015).

In New Zealand, productive land covers 42% of the land mass; whilst this has not increased significantly since 1996, the intensity of farming operations has (Ministry for the Environment & Statistics New Zealand 2015). Agriculture is an important part of GDP, accounting for \$36.7 billion in exports in 2016 (Ministry for Primary Industries 2016), with dairy being the largest export commodity, worth approximately \$12 billion (MacPherson 2015). The South Island, particularly the Canterbury region, has seen a large increase in intensive farms in the last 10 years with an 82% increase in number of farms and vastly increased cattle stocking when compared to farms elsewhere in the country; mean herd sizes of c. 912 compares to 413 nationwide (Burns 2015; MacLeod and Moller 2006).

Intensive farming, and particularly dairying, is recognised to be a substantial contributor to several environmental issues (Foresight 2011; Steinfeld et al. 2006). Within Canterbury, land conversion to agriculture has led to a substantial amount of biodiversity and native habitat loss, with native land cover on the Canterbury Plains currently amounting to less than 1% (Environment Canterbury 2008b; Harding et al. 2009; Walker et al. 2007). Most remaining remnants in the lowlands are small and fragmented, and threats to their longevity are further perpetuated by the invasion of exotic plants (Meurk and Swaffield 2000). In fact, naturalised species are reported to be more numerous in New Zealand than the depauperate indigenous flora (Sullivan et al. 2004). Whilst the native remnants are small and fragmented (Meurk et al. 1995) they contain a large proportion of New Zealand's indigenous and threatened species and are under-represented within the protected areas

network (Environment Canterbury 2008b; Head and Given 2001; Walker et al. 2006). Furthermore, vegetation change has been linked to altered states of ecosystem processes such as nutrient and hydrological cycles, affecting ecosystem services (Ehrenfeld 2003; Ehrenfeld et al. 2005; Jobbágy and Jackson 2004).

Along with the loss of biodiversity and associated ecosystem services, agriculture was responsible for 49% of New Zealand's greenhouse gas emissions in 2014 and is the main contributor to atmospheric nitrous oxide emissions (Brown and Petrie 2006). The waterways surrounding farming operations have been subjected to degradation through deposition of fine sediment and nitrate leaching and nutrient enrichment. (Abell et al. 2011; Allan 2004; Ballantine and Davies-Colley 2014; Jones et al. 2001; Monaghan et al. 2009). Water abstraction for irrigation also poses threats to freshwater systems and, due to low rainfall, Canterbury dairy farms rely more heavily on groundwater and freshwater sources of irrigation than elsewhere in the New Zealand (Zonderland-Thomassen and Ledgard 2012) leading to concerns over water scarcity (Norton and Reid 2013; Postel 2000). Soil is also reported to have been physically and chemically impacted through compaction from cattle trampling and heavy machinery use (Nawaz et al. 2013), and the over-use of phosphorus and nitrogen fertilisers (Ministry for the Environment 2007; Ministry for the Environment & Statistics New Zealand 2015). These effects are well documented. There are also environmental impacts of feed supplements, particularly palm oil kernel extract (PKE) that is used extensively in Canterbury (Foote et al. 2015).

Despite these environmental impacts of agriculture, New Zealand is marketed as being a distinctly "clean, green" environment (Ateljevic and Doorne 2002; Bell 1996). Tourism is built upon this branding, with the New Zealand Tourism website containing a "100% Pure New Zealand" slogan amidst scenic images of clean water, lush vegetation and recreational tourism (New Zealand Tourism 2016). Earnings from the tourism revenue stream were \$11.7 billion in 2015 (MacPherson 2015) and in 2016 tourism exceeded dairy exports by \$2.14 billion (Statistics New Zealand 2016). Tourism also provides 13.2% of the country's jobs (Statistics New Zealand 2016). There are rising concerns that increasing agricultural intensification will negatively impact the tourism and recreational sectors, whilst simultaneously reducing export of primary commodities (Foote et al. 2015). The Ministry for the Environment (2001) reported that there would be a 54% reduction in consumer purchasing of dairy products should the environment be perceived to be degraded.

In light of this knowledge, it is imperative that agronomists and ecologists strive to sustain and develop agriculture and the economy whilst preserving the health of ecosystems and associated ecosystem services, including clean water, clean air and healthy soil (Beddington 2011; Robertson and Swinton 2005). Sustainable intensification is defined as "the pursuit of the dual goals of higher yields with fewer consequences for the environment" (Steinfeld et al. 2006). A concept of sustainable

intensification can be based upon two different perspectives: (i) land sharing; an integrated capacity of biodiversity embedded within productive land or, (ii) land sparing, purposely separated land use areas (Garnett et al. 2013). It has been argued that both approaches have merits and should be evaluated on a site-specific basis (Barral et al. 2015; Merckx and Pereira 2015; Phalan et al. 2011). A main challenge is to ensure that the agricultural land remains productive and the native habitats are resilient to potentially harmful biotic and abiotic influences such as nutrient enrichment, encroachment of invasive species, reduced biotic interactions (e.g. microbes for nutrient mobilisation and insects for pollination) and lack of connectivity (Baudron and Giller 2014).

The present work explores the potential to successfully and sustainably embed indigenous biodiversity within the Canterbury agricultural landscape, as an integrated approach to landscape management, whilst evaluating the obstacles and deficiencies in current knowledge. This research is based on a case-study of a large scale dairy conversion scheme at Te Whenua Hou (Eyrewell Forest) on the Canterbury Plains. Largely during the course of the present work, the scheme has converted 6,764 ha of a plantation pine forest into 20 pivot-irrigated dairy farms (Ngai Tahu Farming 2016). The conversion included a number of initiatives to mediate some of the negative effects of dairy farming, including the establishment of 150 ha of indigenous shrubland to compensate for biodiversity losses and to provide biodiversity gains by creating a network of native vegetation through the proposed land conversion area. In addition, a further area of approximately 150 ha of native planting is associated with farm and paddock boundaries, under pivots and around farm buildings and infrastructure.

Ecological restoration is underpinned with theory and knowledge pertaining to habitat structure and community composition, but also to an understanding of the environmental conditions, ecological processes and interactions of abiotic and biotic components of the ecosystem (Bradshaw 1993; Heneghan et al. 2008; Hobbs and Norton 1996; Lindenmayer et al. 2002). In the Eyrewell Forest region there have been a number of studies which describe the remnant vegetation community assemblages, but only a few have integrated the edaphic conditions or have considered the ecology of these ecosystems (Bretherton et al. 2002; Ecroyd and Brockerhoff 2005; Meurk et al. 1995; Molloy and Ives 1972). Whilst these studies acknowledge the presence of prominent non-vascular plant communities, information relating to the value and role of these communities is limited. Non-vascular plants are known to be an important ecosystem component in many dry habitats or those with extreme temperature and in other parts of the world and have previously been incorporated into restoration projects (Belnap 2006; Maestre et al. 2011). Examples of restoration projects including non-vascular plants in New Zealand are limited to peatland projects or mining sites where mosses and other cryptogams have been used for stabilisation (Buxton et al. 2005).

Much of the native flora in New Zealand has evolved on and is adapted to nutrient poor substrates (Wardle 1991). Soils which are naturally fertile are limited and many have been weathered by high rainfall, leading to acidic soils (McLaren and Cameron 1996; Wardle 1991). Earlier studies have found that increased soil nutrition does not necessarily lead to increased growth rates in native vegetation (Caroline 1999; Franklin 2014; Ogle 1996; Peltzer et al. 2016) and can lead to biodiversity losses within remnants where nutrient enrichment occurs (Bowie et al. 2016; Ecroyd and Brockerhoff 2005). However, the evidence is inconclusive; other studies in controlled environments have pointed to increased growth of some species (Carswell et al. 2005; Esperschuetz et al. 2017a). Clearly, an understanding of the responses of native flora to increased nutrients is important when restoring habitats on soils that have been modified from their natural state and which are planted in a nutrient-enriched environment. Little is known about the nutritional requirements of native seedlings, even though it is well known that lack of such knowledge often leads to failure in restoration projects elsewhere (Bradshaw and Chadwick 1980).

Restoration techniques are varied, and investigation of the suitability of existing methodologies used in New Zealand is included in the present work. Due to the limited native vegetation on the Canterbury Plains, the introduction of plant propagules is important. A number of methods exist, including relying on the natural regeneration of existing plants, integrating viable seed into the environment (Dodd and Power 2007; Lamb and Gilmour 2003; Stanturf et al. 2014) or planting of nursery grown specimens (Anton et al. 2015; Clarkson and Kirby 2016). The method chosen to introduce propagules needs to be tailored to the site and species used and each method has its own compromise with regard to cost and vigour of grown specimens (Dodd and Power 2007; Douglas et al. 2007; Engel and Parrotta 2001; Madsen and Löf 2005), availability of seed within the environment (Meurk et al. 1995; Overdyck and Clarkson 2012; Partridge 1989), and current knowledge of plant ecology and survivorship rate (Ceccon et al. 2015; Close and Davidson 2003; Douglas et al. 2007; Fountain and Outred 1991; Rowarth et al. 2007). The most common approach in New Zealand restoration projects is the planting out of robust, 1-2 year old specimens grown in nurseries. However, this can be costly, requiring each plant to be protected from herbivory, abiotic stress, excessive shading and competition from exotic species, depending on the site and species involved (Gorbunov 2008). Methods of protection vary in cost and effectiveness, including the fencing of entire areas or the use of tree guards around each individual plant (Davis and Meurk 2001; Lai and Wong 2005), and suppressing weeds by herbicides, weed mats or organic mulches (Ladd et al. 2010; Stanton-Clements et al. 2013). It has been argued that restoration techniques should be determined by the ecology of the plant species used and the site specific conditions, not chosen due to their common usage or success elsewhere (Hilderbrand et al. 2005). Further study on restoration methods in all habitats is required to

develop best practice techniques. This is particularly the cases where the use of novel techniques, such as establishing non-vascular ground covers, may be important.

1.2 Research Aims and Objectives

The aim of the work presented in this thesis is to understand and assist the design of successful dryland kānuka shrubland restoration within an intensively farmed landscape matrix on the Canterbury Plains. The thesis demonstrates the process undertaken to establish a restoration program for Te Whenua Hou in Eyrewell Forest. It documents the target vegetation (the dryland kānuka shrubland), explores the important abiotic and biotic components of the community, both vascular and non-vascular plants, and evaluates the potential drivers of community change which are degrading the habitat elsewhere. This underpinning knowledge then informs restoration choices and leads to an investigation of the practical implementation of suitable restoration methodologies.

The project had the following objectives:

- Objective 1: To identify the composition, ecology and associated edaphic environment of indigenous dryland kānuka communities on the Canterbury Plains (Chapter 3);
- Objective 2: To investigate the significance and ecological role of non-vascular plants within dryland kānuka vegetation with regard to chemistry, hydrology and biotic interactions (Chapters 4 and 5);
- Objective 3: To investigate the main environmental determinants of community composition and the effect of nutrient spillover on plants within the kānuka shrubland (Chapter 6);
- Objective 4: To evaluate appropriate methodologies and techniques to restore dryland kānuka habitats into these agricultural landscape matrices (Chapter 7).

1.3 Thesis Structure

The Chapters within the thesis broadly follow the objectives, as set out above. Chapter 2 provides a detailed background and literature review related to the main focus of the project, including descriptions of New Zealand's bryophyte flora and an evaluation of its likely importance in the context of restoration practice. Chapter 3 introduces the site, including remnant dryland shrubland areas of the Eyrewell Forest, and investigates the ecology of the communities and germination requirements of individual species. Chapters 4-7 report experimental trials. These include:

- Chapter 4: Bryophytes and Hydrology – glasshouse and field trials investigating the influence of dominant mosses on the hydrology of the dryland kānuka shrubland remnants;
- Chapter 5: Bryophytes, Chemistry and Biotic Interactions – the influence of dominant mosses on the substrate chemistry and the role they play in assisting or inhibiting germination of plant species within the dryland kānuka shrubland remnants;
- Chapter 6: Nutrient Influences upon native and exotic dryland shrubland communities – glasshouse and field trials exploring the effects of enhanced nutrition upon the health, growth and competitive ability of native and exotic vascular and non-vascular plants within dryland kānuka shrubland; and
- Chapter 7: Restoration Practices – field experiments examining plant introduction and protection practices within the restoration industry.

Chapter 8: provides an overall discussion and conclusions, conclusions and recommendations for further research. Appendices are provided documenting scientific and common names for plants frequently referred to plus abbreviated terms (Appendix A) and supporting information for Chapter 3 regarding soil test data and species lists (Appendix B). Appendix C is the executive summary page from a report delivered to the sponsors of the present work, providing a synopsis of the research and detailing recommendations for the restoration of indigenous habitats in Eyrewell Forest.

Chapter 2

Background

2.1 Introduction

New Zealand was once dominated by forest cover below the tree line with a high incidence of floral and faunal endemism (Daugherty et al. 1993; Wardle 1991). Since the relatively recent human arrival, the cover of forest has decreased by approximately three quarters, resulting in highly fragmented remnant habitats (Ewers et al. 2006; Guild and Dudfield 2009). This is exemplified on the Canterbury Plains of the central eastern South Island which has experienced a 91% reduction in forest coverage from the pre-human indigenous habitat (Ewers et al. 2006).

The Canterbury Low Plains Ecological District (as defined by the Department of Conservation (McEwan 1987)) is a highly modified environment in which indigenous vegetation has been severely spatially reduced mainly due to conversion of land for production and settlement (Environment Canterbury 2008a). The amount of native land cover remaining in some areas is less than 0.1% being categorised as “acutely threatened” (Walker et al. 2006; Winterbourn et al. 2008) and the existing remnants are small and fragmented, often containing non-regenerating populations (Meurk 2008; Meurk et al. 1995). These habitats contain a large proportion of indigenous and threatened species that are underrepresented within the New Zealand protected areas network (Head and Given 2001; Holdaway et al. 2012; Walker et al. 2006), which has led to widespread ecological restoration projects across the country in recent years (Norton 2009; Peters et al. 2015). Most restoration projects concentrate on removing invasive plants and restoring appropriate abiotic conditions before introducing a specific, desired vascular plant assemblage of trees, woody shrubs and other vascular plants (Davis and Meurk 2001). This approach restores some structure and composition of the desired ecosystem but may underestimate the importance of restoring ecological processes and the functional roles of non-vascular plants (Stanturf et al. 2014). New Zealand has high diversity and abundance of lower plants including Bryophytes (mosses and liverworts), which contribute substantially to forest ecosystems in terms of structure, biomass, diversity and potentially function (Fife et al. 2014; Geffert et al. 2013; Hofstede et al. 2001), although the literature relating to their role within New Zealand temperate forest is limited.

This literature review provides critical evaluation of the knowledge base surrounding the broader role of mosses in temperate forests and also reviews methodologies commonly used within ecological restoration practice. I explore the feasibility and potential benefits of moss introduction to restored vegetation assemblages.

2.2 Restoration Ecology: Forests and Plants

Restoration ecology has been defined as, “the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed” (Clewett et al. 2004). It is a complex field, rooted in ecological theories such as those of succession, community assembly rules and alternative states, driven by various ecological forces such as the abiotic environment, dispersal limitation, priority effects and convergent/divergent communities (Cale et al. 2013; Suding et al. 2004; Young et al. 2001). These theories underpin many restoration ecology models striving to understand the workings of the ecosystem, their decline into deterioration, and the methodologies to successfully restore environments on varying spatial levels (Grant 2006; Hobbs and Harris 2001; Hobbs and Norton 1996; van Andel and Aronson 2012). The Society for Ecological Restoration (SER) defines four guiding principles for the practice of restoration (Suding et al. 2015), that restoration will;

- I. “Increase ecological integrity” by introducing or enhancing species composition, function and ecosystem processes;
- II. Create ecosystems which are resilient and self-sustaining;
- III. Is led by historical events and future predictions; and
- IV. Incorporates cultural and social values.

Tongway and Ludwig (2011) described a five-point approach to undertaking restoration projects and emphasised the importance of planning and monitoring and re-analysing to attain success:

1. **Set Goals:** A thorough understanding of precise and measurable goals is recommended as background to the restoration scheme. This will dictate the course of the project;
2. **Define the Problem:** Critical analysis of the factors causing degradation and the ecosystem processes which have been damaged, altered or disturbed;
3. **Design a Solution:** Using information gleaned from step two investigate suitable solutions, mitigation strategies or recreation works;
4. **Apply Technologies:** Use best known technologies in an adaptive manner, amending and altering methodologies where results are not forthcoming or expected;
5. **Monitor:** The process is iterative, monitoring successes and failures and adapting, re-analysing and applying new technologies until restoration goals are achieved.

2.2.1 Restoring the Abiotic Environment

The main hurdles to be overcome when restoring an area are the effects of historical land use on the substrate, dispersal constraints of biota into the restoration area and its spatial character and degree of fragmentation (Honnay et al. 2002). A popular method of approaching the first of these issues has been to amend the abiotic environment (Hilderbrand et al. 2005). Primarily undertaken where the physical and chemical environment has been severely altered by activities such as mining, deforestation or river engineering, techniques to reinstate the original condition are employed. For instance, mitigation may include nutrient removal or addition through manipulation of topsoil and use of fertilisers, hydrology modification or amendment of shade and canopy cover (Bradshaw 1997; Gómez-Aparicio et al. 2004; Hart et al. 2002; Hölzel and Otte 2003; Mabry et al. 2010). This approach without further intervention assumes that the correct abiotic environment will result in the desired ecosystem establishing and was commonly practiced in grassland environments in the 1970s and 1980's (Kiehl 2010). Whilst the importance of the abiotic environment cannot be undervalued, creating conditions similar to those found in a desired reference site does not guarantee its establishment (Bradshaw 1997; Hilderbrand et al. 2005). Further intervention may be required to reinstate important biotic-abiotic interactions or overcome biotic constraints such as a limited species pool preventing natural dispersal or an increase in invasive, non-desirable species within the area (Suding et al. 2004).

2.2.2 Restoring the Biotic Environment

Introduction of biota within traditional forest restoration has tended to focus the planting of larger vascular species, such as trees and shrubs (Atkinson 1994; Greater Regional Wellington Council 2014). This affords instant community assemblage and may accelerate the time to canopy closure (Bergin and Gea 2007; Porteous et al. 1993) although it is noted that often “planting has its pitfalls”; naturally regenerating individuals are often healthier and obviously are more cost effective (Davis and Meurk 2001). When introducing plant species various questions need to be addressed: species selection; reintroduction methods; subsequent management practices. These activities should all be firmly related to the objectives and site-specific conditions of the scheme (Box 1996; Burton 2014).

In restoration projects aiming to achieve a native community the species selected often reflect those found in remnant sites or known from historical data (Davis and Meurk 2001). Sometimes a plant which is not regionally native to an area can be employed to achieve rapid canopy cover and assist largely shade-tolerant native plants to regenerate (Meurk et al. 2016). However, more commonly, indigenous species naturally found in the local environment are selected for introduction (Davis and Meurk 2001; Greater Regional Wellington Council 2014; Sullivan et al. 2009; Waitakere City Council 2005).

Plants can be introduced to an area through a variety of techniques such as allowing natural dispersal and regeneration, growing desired plants in a nursery and planting out once established, or by direct seeding (Stanturf et al. 2014). Direct seeding, the process of broadcasting seed over an area can introduce plants into a restoration area successfully and cost effectively, cutting budget costs by up to 50% (Dodd and Power 2007; Douglas et al. 2007; Engel and Parrotta 2001; Madsen and Löf 2005). However, it is reported that seeds of New Zealand native plants are not long-lived and information regarding germination of many species is lacking, which may account for results from this technique to have high failure rates (Burrows et al. 1994; Rowarth et al. 2007). The method can also produce scattered areas of low or high germination leading it to be viewed by some as a complementary technique to planting (Ceccon et al. 2015; Close and Davidson 2003). Godefroid et al. (2011) analysed a 249 reintroduction projects worldwide using a range of techniques and found that planting out seedlings with bare roots was most effective for survival. However, bare rooted planting is not advisable for some indigenous species such as kānuka (*Kunzea* spp.) due to the fine root system below 10 cm depth (Franklin 2014). It has been noted that planting out overcomes the issues of vulnerability in young seedling survivorship (van Andel and Aronson 2012). Other mechanisms for successful restoration were planting out more plants from a stable population and maintaining adequate management of restoration sites. However, reintroduction can be costly and therefore adequate avoidance strategies from failure should be applied (Gorbunov 2008).

One of the biggest threats to restoration introductions is failure from weed competition (England et al. 2013; Sullivan et al. 2009). Manual weeding can be employed to reduce weed populations, but this is labour intensive and only suitable when infestation levels are low (Davis and Meurk 2001). Methods to target weeds include limiting required resources, such as light through artificial cover (Benayas et al. 2005; Standish 2002) or creating dense plantings or cover crops (Macdonald et al. 2015; Sullivan et al. 2009). Herbicide control of weeds can also yield good results (Harrington and Gregory 2009; Harrington et al. 2015), but can lead to damage of target plants and does not always suppress the weeds effectively (Porteous et al. 1993; Standish 2002). A long-term study found that spraying herbicide actually decreased the population size of many native plants in a grassland whilst the weed species recovered and recolonised (Rinella et al. 2009). Often a combination of weed suppression techniques based on ecological knowledge of the weed species and an understanding of the abiotic conditions of the site is the best approach (Burrows and Watson 2000; Sweeney et al. 2002).



Plate 2.1. Two types of tree guard, a polythene sheath (left) and a corrugated plastic guard (right) held in place with four and one canes respectively.

A commonly used physical method of protection against both weed competition and grazing is the use of tree guards which compose a protective sheath, usually made from plastic that surrounds the plant and is held in place with canes or similar (Plate 2.1). The guards can protect the seedling from herbivory, exposure to winds and dry conditions and competing weeds where weed mats are used in conjunction (Ladd et al. 2010; Lai and Wong 2005). Protection against grazing is important for growth of some palatable species in New Zealand (Wardle et al. 2001) as they evolved in the absence of ungulate herbivores, the scale and type of browsing formerly exerted by resident large extinct flightless birds (mainly moa) would have differed (McGlone and Clarkson 1993).

2.2.3 Is Vascular Plant Reintroduction Enough?

The reinstatement of a large vascular plant assemblage as partial recreation of ecosystem composition and structure offers the assumption that it will facilitate colonisation of associated species, such as herbaceous plants, non-vascular plants and faunal communities, thus eventually generating a functioning ecosystem similar to that found at the reference site. Hilderbrand et al. (2005) described this as the myth of fast forwarding, that it is possible to overcome successional constraints and stabilise environments in order to recreate the reference site in a shorter time frame. In order to produce this effect, processes and functions need to be restored which can take a considerable amount of effort and does not always result in the desired habitat development. Reay and Norton (1999) found some evidence of this approach working when sites had been planted solely with a species of native tree during their study of a number of restoration sites on the Banks Peninsula, South Island New Zealand. The sites were beginning to display important processes, such as dispersal and regeneration of flora

and fauna, within the stands after 30-35 years. However, the restoration and old forest sites were still markedly different in terms of floristic composition.

Numerous studies have revealed that compositional planting alone does not automatically lead to the establishment of the desired community and that further intervention is needed (Suding et al. 2004; Wilkins et al. 2003). There has been a greater call from restoration ecologists to concentrate on restoring the processes and functional traits of an ecosystem rather than the aesthetic and compositional attributes (Anand et al. 2005; Choi 2007; Davis and Slobodkin 2004). Hallett et al. (2013) studied 203 restoration projects listed on the Global Restoration Network (maintained by the Society for Ecological Restoration) and found that ecosystem stability (resilience and self-sustaining habitat goals) were barely cited and social values were represented by only a quarter. It is possible that ecosystems that are diverse in terms of structure and function are more likely to be resilient and adaptable (Suding et al. 2015). This is especially important in light of ecosystem pressures and constraints such as changing environments and possible effects of climate change, species invasions and the dynamism and complexity of ecosystems (Harris et al. 2006; Hilderbrand et al. 2005; Hobbs and Norton 1996).

In recent years, non-vascular plants and their importance in the functioning of communities and ecosystems has received growing attention from conservation organisations (Hallingbäck and Hodgetts 2000; Vanderpoorten and Goffinet 2009). Their value with regards to both community structure and ecosystem function have been recognised, yet these plants often are still overlooked in restoration projects, excepting projects in cool or dry environments or wetland and peatland habitats (Bowker 2007; Bowker et al. 2005; Sottocornola et al. 2007; Vanderpoorten and Goffinet 2009). Although much research has concentrated on the importance of Bryophyta in other environments, there is evidence to suggest mosses have key roles in ecosystem functioning in temperate forests, discussed below.

2.3 Bryophytes

2.3.1 Life History

The bryophyte grouping includes three divisions of plants, namely mosses (Bryophyta), liverworts (Marchantiophyta) and hornworts (Anthocerotophyta). Mosses occur on every continent in both terrestrial and aquatic biomes in a range of habitats, environmental conditions and substrate (Lindo and Gonzalez 2010; Vanderpoorten and Goffinet 2009). They are highly adaptive although species richness is limited in the driest environments indicating their requirements for precipitation or airborne water vapour (Geffert et al. 2013). There are currently over 13,000 species of Bryophyta identified worldwide (Goffinet et al. 2009). Endemism within bryophytes is low in New Zealand when

compared to vascular plants: approximately 20% compared to over 80% vascular plants (McGlone et al. 2001). This is mainly due to diverse reproduction strategies, the ability to disperse long distances and the viability and longevity of spores (Frahm 2008).

Mosses can be divided into acrocarp, cladocarp or pleurocarp species which relates to the position of the sporophyte on the gametophyte and also the growth morphology (La Farge-England 1996; Mägdefrau 1982). Individual plants can grow together which extend the growth form category into life-forms, the colonies created by tightly packed groups of individual plants (see Table 2.1). However, due to the plasticity of moss species to environmental conditions life form can prove difficult to categorise in the field (Bates 1998).

Table 2.1 Life-form of mosses as described by Mägdefrau (1982) and adapted by (Glime 2007).

Life-Form	Description	Example
Annuals	Pioneer species lasting 1 year.	<i>Ephemerum</i> .
Short Turfs	Collection of parallel upright shoots of <2 cm in height. Directional growth is vertical.	<i>Ceratodon</i> .
Tall Turfs	As above but >2 cm. Common in temperate forest floors.	<i>Polytrichum</i> , <i>Sphagnum</i> .
Cushions	Erect shoots tightly packed forming dome shaped clusters.	<i>Grimmia</i> , <i>Leucobryum</i> .
Mats	Interwoven stems extending horizontally.	<i>Racomitrium</i>
Wefts	As mat but ascending shoots. Common on forest floors with loose, new growth each year.	<i>Thuidium</i> , <i>Hypnum</i>
Pendants	Epiphytes; Drooping, hanging down moss.	<i>Dendropogonella rufescens</i>
Tails	Trees and rocks, creeping shoots stand away from substrate.	<i>Leucodon</i>
Fans	Vertical creeping branches on one plane.	<i>Thamnobryum</i>
Dendroids	Stems with leaves at the end of branches to give a tree-like appearance.	<i>Hypnodendron</i>
Streamer	Long, floating moss stems in aquatic habitats.	<i>Fontinalis</i>

Mosses have a stem and leaf structure with the appearance of higher plants but also have “hidden sex organs” being described as “cryptogams” in Linnaean taxonomy (Smith 1938). Critically, the leaves of mosses are sessile and, unlike the true leaves of vascular plants, the tissue lacks a cuticle and stomata in the majority of species (Goffinet et al. 2009). The plants also lack the internal structures of phloem and xylem attributed to higher plants for transporting water and nutrients (Glime 2015). This leads to poikilohydry, a reduced ability to store and regulate water loss and the passive equilibration of internal moisture thresholds to those in the surrounding environment (Proctor 2009; Tuba et al. 1996). Mosses have adapted both physiologically and morphologically to tolerate desiccation and suspend metabolism without extensive damage to tissues, with the ability to recommence photosynthesis upon rewetting (Proctor et al. 2007; Vitt et al. 2014). Water and nutrients are conducted in external capillary spaces and absorbed through stems and leaves either from surface

water or through dry deposition (either rainfall, mist droplets or airborne dust) (Bates 1992; Tuba et al. 1996). However, there is evidence to suggest that mosses can also acquire some nutrients from the substrate on which they grow (Brūmelis et al. 2000; Van Tooren et al. 1990).

The majority of pleurocarpous moss species are ectohydric, gaining water from external sources, but there are a number of endohydric mosses, many acrocarpous in growth form, which have a waxy leaf surface and mainly internal water conduction, as seen in Polytrichaceae (Proctor 2000; Richardson 1981). Endohydric mosses tend to be amongst the largest in stature (Vitt et al. 2014). Whilst these mosses are capable of internal water conduction they are not restricted to it and may resort to endohydric methods only in certain situations (Glime 2015; Zajączkowska et al. 2017).

Unlike vascular plants, mosses are spore producing plants. There is an alternation of generations in mosses from haploid (gametophyte) to diploid (sporophyte) but the haploid stage is the more dominant and long lived (Glime 2015). The haploid stage encompasses the main green plant whilst the diploid stage refers to the sporophyte. The sexual life cycle involves the sporophyte producing spores by meiosis which grow into haploid gametophytes. These then produce eggs or sperm which unite to create a zygote (mitosis) which grows into a sporophyte (Figure 2.1). The spermatozooids must swim from the antheridia to the archegonia to create a zygote signifying the requirement of water (Goffinet et al. 2009). Mosses can be either monoecious or dioecious. Sperm is transported via capillary and gravitational forces of water (Bisang et al. 2004) and some species have “splash cups” which can help propel the gametes (Andersson 2002). There is also evidence that microarthropods can assist in the transport of sperm (Bisang and Hedenäs 2015; Cronberg et al. 2006; Rudolphi 2009).

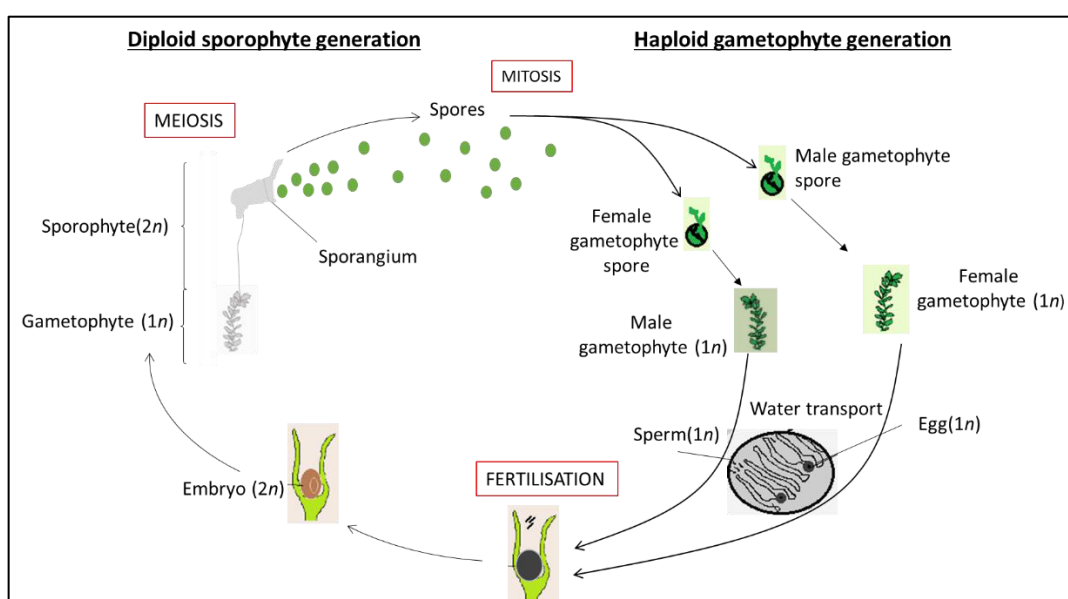


Figure 2.1 Alternation of generation: The sexual life cycle of a moss plant (adapted from Wikipedia Commons, no copyright).

Sporophytes are rare in some species (Longton 1992a; Rydgren et al. 2010) and consist of a seta (stalk) and sporangia (capsule) containing the spores. Sporophytes remain attached to and nutritionally dependent upon the gametophyte (Nath and Bansal 2015). The spores of mosses are usually less than 100µm and released in millions (Frahm 2009). They can be dispersed via a number of vectors with wind being the primary route, but also by water and animals (Bates 2009; Porley and Hodgetts 2005). The spores of plants can remain viable for long periods of time and lead to diaspore banks which would allow the plants to recolonize following disturbance (During 2001).

Mosses also have three main mechanisms for asexual reproduction: branching leading to the establishment of new plants, the development of specialised propagation bodies called gemmae or regeneration from plant parts (Watson 1971). This clonal activity can lead to some species having populations where one sex is dominant, such as *Syntrichia caninervis* (Stark et al. 2000). Vegetative regeneration is highly effective and can act as a means of dispersal; bryophyte plant parts are adhered to animals and transported to new habitats (Heinken et al. 2001; Osorio-Zuñiga et al. 2014; Pauliuk et al. 2011). There is evidence to suggest that some mosses, such as *Sphagnum*, can regenerate from buried plant fractions (Clymo and Duckett 1986), whilst others can grow after being ingested by bats (Parsons et al. 2007).

2.3.2 Bryophyta in New Zealand Forest and Shrubland

In New Zealand there are over 500 species of bryophyte from over 200 genera (Fife 1995). In forests and shrublands the composition of bryophytes is unusually dominated by a limited number of species which can attain over 90% cover, especially in wet habitats (Pfeiffer 2003). There are a number of parameters that can influence the distribution of mosses; some of the most important factors are dispersal mechanisms, climate and vegetation; the latter two affecting the moisture gradient of an area (Frahm 2008; Michel et al. 2011b).

Several studies undertaken on bryophyte communities in New Zealand have found diverse and species rich communities where there is sufficient light and moisture, such as those conditions found in streams and podocarp and beech (*Fuscospora spp.*) forests (Beever and Brownsey 1993; Hofstede et al. 2001; Suren 1996). Kānuka (*Kunzea spp.*) and mānuka (*Leptospermum scoparium*) forest and shrubland appear to be less species rich than taller forests but relatively consistent throughout New Zealand in regard to species composition and structure (Table 2.2). For example, *Hypnum cupressiforme* was found in each site surveyed and formed continuous carpets in association with other species (Beever et al. 1992; Beever and Brownsey 1993; Beever and Brownsey 1990; Beever 1986; Molloy and Ives 1972). Macmillan (1976) described the carpets of *H. cupressiforme* within the Bankside Scientific Reserve on the Canterbury Plains as 12 cm deep around the kānuka stems whilst it provided constant ground cover under native *Festuca* and *Danthonia* grassland. Similarly, Molloy and

Ives (1972) noted the almost continuous cover of *H. cupressiforme* (Plate. 2.2). Despite the abundance of bryophytes within these habitats there is surprisingly little known about the ecological functions within the temperate forest or shrubland.



Plate 2.2. *Hypnum cupressiforme* forming a continuous carpet in kānuka dominated vegetation in the Eyrewell Forest, Canterbury. Growing within the moss is *Leptinella pusilla* which is a creeping tufted perennial native forb of coastal to subalpine New Zealand.

Table 2.2 Summary of the most abundant moss species found within New Zealand mānuka and kānuka habitats.

Area	Reference	No. of species	Most abundant terricolous species
Ship Cove, Marlborough	(Beever and Brownsey 1993)	37	<i>Thuidium furfurosum</i> , <i>Hypnum cupressiforme</i> , <i>Campylopus clavatus</i> , <i>Ptychomnion acicular</i>
Motuhora (Whale Island), Bay of Plenty	(Beever and Brownsey 1990)	17	<i>Thuidium furfurosum</i> , <i>Hypnum cupressiforme</i> , <i>Bryum billardierei</i> , <i>B. campylothecium</i>
Poor Knights Island, Auckland	(Beever 1986)	3	<i>Hypnum cupressiforme</i> , <i>Dicranium billardieri</i>
Canterbury Plains – Bankside Reserve	(Macmillan 1976)	13	<i>Hypnum cupressiforme</i> , <i>Polytrichum juniperinum</i> , <i>Triquetrella papillata</i> , <i>Breutelia affinis</i> , <i>Hedwigia ciliate</i> , <i>Racomitrium langignoaum</i> , <i>Tortella calycina</i> , <i>Campylopus clavatus</i>
Canterbury Plains – Eyrewell Reserve	(Molloy and Ives 1972)	9	<i>Hypnum cupressiforme</i>

2.3.3 The Ecological Role of Bryophyta in Forests

The role of mosses in conjunction with other non-vascular plants (bacteria, fungi, algae and cyanobacteria), commonly referred to as biological soil crusts (Belnap 2006), has been studied extensively in dry environments. Their presence is thought to be vital to allow the ecosystem to remain or regenerate (Bowker 2007). This is especially true in low productivity environments such as tundra or desert where bryophytes can provide substrate stability (Eldridge and Greene 1994), buffer temperature changes (Coulson et al. 1993; Gornall et al. 2011), influence hydrology and nutrient cycles (Belnap 2006; Belnap and Gardner 1993; Evans and Ehleringer 1993; Gornall et al. 2007) and affect the structural composition of habitats, which in turn can impact vascular plant establishment (Gornall et al. 2011; Zamfir 2000). Within boreal forest environments of North America, Scandinavia and Canada the ecosystem function of ground layer and mosses has been extensively studied with regards to their importance in carbon and nitrogen cycling, and processes such as decomposition and their influence on seedling and forest regeneration (Harden et al. 1997; Nilsson and Wardle 2005; Turetsky et al. 2012). Research regarding their functional significance in temperate forests is not as extensive, focusing mainly on the community composition and net production rates (Turetsky 2003). However, the importance of bryophytes in other ecosystems suggests that mosses may play a similar role within temperate forests. Below are some aspects of ecosystems that mosses are known to influence either within forests (temperate or boreal) or other environments.

I. Effects of moss on soil physical properties

In arid environments, where soil surfaces are generally exposed and subjected to high physical and chemical weathering, mosses and lichens can offer stability and a reduction in erosion magnitude (Belnap and Gardner 1993; Calvo et al. 1997). Alam et al. (2012) found that destruction of the bryophyte layer in the shrubland and grassland areas within the hills of Nilgiris, India resulted in reduced binding of the soil mineral horizon and increased landslide incidence. Although these references are not from temperate forest ecosystems, they are relevant to the Canterbury Plains due to the high incidence of soil erosion. As discussed earlier, the Canterbury Plains lies within the dryland zone of New Zealand and comprises a flat landform which has undergone extensive vegetation clearance in the recent past (Rogers et al. 2005). This has exposed the well-drained soils, resulting in high rates of soil erosion, particularly Surface erosion (sheet, rill and wind) (Basher 2013).

Woodland soils tend to be protected from erosion to a certain extent by the vascular plant canopy which prevent rain splash, runoff and sediment loss (Ludwig et al. 2005a). However, in temperate woodlands of south-western Australia, non-vascular organisms are also important for prevention of erosion especially where disturbances such as grazing are an issue (Yates et al. 2000). The presence of the moss layer may also buffer the soil from extreme temperature changes as seen

extensively in arid and boreal environments (Soudzilovskaia et al. 2011; Suzuki et al. 2007; Turetsky et al. 2012).

II. Effects of moss on soil moisture and ecosystem hydrology

Mosses are poikilohydric plants which have no method to avoid dehydration and equilibrate with atmospheric humidity. The water status is a function of many factors including physiology of the plant, local hydrology and availability of water (Bond-Lamberty et al. 2011; Green and Lange 1995). The influence of mosses on the hydrology of an ecosystem is not fully understood (Bond-Lamberty et al. 2011) but it is known that they can increase gravimetric soil moisture by retaining water (Turetsky 2003). It is estimated that the maximum water holding capacity of some mosses within temperate New Zealand forest and grassland habitats is approximately 1400% of their dry weight (DeLucia et al. 2003; Michel et al. 2013). Moisture stored in mosses comes from a variety of sources with some being captured from rainwater. Studies elsewhere have cited the amount of throughfall retention from a boreal canopy to be approximately 23% (Price et al. 1997; Suzuki et al. 2007), whilst a study in New Zealand found mosses added 4.4mm of water holding capacity to a conifer-broadleaf forest ecosystem (DeLucia et al. 2003). The modification of hydrology also impacts other processes such as decomposition and the CO₂ uptake capacity of mosses, with an 85% reduction when mosses are air dried rather than saturated (DeLucia et al. 2003). Mosses also play a role in evapotranspiration, accounting for 49-69% in poorly drained soils and 18-31% in well drained soils under tree canopies in boreal forests (Bond-Lamberty et al. 2011). Many studies have also inferred that soil moisture under a moss layer is higher than under bare soil (Belnap 2006; Oechel and Van Cleve 1986) although there are only a few examples to be found in New Zealand (DeLucia et al. 2003; Michel et al. 2013). Conversely, the presence of a moss ground cover may also affect soil infiltration rates both positively and negatively within dry environments and grasslands (Belnap 2006; Eldridge and Greene 1994; Eldridge et al. 2010).

III. Effects of moss on nutrient and carbon cycles

Mosses are important in the context of nutrient and carbon cycling in many ecosystems, particularly carbon sequestration and nitrogen fixation. Mosses can form associations with nitrogen-fixing cyanobacteria; these are known in a number of species including *Bryum algovicum*, *Grimmia* spp. *Racomitrium* spp. and *Ceratodon purpureus* in a variety of habitats (During and Tooren 1990). The cyanobacteria supplies nitrogen to the mosses retaining it in the ecosystem, potentially preventing losses from the system through leaching (Hawkes 2003; Zielke et al. 2005). Rousk et al. (2016) used ¹⁵N-N₂ pulse labelling and ¹³C-C labelling to detect the fate of N and C within a boreal forest. They found that legumes released nitrogen into the soil within a matter of days whilst lichens and mosses retained the fixed nitrogen. In subarctic regions mosses are considered important drivers for atmospheric N

additions into the ecosystem from this source (Gavazov et al. 2010). In arid juniper woodland environments in North America, Evans and Ehleringer (1993) observed that the soil covering of cyanobacteria, lichens and mosses was the predominant input of nitrogen into the ecosystem. In addition to inputs of N from these associations, certain mosses are also efficient at sequestering nitrogen precipitation inputs (wet and dry) in the live moss and accumulating organic matter making it unavailable for vascular plants in the short term due to the long decomposition period of bryophytes (Bowden 1991; Cornelissen et al. 2007; Jonsdottir et al. 1995).

Mosses account for up to 50% of the total photosynthesis in some ecosystems, including boreal forests, contributing significantly to gross primary production (Kolari et al. 2006; Vanderpoorten and Goffinet 2009). Sequestered carbon is then stored long-term in some ecosystems, such as peatlands (Gorham et al. 2012), due to slow rate of decomposition mediated by the pronounced effect of the moss cover on moisture and temperature (Turetsky 2003). This is most evident in peatlands where degradation is impeded through a number of mechanisms including the physiology of the moss, low pH, increased moisture, low temperatures and anaerobic conditions (Davis 1981; Gorham et al. 2012). In arctic climates, mosses have been known to fix large amounts of carbon, with the majority stored in recalcitrant pools (Rousk et al. 2016; Street et al. 2013). Bona et al. (2013) considered the carbon storage within feather and sphagnum mosses to be sufficiently large that it be included in national-scale carbon models for boreal forests in Canada. However, in a New Zealand temperate rainforest where liverworts were the dominant bryophyte, the levels of CO₂ uptake were reported as much lower than boreal systems at 10% of the total forest floor efflux (DeLucia et al. 2003).

Other nutrients important for plant growth are also influenced by mosses. Wells and Boddy (1995) found that phosphorus inoculated into dead wood was taken up by the widespread pleurocarpous moss found at Eyrewell Forest, *H. cupressiforme*, through the action of a connected saprophytic basidiomycete. This reveals the potential for uptake of nutrients from the substrate thereby preventing use by higher plants. A similar phenomenon was reported in a black spruce forest in Alaska where mosses accounted for 75% of the annual phosphorus accumulation with increased absorption rates of P from solution containing ³²P-labelled phosphate compared to spruce trees (Chapin et al. 1987). However, some mosses have also been reported to leak nutrients from their cells upon rehydration which may be detrimental to the moss but beneficial to surrounding plants (Brown and Buck 1979; Wilson and Coxson 1999).

IV. The influence of moss on vascular plants

The physio-chemical effects of mosses on soil and ecosystem processes as discussed above (protection against erosion, buffering of extreme temperature and moisture loss, potential assistance in nutrient cycling) may help to facilitate establishment of higher plants, particularly in arid and cool environments

(Coulson et al. 1993; During and Tooren 1990; Eldridge and Greene 1994; Eldridge 1998; Van der Wal and Brooker 2004). There is also evidence mosses may affect the germination, establishment and growth of vascular plants.

Germination:

Carpets of moss may inhibit the germination of vascular plants by providing a barrier to the soil or rooting medium, drying the seed or maintaining excess moisture, sustaining temperatures too cold for germination, restricting light levels and increasing the aboveground time available for seed predation (During and Tooren 1990; Head et al. 2004; Sohlberg and Bliss 1987; Zamfir 2000). Conversely, there is evidence to suggest that the moss layer may act to camouflage seed against predation (During and Tooren 1990) and promote regeneration of certain vascular plants in cool or arid environments by providing interspaces to retain seed and vital resources (such as water and nutrients) (Belnap 2006; Sohlberg and Bliss 1987). In temperate grassland in Australia, the moss layer was shown to favour the seed morphology of indigenous plants preventing germination of non-native species (Morgan 2006). Similarly, in the UK, *Hylocomium splendens* was found to facilitate *Calluna vulgaris* growth in heathland environments (Gimingham 1972). However, *H. cupressiforme* was not found to have the same effect (Scandrett and Gimingham 1989) suggesting that interactions may be species specific (Sohlberg and Bliss 1987).

Emergence and Establishment:

The effect of moss on the subsequent emergence of vascular plant species has been observed to have both positive and negative effects in different environments (During and Tooren 1990). In limestone pavements in Sweden, the biomass of vascular plants within moss cushions increased as the size, age and detritus layer of moss cushions increased (Sand-Jensen and Hammer 2012). Rayburn et al. (2012) examined the occurrence of an endemic primrose *Primula cusickiana* and moss cushions in Utah, USA, finding *Primula* responded in terms of higher abundance and vigorous growth when growing in combination with moss. They concluded that the moss acted as a seed trap for the plant and/or buffered extreme abiotic conditions. In other studies it was observed that lichens and mosses may exhibit an inhibitive effect on germination but once vascular plants are established the bryophytes can promote growth (Houle and Filion 2003; Zamfir 2000).

Nutrient supply:

Mosses may limit available nutrients to surrounding vascular plants by intercepting available nutrients from wet and dry deposition. They may also acquire nutrients from the substrate, sequestering and releasing them slowly via decomposition when the moss dies (Oechel and Van Cleve 1986). Decomposition of mosses is slower than for lichens or vascular plants although species variations do

occur (Lang et al. 2009). A number of attributes may account for this delay including their nutritional status and phenolic compounds, acidic environmental conditions, low temperatures and moisture content yet decomposition rates for many species is still unknown (Bates 2009; Brown and Bates 1990; Lang et al. 2009; Turetsky 2003). In contrast, there is evidence to suggest that mosses may supply nutrients, such as phosphorus, to vascular plants through mycorrhizae networks although more research is required to substantiate this link (Brown and Bates 1990; Davey and Currah 2006; During and Tooren 1990).

2.3.4 Bryophytes in Restoration

The literature review has provided information that implies that bryophytes have an important role within forest ecosystems, modifying soil, plant and faunal processes and exhibiting a range of inhibitory and facilitating functional mechanisms. Michel et al. (2013) considered bryophytes critical to the functioning of native tussock grassland systems in New Zealand and the same may be true for forests systems. This raises the question of whether bryophytes should be an integral part of ecological restoration practice.

As discussed previously, bryophytes are widespread in New Zealand whilst presenting low endemism. Two species found within the kānuka stands in the Eyrewell Forest, *H. cupressiforme* and *Bryum argenteum*, are described as widespread and abundant (Atherton et al. 2010; Beever et al. 1992). So why would the requirement exist to actively introduce these species? When mosses are further considered in terms of geographic range, habitat specificity and genetic variation, many mosses, and even those common species such as *B. argenteum* that displays genetic variation between habitats, can be classified as “rare” (Longton and Hedderson 2000). Furthermore, despite the longevity of propagules and the long-distance dispersal mechanisms of many mosses which has allowed plants to colonise every continent, mosses are slow to establish (Belnap 1993; Frahm 2009). This dichotomy can be partly explained by recent habitat fragmentation. Studies have illustrated the negative affect of habitat fragmentation on the composition of moss communities (Perhans et al. 2009; Pharo and Zartman 2007). Long distance dispersal cannot overcome the barriers of fragmentation at a spatial or temporal scale, especially with regard to species found within old growth forest (Snäll et al. 2003).

Bryophytes have been used in restoration projects in arid, cool or wet environments with varied success (Bowker 2007; Forbes 1993; Graf and Rochefort 2010; Mälson and Rydin 2007). Cole et al. (2010) found that in the Mojave Desert bryophyte soil crusts could be transplanted to grow in the same fashion as those crusts which remained in the parent sites. Research into bryophyte restoration in forest ecosystems is limited with the exception of studies within boreal forests or with epiphytic species. The introduction of epiphytic bryophytes as fragments on branches or within net bags had varying success in both tropical and boreal forest environments (Hazell and Gustafsson 1999; Nadkarni

et al. 2000; Rosso et al. 2001). In a northern hardwood forest in the USA, Kimmerer (2005) found that terricolous mosses did not establish onto artificially created treefall mounds within northern hardwood forests in the USA unless introduced by vegetative fragments.

The likelihood of species dispersing into an area is greatly reduced when it is situated more than 200m from an existing population, even where the receptor site is close to a donor site, the understorey may take up to 100 years to establish (Honnay et al. 2002). The dispersal mechanisms of mosses, namely clonal expansion, germination of diaspores in the soil and dispersed diaspores from nearby populations (Rydgren and Hestmark 1997), means that unless donor sites are adjacent or the diaspores are already present in the soil it may be necessary to inoculate species to gain the desired community. In addition, the microsite conditions, including edaphic conditions and light availability, will also determine the success of regeneration (Caners et al. 2009; Frego 1996). In boreal forests with limited fragmentation, the amount of variation within a forest with regard to microsites is important with regard to bryophyte diversity (Fenton and Bergeron 2008). Therefore, in addition to the functional role of bryophytes within the forests and shrublands of New Zealand, ways in which the species can be restored into habitats need to be explored.

Chapter 3

Eyrewell Forest: Kānuka Shrubland Description and Ecological Study

This chapter introduces the main study site at Eyrewell Forest and provides descriptions of the known remnant indigenous communities and major plant associates. I also provide a description and details of an ecological study that I undertook to assess the remnant sites of the immediate area with regards to community assemblage, regeneration and edaphic conditions.

3.1 The Eyrewell Forest

Eyrewell Forest is a rural area situated within the Canterbury Plains, immediately north of the Waimakiriri River, with Kaiapoi to the east and the Canterbury foothills to the west (Figure 3.1). The area comprises stony, free draining soils of the Lismore series (a stony silt loam of the Pallic Orthic Brown Soil classification) and supports a mosaic of agricultural land intersected with watercourses and tree-lined field boundaries. The forest itself comprised a large area of land (6,764 ha) which was planted as production pine forest in the early 1930's, mainly radiata pine (*Pinus radiata*). Prior to this the area was relatively unproductive due to the dry soils and mainly used for sheep farming. The New Zealand Forestry Service note that parts of the area were “covered in dense mānuka (*Leptospermum scoparium*), which were up to 30ft tall” (Wendelken 1966). The native shrubland was “crushed with a heavy roller” and inter-planted with the pine trees. However, the forest suffered substantial damage from failure of trees due to soil conditions, extensive wind throw in storm events and fires (Heath 1979; NZ Herald 2015; Wilson 2015). The forested area was Crown owned until 2000 when Ngāi Tahu Property purchased the land under the Ngāi Tahu Settlement (Ngai Tahu Farming 2016). As the forestry licences expired the area, now known as Te Whenua Hou, was converted from plantation to pasture, a process still ongoing in 2017 (Plate 3.1). Ngāi Tahu set aside 150 ha of land for indigenous habitat restoration and an additional 150 ha for landscape planting on the margins of farms, paddocks and around farm buildings (Ngai Tahu Farming 2016).



Plate 3.1 Eyrewell Forest during conversion from pine plantation to pivot-irrigated dairy farm in 2014.

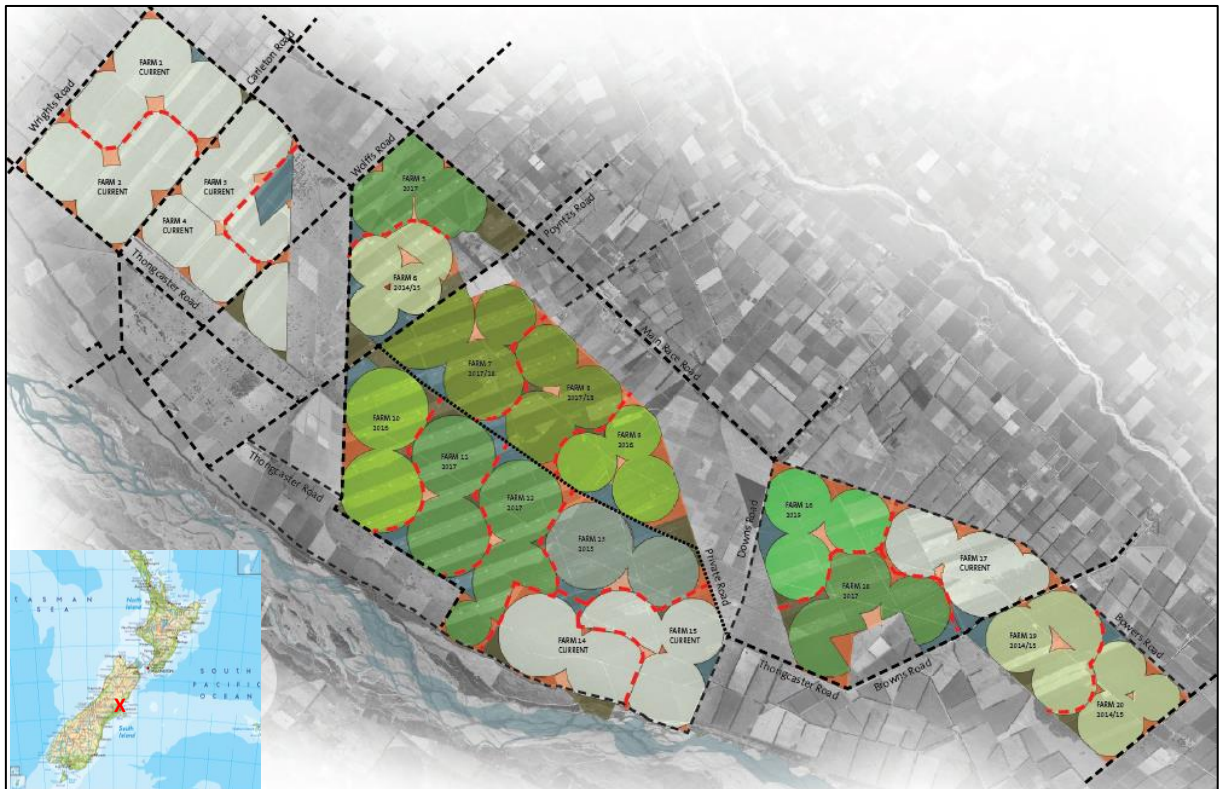


Figure 3.1 Plan illustrating the footprint of the Eyrewell Forest plantation site and Ngāi Tahu's design to convert the plantation into a number of irrigated pasture farms. Green shaded areas outside of the centre pivots denote areas set aside for restoration but corridor restoration is not shown. Inset is a map showing the location of the site within New Zealand.

3.2 Remnant Vegetation Communities and Study Sites

The Eyrewell Forest (Te Whenua Hou) is situated within the Canterbury Low Plains Ecological District (ED) which historically was thought to contain approximately 6-25% kānuka (*Kunzea* sp.) forest with additional 1-5% of kānuka-mānuka (*Leptospermum scoparium*) shrubland in dryland, stable gravel areas but now less than 1% remains and only 28% of that figure is protected (Cockayne 1928; Harding et al. 2009; McGlone 2001). Meurk et al. (1995) documented six significant areas of kānuka shrubland

remaining within the Low Plains ED, totalling less than 200 ha. All remnant areas were less than 20 ha and the protected reserve fragments were less than 5 ha. A limited number of fragments are protected under QEII covenants, Natural heritage Fund land, Ngā Whenua Rāhui land and held as Department of Conservation (DOC) reserves. However, legal protection does not necessarily safeguard the habitat as illustrated by the account of 2.4 ha of indigenous protected shrubland being cleared in 2003 by a private company (QEII National Trust 2014). Furthermore, legal protection does not afford security from natural hazards as demonstrated by the events at DOC Medbury Scientific Reserve; an area of more than 55ha of kānuka forest and associated native grass, herb and moss covered area within the Low Plains ED which was largely destroyed by fire, losing 70% of the vegetation (Ecroyd and Brockerhoff 2005; Head et al. 2004; McEwan 1987).

Examples of DOC protected dryland remnant areas are Eyrewell, Bankside and Medbury Scientific reserves; the latter being described above. Bankside comprises 2.6 ha of open kānuka shrubland and native grassland on dry Eyre soils (Williams and New Zealand Department of Conservation 2005); Eyrewell Scientific Reserve (in this thesis referred to as ESR) is similar to Bankside comprising 2.3 ha of dense kānuka, associated native shrubs, native grassland and moss-herb fields (Plate 3.2) (Williams and New Zealand Department of Conservation 2005). This is a mature shrubland comprising a substantial cover of plains kānuka (*Kunzea serotina*) associated with an understory of prickly mingimingi (*Leptecophylla juniperina*) and an almost continuous ground cover of the pleurocarpous moss, *Hypnum cupressiforme* (Molloy 1972; Molloy and Ives 1972; Wardle 1991). The area also contains other native shrubs and grasses with some exotic species becoming dominant in the understorey of the more open canopy areas (Ecroyd and Brockerhoff 2005). This description aligns largely with the “Kānuka shrubland with Coprosma and prickly mingimingi” alliance as defined by Landcare Research (Wiser et al. 2013; Wiser et al. 2011). However, it should be noted that none of the National Vegetation Survey plots used to define the shrubland communities are located within the Eyrewell Forest area (there are relatively few within the whole of the Low Plains ED due to the lack of indigenous vegetation). This is reflected by the fact that the description offered by Wiser et al. (2013) notes that the kānuka alliance contains frequent *Leucopogon fasciculatus* a species that has not been reported in the Eyrewell Forest shrublands (Ecroyd and Brockerhoff 2005). Furthermore, Meurk et al. (1995) found compositional differences in many kānuka stands within the area, possibly indicating that the shrubland found within the Eyrewell area is a complex alliance which has not been properly described due to lack of survey effort (Bretherton et al. 2002). It could also reflect the influence of the surrounding, highly modified agricultural environment with its history of multiple disturbances.

Bankside Scientific Reserve lacks the prickly mingimingi association and has now been substantially influenced by the intensive dairy farming surrounding the reserve leading to an increase in exotic grasses and shrubs due to increased irrigation and nutrient availability (Bowie et al. 2016;

Emberson et al. 2011). Bowie et al. (2016) recorded only 19 indigenous species compared to 113 observed in 1970, and 27 new exotic species were found.



Plate 3.2. The author and Shanshan Li in the dense kākūka stands within the DOC owned Eyrewell Scientific Reserve (top left). Google Earth images of ESR (top right) and the privately owned Spencer-Bower reserve (SBR, bottom).

In addition to formally protected areas, there are a number of smaller kākūka shrublands within the Eyrewell Forest region which are under private ownership (Meurk et al. 1995). One notable area is located adjacent to Hetherton Road, herein referred to as SBR (after the owners of the reserve, Spencer-Bower Reserve, Plate 3.2). This area is similar in composition to the ESR but lacks the *Coprosma* affiliation and has a more dominant understory of prickly mingimingi.

The pine plantation footprint of Eyrewell Forest also harbours kākūka habitats under the pine canopy and along watercourses, road verges, forest stand boundaries and converted paddock and

farm margins. The forest has been an important haven for indigenous vegetation with the exotic pine trees acting as canopy cover for many shade tolerant indigenous plants and is noted as having potential for conserving kānuka remnants (Ecroyd and Brockerhoff 2005; Harding et al. 2009). It has been referred to as an undervalued “little known resource” (Bretherton et al. 2002) containing a complex mix of species, some of conservation significance (Brockerhoff et al. 2005; Ecroyd and Brockerhoff 2005; Meurk et al. 1995). Plains kānuka is the most frequent species with occasional plant associates, including prickly mingimingi, hypnum moss (*Hypnum cupressiforme*) and a notable plant species, pomaderris (*Pomaderris amoena*) which is at the southern limit of its range (Molloy and Ives 1972).

Bretherton and Given (2002) considered that the Kānuka shrublands present on the Canterbury Plains offer research opportunities into effects of the fragmentation, management and design of agricultural landscapes to achieve economic and biodiversity gains. In addition, they suggest the fact that there appears to be no formal description of the habitat within the Low Plains ED indicates that it could be a much localised, significant community and a “little known resource” (Bretherton et al. 2002).

3.2.1 Community Characteristics and Associations

The kānuka shrubland remnants within and surrounding the Eyrewell Forest comprise of dense stands of *Kunzea serotina*, endemic to New Zealand (New Zealand Plant Conservation Network 2013a). Botanically, the shrublands support a complex mix of indigenous vascular plant species, but the most notable are prickly mingimingi and hypnum moss (Ecroyd and Brockerhoff 2005; Meurk et al. 1995; Molloy and Ives 1972). Two further associates which are not as abundant are also included in the experimental work in this chapter; one due to its strong association with the Canterbury Plains (*Carmichaelia australis*) and one due to the importance within the area (*Pomaderris amoena*). Native broom (*C. australis*) was present in many of the sites that Meurk et al. (1995) studied and also in every habitat (apart from pasture) studied by Ecroyd and Brockerhoff (2005). Below is a brief description of these plants and the moss commonly found within the shrubland.

***Kunzea serotina* De Lange & Toelken (2014) (Myrtaceae)**

Kānuka and mānuka (Myrtaceae) were formally described as *Leptospermum ericoides* and *L. scoparium* respectively until they were later revised and separated into two genera; *Kunzea* and *Leptospermum* (Connor and Edgar 1987; Thompson 1983). Both species have seeds released from serotinous capsules following fire, a characteristic which is uncommon in the New Zealand flora (Perry et al. 2014). *Kunzea* is a genus also known in Australia but New Zealand has 10 endemic species (de Lange et al. 2006; de Lange 2014). *Kunzea serotina*, referred to as plains kānuka (Plate 3.3), can be found in areas of stony soils that are frost-prone from 30-2000 m a.s.l. It is found on both North and South Islands, from the

central volcanic plateau in the north to central Otago in the south (New Zealand Plant Conservation Network 2013a).

Kānuka is a fast growing, relatively short lived (80-150 years) shrub or tree, preferring drier, free draining soils (Burrows 1973). Burrell (1965) noted that plants are usually 1.5 m tall and 7 years of age before they are sexually viable. The plant is xeromorphic (Johnson 1984), a characteristic that enables it to grow in areas which are inhospitable to taller forest species, as in the Canterbury Lower Plains ED (Burrows 1973). It has been described as an r-strategist due to its small, wind dispersed seed and seedling phenotypic plasticity including an observed increase in stem diameter and number of stems when in the open as opposed to under a closed canopy (Kirschbaum and Williams 1991). However, the plant does not have the ability to disperse readily over long distances although it produces large soil seed banks close to parent stands spread by wind or drainage (Kirschbaum and Williams 1991; Ogden 1985). Natural and human caused fires probably increased its cover compared to the larger forest trees such as totara (*Podocarpus totara*) (Meurk 2008).



Plate 3.3. Kānuka shrubs along a watercourse embankment at Te Whenua Hou.

Kānuka seedlings can germinate in a variety of conditions and seeds can remain viable for at least 5 months if kept dry. However, germination is slower under dark conditions and seedlings resulting from buried seed do not survive (Burrows 2006). Seedlings establish effectively on bare soil or in lightly grazed pasture more so than under canopy (Wilson 1994) and results indicate that freshly deposited seeds tends to germinate more readily than those found in the soil seed bank (Allen et al. 1992).

Mycorrhizal associations are prevalent and can facilitate nutrient and moisture uptake by plants and provide resistance to fungal pathogens (Allen 1991). Kānuka forms both ecto- and arbuscular mycorrhizal symbiotic associations (McKenzie et al. 2006). The plants can also act as a host to promote generic mycorrhizal infection of other species, such as indigenous beech and exotic conifers, producing enhanced establishment and growth (Davis and Smaill 2009; Dickie et al. 2012).

In general, little is known about the nutrient status of indigenous seedlings in the field; elsewhere this often leads to failure of planting (Bradshaw and Chadwick 1980). Kānuka seedlings can tolerate shade and root competition more than mānuka (Burrows 1973) but are still vulnerable to competition from herbaceous plants leading to successful establishment mainly in open areas or via gap phase regeneration (Allen et al. 1992; Esler 1967; Singer and Burgman 1999). In Australia, Kirschbaum and Williams (1991) found that kānuka recruitment was rife under the closed, possibly senescent kānuka canopies, indicating that populations studied were somewhat tolerant of intraspecific competition. Kānuka plants were found to be self-replacing following wind throw or mortality of older individuals (Payton et al. 1984). Supporting this, kānuka stands have been found to regenerate under exotic pine plantation at the site of the present study, Eyrewell Forest, where timber thinning seemed to encourage the native understorey (Brockerhoff et al. 2003). Thus, kānuka communities can replace themselves continually when disturbed through natural disturbances (Smale 1994).

***Leptecophylla juniperina* (J.R Forst. & G Forst) C. M Weiller (1999) (Ericaceae)**

With regard to plant associations, Meurk et al. (1995) surveyed five key kānuka sites in the Eyrewell area and found that prickly minigimigi, *Leptecophylla juniperina* (J.R.Forst. & G.Forst.) C.M.Weiller, was the main woody associate (Plate 3.4). Previously known as *Cyathodes juniperina*, this is an indigenous plant found across New Zealand and Tasmania, Australia within the southern heath family, Ericaceae (Flora Committee Landcare Research 2013). The plant is common across New Zealand in shrublands and forests from coastal to montane habitats and forms a bushy, prickly shrub with white/cream flowers and red or white berries (or various shades between the two colours) (New Zealand Plant Conservation Network 2013b). It has Maori significance having been used medicinally, for scent, and the berries were eaten (Landcare Research 2014a).



Plate 3.4. *Leptecophylla juniperina* under the Eyrewell Forest pine canopy with white drupes. The species can also have red drupes or any colour in between.

Prickly mingimingi also appears to be associated with kānuka in other parts of New Zealand. When plants were observed regenerating under stands of kānuka and invasive gorse (*Ulex europaeus*) in Nelson, there was a complete loss of prickly mingimingi and partial loss of *L. fasciculatus* under gorse as opposed to beneath the kānuka stands (Sullivan et al. 2007). There were thought to be many possible reasons, including altered moisture levels in the soils, a change in soil nutrient status, different soil fauna under the two different canopies or mycorrhizal associations.

Prickly mingimingi belongs to the sub-family Styphelioideae, associated with the Vaccinoideae. They have specialised distal roots which are colonised by ascomycete fungi to form ‘ericoid mycorrhiza’ (Smith 1997b). A number of the fungi associated with ‘ericoid mycorrhizae have been found to be associated with Kānuka (McKenzie et al. 2006; Smith 1997b) leading to the assumption that fungal infection may be an important factor in assisting prickly mingimingi to grow under the Kānuka canopy.

***Carmichaelia australis* R. Br (1825) (Fabaceae)**

Common native broom (*Carmichaelia australis*) is a widespread endemic shrub of the Fabaceae (Plate 3.5). It is found in a range of habitats from montane and river terraces to shrublands, forest and swamps across the two mainland islands of New Zealand, except the southern part of the South Island (Heenan 1996). Members of the Fabaceae are colonisers that harbour nitrogen fixing bacteria within root nodules containing rhizobia (Lewis et al. 2005). McCallum (1996) found that *C. australis* associates with *Mesorhizobium* spp. and Bellingham et al. (2001) found that nitrogen input increased along a chronosequence in which a species of native broom had encroached, as did the woody vascular plant element, illustrating the role of broom as a facilitator in primary succession.



Plate 3.5. The flattened cladodes and the purple veined flowers of *Carmichaelia australis*.

***Pomaderris amoena* Colenso (1886) (Rhamnaceae)**

Pomaderris amoena is an endemic New Zealand species which is also present within the kānuka stands at Eyrewell (Plate 3.6). Here it is at the southern limit of its range in New Zealand. Most populations occur in the northern half of the North Island, or at the southern tip surrounding Wellington (New Zealand Plant Conservation Network 2014). This plant is often found in shrubland, rocky montane or coastal environments (Duguid 1990; New Zealand Plant Conservation Network 2014), which are perhaps with similar environmental conditions to the drought prone Canterbury Plains. This species appears to reproduce asexually via apomixis (Harvey and Braggins 1985). Little is known about the ecology of the plant, although a similar species in Tasmania, *Pomaderris apetala*, was found the regenerate profusely following fire and form ecto-mycorrhizae associations (Warcup 1980b; 1991). In addition, plants from the same genus also found in New Zealand, *P. hamiltonii* and *P. kumeraho*, showed increased germination rates when heat treated with boiled water (Haines et al. 2007). These plants also reacted to smoke treatment when seeds were warmed during germination indicating the importance of fire for regeneration. Dispersal of seeds of some *Pomaderris* species have been reported to be carried out by ants in Australia (Gardner 1996).



Plate 3.6. *Pomaderris amoena* plant in flower situated along a road boundary under the pine canopy at Eyrewell Forest.

***Hypnum cupressiforme* Hedw. (1801) (Hypnaceae)**

The remnant kānuka stands at Eyrewell Forest have a thick carpet of plait moss, *Hypnum cupressiforme* (hypnum moss, Plate 3.7), which covers the surface of the soil and appeared to be the main contributor to soil organic matter (Molloy and Ives 1972). The moss is an associate with kānuka forest in many areas of New Zealand (Beever and Brownsey 1993; Beever and Brownsey 1990; Smale 1994; Wilcox et al. 2004). The role of the moss within these communities has been little studied although, globally, there is evidence that bryophytes are often major contributors to hydrology, nutrient and carbon cycles within ecosystems (Belnap 2006; Porley and Hodgetts 2005; Vanderpoorten and Goffinet 2009).



Plate 3.7. *Hypnum* moss carpeting the forest floor of a kānuka remnant site in Eyrewell Forest

Hypnum is a pleurocarpous moss which is widespread globally and found in a number of habitats, preferring acidic substrate from sand dunes and heathlands to woodlands in dry environments (Michel et al. 2011b; Porley and Hodgetts 2005). The moss is extremely variable and has been described in many varieties and alongside closely related species (Smith 1997a). It is dioecious and can be dispersed via a number of vectors including wind, water and animals or grown from diaspore banks (Bates 2009; During 2001; Porley and Hodgetts 2005). Diaspore banks may be significant in the maintenance and resilience of plant communities, but are poorly known (Maciel-Silva et al. 2012).

Like most bryophytes, hypnum moss photosynthesises in moist conditions and is tolerant of desiccation by employing poikilohydry which allows the plant to dehydrate without damage to tissues, commencing photosynthesis upon rewetting (Proctor et al. 2007). Hypnum is an ectohydric moss and is effective at absorbing water and nutrients found in this surface water or through dry deposition (Tuba et al. 1996). For this reason, the moss has been used as a bio-indicator to monitor atmospheric deposition of many heavy metals as it can absorb concentrations of heavy metals over its large surface area when found close to pollution sources (Proctor 2009; Richardson 1981; Ward et al. 1977).

3.3 Eyrewell Kānuka Habitats: Ecological Study

The composition of the kānuka remnants within and surrounding the Eyrewell forest have been studied on various occasions as described above, but, limited studies have been carried out on the ecology of these stands. The aim of this chapter was to investigate the edaphic conditions, and seedling occurrence and species composition of kānuka habitats in different locations, thus considering the drivers of ecosystem maintenance and regeneration. The study explored differences in the kānuka stands that existed as remnants, regenerating under pine plantations, and those within field boundaries with regard to abiotic and biotic parameters as a direct result of substrate, dispersal constraints of biota into the restoration area and the spatial considerations of location, size, fragmentation and adjacent land use; the important factors described by Honnay et al. (2002). Field work aimed to:

- i. Investigate the composition the kānuka stands with regard to exotic species present and total species richness;
- ii. Understand the edaphic conditions of the kānuka shrublands within the Eyrewell Forest; and
- iii. Understand the regeneration of the main woody associates.

3.4 Methodology

3.4.1 Habitat Survey

Site Selection

A total of twelve sites containing kānuka were identified either within the Eyrewell Forest footprint or adjacent to the Ngāi Tahu owned land. These sites consisted of kānuka habitats in three different environmental conditions (Figure 3.2):

- Four kānuka stands regenerating under the pine forest canopy. These sites were chosen due to the plant assemblages containing both *K. serotina* and *P. amoena*;
- Four kānuka stands along field boundaries, surrounded by irrigated farm pasture. The sites were chosen to be representative of the whole of the site from east to west; and
- Four stands of remnant vegetation, as previously described by Meurk et al. (1995). These sites were of limited existence, and the most accessible sites that bordered the Eyrewell Forest area were chosen.

The last habitat category contained three sample locations with one encompassing two separate sites due to its size (descriptions of ESR, SBR and kānuka under the pine are outlined above in Section 3.2). The transects in this habitat were separated by at least 100 m. The location of the sites can be viewed in Table 3.1.

Table 3.1 Location, ownership and classification of kānuka shrubland ecology study sites.

Habitat Classification	Location	Name and Owner
Under Pine Forest	43°42'53.01" S, 172°31'66.38"E	C4, Ngāi Tahu
	43°41'80.9" S, 172°27'48.9"E	C1 Ngāi Tahu
	43°42'08.7" S, 172°28'72.1"E	K3, Ngāi Tahu
	43°39'20.4" S, 172°28'50.8"E	P1, Ngāi Tahu
Field Boundary/ Grassland	43°39'50.4" S, 172°19'36.7"E	K1, Ngāi Tahu
	43°42'42.5" S, 172°25'25.8"E	K2, Ngāi Tahu
	43°42'46.8" S, 172°32'17.6"E	K4, Ngāi Tahu
	43°42'26.7" S, 172°45'02.6"E	K5, Ngāi Tahu
Kānuka Remnant	43°38'33.3" S, 172°19'38.3"E	ESR, DOC
	43°40'77.5" S, 172°35'28.3"E	Downs Road, Private Ownership
	43°42'70.6" S, 172°42'67.5"E	SBR 1, Private Ownership
	43°42'88.9" S, 172°43'29.1"E	SBR 2, Private Ownership



Figure 3.2 Study habitats showing examples of field boundary (top left), under the pine (bottom left) and the remnant habitat with the dense kānuka stand (top right) and continuous moss carpet (bottom right).

All field data were collected during the summer/early autumn months of 2014 (January to March). At each site, sampling points were located within a 5 m x 5 m quadrat along a 100 m linear transect at 20 m, 40 m, 60 m and 80 m. The transect was positioned roughly in the centre of the habitat, parallel to a fenceline where possible. The measurements taken were based on the REECE methodology which divided the vegetation into 6 vertical tiers from the ground up to the canopy. The shrubland tier class was employed due to the majority of the vegetation being under 12m in height. In each tier percentage ground cover, canopy cover, species present and their percentage cover class was recorded (Hurst and Allen 2007). Litter cover was also reported, defined as dead plant parts on the surface, including pine needle and dead wood. In addition, the number of seedlings present within the quadrat was recorded for *K. serotina*, *P. amoena* and *L. juniperina*. A seedling was defined as a young plant under 50 cm in height. Light intensity readings (Lux) were also taken but these produced varying results due to fluctuating cloud cover and are not presented here. Instead, percentage canopy cover of the 5 m quadrat was estimated and gave an indication of light availability.

Chemical Analysis

At the first sampling point on each transect, three soil cores of 5 cm diameter were taken at random locations within the quadrat and bulked. Topsoil (the top 7.5 cm) and subsoil (7.5 cm to 15 cm deep) were sampled separately using a soil corer. Chemical analysis was carried out following standard methodologies (Blakemore et al. 1987). Fresh soil samples were sieved to 4mm and stored at 5°C kept in the fridge in air tight polythene bags for no longer than two days. To gain gravimetric moisture content, 10 g of field moist soil was dried at 105°C for 24 h. A further 4g sub-sample of field moist soil was extracted with 2 M KCl (potassium chloride) to estimate nitrate (NH_3^-) and ammonium (NH_4^+) concentrations. The soil solution (soil and 40 mL KCl) was shaken for 1 h, centrifuged at 2000 rpm for 10 minutes and then filtered through Whatman No. 41 filter paper. The extracts were analysed using Flow Injection Analyser (FIA) (Foss FIAstar 5000 triple channel with SoFIA software version 1.30. Foss Tecator, Hoganas, Sweden).

The remainder of the samples were air dried at 25°C for four days, ground and sieved to 2mm. Plant available P was analysed following the Olsen P method (Watanabe and Olsen 1965). Samples of air dried soil (1 g) were shaken for 30 minutes with 20 mL 0.5 M sodium bicarbonate (NaHCO_3). The extractant was centrifuged for 10 minutes at 2000 rpm and filtered through Whatman No. 42 filter paper. Murphy Riley colour reagent (5 mL) (Murphy and Riley 1962) was added to the filtrate (10 mL) and the sample was made up to 50 mL with dionised water. Absorbance was measured after 30 minutes and read at 880 nm on a Shimadzu UV mini-1240 spectrophotometer. Total organic carbon was measured via loss on ignition using 10 g air-dried soil (following 2h at 105°C) which had been ignited in the muffle furnace at 500°C for 4 h. Soil pH was analysed using a suspension of 10 g air-dried soil with 25 mL dionised water left to stabilise for 12 h and analysed using S20 SevenEasy™ pH meter (Mettler-Toledo, Switzerland). Total carbon and nitrogen were analysed using a LECO CNS Elemental Analyser (LECO Australia Pty Ltd, NSW, Australia).

Data Analysis

Plant species information was collated and organised using a phytosociological classification similar to that used in the National Vegetation Classification system (NVC) employed for UK plant communities (Rodwell 2006). The cover of individual species and their frequency of occurrence within each plot were calculated to produce floristic tables detailing the species characteristic of the habitat (constant species with 60-100% frequency). The Braun-Blanquet cover scale (Braun-Blanquet 1932) and the NVC frequency grading classes were used (Table 3.2).

Table 3.2 Grading of plants within the phytosociological system with regard to frequency and cover classes. Frequency is measured as percentage occurrence within a number of plots and cover is measured as mean percentage cover of a plant within a plot.

NVC Frequency Class	
Frequency Class	Frequency between Plots
I (scarce)	1-20%
II (occasional)	21-40%
III (frequent)	41-60%
VI (constant)	61-80%
V (constant)	81-100%

NVC Cover Class	
Cover Class	% Cover Within Plots
1	1-5%
2	6-25%
3	26-50%
4	51-75%
5	76-100%

Plant species were assessed as being either pioneer (colonisers following disturbance preferring full light conditions), secondary (those species replacing pioneer species and facilitating transition to a more stable state tolerating intermediate light conditions) or primary (those species likely to form communities within the steady state being shade tolerant) as defined by Brockerhoff et al. (2003), Moles and Drake (1999) with additional information gleaned from New Zealand Plant Conservation Network (2017) and Rose et al. (2006).

Mean values and standard errors were calculated for each measured parameter. One-way Analysis of Variance (ANOVA) and post-hoc Tukey Honestly Significance Difference (HSD) tests were used to determine differences of non-vascular plant cover, canopy cover and litter cover in each of the habitats. Data were transformed where appropriate to meet the normality assumptions of the test. The seedling data count data could not be acceptably transformed (O'Hara and Kotze 2010) therefore differences of seedling occurrence in habitat, non-vascular plant cover, canopy cover and litter cover was analysed using the Kruskal-Wallis non parametric test.

To determine differences in soil chemistry within each habitat, a multivariate (MANOVA) test was applied to both topsoil and subsoil. To gain Gaussian distributions the gravimetric moisture content, Olsen P, NO_3^- and NH_4^+ data was transformed using Johnson transformation. A one-way ANOVA with post-hoc Tukey (HSD) test was applied to test for differences across habitats for each parameter.

Principal Component Analysis (PCA) were performed for all non-transformed biotic data and soil parameters in the topsoil and subsoil. Principal components were tested for linear relationships

using the Pearson's Correlation Coefficient (r). All data analysis was carried out using Minitab® (Version 17.2.1., Minitab Inc., Sydney, AUS) and graphs created using SigmaPlot (Version 12.3, Systat Software, San Jose, CA).

3.4.2 Individual Species Investigation

A series of germination trials were undertaken to assess optimal germination conditions for four of the associate species within the kānuka shrubland. *L. juniperina*, *P. amoena*, *C. australis* and *K. serotina* seeds and drupes were sourced from the Eyrewell Forest area and were cleaned and inspected under a microscope for damage prior to storage for two months in at 3-5°C. For *L. juniperina*, the drupes were separated into their fruit colour (red or white) and the flesh was removed to reveal the endocarp before being stored.

Seeds were subjected to a number of conditions: light conditions simulating open, full light environments; dark conditions simulating those found under a canopy or buried in leaf litter; and soil to assess the benefit of any soil microbes. The light condition involved seeds being placed on moist filter paper inside a petri dish and covered in a transparent plastic (PVC) wrap. The dark conditions were achieved by placing the petri dish into a sealed black polyethylene bag. The soil condition replaced the moist filter paper with moist soil collected from the pine areas of Eyrewell Forest.

Twenty seeds of each species apart from *L. juniperina* (10 seeds) and *C. australis* (5 seeds) were sprinkled into each petri dish and exposed to the conditions outlined above following exposure to one of eleven treatments (Table 3.3). *L. juniperina* seeds were also fed to native birds and lizards at Orana Wildlife Park, Christchurch, but the animals refused to eat them. There were three replicas of each treatment apart from for *P. amoena* where there were five. With the exception of the *L. juniperina* seeds which were smoke-treated and kept in glasshouse conditions, all seeds were placed inside an incubator with a 12 hourly temperature fluctuation of 10°C and 25°C. Petri dishes were checked every 2-4 days with germination incidence recorded and watered as required. The germination tests ran until all the seeds had germinated or for 6 months for all species except *L. juniperina* which ran for 1 year.

To assess the longevity of seeds, *K. serotina* and *P. amoena* seeds which had been stored at 5°C for two months before being kept dry conditions at room temperature for 8 months were also subjected to the germination experiment. Following the results from the initial experiment, *K. serotina* seeds ($n=50$) were placed into petri dishes (three replicates) and into the incubator under light conditions. *P. amoena* seeds ($n=20$) were subjected to three treatments of control, heat treated in boiled water and heat treated at 105°C for 1 h and also placed into the incubator under light conditions.

Table 3.3 Species and treatments involved in the laboratory/glasshouse germination experiment. Treatments carried out on species marked with an X.

Species											
Treatment	<i>K. serotina</i>		<i>L. juniperina</i>			<i>C. australis</i>		<i>P. amoena</i>			
	Light	Dark	Light	Dark	Soil	Light	Dark	Light	Dark	Soil	
Control – placed on filter paper/soil	X	X	X	X	X	X	X	X	X	X	
Hot Water - in boiled water (12 h)			X	X	X			X	X	X	
Warm Water – in c.50°C water (12 h)			X	X	X						
60°C 1hour –1 h at 60°C prior to trial			X	X				X	X		
60°C 1 day – 24h at 60°C prior to trial			X	X				X	X		
105°C 1hour- 1 h at 105°C prior to trial			X	X				X	X		
105°C 1day- 24 h at 105°C prior to trial								X	X		
Buried in 2 cm soil						X					
Smoke treated – held in smoke (3mins)			X	X	X						
Smoke treated & grown with Kānuka					X						
Weta Eaten – flesh eaten by weta			X	X							
Cracked Drupes – endocarp cracked with seed intact prior to experiment			X	X	X						
Exposed seed prior to experiment			X	X							

Data Analysis

Means (\pm SEM) of germination rates within each treatment and condition were calculated for each species and illustrated using SigmaPlot (Version 12.3, Systat Software, San Jose, CA). Differences in germination rates between treatments and conditions were analysed using a one-way ANOVA with post-hoc Tukey (HSD) test. Statistical analysis was undertaken using Minitab® 17.2.1 (Minitab Inc., Sydney, AUS).

3.5 Results

3.5.1 Field Habitat Survey

Plant Community

The kānuka field boundaries and the remnant areas had the highest species richness of 38 and 39 (full species list in Appendix B1). However, the field boundaries had the lowest representation of native plants with only 25% of the total flora being indigenous whilst the remnant areas contained the highest (Table 3.4). The pine forest and field boundary kānuka stands contained more exotic species;

particularly grasses and herbs synonymous with agriculture, open habitats and disturbance, became more frequent.

Table 3.4 All constant species of 60-100% frequency amongst the plots (frequency classes IV-V) in each vegetation type. The plants cover class is denoted in brackets (1-5). Plants in bold type are native species. Also displayed are data illustrating vascular species richness and percentage of native plants (**= $p<0.05$) plus number of plants classified as pioneer, secondary or primary species.

Constant Species	Remnant	Field Boundary	Pine
<i>Agrostis capillaris</i>		V(3-5)	IV(2-4)
<i>Anthoxanthum odoratum</i>	V (2-3)		IV(2-3)
<i>Dichondra repens</i>	IV (1-3)		
<i>Cytisus scoparius</i>		IV(1-4)	
<i>Festuca rubra</i>		V(2-5)	
<i>Hypnum cupressiforme</i>	V (3-6)	IV(2-4)	
<i>Hypochaeris radicata</i>	IV (1-3)	IV(2-3)	
<i>Kunzea serotina</i>	V (3-6)	V(3-4)	
<i>Leptecophylla juniperina</i>	V (1-4)		
<i>Lolium perenne</i>		IV(2-4)	
<i>Pinus radiata</i>			V(2-5)
Vascular species Richness **	38 ^{ab}	39 ^a	25 ^b
Mean % Native Vascular Species **	53 ^a	25 ^b	40 ^b
Number of vascular pioneer species (% native)	24 (25%)	30 (10%)	19 (26%)
Number of vascular secondary species (% native)	12 (100%)	8 (87.5%)	5 (100%)
Number of vascular Primary species (% native)	2 (100%)	1 (100%)	1 (100%)

The individual parameters of non-vascular plant, canopy and litter cover within each habitat type differed. The remnant areas had the highest cover of non-vascular plants and canopy cover. The litter cover was highest within the pine forest due to an almost continuous covering of pine needles (Figure 3.3).

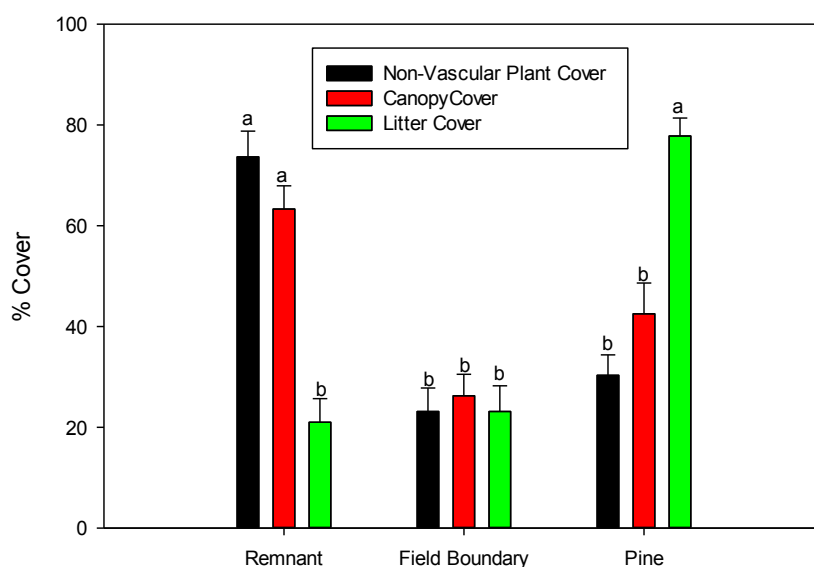


Figure 3.3 Mean values (±SEM) for percentage cover of moss, litter and canopy within each kānuka habitat. Means that do not share a letter are significantly different ($p<0.05$).

Seedling regeneration of native species varied in each habitat with *L. juniperina* found almost exclusively within the remnant vegetation habitats ($p < 0.05$). *K. serotina* and *P. amoena* seedlings were found more frequently in the kānuka field boundary and pine plantation habitats ($p = 0.008$ and $p = 0.029$ respectively, Figure 3.4). *K. serotina* was not significantly influenced by non-vascular plant, canopy or litter cover. *P. amoena* was more frequent with increased litter cover ($p = 0.008$) and *L. juniperina* was found more frequently with non-vascular plant cover ($p = 0.033$).

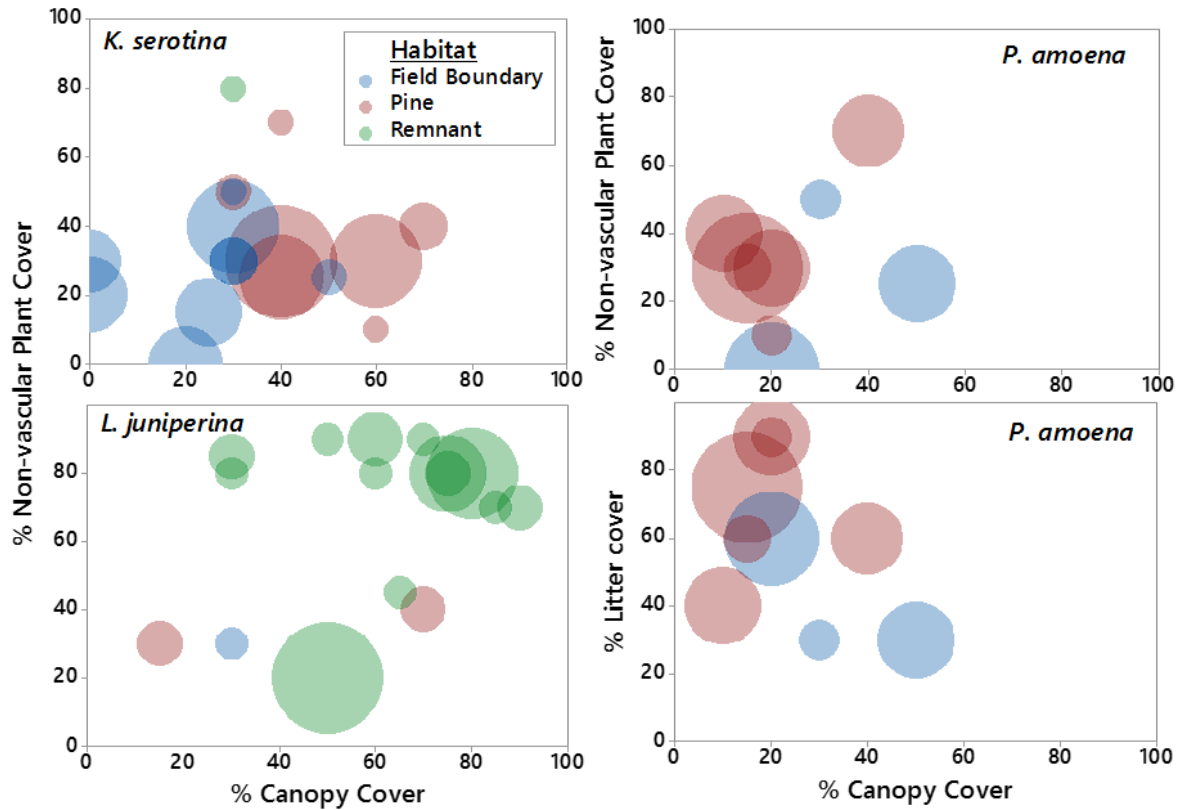


Figure 3.4 Presence and number of seedlings, shown by the size of each bubble, against the percentage cover of non-vascular plant ground cover and canopy in each habitat (colour-coded).

Principal component analysis of the parameters directly influenced by biota revealed a difference in the ordination space occupied by each habitat (Figure 3.5). The remnant habitat was weighted towards the negative end of axis one influenced by the number of *L. juniperina* seedlings, percentage of native species, moss cover and canopy cover. The pine and field boundary habitats were further distinguished by axis two influenced by litter cover and species richness. Axis one explained 32.5% of the variance whilst axis two explained 18%.

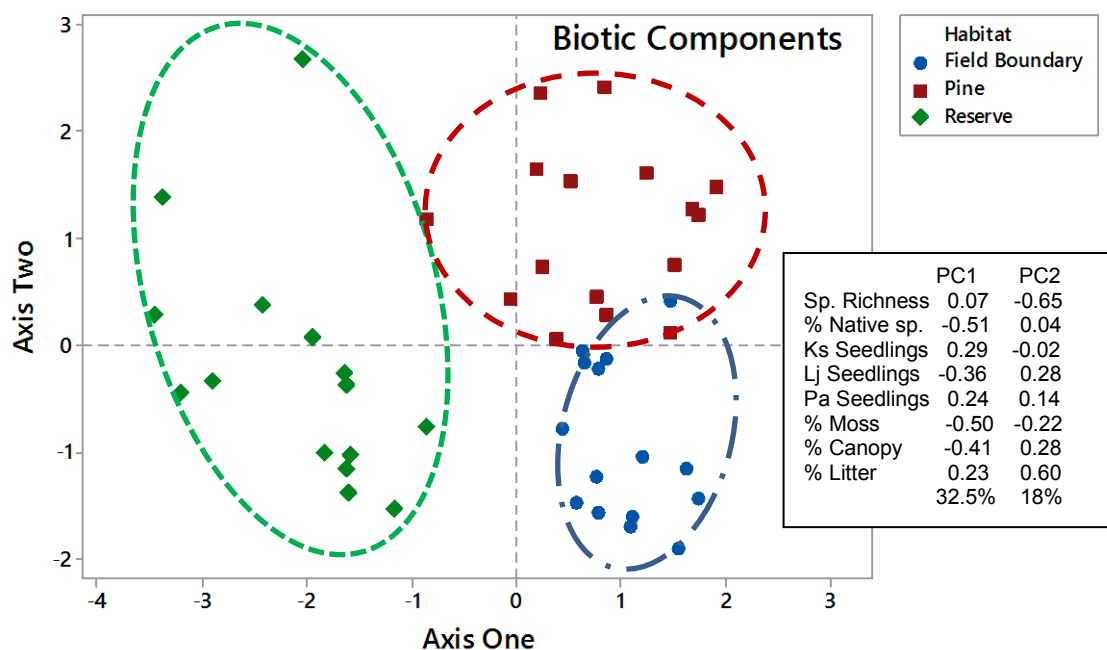


Figure 3.5 PCA of biotic components for each habitat type. The points represent each survey point in each habitat replicate and the relative loadings are tabulated adjacent to the figure. The seedlings are annotated as such: Ks= *Kunzea serotina*, Lj= *Leptecophylla juniperina*, Pa= *Pomaderris amoena* and the ellipses group the habitats by their relative loadings.

Soil Data

In topsoil, the remnant habitat areas had the least nutrient rich substrate with lowest pH, lowest nitrate and lowest available phosphorus (Figure 3.6). The remnant habitats also had the highest total organic carbon (TOC) and total nitrogen. Conversely, the grassland had the highest pH, nitrate and available phosphorus. The pine habitat was characterised by high carbon: nitrogen ratio. Gravimetric soil moisture and ammonium concentrations were not significantly different between habitats within both topsoil and subsoil.

There were fewer differences between habitats for the subsoil but results reflected those of the topsoil. The grassland had the highest concentration of nitrate and the highest pH. The carbon: nitrogen ratio was also highest in the pine habitat (Figure 3.7).

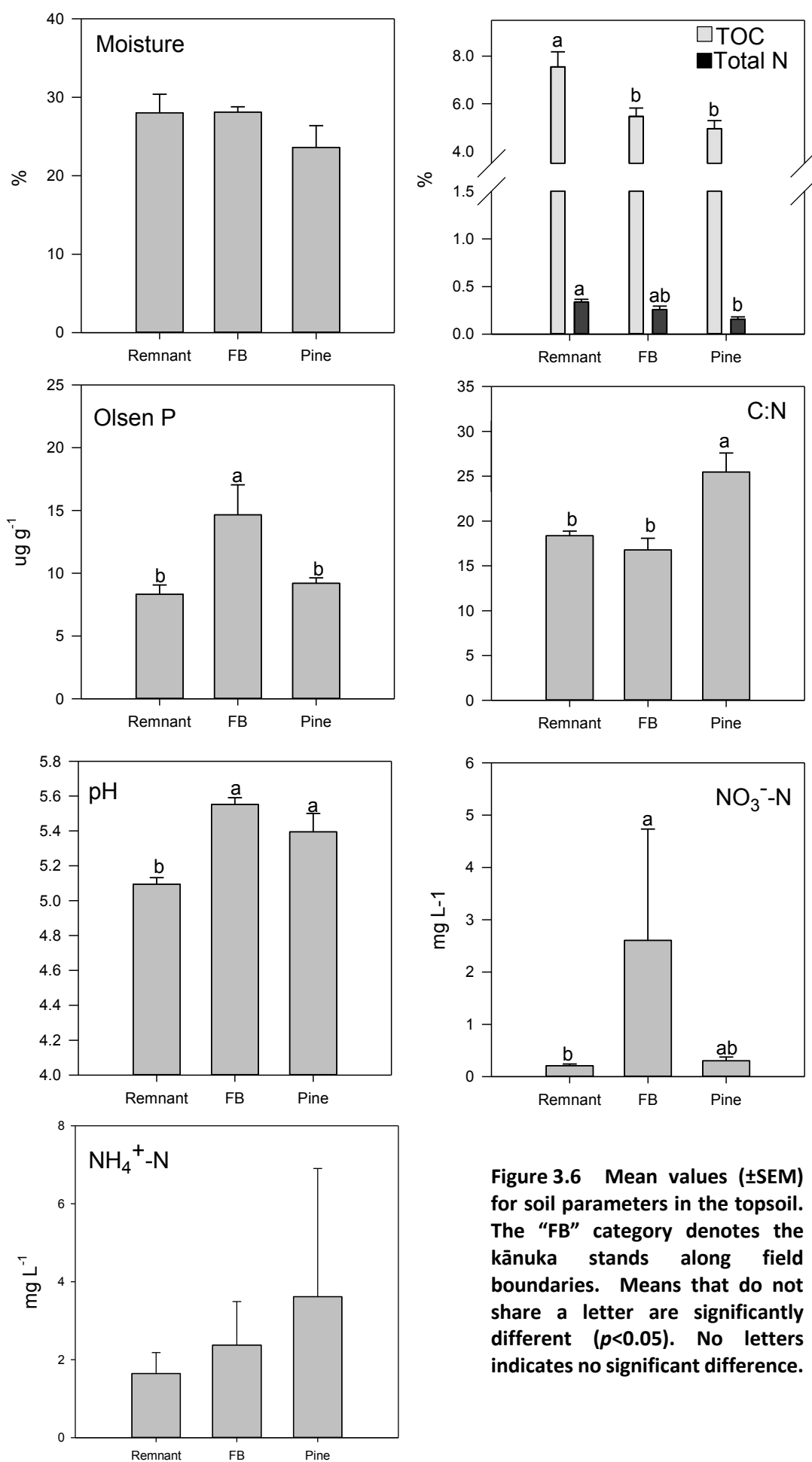


Figure 3.6 Mean values (\pm SEM) for soil parameters in the topsoil. The “FB” category denotes the kānuka stands along field boundaries. Means that do not share a letter are significantly different ($p < 0.05$). No letters indicates no significant difference.

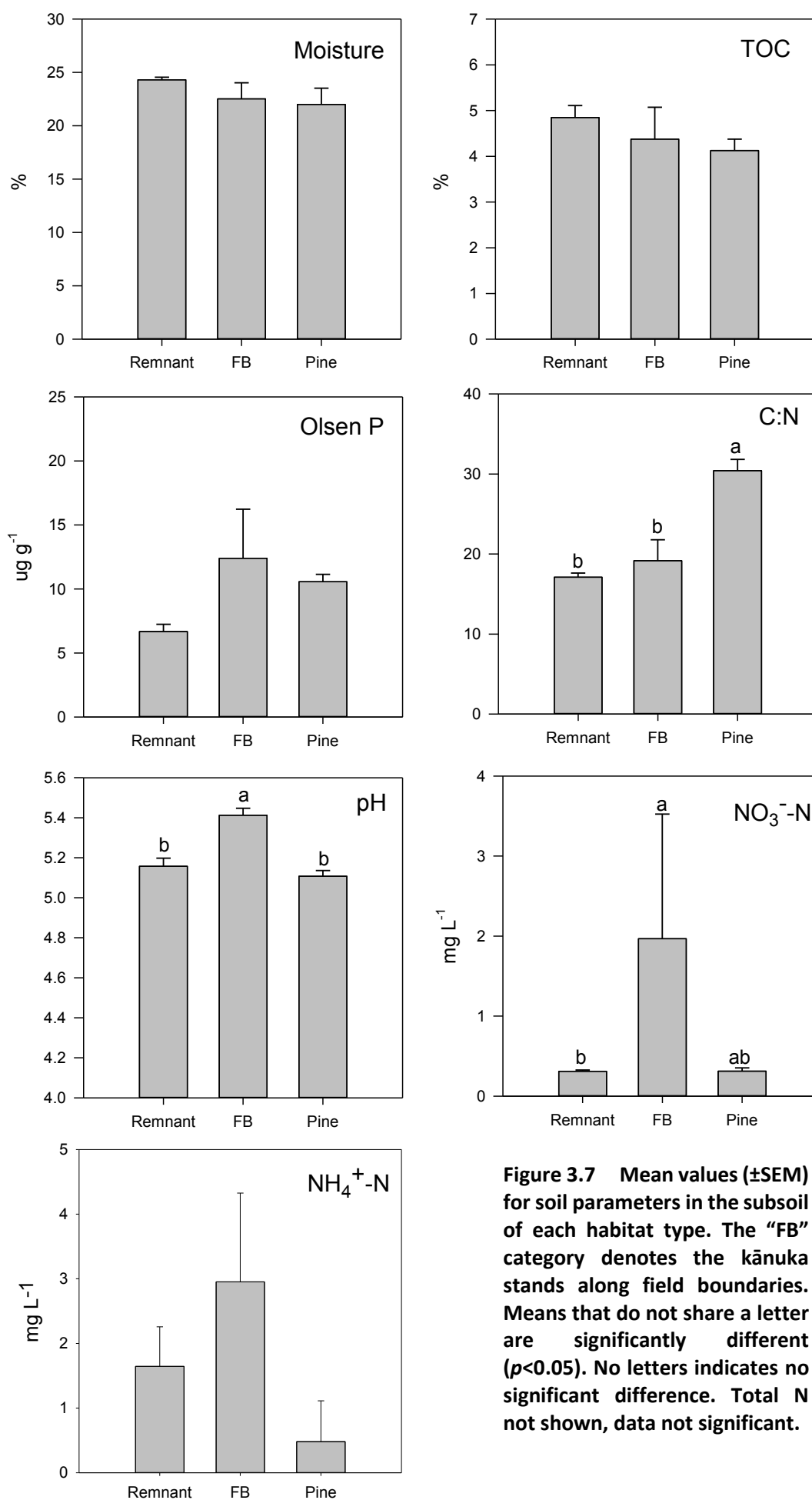


Figure 3.7 Mean values (±SEM) for soil parameters in the subsoil of each habitat type. The “FB” category denotes the kānuka stands along field boundaries. Means that do not share a letter are significantly different ($p < 0.05$). No letters indicates no significant difference. Total N not shown, data not significant.

Multivariate analysis of soil data revealed differences between habitats, each occupying a different ordination space (Figure 3.8). For topsoil, axis one explained 37% of the variation whilst axis two accounted for 32%, together accounting for over 69% total variance. The remnant habitats were separated from the field boundary habitats along axis one which was influenced by nitrate and Olsen P. The remnant habitats were also separated from the pine habitats along axis two, which was weighted for total organic carbon, moisture and carbon: nitrogen ratio.

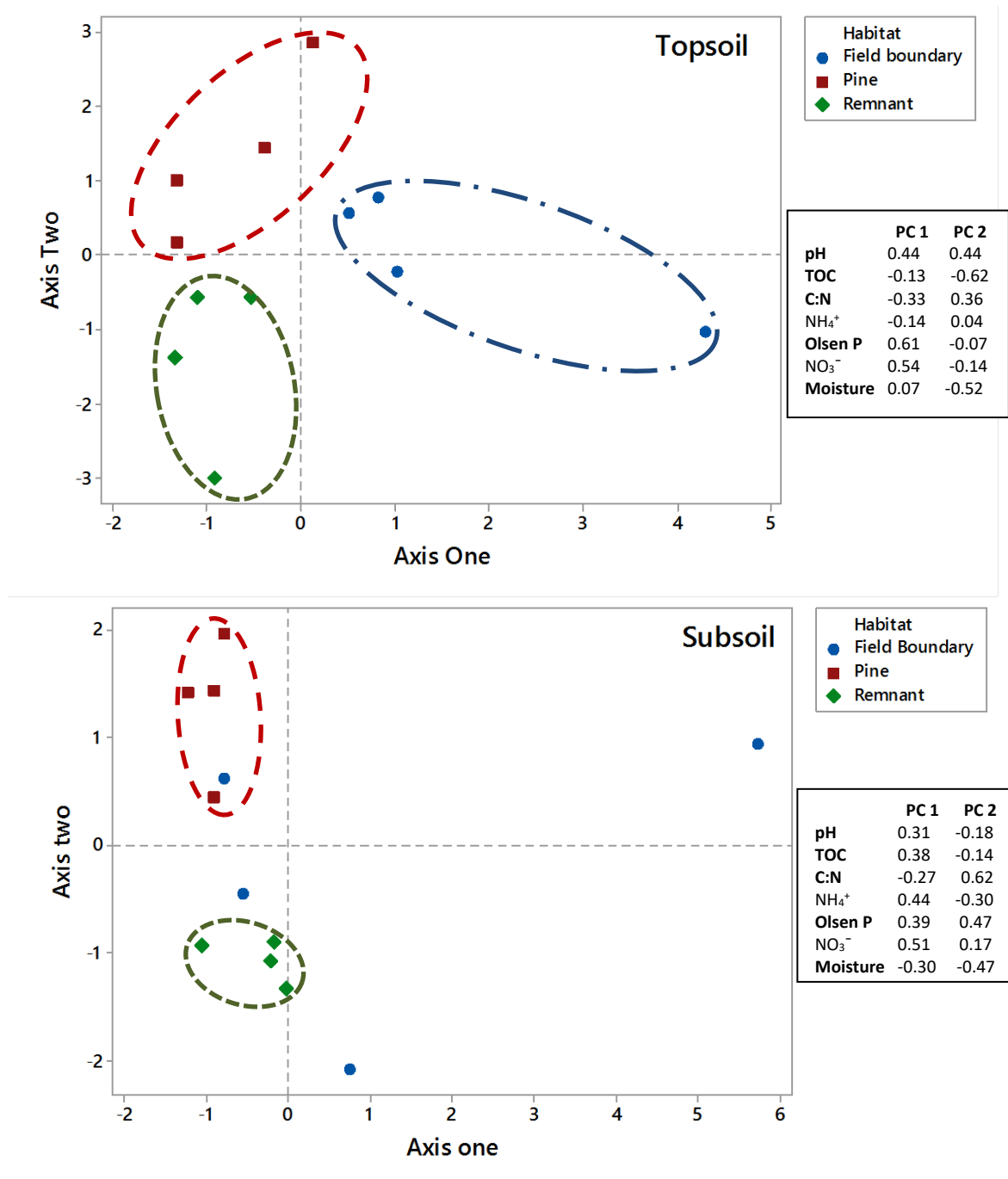


Figure 3.8 PCA of soil characteristics in both topsoil and subsoil. The points represent each habitat replicate and the relative loadings are tabulated adjacent to the figure. Ellipses group the habitats by their relative loadings.

The subsoil analysis also revealed a similar pattern with the remnant and pine habitats occupying a particular space along axis one but separated along axis two. The field boundary habitats were more variable and therefore less defined. Axis one accounted for 51% of variance whilst axis two accounted for 24%. Axis one was positively weighted for pH, total organic carbon, NH_4^+ and NO_3^- whilst axis two was weighted for C:N ratio and Olsen P.

3.5.2 Individual Species Investigation

None of the *L. juniperina* seeds germinated within 1 year of being exposed to any of the treatments. *K. serotina* seeds only germinated in light conditions, with a germination rate of 63% for fresh seed and 56% for 1 year old seed (not significant). Time to germination was similar for both sets of seeds, fresh seed germinating after 15 days and year-old seed after 14 days. *C. australis* was the fastest species to reach germination, at 12 days. There was no significant difference between the treatments, although in one of the dark treatment replicates, 100% of seeds germinated (Figure 3.9).

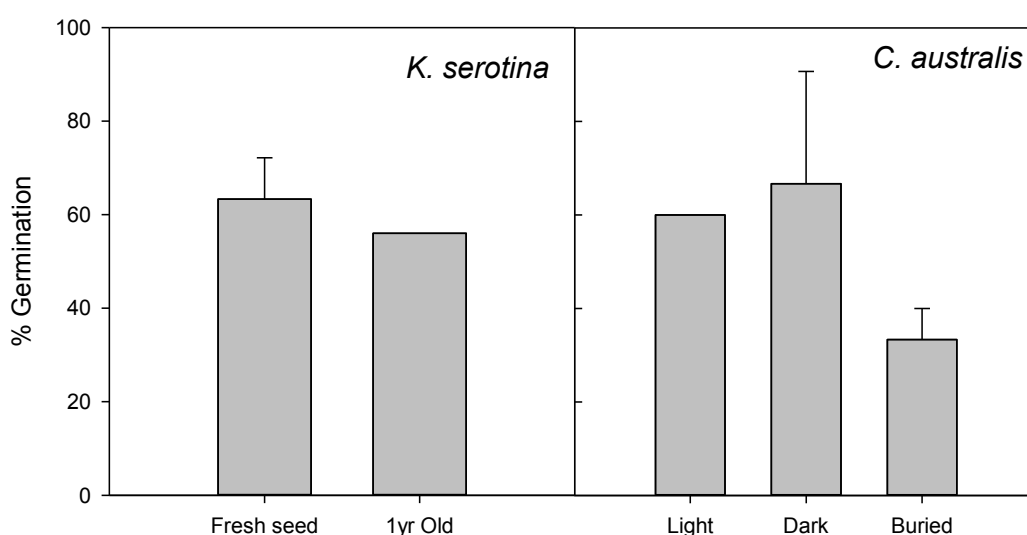


Figure 3.9 Mean (±SEM) germination rates for *K. serotina* seeds which had been kept in light conditions and sown as fresh seed (stored at 5°C for one month) or 1yr old seed (stored at 5°C for 12 months prior to the experiment). On right, mean (±SEM) germination rates for *C. australis* seeds which had been kept in light or dark conditions or buried under 2 cm soil. Results are not significantly different.

The first fresh *P. amoena* seed to germinate was within the boiled water treatment after 17 days. For the 1 year old seeds this was extended to 53 days and the first seed to germinate was within the 105°C for 1 h treatment. No seeds germinated within the dark conditions or within the warm water, 60°C for 1 h or 1 day and the 105°C for 1 day treatment. Fresh seed appeared to require heat treatment to germinate whilst there was no significant difference between germination rates in the older seeds. Fresh seed produced an approximate 20% increase in germination than the older seeds (Figure 3.10).

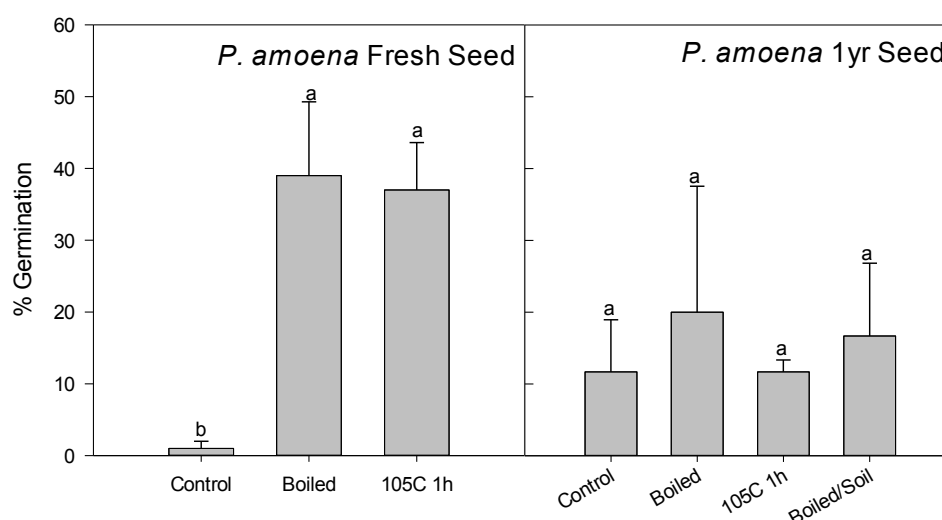


Figure 3.10 Mean germination (\pm SEM) rates for fresh *P. amoena* seeds which had been kept in light conditions and subjected to three treatments (control, placed in boiled water for 12 h or subjected to 105°C for 1 h and placed on moist filter paper too germinate). Seed which had been stored at 5°C for one year was also subjected to the treatments with the addition of seeds which had been subjected to 105°C for 1 h and sown on soil. Means that share a letter are not significantly different ($p=0.05$).

3.6 Discussion

3.6.1 Soil Parameters

Soil nutrients were highest in the field boundary habitats that adjoined irrigated dairy pasture, displaying evidence of nutrient spillover. The dairy pasture had been subjected to fertiliser additions of superphosphate at rates of between 300-500 kg ha⁻¹ and lime at 3000-6000 kg ha⁻¹ in the initial conversion with subsequent soil tests advising future fertiliser use to maintain pasture health (Ngāi Tahu Farming Ltd, pers. comm.).

Olsen P concentrations were at least 5 mg kg⁻¹ higher in the field boundaries than the other habitats surveyed. Phosphorus is known to be carried in surface runoff, mainly in particulate form, from pasture with elevated phosphorous status (Gillingham and Thorrold 2000; Nash and Murdoch 1997; Pierzynski and McDowell 2005) and runoff into the field boundaries is likely. However, the 15 mg kg⁻¹ concentrations were still below the recommended threshold for agricultural pasture of 20-30 mg kg⁻¹ (Roberts and Morton 1999). Similarly, nitrate levels were highest in the field boundaries but only by 2.3 mg kg⁻¹. Water soluble nitrate is usually transported out of the pasture system by percolating through the stony soil profile although there also can be surface runoff, especially when levels are high (Hubbard et al. 1989). At a similar site nearby, Bowie et al. (2016) found that although water flow from pasture into a native habitat reserve was evident there was no marked difference in nitrate levels suggesting the nitrate had been leached through the very porous Eyre soil. There is also

the possibility of introduction of nutrients at the time of spreading. The spinning disc spreader which distributes lime and superphosphate can often misplace the fertiliser with substantial amounts being found within the adjacent vegetation (Rew et al. 1992; Tsiouris and Marshall 1998).

The soils of the pine and remnant sites had a lower nutritional status, reflecting the natural low fertility of the Lismore soil (Kear et al. 1967). The pine habitats had a significantly higher C:N ratio which reflects previous accounts of plantation radiata pine forest with decreased nitrogen and organic carbon leading the higher C:N ratios than native areas (Turner and Lambert 1988). This would seem to be the case in Eyrewell Forest with the significantly higher levels of TOC in remnant habitats possibly contributing to the lower pH in topsoil (Ritchie and Dolling 1985).

The lowest pH was recorded within the remnant areas which may be limiting availability of macronutrients (Figure 3.11). One anion which is not limited at low pH is aluminium. In acidic soils aluminium (Al^{3+}) is released from clay particles into the soil solution available uptake by plants (McLaren and Cameron 1996). Independent soil tests carried out on the soils at Eyrewell have shown Al levels to exceed the $<3 \text{ mg kg}^{-1}$ recommended limit and reach levels of 26 mg kg^{-1} in some areas (Appendix B.2) and soil tests from 2008 show the pH within the Eyrewell Scientific reserve to be 4.6 (DOC, pers. comm.). Aluminium can be highly toxic to plants affecting growth, particularly of roots (Delhaize and Ryan 1995; Matsumoto 2000). There are a number of plants which can tolerate aluminium rich soils and do so via a range of mechanisms including producing root exudates which chelate aluminium or detoxifying the aluminium within cells by forming organic acid complexes (Brunner and Sperisen 2013; Ma et al. 2014; Ma et al. 2001).

An interesting discovery was that the moisture levels between the three habitat sites were not statistically different despite the grassland field boundaries being subjected to irrigation. The mean percentage moisture was approximately 5% lower in the pine habitat but almost identical between the remnant and field boundary habitats. The Lismore soils of the area are well drained with a low water holding capacity found within Class A of the hydrological soil grouping (Landcare Research 2011; Woodward et al. 2001). Although these soils are free draining the influence of the pivot irrigators adjacent to the field boundaries was not discernible. A possible reason could be the high percentage of moss ground cover found within the remnant habitats. Moss is known to support higher soil moisture levels in other climatic regions and vegetation (Belnap 2006; Xiao et al. 2016).

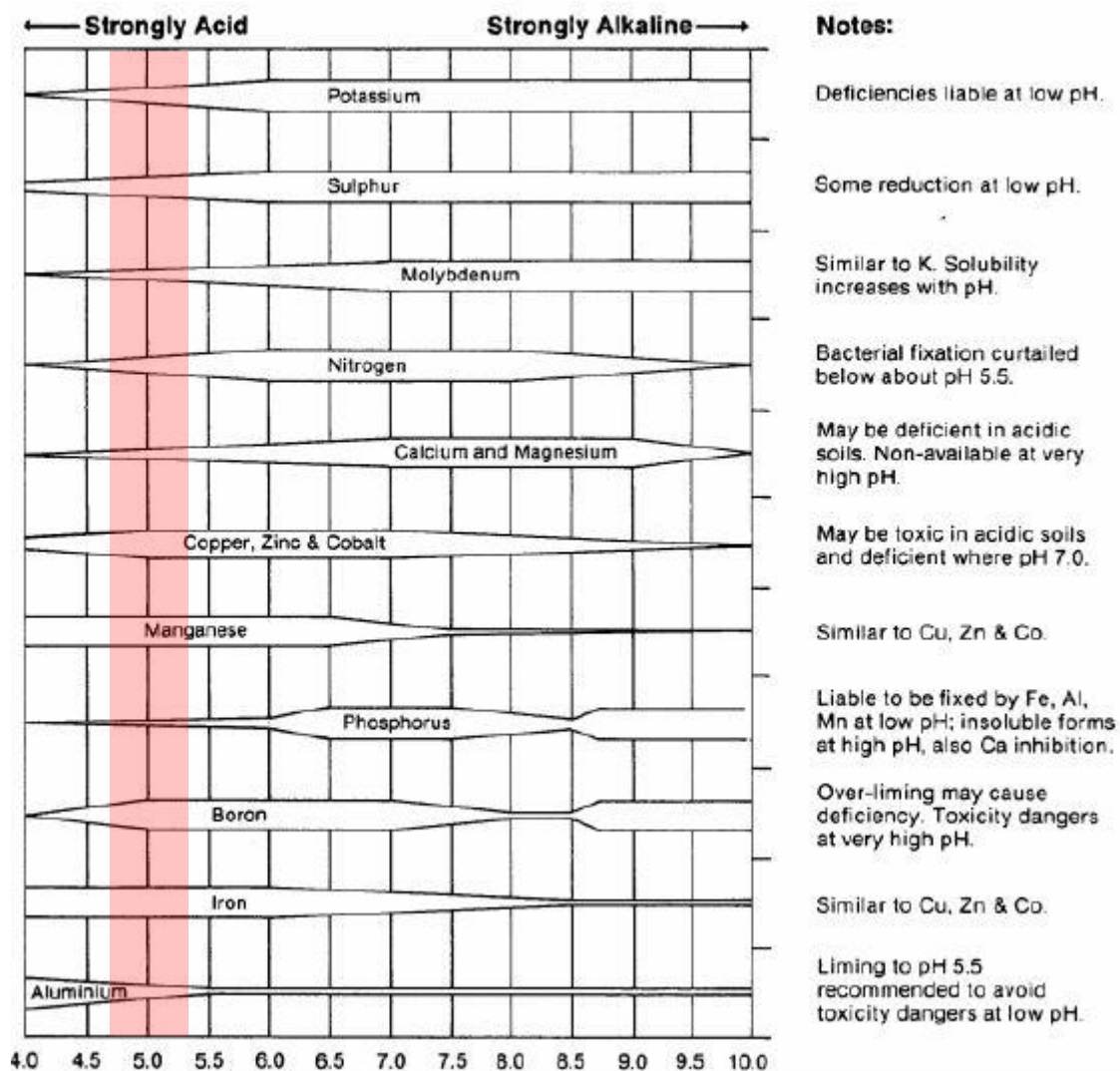


Figure 3.11 The effect of pH on the availability of nutrients in soil (Truog 1948). The shaded area indicates the pH range of the remnant areas.

3.6.2 Individual Species

The germination study found that *K. serotina* seeds germinate readily in light conditions and if kept dry and cool, viability can persist for at least a year, retaining similar germination rates to fresh seed. These results correlate with previous research pertaining to kānuka seeds being able to withstand an amount of drying and seed bank storage, from a few months to 15 months (Burrows 1997; Sem and Enright 1996). However, it is generally accepted that most New Zealand species have limited seed banks and most kānuka seedlings regenerate from fresh seed rain rather than seed bank sources (Bray et al. 1999; Standish et al. 2008). The seedlings of kānuka were found mainly in the pine and field boundary sites that had the most disturbance and reduced canopy cover leading to higher light levels. This supports the gap-phase strategy theory as disturbance and open ground was limited in the remnant habitats leading to fewer opportunities for regeneration (Allen et al. 1992; Kirschbaum and Williams 1991).

P. amoena was also found within the pine and field boundary sites and did not germinate in dark conditions, revealing it to be a pioneer species with a preference for light conditions. The rate of germination was less than 50% and appeared to be encouraged by heat treatments in the fresh seed with capacity for germination being reduced more than fourfold when not heat treated. The use of heat treatment for breaking dormancy of this species was further confirmed as seedlings were found frequently in one field boundary location where the adjacent pine forest had been burnt just two years previous to the study (Ngāi Tahu Farming Ltd., pers. comm.). The *Pomaderris* genus includes many species which germinate following fires (Cunningham and Cremer 1965; Ooi et al. 2014). Both Warcup (1980a) and Haines et al. (2007) found an increase in germination for *Pomaderris* sp. when subjected to heat treatment, although, different species are likely to have different heating and scarification requirements to break dormancy (Haines et al. 2007).

C. australis seeds germinated in both light and dark conditions, even when buried in 2 cm of soil. This species is considered to be a pioneer species due to its ability to tolerate full light, but it would appear that it can also tolerate shade. It was only recorded within the remnant areas but no seedlings were found in the field. Grüner and Heenan (2001) found that propagules were long-lived (24 years) and were viable following scarification such as heat treatment of boiling water.

The study failed to germinate any *L. juniperina* seeds. The DOC nursery at Mokukarara have experienced similar problems (Nicky Robb, pers. comm.). There is a lack of knowledge of germination requirements of this species although plants within the same genus are known to respond well to extended periods of warming (Baskin et al. 2005). The plant is also known to be eaten by some birds now not present upon the Canterbury Plains which may be an important feature of dormancy breaking (Horrocks et al. 2008). Nonetheless, seedlings were reported, found mostly in the remnant habitat, suggesting the plants prefer shade conditions in damp, acidic areas.

3.6.3 Community Composition

None of the plant species found were listed under any category other than “Not Threatened” within the Threatened Plant List (de Lange et al. 2013). The highest numbers of native species were found within the remnant areas which had the lowest soil nutrient status. Nutrient addition has been found to decrease species richness in a number of habitats worldwide and greatest diversity is thought to be found in intermediate levels of nutrients (Clark and Tilman 2008; Goldberg and Miller 1990; Tilman 1984). The total number of species recorded within each habitat was much less than found by Ecroyd and Bockerhoff (2005) who detailed a total of 118 species with 60% being adventive. The total in the present research was 38 species (47% exotic). This may be explained by surveying on only one day

rather than throughout the year; it is probable that a large number of native species were missed, particularly annual species or herbaceous species. Table 3.5 displays the native plants recorded by Ecroyd and Brockerhoff (2005) along with their life-form. Interestingly of the 47 species, 20 are herbaceous plants and a further 19 are small stature grasses, sedges, ferns or orchids which have a shallower rooting system than trees and shrubs. This is especially noteworthy given the dry environment (Schenk and Jackson 2002).

Table 3.5 List of native species and their life-form found by Ecroyd and Brockerhoff (2005) in the Eyrewell Scientific Reserve in 2003.

Species		Species	
Forbs		Shrubs	
<i>Brachyglottis bellodioides</i>	Forb	<i>Carmichaelia australis</i>	Shrub
<i>Celmisia gracilentia</i>	Forb	<i>Coprosma crassifolia</i>	Shrub
<i>Clematis quadribacteolata</i>	Forb	<i>Coprosma intertexta</i>	Shrub
<i>Dichondra repens</i>	Forb	<i>Coprosma propinqua</i>	Shrub
<i>Epilobium alsinoides</i> ssp. <i>atriplicifolium</i>	Forb	<i>Discaria toumatou</i>	Shrub
<i>Epilobium cinereum</i>	Forb	<i>Kunzea serotina</i>	Shrub/tree
<i>Euchiton audax</i>	Forb	<i>Leptecophylla juniperina</i>	Shrub
<i>Galium propinquum</i>	Forb	<i>Melicytus alpinus</i>	Shrub
<i>Gallium microphyllum</i>	Forb	<i>Pomaderris amoena</i>	Shrub
<i>Hydrocotyle moschata</i>	Forb		
<i>Hypericum gramineum</i>	Forb	Monocotyledons	
<i>Lagenophora strangulate</i>	Forb	<i>Carex breviculmis</i>	Sedge
<i>Leptinella pusilla</i>	Forb	<i>Carex goyenii</i>	Sedge
<i>Leucopogon fraseri</i>	Forb	<i>Deyeuxia avenoides</i>	Grass
<i>Nertera setulosa</i>	Forb	<i>Dichelachne crintia</i>	Grass
<i>Oxalis exilis</i>	Forb	<i>Festuca novae-zelandiae</i>	Grass
<i>Senecio glomeratus</i>	Forb	<i>Luzula rufa</i> var. <i>rufa</i>	Sedge
<i>Senecio minimus</i>	Forb	<i>Microlaena stipoides</i>	Grass
<i>Solanum laciniatum</i>	Forb	<i>Microstis unifolia</i>	Orchid
<i>Stackhousia minima</i>	Forb	<i>Poa cita</i>	Grass
<i>Wahlenbergia violacea</i>	Forb	<i>Poa pusilla</i>	Grass
Ferns		<i>Rhytidosperma gracile</i>	Grass
<i>Asplenium flabellifolium</i>	Ferns	<i>Rytidosperma unarede</i>	Grass
<i>Hypolepis ambigua</i>	Fern	<i>Thelymitra longifolia</i>	Orchid
<i>Microsorium pustulatum</i>	Fern	<i>Thelymitra pauciflora</i>	Orchid

In this study, the remnants contained the most native species and, although native species only accounted for 25% of pioneer species, all secondary and primary species were native (Table 3.4). The pioneer species were mainly exotic, with sweet vernal grass (*Anthoxanthum odoratum*) and cats ear (*Hypochaeris radicata*) being prominent components. Ecroyd and Brockerhoff (2005) also found these species, plus *Crepis capillaris* and *Agrostis capillaris*, to be of concern as they replaced the indigenous grassland and shrubland understory. These two exotic grass species also featured heavily in the pine habitats which were dominated by exotic species. The pine areas contained the least species but also had 100% native secondary and primary species present, similar to the remnant areas.

In contrast, the grassland habitats were species rich but contained a majority of non-native (80%) and pioneer species. These habitats were long narrow strips of land surrounded by the exotic grassland pasture which had experienced recent disturbance due to the conversion of adjacent land with high nutrient inputs. Obviously, an important aspect relating to seedling occurrence is the proximity of seed producing plants (Hill and Read 1984; Moles and Drake 1999; Williams and Karl 2002). Native plant diversity was limited within the field boundary and pine habitats, with limited propagule availability opportunities.

In many situations kānuka is viewed as a secondary species, often present as a seral stage which succeeds and outcompetes a dominant mānuka community before development of podocarp forest (Allen et al. 1992; Bergin et al. 1995; Esler 1967; Esler and Astridge 1974). Smale et al. (1997) reported that soil depth under kānuka forest in the East Cape increased from 20 cm in year 10 to approximately 58 cm by year 70 showing good carbon sequestration and facilitating the regeneration of primary tree species. However, in more dynamic habitats, such as sand dunes, kānuka communities may be dominant due to isolation from primary species propagules (Smale 1994). The kānuka community within Eyrewell forest does not have the instability of substrate like the sand dune but the poor soils are thin and droughty (Kear et al. 1967). Added to the fact that propagules from other native species are scarce, this has led to the suggestion that kānuka shrubland has been the naturally dominant vegetation here for at least for approximately 200 years (Meurk and Hall 2006). Therefore, kānuka has been given a primary species status in this study in spite of the characteristics of the plant, such as seed size, seed dispersal mechanism and seedling regeneration in gaps (Swaine and Whitmore 1988).

3.6.4 Community Composition Influences

Allen et al. (2003a) described six influences on the structure and composition of forests in New Zealand: disturbance, herbivory, species effects, climate and soil, dispersal and time. Combinations of these factors are the probable cause for the difference in community assemblage across the three habitat types. Below is a brief description of each influence, with the exception of time, which is the overarching component in the ecosystem development.

Soil Nutrition and Climate

The soils in each habitat were of the same descriptive classification (Lismore), but nutrient enrichment from land-use practices had undoubtedly altered the chemical composition of field boundaries. Invasion of exotic species into disturbed and nutrient enriched areas has been noted in a number of habitats throughout the world (Hobbs and Huenneke 1996; Lake and Leishman 2004; McIntyre and Lavorel 1994). Invasive species tend to have traits of rapid acquisition of resources (Blackshaw et al. 2004; Peltzer et al. 2016) and can become dominant components of communities, often outcompeting

the native species (Caplan et al. 2017). Nutrient spillover from neighbouring land can increase productivity of some plant species but decrease the overall species diversity over time due to shifts in habitat composition (Kleijn and Snoeiijing 1997; Schmitz et al. 2014), perhaps similar to the field boundary habitats in the present study.

Dispersal

Dispersal opportunities are dependent upon available propagules. (Zefferman et al. 2015) noted that abiotically limited environments can lead to a reduction in invasion by exotic species, which then do not provide as many new propagules. There are limited supplies of propagules of native species on the Canterbury Plains leading to a lack of dispersal opportunities. The majority of land contains exotic species providing a wider opportunity to encroach (Meurk and Swaffield 2000).

Disturbance and Herbivory

Nutrient enrichment and disturbance can often lead to increases in exotic plant invasion (Lake and Leishman 2004). It is known that at least one remnant location (within ESR) has been fenced from cattle and other disturbances for at least 45 years. Two other remnant sites (SBR) were fenced with an electric hotwire which was intermittently effective at preventing cattle from entering. However, it is assumed that disturbance within these areas were lower than the other two habitat types which were influenced by disturbances from the plantation and farming work of adjacent land. In some ecosystems the interplay intermediate disturbance can provide a variety of niches and maintain biodiversity (Collins and Glenn 1997; Grime 1973; Roxburgh et al. 2004), but the relationship is not straight forward (Mackey and Currie 2001). In Europe, whilst increased disturbance levels can cause biodiversity loss, so too can the lack of disturbance and management (Jansen et al. 2009; Pärtel et al. 1999; Reitalu et al. 2014). Alpert et al. (2000) describes the relationship of abiotic stress and disturbance on a community illustrating that when stress low and disturbance is either too high or too low invasions can typically occur (Figure 3.12). However, in systems which have evolved without or with little disturbance minimal disturbance may promote species richness. A study in New Zealand found that disturbance was a prerequisite for four invading exotic weed species (Jesson et al. 2000) and in Australia, McIntyre and Lavorel (1994) found that a decrease in grazing pressure increased diversity. In New Zealand, the extensive degree of isolation from successfully competitive fauna and flora which has evolved overseas and the lack of mammals prior to human settlement makes it particularly susceptible to disturbance and invasion (Meurk and Hall 2006; Meurk et al. 2016). In New Zealand it is thought that, historically, disturbance would have been local and generally low in intensity; in Canterbury the main forms may have been infrequent fires by lightning strike and the action of large footed birds scarifying the ground for food (Rogers et al. 2005).

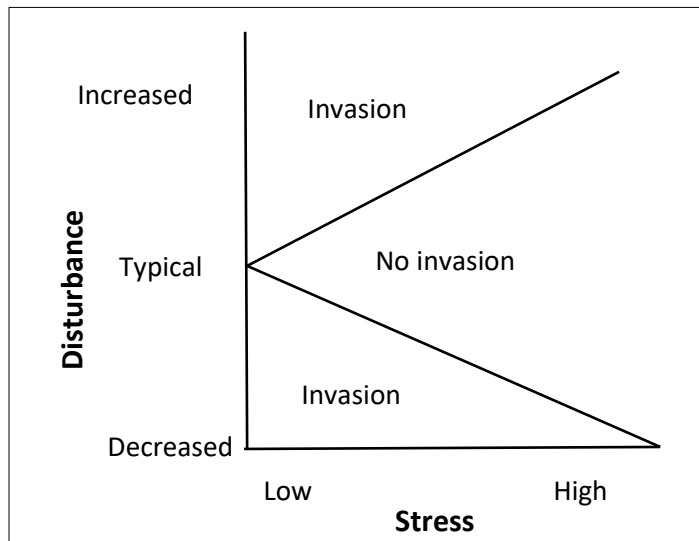


Figure 3.12 Adapted model for invasibility and the role of stress (e.g. nutrient availability in this case) and disturbance (e.g. ungulate grazing and trampling) from Alpert et al. (2000).

Species Effects

Once exotic species have encroached into a habitat they can often change community composition by altering nutrient cycling processes which in turn can be further detrimental to native species (Ehrenfeld 2003; Putten et al. 2013). For example, with the exception of nitrogen-fixing plants, litter from perennial ryegrass (*Lolium perenne*) and other turf-forming grasses may contain higher nutrient concentrations than native species litter (Dickinson 1984; Dickinson et al. 2015; Zhong et al. 2016). The litter from exotic species often decomposes more rapidly than native species (Ehrenfeld et al. 2001) and positive feedbacks from plant-litter to soil can be found (Hobbie 2015). Conversely, litter from New Zealand native plants was shown to reduce soil nitrate by 85% (Zhong et al. 2016). It is generally agreed that most New Zealand plants have adapted to low nutrient environments (Wardle 1991), therefore positive feedbacks, increasing soil nutrition could benefit exotic pioneer species rather than indigenous plants.

3.6.5 Conclusions and Restoration Implications

This study indicates that in order to restore kānuka shrubland a number of factors which have led to the decrease in native biodiversity of the field boundary and pine habitats need to be addresses:

- I. It is important that native propagule sources are introduced to the area (Standish et al. 2008).
- II. It may be necessary to mitigate for past and future nutrient enrichment from neighbouring land-use through processes such as leaching, runoff and windblown chemicals. It is possible that the edaphic changes in field boundary habitats have driven the increased number of exotic species. It is known that kānuka can tolerate a range of conditions and can grow superiorly to

woody exotic species (Cieraad et al. 2015a), but it does not regenerate successfully within long grass swards (Allen et al. 1992). Therefore, invasion of more generalist species such as non-native grasses can not only compete for space and cover within the habitat but also produce high biomass and N availability producing a cumulative effect of out-competing native species (Ehrenfeld 2003; Eliason and Allen 1997; Morgan 1998).

- III. The role of disturbance, especially when planting into a remnant patch or newly created reserve, should be considered and disturbance kept to a minimum to reduce invasions. In this study seedlings of two species investigated appeared to require some disturbance for regeneration under the kānuka canopy due to only germinating in light conditions; kānuka and pomaderris (Plate 3.8). This supports the theory that small-scale disturbance, such as that historically offered by foraging bird species, is required to retain the heterogeneity of native habitats (Hobbs and Huenneke 1996; Rogers et al. 2005). In addition, both species respond to fire, with *K. serotina* being serotinous and heat stimulating germination in *P. amoena*.

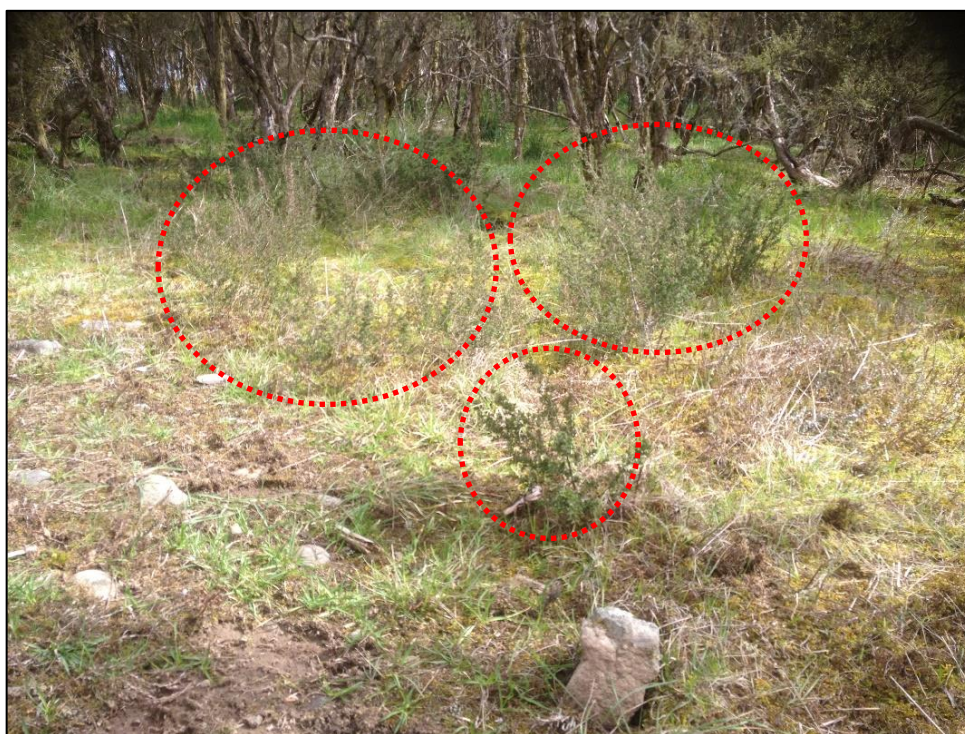


Plate 3.8. *Pomaderris amoena* plants (within red circles) found in the Eyrewell Scientific Reserve growing in an area of low canopy and disturbance, presumably by birds and hedgerows, to the ground layer revealing bare soil.

3.6.6 Onward Project Development

The most striking differences between the remnant and field boundary habitats, despite the number of native species, were the low soil nutrient concentration, high soil TOC and the high cover of canopy

and moss ground cover. In addition, the soil moisture content, although not significant, was higher than the field boundaries which were subjected to irrigation from neighbouring pasture. As discussed, it is likely that soil nutrient enrichment was partly a consequence of nutrient spillover from the adjacent farmland. However, non-vascular plants have significant influence on nutrient and hydrological cycles in boreal forests and New Zealand temperate rainforests (Bonan and Shugart 1989; DeLucia et al. 2003; Nilsson and Wardle 2005) and can be effective at sequestering carbon (Bona et al. 2013; Harden et al. 1997). The remnant habitats contained over 70% non-vascular plant cover and therefore the ecological role of this ground cover layer with regard to hydrology and soil chemistry within dryland kānuka shrubland deserves further investigation.

Furthermore, many of the plants found within the remnant areas were small, stature herb species (Table 3.5, Plate 3.9) with shallow roots. These species may depend upon the potential moisture and soil temperature buffering services provided by the moss cover (Xiao et al. 2016). Further research is required to establish the seedling requirements of the main community associates including in-depth investigation of the function of the moss layer with regards to plant interactions to determine whether it is an important component in initial stages of restoration.



Plate 3.9. Examples of plants growing in the moss layer at the Eyrewell Scientific Reserve. *Leucopogon fraserii* within *Racomitrium lanuginosum* (left), *Leptecophylla juniperina* seedling within *Thuidium furfurosum* (centre) and *Leptinella pusilla* within *H. cupressiforme* (right).

The soils of the remnant habitat were moderately acidic and allowed for Al^{3+} to be found in the soil solution for plant uptake. Without modification of soils the plants within the remnant areas are surviving in potentially toxic soils. Could it be possible that some of the main species found within the remnant areas are tolerant of Al^{3+} ? Two of the defining plant species within the remnant area could possibly have Al^{3+} tolerance; a kānuka species in Australia was found to be highly tolerant of aluminium in soils (Mikli 2001) and sweet vernal grass in some parts of the world has also been shown to tolerate high levels of Al^{3+} (Gould et al. 2014). This question requires more insight into the tolerance of plants

within the community but could have wide reaching consequences for management of the sites for eradication of exotic, non-tolerant species.

The literature review and field study have clearly indicated that further study and understanding of the germination and propagation of native species, especially *L. juniperina*, is required to successfully restore indigenous habitats (Rowarth et al. 2007).

Chapter 4

The Role of Non-vascular Plants on Hydrology within a Kānuka Dryland Shrubland

4.1 Introduction

Vascular plants are known to assert a strong influence on the hydrology of an ecosystem, as demonstrated by the concept of the Soil-Plant-Atmosphere Continuum (Bond et al. 2008), from uptake through roots to evapotranspiration (Bosch and Hewlett 1982; Chapin III et al. 2011; Kramer and Boyer 1995). The morphology of the plants modify rainfall interception and prevent runoff (Ludwig et al. 2005b; Starks et al. 2014). Non-vascular plants, on the other hand, have received much less consideration in this context despite being ubiquitous throughout the world, occurring on every continent in a range of terrestrial and aquatic habitats, environmental conditions and substrate (Lindo and Gonzalez 2010; Vanderpoorten and Goffinet 2009). With the exception of key habitats, such as peat bogs, these lower plants have been overlooked for a number of reasons including their small stature (Eldridge 1999), the difficulty with identification (Cornelissen et al. 2007; Turetsky et al. 2012) and the inaccessibility of the growth habitat (Boch et al. 2013; Kiebach et al. 2016). However, in the last decade these highly adaptive organisms have slowly been gaining attention and are increasingly being recognised as important components of ecosystems (Hallingbäck and Hodgetts 2000; Longton 1992b) often contributing a large proportion of the photosynthetic biomass (Binkley and Graham 1981; Hofstede et al. 1993) and influencing the functional properties to the ecosystem (Davis 1981).

Water use within vascular plants is well studied; to overcome the scarce and unpredictable nature of water in ecosystems, vascular plants have evolved internal transport structures in which roots access water from the soil, and cuticles and stomata on the leaves regulate evapotranspiration (Raven 1977). Of course, mosses possess an alternative system for water regulation without a vascular structure (Proctor 1982). Although morphologically similar, the leaves of bryophytes are often only one cell thick, and lack a cuticle and vascular apparatus (xylem and phloem) (Glime 2015; Richardson 1981). Hence, most mosses are poikilohydric; having a reduced ability to regulate water loss leading to the equalising of internal and external moisture levels (page 12 of this thesis) (Proctor 2009; Tuba et al. 1996). Instead of internal transport of water and nutrients, conduction is external in capillary spaces with absorption through stems and leaves either from surface water or through dry deposition (either rainfall, mist droplets and airborne dust) (Bates 1992; Tuba et al. 1996). Mosses which employ this method are referred to as ectohydric and readily rehydrate when provided with water. Some mosses can also gain solution from the substrate upon which they grow, those assessed as

‘endohydric’, comprising modified rhizomes which act similarly to vascular structures and with leaves containing a waxy cuticle (Proctor 2000; Richardson 1981). These mosses also retain the ectohydric ability and can gain water through external channels (Zajączkowska et al. 2017).

Given this dependence on and ability to intercept atmospheric moisture, it is logical that mosses may exert some influence over the hydrological cycle. Research on this has been conducted in biomes with temperature extremes such as in arctic (Gornall et al. 2007; Zimov et al. 1995), boreal (Betts et al. 2001; Betts et al. 1999) and arid systems. In arid systems, Belnap (2006) comprehensively explored the research attributing the hydrological influence of Biological Soil Crusts (BSCs; including bryophytes and cyanobacteria, green algae, microfungi, bacteria and lichens) to their effect on processes, such as infiltration, surface runoff, water holding capacity. BSCs in arid habitats are also shown to facilitate encroachment by vascular plants which further alters the water cycle. Similarly, in Sweden, mosses found on limestone pavements were able to support vascular plants due to the increase in moisture within the cushions (Sand-Jensen and Hammer 2012). In the arctic, mosses contributed highly to evaporation rates resulting in a loss of water from the system (McFadden et al. 1998), whilst in a boreal forest mosses were shown to be important elements of the water cycle influencing surface hydrology, substrate temperature and evapotranspiration (Betts et al. 1999; Bonan and Shugart 1989; Van Cleve and Viereck 1981).

Despite this knowledge base, the hydrological influence of bryophytes within ecosystems is often not fully understood (Bond-Lamberty et al. 2011), as is exemplified in New Zealand where research surrounding the topic is scarce (DeLucia et al. 2003; Michel et al. 2013). This is not related to a lack of moss diversity or abundance: New Zealand has over 500 species of Bryophyta from over 200 genera, with approximately 20% of those being endemic (Fife 1995). These can attain over 90% ground cover in forests and shrublands, especially in wet habitats (Pfeiffer 2003). However, even in some of the driest habitats within New Zealand mosses, particularly *Hypnum cupressiforme*, can form almost continuous carpets under kānuka shrubland communities (Beever 1986; Macmillan 1976; Molloy and Ives 1972).

In chapter 3 of this thesis, kānuka shrubland habitats which contained high percentage cover of moss (remnant habitats) had similar soil moisture to those kānuka habitats which were receiving irrigation from neighbouring land use. Therefore, the objective of this chapter was to investigate the role of the moss layer in the hydrology of the remnant habitats with reference to similar studies elsewhere (excluding those in peatland and bog mosses). The aims were:

- i. To investigate the water holding capacity of the moss layer;

- ii. To investigate the interaction of the moss layer with the soil beneath with regards to transfer of moisture and influence on temperature regime; and
- iii. To investigate potential interactions between moss layer, soil moisture and the growth of vascular plants.

4.2 Methodology

4.2.1 Study Sites

Field experiments were undertaken in kānuka shrubland remnants located at Eyrewell Forest. Two kānuka shrubland remnants were selected; a privately owned Spencer-Bower Reserve (SBR – 43°42'91.13" S, 172°43'53.98" E) and a Department of Conservation owned Eyrewell Scientific Reserve (ESR – 43°38'31.56" S, 172°19'46.43"). These habitats comprised canopies of kānuka (*Kunzea serotina*) with an understory dominated by prickly mingimingi (*Leptecophylla juniperina*) and in excess of 75% ground cover of bryophytes, mainly hypnum moss (*Hypnum cupressiforme*). The predominant soil type at the sites is Lismore, a free draining, shallow, silty loam (Landcare Research 2011). Further description of soils and the remnant habitats can be found in Chapter 3.

4.2.2 Rainfall, Canopy Interception and Moss Cover

In order to determine the proportion of rainfall being intercepted by the canopy and, subsequently the amount available for the understory, bulk rain gauges were set up outside of the kānuka canopy within ESR (n=5) and SBR (n=3). Smaller rain collectors (40 cm diameter, 50 cm tall cylinders) were placed at random locations under the kānuka canopy at both locations (ESR n=48, SBR n=30). The amount of rain collected was measured immediately following a rainfall event on six occasions for ESR and three for SBR during the summer and autumn months 2015/16. The amount of water intercepted was determined by the following equation, taking into account of the size difference in rain gauge equipment:

$$\%UR = (UC_{\mu} / OC_{\mu}) * 100$$

Where %UR is the percentage of rain reaching the understory, UC_{μ} is the mean rainfall volume (mm) collected under the canopy and OC_{μ} is the mean rainfall volume (mm) collected outside of the canopy.

Additional data were collected at each of the 48 rainfall collection sites under the canopy at random locations within ESR. A 1 m² quadrat was positioned with the smaller rain collecting pot in the centre. Mean bryophyte depth was calculated by recording depth in the four corners and the centre

of plot and total non-vascular plant cover and *H. cupressiforme* cover recorded as a percentage of the total quadrat area.

4.2.3 Water Holding Capacity

To protect the native bryophyte communities at Eyrewell turves of *H. cupressiforme* moss were collected instead (where possible) from areas under the pine forest which were to be felled and removed. Whole turves of moss positioned above the soil layer, including the brown, non-photosynthetic dead and decaying moss (herein known as dead moss layer), were collected from ten different locations. Live, green moss was separated from the dead layers and cut into twenty 5 cm squares of 1.5 cm depth, trimming depth from the underside of the turve. Another twenty squares of moss turves were cut but including 0.75 cm live moss and 0.75 cm dead moss layer to bring the total depth to 1.5 cm. Water Holding Capacity was calculated by fully soaking the moss turves in water for 24 h in a sealed container. The moss was carefully and lightly blotted and then immediately weighed, oven dried at 60°C for 72 h and then reweighed. The percentage increase from dry weight to saturated weight was calculated along with the water Holding Capacity (WHC) was calculated following an adapted equation from Michel et al. (2012):

$$WHC = (WM - DM)/DM$$

Where WM is the mass of the saturated moss and DM is the dry mass following drying in the oven.

This procedure was also followed for turves of moss containing live and dead moss for five different moss depths: 1 cm, 2.5 cm, 5 cm, 7.5 cm and 10 cm. The turves were trimmed carefully with scissors removing excess dead moss from the underside of the turves and attain the desired depth.

4.2.4 Hydrology of Moss Cover

The fate of water reaching the surface of the *H. cupressiforme* carpet was measured in a separate experiment in a glasshouse environment. Turves of moss containing more than 90% *H. cupressiforme* were collected and stored for 3 days inside a mist chamber within a glasshouse consisting of a polythene covered wooden frame with automatic water for 10 seconds every 2.5 h (Plate 4.1). The moss turves were trimmed, as above, to five different depths; 1 cm, 2.5 cm, 5 cm, 7.5 cm and 10 cm. Five cores of 5 cm diameter were cut from each moss depth and placed inside plastic cylinders (constructed from acetate and sealed with adhesive tape) connected to a wire mesh stand (to allow drainage) and open at the top (to allow evaporation). The plastic encased the whole depth of the moss turve to allow evapotranspiration only from the open ends of the sleeve and limit lateral evapotranspiration (simulating the moss layer under a kānuka canopy). The wire mesh stands

supporting the cylinders were placed in trays to capture drainage water (adapted from Price et al. (1997)). The turves were then left for 24 h in a covered room with netted walls and subjected to the ambient conditions to equilibrate to the environmental conditions (Bond-Lamberty et al. 2011).

Each turve was weighed prior to the addition of water and at 30 minute intervals for a total of four hours throughout the drying process. Any water drained was collected and measured. This process was carried out for three water volumes of 2 mL, 5 mL and 10 mL. There was one day between each experiment to allow the moss to return to ambient conditions and the three experiments took place on 11th, 13th and 15th February 2016. The weather on these days varied with temperatures of 25°C, 23°C and 31°C respectively with rain on the middle day. The water was administered by dropping water onto the surface of the moss in a regimented manner using a pipette, ensuring that the whole area received water. Difference in weight change between the moss turves and the amount of water drained provided an estimate of water retention within the moss and evapotranspiration rate.



Plate 4.1. Mist chamber located within a glasshouse and consisting of a polythene covered wooden frame with automatic watering system located inside (left). Many moss experiments were undertaken within this humid environment (right).

4.2.5 Hydrology Interactions of Moss, Soil and Vascular Plants

Field Study

Soil moisture in the field was measured on 2 occasions, during winter and summer months. Six categories of moss depth were identified and sampled within the ESR remnant: 0.1-1 cm; 1.1-2 cm; 2.1-4 cm; 4.1-6 cm; 6.1-8 cm; and >8 cm. Three soil cores of 5cm diameter and 7.5 cm depth were collected and bulked under each moss depth (n=c.20). Fresh soil samples at 5°C in air tight bags overnight and sieved to 2 mm the next day. Gravimetric soil moisture content was determined following standard methodologies (Blakemore, Searle, & Daly, 1987).

Soil temperature was also measured under three different depths of moss: 0 cm, 3 cm and 9 cm ($n=4$) for 19 days during the winter month of June 2016 and the summer months of December 2016 and January 2017. Data were collected using HOBO® Pro v2 weatherproof data loggers with two external temperature probes connected with 1.8m cables (U23-00x). The probes were placed 5 cm into the soil by lifting the moss turve, making a small incision into the soil using a soil knife, placing in the probe before gently sealing the opening and replacing the moss layer (where applicable). Data loggers were left *in situ* for 3 days to calibrate before data collection began. During the winter 4 data loggers under the 9 cm moss failed and therefore data are only available for bare soil ($n=4$) and 3 cm ($n=4$) moss covering. Similarly, in the summer, two of the data loggers in the thick moss treatment failed ($n=2$).

Laboratory Study

Two trials were set up in the laboratory to explore the transfer of water from moss to soil. The first involved fully saturated moss turves encased in the plastic cylinder as described above but without the wire mesh and stand. The depths were 0 cm, 1 cm, 2.5 cm, 5 cm, 7.5 cm and 10 cm, with five replicas of each. Eyrewell soil had been previously collected, sieved to 4mm and left to air dry for 24 h. Eight samples of this soil were taken to determine the soil moisture at the start of the experiment and the rest was used to fill 5 cm diameter, clear plastic pots. Each moss column was placed on top of a pot of soil and sealed with tape to create a single column of soil and moss (Plate 4.2). The weight of the whole column, including the moss and soil, was measured every hour for 6 h. At the end of the experiment soil samples from both the top 2.5 cm and the bottom 2.5 cm of the pot were analysed for soil moisture content.

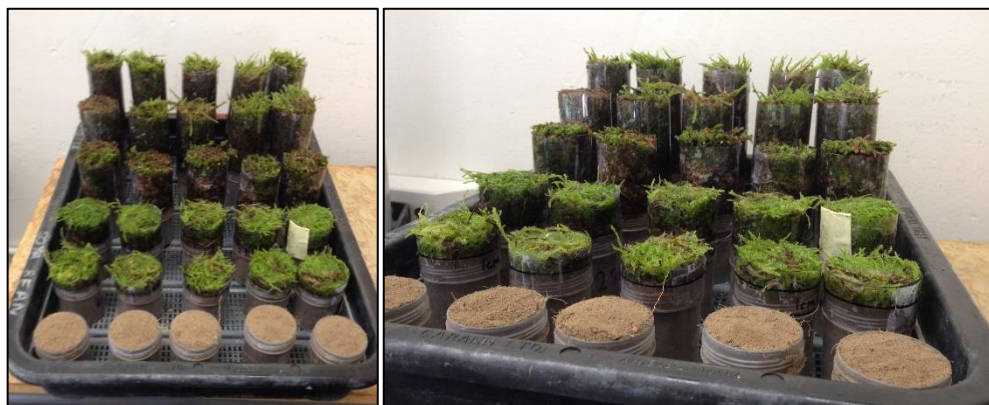


Plate 4.2. Experiment to assess transfer of water from saturated moss of different depths (1, 2.5, 5, 7.5 and 10 cm) encased in plastic tubes to dry soil beneath.

The above trial was repeated to explore the transfer of moisture from soil to moss. In this experiment, instead of saturating the moss, it was air dried for 7 days, then placed on soil which had approximately 30% moisture content. The weight of the soil, moss and the whole column was measured every hour for 6 hours and the moisture content of the moss and soil calculated as above.

To determine whether the moss layer affected moisture availability for vascular plants an experiment was set up in the greenhouse with 12 kānuka seedlings growing in 1L pots. Once the seedlings were 10 cm tall they were subjected to one of four treatments of control (no moss, bare soil), 3 cm moss and 6 cm moss layers and saturated with water. The moss turves for the two moss treatments (having been stored inside the mist chamber for three days) were cut to fit precisely in the pot and enclosed in plastic tube where the moss was above the pot. Pots were left inside the mist chamber for two days before being saturated with water again. The pots received no water from this point and were removed from the mist chamber. The plants were monitored and the time taken for each kānuka plant to reach water stress was recorded. Water stress was defined as the point that the upper leaves showed signs of wilting by colour or texture change and warping the shape.

4.2.6 Statistical Analysis

Differences between that amount of rainfall (\pm SEM) being retained in the kānuka canopy were analysed using SigmaPlot 12.3 (Systat Software, San Jose, CA). Linear regression was used to determine the relationship between the amount of rainfall and moss cover, moss depth and *H. cupressiforme* cover at the individual collection sites under the canopy also using SigmaPlot.

A Two-sample t-test was used to determine the significance of the Water Holding Capacity between live moss and live moss and dead moss samples. The relationship between moss depth and WHC was modelled using linear regression.

A simple regression was calculated to assess the relationship between the depth of moss and the amount of evaporation and drainage of different water applications in the laboratory experiment. The response variable was normalised where required using appropriate transformation after three outliers were deleted for the 10 mL treatment. Linear regression was also used to predict soil moisture and soil temperature in the field and a quadratic model linear regression was used for the laboratory experiment based on moss depth with Pearson's correlation defining the relationship.

For soil temperature the mean temperature (\pm SEM) for each treatment and each hour within the data collection period was calculated. A two-way ANOVA with repeated measures assessed the difference in temperature between treatment and hourly intervals. The mean soil moisture was calculated and the difference was calculated using one-way ANOVA with post-hoc Tukey HSD test.

To assess the difference in time taken for kānuka plants to become water stressed a one-way ANOVA was undertaken with post-hoc Tukey HSD test to define the relationship. All statistical analysis was carried out using Minitab® (Version 17.2.1., Minitab Inc., Sydney, AUS) and graphs were compiled on SigmaPlot.

4.3 Results

4.3.1 Rainfall and Moss Cover

The mean amount of rainfall intercepted by the canopy in both remnant sites was approximately 25%, leaving 75% available for the forest floor (Figure 4.1). The duration and intensity of rainfall varied but the maximum total rainfall recorded during one event was 23mm and the least was 3mm.

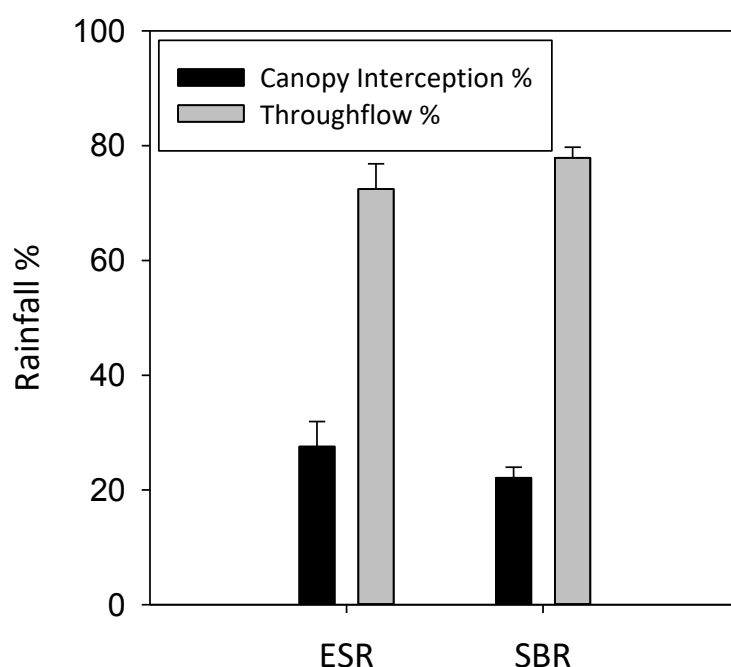


Figure 4.1 Percentage of rainfall (\pm SEM) intercepted by the kānuka canopy and the throughflow to the forest floor (including onto the moss layer) below at two remnant sites in the Eyrewell Forest area.

Within ESR, there was a positive linear relationship between rainfall reaching the surface of the moss carpet during the summer and autumn months and moss depth, moss cover and *H. cupressiforme* cover. However, correlation (r) and R^2 values were weak for each measurement indicating other factors that are also affecting the parameters (Figure 4.2).

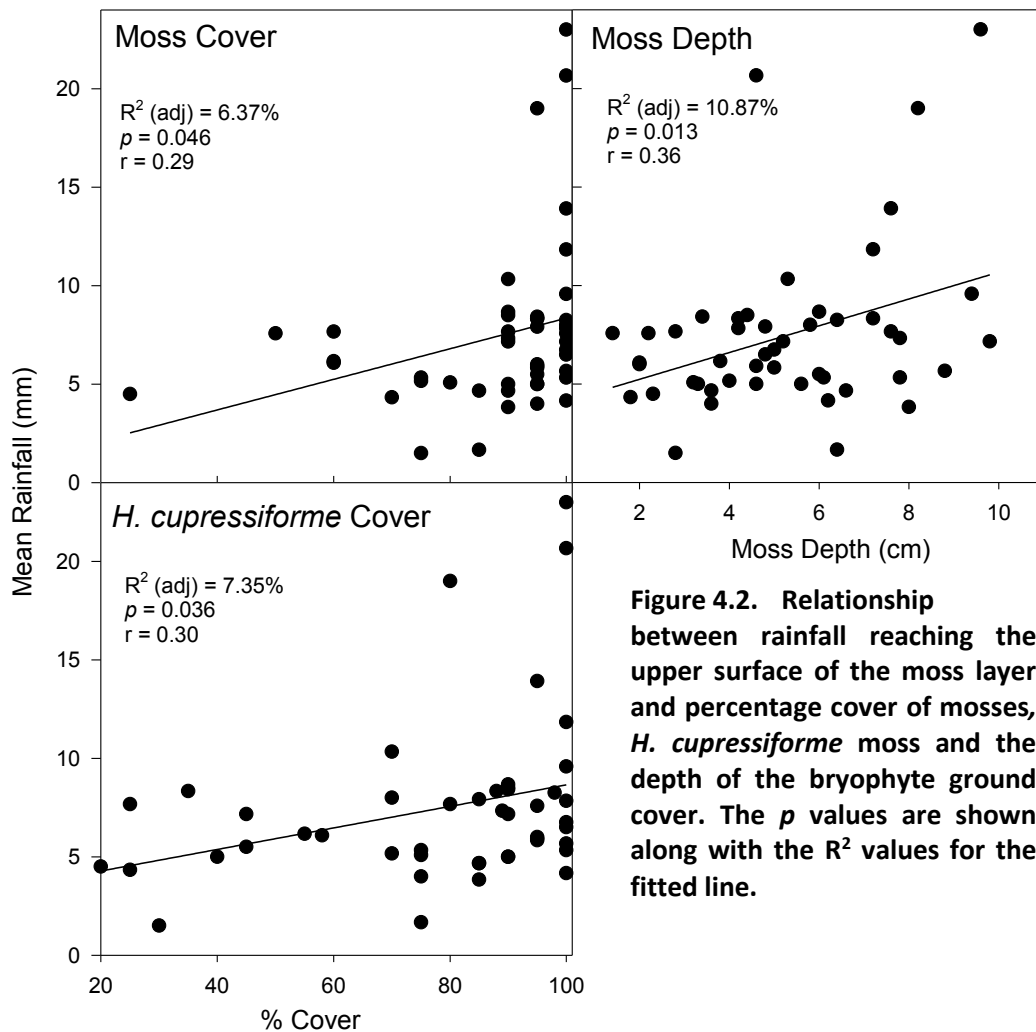


Figure 4.2. Relationship between rainfall reaching the upper surface of the moss layer and percentage cover of mosses, *H. cupressiforme* moss and the depth of the bryophyte ground cover. The p values are shown along with the R^2 values for the fitted line.

4.3.2 Water Holding Capacity

The live moss was able to hold a substantial amount of water (c. 15 times its dry weight) and was significantly more effective at storing water than the dead moss layer with a mean WHC value of 15 compared to 9 (Table 4.1). The 0.75 cm dead moss layer retained only 23% that of the total capacity of the live moss.

Table 4.1 The difference between water holding capacity of live moss and live moss with dead moss layer based on turves comprising 1.5 cm live, green moss and combinations of 0.75 cm live moss and dead moss layer ($p < 0.001$). Capacity is expressed as the WHC value or the percentage increase from dry weight to fully saturated.

Moss Type	WHC	Increase (%)
Live Moss	14.76 ± 0.45	1576.44 ± 44.65
Live moss and dead moss layer	8.75 ± 0.55	975.49 ± 55.31

When investigating the effect of the dead moss and the live moss together as a natural column found in the field there was a strong and significant positive linear correlation between water holding capacity and depth of the moss (Figure 4.3).

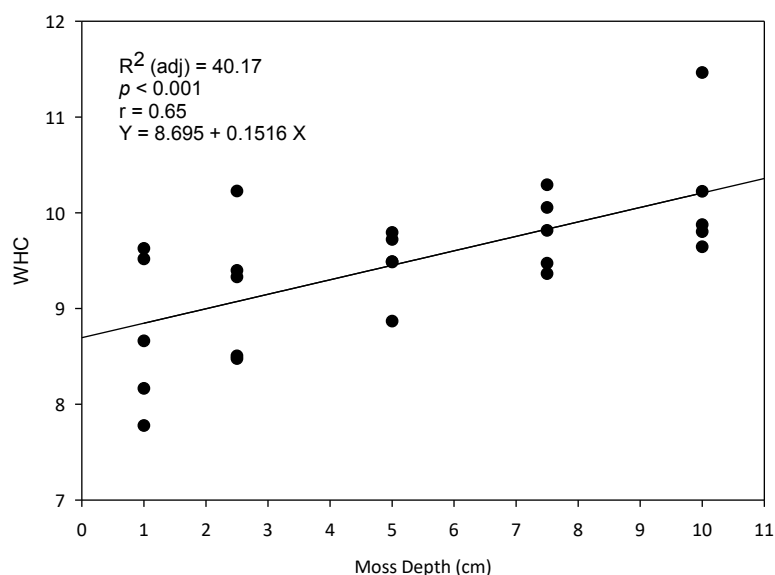
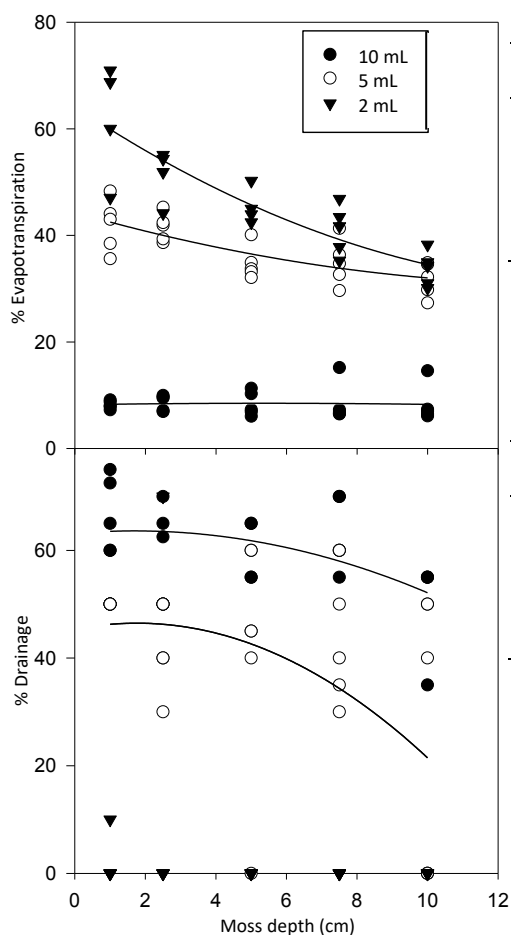


Figure 4.3. Water holding capacity (WHC) values occurring for each moss depth (1, 2.5, 5, 7.5 and 10 cm) including dead moss layer with linear regression line.

4.3.3 Hydrology within Moss

There was a significant negative relationship between the depth of the moss and the amount of evaporation in all water treatments (Figure 4.4). Evapotranspiration rates for 10 mL applications were low and less variable than 2 mL and 5 mL applications. There was little drainage from any moss depth when 2 mL was applied but a significant negative correlation between moss depth and drainage was found at the 5 mL and 10 mL amounts (Figure 4.4).



	<i>p</i>	<i>r</i>	<i>R</i> ²
2 mL	<0.001	-0.73	53.19
5 mL	<0.001	-0.73	52.84
10 mL	0.002	-0.61	37.72

	<i>p</i>	<i>r</i>	<i>R</i> ²
2 mL	0.325	-0.21	4.22
5 mL	0.013	-0.49	24.01
10 mL	0.016	-0.48	22.66

Figure 4.4.

Evapotranspiration and drainage values (shown as a percentage of the total water applied) occurring at each moss depth (1, 2.5, 5 7.5 or 10 cm) for the three water applications (either 2 mL, 5 mL or 10 mL). Significance and relationship is detailed in text adjacent to the figure.

4.3.4 Moss and Soil Interaction

Field Study

The gravimetric soil moisture in the field was found to be positively correlated with moss depth during the summer months and negatively correlated during the winter months (Figure 4.5). The standard error was largest at the 0.1 cm moss depth in both summer (± 1.16) and winter (± 2.86) due to the outlying data points both above and below the regression line.

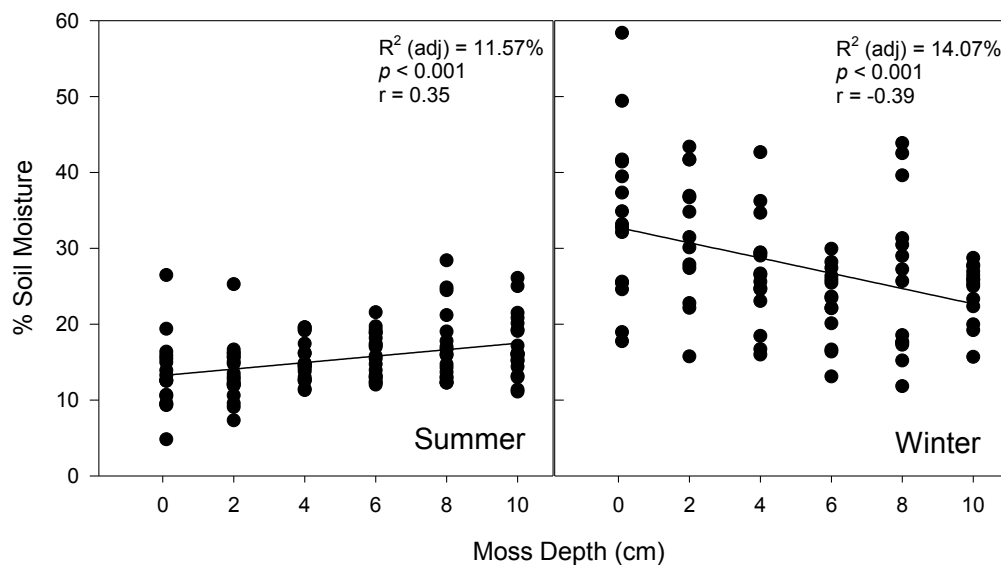


Figure 4.5. Gravimetric soil moisture beneath moss of increasing depth (0.1, 2, 4, 6, 8 and 10 cm). The spread of data points shows the variability in soil moisture most evident at 0.1 cm depth.

There was less fluctuation of soil temperature when the surface was covered by moss (Figure 4.6). In the winter experiment, the temperatures were lower under moss cover in the daytime and higher during the night. The mean diurnal fluctuation of bare soil was 1.34°C compared to 0.43°C under a thin layer of moss. The variance of temperature within both day and night was also greater in the bare soil treatment where the SEM was 0.13 and 0.10 respectively compared to 0.08 and 0.06 when a moss layer was evident.

During the summer months the same pattern was repeated for both bare soil and thin moss with a diurnal fluctuation of 1.39°C for bare soil and 0.68°C for thin soil. The thick moss had a temperature fluctuation higher than the thin moss of 0.95°C .

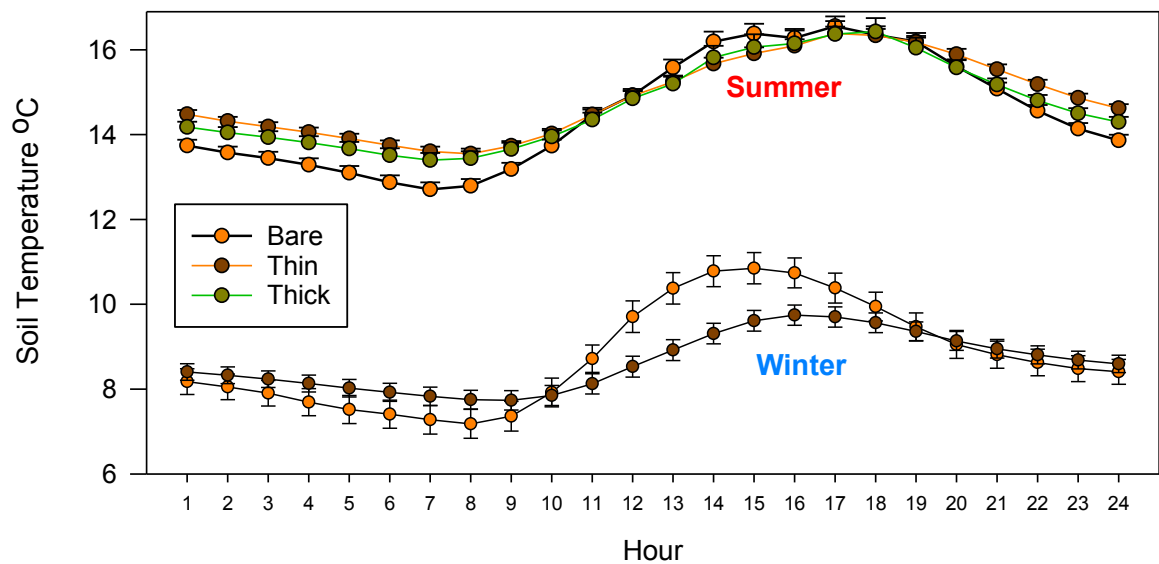


Figure 4.6. Mean soil temperature at 5 cm depth ($n=4$ except thick moss $n=2$) (\pm SEM) for bare soil, soil under a thin (3 cm) and a thick (9 cm) layer of moss ($p<0.05$).

Laboratory Study

The laboratory results mirrored the field results with a strong positive correlation between moss depth and soil moisture suggesting a greater transfer of moisture to soil with increased depth of moss due to increased water holding capacity of the moss (Figure 4.7). However, the curvilinear nature regression suggests a peak in soil moisture at 8 cm depth.

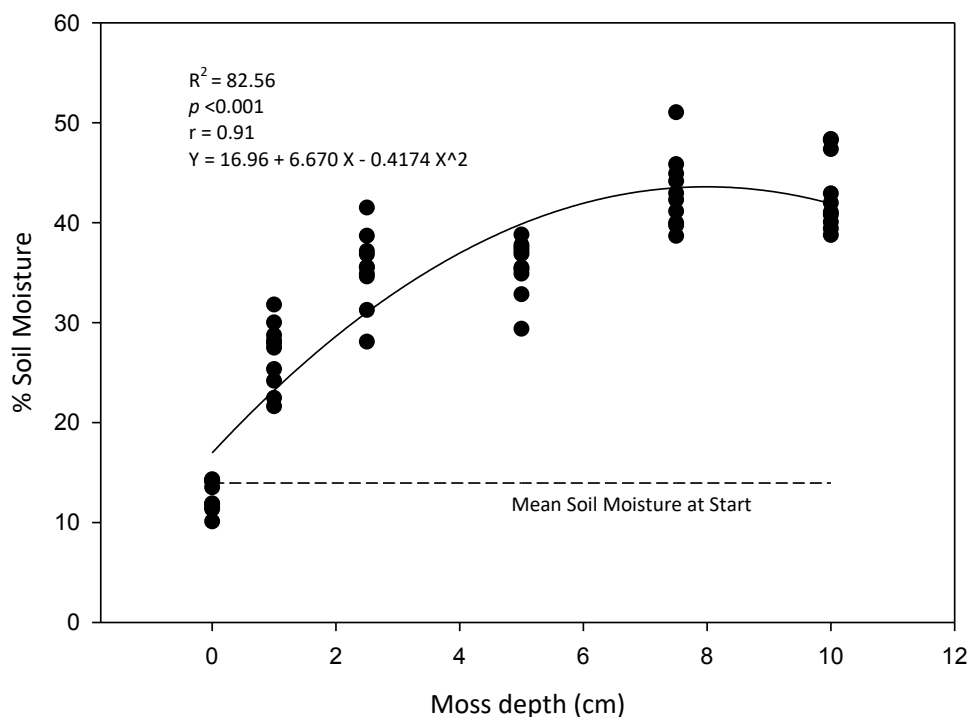


Figure 4.7. Gravimetric soil moisture when in contact with different depths of saturated moss for 6 h (0, 1, 2.5, 5, 7.5 and 10 cm). Dotted line represents the soil moisture of all samples at the start of the experiment. The soil moisture decreased through evaporation when no moss layer was present ($p<0.001$) and there was a positive relationship in moisture increase with depth of moss.

The addition of a moss layer reduced the soil moisture loss by at least 50%. Figure 4.8 illustrates the total moisture loss from the soil and how much was attributed to evaporation and that gained by the moss. The amount of moisture gained by the moss over six hours increased with depth and displayed a strong positive relationship. The evaporation decreased with moss depth and a strong negative relationship was determined. However, the values of moisture gained by the moss from the soil were low ranging from 0.02 mL to 0.26 mL over 6 h.

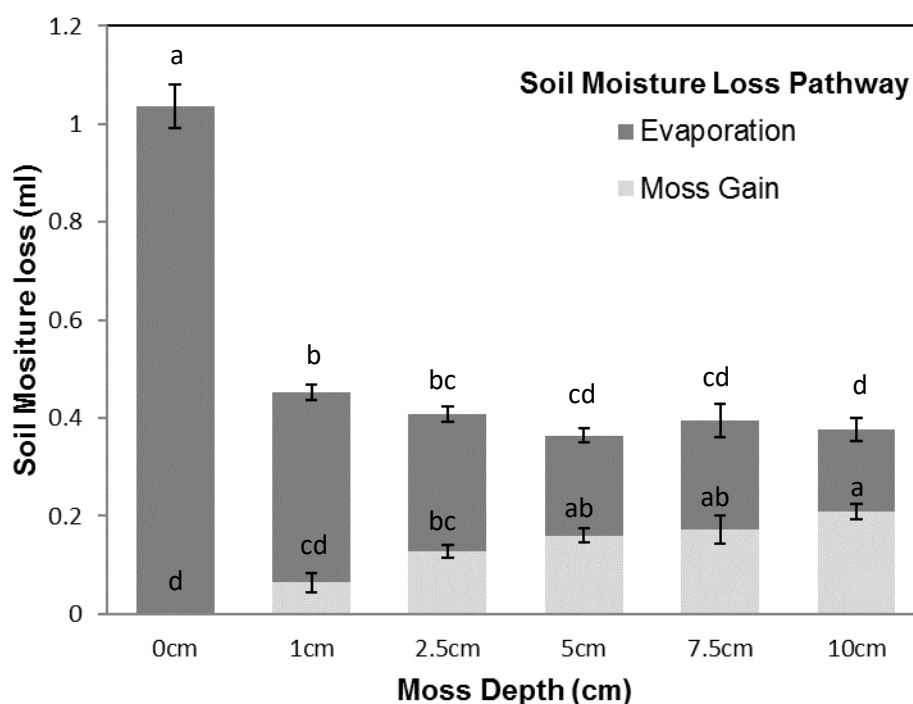


Figure 4.8. Amount of soil moisture lost (mL) from saturated soil over the course of 6 hours due to evaporation (dark grey bars) or movement into the moss column at different depths of air dried moss (light grey bars). Means that share a letter above bars indicate no significant difference between evaporation and treatment ($p < 0.001$, $R^2 = 72.13$, $r = -0.69$). Letters which are the same within the bars indicated no significant difference in moisture gained by moss and treatment ($p < 0.001$, $R^2 = 76.21$, $r = 0.83$).

4.3.5 Moss-Soil-Kānuka interaction

The addition of a moss layer extended the time to the water stress point for the kānuka plants by over a week (9 days) for thin moss and by almost two weeks (13 days) for thick moss (Figure 4.9).

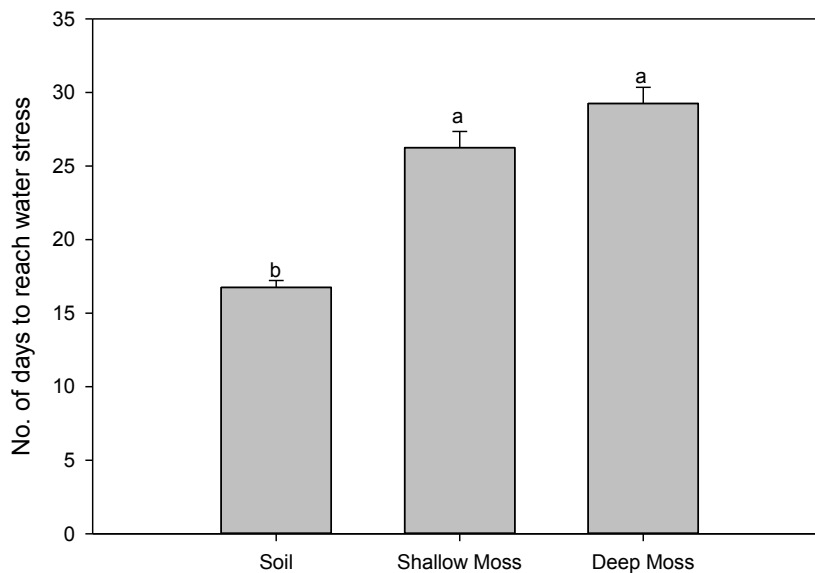


Figure 4.9. Amount of time taken for kākūka plants to reach water stress point (wilted, discoloured leaves) when planted in treatments with different depths of moss (0, 3 and 6 cm). Letters which are the same indicate no significant difference ($p < 0.05$).

4.4 Discussion

4.4.1 Rainfall and Habitat

Hypnum cupressiforme growing within the kākūka shrubland was increasingly present in areas with higher canopy infiltration of rainfall but the relationship was weak indicating that the moss was influencing the hydrology of the habitat rather than simply being distributed in preferentially wet areas. *H. cupressiforme* is known to establish in dry areas and is negatively correlated with rainfall (Michel et al. 2011b; Segreto et al. 2010). The kākūka shrubland community in the present study is situated on the Canterbury low plains in the rain shadow of the Southern Alps. It typically receives approximately 600mm annual rainfall, distributed throughout the year but peaking slightly in the winter months of June, July and August (Macara 2016; NIWA 2017). Approximately 25% of rainfall was intercepted by the kākūka canopy, preventing it from reaching the understory vegetation and soil. Taking this into consideration, the total rainfall input for the understory and ground layer species may be less than 450 mm year⁻¹. This supports the idea that the kākūka communities on the plains inhabit a near semi-arid environment (Quinn and Woodward 2015).

Throughflow of rainfall in kākūka and mānuka stands on the North Island have estimated higher rates of 37-42% (Aldridge and Jackson 1968; Rowe et al. 1999) but these sites had a higher total rainfall figure which may have altered the intensity and the throughflow figures. In a beech forest (*Northofagus*) in Nelson the throughflow was calculated as 22-35%, assuming stemflow of 2% (Rowe 1983). Stemflow was not calculated for this study but previous findings have indicated it does not make a large contribution to throughflow amounting to 1% for kākūka stands (Rowe et al. 1999) and boreal forests (Price et al. 1997).

4.4.2 *H. cupressiforme* in Kānuka Stands

This study confirmed that the live *H. cupressiforme* has a high water holding capacity (DeLucia et al. 2003; Michel et al. 2013) retaining more moisture than the decaying horizons beneath. However, the build-up of decaying moss fragments under the live moss can help to retain water within the moss layer through prevention of drainage to the underlying soil. Over time, the retention of water above the soil surface facilitates transfer from the wet moss to dry soil (Bu et al. 2015), even in repellent soils with waxy organic covering on soil particles, due to the constant contact (DeBano 1981). This is reflected in the field study results that illustrated less fluctuation of soil moisture between the summer and winter months; only 5% difference under moss cover compared to over 20% difference with bare ground. This reduction in fluctuation was mirrored in the temperature results, both winter and summer results showed a variation of approximately 1.3°C in bare soil and approximately 0.5°C in moss covered soil.

This study did not find significant difference in temperature during the summer daytime period under moss or within bare soil. However, during the summer months with low rainfall, the moss is likely to assist in prevention of evaporation from the soil therefore retaining higher moisture levels as found in the present work (Figure 4.5). Many studies have cited the additive benefits of moss cover in reducing evaporation from soil (Chamizo et al. 2013; Xiao et al. 2010). Moreover, the presence of a moss layer can decrease runoff and retain more moisture within the soil-plant continuum (Alam et al. 2012; Chamizo et al. 2012). In addition to the water input of rainfall, mosses have the ability to harvest water vapour from the atmosphere from fog and dew (Wang et al. 2014b; Zhang et al. 2008). During the summer months, the moss may have been exploiting this resource and in turn supplying the dry soil with small inputs of moisture further explaining the increased soil moisture under the moss layer.

A possible explanation of drier soils during the winter months could be due to the moss capturing and retaining water, preventing filtration to the soil. In boreal forests it has been reported that moss cover can intercept 23% of total rainfall (Price et al. 1997; Suzuki et al. 2007). In addition, some studies have found that the presence of a biological soil crust can increase water repellency (Lichner et al. 2012) and repellency is greater when the soil is dry (Dekker and Ritsema 1994). However, these studies were mainly undertaken in arid environments and there is evidence to suggest that infiltration rates may increase after each rainfall event (Bu et al. 2015). Furthermore, studies in which the biocrusts are dominated by moss have been shown to increase infiltration due to their morphology and effects on soil (Belnap et al. 2013; Chamizo et al. 2016b).

During the winter months, the soil temperatures would have been warmer under the moss carpet which may have added to an increase in soil moisture evaporation. The moss was metabolically active during these wetter months (Pitkin 1975; Proctor et al. 2007) possibly further reducing soil

moisture due to evapotranspiration. *H. cupressiforme* can show vigorous growth in woodlands and grasslands of Europe when compared to other moss species (Zechmeister 1998) and, although studies detailing the precise effect of *H. cupressiforme* to evapotranspiration rates in temperate woodlands, mosses are known to be substantial contributors within boreal forests (Heijmans et al. 2004). However, in general, although the moss may have imposed an increased water demand upon the ecosystem many studies have indicated a trend for increased soil moisture under moss layers (Bu et al. 2015; Chamizo et al. 2016a) but perhaps only in the upper surfaces (Xiao et al. 2016).

4.4.3 Conclusions and Implications for Vascular Plants

The effect of the terricolous moss cover in this low rainfall environment is clearly significant but it is complex and involves multifaceted interactions. The interwoven components of hydrology within an ecosystem with regards to the moss layer were set out by Chamizo et al. (2016a) (Figure 4.10). They investigated results for many studies and concluded that in general BSCs can reduce soil moisture evaporation, increase soil moisture and harvest non-rainfall water such as fog and dew (Wang et al. 2014b; Zhang et al. 2008). At a more regional scale, components such as topography and climate also influence the relationship.

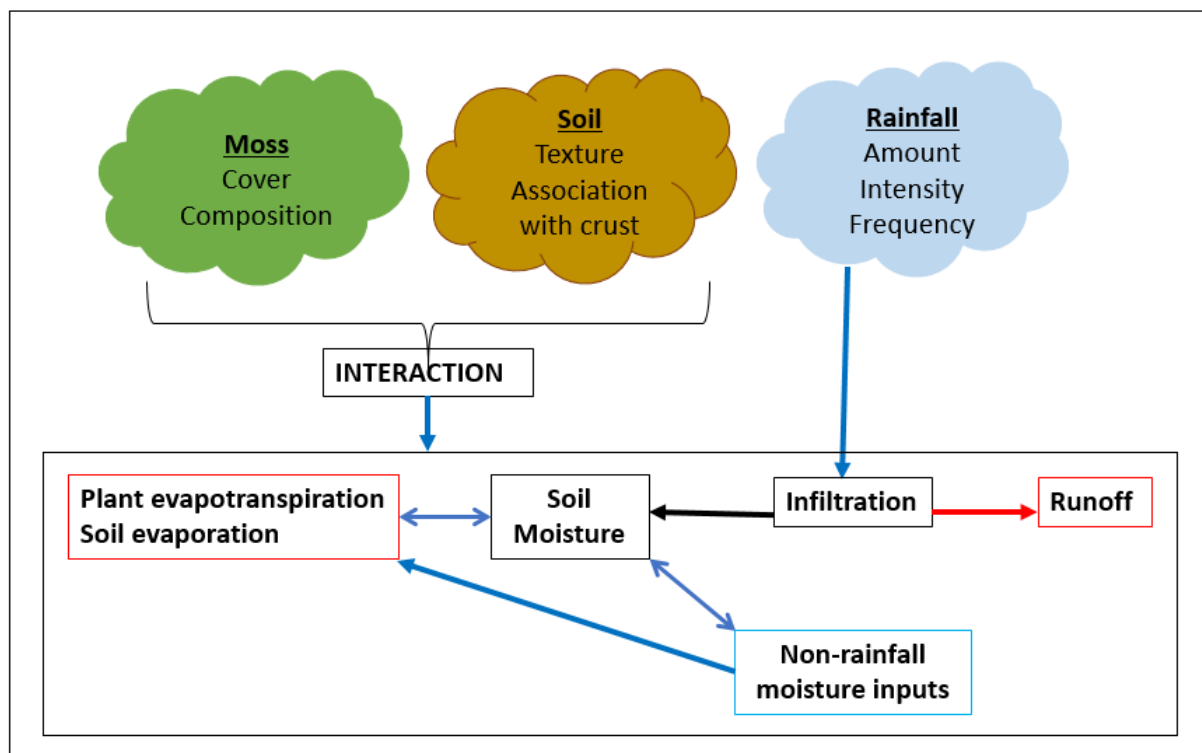


Figure 4.10. Complex interactions between features of biological soil crusts, soil parameters and water inputs at a local scale. The blue arrows and boxes show water inputs and the red shown water lost from the system. Adapted from Chamizo et al. (2016a).

The present study has indicated that the moss is a stabilising agent within the kānuka shrubland, buffering extremes of both soil moisture and temperature. In these dry, well-drained soils the stabilising effect could benefit vascular plants in the area. The drainage of water and leaching of nutrient solution is common in the Lismore soils (Landcare Research 2011). Therefore, instead of rain water being immediately lost through drainage, it is retained within the ecosystem and steadily transferred to the soil. The input of non-rainfall water harvested by the moss carpet would potentially also be useful in this system, inputting further moisture into the dry soil, especially in the dry summer months.

Vascular plants in the Lismore soil reach wilting point at around 10-15% soil moisture (Drewitt 1979). In this study, the deep moss-covered soil (10 cm) displayed gravimetric soil moisture between 16.01% and 24.28% as measured in the winter and the summer respectively, compared to 13.38% and 34.19% for bare ground. This indicates that the moss covering was preventing the soil reaching wilting point in the summer at least for the upper 7.5 cm. Many New Zealand native plants have shallow roots (Franklin 2014) and would be able to utilise the added soil moisture. The results of this study show that the presence of moss prevented water stress in young kānuka plants for almost two weeks.

This ability to conserve water may have important implications for climate change. The kānuka shrublands of the Canterbury Plains comprise a xerophytic community due to the climate and soil leading to water limitation (Bowie et al. 2016; Meurk et al. 1995; Molloy and Ives 1972). In New Zealand, climate change is anticipated to alter rainfall patterns and increase drought frequency further exacerbating the soil moisture deficit in these habitats (Allen et al. 2014; Ministry for the Environment 2016). Therefore, the presence of a moss layer is likely to be beneficial, preserving enough moisture in the soil to maintain the xerophytic plant communities until the next rainfall event or period of available water vapour, such as fog or dew, as seen in many arid environments (Chamizo et al. 2016a).

Alternatively, it could be argued that the interaction of the bryophyte layer with the hydrology of the habitat is detrimental for the ecosystem by relieving competitive exclusion due to water limitation and leading to invasion of exotic species. The kānuka shrublands on the Canterbury Plains have been invaded by a multitude of exotic species from horticulture and agriculture (Bowie et al. 2016; Meurk and Swaffield 2000; Meurk et al. 2016). The majority of these exotic species are found in temperate climates in which nutrients and moisture are more readily available than on the Canterbury Plains, hence the reason for fertilising and irrigation pivots within pastures (McLaren and Cameron 1996). This is evident in some dryland communities, the increased moisture facilitates vascular plant growth (Brown and Archer 1999) which in turn diminishes the bryophyte layer due to competition for light and an input of smothering litter and nutrients detrimental to the mosses (Sveinbjornsson and Oechel 1992; van der Wal et al. 2005). However, bryophytes can also interact with vascular plants in

inhibitory ways which may be highly species and site specific (Jeschke and Kiehl 2008; Michel et al. 2011a; Zamfir 2000). Further research into the interaction of the moss layer with vascular plants within the kānuka shrubland is needed and is addressed in the next chapter.

Conclusion

The moss ground layer within remnant kānuka communities at the sites studied have a significant influence on the hydrology of the habitat, modifying soil moisture and temperature with associated effects on vascular plants. Mosses assist in the soil moisture gradient in the kānuka stands and are a significant determinant of community assemblage. Walker et al. (1999) suggested that soil moisture governed the diversity of vascular plants in Central Otago, New Zealand. This corroborates the findings of earlier studies. Michel et al. (2013) found changes in ecosystem functioning related to hydrology when mosses were removed from a New Zealand grassland habitat and argued the importance of restoring bryophytes in degraded grassland systems. The same would appear to apply for mosses in dry kānuka shrubland communities.

Chapter 5

Interaction between Non-vascular Plants, Soil Chemistry and Biotic Communities within a Kānuka Dryland Shrubland

5.1 Introduction

The evolution of vascular plants some 400 million years ago was an important global event exerting significant changes over energy and nutrient fluxes (Kenrick and Crane 1997). Due to this fact and the ease at which seed plants can be obtained, grown and studied, there has been a vast amount of research on the influence of vascular plants on nutrient cycling (Aerts and Chapin 1999; Grime et al. 1997; Vries and Bardgett 2016). This includes studies of uptake and use of nutrients, biomass turnover, litter quality and production of root exudates which stimulate soil microorganisms and herbivores (Chapin III et al. 2011; Hobbie 1992). However, before the arrival of vascular plants, non-vascular plants were harnessing atmospheric nutrients and beginning weathering processes which had a substantial effect on global ecosystems (Asplund and Wardle 2016; Bateman et al. 1998; Cornelissen et al. 2007; Lenton et al. 2016; Porada et al. 2014). Non-vascular plants are being increasingly recognised as important components of nutrient cycling and ecosystem structure and function (Hallingbäck and Hodgetts 2000; Longton 1992b).

Acquisition of nutrients in mosses involves absorption through stems and leaves harvesting nutrients from surface water or through dry deposition (rainfall, gases, mist droplets and airborne dust) (Bates 1992; Glime 2015; Tuba et al. 1996). Although this is thought to be the main pathway, there is evidence to suggest that some mosses can also gain nutrients from the substrate (Brūmelis et al. 2000; Van Tooren et al. 1990) possibly in association with saprophytic basidiomycete (Wells and Boddy 1995). Storage of ions within the moss can be intracellular, within the cells and cytosol (such as potassium) but can also be held on extracellular exchange sites (e.g. calcium and lead) (Bates 2009; Brown and Buck 1985). Furthermore, the morphology of many mosses enables dust particles (and thus nutrients) to be trapped between overlapping leaves or within the clasping leaves or held in solution between the leaves and cells (Brown and Bates 1990; Glime 2015).

Associations with microbes increase the ability of bryophytes to gain nutrients such as the association with cyanobacteria which fix nitrogen (Hawkes 2003; Zielke et al. 2005). It is estimated that the activity of these non-vascular associations account for almost half of terrestrial nitrogen fixation (Elbert et al. 2012). Associations are known in a number of species including *Grimmia* spp., *Funaria hygrometrica*, *Bryum* spp, *Racomitrium* spp., *Polytrichum juniperina* and *Ceratodon purpureus*, the latter four of these species known within the moss layer in the Eyrewell Scientific Reserve (ESR) (During

and Tooren 1990; Solheim and Zielke 2002). There is also evidence of associations between *Hypnum* spp. and *Phormidium* sp. and *Gloeocapsa punctata* in the UK (Whitton 2012). This infers that through a number of processes and associations bryophytes can access nutrients from the atmosphere which are not available to vascular plants restricted to the soil pool resulting in both positive and negative effects on higher plant growth in different environments (During and Tooren 1990).

Mosses can be a potential source of nutrients to vascular plants supplying leaked nutrients from their cells upon rehydration, a process which may be detrimental to the moss (Brown and Buck 1979; Wilson and Coxson 1999). The mycorrhizae networks may also supply nutrients to surrounding plants although more research is required to substantiate this link (Brown and Bates 1990; Chapin et al. 1987; Davey and Currah 2006; During and Tooren 1990). Other benefits of mosses to vascular plants include provision of nutrients to germinating seeds and seedlings, buffering the harsh abiotic conditions (Belnap 2006; Rayburn et al. 2012; Sand-Jensen and Hammer 2012; Sohlberg and Bliss 1987) and protection against seed predation due to camouflage within the moss carpet (During and Tooren 1990). Mosses may facilitate growth of indigenous plant species preferentially over exotic (Gimingham 1972; Morgan 2006).

Mosses may also limit available nutrients to surrounding vascular plants. Certain mosses have been found to be efficient at sequestering nutrient precipitation inputs in the live moss and accumulating organic matter making it unavailable for vascular plants in the short term due to the length of time it takes to decompose (Bowden 1991; Cornelissen et al. 2007; Jonsdottir et al. 1995). Decomposition can be slow for a number of reasons including acidic environmental conditions, chemical composition, low temperatures and moisture content yet decomposition rates for many species is still unknown (Bates 2009; Brown and Bates 1990; Turetsky 2003). Additionally, although some mosses make associations with cyanobacteria which fix nitrogen, Rousk et al. (2016) found that lichens and mosses retained the fixed nitrogen rather than releasing it into the soil. Furthermore, studies in the arctic have shown the influence of moss cover on abiotic factors such as soil temperature and moisture content which in turn limit microbe activity and rates of mineralisation and nitrification (Gornall et al. 2007; Van Cleve and Yarie 1986).

Bryophytes and lichens may also inhibit regeneration of vascular plants by forming a barrier to the soil, thereby preventing seeds reaching the substrate, drying the seed or maintaining excess moisture, sustaining temperatures too cold for germination, reducing available light levels, increasing the above ground time available for seed destruction, and through allelopathy (During and Tooren 1990; Head et al. 2004; Michel et al. 2011a; Sohlberg and Bliss 1987; Zamfir 2000). However, interactions are complex and although germination may be inhibited, the growth of those plants which do establish can be promoted by bryophytes (Houle and Filion 2003).

In the work reported in the present chapter, I sought to explore the relationship between *Hypnum cupressiforme* and vascular plants within the dryland kānuka (*Kunzea serotina*) shrubland of the Canterbury Plains, New Zealand. Results were discussed with reference to similar studies, focussing on bryophytes in forest habitats and generally excluding those found in peatlands and bog mosses, *Sphagnum* spp. that have a different chemistry and altered environmental conditions. The aims were:

- i. To investigate the potential input and output of nutrients to and from the *H. cupressiforme* carpet; and
- ii. To explore the interactions with vascular plants and the effect of the moss carpet on germination of native species.

5.2 Methods

5.2.1 Nutrient Interactions of *H. cupressiforme*

To assess the chemical composition, ten cores (2.5 cm diameter, 5 cm depth) of *H. cupressiforme* were collected from random locations within a remnant vegetation stand (under canopy) at SBR (see Chapter 3 for location and description). The moss was carefully separated into the green, live moss and the dead moss layer. Half of the live moss was set aside whilst the other half was thoroughly washed and rinsed under deionised water. This process resulted in ten samples of live washed moss, ten samples of unwashed live moss and ten samples of dead moss layer. All were dried in an oven at 60°C for 4 days, ground and sieved to 2mm before being microwave digested in 5M HNO₃ and analysed using a Varian 720-ES Inductively Coupled Plasma Optical Emission Spectrometer fitted with an SPS auto-sampler and ultrasonic nebuliser (ICP-OES) for elemental concentrations (Blakemore et al. 1987). Four soil samples, each made up of four samples which were bulked, were collected from kānuka stands under the moss layer within the Eyrewell Forest. Samples were collected using a soil corer (2.5 cm diameter and 5 cm depth), air dried for 4 days at 25°C, ground and sieved to 2 mm. The samples were then also analysed for macro and micronutrients using ICP-OES.

In addition, rainwater samples collected under and outside of the kānuka canopy were also analysed to evaluate the input of nutrients from the rainfall and the rain filtering through the canopy as throughflow. The rain was collected in 40 cm diameter, 50 cm tall cylinders immediately following rainfall event on 28 May, 2016. Two remnant vegetation stands were surveyed, ESR and SBR (see Chapter 3 for descriptions), both under the canopy (n=5) and adjacent to the stands outside of the canopy (n=4). Each sample was filtered to remove the small amounts of debris which had fallen into the pots and then analysed using ICP-OES.

Leachate from moss turves, and hence the potential addition of nutrients into the soil from moss, was assessed by pouring deionised water (10 mL) over unwashed turves of 5 cm depth, comprising live and dead moss layers, and collecting the drainage over 2 hours (n=10). The drained water was sieved to remove debris and analysed using ICP-OES.

To assess the effect of moss cover upon available nitrogen in the soil, soil samples were collected under four depths of moss (0, 3, 6 and 9 cm) during the winter and summer months within the remnant kānuka stand at ESR (n=8). Soil samples under three depths of moss (0, 3 and 8 cm) were also collected from the SBR remnant in the summer and winter months (n>13). Soil samples were returned to the laboratory immediately, sieved to 4 mm and analysed for KCl extractable nitrate and ammonium using standard methodologies described in Chapter 3 (Blakemore et al. 1987).

5.2.2 Moss – Seedling Interactions

Laboratory Experiment

The effect of the moss layer on establishment and growth of vascular plants was investigated in a controlled laboratory experiment. Five seeds of native broom (*Carmichaelia australis*) and ten of *K. serotina* and *Pomaderris amoena* were placed together in 20 trays, each containing one of five treatments (n=4): a control of Eyrewell soil to 2 cm depth and four moss treatments of Eyrewell soil covered in *H. cupressiforme* moss layers of 2.5, 5, 7.5 and 10 cm depth. Following the results of the germination trials outlined in Chapter 3, the *P. amoena* seeds were pre-treated in boiled water and left soaking as the water cooled to room temperature for 12 h immediately prior to use. The moss turves were cut using scissors, trimming the dead moss layer, to achieve the required depth. The trays were located inside a mist chamber located in a glasshouse with positions of the trays being rotated weekly to avoid location effects (mist chamber described in Chapter 4, Plate 4.1). Seeds were left to germinate and establish for 6 months, apart from *K. serotina* which was left for 4 months, and were checked for germination on a weekly basis. On completion of the experiment, plants were carefully harvested, washed to remove soil particles and oven 72 h at 60°C to gain dry biomass weight. Soil samples were also analysed for soil moisture content and KCl extractable nitrate and ammonium using standard methodologies outlined in Chapter 3.

Field Experiment

The germination and establishment of native seeds within the moss layer was investigated in the field. Eight plots of 0.5 m² within a kānuka field boundary at Eyrewell Forest were marked out (43°39'48.2" S, 172°25'27.5"E). Four of the plots had at least 80% ground cover of *H. cupressiforme* at over 1 cm depth whilst four were control plots with at least 80% bare soil. Seeds from three native species were collected from remnant plants in Eyrewell Forest and hand sown over the plots (Table 5.1). Number of

seeds was dictated by the amount available for collection. Plots were monitored monthly for ten months and germination monitored.

Table 5.1 Seed quantity used within field plots in the vascular plant germination experiment.

Species	Approx. No. Seeds/plot
<i>K. serotina</i>	5200
<i>P. amoena</i>	1500
<i>C. australis</i>	60

5.2.3 Statistical Analysis

Nutrient Interaction

Macronutrient, trace element and available nitrogen data were analysed using SigmaPlot 12.3 (Systat Software, San Jose, CA). Using Minitab® 17.2.1 (Minitab Inc., Sydney, AUS), one-way ANOVA (with post-hoc Tukey HSD test) was used to explore the differences in chemical element concentration between the various components of moss and soil and differences in throughflow, rain water and moss leachate. Two-way ANOVA was used to assess the differences and interaction between the rain water sampling site and location (whether under the canopy as throughflow or collected in the open). The distribution of ammonium in winter in ESR could not be transformed to meet the normality assumptions of the ANOVA test, therefore a Kruskal-Wallis test was used.

Biotic Interaction

Means (\pm SEM) of seeds germinated, established and seedling biomass were calculated using SigmaPlot. A suitable transformation for the data could not be achieved to enable use of parametric statistical tests. Therefore, Kruskal-Wallis tests were performed to analyse the difference of the moss treatment on the germination, establishment and biomass of each species. The difference in soil moisture and nitrate was assessed using one-way ANOVA with post-hoc Tukey comparison.

5.3 Results

5.3.1 Nutrient Interaction of *H. cupressiforme*

The soil and moss concentrations of all elements were comparable to ranges recorded in other studies with the exception of low Ca in soil and low P and K concentrations in moss (Table 5.2). Soil contained significantly more aluminium, magnesium and iron compared to *H. cupressiforme*, both live and dead plant matter, but had lower concentrations in all other nutrients except potassium (Figure 5.1). Live moss (both washed and unwashed) contained the most phosphorus and copper whilst calcium, sulphur

and manganese were highest in the dead moss layer. Concentration of elements were similar between the washed and unwashed moss fragments with significant differences only observed in Cu and Ca.

Table 5.2 Range element concentrations (mg kg^{-1}) generally found in soil, plant matter and rain water following Allen et al. (1989) and a number of studies for *Hypnum cupressiforme* for moss (see footnote).

	Soil	Rain water	Plants	Moss
N	1000-5000	0.05-0.8	10 000-30 000	10 400-27 900*
P	200-1500	<0.01 ⁺⁺⁺	500-3000	2800-3500 ^{***}
K	3000-20 000	0.1-1	5000-50 000	5000-8000 ^{**} / ^{***}
S	300-3000	0.4-4	800-5000	1133-2500 ⁺⁺ / ^{***}
Ca	5000-20 000	0.1-3	3000-25 000	2000-20 000 ^{**} / ^{***}
Cu	5-80	0.0002-0.002	2.5-25	2.5-569 ⁺
Fe	5000-100 000	0.005-0.15	40-500	138-3959 ⁺
Mg	2000-20 000	0.1-2	1000-5000	1600-4700 ^{**} / ^{***}
Mn	200-2000	0.0004-0.003	50-1000	38-825 ⁺
Zn	20-300	0.001-0.015	15-100	8.7-786 ⁺
Al	10 000-120 000	0.002-0.1	100-1000	90-5959 ⁺

+ = Carballeira et al. (2008): Elemental ranges of *H. cupressiforme* in five studies in Spain and France.

++ = Bargagli et al. (2002): Elemental concentration of moss in 90 sites in Italy.

+++ = Pfafflin and Ziegler (1992) and Hou et al. (2012)

* = Skudnik et al. (2015): N concentration in 103 forest samples (under canopy) in Slovenia.

** = Rasmussen and Johnsen (1976): Elemental concentrations of moss taken from trees in Denmark.

*** = Sardans and Penuelas (2008): Approx. concentrations of elements in a 6 year study in Catalonia.

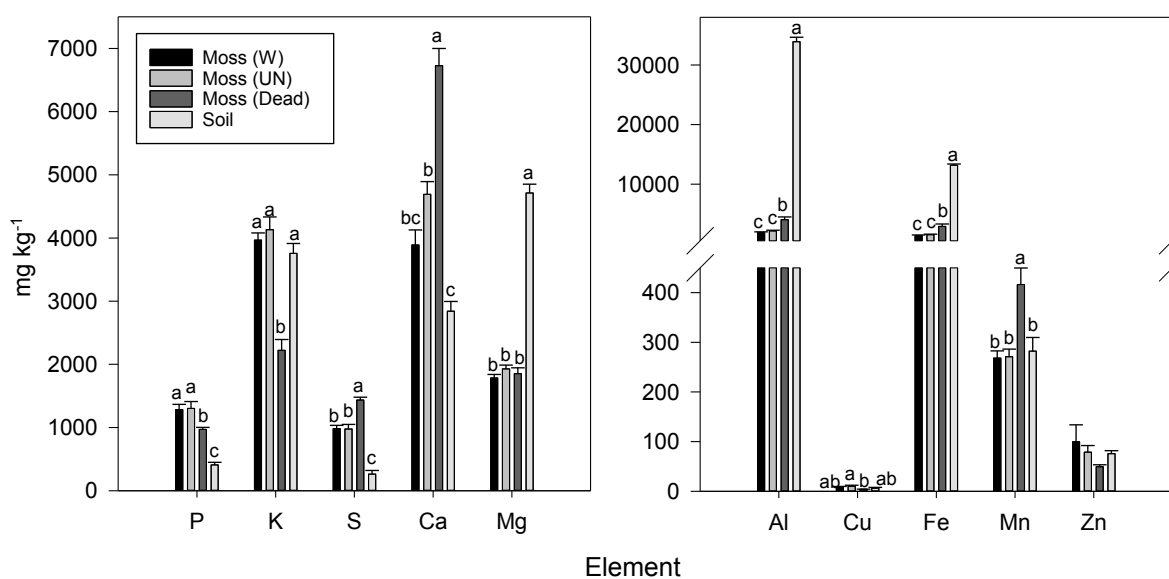


Figure 5.1 Mean (\pm SEM) pseudo-total macronutrient and trace element concentration contained within the soil at Eyrewell ($n=4$), live *Hypnum cupressiforme*, both washed in deionised water (W) and unwashed (UN), and in the dead moss layer ($n=10$). Means that do not share a letter are statistically different (significant to $p<0.05$).

The collection site (either ESR or SBR) had no significant effect on the nutrients within the rain water and throughflow. All concentrations were within the range observed in other studies with the exception of low K and high Cu concentrations in rain water (Table 5.2). Concentrations of elements generally followed kānuka throughflow>moss leachate>rain water apart from P where moss leachate contained the highest concentration, although this was not significant (Figure 5.2). Both copper and iron concentrations were below the detectable limits within the moss leachate.

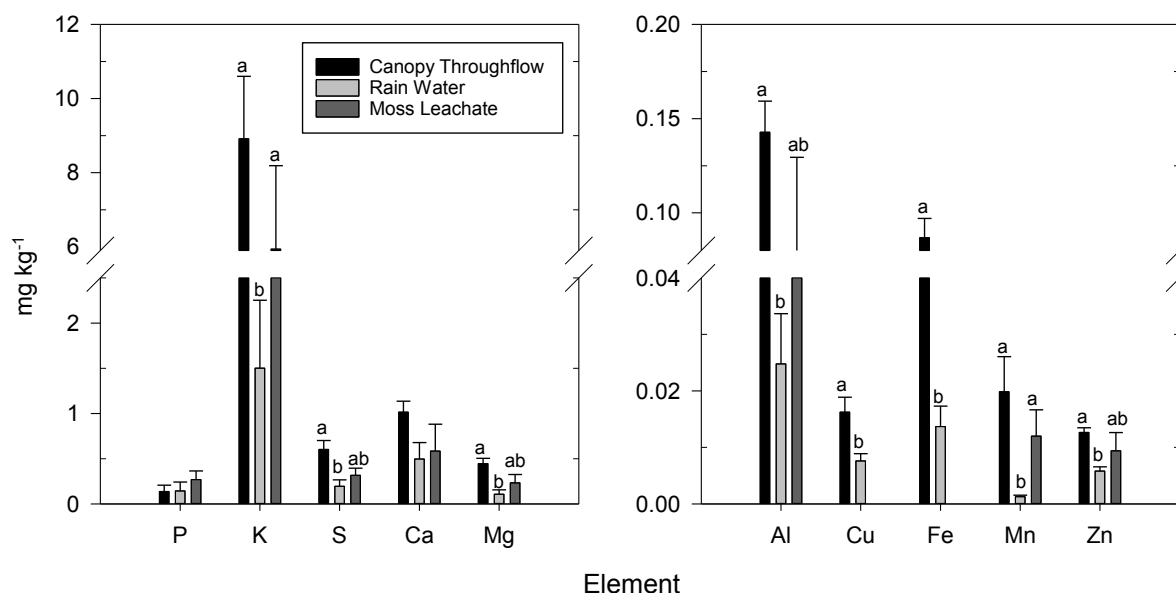


Figure 5.2 Mean (\pm SEM) pseudo-total macronutrient and trace element concentration contained in rainfall ($n=10$), the rain flowing through the kānuka canopy ($n=10$) and within deionised water drained through *Hypnum cupressiforme* turves of 5 cm depth ($n=8$). Means that share a letter are not statistically different (significant to $p<0.05$) and no letters denotes no significant difference. Cu and Fe were below detection limits in the moss leachate.

Available Soil Nitrogen Interaction

Nitrate concentrations were low all cases with a mean high of 3.67 mg L^{-1} in the SBR remnant in summer although within the range for typical soils ($1\text{-}20 \text{ mg L}^{-1}$, as detailed by Allen et al. (1989)) (Figure 5.3). There was a general trend for nitrate concentrations to be highest in bare soil compared to moss cover although the difference was less than 0.7 mg L^{-1} in all instances apart from SBR in summer where the difference was 2.40 mg L^{-1} .

Ammonium concentrations were also within the range expected for typical soils, $2\text{-}30 \text{ mg L}^{-1}$ (Allen et al. 1989), followed a similar pattern with higher concentrations in bare soil than in soil under moss. The difference between the highest and lowest mean concentrations was more visible than observed for nitrate, ranging from $3.01\text{-}11.48 \text{ mg L}^{-1}$.

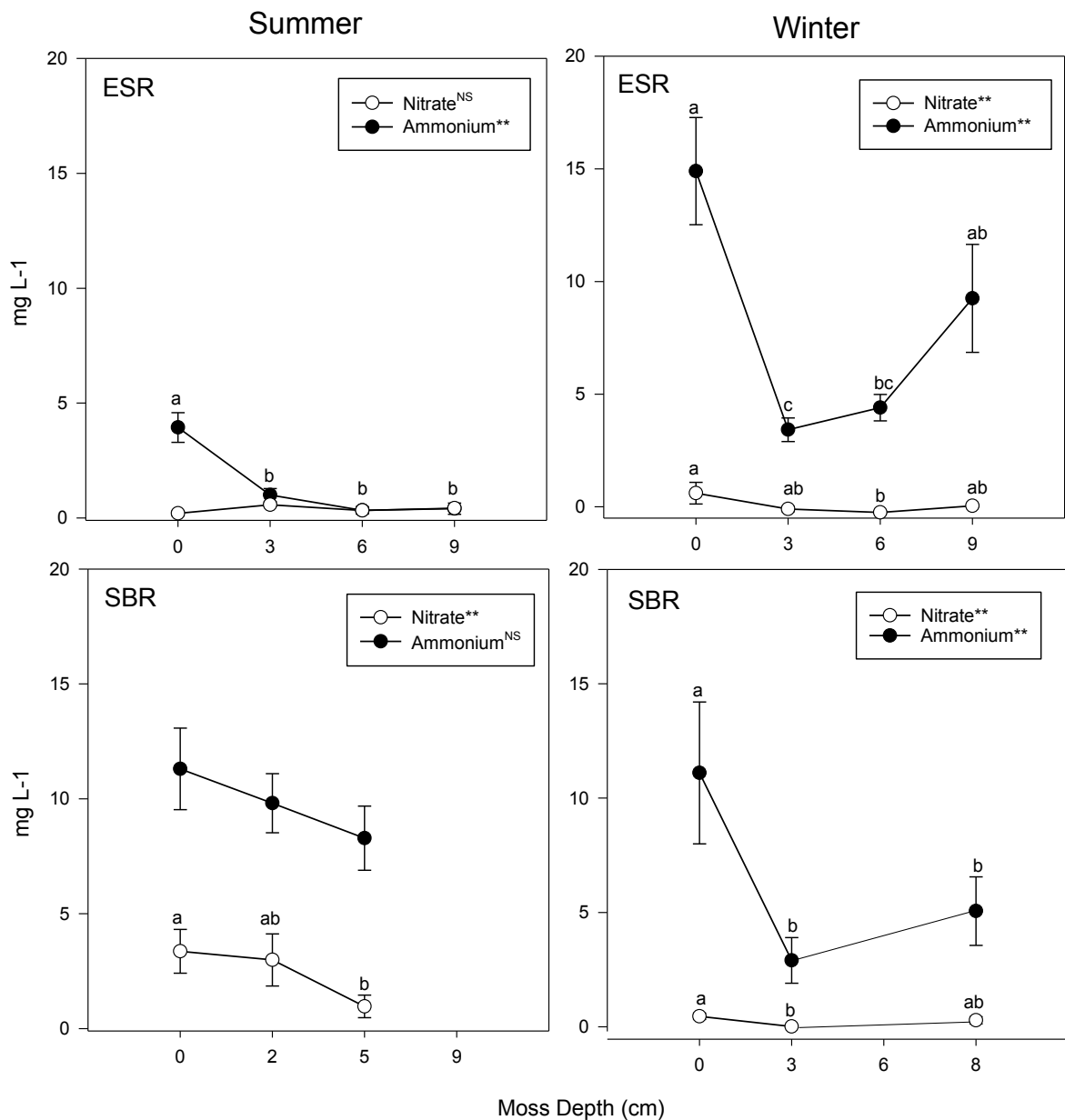


Figure 5.3 Nitrate and ammonium (means (\pm SEM)) within soil samples under different depths of moss from under the kānuka canopies at SBR and ESR during the summer and winter months. Means that are significantly different within either nitrate or ammonium are denoted by $**=p<0.05$. No letters indicates no significant difference (NS).

5.3.2 Biotic Interactions

Laboratory Experiment

Germination and establishment rates for all species were highest on bare soil (Figure 5.4). The total dry biomass for *K. serotina* was low but highest in the bare soil treatment whilst the moss cover increased biomass for *C. australis* and, more visibly and significantly, for *P. amoena* in which robust plants were observed in the 7.5 and 10 cm moss depth treatments. All seeds in the moss treatments

germinated without contact with the soil and observation of rooting revealed that all plants were rooting into the dead moss layer beneath the live moss in addition to the soil (Plate 5.1).

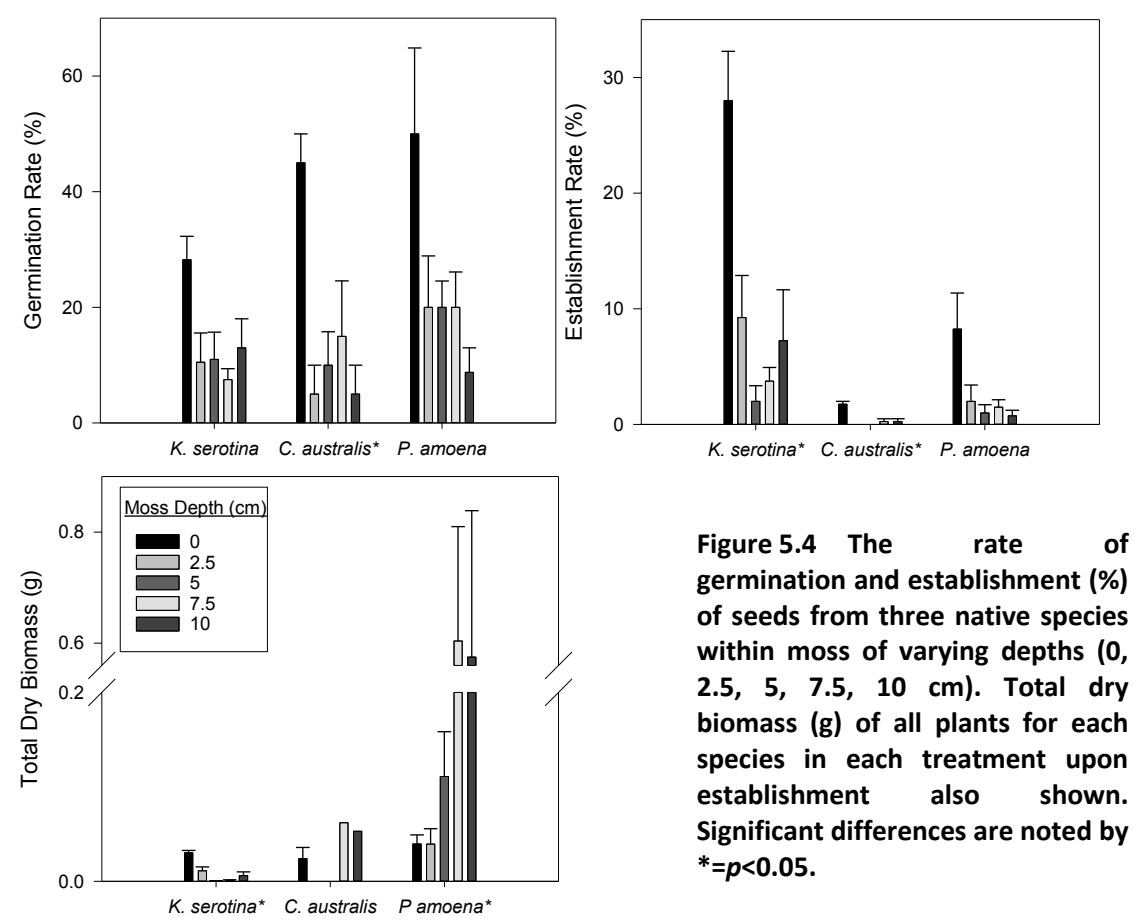


Figure 5.4 The rate of germination and establishment (%) of seeds from three native species within moss of varying depths (0, 2.5, 5, 7.5, 10 cm). Total dry biomass (g) of all plants for each species in each treatment upon establishment also shown. Significant differences are noted by *= $p < 0.05$.

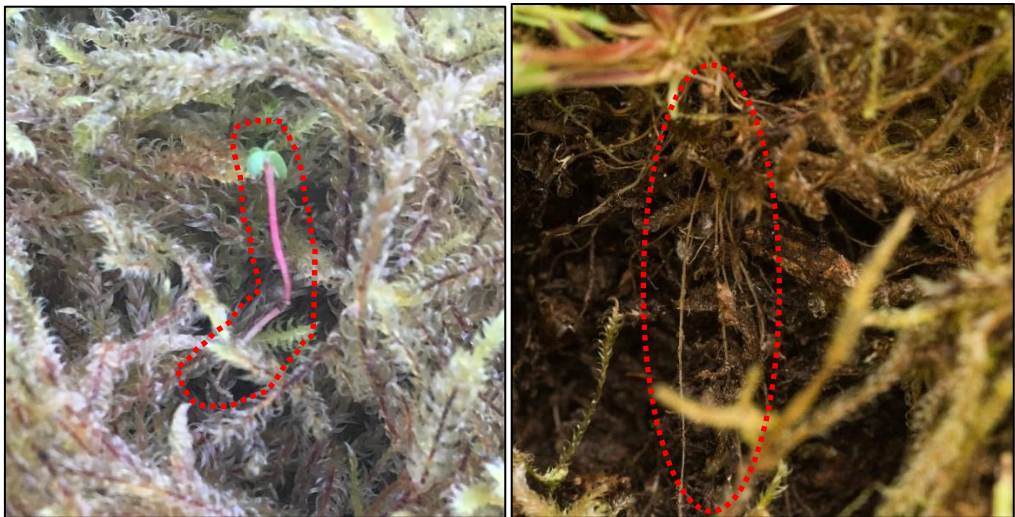


Plate 5.1. Young kānuka plant freshly germinated within moss layer (left) and roots within the dead moss material highlighted within the red ellipse (right).

Soil moisture, ammonium and nitrate concentrations were not significantly different between the treatments but there was a general trend for nitrate to decrease under deeper moss (Table 5.3).

Table 5.3 Soil nitrate and percentage soil moisture within each treatment at the completion of the plant establishment experiment.

Treatment	% Soil Moisture (\pm SE)	NO ₃ ⁻ -N Concentration mg L ⁻¹ (\pm SE)	NH ₄ ⁺ -N Concentration mg L ⁻¹ (\pm SE)
Control	15.57 (5.76)	0.38 (0.19)	3.11 (0.88)
2.5 cm	38.40 (2.67)	0.62 (0.14)	4.90 (1.41)
5 cm	25.68 (9.28)	0.45 (0.25)	3.26 (1.02)
7.5 cm	23.46 (5.38)	0.19 (0.06)	2.19 (0.67)
10 cm	26.86 (8.07)	0.17 (0.09)	2.73 (0.92)

Field Experiment

None of the seeds within the field experiment germinated. This was most probably due to the extremely dry winter and spring that was experienced in Canterbury in 2015/2016.

5.4 Discussion

5.4.1 *Hypnum cupressiforme* Chemistry

H. cupressiforme has been used as an environmental pollution monitoring species due to its ability to bioaccumulate heavy metals (Proctor 2009; Ward et al. 1977). The results of the present study show elemental concentrations in moss tissues within the ranges reported for the same species elsewhere, with the exception of potassium and phosphorus which were both lower (Table 5.2).

The live moss plant parts contained higher concentrations of P, K and Mg than the dead moss as would be expected for elements required for metabolic function (Bates 2009). As such, *H. cupressiforme* has been shown to translocate P effectively to the growing tips (Bates 1994; Phuyal et al. 2008); a process also known for Mg in other mosses (Brūmelis et al. 2000). Phosphorus is known to be absorbed by moss plants with canopy leachates being an important source as evidenced by mosses increasing growth under canopy and the increased nutrient concentration of canopy throughfall and stemflow (Farmer et al. 1991; Skudnik et al. 2015; Tamm 1953). Wells and Boddy (1995) reported that, in addition that from deposition, uptake of phosphorus could also be achieved from substrate through associations with saprophytic basidiomycete. However, the P and K concentrations recorded in the present study are lower than those cited for *H. cupressiforme* in other studies (Table 5.2). Elemental concentrations within moss tissue can be variable and Fernández et al. (2010) showed the range of K to be 3 650 - 10 263 mg kg⁻¹ in *Pseudoscleropodium purum* due to the moss being sampled under different conditions with regard to season, site and treatment of sample (dried, frozen, acclimatised). Furthermore, K and P are known to have a high capacity for leaching from washed samples and periods of desiccation can alter element concentrations (Sardans and Penuelas 2008; Smith 1978; Tamm 1953).

The dead moss layer is known to bind cations (Brown and Bates 1990; Giordano et al. 2009) and in the present study it contained the highest concentrations of Ca, SO_4^{2-} and Mn. Calcium concentrations differ with age of tissues and tend to increase in older tissues possibly reflecting the content of the cell walls (Bates 1979; Tamm 1953). Copper was highest in the unwashed live moss and the lowest in the dead moss possibly reflecting both metabolic requirement and dust build up on the surface (Glime 2015). The soil also had similar concentrations for Cu, 6.3 mg kg^{-1} (± 1.5), although this was low for typical soils (Table 5.2) and lower than the reported average of 9 mg kg^{-1} by Havlin et al. (1999). The concentration of Zn was not significantly different between plant parts and soil. The soil concentration was slightly higher than previously reported for the Lismore series (10 mg kg^{-1}) but almost 50% of this figure is stored in the residual fraction of the soil and completely unavailable to plants (Chowdhury et al. 1997). Unsurprisingly, soil had the highest levels of Fe and Al, both very abundant minerals yet often found in unavailable forms to plants (Kear et al. 1967; McLaren and Cameron 1996).

The Al and Fe concentrations in Canterbury *H. cupressiforme* were reportedly higher than those recorded in moss samples from Turkey (Koz et al. 2012) and Fe, Mn and Zn was higher than concentrations recorded in the vicinity of mining sites in Italy (Bargagli et al. 2002). The wide range of concentrations recorded in moss species must be compared with care due to a number of factors affecting the result: sampling strategy; species specific seasonal variation; sampling handling and preparation; environmental condition of the site; site location with regard to altitude and proximity to the sea (Fernández et al. 2015). Within Canterbury, the difference in concentrations could be partly attributed to the lower rainfall on the Canterbury Plains (not washing deposited elements from plants and into the soil as often as areas of high rainfall), the low soil pH and, hence, exchangeable Al (Whitley et al. 2016) and chelation of other elements within the Lismore soils, and the significant atmospheric deposition Zn, and to a lesser extent Cu, in New Zealand (Gray et al. 2003; Halstead et al. 2000). Sampling strategy and handling may have impacted results due to the samples only being collected in one season (not accounting for season variation), washed samples being subjected to running water for between 1-5 minutes allowing leaching of nutrients (Smith 1978) and, whilst care was taken to ensure the moss fragments were processed rapidly, samples may have been subjected to desiccation which can underestimate elemental concentrations (Fernández et al. 2010).

5.4.2 Nutrient transfer to Soil

Throughflow

The moss leachate was acquired using deionised water. This is not truly reflective of the field environment where the rain water would contain nutrients and be acidic. The hydrogen ions of rainfall in New Zealand has been recorded between 6.0 and 6.7 (Gray et al. 2003), with 6.1 pH for Christchurch

(Fergusson and Stewart 1992) and 6.3 recorded in Westland, South Island (Cox et al. 2015). Our laboratory measurements, following standard methodologies, recorded a pH of 6.0 (McLean 1982), adequately reflecting the pH of Christchurch rainfall. Due to the deionised water not containing the nutrients found in rainwater or canopy throughfall there can be no calculation of nutrient interception from wet deposition by moss and the subsequent prevention of transfer to the soil and rooting zone as described elsewhere (Oechel and Van Cleve 1986). However, the results stated here show the ions contained in the canopy drip, rainwater and moss leachate (without nutrient input from other sources) reaching the bare soil or moss carpet for potential uptake.

The concentration of elements in rainwater, kānuka throughfall and moss leachate was minimal being less than 1 mg L⁻¹, with the exception of potassium, indicating that each rain event does not transfer high concentrations of elements into the soil from any of these pathways. Canopy interception and throughfall of rainfall varies with regard to the intensity and duration (Rutter 1975) and it has been estimated that rainfall of 10-30 mm is required to ensure percolation to the forest floor (Geiger et al. 2009). Taking a more conservative approach and assuming that kānuka and moss throughfall would only occur at rainfall amounts higher than 5 mm day⁻¹, for the year August 2015-2016 there would have been 24 days of throughfall, a relatively low number (Table 5.4). Despite, previously discussed, higher atmospheric deposition Zn, and to a lesser extent CU (Gray et al. 2003), concentrations remain low in Lismore soils (Haynes 1997).

Table 5.4 Annual rain events and amounts from August 2015-2016 in Canterbury. Data acquired from New Zealand's National Climate Database, CliFlow for the Rangiora EWS weather station (NIWA 2017).

Total Rain Days	106
Max. rain (mm day ⁻¹)	30.8
No. days rainfall > 20 mm	5
No. days rainfall 5-20 mm	19
No. days rainfall 1-5 mm	39
No. days rainfall 0.5-1 mm	16
No. days rainfall 0-0.5 mm	27

The concentration of most elements leached out of the moss was lower than those in the kānuka throughfall solution but higher than the rainfall concentrations. The exception was Al, Fe and Cu which were not detectable within the moss leachate. It is known that soils are often deficient in Cu, Zn, Mn and Fe in New Zealand (Condon et al. 2000; Haynes 1997; Will 1990). The present results suggest that *H. cupressiforme* may further exacerbate this deficiency by retaining these elements nutrients and possibly recycling them within old and new tissues, preventing transfer into the soil (Wells and Brown 1996).

Potassium was the element with the highest concentration reflecting its mobility and easily leached status (Aerts and Chapin 1999). In addition, P concentration within moss leachate was higher than rain or canopy throughflow despite the element known to be effectively recycled within tissues (Bates 1994). Mosses are known to leak nutrients from cells upon rewetting following periods of desiccation often being absorbed by neighbouring cells (Bates 1997; Coxson 1991; Gupta 1977; Wilson and Coxson 1999). It is possible the moss used in this experiment was subjected to water stress and released a pulse of nutrients into the leachate. This is supported by Smith (1978) who noted that running water over moss tissues can cause the leaching of both K and P. This may be an example nutrient transfer from moss to vascular plants as soluble P is available for uptake, especially important in P limited forests (Vitousek et al. 2010). However, soluble P can also undergo P-sorption onto Al and Fe oxides which are plentiful in the Lismore soil series found at Eyrewell which would prevent uptake (Holford 1997). Regardless, the amount in the moss leachate is inconsequential compared to the typical uptake of 20-30 kg P ha⁻¹ of wheat to produce 8000 kg grain (Yli-Halla 2016).

Decomposition

Another pathway of nutrient transfer from plant to soil is decomposition. In vascular plants decomposition plays an important role in delivering nutrients back to the soil. In cooler habitats, mosses can be used as a food source by large mammals such as reindeer (Van der Wal and Brooker 2004), but these are absent in the kānuka dryland shrublands. Insect herbivores can accelerate decomposition rates (Chapman et al. 2003) but there are limited insect herbivores of mosses thought to be a function of chemical deterrents within tissues (Gerson 1982; Glime 2006). As a result, mosses undergo slow microbial decomposition, often via the action of fungi (Glime 2015).

In mosses, although they do not contain lignin (associated with slow decomposition rates (Berg and McClaugherty 2003; McLaren and Cameron 1996)) the litter can be highly recalcitrant making it resistant to decomposition (Coxson and Nadkarni 1995; Hobbie 1996). Bryophytes exhibit longer decomposition periods than either lichens or vascular plants (Lang et al. 2009). This provides a short-term nutrient sink in mosses which retain nutrients and release them once the slow process of decomposition allows (Rousk et al. 2014; Weber and Cleve 1984). In an Antarctic study, all soil nutrients were lower under moss cover than under vascular plant cover (Roberts et al. 2009). However, although most research indicates that moss decomposition is slow the majority of these studies have been carried out in polar, boreal or bog/fen habitats with little research undertaken in temperate forests (Brown and Bates 1990; Hobbie 1996; Lang et al. 2009; Moore et al. 2007; Nakatsubo et al. 1997). Temperature is an important aspect of decomposition and rates may be faster in temperate zones (Nakatsubo et al. 1997) although still likely to be reduced when compared to vascular plants due to the chemical composition of the mosses, such as the lignin-like structural compounds and soluble phenolics (Bates 2009; Lang et al. 2009).

Impact on the Nitrogen Cycle

A major constraint of the current study is the lack of N analysis of moss litter or leachate. However, the present study observed a trend for reduced soil nitrate and ammonium concentrations under moss than in bare soil intimating litter transfer from plant to soil. In cooler climates this is in part attributed to the effect of the moss cover on abiotic factors such as temperature and moisture, in turn affecting the microbes which facilitate ammonification and nitrification (Gornall et al. 2007). In Canterbury, there is a temperature variation in soil under moss between seasons but the soil temperature is held between approximately 8-15°C with 5-35% gravimetric soil moisture throughout the year (see Chapter 4). Considering nitrification is inhibited below 5°C, optimal between 25-30°C, and limited by waterlogging but still viable after permanent wilting point (Haynes 2012) these parameters are not thought to exert significant influence over soil microbes. The main process governing available soil nitrogen in this study is thought to be interception and use by moss plants.

H. cupressiforme extracts have anti-microbial properties (Altuner et al. 2014; Çolak et al. 2011) although no studies have directly tested effects of the extracts on mineralising or nitrifying bacteria. Walters (2001) found that there was little microbial activity when investigating mineralisation of *H. cupressiforme* litter for 180 days with rates of 1.7-3.9 % N mineralised in dead and live plant matter in two different temperature conditions. However, the amount of N as nitrate following 180 days at 25°C was high, up to 94%, indicating that nitrifying bacteria activity was prolific. The soil temperature in the present study would not be so high and Walters (2001) for 5°C produced no NO₃⁻ after 180 days. In contrast, Sedia and Ehrenfeld (2005) found increased ammonification and less nitrification in incubated soils contained an extract from *Polytrichum* spp. moss when compared to control soil. They concluded that this was most probably due to an inhibitory effect of the moss extract on the microbes but noted that further research would be required to establish the link. In comparison to the present study, the reduced nitrate concentrations could have resulted from effects on the microbe community but the scope of the study cannot substantiate this.

H. cupressiforme is thought to assimilate ammonium more readily than nitrate (Brown 1992). In the present study, reduced soil ammonium levels in the winter months may be explained by the metabolic activity of the moss. *H. cupressiforme* and other mosses have been shown to be effective at absorbing nutrients, including NH₄⁺ in significant quantities, from wet and dry deposition, acting as a barrier to the soil and the rooting zone (Rasmussen and Johnsen 1976; Van Tooren et al. 1990). The difference between ammonium levels in summer were only visible at the ESR site and were not as large as those observed in the winter months, whereas the nitrate was only significantly reduced at the SBR site. This may be due to the reduction of photosynthesis in moss during this warmer, drier period.

Soil ammonium was most reduced beneath moss at intermediate depths of 3 cm where it is possible that lack of the thick, dead plant layer underneath the moss allows for NH_4^+ uptake from the soil in addition to that deposited on the surface (Ayres et al. 2006; Liu et al. 2013; Wang et al. 2014a). An increase of moss depth and associated dead moss layer increases the distance of active growing shoots from the soil surface making soil NH_4^+ unattainable which potentially explains the increase in ammonium concentrations beneath deeper moss depths. Similarly, Bates (1994) found that *Pseudoscleropodium purum*, a moss of acidic environments which generally grows on a mat of its own litter similar to *H. cupressiforme*, had effective mechanisms for retaining N and P within cells and relied on wet and dry deposition due to the lack of connectivity of live moss tissue with soil.

Soil microbes transform ammonium into nitrate and therefore, a reduction in NH_4^+ would be expected to lead to a reduction in NO_3^- levels. Nitrate concentrations were significantly lower under deeper moss in each site and season, with the exception of the summer month in ESR. It is assumed that the moss carpet is influencing the nitrate soil levels but the reduction in winter and lack of difference in summer could also be due to higher rainfall and more leaching in those months (Di and Cameron 2002). However, this is unlikely due to moss intercepting moisture from rainfall as demonstrated by the drier soil under deeper moss during winter months (see Chapter 4).

In addition to preventing NH_4^+ , and therefore NO_3^- , reaching the soil by intercepting and utilising this nutrient resource, mosses are also thought to sequester N for long periods in tissues (Oechel and Van Cleve 1986). Mosses are effective in their use of N, recycling it within tissues and therefore delaying the amount of time it takes to release them into the soil and vascular plant rooting zone below (Eckstein and Karlsson 1999; Turetsky 2003). A study of feathermosses within a boreal forest found that mosses had been sequestering nitrogen at $1.8 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for the past 5000 years (Lagerström et al. 2007).

5.4.3 Influence of Moss on Vascular Plants

In the glasshouse experiment, a moss cover negatively affected germination of all species. More seedlings germinated and became established on bare soil. Dormancy of seeds and inhibition of germination can be a response to far-red light conditions altered by the moss cover thereby inhibiting germination (Van Tooren 1990; Van Tooren and Pons 1988). Both *K. serotina* and *P. amoena* do not germinate as effectively in dark conditions (demonstrated in Chapter 3). The small seeds of each species were observed to drop through the moss carpet following initial sowing, limiting the exposure to light levels and possibly preventing germination. *C. australis* was shown to germinate in dark conditions and when buried in 2 cm soil (Chapter 3) implying that the response to light does not trigger dormancy. However, the germination rates for this species were lower in the moss treatments. Allelopathic substances in some mosses can inhibit germination (Michel et al. 2011a; Zamfir 2000).

Hypnum plumaeforme is thought to possess this trait but it is not reported in *H. cupressiforme* and the simplicity of the current study cannot determine such effects (Kato-Noguchi et al. 2009; Zamfir 2000). Steijlen et al. (1995) found that Scots Pine seed germination and plant survival in the field was influenced by moisture and the allelopathic effects of *Pleurozium schreberi*, with the moss acting as a barrier to nutrients.

The field experiment in the present study produced no seedlings, explained by drought conditions during the experiment drought (NIWA 2017) which was exacerbated by the act of disturbing the free draining, droughty Lismore soil. The lack of water may have been further exacerbated by the desiccation of the moss layer in the summer months. Jeschke and Kiehl (2008) found the most significant danger to young plants was desiccation and high temperatures. However, in some water stressed environments the moisture retained within the moss cover can aid seed germination and plant establishment (Gornall et al. 2011; Morgan 2006; Parker et al. 1997). This did not appear to be the case in the present study.

The lack of seedling establishment within *H. cupressiforme* may be due to the dense mats acting as a barrier and preventing the radicle from reaching the soil (Jeschke and Kiehl 2008; Leemans 1991; Sedia and Ehrenfeld 2003; Van Tooren and Pons 1988). In addition, the interwoven mat of the moss carpet may prevent emergence due to the morphology of the dicotyledonous, trapping the seedling (Hörnberg et al. 1997; Zamfir 2000). Casanova-Katny and Cavieres (2012) found that moss facilitated germination but not survival of an Antarctic grass species due to an interplay of effects including moisture, temperature and nutrients. In New Zealand, Allen et al. (2003b) found that the moss/ litter layer facilitated germination of *Chionochloa macra* grass seeds when buried to 1cm when compared germination rates in *Poa* grass turf, mainly due to microclimate and *Poa* grass competition effects.

Germination and establishment rates were lower within the moss treatments but, for those *C. australis* and *P. amoena* plants that survived, biomass was higher. Therefore, if the seedlings were able to root into the soil there were benefits from being within the moss substrate (Plate 5.2). Mosses are known to provide microclimates that can provide shelter for vascular plants, with some species relying on the presence of the moss for survival (Ren et al. 2010). It is speculated that less fluctuation in soil moisture under the moss layer influenced growth. In addition, soil conducts heat more effectively than air (McLaren and Cameron 1996) and due to the moss being primarily filled with air pockets whilst the soil contains water, the heat transfer and fluctuation would be more pronounced in the control treatment.

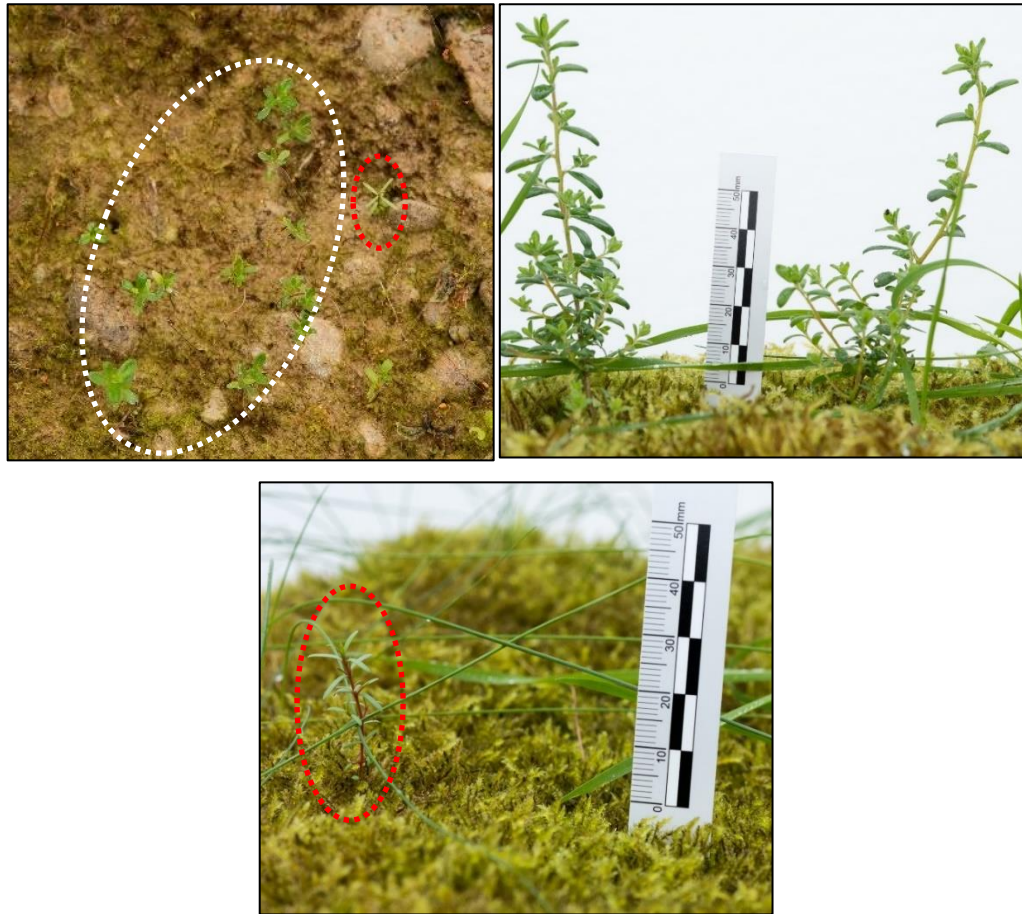


Plate 5.2. *K. serotina* (inside the red elipse) and *P. amoena* (inside the white elipse) seedlings within the bare soil treatment (top left) compared to the larger, more robust plants found within the moss treatments (*P. amoena* top right and *K. serotina* lower).

Supporting the idea that soil under moss contained less nitrogen was the nodulation behaviour of *C. australis*. *C. australis* is a nitrogen-fixing plant and produces nodules containing rhizobia (*Mesorhizobium* spp.) (McCallum 1996). Only plants rooting in the moss treatments produced nodules (Plate 5.3). It is noted that nitrate can be inhibitive to nodulation (Brewin 1991; Carroll and Mathews 1990) implying that the plants within the moss treatments were nitrogen limited and therefore produced nodules. This may explain the biomass decrease in *K. serotina* growing within moss. Studies have reported an increase in biomass with N application (Esperschuetz et al. 2017a), suggesting that N is a limiting nutrient for the species. The soil under deep moss layers in the present study contained less available N than bare soil possibly resulting in decreased growth.



Plate 5.3. *C. australis* with root nodules (red dotted circle) which was planted in soil covered with 7.5 cm moss (left) and another plant of the same species rooting in bare soil with no nodule development (right).

5.4.4 Conclusions

The findings of the present study have illustrated that the *H. cupressiforme* layer within Eyrewell forest comprises a suite of chemical elements which are sequestered within decaying matter and are not readily leached or rapidly decomposed, restricting transfer to the soil below (Oechel and Van Cleve 1986; Pouliot et al. 2009). The use of available nitrogen by *H. cupressiforme* and prevention of this element reaching the soil caused sufficient N deficiency to promote a nodulation response in *C. australis*. This alteration of soil nutrition may indirectly shape soil microbial communities (Delgado-Baquerizo et al. 2017) and there are suggestions that *H. cupressiforme* may have plant extracts which further modify microbial communities. Microbial communities are central to biogeochemical cycles (Philippot et al. 2013), by the action on these communities, moss exerts further importance on ecosystem functioning. This requires further study in the context of the present study.

Despite the decrease in soil nutrition, plants which were able to root in the soil below the moss layer experienced increased growth, probably due to the moisture and temperature buffering effects of the moss carpet. *H. cupressiforme* acts as a barrier to the soil, preventing seeds of vascular plants from germinating and establishing. However, for plants which are not limited by nitrogen, the beneficial effects of the moss carpet on abiotic conditions may produce increased biomass in those individuals that do manage to establish.

The soils within the Eyrewell Forest region are generally nutrient poor (Landcare Research 2014b; Leathwick et al. 2003; McLaren and Cameron 1996) and many native plants are well adapted to this (Craine and Lee 2003; Lee et al. 2001). Conversely, the exotic species which surround the native remnants are adapted to more fertile soils (Meurk and Swaffield 2000) and probably experience a competitive disadvantage with deep moss layers. The moss layer may provide an effective tool in preventing spread of invasive plants by retaining a nutrient poor substrate, even in the presence increased nutrient deposition.

In conclusion, moss exerts a significant influence on nutrient cycling and biotic interactions in kānuka shrubland communities. Combined with the conclusions of Chapter 4 on its influence on the hydrological cycle, this provides evidence to suggest that mosses should be incorporated into ecological restoration and additional functional studies of dryland kānuka ecosystems.

Chapter 6

Effects of Enhanced Nutrition on Dryland Kānuka Shrubland Communities

6.1 Introduction

Native New Zealand vascular flora, although of rather limited species richness at c.2500 species (Wilton and Breitwieser 2000), is highly specialised with endemism of approximately 85% (Wardle 1991). It is thought that this high level of speciation occurred due to a number of factors including lengthy isolation, geological and climatic fluctuations, and co-evolution with flightless birds and lizards in place of mammals elsewhere (Cooper and Millener 1993; Lee et al. 2001; McGlone et al. 2001; Waters and Craw 2006). New Zealand is a global biodiversity hotspot, with high values placed on the importance of biological conservation (Kier et al. 2009; Mittermeier et al. 2005). As a consequence, New Zealand has many protected natural areas but these are mostly in mountainous, wet, steep or cool environments and land outside of this has gained limited protection leading to impoverished indigenous areas in the drier, flatter, warmer environments (Cieraad et al. 2015b; Holdaway et al. 2012). This is exemplified on the Canterbury Plains, South Island, where less than 0.1% native habitat remains in some areas and what is surviving is highly fragmented (Environment Canterbury 2008b; Harding et al. 2009; Meurk et al. 1995; Winterbourn et al. 2008).

Diversity is also reflected in New Zealand soils which contain 15 orders and approximately 2000 soil series' (Hewitt 1992; Hewitt 2010). Naturally fertile soils are limited (Leathwick 2002) and much of the soil resource has been leached of nutrients through weathering (McLaren and Cameron 1996). Therefore, it is suggested that New Zealand plants have evolutionary adaptations to tolerate nutrient poor substrates (Craine and Lee 2003; Lee et al. 2001). In support of this, studies have found that increased soil nutrition does not lead to increased growth rates in some native species (Caroline 1999; Franklin 2014; Ogle 1996; Peltzer et al. 2016) and nutrient enrichment can lead to a decline in indigenous flora (Bowie et al. 2016; Ecroyd and Brockerhoff 2005). However, the response may be highly species-specific as shown by experiments under controlled growing conditions (Carswell et al. 2005; Esperschuetz et al. 2017a).

In stark contrast to naturally nutrient poor soils, there has been a steady increase in artificially fertile land. Productive land, particularly associated with agriculture and horticulture, currently covers approximately 42% of New Zealand (Ministry for the Environment & Statistics New Zealand 2015). Pasture alone amounts to approximately 40% of land cover: 95% of productive land. Plant species used in the pasture mix invariably includes grasses which are exotic with perennial ryegrass (*Lolium perenne*)

being the most widespread and abundant, in association with leguminous herbs and other grasses in areas of lesser fertility (Charlton and Stewart 1999; Lee et al. 2012). To enable high productivity, fertiliser use has been essential (Di and Cameron 2002). Nitrogen fertiliser use has increased tenfold since 1985 with granular urea being applied most commonly (Ministry for the Environment 2007; Quin et al. 2015). Phosphorus additions have increased due to superphosphate and cattle effluent (Jenkins and Jenkins 2012). The increased soil surplus of both nitrogen and phosphorus was estimated to have increased by 41% and 128% respectively in the years 1990-2004 (Statistics New Zealand 2009). This has led to a high vulnerability of loss from the soil through leaching, surface runoff and volatilisation. These highly amended pasture systems pose significant eutrophication risk to natural terrestrial and freshwater environments, especially to fragmented areas of indigenous habitats containing plants adapted to low nutrient environments located adjacently (Chobtang et al. 2017; Didham et al. 2015). Unintentionally, fertilisers can also be spread directly into adjacent habitats either by aerial drift, water runoff or misplacement (Duncan et al. 2008). Tsiouris and Marshall (1998) found that commonly used methods of spreading granular fertiliser, such as disc spreaders, can lead to misplacement in the landscape. This is supported by numerous studies documenting effects on native floral and faunal composition in boundary vegetation (Didham et al. 2015; Kleijn and Snoeiijing 1997; Kleijn and Verbeek 2000; Marshall and Moonen 2002; Maudsley 2000; Willi et al. 2005).

A disparity exists between the need to protect native flora and fauna of New Zealand which is based on low nutrient soils and the requirement to maintain productive land which is often intensive, high nutrient farming. To enable protection and enhancement of indigenous habitats, the ecology of native plants, including the response to increased nutrients, and the biotic interactions need to be understood (McDonald et al. 2016). Whilst enhanced soil nutrients may inhibit growth of some native plants, nutrient-stressed environments and nutrient deficiency in plants can be the source of failure in restoration projects (Bradshaw and Chadwick 1980). Similarly, the effects of increased nutrients on the establishment of exotic flora, and the possible consequences of invasion and competition into native habitats, should be considered (D'Antonio and Meyerson 2002; Kimball et al. 2016).

The research reported in the present chapter aimed to consider the response of native plants and communities to nutrient enrichment in this nutrient poor environment. The plants chosen were those xerophytic species which are main woody associates within the kānuka shrubland and the most common non-vascular plant. I investigated responses of selected species to increases in soil pH, nitrogen and phosphorus in a controlled environment. This was an attempt to evaluate the effects of potential dispersal of agriculturally applied fertilisers into adjacent indigenous landscapes and the effect on the establishment, growth and sustainability of both native and non-native species.

6.2 Methodology

6.2.1 Vascular Plants and Elevated Soil Nutrients

The effect of fertilisers on native vascular plants was investigated in a glasshouse experiment. Thirty-five specimens of plains kānuka (*Kunzea serotina*), Banks Peninsula kānuka (*Kunzea robusta*), New Zealand broom (*Carmichaelia australis*), prickly mingimingi (*Leptecophylla juniperina*) and pomaderris (*Pomaderris amoena*) of equal age (approximately one year) were sourced from local nurseries; the Department of Conservation Motukarara nursery and Matai Nurseries, Waimate. Plants had been grown from seed sourced from the Canterbury Plains except for *K. robusta* seed which was collected from Banks Peninsula, Canterbury. Each plant specimen was carefully removed from the original pot and cleaned of potting mix as far as possible without damaging the roots. The planting medium was soil collected under the pines at Eyrewell Forest. Soil physico-chemistry is described in Chapter 3. Seven soil amendment treatments, based on nutrient quantities applied to Eyrewell Forest soils to enable forest-pasture conversion, were introduced with 5 replicates (Table 6.1). Lime and phosphorus treatments were mixed into the soil prior to potting the plants. Nitrogen was applied in two applications, two and four weeks after planting. Plants were repotted into the medium in 2.5 L pots for *Kunzea* spp., 2 L pots for *P. amoena* and 1.5 L pots for *L. juniperina* and *C. australis*.

Table 6.1 Treatments applied to pots in the elevated nutrient enrichment experiment. Application rates were adjusted to the surface area of the pots.

Treatment	Treatment Code	Chemical Application
Control	C	None
Low nitrogen	N1	300 kg ha ⁻¹ urea (NH ₂ CONH ₂)
High nitrogen	N2	600 kg ha ⁻¹ urea (NH ₂ CONH ₂)
Lime	L	6000 kg ha ⁻¹
Phosphorus	P	470 kg ha ⁻¹ Superphosphate
Low nitrogen & lime	NL	As above
Low nitrogen, lime & phosphorus	NPL	As above

The plants were grown for nine months (June 2014 – February 2015) situated in a complete randomised block and watered as required. On completion of the trial, the above and below ground plant biomass was harvested. Roots were washed thoroughly, and above and below ground biomass was dried for 72 h at 60°C. Both fresh and dry biomass data were collected but only dry biomass data are reported. Soil samples from each pot were collected (approximately 100 g) and immediately sieved to 4 mm and analysed for gravimetric moisture content. Soils were stored below 5°C for less than a week before being analysed for KCl extractable ammonium and nitrate. The remainder of the soil was

dried for 3 days and pH and Olsen P were determined following standard methodologies (see Chapter 3 for in depth procedures following (Blakemore et al. 1987).

Data Analysis

Mean values (\pm SEM) for pH, Olsen P, NO_3^- and NH_4^+ were calculated to determine the nutrient status in each treatment at the end of the experiment using SigmaPlot 12.3 (Systat Software, San Jose, CA). Two-way ANOVA was used to assess the effect of treatment, species and their interaction on soil parameters. To assess the difference between treatments on total biomass of each plant species, a One-Way ANOVA with post-hoc Tukey HSD was used. Data were transformed prior to analysis to fit the normality assumptions of the test where appropriate. Analysis was undertaken using Minitab® (Version 17.2.1., Minitab Inc., Sydney, AUS).

6.2.2 Moss and Nutrient Exposure

Glasshouse Trial

A glasshouse trial was set up to investigate the effect of commonly used agricultural soil amendments which can potentially be windblown or unintentionally spread onto communities of bryophytes. Turves of moss, comprising more than 95% *Hypnum cupressiforme*, were collected from sites to be cleared within Eyrewell Forest and cut into 5 cm squares of 2 cm depth. Each turve was placed on a 2 cm layer of soil collected from the pine forest and located within a mist chamber situated within a glasshouse and automatically watered for 10 seconds every 2.5 h (full description in Chapter 4, Plate 4.1). The turves were left for ten days to acclimatise before receiving one of eight treatments containing lime or superphosphate, with 5 replicates of each (Table 6.2). The chemical additions were applied simulating windblown residues from neighbouring pastures by sprinkling through a sieve directly onto the moss. The treatment amounts were based on the quantities of chemicals applied to the surrounding dairy farms at Eyrewell with levels of lime between 3-6 t lime ha⁻¹ and 300-500 kg superphosphate ha⁻¹ and scaled down to the area of each moss turf.

The position of the turves within the glasshouse was randomly re-arranged on a weekly basis to ensure all environmental conditions were encountered with no location effects. Turves were maintained for six months during the spring and summer months (September 2015 to March 2016). Turf volume was measured every second month, from the measurements of height, width and length, and species composition, cover and health were noted. On completion of the experiment fresh volume of the turves was calculated before the moss samples were gently cleaned of soil and dried at 60°C for 72 h to gain the dry biomass of each turve.

Table 6.2 Treatments applied to moss turves within the elevated nutrient experiment (kg ha^{-1}).

Treatment (Code)	Quantity
Control (C)	None
Low Lime (LL)	1500
Medium Lime (ML)	3000
High Lime (HL)	6000
Low Lime / low P (L&P)	1500 / 117.5
Low P (LP)	117.5
Medium P (MP)	235
High P (HP)	470

Field Trial

To investigate the potential spread and effect of nutrients from adjacent farmland into indigenous habitats, 2 remnant kānuka habitats were sampled during December-January, 2016; DOC Eyrewell Scientific Reserve (ESR), surrounded by irrigated dairy pasture since the 1990's, and Spencer-Bower remnant (SBR), adjacent to pine forest to the north and converted to irrigated dairy pasture to the south in 2014 (see Chapter 3 for full description). Within each remnant, 6 transects of 80 m were identified starting at the fenceline, adjacent to the pasture, and towards the opposite edge (south-north, centre point approximately 50 m). Measurements were taken at the fenceline and at 10 m intervals. At each sampling point plant data, for vascular and non-vascular plants, was collected using a 1 m² quadrat, recording species composition and percentage cover. Three soil cores (2.5 cm diameter and 7.5 cm depth) were sampled and bulked from each quadrat. Soil samples were analysed for soil moisture, pH, Olsen P and KCl extractable N following standard methodologies defined in Chapter 3.

Data Analysis

Treatment effect on dry biomass of moss turves in the glasshouse was analysed using one-way ANOVA. Mean growth (both volume and dry biomass) were calculated (\pm SEM) for intervals throughout the experiment, and plotted using SigmaPlot 12.3. For the field experiment, data were amended for outliers and correlation between the distance from the fenceline into the centre of the remnants (50m) and the biotic and abiotic parameters in the field trial was defined by a simple linear regression, using Pearson's Correlation (r) to define the relationship. One-way ANOVA (with post-hoc Fisher's Least Significant Difference test) tested differences in abiotic and biotic parameters at 10 m intervals.

PCA was performed to determine the influential parameters on both non-native grass cover and bryophyte cover using Minitab 17.1.2 Statistical Software. These parameters were explored further through simple linear regression and curve fitting using SigmaPlot 12.3 with the relationship

defined by Pearson's correlation in Minitab 17.1.2. All parameters measured were reduced using best subsets regression to gain parameters to enter into a regression model. These parameters were then modelled using least squares estimation to determine the main factors influencing both cover of exotic grass species and bryophytes.

6.3 Results

6.3.1 Vascular Plants and Elevated Soil Nutrients

Soil Parameters

The treatments altered soil pH and concentrations of available nitrogen and phosphorus in relation to the amendment (Table 6.3). The lime treatment raised the pH whilst nitrogen and superphosphate reduced it (by one pH unit in the case of N2). Nitrate levels were highest with urea addition, N2 significantly raised levels of ammonium. Olsen P was only calculated for two species in the experiment (*P. amoena* and *C. australis*) making the two-way ANOVA unfeasible; however, the treatments containing superphosphate significantly elevated soil Olsen P concentrations. There were interactions between species and treatment effects for pH, NO_3^- and NH_4^+ .

Table 6.3 Soil physico-chemistry in 7 differently amended soils (n=20) for four native plant species (n=35, Table 6.2 for treatment codes). Olsen P could only be assessed at treatment level due to only two plants being analysed. Means that do not share a letter are significantly different ($p < 0.05$, Tukey HSD). No letters denotes no significant difference and ***= significant $p < 0.05$.

	pH	NO_3^- -N	NH_4^+ -N	Olsen P	Mortality (out of 35)
Treatment					
C	5.1 ^a	0.97 ^c	0.96 ^b	5.594 ^{cd}	0
N1	4.5 ^c	49.38 ^b	7.07 ^b	5.166 ^d	4
N2	4.1 ^d	143.9 ^a	99.02 ^a	7.057 ^c	3
L	5.3 ^a	3.56 ^c	3.04 ^b	4.809 ^d	3
P	4.9 ^b	0.76 ^c	0.61 ^b	14.799 ^a	2
NL	4.7 ^{bc}	34.31 ^b	8.77 ^b	4.725 ^d	1
NPL	4.8 ^b	22.49 ^{bc}	11.13 ^b	12.142 ^b	3
p value	<0.001	<0.001	<0.001	<0.001	
Species					
<i>P. amoena</i>	4.9 ^a	30.1 ^{ab}	10.19	Not available	4
<i>L. juniperina</i>	4.8 ^{ab}	8.52 ^b	27.75		1
<i>K. robusta</i>	4.8 ^{ab}	45.5 ^{ab}	17.71		6
<i>K. serotina</i>	4.6 ^b	49.32 ^a	16.38		4
<i>C. australis</i>	4.8 ^{ab}	47.0 ^a	20.75		0
p value	0.069	0.019	0.469		
Significant Effects					
Species	***	***	***	Not available	
Treatment	***	***	***		
Species x Treatment	***	***	***		

Not all plants remained healthy during the trial and a number of plants died in all treatments apart from the control and in each species apart from *C. australis*. Despite this, replicates within each treatment remained at least n=4, with the exception of *P. amoena* (treatments N1, NL and NLP) which were reduced to three replicates.

Plant Biomass

Total biomass for *L. juniperina* was not significantly affected by treatment (Figure 6.1). *K. serotina*, *K. robusta* and *P. amoena* experienced greater biomass when grown in soils amended with nitrogen with reduced growth in P, lime and control conditions. There was a trend for total biomass of *C. australis* to be greatest under those treatments including lime, NPL treatment producing most biomass.

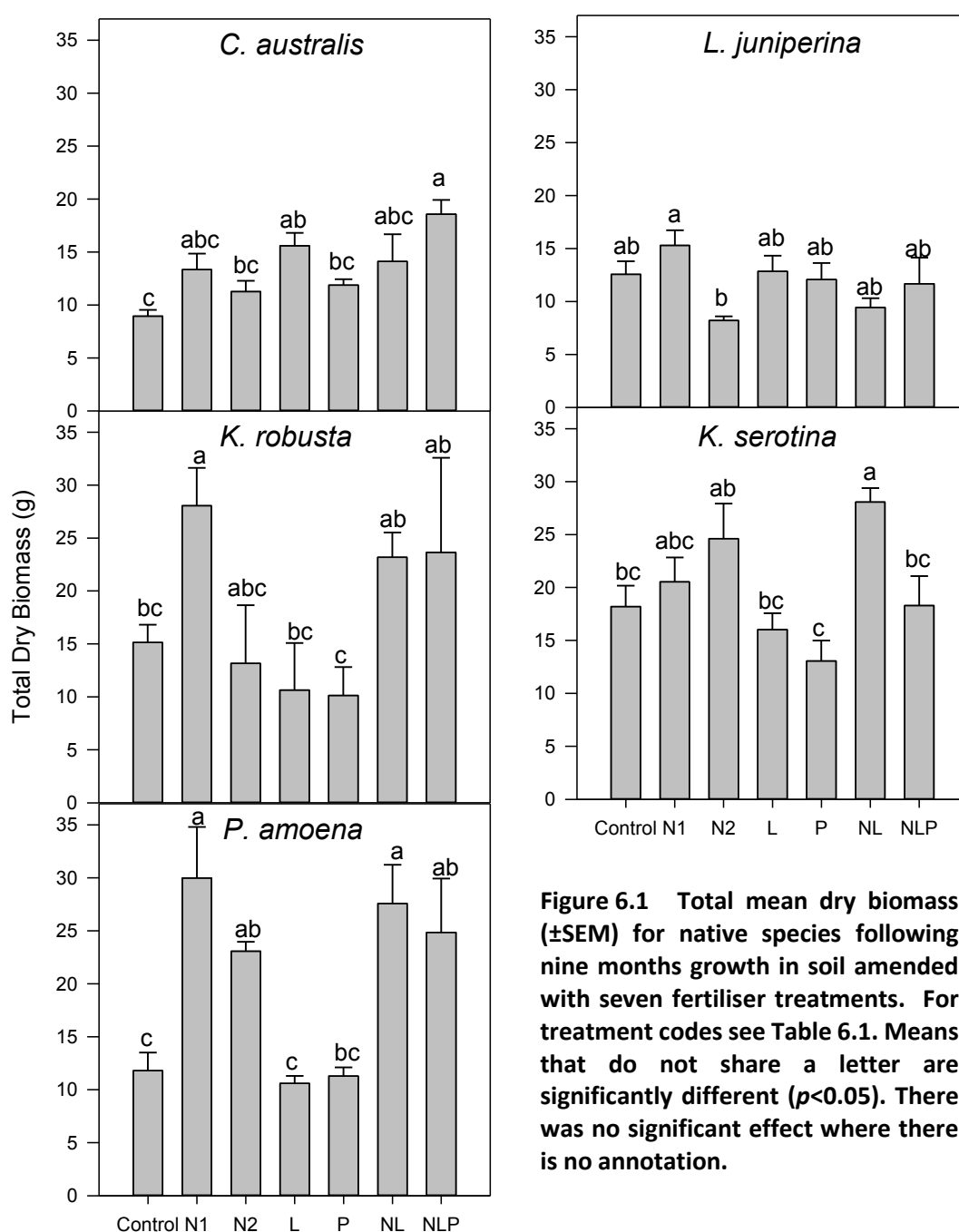
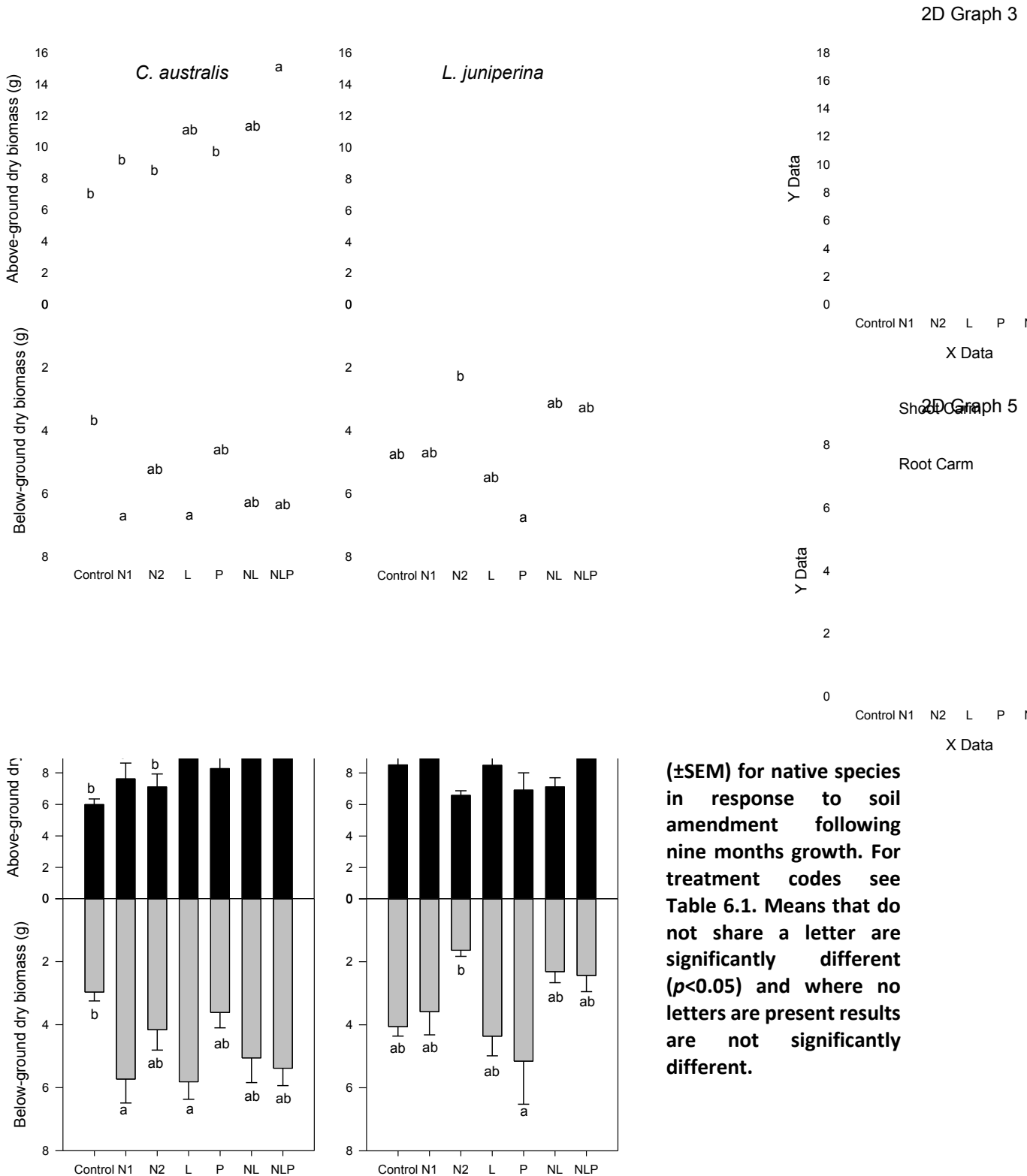


Figure 6.1 Total mean dry biomass (\pm SEM) for native species following nine months growth in soil amended with seven fertiliser treatments. For treatment codes see Table 6.1. Means that do not share a letter are significantly different ($p < 0.05$). There was no significant effect where there is no annotation.

Root and shoot biomass showed a similar pattern; *Kunzea* species and *P. amoena* achieving the highest biomass in treatments which contained nitrogen and lowest in phosphorus, lime and control treatments (Figure 6.2). *L. juniperina* was not significantly affected by treatment for shoot biomass although the root biomass was greatest under elevated phosphorus conditions and lowest under high nitrogen. Shoot biomass for *C. australis* mirrored that of the total biomass result with elevated growth in treatments that contained lime.



6.3.2 Moss and Nutrient Exposure – Glasshouse

The treatments of lime and superphosphate applied to the moss turves showed no significant difference in dry weight over six months. This is despite the fact that turves in the phosphorus treatments all showed signs of poor health with discoloured, and in some cases, dying leaves. With regards to volume of moss when fresh, the only treatment that had more growth than the control was the low lime treatment (Figure 6.3). All other treatments reduced the spread and health of the moss with treatments containing phosphorus being most detrimental.

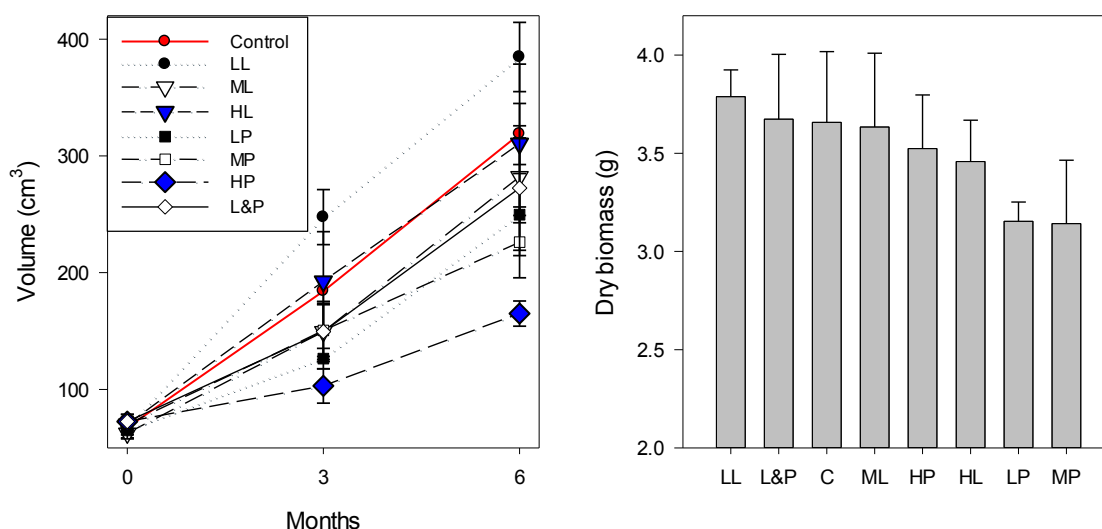


Figure 6.3 The volume of moss (cm³) at the start of the experiment and after 3 and 6 months growth experiencing different treatments (left, $p=0.006$), and dry biomass after 6 months (right, $p=0.64$). Treatment codes shown in Table 6.2.

6.3.3 Moss and Nutrient Exposure – Field Study

Clear patterns of plant cover were observed in the context of distance from the fence line (Figure 6.4). The bryophyte cover increased into the centre of the remnants, with cover reducing after this point. Conversely, there was a negative correlation between exotic grass cover, and associated litter, soil nitrate and Olsen P concentrations from the fence line into the centre of the remnants. Olsen P concentrations were significantly higher at 0 and 10m from the fence line, reaching the lowest concentration from 20 m (Figure 6.5). Nitrate concentrations were significantly highest between 0 and 20 m, reaching lowest concentrations at 30 m and beyond, into the remnant. Following this trend, the grass litter cover was lowest from 30 m onwards whilst the moss cover was highest from this point into the remnant.

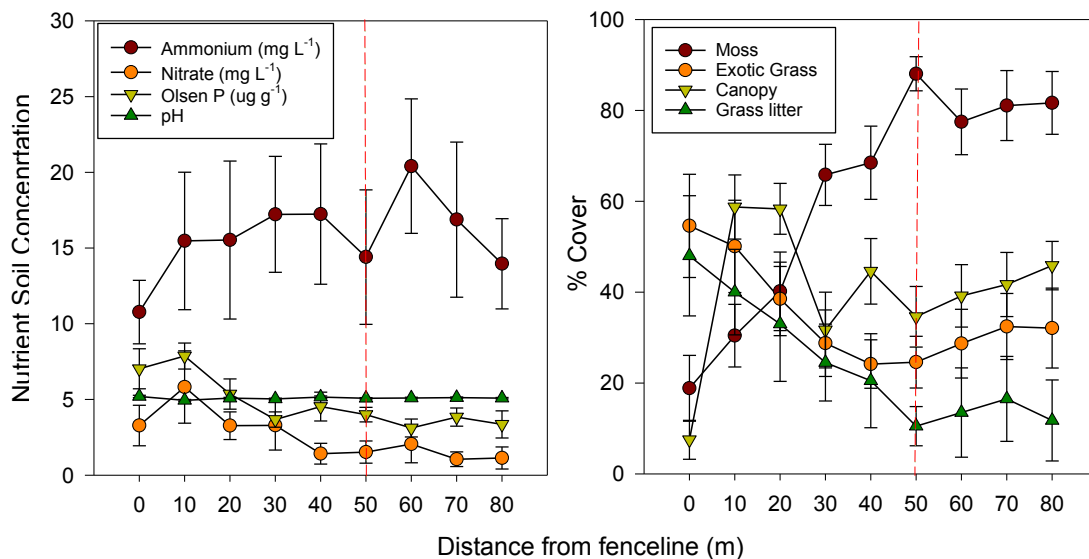


Figure 6.4 Relationship of measured parameters to distance from fenceline adjacent to irrigated pasture into the centre of the remnants dryland kānuka stands (centre point noted by red dotted line). Biotic parameters include percentage cover of moss ($r=0.71$, $p<0.001$, $R^2(\text{adj}) = 49.2\%$), exotic grass ($r=-0.39$, $p=0.002$, $R^2(\text{adj}) = 13.7\%$), canopy (not significant) and grass litter ($r=-0.38$, $p<0.001$, $R^2(\text{adj}) = 12.7\%$). Soil parameters include NH_4^+ (not significant), NO_3^- ($r=-0.30$, $p=0.023$, $R^2(\text{adj}) = 7.4\%$), Olsen P ($r=-0.39$, $p=0.001$, $R^2(\text{adj}) = 13.9\%$) and pH (not significant).

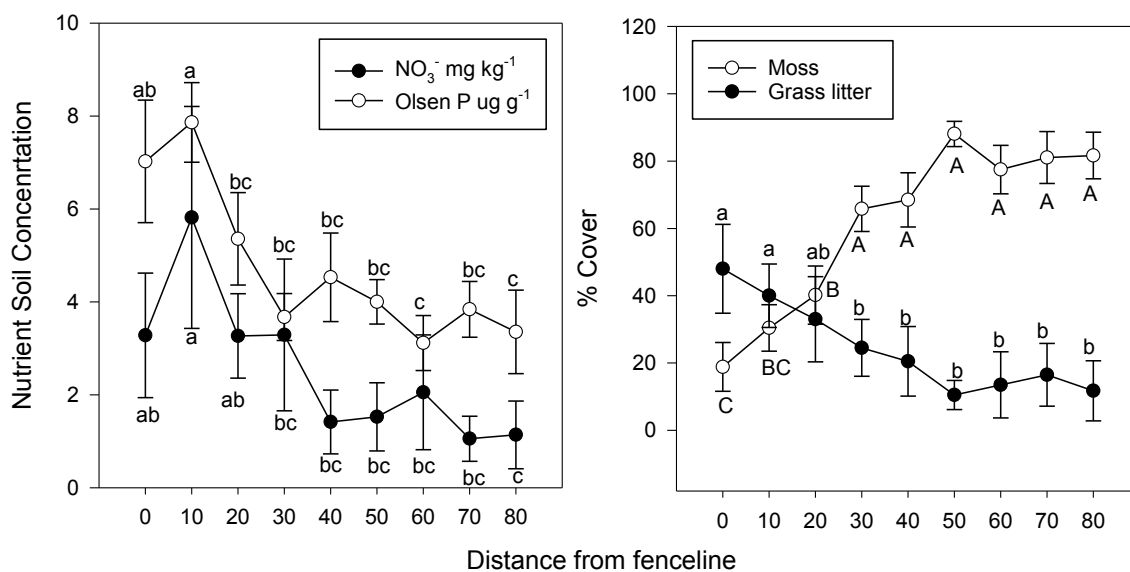


Figure 6.5 Nitrate and Olsen P concentrations and moss and grass litter cover recorded at 10m intervals into the remnants. Means that do not share a letter (either abc or ABC) are significantly different ($p<0.05$).

Multivariate analysis illustrated the factors associated with the cover of moss within the two remnants (Figure 6.6). The highest bryophyte cover was positioned along the negative of both axes. Axis one was positively weighted for percentage of grass cover and grass litter cover whilst axis two was positively weighted for canopy cover, NO_3^- and NH_4^+ .

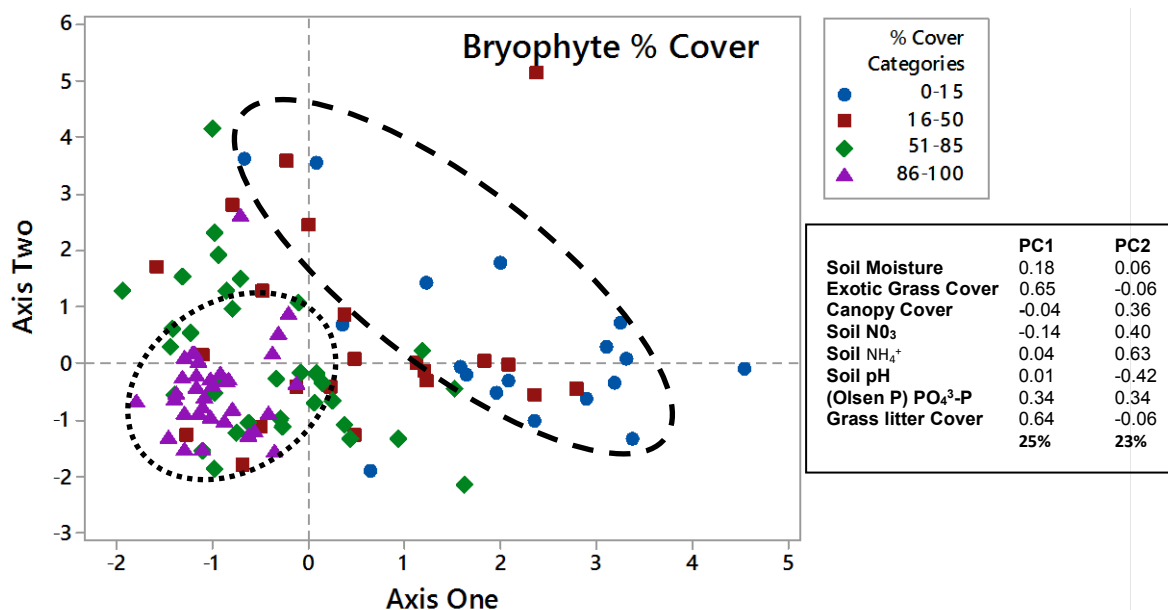


Figure 6.6 PCA of biotic and abiotic components in relation to bryophyte cover. The categories represent percentage cover of bryophytes: 0-15%, 11-50%, 51-85% and 86-100% with points representing single sampling locations. The relative loadings are tabulated adjacent to the figure and ellipses demonstrate the broad ordination space occupied by the highest % moss cover (dotted line) and lowest % moss cover (dashed line).

Multivariate analysis of abiotic and biotic parameters when grouped by exotic grass cover demonstrated that the highest grass cover occupied a distinct ordination space (Figure 6.7). Axis one was weighted for bryophyte cover and depth and axis two for bryophyte depth and soil NH₄⁺.

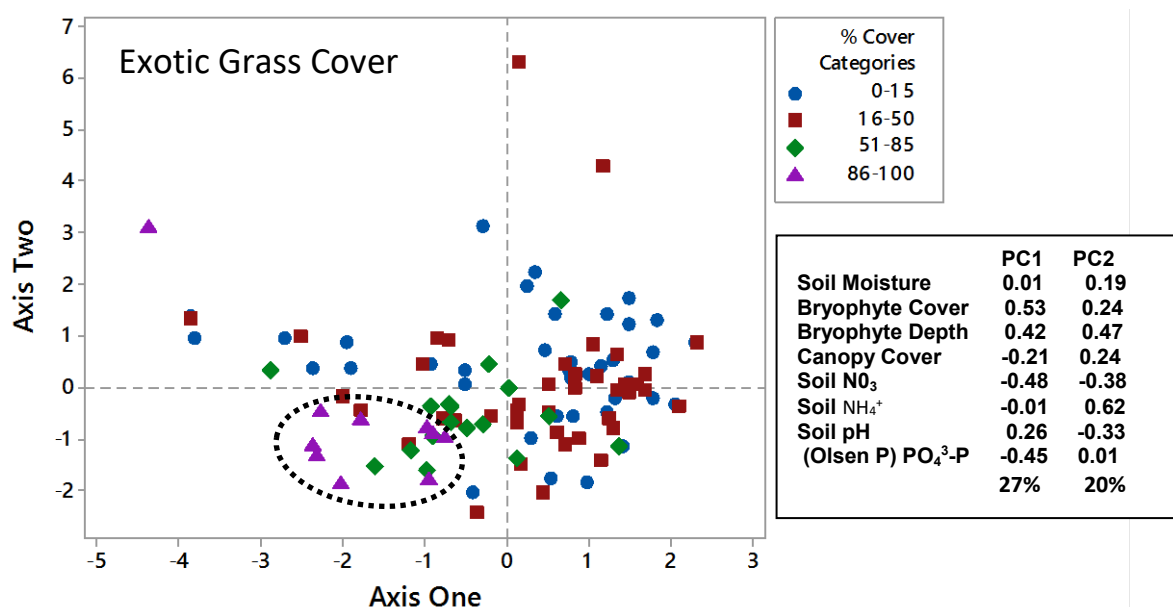


Figure 6.7 PCA of biotic and abiotic components determining exotic grass cover in four categories (0-15%, 11-50%, 51-85% and 86-100%). The relative loadings are tabulated adjacent to the figure and the oval overlaid on the figure demonstrates the ordination space occupied by the highest grass cover category.

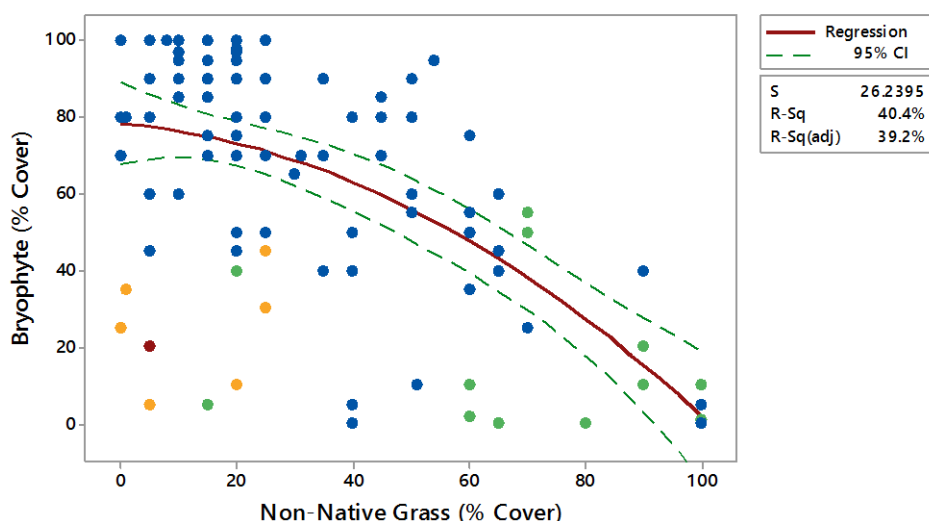


Figure 6.8 Simple quadratic regression for percentage cover of bryophyte and non-native grass ($r = -0.62$, $p < 0.001$, R^2 (adj) = 39.2%). The blue circles represent the data points of moss and grass cover. The coloured circles represent samples where percentage cover of moss and grass was low due to presence of >80% kānuka litter (orange), grass litter (green) or bare ground (brown). Regression excluding these points resulted in R^2 (adj) 52.1%.

Percentage cover of bryophytes was negatively correlated with the exotic grass cover (Figure 6.8). A number of data points were below the 95% confidence bands where both moss and grass cover were low owing to replacement by kānuka litter, grass litter or bare ground (% cover > 80).

Bryophyte cover was significantly reduced by the cover of exotic grass litter and an increase in soil nitrate and Olsen P (Figure 6.9). Exotic grass cover was significantly reduced by bryophyte cover and a reduction in available P. Multiple regression revealed that parameters most influencing bryophyte cover were cover of grass litter cover, NO_3^- , Olsen P and NH_4^+ ($p < 0.001$, R^2 (adj) = 66.9%). For exotic grass cover the influential parameters were moss cover, NO_3^- , Olsen P and soil moisture ($p < 0.001$, R^2 (adj) = 60.9%).

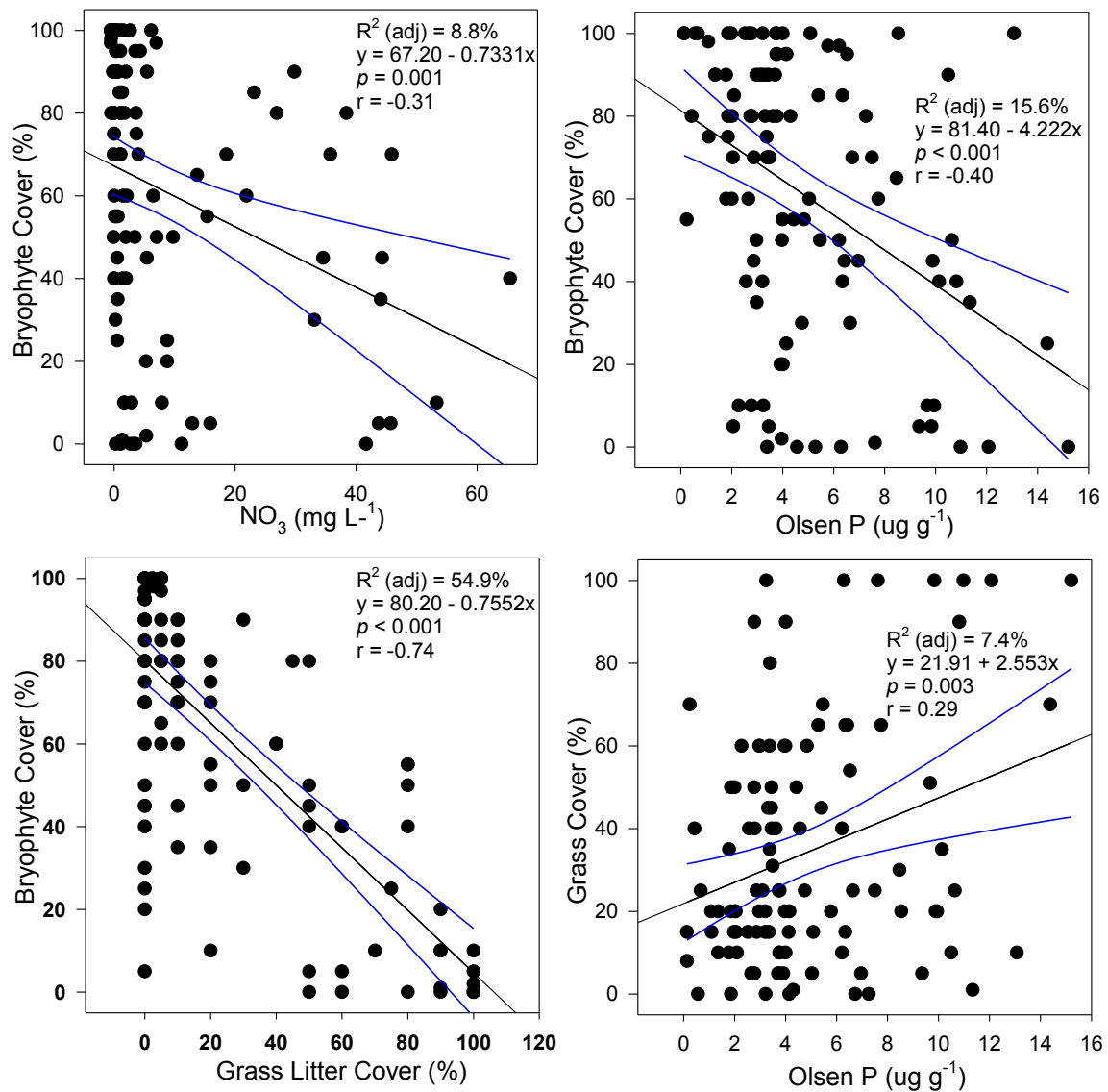


Figure 6.9 The relationship between percentage cover of bryophyte and exotic grass and the interactions with nitrate and Olsen P soil concentrations. Regression line shown in black with 95% confidence limits shown in blue.

6.4 Discussion and Conclusions

6.4.1 Vascular Plants and Elevated Soil Nutrients

The glasshouse trial revealed that both kānuka species and *P. amoena* produced more biomass in treatments with added nitrogen. This contrasts with the results obtained by Franklin (2014) and Stevenson and Smale (2005) who found no significant biomass increase in native plants and a preference for nutrient poor substrate. However, in both of these studies the control substrate used to grow the natives (even when topsoil had been removed by Stevenson and Smale (2005)) contained higher concentrations of nutrients than the Lismore soil used in the present trial collected from Eyrewell Forest. Nitrogen is a principal constraint to net primary production (LeBauer and Treseder 2008; Vitousek and Howarth 1991), and in the present study fertilisation alleviated N limitation, as

observed with plants of similar low nutrient environments elsewhere in the world (Britton and Fisher 2008; Sharifi et al. 1988; Zhao and Liu 2009). In a comparable study, Caroline (1999) found that fertiliser did assist plant growth in the first year but this increase subsequently diminished. This was attributed to the fertilised plants investing in foliar growth whilst the non-fertiliser plants invested in root growth leading to enhanced performance in later years. This was not found in this study, the control plants had similarly low above and below ground biomass.

Esperschuetz et al. (2017a) and Esperschuetz et al. (2017b) found increased biomass in both *K. robusta* and *Leptospermum scoparium* when subjected to a biosolid amendment or urea application: the former increased both nitrogen and phosphorus soil concentrations. Hawkins and Sweet (1989) also reported increase in growth of rimu (*Dacrydium cupressinum*), a tree usually located in nutrient poor sites, in response to increased nutrients. Phosphorus, typically also of limited extent in New Zealand soils (Langer et al. 1999; Livesley et al. 2008; McLaren and Cameron 1996), did not promote biomass in the kānuka species or *P. amoena* in this study, agreeing with the findings of Franklin et al. (2015). Addition of phosphorus also reduced growth of *L. juniperina* as did the high levels of urea but total biomass was not significantly affected for this slow growing species.

C. australis had increased aboveground biomass in all treatments containing lime additions, a result reported elsewhere for other nitrogen fixing species (Basu et al. 2008; Mohammadi et al. 2012). The lack of response to nitrogen treatments may be due to the fact that it is a nitrogen fixing plant and is therefore not limited by soil available nitrogen. To support this, results, not shown here, indicate that there was no nodulation within plants rooting into substrate where nitrogen was applied (N1, N2, NL and NLP; Shanshan Li, pers. comm.) with nitrate often preventing nodulation (Brewin 1991; Carroll and Mathews 1990). The benefits of lime application may equate to an increase in the solubility of nutrients required for growth of plant and nodulation, especially phosphorus, sulphur and potassium (Divito and Sadras 2014; Sulieman et al. 2013), and the immobilisation of those elements harmful to plants, such as aluminium (Truog 1948). Furthermore, nitrogen-fixing plants have a high requirement for phosphorus due to metabolic activity associated with nodule formation (Sulieman et al. 2010). This was reflected in the present study with plants exhibiting increased growth in the combined treatment (NPL). Surprisingly, increased biomass of shoots did not occur in the P treatment although root biomass was increased compared to the control. This may be explained by the plant directing reallocating N to the roots and nodules rather than aboveground biomass (Sulieman et al. 2010).

6.4.2 Moss and Nutrient Exposure

There was no significant difference in biomass between the fertilised and non-fertilised bryophyte turves. This may be reflective of the fact that moss is slow growing (Furness and Grime 1982; Zechmeister 1995), and the trial may have benefited from a longer period of study. In spite of this, the

visual health and volume of the moss differed significantly, with lime addition increasing volume. This is partly explained by the observation of stress in the phosphorus treatments that had caused the moss turves to senesce and compact, thus decreasing total volume (Plate 6.1). Senescence or “browning” is known to be observed when bryophytes come into direct contact with fertilisers due to their lack of a protective cuticle (Hallbäck and Zhang 1998; Jäppinen and Hotanen 1990). Application of lime can also cause this damage but some species are able to regain health (Hallbäck and Zhang 1998). An optimum pH of 4.5-6.5 has been shown to increase biomass and species diversity in bryophytes (Virtanen et al. 2000) although results for *H. cupressiforme* have differed, with increases (Dulière et al. 2000; Hallbäck and Zhang 1998) and decreases in northern hemisphere forests (Rodenkirchen 1992). However, with more time, the current experiment may have resulted in death of turves with phosphorus application and increasing growth of those with lime application.



Plate 6.1 Moss turves after 6 months growth in two different treatments: low lime (left) and medium phosphorus (right).

6.4.3 Field Trial

There appeared to be a negative relationship between the distance from the intensive farmland into the centre point of the remnants and the concentrations of nitrate and, more particularly, phosphorus in soil. The relationship for nitrate was less robust as is expected due to nitrate being water soluble and likely to percolate through the soil in a vertical rather than horizontal flow (Bowie et al. 2016; Hubbard et al. 1989). However, the ammonium concentrations did not differ significantly throughout the remnants, possibly a consequence of low variation of NH_4^+ inputs and low variation of mineralisation rates and clay mineral exchange sites. NO_3^- is formed by the biological oxidation of NH_4^+ , a process known as nitrification, and therefore the ammonium content controls nitrification. In the present

study, NH_4^+ remained relatively constant whilst NO_3^- concentrations decreased into the centre of the remnants. Nitrification can also be inhibited by other factors such as low soil pH, low soil nutrient status and high aluminium content (Haynes 1986). The pH of the soils did not differ along the transect, which also governs Al availability, but nutrients decreased, as demonstrated by the negative relationship of Olsen P concentrations to distance from the fenceline.

Another factor which may explain the NO_3^- concentrations is the presence of native plant species in the remnant habitat. The moss ground cover increased into the centre of the remnants which were dominated by the kānuka canopy species. *H. cupressiforme* can influence soil mineral N concentrations (as discussed in Chapter 5, section 5.2.4). Despite the present study being undertaken in summer when mosses are generally inactive, they may still decrease nitrate soil concentrations (Figure 5.3). Furthermore, studies have shown the kānuka may inhibit nitrification (Esperschuetz et al. 2017b) possibly through the action of enzyme inhibitory plant extracts that impede nitrifying bacteria activity (Kellam et al. 1992) and through roots exudates which are known to have antimicrobial properties (Prosser et al. 2016). However, the action of kānuka on the nitrification process is under debate due to the lack of evidence in a similar study by Franklin et al. (2017). In this study, inhibition of denitrifying bacteria was suggested but no evidence of effect on nitrification was observed.

Olsen P concentrations were higher at the edge of the remnant habitats. Leaching of phosphorus from soil can occur in periods of heavy nutrient loading and heavy rainfall, but it is usually only carried as sediment in surface runoff (Gillingham and Thorrold 2000; Nash and Murdoch 1997). It is known that exotic grass and weed species respond well to additions of nitrate and phosphorus (Blackshaw et al. 2004; Moir et al. 2003; Smith et al. 2012). Pasture grasses are generally those species of temperate environments accustomed to more fertile edaphic conditions than found in the Lismore soils at Eyrewell (Charlton and Stewart 1999). Therefore, the transport of such nutrients into the remnants from the neighbouring pasture frees the grasses from nutrient limitation enabling encroachment and an increase productivity. Another factor intensifying exotic weed encroachment was disturbance. Both of the reserve had experienced grazing in the previous years; in ESR pigs had escaped into the reserve for a period less than a month (pers. comm., Anita Spencer (DOC)) and cattle had been seen in SBR prior to erection of permanent fencing. Disturbance can facilitate exotic weed invasion in New Zealand native habitats (Jesson et al. 2000; Kelly and Sullivan 2010) and it is assumed that the disturbance to the understory vegetation provided this.

Conversely, results of this study showed that there was a strong relationship between the increase in exotic grasses and a reduction in moss cover (Figure 6.7). It is likely that the moss reduction was a function of the competitive, shading and smothering presence of the grass and associated litter.

van der Wal et al. (2005) found that shading due to increased growth of vascular plants in response to nitrogen input reduced the growth of mosses, although this is a species specific effect.

Positive feedbacks between exotic grass invasion and nutrient enrichment can also occur. Perennial ryegrass (*Lolium perenne*) and other turf-forming grasses contain higher nitrogen concentrations than native species litter (Dickinson 1984; Dickinson et al. 2015; Hahner et al. 2013). Litter from exotic species often decomposes more rapidly than native species (Craine and Lee 2003; Ehrenfeld et al. 2001) creating rapid cycling of nutrients into the ecosystem (Hobbie 2015). This can change community composition by altering nutrient thresholds which in turn can be detrimental to native species (Ehrenfeld 2003; Putten et al. 2013). Furthermore, during the field survey work, a disc spreader passed by the ESR remnant and particles of lime were thrust at least 2 m into the remnant, a phenomena encountered by Tsiouris and Marshall (1998). Although the pH did not differ significantly along the transect it is an example of direct eutrophication.

6.4.4 Patch Size Effects

Results of the present study showed nutrient spillover and weed encroachment into the remnants, decreasing with further distance from the fenceline, reaching stable levels at around 30 m into the site. Bowie et al. (2016) found native species richness was peaked at 20 m into a similar kānuka dryland shrubland reserve (DOC Bankside Scientific Reserve, full description in Chapter 3). However, the effects of adjacent farming land on pH and Olsen P concentrations were much more pronounced, partly owing to the topography of the site.

Fragmented and small remnant patches of native vegetation are known to experience edge effects and high loadings of nutrients, such as P, from adjacent land use. In Australia, Duncan et al. (2008) found that nutrient enrichment was negatively related to patch size, particularly for P, from aerial and water deposited nutrients and Stevenson (2004) reported a 10 fold increase in P in small remnant habitats compared to native reference forests found in Hamilton, New Zealand. Furthermore, a large scale study of fragmented remnants in the Waipa district, North Island, confirmed that nutrient loadings increased with the increasing intensification of adjacent land use (Didham et al. 2015). They found that the soil physico-chemistry was altered over 50 m into the centre of fragmented remnant habitats when compared with reference forest soils sampled from forest, 400 m into the centre of the habitat. Due to the lack of large areas of kānuka shrubland remaining on the Canterbury Plains, the present study could not compare reference site soil composition with the small remnants studied. However, the results suggest that edge effects are experienced at least 30m into the remnant habitats at Eyrewell Forest. This implies that a reserve with an area of 1 ha would have an interior of only 1600 m² which would be suitably buffered from effects of neighbouring land use.

6.4.5 Restoration Considerations and Recommendations

In this study the native vascular plants and moss appeared to increase biomass under nitrogen or lime addition but reduce when phosphorus was applied. This suggests likely negative effects on biomass in the field where fertiliser spillover is experienced. However, the amount of superphosphate applied to the pots was large (470 kg ha^{-1}) reflecting the amount applied to soils to enable initial conversion of Eyrewell pine forest to dairy pasture. Subsequent amounts applied per farm to maintain grassland are likely to be lower, e.g. in 2003-2004 average amount per application was 50 kg ha^{-1} (Basset-Mens et al. 2009). This indicates that spillover of this concentration would not be anticipated. In addition, when fertiliser was applied in combination (NPL) the results were largely beneficial. Therefore, this study suggests that the main issue affecting the remnant communities would be invasion rather than growth inhibition. “A plant community becomes more susceptible to invasion whenever there is an increase in the amount of unused resources” (Davis et al. 2000). This statement exemplifies the situation within the kānuka remnants where nutrient enrichment may occur in an ecosystem adapted to low fertility soil.

The present study revealed that local disturbance from nutrient spillover and weed encroachment, which in turn are changing soil conditions and community structure. This situation is reported from native habitats throughout the world (Bowie et al. 2016; Harrison 1999; Huenneke et al. 1990; Uzêda et al. 2016). The cumulative effect of nutrient enhancement and replacement leads to the replacement of native species, including the mosses. Mosses exert influence on the soil hydrology and nutrient cycles and interact with the germination and establishment of the vascular plants as discussed in Chapter 4 and 5 of this thesis. In addition, mosses have been found to drive community assemblage change in arthropods (Pehle and Schirmel 2015).

Tall grass swards prevent recruitment of the main canopy species (Allen et al. 1992), the seeds of which require light for germination. There is a risk this may lead to “living dead” stands through lack of regeneration (Meurk et al. 2016). In order to mitigate these issues and to design functionally resilient habitats, there appear to be two requirements: disturbance should be eliminated as far as possible (e.g. prevention of nutrient spillover) and structural and functional processes within the kānuka shrubland restored.

Prevention of spillover

There are a number of nutrient enrichment mitigation strategies, many of which relate to farming methods such as timing and amount of fertiliser application, controls surrounding tillage and fencing

of remnants (Schoumans et al. 2014; Stark and Richards 2008). Other methods suggested are physical prevention of nutrient rich drainage into native areas. Bowie et al. (2016) commented that a bund preventing surface runoff had attracted cattle which in turn had flattened it, and presumably experienced high nutrient inputs from cattle excrement, identifying the importance of fencing. Another, feasible landscape tool to mitigate nutrient enrichment is the creation of a buffer strip separating the remnant from the source of pollution.

The use of soakaways, gravel filled drains surrounding remnant areas allowing percolation of runoff rather than allowing spillover into the reserve, have been installed at a nearby Canterbury remnant kānuka stand (Bankside Scientific Reserve) but results of the effectiveness are yet to be examined (Anita Spencer (DOC), pers. comm.). This technique may be particularly effective due to the elevated concentrations of Olsen P at least 30 m into the reserve. Olsen P was positively correlated with encroachment of exotic grasses, as was nitrate. However, used in isolation and in the well-drained Lismore soils, these may facilitate movement of nutrients to water courses leading to eutrophication (Edwards and Withers 2008; Jarvie et al. 2006; Withers et al. 2011). Therefore, planting is recommended in the soil above the soakaway to increase capacity for nutrient uptake into foliage rather than leaching to water courses.

Kānuka has been shown to be capable of luxury uptake of N and has a reducing effect on the action of denitrifying bacteria, limiting release of N_2O from the soil (Franklin et al. 2015). There is also a suggestion that the plant may inhibit nitrifying microbes, thereby reducing NO_3^- in soils (Esperschuetz et al. 2017b). Therefore, a densely planted buffer, at least 10 – 20 m wide (Bowie et al. 2016), in the soil layer on top of the soakaway may mediate nitrogen losses to water courses and the N uptake capacity may assist in maintaining low soil N, reducing potential for spillover. Planting along water courses are known to trap sediment in runoff (Leathwick et al. 2003), although more so for the coarse particles (Barling and Moore 1994). Didham et al. (2015) suggest planting a mosaic of species due to the plants varying ability to prevent sediment (grasses) or soluble nutrient (woody species) movement into neighbouring land. Furthermore, Franklin et al. (2015) found that some native monocotyledons, pukio (*Carex virgata*) and harakeke/flax (*Phormium tenax*), accumulated large amounts of N in foliage, suggesting these plants could offset nitrate leaching at farm boundaries (Figure 6.10).

Restoring Ecosystem Processes

Restoring processes within the reserve would primarily require the reserve to be adequately fenced from grazing cattle; beneficial functioning services offered by species within the habitat are unlikely to be felt in the presence of grazing (Zhang et al. 2016). Supplementary planting of kānuka within the

disturbed edges of the habitat that are particularly prone to weed invasion is recommended to supplement for the lack of regeneration in those areas. Native plant litter from species such as kānuka has been shown to influence soil nutrient cycles, particularly with regards to lower NO_3^- concentrations (Zhong et al. 2016). Finally, it is recommended that small non-vascular plants fragments are collected from within the reserves and introduced back into the edge habitats to restore the functional traits of the moss with regards to hydrology and nutrient cycling (Figure 6.10). This is given more attention in Chapter 7.

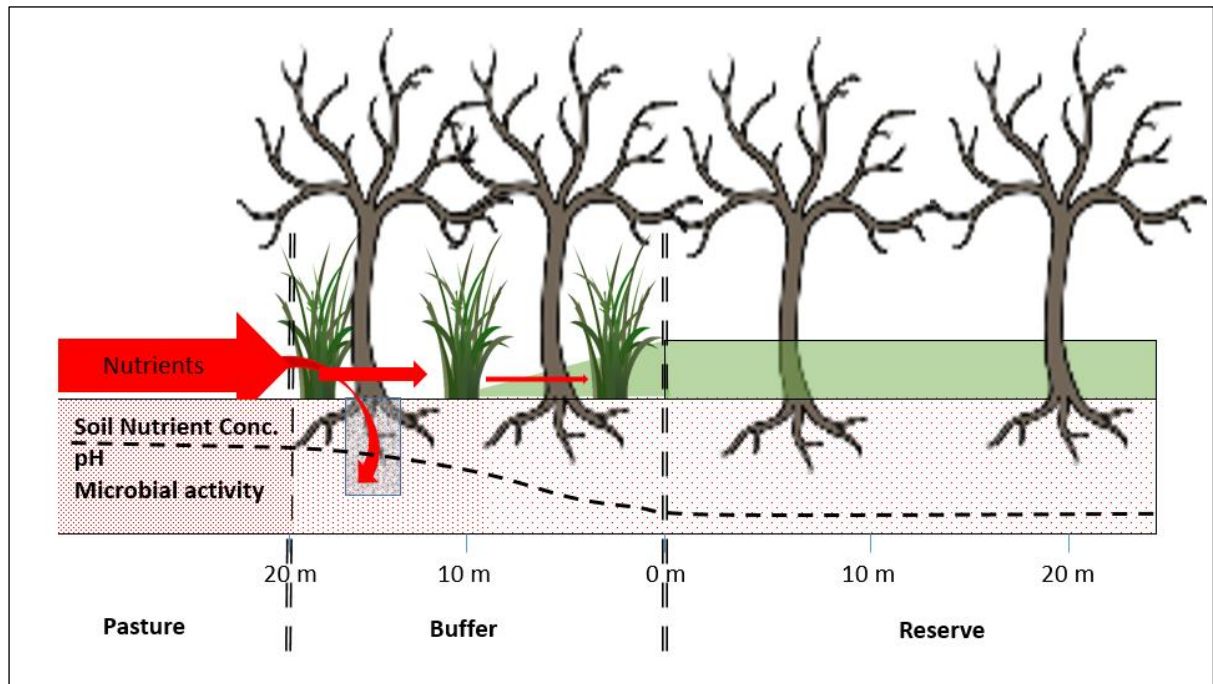


Figure 6.10 Illustration of design for restoration and remnant kānuka stands to assist with limiting nutrient spillover, weed encroachment and maintaining ecosystem function. Nutrient from the pasture in runoff are lost from the system through drainage though the soak away (grey rectangle) 5 m from the perimeter of the fenced buffer zone. Runoff sediment is trapped in roots of the planted buffer, plants known for high uptake of N and with a reduction effect on the microbial community. Moss layer is restored to the edge of the reserve maintaining ecosystem function.

In conclusion, the remnant kānuka stands were found to experience spillover effects from neighbouring farmland. In light of the fact that many native species do not respond positively to increased soil nutrients, particularly phosphorus, and competition by weeds, it appears crucial that mitigation for this protects remnants and created areas of native habitat. By combining a number of techniques, the effects of spillover and exotic weed encroachment may be lessened, enabling the survival of kānuka shrubland communities within the multifunctional landscape.

Chapter 7

Restoration of Native Dryland Vegetation Embedded in Irrigated Dairy Pasture

7.1 Introduction

“Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (Clewett et al. 2004). In New Zealand, the high levels of endemism within flora and fauna (Kier et al. 2009; Mittermeier et al. 2005), and factors such as the loss of three quarters of indigenous land cover (Cieraad et al. 2015b; Ewers et al. 2006; Walker et al. 2006) and a renewed respect for indigenous habitats has led to a recent increase in native ecological restoration projects around the country (Meurk and Swaffield 2000; Meurk et al. 2016; Norton 2009; Peters et al. 2015).

Restoring areas which have been degraded, damaged or destroyed includes many abiotic and biotic challenges such as considering the effects of historical land use on the substrate, dispersal constraints of biota into the restoration area and its spatial character (Honnay et al. 2002). In the low lying, drier areas of the Canterbury Plains, indigenous habitat has been reduced by over 99% (Cieraad et al. 2015b; Walker et al. 2006) leaving small, fragmented islands of native vegetation and habitat (Meurk et al. 1995). Small remnant habitats mean large edge effects from adjacent farming landscape (e.g. Bowie et al. (2016)) and also lead to a limited pool of propagules and large distances between patches of native vegetation. Seed banks are often devoid of native species, which mean it is necessary to physically introduce plants into restoration areas (Overdyck and Clarkson 2012).

Introduction of native species can take many forms, such as natural regeneration or purposefully spreading seed through a number of techniques (Dodd and Power 2007; Lamb and Gilmour 2003; Stanturf et al. 2014), but by far the most common method is the planting of nursery grown specimens (Anton et al. 2015; Clarkson and Kirby 2016). Each method has its own challenges such as cost, time to achieve canopy closure and survival rates. Natural regeneration is relatively low cost but is reliant on available propagules which may be absent within the fragmented landscape (Meurk et al. 1995; Overdyck and Clarkson 2012; Partridge 1989). It may also result in an altered trajectory and a different community than the proposed target determined at the start of the project (Sullivan et al. 2007). Applying seed to an area (direct seeding) can be useful at overcoming this issue and can also reduce costs of restoration (Dodd and Power 2007; Douglas et al. 2007; Engel and Parrotta 2001; Madsen and Löf 2005). In addition, plants germinated *in situ* can be more robust than those transplanted into an area. However, direct seeding can lead to high germination failure rates and low establishment (Ceccon et al. 2015; Close and Davidson 2003) due to a number of factors including gaps

in plant knowledge relating to seed dormancy, site preparation, sowing rates and management (Douglas et al. 2007; Fountain and Outred 1991; Rowarth et al. 2007). Planting out robust specimens grown off-site may overcome issues of vulnerability of young seedling survivorship (van Andel and Aronson 2012) but can be costly, requiring each plant to be protected from harm from herbivory, abiotic stress, excessive shading and competition (Gorbunov 2008).

One of the most commonly used techniques for protection for young planted specimens against herbivores and harsh abiotic conditions is the use of tree guards (Davis and Meurk 2001; Lai and Wong 2005). Commonly, tree guards comprise a sheath which is held in place with canes secured in the ground, fully enclosing the young plant (Ladd et al. 2010; Lai and Wong 2005). There are a number of commercial suppliers providing a range of similar products (e.g. KBC Combiguard®, Tubex® tree shelter tubes and Ascot Seedling Protection® tree guards). These guards are often coupled with a weed mat that sits around the base of the plant thereby offering further protection against frost and preserving soil moisture. Correct installation of tree guard and mat is important as adverse weather condition can cause them to be lost or collapse and smother seedlings (Close and Davidson 2003). The material used and its properties, such as rigid plastic, polythene or shade cloth, can affect plant growth by influencing the temperature and light increment immediately surrounding the plant (Close et al. 2009). Studies in dry areas of China and Australia have found that combined tree guard and weed mat treatments are cost-effective methods for increasing survivorship and growth of plants in the long term (Ladd et al. 2010; Lai and Wong 2005; Stanton-Clements et al. 2013). In New Zealand, tree guards are commonly used and recommended for restoration plantings (Davis and Meurk 2001).

The competitive exclusion principal notes that populations of non-breeding, sympatric populations occupying the same ecological niche cannot stably co-exist (Hardin 1960). Those species with the ability to effectively acquire and use limited resources or produce offspring rapidly can increase fitness and competitiveness (Tilman 1982). In New Zealand, there are as many naturalised exotic vascular plant species as there are native and 1366 species are naturalised in Canterbury (Howell 2008). Mahon (2007) compiled a list of 328 exotic plant species considered to be weeds and the most common species recorded within the kānuka stands at Eyrewell Forest are shown in Table 7.1 (Chapter 3, Appendix B1). All species listed can compete for soil nutrients, water, light and space within restoration plantings.

Weed control is considered one of the most important aspects of seedling growth and inefficient control can lead to failure (Close and Davidson 2003; Davis and Meurk 2001; Ladd et al. 2010). Targeted chemical control, particularly with glyphosate, is a common method for controlling invasive plants but it is not guaranteed to eradicate a plant and can have damaging effects on the environment, the handler and surrounding biota if not carried out correctly (Rinella et al. 2009;

Standish 2002). Used in conjunction with tree guards, weed mats are useful at suppressing weeds within the immediate vicinity of the young plant without disturbing surrounding land and allowing pioneer weed species to invade (Lai and Wong 2005). Similarly, using organic or inorganic mulch such as gravel or chipped bark, if applied at a sufficient depth, may suppress the weeds (Winkel et al. 1996). Mulch has also been shown to promote growth of plants in degraded soils by retaining moisture (Singh and Saggarr 1997) and ameliorating soil temperatures (Blanco-García and Lindig-Cisneros 2005) in harsh environments although this is likely to be species and mulch type dependent (Athy et al. 2006). With every restoration technique, implications of site and species need to be considered to achieve success (Wade et al. 2008).

Table 7.1 Exotic weed species frequently recorded in kānuka stands at Eyrewell Forest in the ecology study (Chapter 3), noted to be a problem by Ecroyd and Brockerhoff (2005) in their study of the Eyrewell Scientific Reserve (*) or commonly observed at Eyrewell forest ().**

Weed Species	Family	Growth Form	Habitat preference
<i>Agrostis capillaris</i>	Poaceae	Grass	Low fertility habitats
<i>Anthoxanthum odoratum</i>	Poaceae	Grass	Low fertility habitats
<i>Cytisus scoparius</i>	Fabaceae	Shrub	Common on dry/acid soils
<i>Festuca rubra</i>	Poaceae	Grass	Free draining, neutral pH soil
<i>Hypochaeris radicata</i>	Asteraceae	Herb	Disturbed sites, low fertility
<i>Lolium perenne</i>	Poaceae	Grass	Moist sites, moderate pH (6-7)
<i>Pinus radiata</i>	Pinaceae	Tree	Low fertility sites
<i>Crepis capillaris</i> *	Asteraceae	Herb	Disturbed sites, low fertility
<i>Ulex europaeus</i> **	Fabaceae	Shrub	Common on dry/acid soils
<i>Dactylis glomerata</i> **	Poaceae	Grass	Wide tolerance (moderate fertility)

As discussed in previous chapters of this thesis, moss is considered to have an important role in the ecological function of an ecosystem. Research into bryophyte restoration is less extensive than for vascular plants (Brooker et al. 2011). Nonetheless, there have been successful introductions of bryophytes by introduction of plant fragment into grasslands in Germany (Jeschke 2012), heathlands in Scotland (Flagmeier et al. 2016), in alpine environments with small turves (<25 cm²) (Aradottir 2012), and both tropical and boreal forest environments (Hazell and Gustafsson 1999; Nadkarni et al. 2000; Rosso et al. 2001). In Westland, New Zealand, restoration of the bryophyte layer has received scant attention. A scheme in an old coal mine aimed to increase the trajectory of natural succession and stabilise soil by applying a mixture of small bryophyte fragments in solution to bare ground (Buxton et al. 2005). Results were varied, from 0.1-28% cover, in relation to the amount of bryophyte fragments in the mixture, the site physical condition (e.g. aspect, slope) and the microclimate conditions.

The suite of experiments reported in the present chapter aimed to determine the main practical challenges for a large-scale ecological restoration project at Eyrewell Forest on the Canterbury Plains. As described in Chapter 3 of the thesis, the former pine forest had been purchased

by Ngāi Tahu and was undergoing conversion from pine forest to dairy pasture setting aside areas of land for ecological restoration. It was agreed by the stakeholders that dryland kānuka shrubland communities would be restored in 18 areas set aside for active restoration (referred to as reserves and illustrated on Figure 7.1). The reserves, plus two reserve areas where passive restoration would occur, would amount to over 150 ha of indigenous habitat with another 150 ha of native landscape and shelterbelt planting. The previous chapters in this thesis provided knowledge on the plant communities and abiotic conditions of the dryland kānuka communities and informed design of the reserves to be established.

The aim of this chapter is to describe experiments carried out to investigate different restoration techniques appropriate to these habitats. The main objectives were:

- i. To investigate the role of tree guards with regard to providing shelter from herbivores and climatic stresses (sections 7.2.2, 7.2.3);
- ii. To establish whether seedlings would benefit from nutrient application to increase growth in the nutrient poor soils (sections 7.2.4, 7.2.5);
- iii. To examine different methods of weed control in the field (7.2.5, 7.2.6); and
- iv. To explore a range of propagule introduction techniques (7.2.6, 7.2.7).

7.2 Methods

7.2.1 Study Site and Species Used

The study site was Eyrewell Forest (43°43'21.04" S, 172°33'39.46"E) located on the Lower Plains Ecological District of the Canterbury Plains (McEwan 1987). The climate of the region is dry with a prevalence of strong north-westerly föhn winds, warm summers, cool winters and low rainfall leading to low humidity and high evapotranspiration rates (Macara 2016; McGowan and Sturman 1996; Molloy and Ives 1972). The droughty Lismore soils are naturally of low fertility (Kear et al. 1967) and were previously afforested with *Pinus radiata*. The site had been recently cleared to facilitate conversion from pine forest to dairy pasture leaving 18 areas (reserves) ranging in size from 0.55 – over 10 ha (Figure 7.1). Each cleared reserve site comprised a mosaic of bare ground, woody debris and self-seeded annual weed and grass species which tended to die back in the winter months. Gorse (*Ulex europaeus*) and broom (*Cytisus scoparium*) were also present on all sites. A high density of hares (*Lepus europaeus*) was present on the site as the main herbivore.

Previous experiments and studies had reported the low fertility of the natural soils supporting the kānuka communities in the area (Chapter 3 of this thesis). The detrimental effect of nutrient

enrichment on species composition and encroachment of exotic grasses and weeds had been previously described (see Chapter 6) and this determined the requirement to protect the reserve areas from nutrient spillover from neighbouring farmland. Therefore, every reserve had a 5 m wide buffer of kānuka (*Kunzea serotina*) or mānuka (*Leptospermum scoparium*) planted around the perimeter. All experiments reported in the current chapter were undertaken within the centre of these reserves, inside the buffer plantings, with the exception of the tree guard trial where the buffer plantings were used for the experiment.

The dominant canopy species of the dryland shrubland, *K. serotina*, was used in all experiments. *Pomaderris amoena* was also used due to its local importance. This plant is at its southern boundary in the Eyrewell Forest and previous attempts to grow the plant from seed or transplanting of seedlings had been relatively unsuccessful which acted as a driver to reintroduction of the species in all restoration plots. Mānuka was used within fertiliser, weed and direct seeding experiments due to its former presence within the kānuka shrubland community alliance and a desire by stakeholders to produce mānuka honey. Finally, *Carmichaelia australis*, *Ozothamnus leptophyllus* and *Carex goyenii* were used in the direct seeding trials due to these plants being located adjacent to reserves with viable propagules.

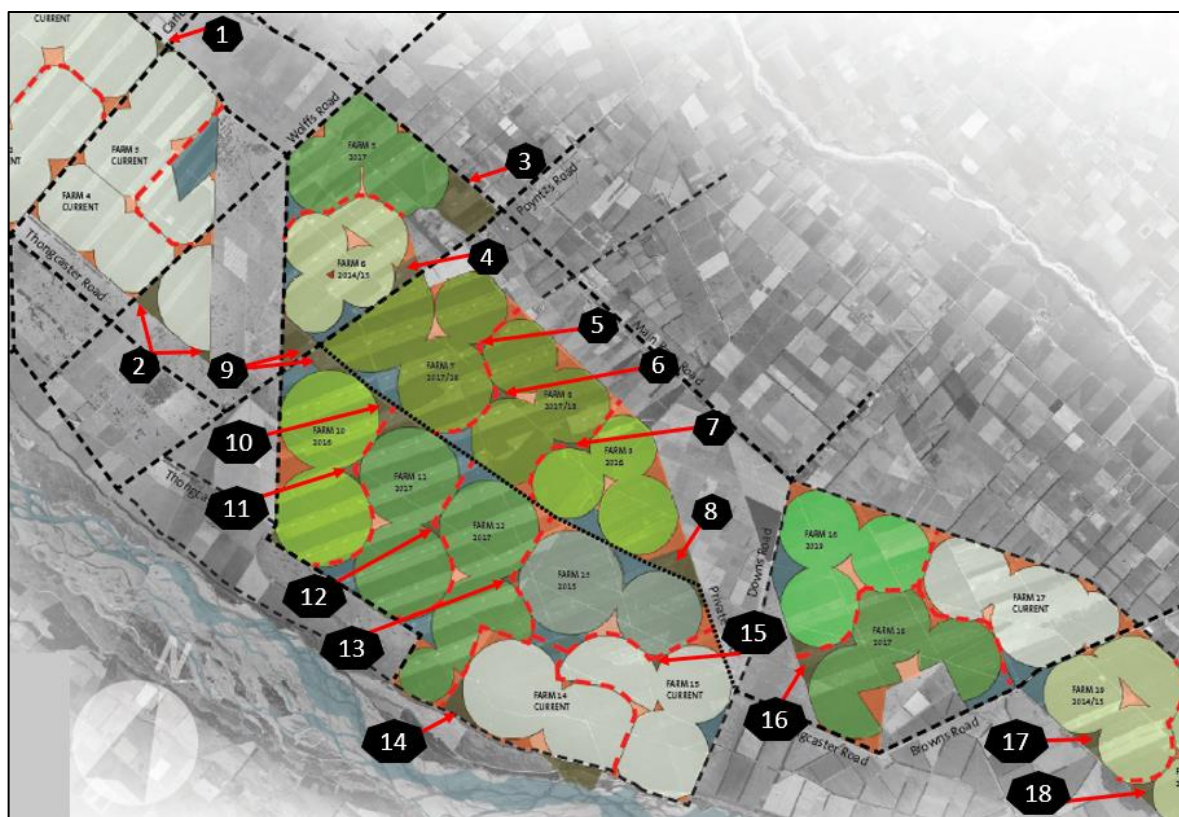


Figure 7.1 Location of reserves within the agricultural landscape matrix at Eyrewell Forest. Experiments were located in reserves 2, 14, 15, and 17.

7.2.2 Tree Guard Mortality Trial

In winter 2015 (April/May), one year old saplings of *Kunzea serotina* and *Pomaderris amoena* grown from seed collected from Eyrewell Forest were planted in reserve 17 in three treatments:

- Planted with tree guard (TG);
- Planted without tree guard (NTG); and
- Planted without tree guard with netting along the fenceline for wind protection (NTGN)

There were two types of tree guard used; a thin polythene sleeve held in place by four canes (TG1) and a rigid corrugated plastic guard which was held in place with one stake (TG2) (Plate 7.1). Each type had a mulch weed mat installed around the base of the plant made from Eco Wool™ (wool waste fibre). All treatments received one dose of hare and rabbit repellent (Grazers Rabbit Repellent, Hunkin Garden Products Ltd) upon planting with the exception of the TG1 treatment where plants were left for 12 hours without the tree guard or repellent.



Plate 7.1. Two types of tree guard were used within the experiment: a polythene sleeve held with four canes (left) and a rigid plastic guard held with one cane (right).

K. serotina was planted closest to the edge of the reserve and the *P. amoena* plants were at least 10 m into the reserve (Figure 7.2). Within each treatment the plants were arranged in 1, 2 or 3 rows to establish whether denser plantings have a positive effect on elements such as wind and sun damage.

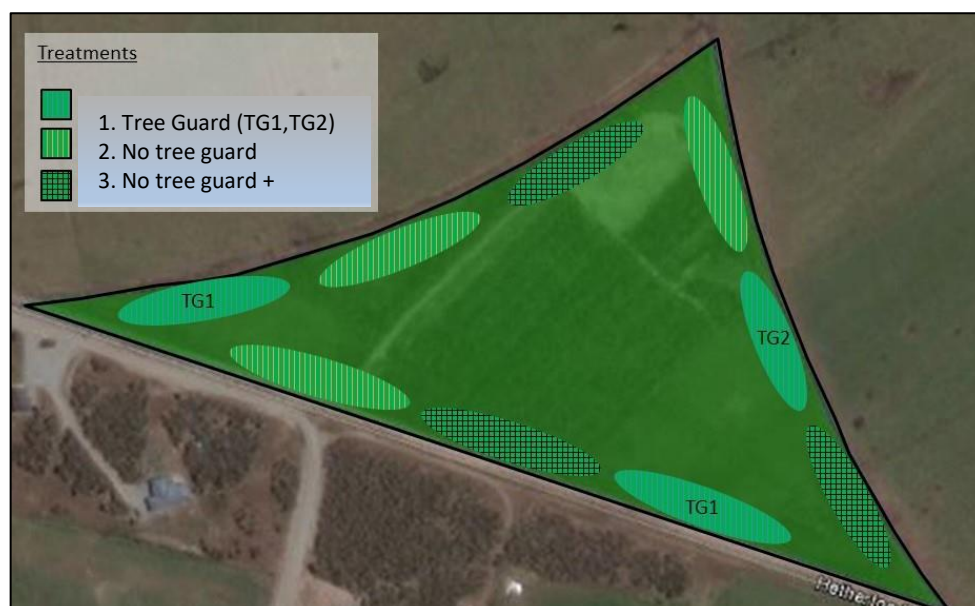


Figure 7.2 Arrangement of treatments for the tree guard experiment within a restoration area created within a restoration area in Eyrewell Forest.

Planting was at 1 m spacing, in a herringbone arrangement where multiple rows were present. For *K. serotina*, the plants were arranged in 1-3 rows, each containing 18 plants: there were three replicates of each treatment and a total of 108 plants per replicate, amounting to a total of 972 plants (see Figure 7.3). For *P. amoena*, plants were also arranged in three rows but with 4 plants per row except when three rows were present (Figure 7.3). There were 20 plants per replicate with three replicates of each treatment amounting to a total of 180 plants. Numbers of plants for this species was lower due to it not being a popular nursery grown species and therefore more difficult to obtain.

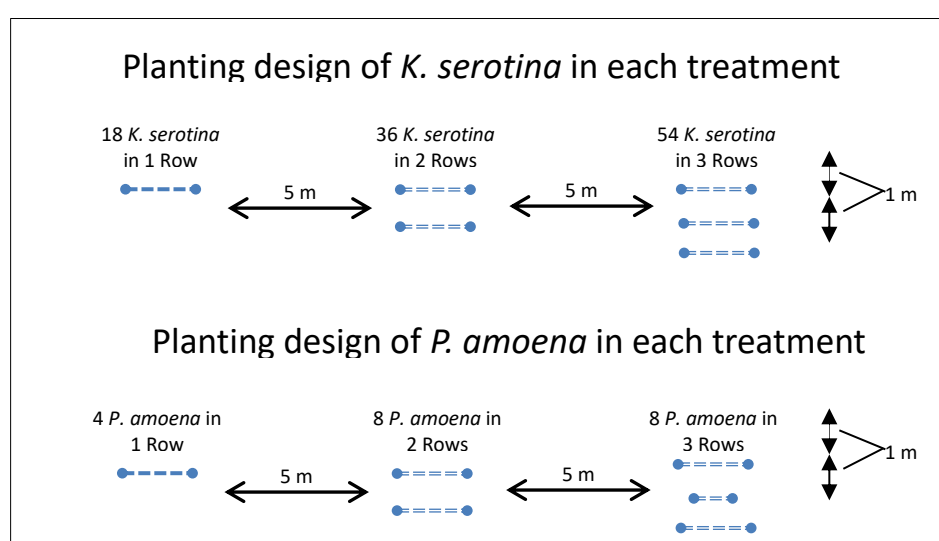


Figure 7.3 Planting configuration plan of the two plant species within each treatment.

Plants were monitored for 9 months without irrigation and plant survival and signs of injury recorded every three months. Signs of injury included damage by hares, drought or nutrient deficiency such as chlorosis. Mortality due to herbivory was recorded when a plant had noticeably been destroyed by hares such as being dug out of the ground or chewed and snapped at the base of the stem.

7.2.3 Tree Guards and Abiotic factors

To ascertain the effect of tree guards on ambient temperature surrounding the plant, weatherproof data loggers (HOBO® Pro v2) with two external temperature probes connected with 1.8 m cables (U23-00x) were used, with one placed inside the TG1 guard and the other at 1 m distance in the adjacent environment. Data loggers were acclimatised for two days before logging temperature hourly for 7 consecutive days in two periods of the growth season (November 2015 and January 2016). Six data loggers were used but 2 failed in the outside treatment equating to 3 replicates inside TG1 and only in the adjacent environment.

The effect of tree guards on soil temperature was also investigated in summer (January 2016) and winter (August 2015). Temperature probes were placed a 4 cm deep holes cut into the soil with a soil knife and subsequently closed around the probe. The temperatures were recorded under planted *K. serotina* plants in four treatments during the winter months which were;

- Control of bare ground with no tree guard or weed mat;
- Tree guard and bare ground (no weed mat);
- Tree guard and mulch weed mat made from Eco Wool™ (wool waste fibre); and
- Tree guard and moss ‘weed’ mat comprising a turve of *Hypnum cupressiforme* (approximately 3 cm depth and an area of 20 x 20 cm) collected from under pines trees in the neighbouring forest).

In summer there were three treatments, all excluding the control. After a period of 2 days *in situ*, data were recorded for 8 days recording temperature at hourly intervals. All treatments had 2 replicates during the summer experiment and 4 in the winter experiment. However, during the winter experiment 2 data loggers failed in the “tree guard and mat” treatment reducing the number of replicates to 2.

Soil moisture was also evaluated across the three treatments in both summer and winter. Soil samples were taken using a soil corer (2.5 cm diameter and 5 cm depth), immediately sieved to 4 mm and oven dried at 105°C to gain the percentage soil moisture (see Chapter 3 for detailed methods).

7.2.4 Fertiliser Tablets

To investigate the effect of fertilisers on New Zealand native plantings, slow release fertiliser tablets (Terra Pro ©) were used for planting *K. serotina* and *Leptospermum scoparium* into reserve 15. The plants were grown from seeds that were sourced from established plants in Eyrewell Forest and grown in a nursery for 1 year before planting. There were two treatments for *L. scoparium* (n=15): with fertiliser tablet and without. The fertiliser tablet was placed directly adjacent to the roots before planting. For *K. serotina* a third treatment was introduced (n=54): fertiliser tablet and soil which used a fertiliser tablet covered with soil prior to the plant being placed into the ground. All *K. serotina* plants were placed inside tree guards to prevent herbivory whilst none of the *L. scoparium* plants were protected.

The *L. scoparium* plants were monitored after 7 months and *K. serotina* plants for 9 months. There was no additional watering of the plants throughout the growth period and the heights were measured at the start and end of the trial. On completion, five plants were harvested in each treatment, the shoots and roots were cleaned of any soil and dried at 60°C for 4 days and weighed to assess the dry above and below ground biomass.

7.2.5 Weed Control

To investigate different methods of weed control, an experiment was set up in two adjacent reserve (reserve 2) which had been cleared two years previous due to a forest fire in the area. A total of thirty 10 m² plots, half containing planted *K. serotina* and half with planted *L. scoparium* during the winter months of 2014 and all with regenerating *P. amoena*, were marked out. Three replicates of each of four treatments were applied around the plants in summer 2015/6:

- Control, do nothing;
- Glyphosate (Roundup®), spraying of whole plot;
- Sawdust and dairy shed effluent mulch to 10 cm; and
- Compost mulch (Selwyn District Council municipal green waste compost) to 10 cm.

Prior to the start of the trial gorse (*Ulex europaeus*) and broom (*Cytisus scoparius*) were manually removed from each plot with glyphosate applied to stumps and the plant composition and percent coverage for each species occurring in the plot was recorded (Plate 7.2). Treatments were applied without regard to the regenerating *P. amoena* often covering young plants. After 11 months, the plant composition and percentage cover was measured again. In addition, five plants of each species were

harvested within the control, compost and sawdust treatments. They were cleaned of soil and dried for 3 days at 60°C to obtain dry biomass data.



Plate 7.2. The reserve without any treatment showing the weeds surrounding the plants (left) and student, Paula Greer, applying compost to one of the plots (right).

7.2.6 Direct Seeding

To assess the success of direct seeding and the effect of moss cover on seed germination and plant survival a total of 50 2 m² plots were marked out in reserve 14: 25 plots in an area which had been disturbed by a tractor and ripper, ripping the top 10 cm of soil (ripped), and 25 within an area which retained regenerating, exotic grasses and annual herbaceous vegetation (unripped). Five replicates of each treatment were established in each area (ripped and unripped) (Table 7.2).

Table 7.2 Description of treatments within the seeding experiment.

Treatment	Prescription
Control	Remained as <i>in situ</i>
Seeded	Received seeds
Glyphosate	Plot sprayed with glyphosate prior to addition of seeds
Forest litter	Received forest litter at 5 cm thick and containing >50% <i>H. cupressiforme</i> .
Forest litter + seeded	Received forest litter (as above) <i>in</i> addition to seeds.

The forest litter was harvested from underneath the pine forest to be felled and included pine needle litter and moss and lichen fragments. The amount of seeds used per plot was dictated by the amount that could be harvested (Table 7.3 details the seeds used). Germination and establishment of plants was monitored every week for the first two months and every three months subsequently for 9 months.

Table 7.3 Approximate number of seeds used within the seeding experiment.

Species	Approx. No. Seeds/m ²
<i>K. serotina</i>	1300
<i>P. amoena</i>	375
<i>Carmichaelia australis</i>	15
<i>Ozothamnus leptophyllus</i>	80
<i>Carex goyenii</i>	15
<i>L. scoparium</i>	200

7.2.7 Moss Introduction

Moss turves containing at least 80% composition of *H. cupressiforme* (approximately 3 cm depth and an area of 20 x 20 cm) were collected from under the pine forest and placed around the base of 12 seedlings planted in tree guards. The seedlings were *K. serotina*, *P. amoena* and *L. juniperina*. The survivorship of the moss was determined after 1 year *in situ* through observation of health.

Moss fragments were also collected from the neighbouring pine forest along with pine needle litter (at least 50% *H. cupressiforme*) and applied to 20 2 m² plots at 5cm depth in the direct seeding experiment (described above). The change in percentage cover of moss was recorded following 9 months.

Data Analysis

In the tree guard experiment, two-way ANOVA was used to detect differences in mortality with regards to treatment and the row arrangement of plants. One-way ANOVA was used to determine significant differences in percentage mortality. Distribution of herbivory data was skewed and no appropriate transformation was found; differences were tested using Kruskal-Wallis.

For both soil and air temperature the mean temperature (\pm SEM) for each treatment and each hour within the data collection period was calculated. Two-way ANOVA with repeated measures assessed the difference in temperature between treatment and hourly intervals. The mean soil moisture was calculated and the difference was calculated using one-way ANOVA with post-hoc Tukey HSD test.

Height and biomass of plants in the fertiliser tablet experiment were assessed using one-way ANOVA with post-hoc Tukey HSD test.

For the weed experiment, changes in mean percentage cover for bare ground, native and exotic plant cover and naturally regenerating *P. amoena* were calculated. The mean biomass of *K.*

serotina and *L. scoparium* growing in plots (\pm SEM) was also calculated. Differences between treatments for each parameter were assessed using a one-way ANOVA with post-hoc Tukey HSD test.

Mean (\pm SEM) and total germination rates, and plant survivorship for seedlings in the direct seeding experiment were calculated across all treatments. Due to a lack of data a suitable transformation could not be found so differences were assessed using non-parametric Kruskal-Wallis test.

For moss introduction experiment, changes in mean percentage cover moss were calculated in the two ground conditions. Differences between treatments were assessed using a one-way ANOVA with post-hoc Tukey HSD test.

7.3 Results

7.3.1 Combiguard and Mortality

Mortality rates of *K. serotina* were not significantly affected by the physical arrangement of rows or the interaction between row and treatment. There was higher mortality of *K. serotina* for those plants without tree guards (NTG and NTGN). Herbivory levels were low in all treatments (<10%). Herbivory rates were higher for *P. amoena* when not protected by a tree guard and lowest mortality and herbivory was experienced in TG1 (Table 7.4).

Table 7.4 Percentage mortality rates and the percentage of which were attributed to herbivory due to hare damage for *K.serotina* and *P. amoena* in all treatments (TG1= with tree guard 1; TG2= with tree guard 2; NTG= without tree guard; NTGN = no tree guard with wind netting). ** denotes significance at $p<0.05$.

Treatment	<i>K. serotina</i>			<i>P. amoena</i>		
	No. of plants	% Mortality**	% Herbivory Mortality	No. of plants	% Mortality**	% Herbivory Mortality**
TG1	216	23.15	0.93	40	52.50	0
TG2	108	41.67	1.85	20	85.00	0
NTG	324	78.70	8.95	60	86.67	36.67
NTGN	324	80.86	4.63	60	91.67	45.00

The plantings occurred in the late autumn/winter period and the majority of mortality occurred during the summer months for *K. serotina* (January 2016) (Figure 7.4). For *P. amoena*, there

was a steady increase in mortality in all treatments except TG1 which saw a sharp increase during the final measurement period in summer.

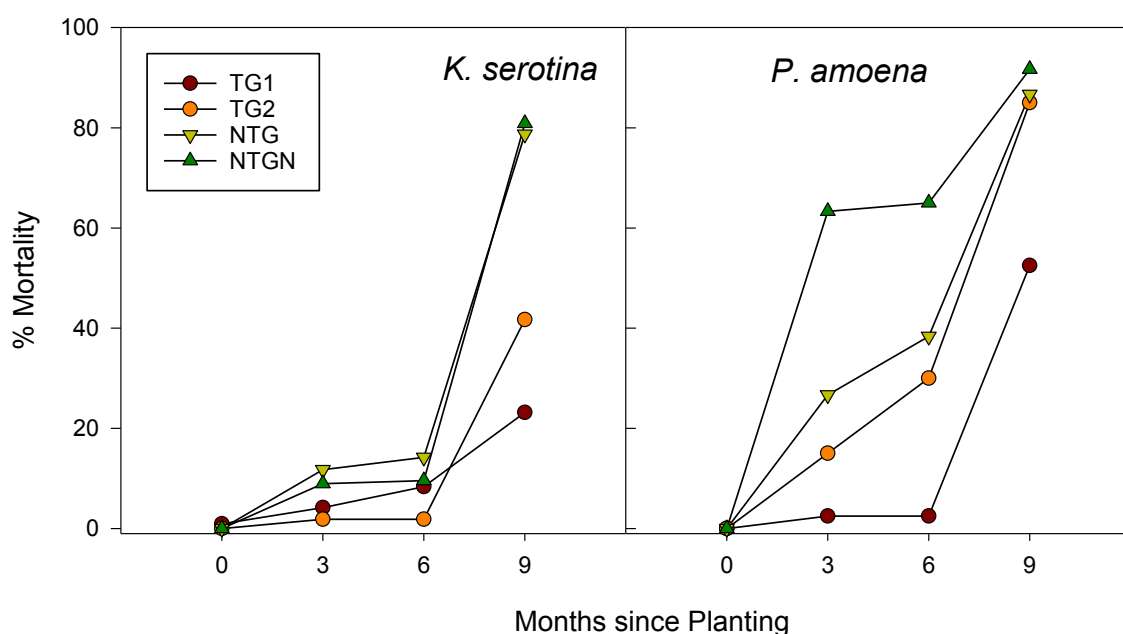


Figure 7.4 Cumulative mortality rates for *K. serotina* ($p < 0.001$) and *P. amoena* ($p < 0.01$) in each treatment recorded every 3 months from planting. Planting occurred in autumn/winter (April/May 2015) and measurements were taken in the following July, October and January.

7.3.2 Tree Guards and Abiotic Factors

Fluctuations in air temperature each day were reduced by the TG1 treatment showing the highest night-time and the lowest daytime temperatures in both November and January (Figure 7.5). Results were highly significant in both data collection periods for difference in temperatures recorded each hour and per treatment each hour. Treatment did not affect mean overall daily temperature in either period (Table 7.5).

Table 7.5 Results of two-way ANOVA with repeated measures assessing the difference in temperature when influenced by treatment alone, time of day or the interaction of both.

November	DF	F-Value	P-Value	January	DF	F-Value	P-Value
Treatment	1, 624	0.96	0.338	Treatment	1, 624	7.72	0.075
Hour	23, 624	21.50	0.000	Hour	23, 624	43.94	0.000
Treatment*Hour	23, 624	2.50	0.000	Treatment*Hour	23, 624	2.22	0.001

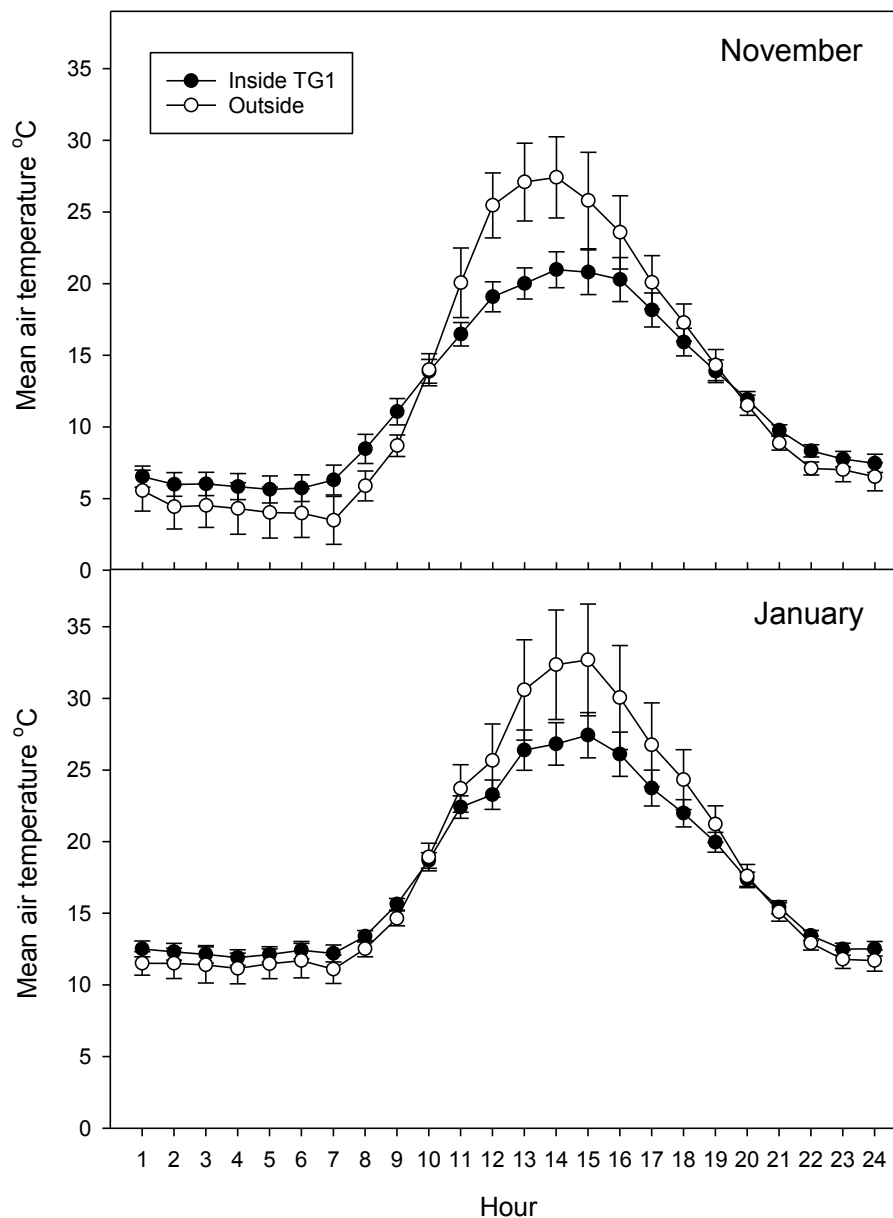


Figure 7.5 Mean air temperature (°C) (\pm SEM) during November 2015 and January 2016 both within the TG1 tree guard and outside. The means are shown as temperature per hour calculated from an average over 7 days.

The use of weed mats significantly reduced the soil temperature in winter ($p < 0.05$). The moss mat reduced temperature fluctuations in winter to 4.1°C whilst the mulch weed mat and no mat treatments were slightly higher (6.2°C and 6.4°C respectively) and the no tree guard treatment had a fluctuation of 7.5°C (Figure 7.6). In summer, treatment affected the overall daily soil temperature ($p < 0.05$) but was not significantly different each hour. The temperature ranges found the mulch weed mat to have the least fluctuation, 8.0°C between the highest and lowest temperature. For the moss mat the temperature varied 9.0°C whilst it was highest under no mat at 10.1°C.

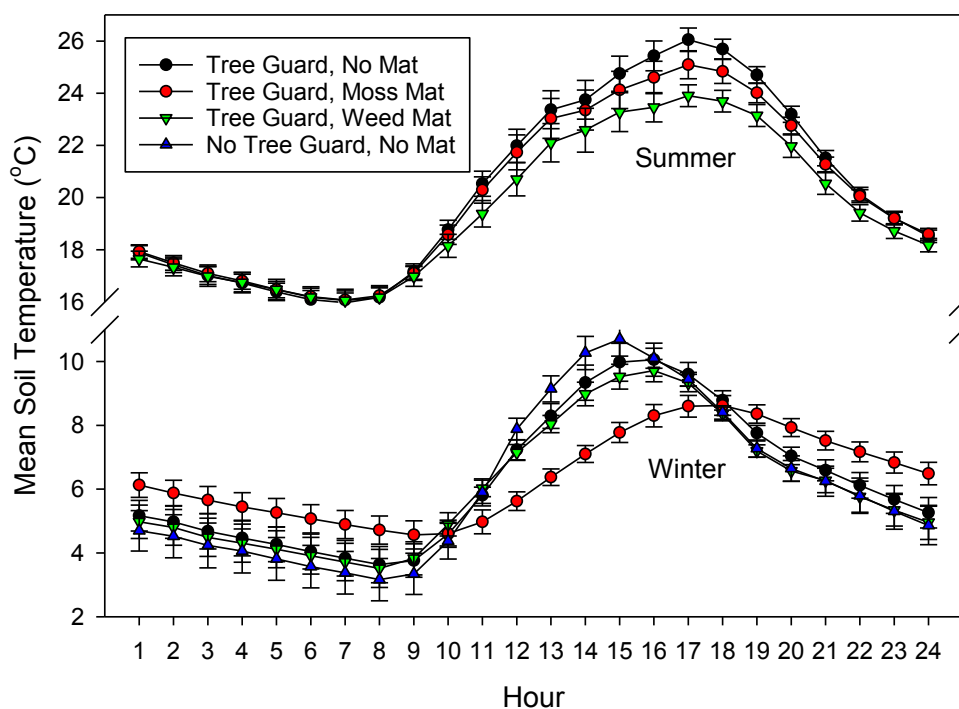


Figure 7.6 Mean soil temperature (°C) (\pm SEM) during the winter and summer months for a range of treatments involving tree guards and weed mats. The means are shown as temperature per hour calculated from an average over 8 days.

The percentage soil moisture did not differ significantly between treatments in summer. However, in winter the mulch weed mat and the moss mat had higher levels of soil (Figure 7.7). In winter, soil moisture was elevated due to the samples being taken following a period of rain. The “no mat” treatment had the least amount of fluctuation between seasons and the amount of fluctuation between seasons for each treatment was as follows: wool mat 39.92%, moss mat 22.98% and no mat 18.70%.

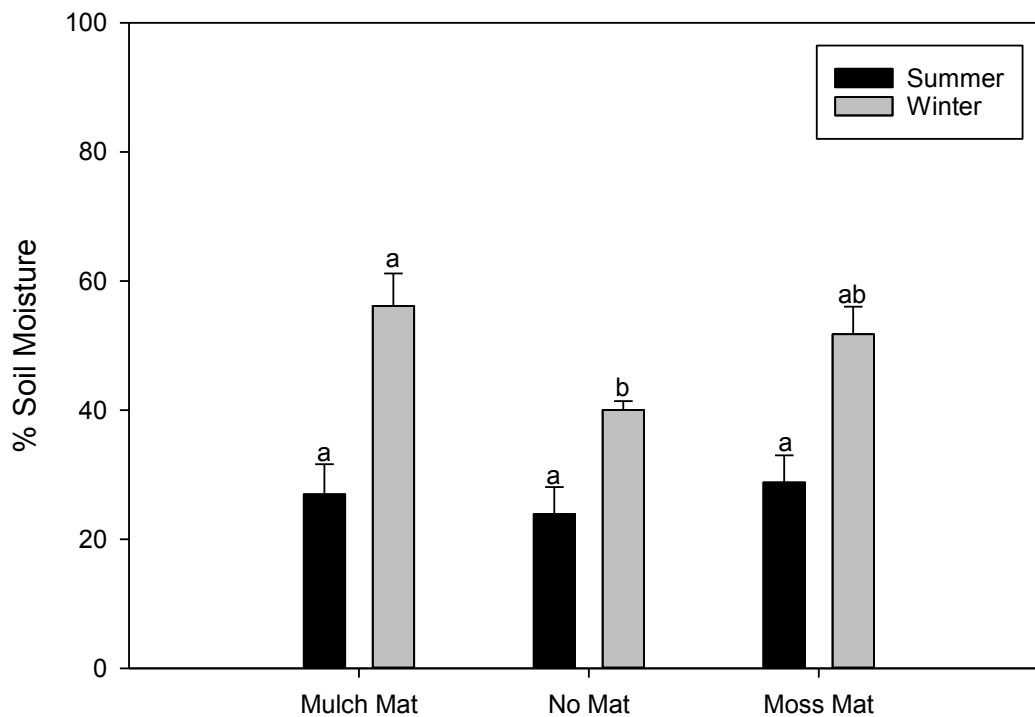


Figure 7.7 Mean % soil moisture (SEM) taken within tree guards and under three treatments during the summer ($p=0.425$) and winter months ($p<0.01$). Means which do not share a letter are significantly different.

7.3.3 Fertiliser Tablets

The addition of the fertiliser tablet decreased the biomass of *L. scoparium* by over 30 cm but there was no significant difference between treatments for *K. serotina* (Figure 7.8). The mean height of *L. scoparium* shoots was significantly lower in the fertiliser treatment although many had been subjected to herbivory and over half of the plants were dead at the end of the trial (Table 7.6). The results of *K. serotina* showed signs of hare damage on every plant and stem heights were not statistically different between treatments but echoed the biomass findings.

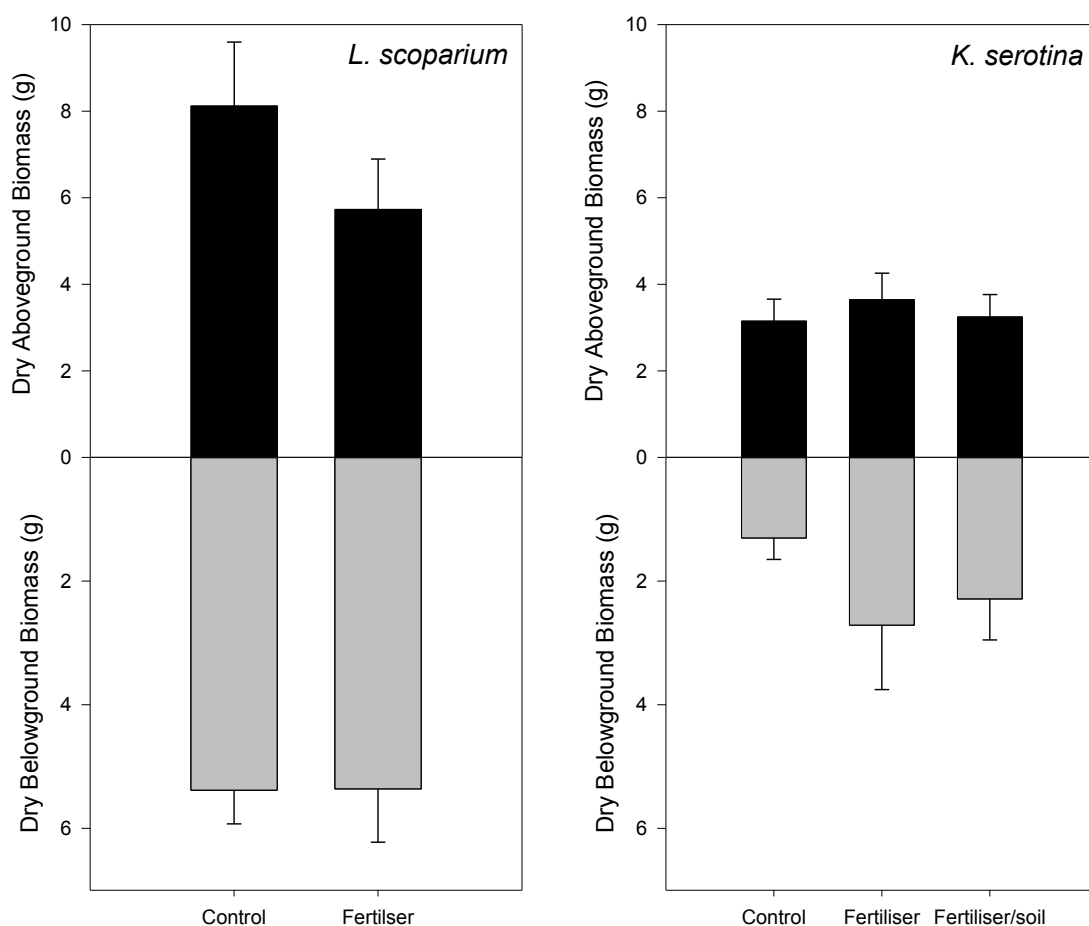


Figure 7.8 Mean above and below ground biomass (\pm SEM) for *L. scoparium* ($p=0.032$, $n=5$) and *K. serotina* ($p=0.810$, $n=5$) when planted with or without fertiliser tablet, or with fertiliser tablet covered with soil.

The roots of the plants which survived showed no sign of damage due to presence of fertiliser tablet. Fertiliser tablets were found intact with little interaction with the roots of either plant (Plate 7.3).

Table 7.6 Mean height (\pm SEM) of *K. serotina* and *L. scoparium* plants following 7 months growth during the growing season subjected to treatments (C= control, FT = Fertiliser tablet, FTS = Fertiliser tablet covered with soil). Means that do not share a letter are significantly different ($p<0.05$)

Treatment	<i>K. serotina</i>		<i>L. scoparium</i>	
	Height (cm)	Mortality (%)	Height (cm)	Mortality (%)
C	26.37 ^a	7 ^a	42.71 ^a	6.7 ^b
FT	27.06 ^a	7 ^a	37.00 ^b	53.3 ^a
FTS	26.98 ^a	6 ^a		
<i>p</i> value	0.909	0.946	0.032	0.007



Plate 7.3. Fertiliser tablets remained relatively intact in the soil surrounding the *L. scoparium* and *K. serotina* plants after 7 and 9 months growth. There was little evidence of interaction with plant roots.

7.3.4 Weed Experiment

Treatment had no significant effect on plant biomass of either *K. serotina* or *L. scoparium* although there was reduced growth for *L. scoparium* and increased growth for *K. serotina* in the sawdust treatment (Figure 7.9).

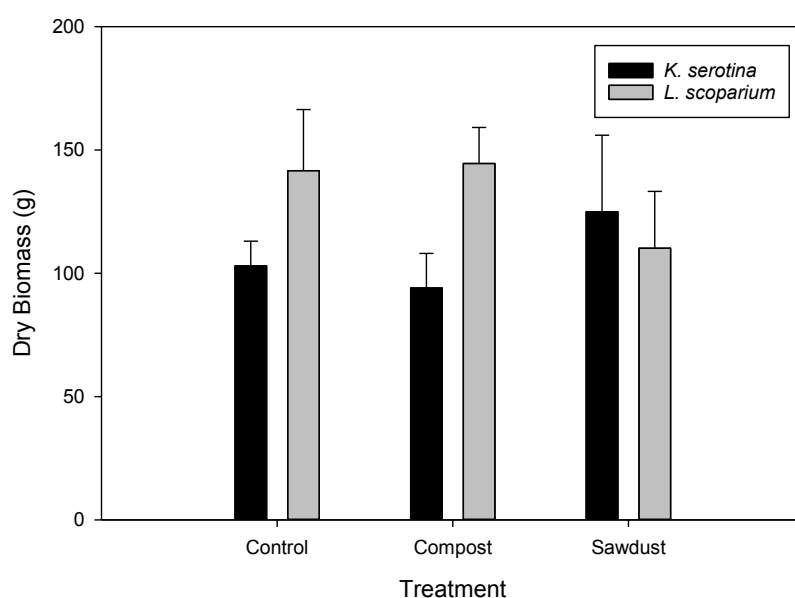


Figure 7.9 Dry biomass of *K. serotina* and *L. scoparium* in each treatment following 11 months growth. No result was significant ($p < 0.05$).

The control and glyphosate treatments retained more bare ground than the sawdust and compost treatments despite these latter two covering the plot at the start of the experiment producing 100% bare substrate. Weed species rapidly established in plots with sawdust and compost, up to 100% plant cover in many instances (Figure 7.10). There was no significant changes in the amount of naturally

regenerating *P. amoena* or total native species between the treatments although there was a general increase in abundance across all plots and a trend for more to be found within the control and glyphosate plots. Naturally regenerating species included *Senecio glomeratus*, *Euchiton sphericus*, *E. audax* and *Deyeuxia avenoides* (Plate 7.4).

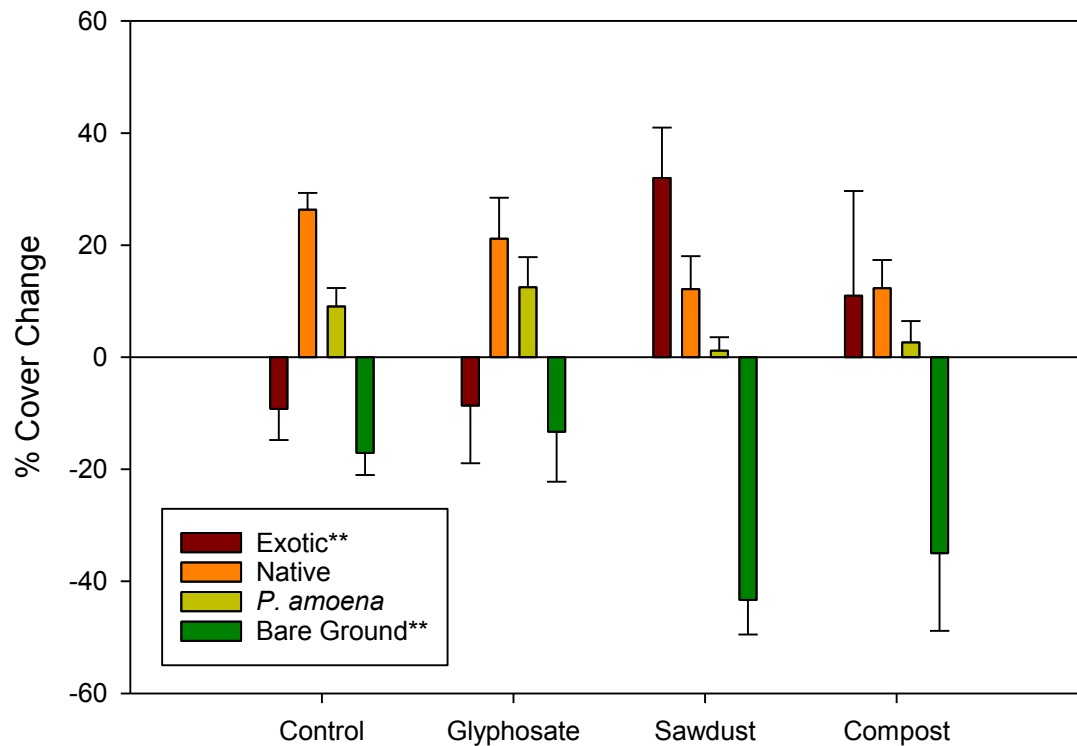


Figure 7.10 Change in percentage cover of bare ground and plant species in relation to treatment. ** denotes significance ($p < 0.05$).



Plate 7.4. Naturally regenerating native plants in Reserve 2: *K. serotina* (red arrow) and *P. amoena* (white arrow, left); and, although not found within experimental plots, *O. leptophyllus* (right).

7.3.5 Direct Seeding

Initial germination rate of seedlings was highest in the ripped reserve (624 compared to 58), but survival rates were low. Seeds only germinated in plots which had seeds added reflecting the lack of a soil seed bank. The highest survival rate was in the unripped reserve (Figure 7.11).

Survival rates for all species were low with a total of 2 *L. scoparium* seedlings surviving in the glyphosate treatment along with 8 *K. serotina*. *P. amoena* seedlings were found in the seeded treatment in both ripped and unripped condition and the glyphosate treatment in the ripped area (Figure 7.11).

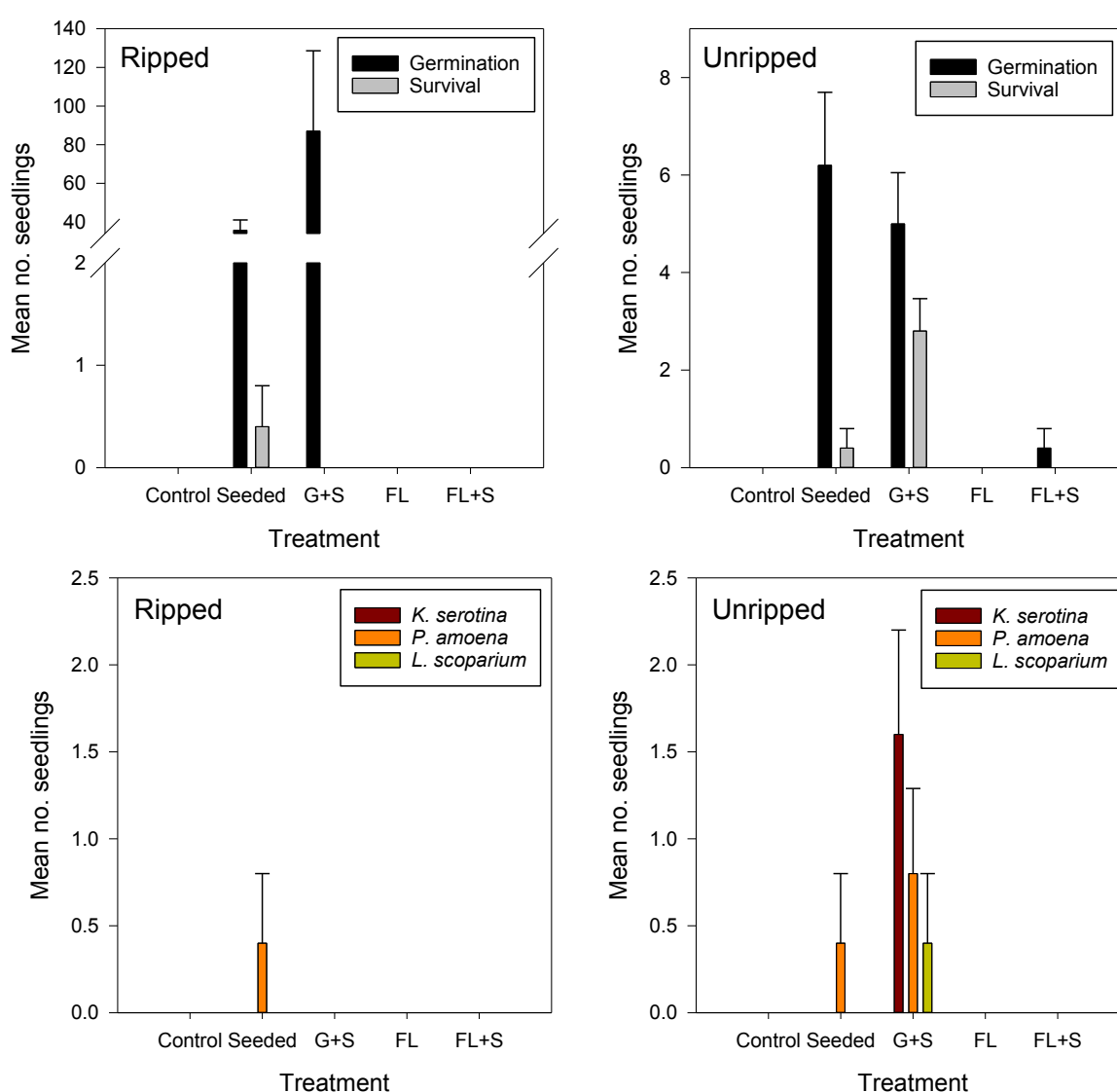


Figure 7.11 Mean germination and survival rate (\pm SEM) for seeds within treatments and conditions (top graphs, $p < 0.05$) and mean number of seedlings (\pm SEM) surviving after 9 months within each treatment separated by species (bottom graphs). Treatment codes are G+S=Glyphosate and seeded, FL= forest needle litter and FL+S= forest needle litter and seeded.

7.3.6 Moss Introduction

All moss turves survived in the field when placed around the base of planted seedlings (Plate 7.5). There was no growth of the moss turves, but all seemed healthy. The moss cover increased in the ripped area of the reserve whilst there was a significant decrease in the unripped reserve (Figure 7.12).



Plate 7.5. Moss turf placed around base of plant inside a tree guard (red, dashed line, tree guard removed for picture). The moss remained healthy for 1 year in the field. Picture taken in late summer.

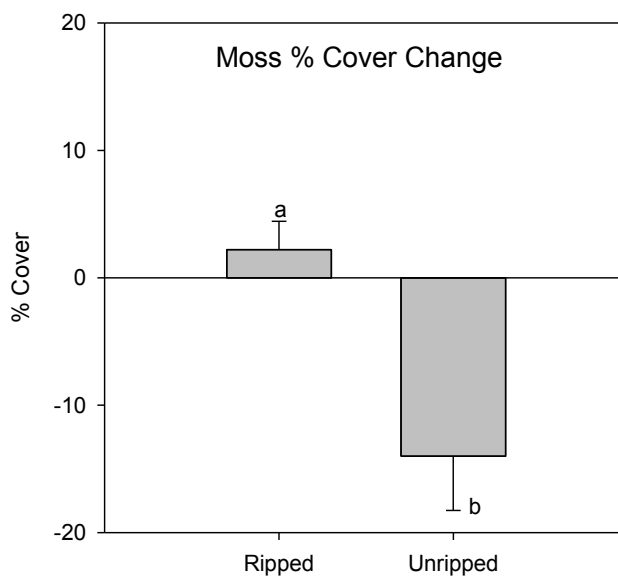


Figure 7.12 Mean change in percentage cover of moss (\pm SEM) when applied as fragments from a neighbouring pine forest stand and left to grow for 9 months. Letters denote significant difference ($p < 0.01$).

7.4 Discussion and Conclusions

7.4.1 Use of tree Guards

This study illustrated that using tree guards improved restoration success. TG1, the polythene tree guard held in place with four canes, was shown to be the most effective guard, reducing the mortality rates in both species. Hares were the only herbivore sighted within the area and plant wounds were consistent to those caused by hares, with bite marks of 45° angle (Wong and Hickling 1999). The levels of herbivory were generally low for *K. serotina* (less than 15%) reflecting the dietary preference for grasses and the unpalatable nature of the plant (Blay 1989; Farmilo et al. 2016). Whilst there is little existing knowledge of the plant-herbivore interactions for *P. amoena*, herbivory in this study was close to 50% without tree guards implying greater palatability. In addition to direct effects of herbivory and although not reported in this short-term study, detrimental indirect effects of unprotected plants may have unfolded in the longer-term such as herbivores preventing plants from producing viable seed leading to population depletion (Grüner and Norton 2006).

Herbivory was not the main cause of mortality for all plants. Competition from weed species is one of the main reasons for failure in planting schemes (Anton et al. 2015; Sullivan et al. 2009). Weed mats, coupled with the tree guards, limit competition from the surrounding plants (Grose 2012; Lai and Wong 2005; Sweeney et al. 2002). However, *K. serotina* plants within the TG2 treatment experienced over 1.5 times more mortality than those in TG1 despite having shelter and weed mat. For *P. amoena* the mortality rate of TG2 (the rigid corrugated plastic guard) was similar to those receiving no protection suggesting that herbivory or competition was not the main cause of death.

Mortality rates increased during the spring and summer months when the dry and warm north-westerly winds notable of the Canterbury Plains were evident (McGowan and Sturman 1996). In addition, 2015 had the lowest recorded historical rainfall, almost halving regional annual averages to approximately 300mm (Gabites et al. 2015; NIWA 2017). Desiccation is an important determinant of seedling mortality (Morales et al. 2016; Norton and Young 2016), and tree guards can induce water stress by increasing temperature inside the plastic sheath (Close et al. 2009; Dupraz and Bergez 1999; Kjølsgren and Rupp 1997). Where temperatures are high inside the sheath, ventilation has been found previously to be useful (Bergez and Dupraz 2000). Results from the present study showed that temperatures were generally lowered inside the sheath and TG2 offered ventilation to the plant from the base by creating gaps between the rigid plastic and the undulating ground, but mortality with this tree guard was high. TG1 guards had the highest survival rates and were able to completely enclose the plant, protecting it from the prevailing winds, due to the flexible polythene material which followed the contour of the ground and was held down with rocks.

Overall, desiccation in summer was the likely major cause of mortality in the experiment but tree guard usage is beneficial. TG1 protected seedlings through reduced air and soil temperature and maintaining soil moisture which probably also led to lower transpiration rates (Bergez and Dupraz 1997).

7.4.2 The Cost of Planting

The use of tree guards demands an initial financial investment but when factoring in costs due to plant losses from herbivory, competition and desiccation previous studies have found them to be cost efficient (Lai and Wong 2005). In the present study, the cost for each surviving *K. serotina* plant at the end of the experiment using the exact costings provided by the landscapers was \$17.42 for no tree guard, \$13.12 with TG2 and \$8.62 for TG1. The higher mortality rate in *P. amoena* led to costs of surviving plants being \$33.24 for no protection, \$13.64 for TG1 and \$43.20 for TG2 reflecting increased cost of a tree guard which offered minimal protection against desiccation. Therefore, even with the initial investment, TG1 proved to be the most cost efficient at the end of the experiment.

7.4.3 Fertiliser tablets

In agreement with previous glasshouse and field trials reported in Chapter 6 of this thesis and elsewhere, (Franklin 2014; Stevenson and Smale 2005), the fertiliser treatment showed no visual signs of damage to roots and had limited effect on the biomass of *K. serotina*. *L. scoparium* experienced significantly reduced biomass, height and mortality as a result of fertiliser addition, a result which was similar in the field with the addition of sawdust and dairy shed effluent (section 7.3.4, although not significant). This probably reflects the fact that native plants are well adapted to low nutrient environments (Craine and Lee 2003; Wardle 1991).

These results have to be interpreted with caution, and the length of the growing season may not have been long enough. In Australia, significant effects of fertiliser tablets on growth have been reported after just 6 months in north-eastern Tasmania where annual rainfall was 980 mm (Neilsen 1996) and after one year following a drought season (annual rainfall 882.1 mm) (Ruthrof et al. 2011). Fernández-Sanjurjo et al. (2014) found that 80% of N, P and K nutrients were leached from the tablet within the first 1.5 years of rainfall for an area with annual rain of 950 mm. Therefore, in the present study it can be assumed that the plants will have received some rhizosphere nutrient enrichment from the tablet. Rainfall reported in those other studies were higher than those occurring in Canterbury; the average annual rainfall for the Canterbury Plains is between 600-800 mm (Macara 2016) and in 2016 the average rainfall for the area was less than 550mm (NIWA 2017). Weggier et al. (2008) carried out trials on the effects of fertilisers with varying levels of drought and concluded that in times of reduced rainfall, although fertiliser addition may increase biomass, it can also increase drought stress. This may

have been the cause of negative effects of fertiliser on *L. scoparium* where an initial growth led to drought stress and mortality (White et al. 2009), leaving plants with lower biomass surviving. Previous studies which have reported increased biomass of *Kunzea* sp. and *L. scoparium* and with added nutrition have been undertaken in glasshouse environments where water stress was not a factor (Esperschuetz et al. 2017a; Esperschuetz et al. 2017b). Therefore, in low rainfall areas (annual precipitation of 630mm) site preparation and species selection may be more important than fertilisation (Ruthrof et al. 2013) and the importance of site conditions promoting early growth prior to periods of stress could assist survivorship (Ruthrof et al. 2011).

7.4.4 Weed Control

Mulch application can be an effective weed control method reducing both annual and perennial weeds, providing the mulches do not have weed seeds within them (Jodaugienė et al. 2006; Pupalienė et al. 2015; Ramakrishna et al. 2006). In contrast, the addition of organic material in this trial promoted weed invasion whilst the control and glyphosate treatments resulted in the least weed encroachment and the greatest native species regeneration.

The exotic and herbaceous weed species found within the Canterbury Plains are those synonymous with agriculture and horticulture (Meurk and Swaffield 2000) which have evolved in more fertile soils requiring nutrient input to survive in the low fertility soils of New Zealand (McLaren and Cameron 1996). It is anticipated that the mulches in this experiment provided nutrient input (Tiquia et al. 2002), increased soil moisture (Stratton and Rechcigl 1998) and introduced invertebrates which assist in nutrient cycling processes (Jordan and Jones 2007), and enabled the rapid spread of exotic species. Furthermore, although the mulch was spread to 10 cm thickness on plots, over time this depth reduced due to compaction of material. Research has shown that mulch depth less than 7.6 cm can enhance weed growth (Kuhns 1992). In contrast, droughty Lismore soils were maintained in their nutrient poor state within the control and glyphosate treatments and assisted in the prevention of exotic plant invasion due to resource limitation (Zefferman et al. 2015).

The conclusion of this part of the present study is that herbicide application for weed control is most effective and mulches should be avoided to prevent nutrient enrichment and encroachment of exotic species.

7.4.5 Direct Seeding

Native seedlings were only recorded in plots where seeds had been added, reflecting the depauperate seed bank of the area (Overdyck and Clarkson 2012; Partridge 1989; Sem and Enright 1996) and the proximity of the plots to an abundance of exotic seed and competition (Rowarth et al. 2007). The high

initial germination rates recorded in the ripped area may have been a result of the increased bare soil resource (Stevenson and Smale 2005) but very few of these survived. The majority of surviving seedlings were recorded mainly within those plots which had not been disturbed through ripping and with added seed and glyphosate application (Plate 7.6).



Plate 7.6. An unripped plot 9 months following glyphosate application showing the weed control (left) and a young kākūka seedling within the plot next to a pen (right).

The success of direct seeding projects is inextricably linked to the environmental conditions and particularly soil moisture and competition (Dodd and Power 2007; Douglas et al. 2007; Erickson et al. 2016). In this trial, whilst the effect of competition was somewhat alleviated in the ripped plots, the area experienced a drought (NIWA 2017) which was exacerbated by the act of disturbing the free draining, droughty Lismore soil (Blevins et al. 1971; Blevins 1984; Stevenson and Smale 2005). Conversely, it is assumed that competition within the unripped plots was higher, even in the glyphosate plots, leading to decreased germination but higher survival due to increased soil moisture as a result of the intact soil and the surrounding ground cover. These findings support those of Ledgard et al. (2008) who found that disturbing the soil led to an increased amount of seedlings except in the comparable site of Medbury Scientific Reserve, which has similar abiotic characteristics as the study site. Disturbing the soil in this study led to a decreased number of seedlings and the herbicide treatment, alleviating competition but retaining soil structure and moisture, produced the best results.

Mulch addition can increase soil moisture by preventing soil evaporation (Ji and Unger 2001; Prihar et al. 1996; Stratton and Rechcigl 1998) yet in this experiment there were no seedlings found within the forest litter/mulch treatment. The litter comprised of pine needles from the radiata pine trees with moss and lichen fragments (*Hypnum cupressiforme*, *Cladonia* sp., *Racomitrium lanuginosum*). Within the dry environment, the moss and lichen species may have exerted further

competition on the seeds and seedlings for moisture leading to mortality due to desiccation (Jeschke and Kiehl 2008). In addition, factors discussed in Chapter 5 of this thesis may have been in operation: litter layer creating a barrier to the soil (Jeschke and Kiehl 2008; Leemans 1991; Sedia and Ehrenfeld 2003; Van Tooren and Pons 1988); the litter trapping seedlings under the moss layer (Hörnberg et al. 1997; Zamfir 2000); pine litter, bryophytes and lichens exerting allelopathic effects upon the germination species (Lill et al. 1979; Michel et al. 2011a; Zamfir 2000); or a combination of many factors (Casanova-Katny and Cavieres 2012; Steijlen et al. 1995).

In conclusion, direct seeding of the reserve at Eyrewell produced poor results. The planting design developed for the restoration of the reserves at Eyrewell Forest aimed for at least one plant every 1-1.5 m. The results from this study would not achieve this goal. Furthermore, canopy cover is often a goal of restoration projects to assist in weed control (Standish 2002) and the amount of time taken for seeded individuals to overtop the weeds in this nutrient poor, dry environment would be considerable. Therefore, this method is deemed to be a complimentary technique used in addition to other methods.

7.4.6 Moss Introduction

Hydroseeding as a method for introducing bryophytes into the dry site of Eyrewell Forest were discounted following a series of glasshouse and field trials which recorded no results. The field conditions were too dry and led to death of moss after three months, even when protected by a polythene sheet to retain moisture (not reported here). The protection afforded by tree guard allowed all moss turves to remain healthy throughout the experiment. It is assumed that the decreased temperature and increased soil moisture assisted the survival of the moss although no significant growth of the mat was observed.

The moss applied as plant fragments in the ripped area of the reserve allowed for increase in cover of 2% whilst moss cover was reduced in control plots. This is likely due to the competition and shading from exotic grasses (van der Wal et al. 2005) and echoes findings of the field experiment in Chapter 6 where grass cover was negatively correlated with moss cover. The act of ripping the soil created bare ground and less competition from neighbouring weeds (Plate 7.7). Suppression of exotic weeds, including pasture grass (*Lolium perenne*), may have been further enhanced through application of pine needle litter which is thought to have allelopathic extracts inhibiting germination of seeds (Lill et al. 1979) and inhibit biological nitrification, further limiting encroachment of weed species (Cooper 1986; Suescun et al. 2012). Furthermore, the distribution of mosses is known to be partly governed by microsite conditions in an environment (Lee and Roi 1979; Watson 1980). The act of ripping the soil allowed for hollows in the plots which created favourable microclimates for moss growth.



Plate 7.7. Moss fragments and pine litter applied to plots in a control area (left) and a ripped area (right) of a reserve at Eyrewell Forest. Moss cover changed within the control plots due to competition effects from grass species.

7.4.7 Evaluation of the Implications for Restoration

The current suite of field experiments clarifies some of the main challenges facing restoration in the area. Aside from herbivory from hares, the most important difficulties to overcome are the protection of introduced propagules from desiccation, and preventing competition from weeds by retaining a low nutrient substrate. In light of the findings from these investigations the following recommendations to improve efficiency of restoration at the site are:

1. Vascular Plant Introductions – This study has shown that direct seeding of the area cannot be relied upon for complete restoration and is often not viable in times of drought (Chirino et al. 2009). However, this could be more economically feasible and may be a complimentary technique alongside planting when conditions are favourable (Ceccon et al. 2015). Further studies are required.
2. Non-vascular Plant Introductions – For successful introduction of moss the plants need to be protected against the effects of desiccation and competition from weed species.
3. Preventing Desiccation – The main factor affecting the survival of planted specimens in the area appeared to be desiccation in summer. To achieve successful establishment, protection is advised through the use of tree guards or else irrigation may be required through the dry summer and autumn months. However, it may also be beneficial to ensure the site remains at a low soil moisture gradient, creating stress and out-competing exotic weeds and grasses

(Meurk and Greenep 2003) as irrigating may relieve resource limitation and encourage weed growth (Devine et al. 2007). Further studies would be required to assess irrigation effects. Other practices to overcome water stress could be explored including direct watering techniques (Bainbridge 2002), alternative propagation techniques such as using hydrogel in the potting mix which retains moisture or developing plants with longer roots in deeper containers so that they can reach the lower soil horizons with more stable moisture concentrations (Chirino et al. 2009). The trial reported in this chapter suggests that weed mats are not essential to maintain soil moisture where tree guards are used.

4. Herbivory protection – It is recommended that tree guards or herbivore proof fencing be used where palatable species (e.g. *P. amoena*) are planted.
5. Fertiliser Use – The results from this trial suggest that for *K. serotina* and *L. scoparium* gain no apparent benefit from fertiliser tablets in the field, at least in the short-term, and the use may be detrimental to growth.
6. Weed Control – The addition of organic material into this nutrient poor area is likely to enhance growth and spread of exotic species. One exception would be nutrient poor forest litter from the pine stands that includes acidic needle litter, moss and lichen fragments which may inhibit seed germination of weeds.
7. Disturbance - Disturbance to soil is likely to favour encroachment of weed species and increased competition for vascular and non-vascular plants. Disturbance should be avoided.
8. Economics - With regard to cost, the tree guards present an initial financial outlay but offer protection against desiccation, herbivory and weed competition, and so, pay for themselves in planting success. The cost of fencing, irrigating and controlling weeds in large-scale projects can be expensive and far outweigh the cost of tree guards (Clapperton and Day 2001; Hayward and Kerley 2009; Lai and Wong 2005; Pimentel et al. 2002). In this experiment, the tree guard (TG1) saved \$8.80 per *K. serotina* plant and \$19.60 per *P. amoena* plant.

Chapter 8

Overall Conclusions, Recommendations for Ecological Restoration and Implications for Further Research

The conversion of the Eyrewell Forest into the irrigated dairy farms of Te Whenua Hou and this study of the opportunity for ecological restoration has provided valuable insight into the ecology of kānuka shrubland remnants on the Canterbury Plains. The work has also highlighted research gaps and opportunities into the most efficient and cost-effective techniques for ecological restoration. Conclusions are discussed here with reference to the objectives listed in Chapter 1 (pg. 5).

8.1 Community Composition, Ecology and Edaphic Conditions

- *Objective 1: To identify the composition, ecology and associated edaphic environment of indigenous dryland kānuka communities on the Canterbury Plains.*

This study has confirmed that kānuka stands in the Eyrewell Forest area are primarily characterised by low nutrient soil and a dry climate with seasonally extreme temperatures. The native vegetation and habitats contain plants with xerophytic adaptations which evolved in these conditions (Craine and Lee 2003; Molloy and Ives 1972; Wardle 1991). Total organic C in topsoil was approximately 2% higher in the remnant vegetation than in the disturbed kānuka stands under pine or in field boundaries, as was also the case for total N (0.08% higher). The topsoil was lower than either the pine forest or field boundary for Olsen P ($6.3 \mu\text{g g}^{-1}$ lower), lower mineral nitrogen ($2.4 \text{ NO}_3^- \text{-N mg L}^{-1}$ and $19.36 \text{ NH}_4^+ \text{-N mg L}^{-1}$ lower), pH (0.46 lower) and a lower C:N ratio (7.11 lower).

Remnant kānuka dominated habitats were found to contain most native species, corroborating the findings of Ecroyd and Brockerhoff (2005). These are represented as small island refugia, with wide distances between them, within which soil nutrients are not enriched, where there is a lesser degree of disturbance and where native propagules are present. Initially, the study focused on the canopy species, *K. serotina*. High germination rates, length of viability and lack of germination in dark conditions indicates that where propagules and small-scale disturbance are present with limited tall grass competition indicate that kānuka is sustainable as the dominant and single canopy species (also suggested by Allen et al. (1992)). Larger tree species originally present in, and adjacent to, these stands included totara (*Podocarpus totara*) (Harding et al. 2009) but this species has long disappeared from the landscape and will not emerge unless introduced.

A small shrub at the southern limit of its boundary, *Pomaderris amoena*, was thought, prior to the present work, to be difficult to germinate and transplant (New Zealand Plant Conservation Network 2014). There were concerns about the survival of the species in the Eyrewell Forest area once the forest was cleared, prompting a particular focus on this species. The present work found that small-scale disturbance within the habitat was important for germination and establishment of *P. amoena*. It was found in disturbed habitats of field boundaries and pine forest with relatively open canopies and the seeds only germinated in light conditions. Seed dormancy was successfully broken through heat treatment with boiled water. This dormancy breaking mechanism was further illustrated in the field by the natural, and prolific, regeneration of the species in an area (reserve 2, Figure 7.1) which had experienced fire two years previous (Glen Clayton (Ngāi Tahu Farming Ltd), pers. comm.) reported in Chapter 7 (Figure 7.10). Fire stimulated germination is relatively rare in New Zealand, and previously thought to be to mainly confined to kānuka and mānuka (Myrtaceae) with scant evidence for other species. The research presented in this these show that *P. amoena* may also be fire stimulated.

Other frequent woody understory plants included the native broom, *Carmichaelia australis*, and *Leptecophylla juniperina*. *C. australis* seeds showed no dormancy and germinated in light and dark conditions despite being found only in the remnant habitats with high canopy cover and shade. The lack of this species elsewhere is likely explained by lack of propagules and competition from exotic species, including nitrogen-fixing broom (*Cytisus scoparius*) and gorse (*Ulex europaeus*). *C. australis* is much less invasive and does not compete with these species. *L. juniperina* seeds could not be germinated in the laboratory or glasshouse despite repeated attempts but seedlings were found most frequently in the shadier, remnant habitats. Longer-term trials and alternative methods of propagation require further investigation to assist in restoration projects.

A particularly interesting component of the kānuka community was the moss ground layer, dominated by *Hypnum cupressiforme*. This species was present in every kānuka dominated habitat studied, with high percentage cover in the remnant vegetation stands. The limited research surrounding the functional role of moss in kānuka dryland shrubland indicated a requirement for further investigation and provided a strong focus for the present project. Further evaluation of any investigations into the ecological role of moss in kānuka shrubland is discussed in the following section.

The main findings from the community ecology study are summarised as:

- Remnant habitats are supported by low nutrient soil and contain a high percentage of moss ground cover;

- *K. serotina* (kānuka) is the single canopy species which grows well in these conditions, providing small-scale disturbance (due to its reliance on light conditions for germination) and propagules are present;
- *P. amoena* requires heat treatment to break seed dormancy, requires light conditions and small-scale disturbance but regenerates successfully in the field following fire;
- *C. australis* seeds can germinate in a range of conditions providing propagules are present and viable;
- *L. juniperina* is found in low shade, low disturbance environments and seed dormancy is very hard to break; and
- In addition to the species stated in the present project, 5 other species of shrubs, 21 forb species, 17 grasses and sedges and ferns are natural components of these ecosystems (see page 50) and require further study.

8.2 Importance of Non-Vascular Plants

- *Objective 2: To investigate the significance and ecological role of non-vascular plants within dryland kānuka vegetation with regard to chemistry, hydrology and biotic interactions.*

Hypnum cupressiforme carpets were found to have an important ecological role in the functioning of remnant kānuka communities in terms of influencing hydrology, nutrient cycling and biotic interactions. Soil moisture and temperature were affected by the moss layer with deeper layers of moss reducing fluctuations between summer and winter months through rainfall interception, water-holding capacity, infiltration, evapotranspiration, and an accumulation of insulating dead plant matter (>10 cm thick in places) affecting soil moisture evaporation and temperature (Chapter 4). By influencing the soil moisture gradient, mosses are significant drivers of community composition and, as such, crucial to ecosystem functioning. These findings are corroborated by studies elsewhere; Michel et al. (2013) argued that bryophytes were critical components to hydrological cycling in New Zealand grassland habitats.

In addition to the physical effects on soil of the moss layer, *H. cupressiforme* was found to alter soil nutrient status, reducing availability of nutrients; capturing nutrients and storing in tissue which is slow to decompose with minimal leaching of nutrients to soil from water percolating through the moss carpet. Concentrations of mineral nitrogen in the soil were influenced through interception and use, such that it caused a nodulation response in *C. australis* growing in a moss layer when no response was recorded without moss. Microbial communities can be modified by alteration of soil nutrition and the

action of plant extracts, suggested to be present in *H. cupressiforme*, leading to further importance of the moss to ecosystem functioning.

Despite the soil nutrient limitations beneath the moss layers, the presence of moss increased biomass of *P. amoena*, but decreased growth of *K. serotina*. The increase was suggested to be a function of the mosses effect on soil temperature and moisture whilst the decrease in mineral N concentration may be a limiting factor for *K. serotina*. Nonetheless, although moss did not prevent germination of the common woody native species of the kānuka community, there was a negative correlation between depth of moss layer and germination and establishment rate, due to shading of the seed and acting as a barrier to the soil beneath.

The present work indicates there is potential of moss carpets to limit nutrients available for invasive exotic weeds, assisting in retaining the indigenous species composition of the habitat. Additionally, the moss carpet may prevent the germination and survival of exotic plants, either by physical mechanisms or allelopathic effects. More research is required to establish whether the bryophyte and lichen species found within the dryland kānuka shrubland contain such chemicals and whether there is a deleterious effect upon native and/or exotic species.

The kānuka shrubland environment is being invaded and influenced by exotic species, and this knowledge of the effect of moss on germination and establishment of vascular plants may influence management. Concern is raised in the present study and elsewhere that varying growth strategies and root structures of exotic plants may assist them reaching the substrate beneath the moss layer and then benefitting from the buffering effect on soil moisture and temperature (Craine et al. 2001; Craine and Lee 2003).

The results from the suite of experiments on *H. cupressiforme* undertaken in the present work are summarised as:

- *H. cupressiforme* has a high water holding capacity and can retain moisture in the biotic environment, preventing percolation through the soil. The moss layer transfers moisture to the dry soil beneath through capillary and gravitational forces, retaining soil moisture between mean values of 16-26 % annually (compared with 13-34% without moss). The moss layer facilitates wetter soil in summer and drier soil in winter, of benefit to vascular plants;
- The moss layer can buffer soil temperature extremes, with fluctuations in winter of 0.4°C in winter and 0.7-1.0°C in summer depending on moss depth (cf. 1.3 °C in winter and 1.4°C in summer without moss);

- The moss layer intercepts and retains nutrients with little transfer to the soil through leachate or decomposition (which, in any case, is known to be slow). *H. cupressiforme* decreases soil concentrations of mineral N through interception, uptake and sequestration; and
- Moss mats reduce germination and establishment rates of vascular plants. The buffering of soil moisture and temperature within the moss mat can be beneficial to vascular plant biomass if their growth is not limited by N.

8.3 Drivers of Community Change and the Effect of Nutrient Spillover

- *Objective 3: To investigate the main environmental determinants of community composition and the effect of nutrient spillover on plants within the kānuka shrubland.*

The major influences of change on the community were soil nutrient conditions, disturbance and weed encroachment, each potentially impacting habitat restoration and sustainability (Plate 8.1). Soil nutrients exert a strong influence on the communities of the dryland kānuka shrublands. Glasshouse pot experiments showed that some of the native woody species (*K. serotina*, *K. robusta* and *P. amoena*) increased growth and biomass as a response to nitrogen, but phosphorus fertilisation tended to produced negligible or detrimental responses. These results were supported in the field experiments where either organic matter or fertiliser tablets were added (as discussed below). However, exotic species, especially grasses, had a positive growth response to phosphorus and soil nutrients, as found in other studies in similar environments.



Plate 8.1. Invasion of certain areas of dryland kānuka shrubland by exotic grasses such as *Dactylis glomerata* and the invasive herb *Hieracium pilosella*.

Nitrate and phosphate concentrations in the field were correlated with lower moss cover and high exotic grass cover within the remnant areas. The relationship between nutrients and encroaching weed species was also illustrated with the addition of organic matter (sawdust and compost) in the form of mulch in the field. This represented a positive feedback in which higher nutrient content within the plant litter of exotic species led to further enrichment of soil nutrients and promoted further weed invasion. Effects of nutrient spillover in the remnant kānuka habitats were observed up to 30 m into the stands, showing a significant edge effect on these small remnant kānuka habitats (Plate 8.2). Research into potential mitigation strategies for lowering the leaching and runoff of nutrients from intensive agriculture into neighbouring habitats is crucial; this is discussed in section 8.6.



Plate 8.2. Small remnant kānuka stand that has experienced large edge effects of nutrient enrichment from surrounding land use and been encroached by exotic grasses preventing regeneration of native species.

The kānuka stands had a higher incidence of exotic species. In addition, where disturbance had been a factor, such as in habitats directly adjacent to agricultural land and subjected to frequent nutrient runoff, cattle and herbivore browsing. It is known that kānuka cannot regenerate within swards of long grass due to the seed requiring light for germination. Therefore, if grasses encroach, recruitment would cease and old and senescent canopy plants would not be replaced. Weed control and mitigation for nutrient enrichment appears to be of paramount importance in the kānuka remnants.

The findings of the experiments are summarised as:

- Native plants have varying responses to enhanced nutrition; *K. serotina*, *K. robusta* and *P. amoena* all responded with increased growth to enhanced soil nitrogen. *C. australis* increased biomass in soil amended with lime; inhibition of nodulation was found with N fertilisation and in the presence of NO_3^- . *L. juniperina* showed no significant growth response to nutrient amended soils. The moss, *H. cupressiforme*, had increased growth in low lime conditions, when lime was sprinkled on the surface of the moss carpets whilst superphosphate application produced a negative growth response;
- Nutrient spillover, particularly of nitrate and Olsen P, was detected at least 30m into the remnant kānuka habitats. Buffer zones are essential and a 5m buffer zone was originally integrated into the design of restoration plots at Eyrewell Forest;
- Weed encroachment into the remnant habitats was positively correlated with nutrient enrichment, and negatively correlated with moss cover.

8.4 Restoration Techniques

- *Objective 4: To evaluate appropriate methodologies and techniques to restore dryland kānuka habitats into agricultural landscape matrices.*

The Canterbury Plains has a limited diversity of native species, and highly restricted sources of propagules; therefore, plant introductions are required for ecological restoration. Moreover, cost-effective techniques for restoration on a large-scale is required. In the present project 150 ha of standard restoration planting (1.5 m x 1.5 m) would require perhaps 750, 000 plants. At NZ\$7 each, including planting and tree guards, this would cost approximately NZ\$5.3M. Therefore we need to create a template beyond which natural processes will lead to an effective ecological restoration.

The results of the present study illustrated that certain techniques are more appropriate for the dry, nutrient poor stony soils the in Eyrewell Forest region. In all experimental work, the threat of desiccation to young plants was evident. Direct seeding produced varied but low responses due to a period of drought and it is concluded that seeding alone is unlikely to lead to a satisfactory restoration, but a direct seeding may be a useful supplementary method of introducing propagules to an area alongside planting nursery grown specimens. Tree guards were the most cost-effective technique for survivorship when planting seedlings, reducing effects of herbivory and desiccation, and increasing soil moisture and reducing air and soil temperature around the plant. However, the initial investment in

tree guards can be high and other techniques promoting plant survivorship and growth within reserves which do not use tree guards may be required. Cost-effective methods of fencing, chemical herbivore repellents, and irrigation methods need to be explored.

The addition of fertiliser or nutrients had negligible or detrimental growth response for the native species studied and increased cover of competitive weed species. The sawdust and compost used within the weed trial encouraged weed encroachment through addition of nutrients and the suspected soil moisture improvements. The only exception to this was when using mulch which contained materials that potentially have an allelopathic effect on germinating seeds of exotic weeds. The pine needle litter mixed with moss and lichen fragments appeared to prevent seedlings from germinating, including native seeds, but further research is required to investigate this. It is possible that this litter may be useful around planted seedlings to increase soil moisture but retain the low nutrient status and preventing weed encroachment.

Moss introduction was successful when weeds were suppressed and plants were protected from desiccation through shelter by tree guard or the creation of hollows to modify the microsite environment. The combined effect of ripping the soil and applying pine needle litter reduced weed encroachment and allowed moss growth. In the weed control experiment in the field, herbicide was the best method of limiting weed encroachment and promoting native seed germination. This was considered to be due to a lack of soil disturbance which can stimulate weed seed germination and fact that it did not add further nutrients into the soil for invading weed species.

The results from this experimental work is summarised as:

- The development of a cost-effective ecological restoration template that restores ecosystem processes is essential;
- Polythene tree guards provide a cost-effective protection of young plants protecting plants from prevailing winds, increasing soil moisture and decreasing ambient temperature around the plant in summer;
- Addition of fertiliser tablets, or mulch, increased weed encroachment and had negligible or detrimental effects on *L. scoparium* and *K. serotina* growth;
- Weeds were best controlled through use of herbicide and the use of pine litter;
- Direct seeding is not appropriate as a stand-alone plant introduction technique at Eyrewell due to extreme environmental stress; and

- Moss can be successfully introduced as fragments or turves providing it is protected for competition and desiccation by use of tree guards, creation of hollows and application of pine needle litter.

8.5 Further Research

To ensure successful restoration of kānuka habitats, it is suggested that further insight into important interactions and effects of abiotic and biotic components of the community are gained. In summary, and as discussed in relevant sections above, this includes:

- The high soil aluminium concentrations in the Eyrewell Forest area, the low soil pH and the potential implications to plants within the acidic environment merits further discussion and study. Aluminium can be highly phytotoxic to some species whereas others have developed mechanisms to tolerate or exclude it from cells. Responses of the main community associates, such as *K. serotina*, *L. juniperina* and *P. amoena*, in addition to the most frequent exotic species such as *Anthoxanthum odoratum*, *Hypochaeris radicata* and *Hieracium pilosella*, would provide a worthwhile further study. Investigation into habitat partitioning with regard to soil aluminium could have useful implications for management of remnant areas and control of exotic weed species (Andersen et al. 2010)
- Exploration of seed germination and propagation techniques for a wider range of species naturally found in those habitats which are difficult to grow, to ensure full diversity is restored in the habitats;
- More detailed investigation of the Interaction of moss layer with germination, establishment and growth of native and exotic flora and fauna to determine facilitating and preventative effects. This should include investigation of potential allelopathic effects of *H. cupressiforme* and other mosses in the kānuka shrubland communities, across trophic levels, found in the kānuka stands;
- Further trials on plant survivorship and growth within planted reserves investigating alternatives to tree guards are required. Different cost-effective methods including fencing, chemical herbivore repellents, and irrigation trials need to be explored; and
- Implementation and mitigation and a number of mitigation strategies for lowering the spillover and runoff of nutrients from intensive agriculture into neighbouring habitats.

8.5.1 Conclusion

The overarching aim of the studies was:

- *to understand and assist the design of successful dryland kānuka shrubland restoration into an intensively farmed landscape matrix on the Canterbury Plains.*

The present work has provided valuable insights into the ecology of the dryland kānuka shrubland ecosystems and options for ecological restoration. Ecological restoration presented challenges associated with hot, dry summers, stony nutrient poor soils, fertiliser spillover and invasive weeds. One of the main findings is of the importance of moss carpets in the structure and function of dryland kānuka stands. The inclusion in restoration may be critical for establishing and sustaining ecosystem structure, functionality and resilience.

Studies reported in this thesis have informed the practical restoration of Te Whenua Hou, culminating in a more accessible and readable contractual document for the sponsors of the work (Appendix C). The research presented in this thesis outlines the importance of moss inclusion in ecological restoration but also highlights challenges for traditional and future restoration practitioners. This work can be used as a template for similar schemes aiming to restore large areas. The process of thoroughly investigating the remnant and target habitats, formulating restoration goals, implementing and testing restoration technology and monitoring results has contributed to existing scientific knowledge of relevance to ecological restoration.

At the time of writing, planting and experimental restoration work has been established in 11 of the 18 reserves beginning with a minimal 5 m buffer zones planted with kānuka. The current field work took place in 4 of these restoration areas. Novel approaches are constantly being investigated, including land owners trialling direct seeding with treated mānuka seeds sown from a helicopter over large areas. In addition, the scheme is in the process of re-introducing thousands of the emblematic New Zealand tree, totara (*Podocarpus totara*). This represents substantial biodiversity and social gain due to the species spiritual importance to Māori, and the timber used for canoes, carving and building. Furthermore, the wider project is assessing the benefit of the scheme to native birds, invertebrates and lizards for which the restoration will provide a habitat resource. Restoration of areas within the Te Whenua Hou footprint is ongoing and future work will incorporate all findings from the studies presented here and build on this knowledge base.

Appendix A

Plant Species and Abbreviations

List of frequently used plant names and abbreviations used within the thesis.

Scientific Name	Family	Abbreviated Name	English and Māori Names
<i>Carex goyenii</i> Petrie (1882)	Cyperaceae	<i>C. goyenii</i>	Goyens sedge
<i>Carmichaelia australis</i> R. Br (1825)	Fabaceae	<i>C. australis</i>	Common broom, mākaka, maukoro, tainoka, tawao
<i>Leptecophylla juniperina</i> (J.R Forst. & G Forst) C. M Weiller (1999)	Ericaceae	<i>L. juniperina</i>	Prickly mingimingi, tauhinu, prickly heath,
<i>Leptosperma scoparium</i> J.R Forst. & G Forst (1776)	Myrtaceae	<i>L. scoparium</i>	Mānuka, kāhikatoa, red tea tree
<i>Hypnum cupressiforme</i> Hedw. (1801)	Hypnaceae	<i>H. cupressiforme</i>	Cypress-leaved plaitmoss, hypnum moss
<i>Ozothamnus leptophyllus</i> (G.Forst.) Breitw. & J.M.Ward (1997)	Compositae	<i>O. leptophyllus</i>	Cottonwood, tauhinu, tauhinu koromiko, tauhinu korokio
<i>Pomaderris amoena</i> Colenso (1886)	Rhamnaceae	<i>P. amoena</i>	Tauhinu, toehinu, whititiri
<i>Kunzea robusta</i> De Lange & Toelken (2014)	Myrtaceae	<i>K. robusta</i>	Kānuka, white tea tree
<i>Kunzea serotina</i> De Lange & Toelken (2014)	Myrtaceae	<i>K. serotina</i>	Kānuka, white tea tree

List of frequently used abbreviations within thesis:

Aluminium	Al or Al ³⁺	Biological Soil Crust (see section 2.3.3)	BSC
Ammonium	NH ₄ ⁺	Department for Conservation	DOC
Carbon	C	Eyrewell Scientific Reserve (see section 3.2)	ESR
Calcium	Ca	Spencer-Bower Reserve (see section 3.2)	SBR
Copper	Cu	Total Organic Carbon	TOC
Iron	Fe		
Potassium	K		
Magnesium	Mg	Nitrogen	N
Manganese	Mn	Phosphorus	P
Nitrate	NO ₃ ⁻	Zinc	Zn

Appendix B

Supplementary Information for Chapter 3

B.1 List of species found within kānuka habitats within the Eyrewell Forest

Species (native in bold type) found within the field survey either within the remnant shrubland vegetation (R), the field boundary kānuka stands (F) or within the pine forest (P). The successional status is also noted as pioneer species (Pi), secondary species (Se) or primary species (Pr).

Species	R	F	P	Species	R	F	P
<u>Grasses and Sedges</u>				<u>Herbs</u>			
Pi <i>Agrostis capillaris</i>	X	X	X	Pi <i>Anagallis arvensis</i>		X	
Pi <i>Anthoxanthum odoratum</i>	X	X	X	Se <i>Asplenium flabellifolium</i>	X		
Pi <i>Bromus stamineus</i>	X	X		Pi <i>Chenopodium album</i>		X	X
Se <i>Festuca novae-zelandiae</i>	X	X		Pi <i>Cirsium vulgare</i>		X	
Pi <i>Festuca rubra</i>	X	X	X	Pi <i>Dianthus armeria</i>		X	X
Pi <i>Dichlachna crinita</i>	X			Se <i>Dichondra repens</i>	X		
Pi <i>Dactylis glomerata</i>	X	X		Pi <i>Euchiton audax</i>	X		
Se <i>Lolium perenne</i>		X		Pi <i>Euchiton Sphaericus</i>		X	X
Se <i>Poa pulsilla</i>	X			Pi <i>Gallium aparine</i>	X		
Pi <i>Rhytidosperra gracile</i>	X		X	Se <i>Geranium sp.</i>	X	X	X
Pi <i>Vulpia bromoides</i>	X			Pi <i>Crepis vesicaria ssp. taraxifolia</i>		X	
Se <i>Carex sp.</i>	X	X		Pi <i>Hieracium pilosella</i>	X	X	X
<u>Woody Plants</u>				Pi <i>Hypochaeris radicata</i>	X	X	X
Pi <i>Carmichaelia australis</i>	X			Pi <i>Lapsana communis</i>	X	X	
Se <i>Coprosma propinqua</i>	X			Se <i>Leptinella pusilla</i>	X		
Pr <i>Coprosma crassifolia</i>	X			Se <i>Leucopogon fraseri</i>		X	X
Se <i>Coprosma sp.</i>	X			Pi <i>Lotus suaveolens</i>	X		
Pi <i>Cytisus scoparius</i>	X	X	X	Pi <i>Platago lanceolata</i>		X	X
Pr <i>Kunzea serotina</i>	X	X	X	Pi <i>Microtis unifolia</i>	X		X
Se <i>Leptecophylla juniperina</i>	X	X	X	Pi <i>Rumex acetosella</i>	X	X	X
Se <i>Melicytus alpinus</i>	X			Pi <i>Senecio glomeratus</i>	X	X	X
Pi <i>Ozothamnus leptophyllus</i>	X			Pi <i>Senecio jacobaea</i>		X	X
Pi <i>Pinus radiata</i>	X	X	X	Pi <i>Sonchus asper</i>		X	
Pi <i>Pomaderris amoena</i>		X	X	Pi <i>Sonchus oleraceus</i>		X	
Se <i>Rubus fruticosus agg.</i>		X	X	Pi <i>Stellaria media</i>	X	X	
Pi <i>Ulex europaeus</i>	X	X		Se <i>Thelymitre longifolia</i>		X	X
<u>Climbers</u>				Pi <i>Trifolium arvensis</i>		X	
Se <i>Clematis quadribracteolata</i>	X			Pi <i>Trifolium dubium</i>	X		
				Pi <i>Trifolium repens</i>		X	
				Pi <i>Verbascum thapsus</i>		X	X
				Pi <i>Vicia sativa</i>		X	
				Pi <i>Taraxacum officinale agg.</i>		X	X

B.2 Independent soil analysis results for the Eyrewell Forest



agri-nutrients
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ANALYSIS REPORT

Page 3 of 5

Client:	Ngai Tahu	Lab No:	776780	s2chpr1
Address:	PO Box 13-0060 CHRISTCHURCH	Date Registered:	20-Mar-2010	
		Date Reported:	06-Apr-2010	
		Quote No:		
		Order No:		
		Client Reference:		
		Submitted By:	M Kearney	

Sample Name: Carleton Hunter S Lab Number: 776780.5 Sample Type: SOIL Barley (S155)						Sample Name: Carleton Hunter N Lab Number: 776780.6 Sample Type: SOIL Barley (S155)					
Analyte	Level	Optimum	Below	Optimum	Above	Analyte	Level	Optimum	Below	Optimum	Above
pH	pH Units	5.0	5.7 - 6.2			pH	pH Units	5.0	5.7 - 6.2		
Olsen Phosphorus	mg/L	12	20 - 30			Olsen Phosphorus	mg/L	14	20 - 30		
Potassium	MAF units	8	6 - 12			Potassium	MAF units	8	6 - 12		
Calcium	MAF units	< 2	6 - 14			Calcium	MAF units	< 2	6 - 14		
Magnesium	MAF units	14	12 - 25			Magnesium	MAF units	10	12 - 25		
Sodium	MAF units	5	0 - 24			Sodium	MAF units	5	0 - 24		
Sulphate Sulphur	mg/kg	7	10 - 15			Sulphate Sulphur	mg/kg	12	10 - 15		
Aluminium (CaCl ₂ Extractable)*	mg/kg	20.4	0.0 - 3.0			Aluminium (CaCl ₂ Extractable)*	mg/kg	26.6	0.0 - 3.0		
Available Nitrogen (15cm Depth)*	kg/ha	44	100 - 150			Available Nitrogen (15cm Depth)*	kg/ha	43	100 - 150		
Anaerobically Mineralisable N*	µg/g	36				Anaerobically Mineralisable N*	µg/g	30			
Soil Sample Depth*	0-150 mm					Soil Sample Depth*	0-150 mm				
Base Saturation % me/100g	K 2.8 Ca 7 Mg 4.5 Na 0.7 K 0.47 Ca 1.2 Mg 0.75 Na 0.12					Base Saturation % me/100g	K 2.8 Ca 5 Mg 3.2 Na 0.8 K 0.38 Ca 0.7 Mg 0.44 Na 0.11				
Additional Properties	Cation Exchange Capacity (me/100g) Total Base Saturation (%) Volume Weight (g/mL)	17 15 0.80				Additional Properties	Cation Exchange Capacity (me/100g) Total Base Saturation (%) Volume Weight (g/mL)	14 12 0.96			

The above nutrient graph compares the levels found with reference interpretation levels. NOTE: It is important that the correct sample type be assigned, and that the recommended sampling procedure has been followed. R J Hill Laboratories Limited does not accept any responsibility for the resulting use of this information. IANZ Accreditation does not apply to comments and interpretations, i.e. the 'Range Levels' and subsequent graphs.

Analyst's Comments

Samples 1-6 Comment:

For high potential yield crops, desirable soil test levels for nutrients are towards the top of the medium or optimum range.

Samples 1-6 Comment:

The guidelines for interpretation of 0.02M CaCl₂ Extractable Aluminium are: Less than 3 mg/kg unlikely to be toxic to plants; 3-10 mg/kg may be toxic to plants in soils with low Organic Matter; greater than 10 mg/kg toxic to plants.

ANALYSIS REPORT

Page 2 of 2

Ngai Tahu
PO Box 13-0060
CHRISTCHURCH

Lab No: 776780
Date Registered: 20-Mar-2010
Date Reported: 06-Apr-2010
Quote No:
Order No:
Client Reference:
Submitted By: M Kearney

New Block 1st Left 776780.1 SOIL Barley (S158)					Sample Name: Ryecorn 1st Right Lab Number: 776780.2 Sample Type: SOIL Barley (S158)						
	Level	Optimum	Below	Optimum	Above	Analysis	Level	Optimum	Below	Optimum	Above
pH Units	5.0	5.7 - 6.2				pH	pH Units	5.3	5.7 - 6.2		
N mg/L	18	20 - 30				Organic Phosphorus	mg/L	28	20 - 30		
AF units	8	8 - 12				Potassium	MAF units	9	8 - 12		
AF units	<2	8 - 14				Calcium	MAF units	6	8 - 14		
AF units	14	12 - 25				Magnesium	MAF units	15	12 - 25		
AF units	4	0 - 24				Sodium	MAF units	4	0 - 24		
mg/kg	15	10 - 15				Sulphate Sulphur	mg/kg	37	10 - 15		
P mg/kg	20.2	0.0 - 3.0				Aluminium (CaCl2 Extractable)*	mg/kg	4.0	0.0 - 3.0		
K kg/ha	45	100 - 150				Available Nitrogen (15cm Depth)*	kg/ha	70	100 - 150		
µg/g	38					Anaerobically Mineralisable N*	µg/g	63			
h*	0-150 mm					Soil Sample Depth*	0-150 mm				
K	K 2.8	Ca 9	Mg 4.4	Na 0.7		Base Saturation %	K 2.8	Ca 30	Mg 4.8	Na 0.8	
me	K 0.48	Ca 1.5	Mg 0.73	Na 0.11		me/100g	K 0.81	Ca 8.5	Mg 1.05	Na 0.12	
me	Cation Exchange Capacity (me/100g)				17	Additional Properties	Cation Exchange Capacity (me/100g)				22
	Total Base Saturation (%)				17		Total Base Saturation (%)				38
	Volume Weight (g/mL)				0.83		Volume Weight (g/mL)				0.1

Lindsey Hunter N 776780.3 SOIL Barley (S158)					Sample Name: Sedon Lindsey NW Lab Number: 776780.4 Sample Type: SOIL Barley (S158)						
	Level	Optimum	Below	Optimum	Above	Analysis	Level	Optimum	Below	Optimum	Above
pH Units	5.4	5.7 - 6.2				pH	pH Units	5.1	5.7 - 6.2		
N mg/L	20	20 - 30				Organic Phosphorus	mg/L	23	20 - 30		
AF units	8	8 - 12				Potassium	MAF units	8	8 - 12		
AF units	5	8 - 14				Calcium	MAF units	5	8 - 14		
AF units	7	12 - 25				Magnesium	MAF units	15	12 - 25		
AF units	8	0 - 24				Sodium	MAF units	4	0 - 24		
mg/kg	50	10 - 15				Sulphate Sulphur	mg/kg	50	10 - 15		
P mg/kg	1.4	0.0 - 3.0				Aluminium (CaCl2 Extractable)*	mg/kg	6.5	0.0 - 3.0		
K kg/ha	58	100 - 150				Available Nitrogen (15cm Depth)*	kg/ha	57	100 - 150		
µg/g	43					Anaerobically Mineralisable N*	µg/g	43			
h*	0-150 mm					Soil Sample Depth*	0-150 mm				
K	K 2.3	Ca 35	Mg 2.8	Na 1.4		Base Saturation %	K 2.4	Ca 27	Mg 4.3	Na 0.6	
me	K 0.33	Ca 5.0	Mg 0.38	Na 0.20		me/100g	K 0.44	Ca 4.8	Mg 0.77	Na 0.11	
me	Cation Exchange Capacity (me/100g)				14	Additional Properties	Cation Exchange Capacity (me/100g)				18
	Total Base Saturation (%)				42		Total Base Saturation (%)				34
	Volume Weight (g/mL)				0.88		Volume Weight (g/mL)				0.1

ANALYSIS REPORT

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Client: Ngai Tahu Forest Estates Ltd	Lab No: 1140193	s2chpvt
Address: PO Box 130060	Date Registered: 29-May-2013	
CHRISTCHURCH 8141	Date Reported: 05-Jun-2013	
	Quote No: 54854	
	Order No: P. Roberts	
	Client Reference: 4038010	
	Add. Client Ref: Eyrewell	
	Submitted By: Mr R Williams (KA)	

Sample Name: Sth Herrington Rd West Lab Number: 1140193.1 Sample Type: SOIL Mixed Pasture, Dry Stock (Sed.) (S188)		Sample Name: Davis Rd East (Blue 8B) Lab Number: 1140193.2 Sample Type: SOIL Mixed Pasture, Dry Stock (Sed.) (S188)	
Analysis	Level	Optimum	Below
pH	pH Units	5.2	5.8 - 6.0
Olsen Phosphorus	mg/L	12	20 - 30
Potassium	MAF units	8	8 - 8
Calcium	MAF units	2	4 - 8
Magnesium	MAF units	19	8 - 10
Sodium	MAF units	3	
Sulphate Sulphur	mg/kg	7	10 - 12
Aluminium (CaCl ₂ mg/kg Extractable)		13.7	0.0 - 3.0
Total Carbon	%	4.0	
Total Nitrogen	%	0.19	0.30 - 0.60
C/N Ratio*		20.8	
Soil Sample Depth*	mm	0-150	
Base Saturation %		K 2.8 Ca 11 Mg 6.1 Na 0.5	
me/100g		K 0.49 Ca 2.0 Mg 1.08 Na 0.09	
Additional Properties		Cation Exchange Capacity (me/100g)	18
		Total Base Saturation (%)	20
		Volume Weight (g/mL)	0.78
Sample Name: Sth Hunter Rd East (Green 15) Lab Number: 1140193.3 Sample Type: SOIL Mixed Pasture, Dry Stock (Sed.) (S188)		Sample Name: Sth Hunter Rd West (Green 9B) Lab Number: 1140193.4 Sample Type: SOIL Mixed Pasture, Dry Stock (Sed.) (S188)	
Analysis	Level	Optimum	Below
pH	pH Units	5.3	5.8 - 6.0
Olsen Phosphorus	mg/L	11	20 - 30
Potassium	MAF units	7	8 - 8
Calcium	MAF units	1	4 - 8
Magnesium	MAF units	15	8 - 10
Sodium	MAF units	3	
Sulphate Sulphur	mg/kg	6	10 - 12
Aluminium (CaCl ₂ mg/kg Extractable)		12.9	0.0 - 3.0
Total Carbon	%	3.5	
Total Nitrogen	%	0.18	0.30 - 0.60
C/N Ratio*		21.1	
Soil Sample Depth*	mm	0-150	
Base Saturation %		K 2.7 Ca 10 Mg 5.5 Na 0.8	
me/100g		K 0.43 Ca 1.5 Mg 0.87 Na 0.09	
Additional Properties		Cation Exchange Capacity (me/100g)	16
		Total Base Saturation (%)	19
		Volume Weight (g/mL)	0.77
Analysis	Level	Optimum	Below
pH	pH Units	5.2	5.8 - 6.0
Olsen Phosphorus	mg/L	12	20 - 30
Potassium	MAF units	7	8 - 8
Calcium	MAF units	1	4 - 8
Magnesium	MAF units	15	8 - 10
Sodium	MAF units	3	
Sulphate Sulphur	mg/kg	7	10 - 12
Aluminium (CaCl ₂ mg/kg Extractable)		16.3	0.0 - 3.0
Total Carbon	%	4.2	
Total Nitrogen	%	0.20	0.30 - 0.60
C/N Ratio*		20.9	
Soil Sample Depth*	mm	0-150	
Base Saturation %		K 2.5 Ca 8 Mg 5.0 Na 0.8	
me/100g		K 0.40 Ca 1.2 Mg 0.79 Na 0.09	
Additional Properties		Cation Exchange Capacity (me/100g)	16
		Total Base Saturation (%)	18
		Volume Weight (g/mL)	0.82

Appendix C

Restoration Report Summary

Executive summary page from a contractual report to project sponsors detailing restoration. For full report please email author.

Executive Summary

This report offers a non-technical summary of research, development and monitoring undertaken to enable restoration of 150 ha of indigenous habitat onto the Canterbury Plains at Te Whenua Hou dairy conversion.

The Ecology Department at Lincoln University has worked collaboratively Ngāi Tahu Farming Ltd, Mana Whenua Working Group and associated stakeholders (including landscape designers, landscape contractors, DOC, ECAN and private land owners) from Sept 2013 – Aug 2016. The scope of the work was:

- To provide a baseline study of ecological features of remnant vegetation, monitoring flora and faunal species across the Te Whenua Hou site.
- To assist in the design and establishment of reserves in an efficient and cost effective manner.
- To devise a template for the methodology and implementation for the restoration in a lowland dryland landscape.
- To evaluate the ecological and environmental benefits of the restoration activity.

The restoration is designed around the major woody associates found in existing areas of protected vegetation and within remnant stands of natural vegetation. A key discovery of the research has been the identification of the important functional role of non-vascular plants and how these communities should also be an integral part of the restoration. The first stage of the restoration plantings have focussed on the early establishment of kānuka and mānuka as canopy species. Subsequent additional inter-planting is required to restore the composition and function of the habitats and to create sustainable, regenerating ecosystems. Further research into how this can be successfully achieved is required, with significant additional intervention. This requires the introduction of species of local provenance that are currently largely unavailable from nurseries and of which there is little knowledge of their propagation beyond recognition of significant difficulties. Biodiversity values within this landscape has a unique identity, largely revolving a rich assemblage of invertebrates, lizards and birds. The return of this faunal component into the landscape is the major

challenge; restoring appropriate habitat for this to occur will validate the success of the restoration in the longer term. Meanwhile, progress is being made with onward and detailed monitoring. Improvements have already been identified in bird communities, skink presence, high beetle and moth diversity, three flatworm species, leaf-vein slugs, weta and trap-door spiders. The main risk and challenges to the future success of the restoration are:

- Not restoring a satisfactory complement of additional plants and animal species in the reserves,
- Not protecting the reserves against potentially adverse abiotic conditions such as nutrient enrichment,
- Not protecting the reserves against future disturbance and development, and
- A failure to appreciate that ongoing restoration monitoring and research is required.

The report also documents the beneficial ecosystem services that the restoration scheme potentially delivers to Te Whenua Hou, including increased biodiversity, opportunities for provisioning (mānuka/kānuka honey, rongoā prospects), recreational opportunities and regulating benefits (such as carbon sequestration, increasing the soils ability to act as a sink for greenhouse gases, and improving soil quality).

Further targeted research and restoration plantings are required to ensure the completion and success of the project. Recommendations for future research and plantings are included within the report.

Keywords: Ecological restoration, kānuka shrubland, *Hypnum cupressiforme*, moss, hydroseeding, biodiversity

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