Assessment of seedling recruitment under Manuka (*Leptospermum scoparium*) and Kanuka (*Kunzea ericoides*) plantings at Shakespear and Wenderholm Regional Parks

Diane Patricia Quadling

Thesis submitted in (partial) fulfillment
of Master of Applied Science

Auckland University of Technology
Auckland

August 2006

Table of Contents Page

List of Figures — Page

List of Tables — Page

Attestation of Authorship

I herby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previous published or written by another person nor material which to a substantial extent has been accepted for the qualification of any other degree or diploma of a university or other institution of higher learning, except where due acknowledgement is made in the acknowledgements.

……… ….. (Signed)

……………22 January 2007…….. (Date)

Acknowledgements

This thesis is dedicated to the memory of my mother Jan Quadling (1943-2004), who passed away before she could have a chance to correct my grammar and spelling mistakes. She will always be my best friend and I know that she is watching over me.

I would like to acknowledge the support of my supervisors Len Gillman, Tim Lovegrove and Graham Ussher throughout the duration of this thesis, and for their constructive criticism of the numerous drafts, but most of all for their contagious passion towards the environment.

For help with work in the field my heartfelt thanks go to the park volunteers and the Park Rangers at Shakespear, Wenderholm and Botanical Gardens for all of their help in setting up cages, cutting trees, identifying seedlings and for their continued never-ending commitment for the parks.

My personal thanks also goes to everyone who has read drafts, made comments and helped the stress levels, especially my family and friends, in particular; Darrell Quadling, Leigh Quadling-Miernik, Robert Miernik, Alan Quadling, Liz Klenner, Lynne and Richard Booth, Sarah Smellie and Jo Daly. Finally to my partner Simon Booth, words cannot express my thanks for the love and support you have given me through the years of study. Without it, this would not have been possible. Thank you.

Abstract

Exclosure plots were monitored to investigate the impact of browsing on seedling recruitment by *Trichosurus vulpecula, Oryctolagus cuniculus* and *Rattus rattus* on seedlings under *Leptospermum scoparium* and *Kunzea ericoides* plantings in two Auckland Regional Council Parks (ARC), Shakespear and Wenderholm. The number of woody seedlings that established over a 17-month period was recorded. Gaps within the same *Leptospermum scoparium* and *Kunzea ericoides* canopy were created to investigate the influence of the canopy on seedling recruitment. Soil samples were taken to investigate the existing seed bank beneath the same *Leptospermum scoparium* and *Kunzea ericoides* canopies.

At Wenderholm, net change in seedling density differed among treatments (P=0.014). Seedling density increased within the plots that excluded *Trichosurus vulpecula* and *Oryctolagus cuniculus* and within plots that additionally excluded *Rattus rattus*, but declined in the control plots. In contrast at Shakespear, although seedling density increased more within both the exclosure plots than in the control plots, this result was not statistically significant (P=0.728). At Wenderholm, the average seedling height increased within both types of exclosure plots, but declined in the control plots. However, these differences among treatments were not statistically significant (P=0.204). At Shakespear, seedlings increased in height within the *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus* exclosures and declined marginally in the other two treatments. Again, differences in height change among treatments were not statistically significant (P=0.202).

At both regional parks, the greatest cause of mortality within the exclosures excluding *Trichosurus vulpecula* and *Oryctolagus cuniculus* was desiccation. All of the mortalities within the *Trichosurus vulpecula, Oryctolagus cuniculus* and *Rattus rattus* exclosures was unidentifiable. However, within the control plots, at Wenderholm, the greatest identified cause of mortality was browsing and at Shakespear, the only cause of mortality within the control plots was browsing.

Seedbanks at Wenderholm and Shakespear under the *Leptospermum scoparium* and *Kunzea ericoides* plantings were dominated by forb species. A total of 1308 seedlings germinated from soil taken from Wenderholm, with exotic species making up 99.4% of germinations, with exotic species making up 97.9% of germinations. Similarly a total of 801 seedlings germinated from soil samples taken from Shakespear.

At Wenderholm, the number of native seedling germinations within the gaps created in the *Leptospermum scoparium* and *Kunzea ericoides* canopy, was more than twice the number that germinated under the closed canopy. However, this difference was marginally non-significant (*P*=0.065). At Shakespear, the number of native seedling germinations within gaps created in the *Leptospermum scoparium* and *Kunzea ericoides* canopy was similar to the number that germinated under the closed canopy (*P*=0.2603).

The results suggest that at Wenderholm, despite ongoing predator control, *Trichosurus vulpecula* and/or *Oryctolagus cuniculus* have had an adverse effect on the survival and growth of seedlings. The results also suggest that at Shakespear, *Rattus rattus* have had an adverse effect on the survival and growth of seedlings under the *Leptospermum scoparium* and *Kunzea ericoides* canopy. The distance from mature forest may also have had an impact on the dispersal of native seeds within the *Leptospermum scoparium* and *Kunzea ericoides* canopy. The implication of these results for the future management of restoration plantings in regional parks is discussed.

Thesis Layout

Chapter One: Investigates the definition of restoration, the scope for restoration, and the pre-requisites for success. It discusses the best practice for restoration, internationally and nationally. The Auckland Regional Council's Policy for the regional parks is assessed.

Chapter Two: Describes Shakespear and Wenderholm Regional Parks and outlines the methods used for: firstly, assessing the impact of browsing on seedling regeneration by *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus*; secondly, investigating the influence of *Leptospermum scoparium* and *Kunzea ericoides* canopy on seedling recruitment and; thirdly, investigating the seed bank beneath *Leptospermum scoparium* and *Kunzea ericoides* canopies.

Chapter Three: Describes the results found at Shakespear and Wenderholm regional parks. Seedling survival, effects of predation on seedling species, causes of mortality, germinations from seed banks and seedling recruitment into canopy gaps are discussed.

Chapter Four: Evaluates the findings of the results and looks at possible future revegetation and management strategies that could be implemented within the Auckland Regional Council parks.

Chapter Five: Summaries the major findings of this study and discusses the management implications for the Auckland Regional Council for the future of the vegetation programs on the regional parks.

Chapter One: Introduction

1.1. Overview

The common view of ecological restoration is to *"re-instate biotic communities to their original pre-human pristine state"* (Atkinson, 1990; Recher, 1993), or as close to this state as possible. Humans have had large detrimental impacts on ecosystems, through the introduction of plant and animal pests and habitat destruction (Atkinson, 2001; Norton & Miller, 2000; Taylor *et al.*, 1997; Towns & Ballantine, 1993). Conservation of New Zealand native biota is now becoming increasingly dependent on the retention and management of native vegetation (May, 1999; Saunders *et al.*, 1991) and the network of Auckland Regional Council parks help contribute to this. The Regional Council's objectives for restoration in regional parks include the restoration and enhancement of habitats and ecosystems with high ecological values and the protection of under-represented or threatened ecosystems (Auckland Regional Council, 2003a). Forest composition and diversity can both be strongly influenced by pest disturbance (Gillman, 2002). Consequently, understanding the impacts of animal pests, seedling survival and the persistence of weed seedbanks is crucial in developing long-term strategies for restoration in regional parks.

The impacts on the seeds, fruit and foliage of indigenous plants by possums (*Trichosurus vulpecula*) (Atkinson, 1992; Coleman *et al.*,1985; Nugent *et al*., 1997), rabbits (*Oryctolagus cuniculus*) (Gillman & Ogeden, 2003) and rats (*Rattus rattus*) (Atkinson, 2001; Campbell *et al*., 1984; Craig *et al*., 1984; Miller & Miller, 1995; Nugent *et al*., 2001; Wilson *et al*., 2003) have been extensively studied. However, despite the literature detailing the impacts of *Trichosurus vulpecula* on mature plants there is little information on the effect that *Trichosurus vulpecula*, *Oryctolagus cuniculus* and or *Rattus rattus* have on seedlings and hence forest regeneration (Campbell & Atkinson, 2002; Gillman, 2002 Gilman *et al.,* 2002; Nugent *et al.,* 2000; Nugent *et al*., 2001; Wilson *et al*., 2003). Forb and grass species are unlikely to contribute to the canopy, but may have a negative impact through competition with native seedlings. Rahman *et al*. (2001) found that seedbanks were persistent in the soil over long periods of time, and due to continual input of seed.

This study uses small exclosures to investigate the impact of browsing by *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus* on seedling recruitment under

Leptospermum scoparium and *Kunzea ericoides* plantings in two Auckland Regional Council (ARC) parks. This study also investigates the seed banks in the soil under the same *Leptospermum scoparium* and *Kunzea ericoides* canopy. Auckland Regional Council management objectives are reviewed to see if they are being met and alternative management options are discussed. Alternative management options include; the creation of light gaps in the *Leptospermum scoparium* and *Kunzea ericoides* canopy; under planting with mature phase species and; changing the mix of species initially planted.

1.2. New Zealand Flora/Fauna

New Zealand's biota has developed as the result of isolation and long periods of submergence. This has led to a limited number of major plant and animal groups in New Zealand, but it has also contributed to a high percentage of endemic species (Taylor *et al.*, 1997). Prior to the arrival of humans, 800-1,000 years ago, approximately 85% of New Zealand was covered in forest (Atkinson, 1994; Taylor *et al.*, 1997). Fragmentation of the once extensive tracts of native forest in New Zealand has occurred over the last 150 years (Davies-Colley *et al.*, 2000; Taylor *et al.*, 1997; Young & Mitchell, 1994). The exotic vegetation cover that has replaced those large areas of forest, tussock and wetland now extends to over 45% of the country. This is made up of 9.6 million hectares of exotic grasslands, 1.6 million hectares of exotic forests, and almost a million hectares of crops, horticultural land, suburban lawns, gardens and parks (Taylor *et al.*, 1997).

Fragmentation of natural landscapes is often detrimental to biodiversity and results in several changes to the ecosystem (Fahrig & Merriam, 1994; Marzluff & Ewing, 2001). Forest fragmentation can result in large areas being subject to "edge-effects" with an increase in the exposure to sunlight, wind and temperature fluctuations (Davies-Colley *et al.*, 2000; Murcia, 1995). As a result, the ecological conditions vary between the edge and the interior of the forest in terms of the vegetation structure, species richness and microclimate (Davies-Colley *et al.*, 2000; May, 1999; Murcia, 1995; Young & Mitchell, 1994). Remnants are also reduced in their ability to support the original biological diversity (May, 1999).

Since the arrival of humans, our ecosystems have been over-exploited, mammals have been introduced and habitats destroyed (Atkinson, 2001; Norton & Miller, 2000; Taylor *et al.*, 1997; Towns & Ballantine, 1993). Native animals have not developed behaviours or physical attributes to protect themselves and native plants have been unable to develop effective defences against introduced mammalian predators (Atkinson, 2001). Changes within ecosystems also have an impact on the interactions between species. As a result of these changes the natural selection process occurring within native species may be altered (Atkinson, 2001). However, it is difficult to understand or measure the total impact of these introductions as the interactions and flow on effects have not been studied.

Much of New Zealand's conservation effort relies on protected areas (Miller & Hobbs, 2002; Norton & Miller, 2000; Taylor *et al.*, 1997) such as National and Regional Parks. For example, the Auckland Regional Council has a network of 25 regional parks covering over 38,000 hectares of land within the Auckland Region. However, it is not enough to simply protect biodiversity through these areas alone. Isolated areas set aside may not be large enough to function independently of their surroundings (May, 1999; Miller & Hobbs, 2002). The realization is that while we need to retain individual remnants, we must also manage the landscape to ensure we develop optimal long-term outcomes (Bellingham *et al.*, 1999). The current thinking in New Zealand is that more ecologically coherent landscapes are required (e.g. connective corridors and mosaic management) with greater linkage between natural areas and restored areas (Department of Conservation 2000; Miller & Hobbs, 2002).

1.3. Seed Banks

Seed banks under temperate forests are often dominated by herbaceous species with little resemblance in species composition to the existing vegetation (Sem & Enright 1995; Sem & Enright 1996; Rahman *et al.*, 2001; Edwards & Crawley, 1999). However, few studies have been able to establish the importance of seed banks for forest succession, due to the difficulty of separating seedling input from the seed bank and seedling input from seed rain from the surrounding vegetation (Sem & Enright, 1995). Furthermore the high variability in seed density found in studies makes it difficult to interpret how seed banks contribute to secondary succession in forests (Sem & Enright 1995). Burrows (1995) found that New Zealand has a relatively low proportion of seeds with over-winter dormancy and relatively few species that form long-term (i.e. 1 year or longer) seed banks. Therefore, the importance of seed banks as a contributor to the early stages of native plant succession may be important. However, seed banks of weeds species, and their subsequent germination after disturbance, may inhibit native species regeneration.

At Huapai, Enright & Cameron (1988) found that adventive weedy species seeds were abundant in the forest soil seed bank and they suggested that distance from source, modes of dispersal, and the size of the forest patch were all important in determining seed bank composition. Enright & Cameron (1988) also suggested that the number of weedy species might decline as the distance from forest edge increases. Estimations of weed seed bank populations could help to predict future weed infestations. In addition, knowledge about the persistence of weed seed banks is important for developing long-term weed management strategies (Rahman *et al.*, 2001). A four year study in New Zealand found that the number of seeds in a seed bank (in the absence of seed input) remained abundant enough to indicate that natural depletion of the weed seeds would not occur for many years (Rahman *et al.*, 2001).

1.4. Forest Succession

It is widely believed that *Leptospermum scoparium* and *Kunzea ericoides* are early successional species that give way to late successional forest species. Esler & Astridge (1974) found that all *Leptospermum scoparium* and *Kunzea ericoides* communities in the Waitakere Ranges were transitional and some stands were giving way to dominance by *Agathis australis* and others to dominance by *Dacrydium cupressinum*. Esler & Astridge (1974) and Allen *et al*. (1992) have all found that *Kunzea ericoides* seedlings grow in open habitats and form a dense thicket that suppresses growth of other tree species, until there is a substantial reduction in *Kunzea ericoides* stem density after approximately 50 years. Allen *et al*. (1992) found that after 70 years only a scattering of podocarp seedlings had established in *Kunzea ericoides* stands and they were unable to determine whether or not *Kunzea ericoides* was being replaced by mature forest. Esler & Astridge (1974) suggest that after 100 years, *Kunzea ericoides* and *Leptospermum scoparium* will still be present in the Waitakere Ranges. Wilson (1994) also suggests that it may take 100 years for *Leptospermum scoparium* and *Kunzea ericoides* to give way to broad-leaved species.

Gorse (*Ulex europaeus*) has been considered as a possible nursery crop. If left undisturbed, gorse grows vigorously for a few years, growth then slows, and the canopy opens up. Shade tolerant plant species can regenerate through the aging gorse canopy and smother it as gorse requires full light to grow and is unable to regenerate under a canopy shade (Lee *et al*., 1986; Wilson H, 1994; Williams & Karl, 2002). Lee *et al*. (1986) estimated that a canopy of native vegetation and consequently the demise of the gorse could take 50-60 years. Gorse will regenerate in areas where there is no canopy cover (Lee *et al*., 1986). However, unless there is regeneration of shade tolerant plants within the gorse canopy then this method could have detrimental effects by allowing an invasive species to remain. A more desirable alternative nursery crop that could be considered is flax (*Phormium tenax*). Reay and Norton (1999b) found that *Phormium tenax* could provide a suitable site for the regeneration of woody plant species in grass pastures.

1.5. Impact of Animal Pests

Trichosurus vulpecula, Cervus elaphus and *Capra hircus* are considered to be the main mammalian pests in New Zealand (Nugent *et al*., 2001). *Oryctolagus cuniculus* were introduced in 1777, and are now widespread throughout New Zealand. They are most common where annual rainfall is less than 1,000mm, and where plants are grazed by other animals (Atkinson, 2001). Gillman and Ogden (2003) found that *Oryctolagus cuniculus* were responsible for most of the non-trophic damage at Huapai Scenic Reserve and that following *Oryctolagus cuniculus* control all non-trophic animal damage ceased.

Trichosurus vulpecula introduced in the mid-nineteenth century are common throughout most of New Zealand below 1200 metres a.s.l., and reach their highest densities in indigenous mixed hardwood forests (Coleman *et al*., 1985). However, their effect on native vegetation was not recognized until the 1920's (Atkinson, 2001). *Trichosurus vulpecula* eat the foliage from many woody species including *Weinmannia racemosa*, *Metrosideros umbellata*, *Melicytus ramiflorus* and *Pseudopanax* species (Coleman *et al*., 1985). Atkinson (1992) observed extensive possum browsing of *Metrosideros robusta*, *Belschmiedia tawa*, *Melicytus ramiflorus*, *Knightia excelsa* and *Coprosma areolata* seedlings and saplings on Kapiti Island. Nugent et al's (1997) study at Pureora Conservation Park found that woody species made up 80% of *Trichosurus vulpecula* annual diet. Continued browsing by *Trichosurus vulpecula* has resulted in an increase in the level of canopy dieback in New Zealand forest (Payton, 2000). Their generalist and opportunist feeding behaviour means that forest communities which are highly disposed to *Trichosurus vulpecula* damage, may change rapidly (Payton, 2000)

Kiore (*Rattus exulans*) are believed to have arrived with the first Polynesians that reached New Zealand. Norway Rats (*Rattus norvegicus*) arrived around the time of Captain Cook between 1769-1778. However, ship rats (*Rattus rattus*) are believed to have established in New Zealand in the middle of the nineteenth century (Atkinson, 2001). Rats (including kiore) have an impact on New Zealand's forests by eating the seeds, fruits and foliage of indigenous plants (Atkinson, 2001; Campbell *et al.*, 1984; Craig *et al.*, 1984; Miller & Miller, 1995; Nugent *et al.*, 2001; Wilson *et al.*, 2003). However, there is little information on the effect that *Trichosurus vulpecula* or rats have on seedlings and hence forest regeneration in New Zealand (Campbell & Atkinson, 2002; Gillman, 2002; Nugent *et al.*, 2001; Wilson *et al.*, 2003).

1.6. Single Species Management versus Ecosystems Management

Conservation management in New Zealand has previously been conservative and reactive, focusing on protecting species rather than ecosystems (Craig, 1990; May, 1999; Veitch & Bell, 1990; Walker, 1995). The aim of single species management is to protect a particular species to a point where it is self-sustainable within a given timeframe (Clout & Saunders, 1995). Whilst single species conservation can be considered successful, some people argue that the single species approach results in a lack of protection for other species and does not take into account long term ecological and evolutionary problems (Atkinson, 1994; May, 1999; Ogden, 1995).

The alternative to single species management is to take into account the whole ecosystem (May, 1999; Saunders *et al.*, 1991). Ecosystem management (which includes restoration management) is a concept that has been around for over 65 years but has received little support, until recently (Grumbine, 1994). Grumbine (1994) defined ecosystem management as management that "*integrates scientific knowledge of ecological relationships within a complex socio-political and values framework towards the general goal of protecting native ecosystems integrity over the long term goal*". As to which is more important, species management or ecosystem management? I believe that Atkinson (2001) summed up the debate aptly: "*There is no point arguing the merits of single-species and ecosystem approaches in conserving biodiversity: both are needed*".

1.7. Definition of Restoration

There are numerous definitions for restoration, ranging from the most common view of re-instating biotic communities to their original pre-human state (Atkinson 1990; Recher, 1993), to more complex definitions that encompass a self-sustaining environment (Ehrenfeld & Toth, 1997; Simberloff, 1990b), with the latter allowing for alternative land use and the involvement of people (Cairns, 1993; Hobbs, 1993).

There has been discussion over what point in time to define as the original pre-human pristine state. Recreating a specific ecological state of the past is often seen as an unobtainable goal due to climate change, irreversible changes (such as the loss of species through extinction, the introduction of animal and plant pests and pollutants) and because of the non-static nature of natural ecosystems (Atkinson, 1990; May, 1999; Simberloff, 1990b). Atkinson (1990) argues that unless one adopts a very loose definition of the pristine state, then restoring an environment to a pre-human state is an idealistic view, and that can seldom be seen as an achievable goal.

Cairns (1993) defines restoration as "*the creation of healthy self-regulating systems that allows for alternative land-uses and the involvement of people*" which is believed to be a more achievable goal (May, 1999). However, it is suggested here that this definition is too broad, as it could also cover exotic pasture. Cairns (1993) further suggests that if the goal of restoration is the renewal or rehabilitation of degraded landscapes, then the intent is to restore the functional and structural attributes of the degraded ecosystem. Simberloff (1990a) suggests that a restoration will be considered successful if it results in a system in which structure and function cannot be shown to be outside the bounds that are generated by the normal dynamic processes of the ecosystem. Karr (1990) suggests that if the goal of restoration ecology is to *"preserve and restore the original biota to degraded landscapes"* then not only species richness but species integrity of the landscape must be taken into account. Grumbine (1994) believes that definitions for ecosystem management give too much emphasis to the involvement or protection of native areas for humans, rather than the role humans have to play in protecting these areas. Ehrenfeld (2000) and Geist and Galatowitsch (1999) believe that restoration should be recognised for what it is, instead of believing that we are trying to replicate the original biodiversity.

Although there is no consensus on what the definition of restoration should be, these authors have some commonalities in their opinions. The authors appear to agree that restoration must include sustainability and functionality, but are unable to agree as to what level or point in time should be used. For the Auckland Regional Council, it is impossible to achieve restoration, which excludes people; therefore, perhaps the definition of restoration differs for different situations.

1.8. Expanding the Scope of Ecological Restoration

The growing awareness of ecological restoration is forcing consideration of what is required for good restoration. Consideration must be given not only to the flora and fauna but must include an expanded view that encompasses historical, social, cultural, political, aesthetic and moral aspects (Higgs, 1997). Daily (1993) believes that the main restraints on the potential success of restoration efforts are not scientific, but instead are social, political and economic. Bellingham (1990) argues that ecologically sound practices are imperative, but unless they are also economically and socially sound they will never be effectively implemented. Recher (1993) suggests that the need to conserve viable populations is based on ethical, economic and political considerations of the ecological and environmental consequences to the loss of populations and or species. Therefore, society, as well as individual land managers, will ultimately determine the need for restoration, and the extent of it. Higgs (1997) argues that good ecological restoration requires negotiating the best possible outcomes based on ecological knowledge and the interests of the stakeholders. Higgs (1997) also suggests that changes in our understanding of ecosystems has led to a better understanding of restoration as being something that is partially constructed by human values and attitudes. Therefore, changes in our focus to restoration should be to bring back into harmony the relationship between human practices and ecological functions.

Historically, early reserves in the United States, Canada, New Zealand and Australia were established on landscapes considered to have great aesthetic value and are therefore usually found at high elevations (Mendel & Kirkpatrick, 2002; Pressey, 1994). As a result of this the development of the reserve systems in Australia has been in an ad hoc and opportunistic fashion, resulting in large gaps in the protection of biodiversity (Mendel & Kirkpatrick, 2002).

Aesthetic considerations form a major part of people's decisions (Hartley, 1997; Higgs, 1997). When people make decisions about a resource use, these decisions can only be the result of how people value that resource. For example, it might be claimed that decisions made by regional or local government to preserve a particular area of land to reflect the 'intrinsic values' of the area, but in fact they only reflects the values placed upon that area by those particular people (Hartley, 1997). However, Higgs (1997) argues that aesthetic values are important, because they can enhance the public's acceptance of restoration projects.

The Auckland Regional Council has a requirement under Section 6 of the Resource Management Amendment Act (2005) to recognise the relationship of Maori and their culture and traditions with their ancestral lands, water, sites, waahi tapu and taonga. Higgs (1997) suggests that a need to achieve clarity on moral or cultural considerations will strengthen the ability for ecological restoration to generate healthy relationships between the people and the land.

Humans are dependant on the environment, but the failure of the Biosphere II experiment (Allen, 1996) showed very clearly that we do not have a comprehensive understanding of how the ecosystem works. Higgs (1997) and Miller & Hobbs (2002) argue that ultimately the success of biodiversity conservation depends on public support. Co-operation between Local and National agencies, researchers and the public is essential in protecting biodiversity (Craig, 1990; Grumbine, 1991; May, 1999) and ensuring that restoration projects have the best chance of success (Geist & Galatowitsch, 1999). However, the way in which people approach ecological issues can be biased based on their present and past environment (cultural, social and physical) (Craig, 1990).

There are strong pressures from individuals and non-government organizations for governments to preserve ecosystems, which are perceived by the public as having exceptional biological diversity (Recher, 1993). People value nature as well as the ability to use the resource. For the Auckland Regional Council the Regional Community objective in the Parks Management Plan (2003a) is *"to ensure that parks reflect the needs and values of the community"*. To achieve this goal, different parks within the Auckland Region provide for different recreational and environmental experiences, while at the same time supporting the objectives and polices of the Management Plan, including restoration (Auckland Regional Council, 2003a).

The most common obstacle to restoration is obtaining sufficient funds (Holl & Howarth, 2000). The critical issue is that policy makers must ensure that resource allocation decisions take relevant 'environmental' costs and benefits into account (Hartley, 1997; Hughey *et al.*, 2003). Recher (1993) suggests that society as well as managers will determine the need for, and the extent of restoration. To influence these decisions it will be necessary to justify why restoration, conservation and management of biological diversity is necessary and important. For this to be effective, priorities need to be developed that are both ecologically relevant as well as politically and economically realistic (Recher, 1993).

Restoration projects are not just about planting trees, they require involvement with the local community, through consultation with the public and by enlisting volunteers to help plant trees. However, all restoration projects are limited by a lack of funds. These are all things that the Auckland Regional Council has to consider when looking at restoration projects in regional parks.

1.9. Successful Restoration – Best Practice Principles

International and national consensus for ecological restoration is that consideration must be given not only to the flora and fauna, but that it must include the relationship between ecological and cultural restoration, aesthetics as well as community involvement (Higgs, 1997). Best practices internationally and in New Zealand are examined and the strategic documents that are produced are discussed in this section.

Local and Government organisations, both internationally and nationally are obliged, by legislation, to have strategies that identify the aims and outcomes they are trying to achieve (Llewellyn & Tappin, 2003). The ability to clarify the objectives for protected area management is important, as is articulating preferred outcomes against which different management options can be judged (Sutton, 2004). For parks and reserves, public interest in the way that local and central Government agencies perform their duel roles of conservation and provision for recreational opportunities means that management planning becomes a more 'social' function (Sutton, 2004).

1.9.1. International Best Practice Principles

There is a range of different models for environmental management and they will be examined more closely by comparing models from the United States of America (USA) and the United Kingdom (UK). Llewellyn & Tappin (2003) found that although parks in the United States and United Kingdom have strategic documents, in the past they were not referred to on a regular basis, if at all. However, strategies were re-worked in a bid to help attract external funding (Llewellyn & Tappin 2003). Langley (1991) argues that formal strategies help to facilitate the processes of social interaction. They bring parks and potential funding agencies closer by providing a template for the alignment of their interests. They also help to close the gap between what the public sector providers say they do and what they actually do (Llewellyn & Tappin 2003).

National Parks in the United States are owned by the Federal Government (i.e. American people) and management is centralized in Washington DC (Llewellyn & Tappin, 2003). Managers of United States' parks are required to produce four documents: a management plan, a strategic plan, a performance plan, and a performance report (Llewellyn & Tappin, 2003). In the United Kingdom, the governance system of the parks is more complex. The parks are populated by people and therefore are mainly owned by private citizens and run by local boards or committees (Llewellyn & Tappin, 2003). Legislation requires that managers of United Kingdom parks produce three strategic documents: a management plan, a corporate plan and a performance plan (Llewellyn & Tappin, 2003). Keiter (1998) believes that translating these documents into a workable ecosystem management policy is not easy, as managers have to interpret the goals and objectives of the plans. This is similar to what happens with strategic documents in New Zealand.

1.9.2. New Zealand Best Practice Principles

Legislation such as the Reserves Act (1977), Fisheries Act (1983) Conservation Act (1987), Resource Management Act (1991), Biosecurity Act (1993), Forest Amendment Act (1993), and Historic Places Act (1993) provide the legal context for looking after the environment in New Zealand (Norton & Miller, 2000). The main environmental planning legislation in New Zealand is the Resource Management Amendment Act (RMAA 2005). The purpose of the Act is to *"promote sustainable management by allowing for the economic and cultural wellbeing of local communities while providing for the protection of natural resources including native biodiversity"* (RMAA 2005).

The Resource Management Act also sets out the environmental management responsibilities of local authorities (RMAA 2005).

The Department of Conservation (DoC), established in 1987 under the Conservation Act, manages the Crown's conservation estate and it is the main Government organisation responsible for protecting and sustaining biodiversity. The Ministry for the Environment (MfE), established under the Environmental Act (1986), has responsibility in the planning and policy making process and it also provides advice to the central and local Government on environmental issues. Administering bodies, such as the Regional and District Councils, promulgate and enforce resource management polices and plans (Taylor *et al.*, 1997). National parks in New Zealand are owned and managed by the Department of Conservation on behalf of the people of New Zealand while the regional parks are owned and managed by Regional Councils on behalf of the people of those regions. The Department of Conservation (DoC) and Territorial Local Authorities manage a number of parks and reserves. The Department of Conservation manages parks through the Conservation Act (1987) and it focuses on conserving indigenous plants, animals and habitats. The Territorial Authorities generally manage parks through the Reserves Act (1977) and the Local Government Act (2002), and they largely focus on recreational use. The Auckland Regional Council (ARC) provides a mixture of conservation and recreational activities within regional parks (Auckland Regional Council, 2003a).

1.9.3. The Auckland Regional Council

The Auckland Regional Council (ARC) is responsible for managing the regional parks within the Auckland region. The role of the Council, in managing the regional parks network, is set down by two main Acts: the Local Government Act (2002) and the Reserves Act (1977) (Auckland Regional Council, 2003a). The Local Government Act (2002) enables the ARC to acquire and manage regional parks in order to protect special natural and cultural features and to provide for the recreational needs of the people in the Auckland region (Auckland Regional Council, 2003b). The Reserves Act (1977) is designed to protect public land, protect natural and cultural values and ensure the preservation of access for the public (Auckland Regional Council, 2003b). It also allows for classification of different types of reserves and contains provisions for their acquisition, control, management, maintenance, preservation, development and use

(Taylor *et al.*, 1997). Under the Reserves Act (1977) the ARC is required to prepare management plans for its regional parks (Auckland Regional Council, 2003a).

The Council's objectives in the Regional Parks Management Plan incorporate the protection of habitats and ecosystems whilst enabling current and future generations to use and enjoy the parks. It also takes into account the needs of Tangata Whenua, visitor services, education and farming, as well as research and monitoring (Auckland Regional Council, 2003a). The Council's objectives for restoration and enhancement are:

- The restoration and enhancement of habitats and ecosystems with high ecological values;
- The conservation of regionally under-represented or threatened ecosystems;
- The re-introduction of indigenous flora and fauna;
- The provision of ecological corridors for wildlife (Auckland Regional Council, 2003a).

The level of restoration varies between regional parks due to limited resources (Auckland Regional Council, 2003a). Where practical, a minimum impact approach is used by removing livestock and fencing the area to allow it to regenerate naturally without any assistance (Auckland Regional Council, 2003a). The minimum impact approach ranges from doing nothing (natural regeneration) to fencing, weed control and/or scattering seed (assisted regeneration) to planting early pioneer species, propagated from nearby seed source. However, in some situations it is considered necessary to intervene to ensure that ecosystems become self-sustaining. For example, by providing habitat or food for birds, extending forested habitats or linking isolated forested areas (Auckland Regional Council, 1996).

1.10. Pre-requisites for Successful Restoration

The conservation of biological diversity has become one of the important goals of managing forests and therefore it is essential to have measures, which determine the success or failure of a project (Lindenmayer *et al.*, 2000). The goals for restoration can vary from project to project. For example, they may include: weed suppression, establishing canopy cover, increasing native species, or reducing animal and plant pests. Atkinson (2001) argues that although conserving biological diversity is a goal of restoration, a project may have goals that extend across several dimensions: from conserving the genetic variability of populations to safeguarding the successional linkages between communities within the landscape.

Grumbine's (1994) review of papers on ecosystem management found that most authors agreed that setting clear goals is crucial to the success of ecosystem management. He also stated that the overall goal of maintaining ecosystem integrity should include five more specific goals:

1. *Maintain viable populations of all native species in situ.*
2. *Represent, within protected areas, all native ecosystem types across their natural range of variation.*
3. *Maintain evolutionary and ecological processes (i.e. disturbance regimes, hydrological processes, providing connecting habitats etc.).*
4. *Manage over periods of time long enough to maintain the evolutionary potential of species and ecosystems.*
5. *Accommodate human use and occupancy within these constraints.*

The first four goals relate to reducing the loss of biodiversity, the fifth goal recognises that people have a vital (if problematic) role to play (Grumbine, 1994). Cairns (2000) suggested that major ecological restoration will not take place unless society approves the goals and objectives of the restoration.

1.10.1 Goals for Projects

The pre-requisite for success, in restoration, is determined by the ability to set clear and achievable goals (Allen *et al.*, 1997; Atkinson, 2001; Cairns, 1993, 2000; Craig, 1990; Ehrenfeld, 2000; Hobbs & Harris, 2001). However, Atkinson (1990) and Simberloff (1990a) found that restoration goals are usually ambiguous, making it difficult to prove if the restoration project was a success or not.

Ehrenfeld (2000) suggests that the requirement of goals for restoration projects is the most important element, because it sets expectations, drives the detailed plans for actions and determines the amount of monitoring required. However, it must also be recognised that there is no one paradigm or context for setting restoration goals and that goals need to be as specific as possible but developed appropriately for each restoration project (Ehrenfeld, 2000; Pastorok *et al.*, 1997). Hobbs & Harris (2001) also argue that having clear and achievable goals is essential. However, they state that ecosystems are not static and therefore the goals should focus on the desired characteristics for the future, rather than what they were in the past.

Higgs (1997) suggests that restoration goals should focus on 'ecological fidelity' which is comprised of: structural/compositional replication, functional success and durability. However, Hobbs and Harris (2001) suggest that these are general terms, which are difficult to develop into effective goals. Miller and Hobbs (2002) suggest that, to be effective, conservation planning must be based on information derived from well-designed studies which take into account all land uses. Pfadenhauer (2001) suggests that the goals of restoration ecology can generally be described in terms of increased biodiversity, enhanced water retention capacity and avoidance of soil erosion. Rapport *et al.* (1998) suggest that system vigour, organisation and resilience are properties which can be assessed and hence could be used to develop goals for restoration. Recher (1993) argues that a primary goal for restoration must be sustainability and that after the initial input of human resources, the restored landscape must be self-sustaining, in the sense that ecological processes and functions continue without human intervention.

Atkinson (1990) suggests that goals for conservation, specifically ecological restoration have generally focused on returning a site to some historical condition and to restore the ecosystem to a state which is representative of what might have occurred prior to human disturbance. This, according to Atkinson, is unrealistic. Ehrenfeld (2000) suggests that we must recognise that goals must be flexible and that it might be more prudent to develop guidelines for defining a set of conditions under which different kinds of goals are appropriate. Atkinson (2001) points out that all goals rest on people's judgments, and whatever decisions we make are based on how society views this in comparison with other priorities.

The consensus nationally and internationally, for ecological restoration, is that there is a need to set clear concise goals and that consideration must be given not only to the biological aspects but that historical, cultural, aesthetic, social, moral, economics and political aspects must be taken into account (Daily, 1993; Geist & Galatowitsch, 1999; Higgs, 1997; Recher, 1993). Although goals can vary from project to project, all goals must be clearly defined, measurable and monitored on a regular basis to determine if the project is a success. The common goal of restoration is to reproduce the biotic communities to a pre-determined historic or indigenous ecosystem. This can be considered an unrealistic goal and it may not always be practical in a regional park situation, where there is a need to balance the protection of the flora and fauna as well as ensuring open spaces are available for the needs and wants of the public.

1.10.2. Definition and Limitations of Goals

The ideal goal is to restore an ecosystem to its original condition (Cairns, 1999) and often the extent to which this is achieved is the measurement of success. Restoration projects require planning and monitoring to ensure that they are successful. Planning starts with defining the problem, a clear statement of objectives and an understanding of any uncertainties (Pastorok *et al.*, 1997). Defining goals is considered to be the most important step in the planning process, as it ensures that there is a direction for the project to go (Pastorok *et al.*, 1997). It is also important to develop effective and easy to measure success criteria, which relate back to the specific restoration goals (Hobbs & Harris, 2001). These criteria should also be continued throughout the planning, design, implementation and monitoring phases of the project (Pastorok *et al.*, 1997). Pastorok *et al.* (1997) believe that the success of the planning process relies on identifying key ecological process within the ecosystem and understanding those processes in relation to the objectives of the project.

Cortner *et al.* (1994) and Moote *et al.* (1994) found that most observers agree on the basic concept of the five principles governing ecosystem management as follows. First, that ecosystem management goals must be socially defined through a shared vision process that incorporates ecological, economic, and social considerations (Cortner *et al.*, 1994; Moote *et al.*, 1994). Second, that ecosystem management is based upon integrated and comprehensive scientific information that addresses multiple rather than single resources (Grumbine, 1994; Moote *et al.*, 1994). Third, ecosystem management should seek to maintain and restore biodiversity and sustainable ecosystems (Christensen *et al.*, 1994; Grumbine, 1994). Fourth, ecosystem management involves management at large spatial and temporal scales to accommodate the non-static nature of the environment (Cortner *et al.*, 1994). Fifth, ecosystem management requires an adaptive management approach, which must include establishing goals and objectives, which are monitored, re-evaluated and adjusted when required (Cortner *et al.*, 1994; Moote *et al.*, 1994).

Pfadenhauer (2001) believes that in practice a discrepancy exists between the high ideals of restoration goals and reality. Because restoration involves scientific and social interests, limiting factors can include conflict between different restoration goals (Geist & Galatowitsch, 1999; Pfadenhauer, 2001) and the unpredictability of restoration goals due to long-term effects, (Pfadenhauer, 2001). Failure is often the result of underlying human obstacles such as cost constraints, limitations in land allocation and insufficient time and labour (Geist & Galatowitsch, 1999). Hughey *et al.* (2003) suggest that instead of asking what are the priority areas or species that require conservation management and how effective are different management techniques, we should be asking *"Where should scarce resources be invested in conservation management? And which investments in conservation management have been the most successful?"*

1.10.3. Determining Success

Hobbs & Harris (2001) believe that if the goals of a restoration project are composition, structure and function, then the measures to quantify success are unclear. Lindenmayer *et al.* (2000) believe that measures of biodiversity conservation are weak, as they tend to focus on taxon-based indicator species instead of landscape diversity, and they suggest that the relationships between species and ecosystem processes must be researched more. Atkinson (2001) suggests there is a need to understand successional processes to ensure success. Monitoring is also essential to ensure that we can detect mistakes and/or opportunities to improve processes (Atkinson, 2001). Meurk & Swaffield (2000) also believe that the integrity of reporting is essential and must encompass the whole of the ecosystem including the processes involved. Grumbine (1994) suggests that success over the short term means "*making significant, measurable progress towards maintaining viable populations, representing ecosystem types"* but that the long term success is more difficult to recognise.

All measures of success have to be linked back to clearly defined goals. Hobbs and Harris (2001) point out that if the restoration goal is to "*re-establish a diverse vegetation cover resembling that which was present before disturbance'* it is not possible to measure its success. If the alternative goal for example is, to '*re-establish vegetation with 20 trees per hectare, comprising local provenance native species which attain a height of at least 2m within 5 years, and an understorey of native shrubs, forbs and herbs achieving a site diversity of 25+/- species"* it is possible to measure the actual performance and therefore the success of the restoration project. However, it could be argued that these figures are arbitrary and therefore meaningless so that a better definition would be a closed canopy of x number of mature-phase canopy species (pers. comms. Dr Len Gillman, Senior Lecturer, Auckland University of Technology, 2006).

Whilst there is a need to have clearly defined and measurable goals, regular monitoring must also occur to determine if the project has been a success. At present, within the Auckland Regional Council's restoration projects monitoring includes regular checks to ensure that serious environmental weeds are detected, and follow up plantings to fill major gaps resulting from any losses in the initial plantings (pers. comms. Tim Lovegrove, Natural Heritage Scientist, Auckland Regional Council, 2006). However, there is a need to have more detailed information at the start and throughout the project to determine if it is a success. A suggestion is that at the time of planting each block is labeled, and the number of plants of each species planted recorded to enable survival rate to be assessed. Approximate heights and spacing should also be recorded to allow growth rates to be measured at a future date.

1.11. Conflicts between Conservation and Recreation

Protected areas are vital for ensuring the protection of indigenous biodiversity. However, there are inherent conflicts with managing National and Regional parks in balancing the protection of flora and fauna and ensuring open spaces are available for the public to enjoy (Llewellyn & Tappin, 2003; Sutton, 2004). Too much public access threatens conservation, while too little diminishes the public's enjoyment and hence support (Llewellyn & Tappin, 2003). Surveys conducted by the Auckland Regional Council have found that the public wants a diversity of different settings e.g. panoramic views, tracks through vegetation and open spaces. Their key experience-driver is where the park is located and what it has to offer, or enables them to do, including scenic/aesthetic qualities, the environment and ecology is secondary (pers. comms. Neil Olsen, Senior Recreational Advisor, Auckland Regional Council, 2006).

Studies in Australia have shown that tourism can directly or indirectly threaten plant taxa for example, the introduction of weeds, trampling, pathogens, clearing or collecting and that these impacts often occur in conservation areas (Kelly *et al.*, 2003). One way to minimise the impact of tourism is to concentrate particular uses as much as possible. For example, picnic areas and camping areas can be placed in one area of the park, thereby minimising the use of undisturbed areas. The dilemma for parks' management lies in finding ways of bringing in external funding that is consistent with their overall vision whilst also having some flexibility to shift strategic direction, if necessary (Llewellyn & Tappin, 2003). Therefore, a well-planned communication strategy will create a sympathetic and more environmentally aware public, which helps to meet the management-related objectives (Hockings & Carter, 1998).

1.12. Assessment of Auckland Regional Council Management Policy

The role of the Auckland Regional Council in the management of the regional parks network is to meet the conservation and recreational needs of the regional community (Auckland Regional Council, 2003a). The Auckland Regional Council therefore has to have a model in which the recreational needs of people form an important component of the regional parks. This contrasts with the Department of Conservation model, where conservation (e.g. nature and scientific reserves) has the highest priority and the needs of the people are secondary.

The Regional Parks Management Plan outlines the overview and strategic direction for the management of the regional parks network (Auckland Regional Council, 2003a). Within this Plan there are general objectives which include: habitats and ecosystems, indigenous flora and fauna, quality and diversity of the landscape etc. The second half of the Plan (Volume 1) outlines how the general management approach will be applied to each individual park and identifies the focus for park management over a five-year period (Auckland Regional Council, 2003a).

For Shakespear Regional Park, ecological enhancement management actions include: "*Extending valley plantings to provide linkages with existing forests, the eradication or intensive control of all significant introduced plant and animal pests, the protection and enhancement of existing forest remnants, the reintroduction of flora and fauna formerly present and the active participation of the public*". For Wenderholm Regional Park ecological enhancement management actions include: *"Maintaining an integrated pest control programme, protect existing coastal forest ecosystems, and the reintroduction of locally extinct bird species, and other missing flora and fauna as appropriate"* (Auckland Regional Council, 2003a).

However, whilst there is a Management Plan that outlines objectives and goals for the future, which have been developed through a lengthy consultative process with the regional community, these objectives are interpreted by Council officers who then implement the Plan. Therefore, unless strategic documents have clearly defined actions that state how the goals are to be achieved it is very easy for individuals to have different interpretations of how to achieve those goals. Due to the lack of detailed ecological policies, individual's opinions have been able to hold sway. For example, ten years ago, common opinion was to use pioneer species such as *Leptospermum scoparium* and *Kunzea ericoides* and allow nature to do the rest. The Botanical Society with the Auckland Regional Council developed a planting programme which changed the species planted from mixed species to *Leptospermum scoparium* and *Kunzea ericoides* to create a starting point for natural succession.

1.13. Thesis Objectives

The objectives of this study are: 1. to investigate the impact of browsing on seedling regeneration by *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus*; 2. to determine the seedling mortality rate under *Leptospermum scoparium* and *Kunzea ericoides* plantings on two Auckland Regional Council (ARC) Parks; 3. to investigate the influence of *Leptospermum scoparium* and *Kunzea ericoides* canopy on seedling recruitment; 4. to investigate the seed bank beneath *Leptospermum scoparium* and *Kunzea ericoides* canopies; 5. to investigate whether or not the current Auckland Regional Council's policy on revegetation is achieving the desired management outcomes. Three field procedures were undertaken to meet these objectives:

1. seedling recruitment within herbivore mesh cage exclosure plots was measured over a seventeen month period and compared to seedling recruitment in control plots.
2. the seed bank under *Leptospermum scoparium* and *Kunzea ericoides* plantings was investigated by taking soil samples in January and providing suitable conditions for germination for these samples over an 18-month period.
3. the recruitment of seedlings was measured in experimental light gaps created by cutting the pre-existing *Leptospermum scoparium* and *Kunzea ericoides* and comparing this recruitment to that under a closed *Leptospermum scoparium* and *Kunzea ericoides* canopy.

Alternative management options are discussed such as creating light gaps in the *Leptospermum scoparium* and *Kunzea ericoides* canopy, under planting with mature phase species or changing the mix of species initially planted.

Chapter Two: Methods

2.1 Study Sites

Two locations were selected for this study, Shakespear Regional Park and Wenderholm Regional Park.

2.2. Shakespear Regional Park

Shakespear Regional Park forms part of the Auckland Regional Council's network of parks. The park is 376 hectares and is situated at the northern end of Whangaparaoa Peninsula, Ministry of Defence land adjoins the North boundary, to the West is residential land and to the South and East is coastline. The site for this study was located at the northeastern end of the park (NZMG 2674600E 6508400N) at an altitude of 60 metres a.s.l. within two planted blocks (0.760 hectares and 1 hectare respectively) comprising of *Leptospermum scoparium* and *Kunzea ericoides* planted between 1992 and 1994 (Figure 1).

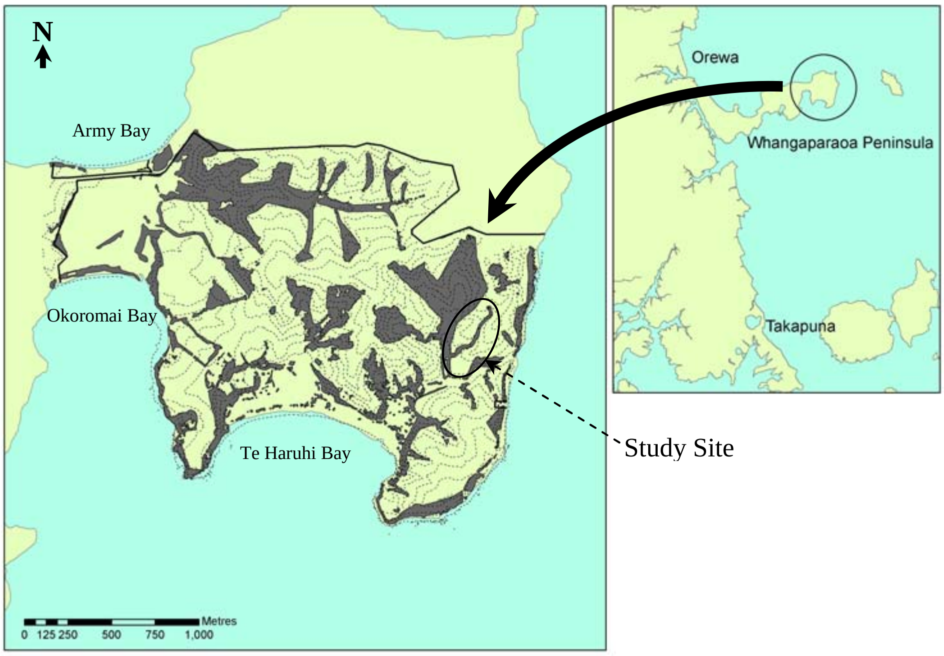


Figure 1. Shakespear Regional Park showing the areas of forest fragmentation, contours and study site (contour interval 50m)

2.2.1. Status of Shakespear Regional Park

Shakespear Regional Park has a Class III status under the Auckland Regional Council Regional Parks Management Plan (2003a). The management objectives of this class are: sustaining social, interactive and informal recreation, with an emphasis on providing recreation opportunities with protection of significant natural and cultural resources (Auckland Regional Council, 2003a). Over the next five years the management actions for ecological enhancement at Shakespear will focus on:

- *"Developing the park as an open sanctuary with initial focus being given to enhancing and restoring the existing habitats to complement the ecological restoration already undertaken on nearby Tiritiri Matangi Island. Once habitat conditions are suitable, the feasibility of reintroducing birds will be investigated with the Department of Conservation.*
- *Enhancing native plant and animal communities through:*
    - *Extending valley plantings to provide linkages with existing forests and planting areas*
    - *The eradication or intensive control of all significant introduced plant and animal pests*
    - *The protection and enhancement of existing forest remnants*
    - *The reintroduction of flora and fauna formerly present, but now absent, including a range of locally extinct bird species; and*
    - *The active participation of the public (Auckland Regional Council, 2003a)"*

2.2.2. Geology of Shakespear Regional Park

Shakespear Regional Park is comprised of two major rock types. The hilly areas are formed of Waitemata sandstones, whereas the flats behind Okoromai and Shakespear Bays are made up of unconsolidated Holocene alluvial and beach deposits. Waitemata Group rocks consist of layered yellow sandstone (20cm-2m thick) and grey coloured mudstone (5-20cm thick). Thick beds of dark coloured volcanic grit (Parnell Grit) were inter-dispersed between the sandstone/mudstone layers (Auckland Regional Council, 2003b). The general soil pattern consists of a coastal strip of weakly to moderately leached yellow-brown earth (Atuani, Puhoi and Warkworth soils) with a central core of strongly leached and podzolised soils (Waikere and Hukerenui). Weakly leached yellow-brown sands (Whananaki soils) have developed on the dunes. Behind the dunes sand-peat complexes are formed. Grey soils developed in the valleys, and Otao soil, derived from volcanic ash, is found behind Pink Beach (Orbell, 1968). The rugged cliffs that surround much of the park have been carved during the last 6,500 years when the sea levels rose to their present level (Auckland Regional Council, 2003b).

2.2.3. Flora and Fauna at Shakespear Regional Park

During the 1860's most of Shakespear Regional Park was covered in *Leptospermum scoparium* and fern and coastal forest was believed to have covered the central area of the park. The large amount of early successional shrubland currently present is believed to have developed following fires started for cultivation purposes by Maori (Auckland Regional Council, 2003b).

Today Shakespear Regional Park is predominantly grazed pasture with *Leptospermum scoparium* and patches of broadleaf forest in the valleys comprising of: *Dacrycarpus dacrydiodes*, *Dysoxylum spectabile*, *Sophora tetraptera*, *Vitex lucens*, *Beilschmiedia taraire*, *Melicytus ramiflorus* and *Myrsine australis*, with an understorey of *Coprosma* species, *Geniostoma rupestre* and *Myrsine australis*. At the eastern tip the dominant trees are: *Vitex lucens*, *Beilschmiedia taraire*, *Corynocarpus laevigatus*, *Dysoxylum spectabile, Sophora tetraptera*, *Metrosideros excelsa*, with scattered *Knightia excelsa* (Auckland Regional Council, 2003b).

Around the cliff edges of the park, mature *Metrosideros excelsa* are dominant. A salt-marsh is located behind Okoromai Bay, and another wetland exists behind the eastern end of Te Haruhi Bay. The Te Haruhi Bay wetland is a Site of Special Wildlife Interest (SSWI) of moderate significance (Auckland Regional Council, 2003b). The adjacent Ministry of Defence land is covered mainly with regenerating native scrub, with an area of remnant forest near the park's northern boundary on the eastern coastal perimeter (Auckland Regional Council, 2003b).

Weed species such as Japanese honeysuckle (*Lonicera japonica*) and privet (*Ligustrum ovalifolium*), are abundant around the YMCA Homestead and the area of bush that covers the western headland around Te Haruhi Bay (Auckland Regional Council, 2003b). Other weed species around the park include: pampas grass (*Cortaderia selloana*), inkweed (*Phytolacca octandra*), Scotch thistle (*Cirsium vulgare*), smilax (*Asparagus asparagoides*) and climbing asparagus (*Asparagus scandens*) (May, 1999).

The diversity of avifauna at Shakespear Regional Park is considered to be low. Tui (*Prosthemadera novaeseelandiae*) and Fantail (*Rhipidura fuliginosa*) are most common (May, 1999). Thirteen native land bird species have been recorded at Shakespear

Regional Park, including spotless crake (*Porzana tenebrosa*) and kaka (*Nestor meridionalis septentrionalis*) which visit from Little Barrier Island and/or Leigh (Auckland Regional Council, 2003b). Kereru (*Hemiphaga novaeseelandiae*), Kaka (*Nestor meridionalis septentrionalis*) and Bellbird (*Anthornis melanura*) are also known to visit when certain plants are in flower (May, 1999). Several species of birds are known to migrate between Tiritiri Matangi and Shakespear Regional Park including: Harrier (*Circus approximans*), Kereru (*Hemiphaga novaeseelandiae*), Bellbird (*Anthornis melanura*) and Tui (*Prosthemadera novaeseelandiae*) (Auckland Regional Council, 2003b). Animal pests at Shakespear include: *Trichosurus vulpecula*, *Rattus norvegicus*, *Rattus rattus*, *Felis catus* and *Mustela* species (Auckland Regional Council, 2003b). All are considered to be a threat to the fauna and/or flora.

2.2.4. Cultural Heritage of Shakespear Regional Park

An archaeological survey in 1976 recorded twenty four archaeological sites, the majority of which were middens. Five pa sites are located on the Peninsula; on the hills to the East and West of Te Haruhi Bay; on the Defence land on the eastern hill of Army Bay; at the northern tip of the Peninsula; and at the head of Waterfall Gully (Auckland Regional Council, 2003b). Human settlement on Whangaparaoa Peninsula extends back almost one thousand years. Pre-European inhabitants were the Ngati Kahu, a hapu of predominantly Te Kawerau decent, who had settlements which were mainly located between Army Bay and Te Haruhi Bay (Auckland Regional Council, 2003b). European history goes back to the 1800's. In 1883, Robert Shakespear began a long association of farming in the area with cattle and sheep. In 1967, land was purchased by the then Auckland Regional Authority, with more land being added to the park in the following year to give its current area of 376.29 ha (Auckland Regional Council, 2003b).

2.2.5. Visitor Numbers at Shakespear Regional Park

Shakespear Regional Park attracts between 500,000-1,000,000 visitors per year (Auckland Regional Council, 2003a). A study conducted in 1987/1988 found that visitor usage of the park is localised to Okoromai Bay, Te Haruhi Bay and Army Bay.

2.3. Wenderholm Regional Park

Wenderholm Regional Park (134 hectares) is situated on the steep ridge of land separating the Waiwera and Puhoi valley catchments and ends in a large coastal headland. Purchased in 1965, it is Auckland's first Regional Park (Auckland Regional Council, 2003b). The site for this study was located near the entrance to the park (NZGM 2663200E 6516900N) at an altitude of 40 metres a.s.l. within a 1.935 hectare area of *Leptospermum scoparium* and *Kunzea ericoides* planted between 1995 and 1997 (Figure 2).

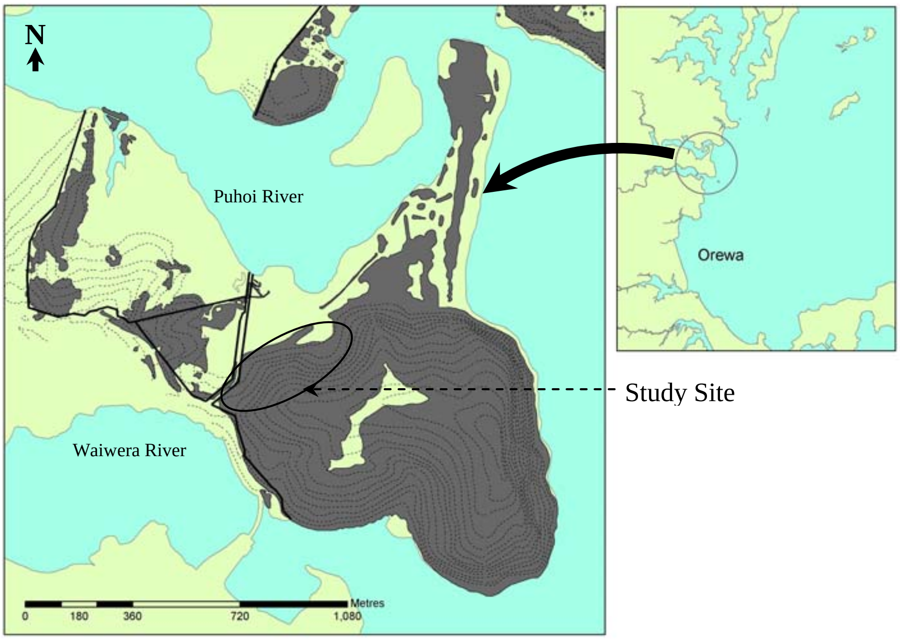


Figure 2. Wenderholm Regional Park showing the areas of forest fragmentation, contours and study site (contour interval 50m)

2.3.1. Status of Wenderholm Regional Park

Wenderholm Regional Park has the same Class III status under the Auckland Regional Council Regional Parks Management Plan (2003a) as Shakespear Regional Park. Over the next five years the management actions for ecological enhancement at Wenderholm will focus on:

"Managing the headland (Maungatauhoro) and salt-marshes as ecological restoration and enhancement areas in accordance with the following actions:

- *Maintaining and integrated pest control programme*
- *Protecting existing coastal forest ecosystems; and*
- *Reintroduction of locally extinct bird species, and other missing flora and fauna as appropriate*

- *Revegetation will focus on coastal plantings, enhancement and wetland restoration plantings (Auckland Regional Council, 2003a)”*

2.3.2. Geology of Wenderholm Regional Park

Wenderholm Regional Park consists of Waitemata Group rocks in the hilly areas; whereas the spit and areas of flat land around the estuaries are made of unconsolidated Holocene sediments (Auckland Regional Council, 2003b). The alternating sandstones and mudstone rocks which formed 20-25 million years ago during the Lower Miocene period are seen in the cliffs around the headland. The beds have an overall low north-to-north-west dip and small faults and several folds visible in the cliff section and shore platform to the south as the result of earth movements 3-20 million years ago (Auckland Regional Council, 2003b). Sand carried northward by long shore drift has been deposited at Wenderholm forming the spit and dunes across the Puhoi River mouth. Subsequently, sediment has settled to form the flat areas behind the spit (Auckland Regional Council, 2003b). The headland soils are well-drained Puhoi light brown clay loom, typical of hilly land in the area. The flats north of the headland and the spit comprise excessively drained Whananaki sand typical of dune complexes. The spit is subject to erosion, particularly near the mouth of the Puhoi River (Auckland Regional Council, 2003b).

2.3.3. Flora and Fauna at Wenderholm Regional Park

The vegetation at Wenderholm Regional Park varies with predominately indigenous coastal forest on the headland extending to the coast and introduced plantings leading to, and surrounding, the colonial homestead. The sandspit is covered in large mature *Metrosideros excelsa*, planted last century. An estuary containing mangroves is located on the northern boundary by the Puhoi River (Auckland Regional Council, 2003b). The coastal *Beilschmiedia taraire* forest on the headland is considered to be one of the best areas of this type in the Rodney Ecological District (Auckland Regional Council, 2003b). The forest has a canopy of *Beilschmiedia tarairi*, *Beilschmiedia tawa*, *Vitex lucens* and *Corynocarpus laevigatus*, with occasional specimens of *Planchonella costata*, *Laurelia novae-zelandiae*, *Nestegis lanceolata* and *Beilschmiedia tawaroa*. *Myrsine australis* and *Leptospermum scoparium* are common on ridges and *Sophora tetraptera* stands are present on the north-facing slopes. *Dysoxylum spectabile*, *Hoheria populnea* and *Rhopalostylis sapida* are also common (Auckland Regional Council, 2003b).

The park also contains older, exotic trees which were planted at various times from as early as the 1860's, these include: Caucasian fir (*Abies nordmanniana*), Bunya bunya pine (*Araucaria bidwillii*), Moreton Bay fig (*Ficus macrophylla*), *Quercus ilex*, cork oak (*Quercus suber*), magnolia (*Juglans regia*), coral (*Erythrina xsykesii*) and Monterey cypress (*Cupressus macrocarpa*) trees and an avenue of London Planetree (*Platanus x acerifolia*) trees along the road entrance (Auckland Regional Council, 2003b).

Seventeen native birds have been recorded in the park, including uncommon species such as the fern bird (*Bowdleria punctata vealeae*), banded rail (*Rallus philippensis assimilis*) and the North Island robin (*Petroica australis longipes*) which was reintroduced recently (Auckland Regional Council, 2003b). Species that are known to visit the park include: the threatened kaka (*Nestor meridionalis septentrionalis*), the long-tailed cuckoo (*Eudynamys taitensis*) and the bellbird (*Anthornis melanura*). A further eleven coastal native species are usually seen on the shoreline or off shore, including; the endangered New Zealand dotterel (*Charadrius obscurus*), the variable oystercatcher (*Haematopus unicolour*) and the reef heron (*Egretta sacra sacra*) (Auckland Regional Council, 2003b).

2.3.4. Cultural Heritage of Wenderholm Regional Park

The Maori name for Wenderholm is Te Awa Puhoi or the 'slow flowing river'. The flat sandy country was known as Te Akeake, and the headland as Maungatauhoro (Auckland Regional Council, 2003b). Twenty-nine archaeological sites have been identified: including a significant Pa on the headland, a waahi tapu on the sandspit, ten sites with pits and/or terraces and numerous midden sites. However, most of the headland sites are now under forest cover (Auckland Regional Council, 2003b). The Maori village, occupied after European contact, was located on the flats at the western end of the park. At the time of European settlement Te Kawerau and Ngati Rongo occupied the area (Auckland Regional Council, 2003b).

The land passed through several hands until the Couldrey family purchased it in 1940. In 1957 they undertook an ambitious restoration programme. In 1965, 127 hectares of land was sold to the Auckland Regional Authority, with the remaining seven hectares being acquired in 1973. Mahurangi Island was added to the Regional Park in 1999 (Auckland Regional Council, 2003b).

2.3.5. Visitor Numbers at Wenderholm Regional Park

Wenderholm Regional Park attracts between 500,000-1,000,000 visitors per year (Auckland Regional Council, 2003a). A study conducted in 1987/1988 found that visitor usage of the park is localised to the sand spit.

2.4. Methods

2.4.1. Seedling Survival within Exclosure Plots

The method used in this study to test whether *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus* inhibited the recruitment of tree and shrub seedlings followed, with some modifications, the method used by Wilson *et al.* (2003). At each regional park the *Leptospermum scoparium* and *Kunzea ericoides* vegetation was divided into a grid with 25 metres between the gridlines. Two exclosure cages (45x45x45cm) and one control plot (45x45cm) were placed under the *Leptospermum scoparium* and *Kunzea ericoides* vegetation. Exclosures consisted of two treatments with different mesh sizes: a possum (*Trichosurus vulpecula*) and rabbit (*Oryctolagus cuniculus*) exclosure (20mm mesh size) and; a possum (*Trichosurus vulpecula*), rabbit (*Oryctolagus cuniculus*) and rat (*Rattus rattus*) exclosure (6mm mesh size). The control plot was marked by wire mesh (20mm mesh size) laid flat on the ground. All exclosures were made of galvanized steel mesh and secured to the ground with weed mat pegs. The treatment methods are referred to hereafter as PRab (possum and rabbit exclosure) and PRabRat (possum, rabbit and rat exclosure) and Control. A total of 48 exclosure and 24 control plots were used at each regional park, to give a sample number of 24 for each treatment per regional park. However, the sample number for Wenderholm was reduced from 24 to 23 as the result of exclosure cages being pushed over, by people, during the study.

Exclosure cages and control plot mesh were left outdoors for two weeks prior to installation so that the rain would remove any coating that might be toxic to the plants. The exclosures and control plots were established under the vegetation in May 2004. At the same time the heights of all seedlings within the exclosures and control plots were recorded by measuring from the highest point to the ground and the genus and species name of each seedling was recorded. Wilson *et al.* (2003) removed existing seedlings from exclosures, whereas in this study pre-existing seedlings were retained, as it was not possible to remove them due to the protected status of the plants in regional parks.

The exclosure and control plots were monitored at 6-8 week intervals until February 2005 and then finally in October 2005, 17 months after establishing the plots. All woody seedlings that emerged were identified. The heights of seedlings which were originally present in May 2004 were recorded; the heights of any new seedlings which emerged were not recorded. All species were identified where possible. However, some browsed seedlings had no leaves and were not identifiable. Although herbaceous seedlings were present in the plots at both Regional Parks there was no attempt to quantify the abundance of them. Cotyledonous seedlings were not counted until they developed true leaves. Leaf litter accumulation on the exclosures was minimal but what did accumulate was tipped into the plot each time the plots were monitored. Accumulated leaf litter on the control plots was left in place.

The abundance of *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus*, was not assessed as continual pest control for these species is carried out on both regional parks, by poisoning, throughout the year**.** Annual culls of *Trichosurus vulpecula* and *Oryctolagus cuniculus*, by shooting, are also carried out on both the regional parks. As a result of the pest controls on the regional parks *Trichosurus vulpecula*, *Oryctolagus cuniculus* and *Rattus rattus* populations are considered to be low (pers comms Tim Lovegrove, Natural Heritage Scientist, Auckland Regional Council, 2006). Pest monitoring at Shakespear from July 2004 to February 2005 resulted in an average of 50 *Rattus rattus*, 12 *Oryctolagus cuniculus* and zero *Trichosurus vulpecula* killed per month. Pest monitoring at Wenderholm from July 2004 to February 2005 resulted in an average of 2 rats killed per month. In May 2005, 13 *Oryctolagus cuniculus* and 21 *Trichosurus vulpecula* were killed at Wenderholm. Stoat control at Wenderholm means that high *Oryctolagus cuniculus* numbers are an on-going problem (pers comms Tim Lovegrove, Natural Heritage Scientist, Auckland Regional Council, 2006).

2.4.2. Germinations from Seed Banks

The method used in this study followed the method used by Edwards and Crawley (1999). Seedbank samples were collected from four 18 metre long, transects set out on a compass bearing of 45^{o} at Shakespear Regional Park and at 180^{o} at Wenderholm Regional Park under the same *Leptospermum scoparium* and *Kunzea ericoides* vegetation that the exclosure plots were located in. Sixteen soil samples were taken at 1.1m intervals along each of the four transects under the *Leptospermum scoparium* and *Kunzea ericoides* canopy at each regional park, resulting in a total of 64 soil samples from each regional park. Soil samples were 5cm in diameter and taken to a depth of 16cm (Edwards & Crawley, 1999). Soil samples were collected in summer (January 2004).

The 64 soil samples from each regional park were combined and broken down by hand into a fine crumb. Roots, rhizomes and stones were removed. The soil was spread evenly in a 0.5-1cm layer over 3-4cm sterile soil in four plastic seed trays (40cmx30cmx5cm) resulting in two trays of soil from Shakespear and two trays of soil from Wenderholm. The trays were placed in a greenhouse at the Auckland Regional Botanical Gardens in January 2004 and monitored until March 2005 (fifteen months). A control tray of sterile soil was located in the glasshouse among the soil samples from the regional parks. Trays were watered every two-three days and the soil was stirred every four months to expose un-germinated seeds. Temperatures in the glasshouse ranged from 10-28^{o} Celsius.

All seedlings (germinable seeds) that emerged were identified and removed. Plants not identified were grown until identification was possible. Moss and liverworts were found in the control trays as well as the sample trays; this is a common occurrence at the Botanical Gardens due to the water (pers. comm. Steve Benham, Conservation Officer, Auckland Regional Botanical Gardens). All moss and liverworts were therefore excluded from the results. All seedlings that emerged from the soil samples collected were identified, counted and categorised into native and exotic species.

2.4.3. Seedling Recruitment into Canopy Gaps

Ten light gaps were created in June 2004 at each Regional Park within the same *Leptospermum scoparium* and *Kunzea ericoides* vegetation in which the exclosure plots were located. Each canopy gap plot was created by removing one or two trees with a saw at the base, until the gap in the canopy foliage was a minimum of one-metre wide. The cut trees were dragged to the edge of the canopy gap. Ten control plots were marked under the closed canopy of *Leptospermum scoparium* and *Kunzea ericoides* five metres from the canopy gap plots. All plots were a minimum of five metres away from the forest edge and any exclosure cages. All seedlings that emerged within a one-metre radius from the centre of each canopy gap plot and each closed canopy control plot were identified in October 2005; 16 months after the gaps were created.

2.5. Statistical Analysis

2.5.1. Exclosure Plots

The net change in seedling numbers for each plot was established by subtracting the final count of seedlings from the initial number of seedlings present at the start of the study. Therefore, a decline in seedling abundance was recorded as a negative net change. The average net change in seedling abundance over 17 months, per plot was calculated and compared among treatments. Seedling changes among treatments were then compared using the Kruskal-Wallis test.

The net change in seedling height for each seedling was calculated by subtracting the final height from the initial height. Therefore, seedlings that became smaller (e.g. from browsing) were recorded as a negative net change. Only those seedlings present at the start of the study were included. The data was initially analysed using ANOVA but residuals were found to be 'not normal' and the data was therefore re-analysed using the non-parametric Kruskal-Wallis test.

The net change in the number of seedlings of each species over 17 months was calculated for each treatment and compared among treatments.

2.5.2. Seedling Recruitment into Canopy Gaps

The hypothesis that more seedlings would establish beneath artificial canopy gaps than under a closed canopy of *Leptospermum scoparium* and *Kunzea ericoides* was tested by comparing the number of seedlings that emerged and remained to the end of the 16-month monitoring period within the gaps, with the number that emerged under the closed canopy using a one-tail Mann-Whitney test. This test was performed for all seedlings and then separately for native seedlings.

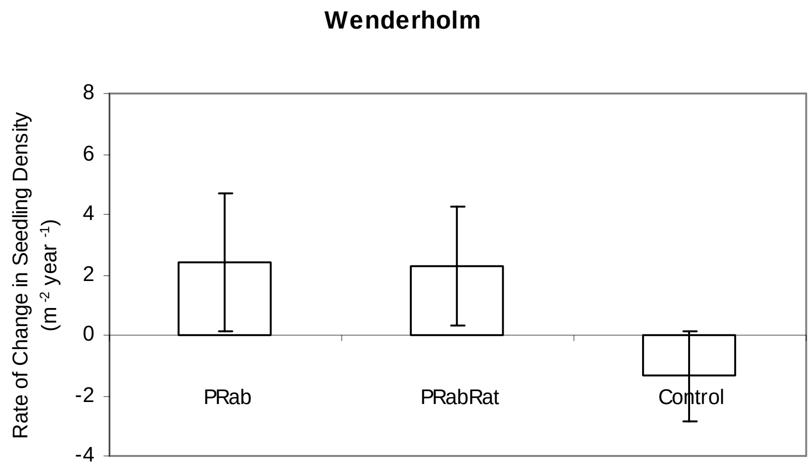
3.1. Seedling Survival

At Wenderholm, there was a significant difference among treatments in the net change of seedling numbers (Kruskal Wallis Test, P=0.014) (Figure 3). The mean density of seedlings within the PRab exclosures increased by 2.42 ±2.29 seedlings m^{-2} $year^{-1}$ (61% increase per year). The mean density of seedlings within the PRabRat exclosures also increased (2.27 ±2 seedlings m^{-2} $year^{-1}$ 52% increase per year). In contrast, the mean density of seedlings within the control plots decreased by 1.36 ±1.48 seedlings m^{-2} $year^{-1}$ (51% decrease per year) (Figure 3). P-values for pairwise comparison are presented in Table 1.

Table 1. P-values for pairwise comparisons of changes in seedling density within possum and rabbit exclosures (PRab); possum, rabbit and rat exclosures (PRabRat); and control plots for Wenderholm Regional Park

	PRab	**PRabRat**	**Control**
PRab		0.9650	0.0211
PRabRat			0.0105

At Shakespear, there was a greater increase in the seedling density within exclosures than in the control plots. However, this difference was not statistically significant (Kruskal Wallis Test, P=0.728) (Figure 3). The mean density of seedlings within the PRabRat exclosures increased by 2.75 ±4.18 seedlings m^{-2} $year^{-1}$ (99% increase per year), and the mean density of seedlings within the PRab exclosures increased by 1.02 ±1.26 seedlings m^{-2} $year^{-1}$ (49% increase per year). However, seedling density within the control plots only increased by 0.15 ±0.53 seedlings m^{-2} $year^{-1}$ (9.9% increase per year) (Figure 3).



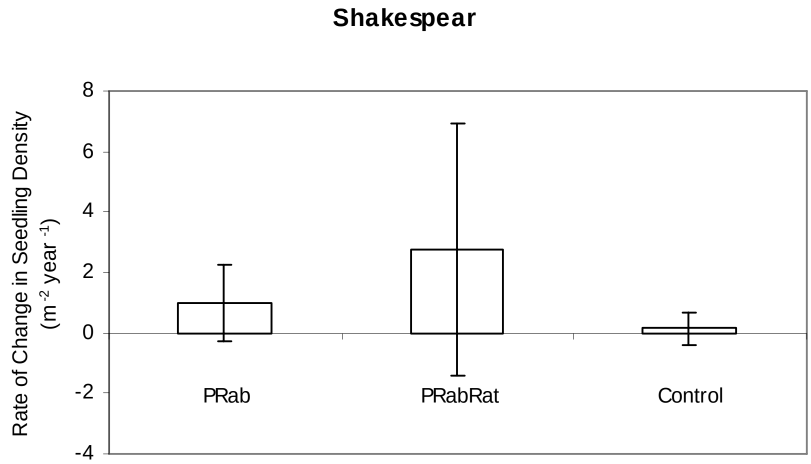
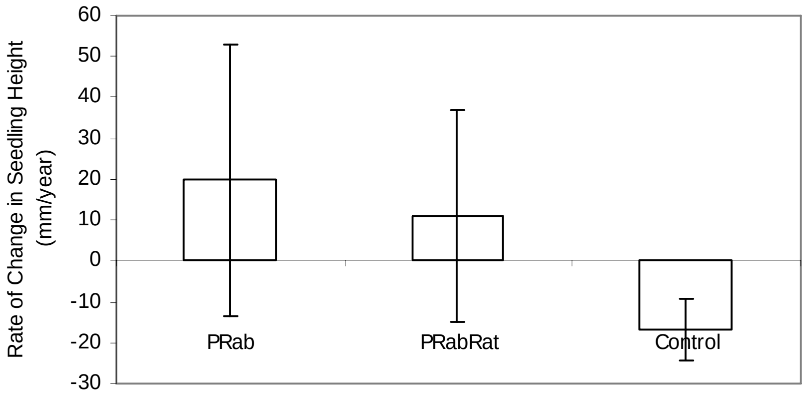


Figure 3. Net change in the seedling density within possum and rabbit exclosures (PRab); possum, rabbit and rat exclosures (PRabRat); and control plots at Wenderholm and Shakespear Regional Parks. 95% confidence intervals shown. Wenderholm n=23 plots and Shakespear n=24 plots for each treatment respectively.

At Wenderholm, there was an increase in the seedling heights within exclosures and a decrease in the control plots. However, this difference was not statistically significant (Kruskal Wallis Test, P=0.204) (Figure 4). The mean height of seedlings within the PRab exclosures increased by 19.8 ±33.17mm/year (22% increase in height) and the mean height of seedlings within the PRabRat exclosures increased by 11.18 ±25.94mm/year (31% increase in height). In contrast, the mean height of the seedlings within the control plots decreased by 16.86 ±7.36mm/year (70% decrease in height) (Figure 4).

Nor was there a significant difference among treatments in the net change in seedling heights at Shakespear (Kruskal Wallis Test, P=0.202) (Figure 4). The mean height of seedlings within the PRabRat exclosures increased by 13.61 ±23.72 mm/year (64% increase in height). In contrast, the mean height of seedlings within the PRab exclosures and control plots decreased (5.88 ±15.82mm/year (35%) and 1.32 ±10.45mm/year (5%) respectively) (Figure 4).

Wenderholm



Shakespear

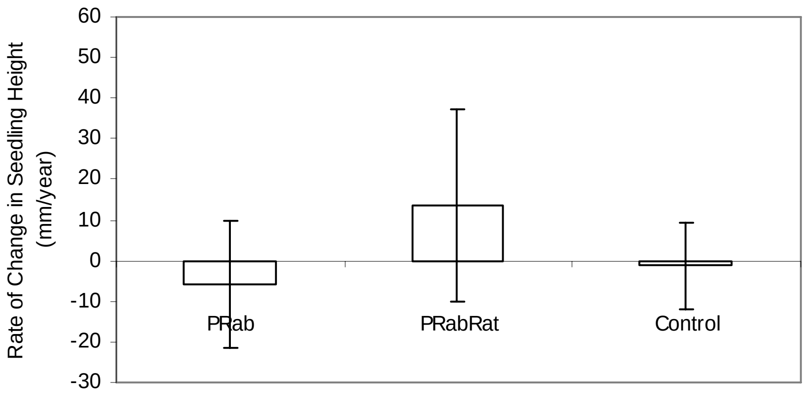


Figure 4. Mean change in seedling heights within possum and rabbit exclosures (PRab); possum, rabbit and rat exclosures (PRabRat); and control plots at Wenderholm and Shakespear Regional Parks. 95% confidence intervals shown. Wenderholm n=23 plots and Shakespear n=24 plots for each treatment respectively.

3.2. Effects of Predation on Seedling Species

3.2.1. Species Originally Present

At Shakespear, *Myrsine australis* was the dominant species originally present within the PRab exclosures; and PRabRat exclosures. *Melicytus ramiflorus* was the dominant species originally present within the control plots. At Wenderholm, *Myrsine australis* was the dominant species originally present within all treatment plots. However, at Wenderholm, *Myrsine australis* also suffered the greatest mortality in both the exclosure plots and control plots (0.31 seedlings m^{-2} $year^{-1}$ (PRab), 0.76 seedlings m^{-2} $year^{-1}$ (PRabRat) and 1.67 seedlings m^{-2} $year^{-1}$ (control)). *Sophora tetraptera* had the largest increase in the number of seedlings in both the exclosure plots and control plots (1.52 seedlings m^{-2} $year^{-1}$ (PRab), 1.51 seedlings m^{-2} $year^{-1}$ (PRabRat) and 0.15 seedlings m^{-2} $year^{-1}$ (control)). *Melicytus ramiflorus* also increased in both the exclosure plots and control plots (1.21 seedlings m^{-2} $year^{-1}$ (PRab), 0.45 seedlings m^{-2} $year^{-1}$ (PRabRat) exclosures, and 0.15 seedlings m^{-2} $year^{-1}$ (control)). *Macropiper excelsum* was not present in the PRab exclosures, but increased by 0.61 seedlings m^{-2} $year^{-1}$ in the PRabRat exclosures and remained unchanged in the control plots (Table 2).

At Shakespear, there was no decline in seedling density of any of the species present. *Melicytus ramiflorus* increased within both the exclosure plots (0.15 seedlings m^{-2} $year^{-1}$ (PRab) and 0.88 seedlings m^{-2} $year^{-1}$ (PRabRat)). By comparison, *Melicytus ramiflorus* and *Myrsine australis* did not increase in abundance in the control plots. *Myrsine australis* was the species that increased the most within the PRabRat exclosures with an increase of 1.59 seedlings m^{-2} $year^{-1}$. It also increased in the PRab exclosures (0.29 seedlings m^{-2} $year^{-1}$). *Coprosma areolata* was the only species that increased in the control plots (1.03 seedlings m^{-2} $year^{-1}$). *Coprosma areolata* increased within both the exclosure plots (0.29 seedlings m^{-2} $year^{-1}$ (PRab) and 0.14 seedlings m^{-2} $year^{-1}$ (PRabRat)). *Coprosma robusta* also increased in both exclosure plots (0.29 seedlings m^{-2} $year^{-1}$ (PRab) and 0.15 seedlings m^{-2} $year^{-1}$ (PRabRat)) (Table 2).

Table 2. Density of seedlings, by species, that were originally present and that emerged during the study at Wenderholm and Shakespear Regional Parks. Wenderholm n=23 plots and Shakespear n=24 plots for each treatment respectively.

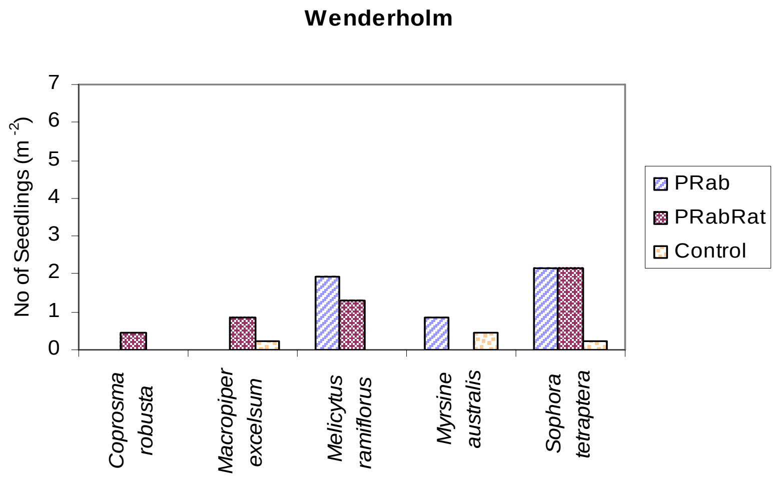
Shakespear	**Seedlings originally present**								
	Possum and Rabbit exclosure (PRab)			**Possum, Rabbit and Rat exclosure (PRabRat)**			**Control plot**		
	May 2004 (m^{-2})	**October 2005 (m^{-2})**	**Difference (m^{-2} $year^{-1}$)**	**May 2004 (m^{-2})**	**October 2005 (m^{-2})**	**Difference (m^{-2} $year^{-1}$)**	**May 2004 (m^{-2})**	**October 2005 (m^{-2})**	**Difference (m^{-2} $year^{-1}$)**
Coprosma areolata		0.41	0.29	0.21	0.41	0.14		0.21	0.15
Coprosma robusta		0.41	0.29		0.21	0.15			
Melicytus ramiflorus		0.21	0.15		1.03	0.88	1.03	1.03	0.00
Myrsine australis	1.65	2.06	0.29	1.44	3.70	1.59	0.41	0.41	0.00
All species combined	**1.65**	**3.09**	**1.02**	**1.65**	**5.56**	**2.76**	**1.44**	**1.65**	**0.15**

Wenderholm	**Seedlings originally present**								
	Possum and Rabbit exclosure (PRab)			**Possum, Rabbit and Rat exclosure (PRabRat)**			**Control plot**		
	May 2004 (m^{-2})	**October 2005 (m^{-2})**	**Difference (m^{-2} $year^{-1}$)**	**May 2004 (m^{-2})**	**October 2005 (m^{-2})**	**Difference (m^{-2} $year^{-1}$)**	**May 2004 (m^{-2})**	**October 2005 (m^{-2})**	**Difference (m^{-2} $year^{-1}$)**
Coprosma robusta				0.21	0.86	0.46			
Macropiper excelsum				0.43	1.29	0.61	0.21	0.21	0.00
Melicytus ramiflorus	0.64	2.36	1.21	0.43	1.07	0.45		0.22	0.15
Myrsine australis	2.15	1.72	-0.31	2.15	1.07	-0.76	3.01	0.64	-1.68
Podocarpus totara	0.21	0.21	0.00						
Prumnopitys ferruginea				0.22	0.22	0.00			
Sophora tetraptera		2.15	1.52		2.15	1.51		0.21	0.15
All species combined	**3.00**	**6.44**	**2.42**	**3.44**	**6.66**	**2.27**	**3.22**	**1.29**	**-1.38**

3.2.2. *New Recruitment of Seedlings*

At Wenderholm, there was a large number of new recruitments of *Sophora tetraptera* in both of the exclosure plots (2.15 seedlings m^{-2} (PRab and PRabRat)), but only a small number in the control plots (0.21 seedlings m^{-2}). New recruitments of *Melicytus ramiflorus* was also high in both of the exclosure plots (1.93 seedlings m^{-2} (PRab) and 1.29 seedlings m^{-2} (PRabRat)) and absent in the control plots (Figure 5).

New seedlings to emerge at Shakespear in exclosures were mainly *Myrsine australis* (1.29 seedlings m^{-2} (PRab) and 6.58 seedlings m^{-2} (PRabRat)), with only a few emerging in the control (0.21 seedlings m^{-2}). *Melicytus ramiflorus* had a large number of new recruits in the PRabRat exclosures (2.47 seedlings m^{-2}), but only a few in the PRab exclosures (0.22 seedlings m^{-2}) (Figure 5).



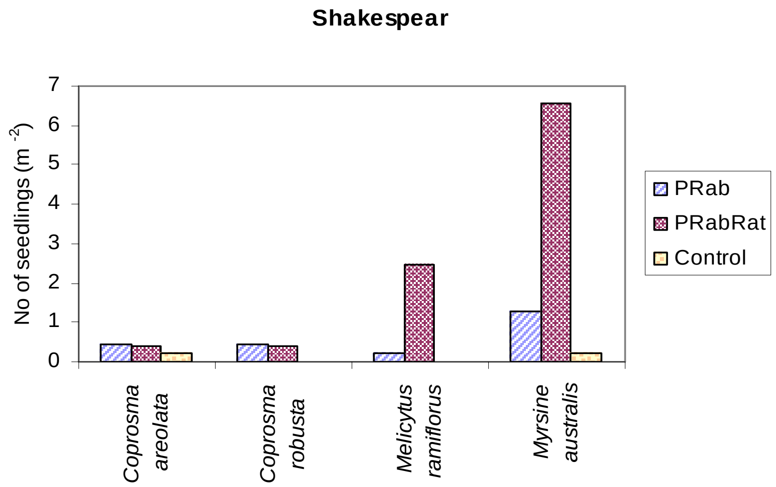
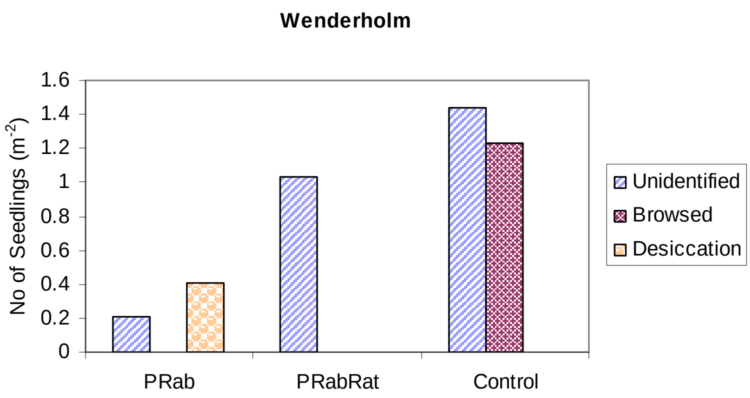


Figure 5. Number of new seedlings that emerged over the total study period (17 months) within possum and rabbit exclosures (PRab); possum, rabbit and rat exclosures (PRabRat); and control plots at Wenderholm and Shakespear Regional Parks. Wenderholm n=23 and Shakespear n=24 exclosure plots respectively.

3.3. Causes of Mortality

At Wenderholm, animal browsing was the greatest identified cause of mortality within the control plots (1.23 seedlings m^{-2}, 46% mortality). However, the cause of most of the seedling mortalities within the control plots could not be identified (1.44 seedlings m^{-2}, 54% mortality) as the seedlings could not be found at the time of examination. None of the seedling mortalities in the PRabRat exclosures could be identified (1.03 seedlings m^{-2}, 100% mortality). In contrast, in the PRab exclosures the greatest cause of mortality was due to desiccation (0.41 seedlings m^{-2}, 67% mortality) followed by mortalities which could not be identified (0.21 seedlings m^{-2}, 33% mortality) (Figure 6). At Shakespear, all seedling mortalities in the control plots was due to animal browsing (0.41 seedlings m^{-2}, 100% mortality). All of the seedling mortalities within the PRabRat exclosures could not be identified (1.03 seedlings m^{-2}, 100% mortality). In contrast, in the PRab exclosures the greatest cause of mortality was due to desiccation (0.62 seedlings m^{-2}, 75% mortality) followed by mortalities which could not be identified (0.21 seedlings m^{-2}, 25% mortality) (Figure 6).



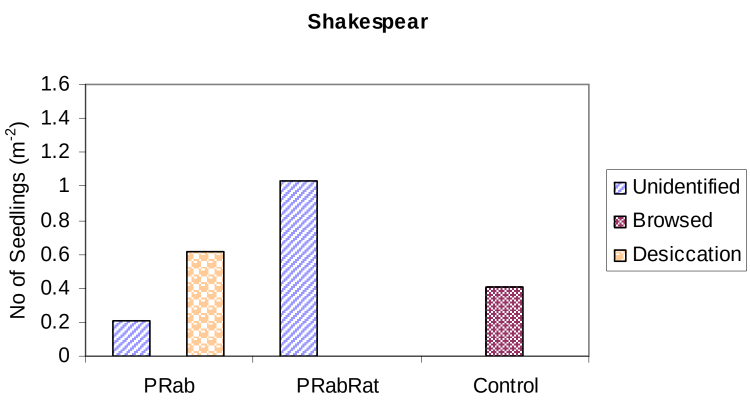


Figure 6. Causes of mortality for seedlings over the total study period (17 months) within possum and rabbit exclosures (PRab); possum, rabbit and rat exclosures (PRabRat); and control plots at Wenderholm and Shakespear Regional Parks. Wenderholm n=23 exclosure plots and Shakespear n=24 exclosure plots for each treatment respectively.

3.4. Germinations from Seed Banks

A total of 1308 seedlings germinated over a period of fifteen months from soil taken from Wenderholm, these included nineteen families and thirty-one species. A total of 801 seedlings germinated from soil samples taken from Shakespear, these included twenty-one families and twenty-nine species (Table 3). Of these, only three native species germinated from soil taken from each park. Species that germinated from Wenderholm were *Cordyline australis*, *Pteridium esculentum* and *Coprosma robusta* and species from Shakespear were *Cordyline australis, Metrosideros excelsa* and *Pteridium esculentum* (Table 3). Exotic species made up 99.4% of viable seed at Wenderholm and 97.9% of viable seed at Shakespear (Table 3). Seed banks at Wenderholm were dominated by *Cyperaceae* species, *Ranunculus repens* and *Lotus suaveolens* (Table 3). Seed banks at Shakespear were dominated by *Isolepis cernua, Lotus suaveolens* and *Poaceae* species (Table 3).

Table 3. Total number of seedlings germinated from combined soil of 0.13 square metres. 64 cylindrical samples ($5cm^2$ by 16cm deep), germinated over a period of fifteen months.

Wenderholm				**Shakespear**			
Family	**Species**	**Common Name**	**No. of seedlings**	**Family**	**Species**	**Common Name**	**No. of seedlings**
Amaranthaceae	*Amaranthus* spp		1				
				Apiaceae	*Daucus carota*		3
Asteraceae	*Carduus nutans*	nodding thistle	13	Asteraceae	*Carduus nutans*	nodding thistle	27
	Conyza canadensis	canadian fleabane	11		*Conyza canadensis*	canadian fleabane	3
	Picris echioides	oxtongue fleabane	21		*Picris echioides*	oxtongue fleabane	5
					Sonchus oleraceus	sow thistle	2
Brassicaceae	*Cardamine hirsuta*	bitter cress	8	Brassicaceae	*Cardamine hirsuta*	bitter cress	25
Caryophyllaceae	*Cerastium glomeratum*	annual mouse-ear chickweed	2	Caryophyllaceae			
	Sagina procumbens	procumbent pearlwort	12		*Sagina procumbens*	procumbent pearlwort	3
	Stellaria media	chickweed	1				
Cyperaceae	*Carex* spp	sedge	29	Cyperaceae	*Carex* spp		4
	Cyperaceae spp		266				
	Isolepis spp		125		*Isolepis cernua*		284
Euphorbiaceae	*Euphorbia* spp	spruge	1				
Fabaceae	*Lotus suaveolens*	hairy birdsfoot trefoil	194	Fabaceae	*Lotus suaveolens*	hairy birdsfoot trefoil	170
	Trifolium repens	white clover	7		*Trifolium repens*	white clover	5
	Vicia spp		1				
				Gentianaceae	*Centaurium erythraea*	centuary	4
				Geraniaceae	*Geranium molle*	dove"s foot cranesbill	7
				Iridaceae	*Libertia* spp		2
Juncaceae	*Juncus* spp	rush	68	Juncaceae	*Juncus* spp	rush	22
Laxmanniaceae	***Cordyline australis***^	cabbage tree	4	Laxmanniaceae	***Cordyline australis***^	cabbage tree	5
				Myrtaceae	***Metrosideros excelsa***^	pohutukawa	5
Onagraceae	*Epilobium* spp	willow herb	6	Onagraceae	*Epilobium* spp	willow herb	23
				Oxalidaceae	*Oxalis corniculata*	horned oxalis	2
Phytolaccaceae	*Phytolacca octandra*	inkweed	6				

Continued…

Table 3. Continued…

Wenderholm				Shakespear			
Family	**Species**	**Common Name**	**No. of seedlings**	**Family**	**Species**	**Common Name**	**No. of seedlings**
Poaceae				Poaceae	*Holcus lanatus*	yorkshire fog	7
	Holcus mollis	creeping fog	3				
					Lolium perenne	perennial rye grass	1
	Poaceae spp		141		*Poaceae* spp		117
Polygonaceae				Polygonaceae	*Polygonum* spp		1
	Rumex obtusifolius	broad-leaved dock	34		*Rumex obtusifolius*	broad-leaved dock	52
Primulaceae	*Anagallis arvensis*	pimpernel	46	Primulaceae	*Anagallis arvensis*	pimpernel	10
Pteridaceae	***Pteridium esculentum*^**	bracken	2	Pteridaceae	***Pteridium esculentum*^**	bracken	7
Ranunculaceae	*Ranunculus repens*	buttercup	236	Ranunculaceae	*Ranunculus repens*	buttercup	3
Rubiaceae	***Coprosma robusta*^**	karamu	1	Rubiaceae			
	Galium aparine	cleavers	1		*Galium aparine*	cleavers	1
Scrophulariaceae	*Veronica serpyllifolia*	turf speedwell,	8				
Solanaceae	*Nicandra physalodes*	apple of Peru	54	Solanaceae			
	Solanum mauritianum	woolly nightshade	2				
	Solanum nigrum	black nightshade	4		*Solanum nigrum*	black nightshade	1
Total			**1308**	**Total**			**801**

^ Native species

3.5. Seedling Recruitment into Canopy Gaps

3.5.1. Abundance and Species Diversity

At Wenderholm, 3.60 seedlings m^{-2} $year^{-1}$ comprising of 12 families and 18 species germinated over sixteen months within the canopy gap plots. In the closed canopy plots 0.54 seedlings m^{-2} $year^{-1}$ of two families and two species germinated (Table 4). At Shakespear, 2.58 seedlings m^{-2} $year^{-1}$ comprising of seven families and ten species germinated over sixteen months within the canopy gap plots. In the closed canopy plots 0.67 seedlings m^{-2} $year^{-1}$ comprising of five families and six species germinated over the same period (Table 5).

Exotic species made up 84.4% of seedlings that germinated within the canopy gap plots and 66.7% within the closed canopy plots at Wenderholm (Table 4). Exotic species made up 77.4% of seedlings that germinated within the canopy gap plots and 36.7% within the closed canopy plots at Shakespear (Table 5).

Canopy gap plots at Wenderholm were dominated by S*onchus oleraceus* (0.85 seedlings m^{-2}), grass species (0.54 seedlings m^{-2}) and *Picris echioides* (0.36 seedlings m^{-2}). In the closed-canopy plots the dominant species was unknown (0.36 seedlings m^{-2}) as they were at the cotyledon stage. The only identified species was *Sophora tetraptera* (0.18 seedlings m^{-2}) (Table 4). Canopy gap plots at Shakespear were dominated by *Galium aparine* (0.99 seedlings m^{-2}), M*yrsine australis* (0.43 seedlings m^{-2}) and grass species (0.40 seedlings m^{-2}). In the closed canopy plots the dominant species were all native, C*oprosma areolata* (0.18 seedlings m^{-2}), *Myrsine australis* (0.13 seedlings m^{-2}) and C*yathea dealbata* (0.11 seedlings m^{-2}) (Table 5).

Table 4. Number of seedlings that emerged over the total study period (sixteen months) at Wenderholm Regional Park within canopy gaps and closed canopy plots, sorted by family.

Wenderholm							
Canopy Gap				**Closed Canopy**			
Family	**Species**	**Common Name**	**Density Seedlings (m^{-2})**	**Family**	**Species**	**Common Name**	**Density Seedlings (m^{-2})**
Asclepiadaceae	*Araujia sericifera*	moth plant	0.02				
Asteraceae	*Carduus nutans*	nodding thistle	0.18				
	Picris echioides	oxtongue fleabane	0.36				
	Sonchus oleraceus	sow thistle	0.85				
Cyperaceae	*Carex species*	sedge	0.07				
Fabaceae	*Lotus suaveolens*	hairy birdsfoot trefoil	0.11	Fabaceae			
	***Sophora tetraptera*^**	kowhai	0.04		***Sophora tetraptera*^**	kowhai	0.18
Haloragaceae	***Haloragis erecta subsp. Erecta*^**	shrubby haloragis	0.02				
Myrsinaceae	***Myrsine australis*^**	red mapou	0.16				
Poaceae	***Oplismenus hirtellus subsp. Imbecillus*^**		0.02				
Podocarpaceae	***Podocarpus totara*^**	totara	0.02				
Solanaceae	*Solanum mauritianum*	woolly nightshade	0.25				
	Solanum nigrum	black nightshade	0.34				
Verbenaceae	***Vitex lucens*^**	puriri	0.02				
Violaceae	***Melicytus ramiflorus*^**	mahoe	0.27				
Unknown	Grass species		0.54	Unknown	Unknown*		0.36
	Thistle species		0.04				
	Unknown*		0.27				
Total			**3.60**	**Total**			**0.54**

^ Native species * Unknown species were at the cotyledon stage

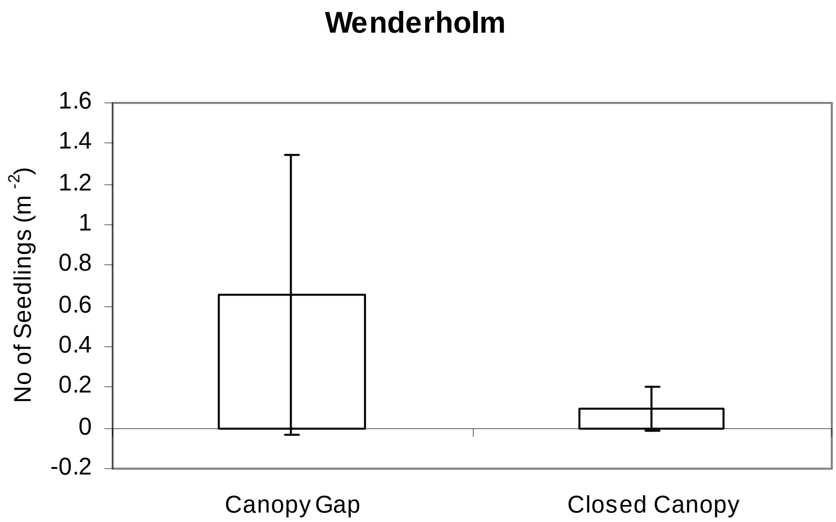
Table 5. Number of seedlings that emerged over the total study period (sixteen months) at Shakespear Regional Park within canopy gaps and closed canopy plots, sorted by family.

Shakespear							
Canopy Gap				**Closed Canopy**			
Family	**Species**	**Common Name**	**Density Seedlings (m^{-2})**	**Family**	**Species**	**Common Name**	**Density Seedlings (m^{-2})**
Apiaceae	*Daucus carota*	wild carrot	0.34	Apiaceae	*Daucus carota*	wild carrot	0.09
Asteraceae	*Carduus nutans*	nodding thistle	0.04				
Cyatheaceae	***Cyathea dealbata***^	silver fern	0.11	Cyatheaceae	***Cyathea dealbata***^	silver fern	0.11
Myrsinaceae	***Myrsine australis***^	red mapou	0.43	Myrsinaceae	***Myrsine australis***^	red mapou	0.13
Polygonaceae	*Rumex obtusifolius*	broadleaf dock	0.02				
Rubiaceae	***Coprosma areolata***^	thin-leaved coprosma	0.02	Rubiaceae	***Coprosma areolata***^	thin-leaved coprosma	0.18
	Coprosma robusta^	Karamu	0.02				
	Galium aparine	Cleavers	0.99				
Unknown	Grass species		0.40	Unknown	Grass species		0.07
	Unknown*		0.20		Unknown*		0.09
Total			**2.58**	**Total**			**0.67**

^ Native species, * Unknown species were at the cotyledon stage

3.5.2. Average Density of All Seedlings

At Wenderholm, there was a greater increase in the mean density of seedlings within the artificially created canopy gap plots (0.65 ±0.69 seedlings m^{-2}) than in the closed canopy plots (0.09 ±0.11 seedlings m^{-2}) (Mann-Whitney, *P*=0.0018) (Figure 7). At Shakespear, the mean density of seedlings also increased more within the artificially created canopy gap plots (0.25 ±0.19 seedlings m^{-2}) than in the closed canopy plots (0.07 ±0.04 seedlings m^{-2}) (Mann-Whitney, *P*=0.029) (Figure 7).



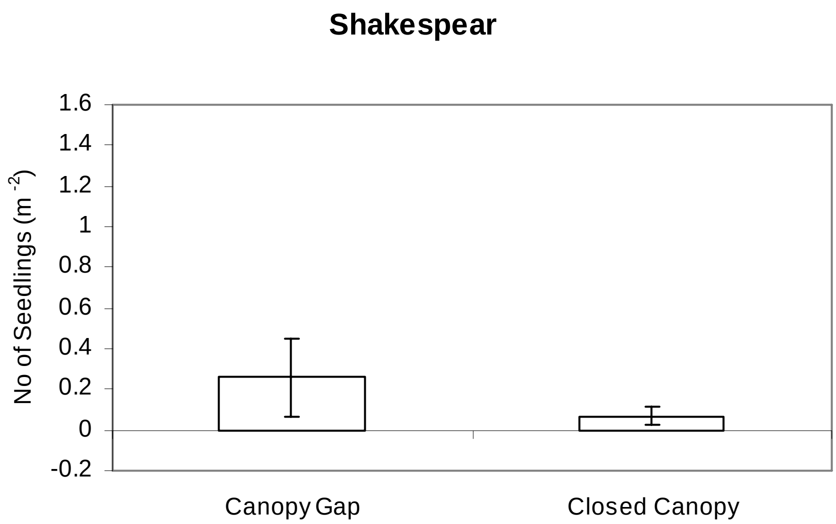
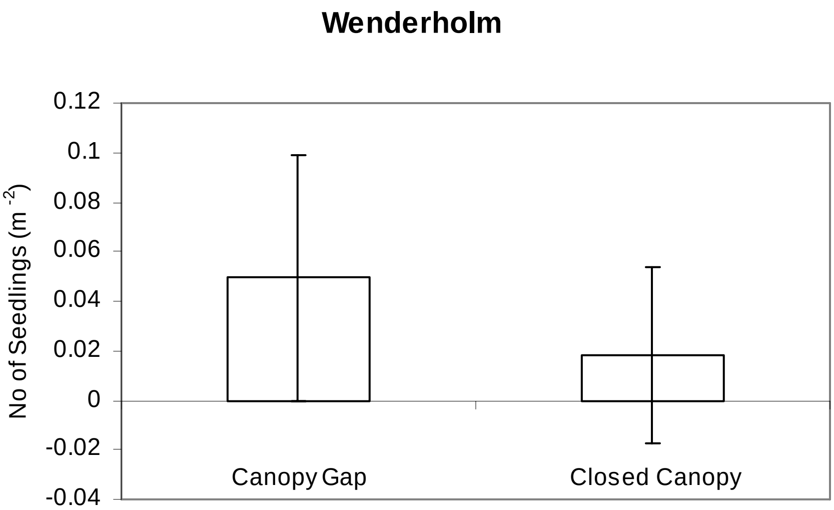


Figure 7. ***Mean density of seedlings over sixteen months within canopy gaps and closed canopy plots at Wenderholm and Shakespear Regional Parks. 95% confidence intervals shown. n = 10 canopy gap plots and 10 closed canopy plots at each park.***

3.5.3. Average Density of Native Seedlings

At Wenderholm, there was a greater increase in the number of native seedlings within the artificially created canopy gap plots (0.05 ±0.05 seedlings m^{-2}) than in the closed canopy plots (0.02 ±0.04 seedlings m^{-2}). However, this difference was marginally non-significant (Mann-Whitney, P=0.0653) (Figure 8). At Shakespear, the number of native seedlings that established within the artificially created canopy gap plots (0.06 ±0.03 seedlings m^{-2}) was similar to the number of native seedlings that established within the closed canopy plots (0.04 ±0.03 seedlings m^{-2}). However, this difference was also not statistically significant (Mann-Whitney, P=0.2603) (Figure 8).



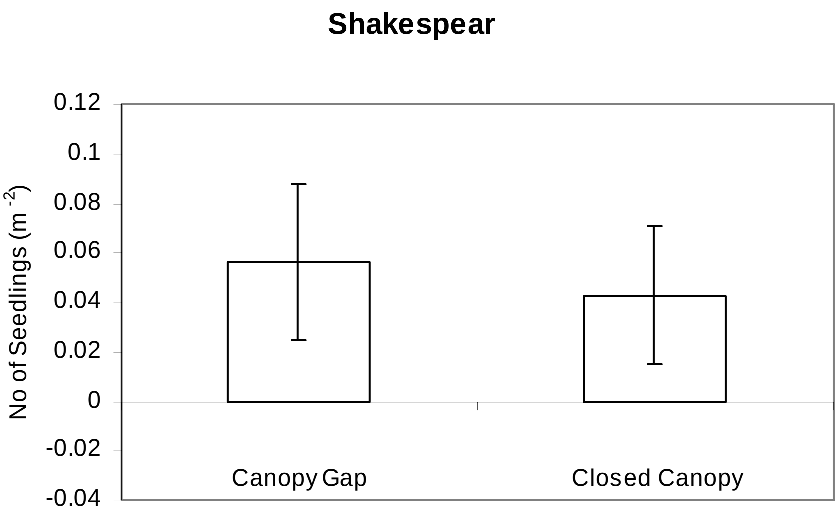


Figure 8.** **Mean density of native seedlings over sixteen months within canopy gaps and closed canopy plots at Wenderholm and Shakespear Regional Parks. 95% confidence intervals shown. n = 10 canopy gap plots and 10 closed canopy plots at each park.

Chapter Four: Discussion

4.1 The Effects of Mammals on Seedling Survival and Growth

At Wenderholm, seedling numbers increased in both PRab and PRabRat exclosure plots, whereas seedling densities declined in the control plots. Although there was not a statistically significant difference between treatments for changes in average seedling heights, the same pattern was evident: heights increased in the exclosures and decreased in the control plots. These results taken together indicate that at Wenderholm *Trichosurus vulpecula* and/or *Oryctolagus cuniculus* are adversely affecting seedling survival through browsing and/or non-trophic damage. At Shakespear, PRab and PRabRat exclosure plots had no statistically significant influence on seedling densities, growth rates or survival. Nonetheless, there was some indication that excluding *Rattus rattus* in addition to *Trichosurus vulpecula* and *Oryctolagus cuniculus* may have allowed greater seedling survival and growth and a longer-term study may be able to demonstrate this.

At both regional parks, excluding *Rattus rattus* in addition to *Trichosurus vulpecula* and *Oryctolagus cuniculus* did not significantly increase seedling densities compared with excluding only *Trichosurus vulpecula* and *Oryctolagus cuniculus*. There are two possible explanations for these results: firstly, *Rattus rattus* may be present but not damaging seedlings or; secondly, there may have been low *Rattus rattus* numbers in Wenderholm. Wilson *et al* (2003) found little influence on seedling numbers at sites with low *Rattus rattus* numbers, but sites with high *Rattus rattus* numbers reduced seedling numbers. Regular control and monitoring in both regional parks shows that at Wenderholm rat populations are very low compared to *Trichosurus vulpecula* and *Oryctolagus cuniculus* and at Shakespear rat populations are higher than *Trichosurus vulpecula* and *Oryctolagus cuniculus* populations.

The greatest cause of mortality within the PRab exclosures at both regional parks was desiccation. The number of mortalities that were unidentifiable, within the PRabRat exclosures makes the results difficult to interpret. No evidence of animal pests (e.g. *Mus musculus*) getting into the exclosures was observed, which would suggest that the mortalities were from natural fluctuations of seedling numbers due to causes such as desiccation, or from damage caused by insects and other invertebrate browsers. Within the control plots at Wenderholm, browsing was the cause of most of the identifiable mortalities, and at Shakespear the only cause of mortality within the control plots was browsing. These results also suggest that *Oryctolagus cuniculus* and/or *Rattus rattus* are having an impact on seedling regeneration.

At Wenderholm, new seedlings to emerge within PRab and PRabRat exclosures included *Sophora tetraptera* and *Melicytus ramiflorus*. At Shakespear, new seedlings to emerge within PRab and PRabRat exclosures included *Myrsine australis* and *Melicytus ramiflorus*, with *Myrsine australis* having the largest increase within the PRabRat exclosures. The effects of *Rattus rattus* on islands have been widely studied and the abundance of seedlings for some woody species increased after rat eradication including *Melicytus ramiflorus* and *Myrsine australis* (Allen *et al.*, 1994; Campbell, 2002; Campbell & Atkinson, 1999, 2002). However there is little information available as to the impact of *Oryctolagus cuniculus* or *Rattus rattus* on *Sophora tetraptera*. Campbell & Atkinson (2002) have suggested that *Rattus rattus* have had an impact on *Sophora microphylla*. Although their exclosure trials did not confirm the results, the abundance of *Sophora microphylla* on Mercury Island in the absence of *Rattus rattus* suggests that *Rattus rattus* may have limited the regeneration of *Sophora microphylla*.

Numerous studies on New Zealand islands have found that *Rattus rattus* have had an adverse effect on seedling numbers (Campbell *et al.*, 1984; Allen *et al.*, 1994; Campbell & Atkinson, 1999, 2002) including the species *Melicytus ramiflorus* (Campbell & Atkinson, 1999) and *Myrsine australis* (Allen *et al.*, 1994). *Trichosurus vulpecula* are also known to have an adverse effect on seedlings, of *Melicytus ramiflorus* (Coleman *et al.*, 1985; Nugent *et al.*, 1997; Fitzgerald & Gibb, 2001) and *Myrsine australis* (Nugent *et al*., 2001). Although previous studies have shown that *Trichosurus vulpecula* are known to eat seedlings over 10cm (Atkinson, 1992; McArthur *et al.*, 2000; Wilson *et al.*, 2003), there appears to be no information to confirm that *Trichosurus vulpecula* eat seedlings under 10 cm. There is little information on the effect of *Oryctolagus cuniculus* on seedlings. However, Gillman & Ogden (2003) found that all non-trophic damage to seedlings at Huapai Scenic Reserve ceased follow *Oryctolagus cuniculus* control, which suggests that *Oryctolagus cuniculus* can have an adverse effect on seedlings.

Gillman & Ogden (2003) also found that non-trophic damage to seedlings decreased with increasing distance from the forest edge. The distance from the forest edge to seedling mortalities was not measured in this study. However, the *Leptospermum scoparium* and *Kunzea ericoides* planting at Shakespear was narrow, 15-20 metres wide, and surrounded by pasture. At Wenderholm, the *Leptospermum scoparium* and *Kunzea ericoides* planting was on the edge of mature forest and ranged up to 50 metres wide. This suggests that *Oryctolagus cuniculus* would be present throughout the *Leptospermum scoparium* and *Kunzea ericoides* plantings in both regional parks.

4.2 Seedling Recruitment from Seed Banks

The seed banks for both Wenderholm and Shakespear regional parks were dominated by forb species. This is consistent with previous studies, which found that forb seeds dominated seed banks (Sem & Enright, 1995, 1996; Tucker & Murphy, 1997; Edwards & Crawley, 1999) and that native woody species are usually poorly represented in seed bank studies (Ogden, 1985; Sem & Enright, 1996). The forb seedlings that germinated in the soil samples at Wenderholm and Shakespear regional parks may have persisted in the seed banks under the *Leptospermum scoparium* and *Kunzea ericoides* canopies from the time when the area was pasture, or they may have dispersed from the pasture at the forest edge. Forb and grass species may have a negative impact on regeneration through competition with native seedlings. The small proportion of bird dispersed native seedlings at Wenderholm included *Coprosma robusta*, *Cordyline australis* and at Shakespear included *Cordyline australis*, suggesting that few seeds are being brought into the planted areas by birds at this time. The only wind dispersed native seedlings at Wenderholm was *Pteridium esculentum* and at Shakespear included M*etrosideros excelsa* and *Pteridium esculentum*.

The results for both regional parks suggest that only a limited range of native species will germinate in the near future. There are three possible explanations for this result: firstly, recruitment of native seeds is limited by seed dispersal; secondly, the *Leptospermum scoparium* and *Kunzea ericoides* canopy cover is limiting the amount of light required for native seedlings to establish; thirdly, native seeds are not surviving in the seed bank under the *Leptospermum scoparium* and *Kunzea ericoides*. Enright and Cameron (1998) found that seed rain was more similar to the existing canopy vegetation than the persistent seed bank. Sem & Enright (1996) estimated that only 10% of seed rain inputs to the forest floor might enter the persistent soil seed bank and that seeds of most species represented in the seed rain are short-lived (i.e. 2 months to 1 year). Therefore, it is possible that seed dispersal and the current closed canopy of the *Leptospermum scoparium* and *Kunzea ericoides* plantings in both of the regional parks is limiting the number of native seedling germinations at this time.

4.3 Seedling Recruitment into Canopy Gaps

The seedlings that emerged within the canopy gaps at both regional parks were mainly forb species. Many more seedlings emerged within the canopy gaps than under the closed canopy. More native seedlings also established within the canopy gaps than within the closed canopy, although this was not a statistically significant difference. Native mature phase canopy species that established in canopy gaps included *Myrsine australis*, *Podocarpus totara*, *Vitex lucens* and *Melicytus ramiflorus* at Wenderholm and *Myrsine australis* and *Coprosma robusta* at Shakespear. Native mature phase canopy species that established under closed canopy included *Sophora tetraptera* at Wenderholm and *Myrsine australis* and *Cyathea dealbata* at Shakespear. At both regional parks, there were less weed species in the closed canopy plots, suggesting that the *Leptospermum scoparium* and *Kunzea ericoides* canopy is effective at suppressing weeds.

The low number of native seedlings in the canopy gaps at both regional parks may be the result of competition with forb species or the short timeframe over which the plots were monitored for this study. It is also possible that the lack of native seedlings found in this study was due to the young age of the *Leptospermum scoparium* and *Kunzea ericoides* plantings, 11-13 years for Shakespear and 8-10 years for Wenderholm. Esler and Astridge (1974) found that the canopy of mature *Leptospermum scoparium* and *Kunzea ericoides* (over 15 years old) is usually thinning out and the canopy gaps increasing. At this point it is possible for relatively shade-tolerant seedlings to establish, but not for pasture species (Williams & Buxton, 1989). It is possible that a combination of continued pest control and opening the canopy may help native seedlings to germinate within the *Leptospermum scoparium* and *Kunzea ericoides* canopy in the regional parks. However, monitoring and weed control would need to continue long-term to ensure that undesirable plant species such as *Araujia sericifera* or *Solanum mauritianum* do not become established. It is therefore recommend a future study which creates artificial light gaps within the *Leptospermum scoparium* and *Kunzea ericoides* canopy be repeated to ascertain what species establish within the

Leptospermum scoparium and *Kunzea ericoides* canopy over a long-term and determine what the possible successional pathway of *Leptospermum scoparium* and *Kunzea ericoides* in the regional parks could be.

4.4 Future Forest Composition

More seedling species established within the exclosure plots at Wenderholm than at Shakespear. Seedling densities and heights also increased more within the exclosure plots at Wenderholm compared to Shakespear. There are three possible reasons for these results: firstly, there is mature forest close to both of the study sites. However, at Wenderholm, it is on the boundary of the study sites and at Shakespear it is approximately 85m from the study sites. Therefore, there may be an increased chance of seed dispersal within the *Leptospermum scoparium* and *Kunzea ericoides* plantings at Wenderholm; secondly, it was observed, by the author, that at Wenderholm there were more species in the canopy within the *Leptospermum scoparium* and *Kunzea ericoides*, which may have attracted more birds and facilitated seed dispersal. Thirdly, although *Oryctolagus cuniculus* were observed at both regional parks on two occasions, pest numbers are considered to be low due to annual pest control. However, on-going monitoring on both regional parks found that at Wenderholm, rat populations are very low compared to *Oryctolagus cuniculus* and *Trichosurus vulpecula* populations and at Shakespear, rat populations are higher than *Oryctolagus cuniculus* and *Trichosurus vulpecula* populations.

The results from this study suggest that despite ongoing predator control *Trichosurus vulpecula*, *Oryctolagus cuniculus* and/or *Rattus rattus* are having an adverse impact on the seedlings under the *Leptospermum scoparium* and *Kunzea ericoides* in the regional parks. Nevertheless, creating gaps in the *Leptospermum scoparium* and *Kunzea ericoides* canopy may increase the chances of native seedling establishment. The results also suggest that the distance to mature forest may have influenced seed dispersal within the *Leptospermum scoparium* and *Kunzea ericoides* canopy. Reay and Norton (1999a) found that restoration plantings at Port Hills were successful despite the use of a species (*Olearia paniculata*) that did not occur naturally at the study site. The revegetation project was a success due to the close proximity of seed sources in the surrounding forest remnants and a high level of seed dispersal by birds into the planted areas.

Previous studies have found that there are alternatives to using *Leptospermum scoparium* and *Kunzea ericoides* as a ‘nursery crop’. Williams and Karl (2002) found that native seed species richness was similar under gorse (*Ulex europaeus*) to that under *Kunzea ericoides* plantings. Seedling emergence and survival under *Ulex europaeus* was believed to be due to openings in the canopy, and the low density of the *Oryctolagus cuniculus* at the sites (Williams & Karl, 2002). However, *Ulex europaeus* could be considered an inappropriate plant species to be introducing into a regional park as it could displace native ecosystems and a more attractive alternative could be flax (*Phormium tenax*) for appropriate locations. Reay and Norton (1999b) found that *Phormium tenax* could provide a suitable site for the regeneration of woody plant species in grass pastures. Reay and Norton (1999b) also found that the distance of individual *Phormium tenax* clumps from the remnant forest did not appear to influence regeneration within clumps, but that *Phormium tenax* clump size did. This was possibly due to birds using the *Phormium tenax* clumps for perch sites as well as a food source (Reay and Norton 1999b), and would suggest that large plantings of flax could be used in the regional parks for areas that are some distance from mature forest.

4.5 Future Management Strategies

Determining whether a restoration project is a success is not just a case of increasing the species richness and abundance, it may be more important to determine if the ecosystem is self-sustaining. According to some researchers, ecosystem function as well as structure must be restored if restoration is to be regarded as successful (Reay & Norton 1999a). As previously discussed, in chapter one, it is impossible to reinstate an environment back to its pristine state due to irreversible changes such as the loss of species through extinction, the introduction of animal and plant pests and pollutants, and because of the non-static nature of natural ecosystems (Atkinson, 1990; Simberloff, 1990b). Therefore, it might be more appropriate for regional parks to have a broader goal, to determine if restoration is a success.

The Auckland Regional Council has a Regional Parks Management Plan, which sets out how the regional parks are to be managed over a five-year period. This plan sets the context for the future use of the parks and the conservation of natural and cultural resources (Auckland Regional Council, 2003a). To achieve the Council’s objectives under the management plan, where practical, a minimum impact approach is used, ranging from doing nothing (natural regeneration) to fencing, weed control and/or scattering seed (assisted regeneration) and planting early pioneer species propagated from nearby seed sources (Auckland Regional Council, 2003a). Up to 80,000 native trees are planted in the regional parks each year using sites which are chosen by various criteria: such as erosion control, riparian management, wetland restoration, creating linkages between existing fragments and restoring forest types that are rare or which have been lost from the region. Planting plans cover up to a ten-year period in advance of planting (Auckland Regional Council, 1996).

Planting methods in the Auckland Regional Council parks have changed in the past due to the different interpretation of the Council's Management Plan by different individuals and political pressure. For example, ten years ago, the common opinion was to use pioneer species such as *Leptospermum scoparium* and *Kunzea ericoides* and allow nature to do the rest. The Botanical Society with the Auckland Regional Council developed a planting programme which changed the species planted from mixed species to *Leptospermum scoparium* and *Kunzea ericoides* to create a starting point for natural succession. Recent planting in regional parks has changed to incorporate *Cordyline australis*, *Coprosma robusta*, *Dacrycarpus dacrydioides* and *Metrosideros excelsa* species along with *Leptospermum scoparium* and *Kunzea ericoides* (pers comms Tim Lovegrove, Natural Heritage Scientists, Auckland Regional Council, 2006).

Depending on what the restoration objective is, future planting methods in regional parks could vary from: planting *Leptospermum scoparium* and *Kunzea ericoides* (to cover a large area and to provide ecological corridors), through to a more managed approach of establishing fast growing seral species and interplanting mature phase species at a later stage. An alternative, which would still achieve the Management Plan objectives, could be to divide the restoration area into spatial modules and have several restoration designs across the area, for example, *Leptospermum scoparium* and *Kunzea ericoide* on slopes, *Cordyline australis*, *Phormium tenax*, *Laurelia novae-zelandiae* and *Dacrycarpus dacrydioides* in wet areas, and *Vitex lucens*, *Corynocarpus laevigatus*, *Melicytus ramiflorus* and *Beilschmiedia tarairi* in valleys (pers. comms. Dr Len Gillman, Senior Lecturer, Auckland University of Technology, 2006). This would imitate the natural environment and also increase resistance following disturbance.

Whilst there is a need to have clearly defined and measurable goals to determine success, regular monitoring must also occur to determine if the project has been a success. A suggestion is that at the time of planting each block should be labelled, the boundaries identified and the number of each species planted recorded to enable survival rate to be assessed. Approximate heights and spacing should also be recorded to allow growth rates to be measured at a future date (pers. comms. Dr Len Gillman, Senior Lecturer, Auckland University of Technology, 2006). The Auckland Regional Council needs to have strategic documents which have clearly defined objectives for each planting plan, and outlines what the objectives are and the timeframe in which it is to be achieved, as well as an on-going monitoring programme to ensure that these goals and objectives are being met at all stages.

Chapter Five: Conclusion

5.1 Introduction

Protected areas are vital for ensuring the protection of indigenous biodiversity. Despite the extensive literature detailing the impacts of *Trichosurus vulpecula* on mature species there is little information on the effect that *Trichosurus vulpecula* or *Rattus rattus* have on seedlings and hence forest regeneration. Consequently, understanding the impacts of animal pests and seedling survival is crucial in developing long-term strategies for restoration projects in regional parks.

5.2 Seedling Recruitment under *Leptospermum scoparium* and *Kunzea ericoides* Canopy

At both regional parks, excluding *Rattus rattus* in addition to *Trichosurus vulpecula* and *Oryctolagus cuniculus* did not significantly increase seedling densities compared with excluding only *Trichosurus vulpecula* and *Oryctolagus cuniculus*. At Wenderholm, seedling densities and seedlings heights increased within the exclosures compared to the control plots, which suggests that *Trichosurus vulpecula* and/or *Oryctolagus cuniculus* were having an adverse effect on seedling survival under the *Leptospermum scoparium* and *Kunzea ericoides* canopy. At Shakespear, seedling densities increased within the exclosures compared to the control plots, and seedling heights only increased within the PRabRat exclosure plots suggesting that *Rattus rattus* were having an adverse effect on seedling survival under the *Leptospermum scoparium* and *Kunzea ericoides* canopy. The on-going pest control and monitoring on the regional parks supports these results.

At Wenderholm, native species in the seed banks included *Coprosma robusta*, *Cordyline australis* and *Pteridium esculentum*. At Shakespear, native species in the seed banks included *Cordyline australis*, *Metrosideros excelsa* and *Pteridium esculentum* suggesting that few species are being brought into the planted areas by birds, or the native seeds, which may have been present in the soil, were dead at the time of collection. The results also suggest that the *Leptospermum scoparium* and *Kunzea ericoides* canopy is efficiently suppressing weeds and that actively managing the *Leptospermum scoparium* and *Kunzea ericoides* plantings by creating canopy gaps may increase the chances of native seedling establishment. It may also be possible to alter the semi-natural successional process within these fragments by inter-planting in artificially created gaps with desired plant species such as *Cordyline australis*, *Coprosma robusta*, *Dacrycarpus dacrydioides* and/or *Metrosideros excelsa* although this might not be the most cost-efficient alternative.

Plantings in regional parks should meet the Council's objectives under the Parks Management Plan, which include: The restoration and enhancement of habitats and ecosystems with high ecological values; the conservation of regionally under-represented or threatened ecosystems; the re-introduction of indigenous flora and fauna and the provision of ecological corridors for wildlife. Plantings could be catered to the area which is being planted. For example, *Cordyline australis*, *Phormium tenax*, *Laurelia novae-zelandiae* and *Dacrycarpus dacrydioides* in wet areas, canopy species such as *Vitex lucens*, *Corynocarpus laevigatus*, *Melicytus ramiflorus* and *Beilschmiedia tarairi* in valleys and for ecological corridors and ridges with *Leptospermum scoparium* and *Kunzea ericoides*.

5.3 Implications for Auckland Regional Council Management

The international and national consensus for ecological restoration is that goals can vary from project to project but often include establishing: weed suppression, canopy closure, an increase in native species and a reduction in animal and plant pests. However, there is a need to set clear, achievable and measurable goals in order to subsequently assess the success of the revegetation project.

As a result of the lack of detailed ecological policies, the opinions of individuals have been able to result in ad hoc changes to restoration approaches. Whilst there is a Management Plan that outlines objectives and goals for the future, these objectives within the Management Plans or Strategic Documents are interpreted by Council officers who then implement them in various ways. Therefore, unless strategic documents have clearly defined actions that state how the goals are to be achieved it is very easy for different individuals to have differing interpretations of how to achieve those goals. The Council's strategic documents must not only define what the goals are but they must also outline how these goals are to be achieved.

References

Allen, E. B., Covington, W. W., & Falk, D. A. (1997). Developing the conceptual basis for restoration ecology. *Restoration Ecology, 5*(4), 275-276.

Allen, J. (1996). *Biospheric theory and report on overall Biosphere 2 design and performance during mission one (1991-1993).* Paper presented at the Fourth International Conference on "Closed Ecological Systems: Biospherics and Life Support", Linnean Society of London.

Allen, R., Lee, W. G., & Rance, B. (1994). Regeneration in indigenous forest after extermination of Norway rats, Breaksea Island, New Zealand. *New Zealand Journal of Botany*(32), 429-439.

Allen, R., Partridge, T., Lee, W., & Efford, M. (1992). Ecology of *Kunzea ericoides* (A. Rich) J.Thompson (kanuka) in east Otago, New Zealand. *New Zealand Journal of Botany, 30*, 135-149.

Atkinson, I. (1992). *Effects of possum on the vegetation of Kapiti Island and changes following possum eradication.* (No. Contract Report 92/95). Wellington: Department of Scientific and Industrial Research, Land Resources, Department of Science and Industrial Research.

Atkinson, I. A. E. (1990). Ecological restoration on islands: Prerequisites for success. In D. R. Towns, C. Daugherty & I. Atkinson (Eds.), *Ecological restoration of New Zealand islands*. Wellington: Conservation Sciences Publication No2. Department of Conservation.

Atkinson, I. A. E. (1994). Ecological measurements for conserving terrestrial biodiversity: A New Zealand perspective. In P. Forey, C. Humphries & R. Vane-Wright (Eds.), *Systematics and conservation evaluation. Systematics Special Volume* (Vol. 50, pp. 63-79). Oxford: Clarendon.

Atkinson, I. A. E. (2001). Introduced mammals and models for restoration. *Biol. Conserv., 99*, 81-96.

Auckland Regional Council. (1996). *Parks Operation Manual*. Auckland: Auckland Regional Council.

Auckland Regional Council. (2003a). *Regional Parks Management Plan: Volume 1 Overview and Strategic Development*. Auckland: Auckland Regional Council.

Auckland Regional Council. (2003b). *Regional Parks Management Plan: Volume 2: Resource inventory*. Auckland: Auckland Regional Council.

Bellingham, M. (1990). *Public involvement in island restoration*. Wellington: Department of Conservation, Science for Conservation Publication No. 2.

Bellingham, P. J., Stewart, G. H., & Allen, R. B. (1999). Tree species richness and turnover throughout New Zealand forests. *Journal of Vegetation Science, 10*, 825-832.

Burrows, C. (1995). Germination behaviour of the seeds of the New Zealand species *Aristotelia serrata*, *Coprosma robusta*, *Cordyline australis*, *Myrtus obcordata* and *Schefflera digitata*. *New Zealand Journal of Botany, 33*, 257-264.

Cairns, J. J. (1993). Ecological Restoration: replenishing our national and global ecological capital. In D. A. Saunders, R. J. Hobbs & E. P. R (Eds.), *Nature conservation 3: Reconstruction of fragmentated ecosystems: global and regional perspectives*. Virginia USA: Surrey Beatty & Sons.

Cairns, J. J. (1999). Balancing ecological destruction: the only hope for sustainable use of the planet. *Aquatic Ecosystem Health and Management, 2*, 91-95.

Cairns, J. J. (2000). Setting ecological restoration goals for technical feasibility and scientific validity. *Ecological Engineering, 15*, 171-180.

Campbell, D. J. (2002). *Changes in numbers of woody plant seedlings on Kapiti Island after rat eradication*. Wellington: Department of Conservation.

Campbell, D. J., & Atkinson, I. A. E. (1999). Effects of kiore (*Rattus exulans*, Peale) on recruitment of indigenous coastal trees on northern offshore islands of New Zealand. *Journal of the Royal Society of New Zealand, 29*(4), 265-290.

Campbell, D. J., & Atkinson, I. A. E. (2002). Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. *Biol. Conserv., 107*, 19-25.

Campbell, D. J., Moller, H., Ramsey, G. W., & Watt, J. C. (1984). Observation on foods of kiore (*Rattus exulans*) found in husking stations on northern offshore islands of New Zealand. *New Zealand Journal of Ecology, 7*, 131-138.

Christensen, N., Bartuska, A., Brown, J., Carpenter, S., D'Antonio, C., Francis, R., et al. (1994). The Report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management. *Ecological Applications, 6*, 665-691.

Clout, M. N., & Saunders, A. J. (1995). Conservation and ecological restoration in New Zealand. *Pacific Conservation Biology, 2*, 91-98.

Coleman, J. D., Green, W. Q., & Polson, J. G. (1985). Diet of brushtail possums over a pasture-alpine gradient in Westland, New Zealand. *New Zealand Journal of Ecology, 8*, 21-35.

Cortner, H., Shannon, M., Wallace, M., Burke, S., & Moote, M. (1994). *Institutional barriers and incentives for ecosystem management: a problem analysis*. Tuson, Arizona, USA: Water Resources Centre, University of Arizona.

Craig, J. (1990). Potential for ecological restoration of islands for indigenous fauna and flora. In D. R. Towns, C. Daugherty & I. Atkinson (Eds.), *Ecological Restoration of New Zealand Islands*. Wellington: Conservation Sciences Publication No. 2 Department of Conservation.

Craig, J., Cowan, P., Dingwall, P., & Innes, J. (1984). *Rodents of New Zealand*: WRLG Research Review.

Daily, G. C. (1993). Social constraints on restoration ecology. In D. A. Saunders, R. J. Hobbs & E. P. R (Eds.), *Nature conservation 3: Reconstruction of fragmentated ecosystems: global and regional perspectives*. Virginia USA: Surrey Beatty & Sons.

Davies-Colley, R. J., Payne, G. W., & Van Elswijk, M. V. (2000). Microclimate gradients across a forest edge. *New Zealand Journal of Ecology, 24*(2), 111-121.

Department of Conservation. (2000). *New Zealand as ecosystems: The ecosystem concept as a tool for environmental management and conservation*. Wellington: Department of Conservation.

Edwards, G. R., & Crawley, M. J. (1999). Herbivores, seed banks and seed recruitment in mesic grasslands. *Journal of Ecology, 87*, 423-435.

Ehrenfeld, J. G. (2000). Defining the limits of restoration: the need for realistic goals. *Restoration Ecology, 8*(1), 2-9.

Ehrenfeld, J. G., & Toth, L. A. (1997). Restoration ecology and ecosystem perspective. *Restoration Ecology, 5*(4), 307-317.

Enright, N., & Cameron, E. K. (1988). The soil seed bank of a kauri (Agathis australis) forest remnant near Auckland, New Zealand. *New Zealand Journal of Botany, 26*, 223-236.

Esler, A., & Astridge, S. J. (1974). Tea tree (*Leptospermum*) communities of the Waitakere Range, Auckland, New Zealand. *New Zealand Journal of Botany, 12*, 485-501.

Fahrig, L., & Merriam, G. (1994). Conservation of fragmented populations. *Conservation Biology, 8*(1), 50-59.

Fitzgerald, B. M., & Gibb, J. A. (2001). Introduced mammals in a New Zealand forest: long-term research in the Orongorongo Valley. *Biol. Conserv., 99*, 97-108.

Geist, C., & Galatowitsch, S. M. (1999). Reciprocal model for meeting ecological and human needs in restoration projects. *Conservation Biology, 13*(5), 970-979.

Gillman, L. N. (2002). *Microdisturbance to forest seedlings in New Zealand due to litterfall and animals.* University of Auckland, Auckland.

Gillman, L. N., & Ogden, J. (2003). Seedling mortality and damage due to non-trophic animal interactions in a northern New Zealand forest. *Austral Ecology, 28*, 48-52.

Gillman, L. N., Wright, S. D., & Ogden, J. (2002). Use of artificial seedlings to estimate damage of forest seedlings due to litterfall and animals. *Journal of Vegetation Science, 13*(635-640).

Gillman, L. N., Wright, S. D., & Ogden, J. (2003). Response of forest tree seedlings to simulated litterfall damage. *Plant Ecology, 169*, 53-60.

Grumbine, R. E. (1991). Cooperation or conflict? Interagency relationships and the future of biodiversity for US parks and forests. *Environmental Management, 15*(1), 27-37.

Grumbine, R. E. (1994). What is ecosystem management? *Conservation Biology, 8*(1), 27-38.

Hartley, P. (1997). *Conservation Strategies for New Zealand*. Wellington: New Zealand Business Roundtable.

Higgs, E. S. (1997). What is Good Ecological Restoration? *Conservation Biology, 11*(2), 338-348.

Hobbs, R. J. (1993). Can revegetation assist in the conservation of biodiversity in agricultural areas? *Pacific Conservation Biology, 1*, 29-38.

Hobbs, R. J., & Harris, J. A. (2001). Restoration ecology: repairing the earth's ecosystems in the new millennium. *Restoration Ecology, 9*(2), 239-246.

Hockings, M., & Carter, B. (1998). An integrated model of public contact planning for conservation management. *Environmental Management, 22*(5), 643-654.

Holl, K. D., & Howarth, R. B. (2000). Paying for restoration. *Restoration Ecology, 8*(3), 260-267.

Hughey, K., F D, Cullen, R., & Moran, E. (2003). Integrating Economics into priority settings and evaluation in conservation management. *Conservation Biology, 17*(1), 93-103.

Karr, J. (1990). Biological integrity and the goal of environmental legislation: lessons for conservation biology. *Conservation Biology*(4), 244-250.

Keiter, R. B. (1998). Ecosystems and the law: Towards an integrated approach. *Ecological Applications, 8*(2), 332-341.

Kelly, C. L., Pickering, C. M., & Buckley, R. C. (2003). Impact of tourism on threatened plant taxa and communities in Australia. *Ecological Management and Restoration, 4*(1), 37-44.

Langley, A. (1991). Formal analysis and strategic decision making. *Omega, 19*(2-3), 79-99.

Lee, W. G., Allen, R. B., & Johnson, P. N. (1986). Succession and dynamics of gorse (*Ulex europaeus L.*) communities in the Dunedin Ecological District South Island, New Zealand. *New Zealand Journal of Botany, 24*, 279-292.

Lindenmayer, D. B., Margules, C. R., & Botkin, D. B. (2000). Indicators of biodiversity for ecologically sustainable forest management. *Conservation Biology, 14*(4), 941-950.

Llewellyn, S., & Tappin, E. (2003). Strategy in the Public Sector: Management in the Wilderness. *Journal of Management Studies, 40*(4), 955-981.

Marzluff, J. M., & Ewing, K. (2001). Restoration of fragmentated landscapes for the conservation of birds: A general framework and specific recommendations for urbanizing landscapes. *Restoration Ecology, 9*(3), 280-292.

May, S. (1999). *Towards re-integration: developing management strategies for Shakespear Regional Park.* Unpublished MSc, University of Auckland, Auckland.

McArthur, C., Goodwin, A., & Turner, S. (2000). Preferences, selection and damage to seedlings under changing availability by two marsupial herbivores. *Forest Ecology and Management, 139*, 157-173.

Mendel, L. C., & Kirkpatrick, J. B. (2002). Historical Progress of Biodiversity Conservation in the protected-area system of Tasmania, Australia. *Conservation Biology, 16*(6), 1520-1529.

Meurk, C. D., & Swaffield, S. R. (2000). A landscape ecological framework for indigenous regeneration in rural New Zealand-Aotearoa. *Landscape and Urban Planning, 50*, 129-144.

Miller, C. J., & Miller, T. K. (1995). Population dynamics and diet of rodents on Rangitoto Island, New Zealand, including the effect of a 1080 poison operation. *New Zealand Journal of Ecology, 19*(1), 19-27.

Miller, J., & Hobbs, R. (2002). Conservation where people live and work. *Conservation Biology, 16*(2), 330-337.

Moote, M., Burke, S., Cortner, H., & Wallace, M. (1994). *Principles of ecosystem management*. Tucson, Arizona, USA: Water Resources Research Centre, University of Arizona.

Murcia, C. (1995). Edge effects in fragmented forest: implications for conservation. *Tree, 10*(2), 58-62.

Norton, D. A., & Miller, C. J. (2000). Some issues and options for the conservation of native biodiversity in rural New Zealand. *Ecological Management and Restoration, 1*(1), 26-34.

Nugent, G., Fraser, K., & Sweetapple, P. (1997). *Comparison of red deer and possum diets and impacts on podocarp-hardwood forest. Waihaha catchment, Pureora Conservation Park.* Wellington: Department of Conservation, Science for Conservation 50.

Nugent, G., Fraser, W., & Sweetapple, P. (2001). Top down or bottom up? Comparing the impacts of introduced arboreal possums and terrestrial remnants on native forests in New Zealand. *Biol. Conserv.*, 65-79.

Nugent, G., Sweetapple, P., Coleman, J., & Suisted, P. (2000). Possum feeding patterns: dietary tactics of a reluctant folivore. In T. L. Montague (Ed.), *The brushtail possum. biology, impact and management of an introduced marsupial* (pp. 292). Wellington: Manaaki Whenua Press.

Ogden, J. (1985). An introduction to plant demography with special reference to New Zealand trees. *New Zealand Journal of Botany, 23*, 751-772.

Ogden, J. (1995). The long-term conservation of forest diversity in New Zealand. *Pacific Conservation Biology, 2*, 77-90.

Orbell, G. E. (1968). *Soils of the Whangaparaoa Peninsula Reserves*: New Zealand Soil Bureau, DSIR.

Pastorok, R. A., MacDonald, A., Sampson, J. R., Wibler, P., Yozzo, D. J., & Titre, J. P. (1997). An ecological decision framework for environmental projects. *Ecological Engineering, 9*.

Payton, I. (2000). Damage to native forests. In T. L. Montague (Ed.), *The brushtail possum. Biology, impact and management of an introduced marsupial* (pp. 292). Wellington: Manaaki Whenua Press.

Pfadenhauer, J. (2001). Some remarks of the socio-cultural background of restoration ecology. *Restoration Ecology, 9*(2), 220-229.

Pressey, R. (1994). Ad Hoc reservations: forward or backward steps in developing representative reserve systems? *Conservation Biology, 8*, 1037-1046.

Rahman, A., James, T. k., & Grbavac, N. I. K. (2001). Potential of weed seedbanks for managing weeds: a review of recent New Zealand weeds. *Weed Biology and Management, 1*, 89-95.

Rapport, D., Costansa, R., & McMichael, A. (1998). Assessing ecosystem health. *Trends of Ecology and Evolution, 13*, 397-402.

Reay, S. D., & Norton, D. A. (1999a). Assessing the success of restoration plantings in a temperate New Zealand forest. *Restoration Ecology, 7*(3), 298-308.

Reay, S. D., & Norton, D. A. (1999b). *Phormium tenax*, an unusual nursery plant. *New Zealand Journal of Ecology, 23*(1), 81-85.

Recher, H. F. (1993). The loss of biodiversity and landscape restoration: conservation, management, survival. An Australian perspective. In D. A. Saunders, R. J. Hobbs & E. P. R (Eds.), *Nature conservation 3: Reconstruction of fragmentated ecosystems: global and regional perspectives*. Virginia USA: Surrey Beatty & Sons.

Resource Management Amendment Act 2005, Wellington, New Zealand. Published under the authority of the New Zealand Government, 2005

Saunders, D. A., Hobbs, R. J., & Margules, C. R. (1991). Biological consequences of ecosystem fragmentation: A review. *Conservation Biology, 5*(1), 18-32.

Sem, G., & Enright, N. (1995). The soil seed bank in Agathis australis (D.Don) Lindl. (kauri) forests of northern New Zealand. *New Zealand Journal of Botany, 33*, 221-235.

Sem, G., & Enright, N. (1996). The relationship between seed rain and the soil seed bank in a temperate rainforest stand near Auckland, New Zealand. *New Zealand Journal of Botany, 34*, 215-226.

Simberloff, D. (1990a). Community effects of biological introductions and their implications for restoration. In D. R. Towns, C. Daugherty & I. Atkinson (Eds.), *Ecological restoration of New Zealand islands*. Wellington: Conservation Sciences Publication No. 2 Department of Conservation.

Simberloff, D. (1990b). Reconstructing the ambiguous: can island ecosystems be restored? In D. R. Towns, C. Daugherty & I. Atkinson (Eds.), *Ecological restoration of New Zealand islands*. Wellington: Conservation Sciences Publication No.2 Department of Conservation.

Sutton, S. (2004). Outdoor recreation planning frameworks: an overview of best practices and comparison with Department of Conservation (New Zealand) planning process. In K. Smith & C. Schott (Eds.), *Proceedings of the New Zealand Tourism and Hospitality Research Conference* (pp. 407-423). 8-10 December, Wellington.

Taylor, R., Smith, I., Cochrane, P., Stephenson, B., & Gibbs, N. (1997). *State of New Zealand's Environment*. Wellington: Ministry for the Environment.

Towns, D. R., & Ballantine, W. J. (1993). Conservation and restoration of New Zealand island ecosystem. *Tree, 8*(12), 452-457.

Tucker, N. I. J., & Murphy, T. M. (1997). The effects of ecological rehabilitation on vegetation recruitment: some observations from the wet tropics of north Queensland. *Forest Ecology and Management, 99*, 133-152.

Veitch, C. R., & Bell, B. D. (1990). Eradication of introduced animals from the islands of New Zealand. In D. R. Towns & I. Atkinson (Eds.), *Ecological Restoration of New Zealand Islands. Conservation Series Publication No 2* (pp. 137-143). Wellington: Department of Conservation.

Walker, B. (1995). Conserving biological diversity through ecosystem resilience. *Conservation Biology, 9*(4), 747-752.

Williams, P., & Buxton, R. (1989). Response to reduce irradiance of 5 species of native and adventive shrub and tree seedlings of native and adventive shrub and tree seedlings from eastern Canterbury. *New Zealand Journal of Ecology*(12), 95-101.

Williams, P. A., & Karl, B. J. (2002). Birds and small mammals in kanuka (*Kunzea ericoides*) and gorse (*Ulex europaeus*) scrub and the resulting seeds rain and seedling dynamics. *New Zealand Journal of Ecology, 26*(1), 31-41.

Wilson, D. J., Lee, W. G., Webster, R. A., & Allen, R. B. (2003). Effects of possums and rats on seedling establishment at two forest sites in New Zealand. *New Zealand Journal of Ecology*, *27*(2), 147-155.

Wilson, H. D. (1994). Regeneration of native forest on Hinewai Reserve, Banks Peninsula. *New Zealand Journal of Botany, 32*, 373–383.

Young, A., & Mitchell, N. (1994). Microclimate and vegetation edge effects in a fragmented podocarp-broadleaf forest in NZ. *Biological Conservation*, *67*, 63-72.