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Vegetation recovery and management of kahikatea (*Dacrycarpus dacrydioides*)-dominated forest remnants in the Waikato Region

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Abstract

The principle aim of this study was to determine whether fencing alone is a sufficient management tool for facilitating the recovery and persistence of indigenous flora in kahikatea-dominated forest patches in the Waikato region. The floral composition of twenty-six kahikatea-dominated forest patches of varied fencing time, management regime and proximity to an urban area (Hamilton City) were sampled using a modified RECCE method in 10x10m quadrats between October 2007 and February 2008. Where woody weed species were present within a forest patch, their diameter at breast height (d.b.h) and reproductive status was noted (presence/absence of flowers and/or fruit). The results of the study demonstrate that, while fencing of a patch and time for native vegetation recovery are important factors in promoting native floral species recovery and ecosystem composition, the combination of patch size, distance of a patch from a main road, and patch location were better predictors of the observed variation in native species cover than fencing time alone; particularly in the layers most affected by grazing. This study indicates that patches less than seven hectares in area, regardless of location, will require continued human intervention to ensure their persistence; and patches in urban areas, irrespective of size, may never become self-sustaining.

Furthermore, the results indicate that medium to low levels of management are the most effective in promoting native flora species recovery and reducing exotic species impacts. Fifteen to twenty years of fencing represents an important stage in the trajectory of a forest fragment where exotic species cover drops below 5%, and native species recruitment is steadily rising. However, the trajectory of floristic change will be different for each patch depending on the length of time since fragmentation, the length of time it has been grazed, how far it is from native seed sources and its surrounding landscape use.

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CHAPTER 1

1.1 Introduction

Biodiversity loss has been identified as one of the leading conservation issues facing the world with the major factors influencing this process being climate change, expanding trade networks and habitat fragmentation (Ehrlich and Ehrlich, 1981; Fischer and Lindenmayer, 2007; Jay, 2005; Mooney *et al.*, 1995). This scenario is no less true in New Zealand where coastal and lowland ecosystems in particular have historically been, and continue to be, pressured by an expanding population and consequent resource demands (Ministry for the Environment, 2007). Seventy three percent of habitable land in New Zealand has been disturbed or modified in some way by humans and 69% is classified as human dominated (Hannah *et al.*, 1994). In the Waikato, lowland forest ecosystems are particularly vulnerable with approximately 11% of the original extent remaining (Burns and Smale, 2002). In particular, kahikatea (*Dacrycarpus dacrydioides*)-dominated patches now represent only 10% of all forest patches in the Waikato region where once they occupied hundreds of thousands of hectares (Leathwick *et al.*, 1995). The kahikatea forest that does remain is often highly modified due to agricultural pressures and rarely contains any old-growth trees spared from logging or clearance (Burns and Smale, 2002).

Within the last twenty years however, there has been a slow but steady move to protect and restore by fencing off and covenanting patches of forest that might not otherwise persist in the landscape. Many of these patches are isolated in a matrix of agricultural land, have been heavily grazed and may be cut off from native seed sources (Environment Waikato, 2007; Leathwick *et al.*, 2003). Consequently, it is not known whether they will be able to recover native biodiversity and ecosystem function without human interference.

1.2 Kahikatea (*Dacrycarpus dacrydioides*)

Kahikatea/white pine (*Dacrycarpus dacrydioides*) (A. Rich.) Laubenf is a dioeceous podocarp with small solitary terminal male cones and small solitary terminal female cones situated on a swollen peduncle. The dispersal structure is an ovoid black nut approximately 4mm long on a succulent red peduncle (Poole

and Adams, 1994). Mature trees may attain heights of up to 60 metres and live for around 600 years with a trunk up to 2 metres diameter often with large buttressed roots, particularly on swampy substrates (Eagle, 2006; Poole and Adams, 1994). On alluvial plains and lowland areas, kahikatea trees may initially form dense mono-specific stands where forest has been subjected to large-scale disturbance by way of flooding and/or wind-throw (Smale, 1984; Whaley *et al.*, 1997) but as the substrate dries and trees mature, the forest may change to dense mixed conifer forest or conifer-broadleaf forest depending on location, topography and substrate (Champion, 1988).

Although kahikatea is still common in both the North and South Islands of New Zealand, its range has been greatly reduced through clearance of lowland alluvial plains for agriculture and forestry. In the North Island in particular, no large extensive tracts of kahikatea-dominated forest remain and in the South Island, only south Westland contains large areas of relatively undisturbed kahikatea forest (New Zealand Plant Conservation Network, 2005; Wardle, 1974).

1.3 Kahikatea-dominated patches

Kahikatea-dominated forest patches are a conspicuous component of the Waikato landscape (Burns *et al.*, 2000) and were once extensive on the wet alluvial lowlands of the North and South Islands (Wardle, 1974). However, through much of the late 19th and early 20th century, forest on alluvial plains was cleared to make way for pastoral endeavours and to provide timber for a growing population (Leathwick *et al.*, 2001b). In the Waikato, although landscape scale clearance of forest occurred, some areas were not needed for production and were allowed to revert back to native vegetation (Jay, 2005). Often, kahikatea stands developed because these areas were still swampy despite extensive drain systems, and therefore kahikatea were more suited to such conditions than other dominant forest trees in the region (Burns *et al.*, 2000). As agriculture intensified with a growing population, many of these re-growth patches were cleared, at least in part and, though some patches remain, they are usually on sites marginal for pasture because of topography or because the land is sensitive to damage (Jay, 2005).

Consequently, in the Waikato, kahikatea-dominated forest now only occurs in fragments, very few of which contain old-growth vegetation (Burns *et al.*, 2000;

Burns and Smale, 2002; Burns *et al.*, 1999; Environment Waikato, 2007). Therefore the persistence of these patches is of vital importance to floristic and ecosystem biodiversity in the Waikato and the rest of the North Island.

1.3.1 Characteristics of kahikatea-dominated forest

Species commonly found in association with kahikatea on semi-swamp to swampy substrates include pukatea (*Laurelia novae-zelandiae*) and swamp maire (*Syzygium maire*) in the canopy, small-leaved shrubs and trees such as *Coprosma rotundifolia*, *Coprosma propinqua*, *Melicope simplex*, *Melicytus micranthus* and *Streblus heterophyllus* in the understorey and numerous ground ferns and sedges (Burns *et al.*, 1999). On drier substrates matai (*Prumnopitys taxifolia*), rewarewa (*Knightia excelsa*), rimu (*Dacrydium cupressinum*) and tawa (*Beilschmeidia tawa*) replace pukatea and swamp maire as emergents or co-dominants in the canopy, *Coprosma grandifolia*, *Hoheria sexstylosa*, hangehange (*Geniostoma ligustrifolium* subsp. *ligustrifolium*), mahoe (*Melicytus ramiflorus* subsp. *ramiflorus*) and pate (*Schefflera digitata*) become more prominent in the understorey and fewer filmy ferns and sedges are found on the ground (Burns *et al.*, 1999; Whaley *et al.*, 1997).

In the North Island, patches of kahikatea forest that have not been highly modified typically contain around 120 indigenous vascular species and may contain regionally and nationally significant species such as *Pittosporum obcordatum*, *Coprosma obconica*, *Myriophyllum robustum*, *Teucridium parvifolium* and *Fuchsia perscandens* (Burns *et al.*, 2000; Burns *et al.*, 1999; Smale, 1984). Modified patches on the other hand, usually only contain regionally common native species and a large suite of adventive species but may have similar vascular plant species richness scores to continuous indigenous forest elsewhere in New Zealand (Burns *et al.*, 2000). However, these species richness scores also include adventive species and composition is different than in unmodified fragments (Burns *et al.*, 1999).

1.3.2 Importance of kahikatea patches

Because it has been at least 125-150 years since fragmentation, many species may have become locally extinct in Waikato patches prior to any botanical study. However, because these patches are all that remain of a formerly widespread

forest type, their persistence is of vital importance for biodiversity and conservation values (Environment Waikato, 2007). With 70% of New Zealand's land surface in private ownership (Ministry for the Environment, 2007), most patches are likely to be found on private land. However, because they are on private land, comprehensive and informed management is unlikely and even in those protected (0.80% of New Zealand's land is in QEII trust covenants (Ministry for the Environment, 2007)), management standard adherence is not assured. Additionally, forest patches on farms can provide important ecosystem and farm management services such as reducing run-off, soil erosion and nutrient leaching, providing shade and shelter for stock and increasing organic carbon in the soil (Environment Waikato, 2007).

1.3.3 Future/threats to kahikatea patches

Kahikatea-dominated patches are often very small (50% of patches in the Waikato are less than 5 hectares in area) and are surrounded by agricultural land (Environment Waikato, 2007). Consequently, they may require active restoration in order to regain ecosystem functioning and become self-supporting; indeed, the smallest may never become self-supporting. The major threats to kahikatea-dominated forest persistence are altered hydrological regimes, competition from adventive species and continued grazing by livestock (Champion, 1988; Environment Waikato, 2007; Walsh, 1898). An altered hydrological regime results in a drier substrate which in turn provides more suitable conditions for shade-tolerant, drier substrate inclined species such as tawa and tītoki (Champion, 1988). Invasion by adventive species can restrict and may even out-compete native species regeneration resulting in altered floristic trajectories (Murphy *et al.*, 2008). In particular, woody weeds have the ability to change vegetative composition resulting in an adventive dominated ecosystem (Denslow and Hughes, 2004). Additionally, continued grazing by livestock removes all vegetation but mature trees, facilitates weed invasion and suppresses regeneration (Atkinson, 2001; Champion, 1988; Cranwell, 1939; Walsh, 1898).

Although active management is advisable, guidelines as to how much management is required in order to achieve the desired biodiversity and ecological outcomes are lacking. In the following chapters I will provide the basis for developing such guidelines first by providing an in-depth analysis of the condition

of native vegetation and the extent of adventive vegetation in twenty-six kahikatea forest patches with different management regimes, different locations relative to an urban centre and different fencing times (Chapter 2). Secondly I will examine the extent of spread of woody weed species in the study patches and ascertain the factors that are associated with their presence (Chapter 3). I will then use this analysis to assess management regime effectiveness, comment on the threats posed to patches in different localities and provide guidelines for future management (Chapter 4).

1.4 Thesis research objectives

This thesis focuses on the vegetative condition of kahikatea-dominated forest patches in the Hamilton basin released from grazing pressure along an urban-rural gradient using a 'space-for-time' substitution. The primary aim of this study was to address the question:

Will fencing alone enable the persistence of the indigenous flora and the condition of the vegetation of kahikatea-dominated remnants in the Waikato region?

Questions arising from this include, but are not limited to:
1) What level of management is required to reduce weed populations to acceptable levels, and encourage native flora species regeneration and recruitment?
2) Will the kahikatea forest patches ever be resilient enough to survive in the longer term?
3) What are the constraints that limit achievement of these outcomes?

1.5 Thesis structure
In the remainder of this introductory chapter I review, from the national and international literature, the main factors affecting kahikatea forest patches. These include fragmentation, surrounding landscape use, water table change, altered nutrient dynamics, grazing and browsing, edge effects, species homogenisation, weed invasion, loss of dispersers/pollinators and reduced ecosystem complexity.

Chapters two and three present the results of a vegetative survey undertaken at twenty-six kahikatea forest patches in the Waikato basin over the 2007-2008 summer. Chapter two specifically explores the effects of location and management effort on the condition of native floral biodiversity and vegetative cover in the layers of kahikatea forest most affected by grazing once grazing pressure has been removed, and addresses the following key questions:

1) Do patches located in urban environments have lower native species diversity and poorer vegetative condition than their peri-urban and rurally located counterparts?

2) Do patches with high management effort contain a greater number and cover of native species and a lower number and cover of adventive species than patches with other management regimes?

3) Are patches with no management effort of all fencing ages dominated by adventive species or have natives persisted?

4) Do patches closer to roads contain higher adventive species numbers and cover than patches further from roads?

Chapter three investigates the impacts of woody weed invasions in kahikatea forest patches to ascertain whether active management is required or whether native species will predominate given time. Specifically, the chapter addresses the following key questions:

1) Is woody weed density and richness related to location and fencing time?

2) Is intensive removal of woody weed species from a patch the best method for controlling woody weed populations or will populations undergo self-thinning?

3) Is there a relationship between native species richness and woody weed richness?

Finally, chapter four summarises the main effects from chapters two and three. Additionally, recommendations are given for further research and for future management of these iconic patches depending on their location along the urban-rural gradient.

Plate 1. The understorey at Piarere/Arnold's bush

1.6 Literature review

1.6.1 Forest Fragmentation

Globally, once extensive forest ecosystems are being fragmented as a result of mounting resource demand. The effects of fragmentation on forested ecosystems are many and varied depending on the degree of fragmentation, the range of original stressors, the spatial extent of species' populations before fragmentation and the composition and degree of modification of the surrounding matrix (Fischer and Lindenmayer, 2007; Hobbs and Yates, 2003). In most cases fragmentation results in the creation of multiple islands of remnant stands that are spatially isolated from the others, are many orders of magnitude smaller than the original habitat and are surrounded by a matrix that differs from the original (Fischer and Lindenmayer, 2007; Hobbs and Yates, 2003). This physical isolation and change in landscape structure often culminates in population size reduction which may then result in species loss and altered ecosystem composition, structure and function (Kupfer *et al.*, 2006; Laurance *et al.*, 2006). Ecosystems may be further altered due to reduced gene flow (Jump and Penuelas, 2006), altered environmental conditions (Bierregaard *et al.*, 1992; Kupfer *et al.*, 2006), increased abundance and incidence of adventive species (Kupfer *et al.*, 2006; Olden, 2006) and changed disturbance regimes (Kupfer *et al.*, 2006).

1.6.2 Surrounding landscape use

In addition to the removal of vegetation and disruption of ecosystem processes, fragmentation is often followed by dramatic changes in the physical structure of the surrounding land (Kupfer *et al.*, 2006). In many cases the surrounding landscape is modified for agricultural production or human settlement resulting in a variety of landscape processes quite different from the original (Hannah *et al.*, 1994). These changes can substantially alter disperser/pollinator behaviour, soil microbial processes, soil nutrient status, water table levels, and may result in alteration of the local microclimate (Hobbs and Yates, 2003; Kupfer *et al.*, 2006). For example, forest fragments in the Amazon surrounded by agriculture or pasture had higher rates of tree mortality and significantly different floristic trajectories than those surrounded by intact forest or secondary re-growth forest (Laurance *et al.*, 2006). Additionally, a gap of as little as 80m was found to be impenetrable to

some insects, mammals and understorey birds when the matrix had no connecting cover (Bierregaard *et al.*, 1992).

Pastoral landscapes in the Waikato basin are vegetatively homogenous, with few hedgerows that could act as reservoirs for native species. Where there are hedgerows they are often composed of invasive adventive shrub species such as Chinese privet (*Ligustrum sinense*), barberry (*Berberis glaucocarpa* and *B. darwinii*), hakea (*Hakea sericea* and *H. salicifolia*) and gorse (*Ulex europaeus*) (McQueen, 1993). Therefore, landscape connectivity to facilitate forest species' seed dispersal is low (Fischer and Lindenmayer, 2007). Conversely, the frequent disturbance and open nature of pastoral landscapes makes it prime habitat for generalist, light-demanding, rapidly-growing, hardy adventive species to spread and colonise thereby facilitating their dispersal to forest patches (Kupfer *et al.*, 2006).

By comparison, although urban areas may contain populations or individuals of forest species in gardens which may be reservoirs for genetic diversity (Roberts *et al.*, 2007), they also contain a large pool of adventive species with the potential to disrupt native ecosystems and out-compete native species (Sullivan *et al.*, 2005). For example, a New Zealand study found that settlement characteristics were much more significant than forest size in predicting exotic plant species number in a forest with the number of houses within 250m of a forest area explaining 66.8% of the variation in the number of exotic plant species in those forests (Sullivan *et al.*, 2005)

Urban environments are often stressed abiotically due to the concentration of anthropogenic influence in a small area. For example, forest fragments in urban areas in New York were found to have daily temperatures 2-3 degrees higher than rural fragments, elevated concentrations of heavy metals, higher soil hydrophobicity, lower leaf litter depth, mass and density, higher concentrations of earthworms and lower biologically available carbon than their rural counterparts (McDonnell *et al.*, 1997). These characteristics will likely result in different species assemblages favouring species with wide environmental tolerances and potentially reduced ecosystem functioning (McDonnell *et al.*, 1997).

1.6.2.1 Water table change

Kahikatea fragments in the Waikato are almost exclusively surrounded by a dairy dominated landscape where once they would have grown on swamp to semi-swamp substrates (Clarkson *et al.*, 2007). In order to produce pasture, the landscape needed to be drained (Crush and Wedderburn, 2002) resulting in substantially reduced water tables within any fragments that have grown or still remain on agricultural land (Environment Waikato, 2007). Consequently, the competitive advantage afforded to kahikatea and associated semi-swamp species is no longer present and species composition is changing to an assemblage that is more competitive on drier substrates (Champion, 1988; Environment Waikato, 2007).

For example, species assemblages in kahikatea-dominated forest with persistently wet soils contain semi-swamp species such as pukatea (*Laurelia novae-zelandiae*), swamp maire (*Syzygium maire*), swamp mahoe (*Melicytus micranthus*), *Hydrocotlye* species, *Leptopteris hymenophylloides* and numerous herbaceous species (Burns *et al.*, 1999). In contrast, highly modified Waikato patches with much drier soils typically contain tawa (*Beilschmiedia tawa*), titoki (*Alectryon excelsus*), mahoe (*Melicytus ramiflorus*), wheki (*Dicksonia squarrosa*), ponga (*Cyathea dealbata*), karamu (*Coprosma robusta*), mapou (*Myrsine australis*), *Diplazium australe*, thread fern (*Blechnum filiforme*), rarely contain herbaceous species (Burns *et al.*, 1999) and pukatea, tītoki, and mahoe become more prominent (Whaley *et al.*, 1997).

1.6.2.2 Changed nutrient dynamics

Agricultural-based landscape use often requires substantial fertiliser inputs for maximal crop growth which leads to eutrophication of water-ways that run through the landscape, changed nutrient cycling patterns and altered soil microbial processes (Flinn and Marks, 2007). The type of fertiliser used for dairying means that species (usually adventive) adapted to high soil nitrogen and phosphorous concentrations have a competitive advantage over those adapted to a less fertile substrate (Hobbs, 2001; Lunt *et al.*, 2007). The major soil types in the Waikato basin are classified as recent soils and consequently have low to very low inherent phosphorous concentrations (Leathwick *et al.*, 2003). Therefore, the influx of phosphorous from agricultural fertiliser use would have markedly changed the

conditions to which Waikato vegetation was adapted. For example, kahikatea-dominated forest elsewhere in the North Island with intact water table levels and low modification of surrounding land contain *Myriophyllum robustum,* a perennial aquatic herb that is in gradual decline (Burns *et al.*, 1999), threatened by wetland drainage, eutrophication of ponds and streams and competition from adventive species (NZPCN, 2005).

1.6.2.3 Grazing of fragments

Livestock grazing of secondary re-growth fragments was common practice in the early 20th century and many farmers continue this practice today (Jay, 2005). These fragments provide(d) shelter and shade for the stock and stabilisation of soil for the surrounding land (Jay, 2005). Grazing of the patches however, has potentially long-lasting effects such as indefinite removal of palatable species from the forest system (Miller, 2006), soil compaction and reduced leaf litter cover (Hobbs, 2001), removal of the understorey layer and removal of perennial herbaceous species (Hobbs, 2001; Lunt *et al.*, 2007). These changes in soil and vegetation structure and composition can lead to reduced water filtration rates (Hobbs, 2001), thereby compounding the problem of draining, reduced organic compound cycling and therefore loss of organic matter from the forest system, a chemically less-buffered system and altered plant species composition (Hobbs, 2001).

Spooner and Briggs (2008) found that exclusion of woodlands from grazing resulted in improved tree and native ground cover regeneration as well as improved soil conditions over as little as five years. However, some of the fragments with long histories of intensive grazing showed no improvement in either soil or vegetation condition when excluded from grazing and therefore may be so degraded that they are unable to regenerate without significant management input. Additional to community composition changes directly due to grazing, livestock grazing effectively increases the amount of edge by removal of the shrub and understorey layers and compaction of the soil (Hobbs, 2001). These effects alter the physical and chemical properties of soil and increase weed invasion much farther into grazed than un-grazed fragments due to frequent disturbance, removal of competition and transport of weedy propagules (Hobbs, 2001). For example, a study of grazed *Eucalyptus*-dominated fragments surrounded by

agriculture in south-western Australia found that soil nitrate levels in the first 75 metres of forest were within the same concentration range as the pasture outside the forest and the resistance of soil to penetration 80m into the forest was as high as that in grazed pasture (Hobbs, 2001).

1.6.3 Edge effects

As one moves from the edges of a forest to its interior, the vegetation composition and structure changes as a result of changes in the abiotic environment such as decreasing light levels, increasing moisture levels, decreased temperature fluctuations and decreasing wind levels (Fischer and Lindenmayer, 2007; Matlack, 1994). The edge of a forest is typically characterised by high light intensity, exposure and frequent disturbance, consequently, the edge of a forest is usually composed of pioneer species capable of swift regeneration after disturbance and tolerant of a wide range of environmental variables (Laurance *et al.*, 2006; Matlack, 1994)

Edge effects refer to the fact that fragmentation results in smaller area to perimeter ratios for each individual remnant and consequently the transition zone from edge to interior takes up a larger portion of the remnant than it would have as a large, continuous forest tract (Young and Mitchell, 1994). For example, a model based on fragmentation effects recorded in Amazonian rainforest found that heat flow at forest/pasture interfaces could penetrate 15 to 20 metres into a fragment depending on forest structure at the edge (Malcolm, 1998). Consequently, for fragments less than 1 hectare in area that are surrounded by pasture, there is no amelioration of temperature fluctuations (Young and Mitchell, 1994).

What constitutes edge habitat differs for different abiotic parameters (Fischer and Lindenmayer, 2007). Vapour-pressure deficits and increased photosynthetically active radiation (PAR) can be detected up to 40 metres into a fragment (Bierregaard *et al.*, 1992), lowered soil moisture content up to 20 metres (Bierregaard *et al.*, 1992) and wind disturbance within 100-200 metres of an edge (Laurance, 1991). This means that species not adapted to exposed conditions, are less likely to be found in fragments with a high proportion of edge habitat (Leathwick *et al.*, 2001a; Matlack, 1994). The species composition of the forest patches are therefore more likely to contain the same suite of species that are

adapted to more disturbance-prone environments resulting in loss of biodiversity and potential ecosystem process disruption (Laurance *et al.*, 2006).

The results are similar for New Zealand forests with vapour pressure deficits, air temperature and wind speed influences from pasture extending at least 40 metres in from a 50-year-old forest edge (Davies-Colley *et al.*, 2000). Edge plots also contained higher plant biomass, greater species richness, greater proportions of 'pioneer' species and greater tree mortality than interior plots (Davies-Colley *et al.*, 2000; Young and Mitchell, 1994). Therefore, any patch smaller than 80m diameter (<0.5 ha) will experience reduced regeneration of native species, increased invasion by weeds and a vegetation composition composed primarily of pioneer species resulting in a homogenised flora (Davies-Colley *et al.*, 2000). With respect to kahikatea patches, edges typically contain a high species richness and vegetative cover of adventive species, have high grass cover, high seedling density, low leaf litter cover and low tree basal area (Smale *et al.*, 2005; Whaley *et al.*, 1997).

1.6.4 Species composition/homogenisation of flora

Studies comparing the floristic composition of edge, fragment and interior plots with continuous tracts of forest have demonstrated increased recruitment of pioneer species in the first 10 metres of the fragment edge (Bierregaard *et al.*, 1992; Laurance *et al.*, 2006; Young and Mitchell, 1994), lower seedling establishment of slow-growing, old-growth taxa (Laurance *et al.*, 2006), increased recruitment of disturbance adapted species in edge plots (Laurance *et al.*, 2006), increased liane abundance and diversity at edges (Laurance *et al.*, 2001) and increased large tree mortality rates (Laurance *et al.*, 2000). Additionally, increased seed predation has been demonstrated at edge versus interior plots and fragmented versus continuous forest (Donoso *et al.*, 2004).

For recently fragmented forest growing on former agricultural fields, these effects may be more pronounced due to the potential loss of local or rare native species from the seed bank and species with weak dispersal mechanisms being unable to penetrate the surrounding agricultural matrix (Vellend *et al.*, 2007). This is supported by a recent study of European and American forests where, although plant species diversity in recent forests approached that found in ancient

woodlands (α-diversity), the variability in plant composition across recent forests was much lower than between ancient forests, which were highly variable in composition (β-diversity) (Vellend *et al.*, 2007). Reduced variability in forest structure and composition has the potential to result in follow-on effects down the food chain which will be particularly pronounced for specialists thereby resulting in a 'knock-down' effect to the entire community (Olden, 2006).

Not only does the homogenization of flora make regional differences less pronounced and threatens the uniqueness of ecosystems, but it may also have ecological and evolutionary consequences for ecosystems such as the creation of 'hybrid swarms' (Olden, 2006). These 'swarms' may genetically eradicate native taxa and disrupt local genotypes which may reduce ecosystem/population resilience to environmental change thereby altering evolutionary trajectories and weakening selection pressures (Olden, 2006).

1.6.6 Adventive species

Weed invasions have been identified as a serious threat to the preservation, conservation and continuation of New Zealand's biodiversity; threatening ecosystem processes (Ehrenfeld *et al.*, 2000), already threatened plants (Miller and Duncan, 2004; Ogle *et al.*, 2000) and suppressing regeneration of native species (Ogle *et al.*, 2000; Standish *et al.*, 2001). There are approximately 24 700 adventive vascular plants resident in New Zealand (Williams *et al.*, 2002), of which 2 390 are considered naturalised or causal (Howell and Sawyer, 2006) and 328 are considered environmental weeds (Howell, 2008). Additionally, woody species comprise just over half of the environmental weeds in New Zealand (Williams and West, 2000) and fourteen new plant naturalisations occur each year (Landcare Research, 1996).

The damaging effects that environmental weeds can have on native ecosystem function in New Zealand has been demonstrated in a study by Standish *et al.* (2001) on the impacts of *Tradescantia fluminensis* (wandering Jew/wandering willie) on native forest regeneration. This weed can regenerate vegetatively from very small fragments and quickly carpets forest floors thereby suppressing native seedling establishment by reducing the amount of light reaching the forest floor. Consequently, only large-seeded, shade tolerant natives such as karaka

(*Corynocarpus laevigatus*) and tawa (*Beilschmiedia tawa*) are able to germinate which could potentially alter ecosystem structure. Similarly, *Clematis vitalba* (Old-man's beard) is a vine that rapidly invades disturbed forest systems and results in loss of forest structure and function as well as suppressing regeneration of other species (Ogle *et al.*, 2000).

Additionally, many native species whose habitat preferences overlap with those of environmental weeds experience reduced seed germination due to the presence of adventive species (Merrett *et al.*, 2007). In particular, woody weeds have the capacity to change vegetative composition and suppress tree regeneration which may result in an ecosystem entirely dominated by a singular adventive species (Hobbs and Yates, 2003; Merriam and Feil, 2002).

1.6.7 Loss of seed dispersers/pollinators

Habitat fragmentation may result in the disruption of plant-pollinator interactions due to the distance between similar vegetation types. Agricultural intensification in particular is generally correlated with a decline in the abundance, diversity and services to crops provided by wild pollinators (Kremen *et al.*, 2007). However, the extent to which pollinator/disperser relationships are disrupted is dependent on the historical continuity of habitat, the availability of alternative food sources in the matrix and the pollination mechanism involved (Kremen *et al.*, 2007). In Amazonian forest patches that were historically part of extremely large tracts of continuous forest, a break of only 80 metres is a strong barrier to movement of some insects and ground-dwelling bird species (Bierregaard *et al.*, 1992). In contrast, no evidence of reduced reproductive output or success has been found in a tropical dry-forest tree pollinated by bats with tree population distances exceeding 10 kilometres (Herrerias-Diego *et al.*, 2006).

The effects of fragmentation on pollination are not limited to animal-dependent pollination; recent (<600 years) bottlenecks, decreased genetic diversity and elevated levels of inbreeding and population divergence have been found even in fragmented wind-pollinated tree populations separated from each other and from continuous forest by only 12 kilometres (Jump and Penuelas, 2006). Although wind can potentially carry seed or pollen large distances, the matrix surrounding the remnant affects the continuity of wind gusts and their direction (Kremen *et al.*,

2007). Lowland Waikato forest patches have an average inter-patch distance of 750 metres (min. 96.7 metres, max. 17, 038 metres) (Leathwick *et al.*, 2001b).

If pollination and/or dispersal mechanisms are disrupted due to fragmentation, remnant habitats may become non-viable as genetic diversity decreases from inbreeding. This will result in reduced seed set, reduced seed rain and therefore reduced regeneration (Kremen *et al.*, 2007). In Amazonian forest fragments, a decrease in sub-canopy species that rely on animal pollination and dispersal has been observed after only twenty years of isolation (Laurance *et al.*, 2006). In New Zealand, although pollinator absence and specialisation has been hypothesised as a potential cause of reduced seed set in a wide-spread native forest shrub (*Alseuosmia macrophylla*), and ten other shrub species have been identified as moderately at risk of pollination limitation, other factors such as competition from adventive species are seen as greater threats to population viability and persistence (Merrett *et al.*, 2007). In many kahikatea-dominated forest patches, most native bird species are either rare or absent (Leathwick *et al.*, 2003) and although introduced species may disperse some species, it is possible that a few, especially those with fruits larger than 10mm diameter, may be dispersal limited (Burrows, 1994; Williams and Karl, 1996).

1.6.8 Summary

At an ecosystem level, all of the previously mentioned effects of fragmentation combine to reduce ecosystem complexity and native biodiversity, which may result in reduced ecosystem stability (McCann, 2000) and therefore threaten ecosystem persistence. This is not only problematic for the ecosystems themselves but also for the ecosystem services they provide. Grazed kahikatea fragments on Waikato farms contribute little to reducing soil erosion and may be traps for livestock making mustering difficult and resulting in injuries to cattle (Environment Waikato, 2007). However, by fencing a fragment, thereby allowing regeneration of native species, farm managers have observed reduced erosion, improved water quality, increased shelter for livestock as well as improved visual amenity values (Environment Waikato, 2007).

Consequently, the health and preservation of forest patches in agricultural landscapes is not only important for native biodiversity but also for landscape

health and farm productivity. The following chapters will explore the effects of location and management effort on the condition of native floral biodiversity and vegetative cover in kahikatea forest in the Waikato basin (Chapter 2); the threat that woody weeds pose to kahikatea forest patches and how they can be managed (Chapter 3); and finally, a summary of the main effects from chapters two and three with recommendations for further research and for future management of kahikatea forest patches.

Plate 2. A large kahikatea at Whewell's bush

CHAPTER 2 –

Location and management matter: vegetative condition of kahikatea (*Dacrycarpus dacrydioides*)-dominated forest fragments in the Waikato basin

2.1 Introduction

Landscape-scale vegetation clearance for agricultural production is a pervasive theme in many regions and countries and has resulted in habitat loss, habitat degradation and interruption of landscape-scale ecological processes (Foley *et al.*, 2005). This scenario is no less true in New Zealand where coastal and lowland ecosystems in particular have historically been, and continue to be, pressured by an expanding population and consequent resource pressure (Ministry for the Environment, 2007). In the Waikato, lowland forest and wetlands were almost completely cleared of vegetation in the late 19th and early 20th centuries to make way for agriculture (Daly, 1990). The resultant mosaic of vegetation is characterised by small fragments of forest and wetland that are usually secondary in growth and far from any continuous, non-modified tract of native forest or wetland (Burns *et al.*, 2000; Burns *et al.*, 1999; Champion, 1988).

Kahikatea (*Dacrycarpus dacrydioides*) is the dominant canopy tree in lowland forest patches in the Waikato due to its light-loving, semi-swamp tolerant nature. These small patches typically persist on farmland that is marginal for farming and may have been used for stock grazing in periods of low resource availability (Jay, 2005). This practice persisted well into the twentieth century; consequently, many of these patches have a long history of grazing (Norton and Miller, 2000). Recently there has been a slow but steady move to protect and restore, by fencing off and covenanting patches of forest that might not otherwise persist in the landscape. However, forest patches with histories of grazed have severely reduced species pools from which to recover compared with non-grazed patches, due to trampling of the soil particularly in the shrub and ground-cover layers of the forest (Atkinson, 2001; Hobbs, 2001; Lunt *et al.*, 2007; Miller, 2006) and may be so degraded that anything resembling the original vegetation is unable to be recovered (Spooner and Briggs, 2008). Additionally, grazing by stock may have

removed stock-palatable species from the landscape completely (Norton and Miller, 2000) and soil nutrient dynamics are sometimes changed substantially (Flinn and Marks, 2007).

Consequently, it is not known whether these patches will be able to recover the native biodiversity and ecosystem function present prior to grazing. Although fencing reduces the disturbance, browsing and compaction caused by livestock and will allow native species to regenerate; it also allows adventive species to respond to the reduction in grazing pressure and grow undisturbed (Champion, 1988). For many adventive species (mostly pasture weeds and herbs), increased native cover will be sufficient to reduce any impact after approximately twenty years (Smale *et al.*, 2005). However, there is a subset of adventive species that, if not managed, may significantly alter ecosystem structure and therefore function (Hobbs and Yates, 2003).

Because wide-spread fencing of patches is still relatively new as a management tool and the impact of many adventive species is still being realised, there is little empirical evidence to suggest the likely outcome. Consequently, the primary aim of this chapter asks: Will fencing alone enable the persistence of the indigenous flora and the recovery of the vegetation of kahikatea-dominated remnants in the Waikato region? Or will adventive species and reduced biodiversity prevail? Arising from this, what level of management is required to reduce the weed populations to manageable levels and encourage native biomass production and native species regeneration and recruitment? And what are the constraints which limit achievement of these outcomes?

2.2 Methods

2.2.1 Sites

Twenty locations in the Waikato region with a total of twenty-five patches of different fencing times each differing in size and distance to an urban area were chosen for this study. In the absence of long-term permanent plots, the best way to measure vegetation change over time is to use a chronosequence of sites in which all factors other than time are standardised (Burrows, 1990). Consequently, the current study employed a 'space-for-time' method with selection of patches primarily based on time since grazing exclusion and location within the Hamilton

basin to achieve an even spread of ages and distance from an urban area and avoid confounding factors such as soil properties and climate. However, as is evident from Table 1, this was not completely possible due to the scarcity of certain remnant age/location combinations.

Table 1. Distribution of forest patches in this study based on fencing time and proximity to an urban area.

Fencing time (years)	Urban	Peri-urban	Rural	Total
0-5	2	2	1	5
6-15	0	3	3	6
16-25	0	4	0	4
26-45	1	2	5	8
46+	2	0	0	2
Total	5	11	9	25

McDonnell and Pickett (1990) and McDonnell *et al.* (1997) conservatively define an urban area as one with a human population density greater than 620 individuals per square kilometre and a rural area as one with population density less than 10 individuals per km^2. These thresholds were followed to classify urban, peri-urban and rural zones for the present study. Other variables measured were the degree of management effort by landowners/caretakers and the distance (in metres) of the nearest edge of the forest to the nearest main road, because transport of seed by vehicles has been shown to be a significant vector in weed invasion (Timmins and Williams, 1991; Von der Lippe and Kowarik, 2007).

Management regime was characterised after discussion with the land owners/caretakers of each remnant. Remnants were classified as having no, low, medium or high management (Table 2). The degree of vegetative recovery was defined according to the percentage of species characteristic of kahikatea-dominated forest present in a patch (Appendix 2). The list was compiled from articles by Clarkson *et al.* (2007) and Burns *et al.* (1999). Acceptable levels of weed presence was defined as less than 5% weed cover in a patch.

Table 2. Management categories used for the study and their categorisation based on weed and pest animal control frequency.

Management regime	Weed control	Pest Animal control
No management (1)	None	None
Low management (2)	- Sporadic - At most annual - Species-focussed - Low intensity	Occasional
Medium management (3)	- Regular - Up to six-monthly - Species-focussed - Intensive	One of the following: - Traps - Bait stations - Hunting
High management (4)	- Regular - More often than six-monthly - Removal of all weed species - Intensive	Two or more of the following: - Traps - Bait stations - Hunting

2.2.2 Data collection

5m by 5m sampling areas (hereafter 'plots') within the patches were chosen by use of a random number table and a transect running the entire length of the study area. For example, a number from the table of 59 12 L would result in a plot being placed 59 metres along the transect into the patch and 12 metres left of the transect. The number of plots within any given site was determined in-field by use of a running mean to ensure sampling adequacy was attained (Mueller-Dombois and Ellenberg, 1974). This resulted in the measurement of 196 plots totalling 4900 square metres – a sampling intensity of 0.80%.

All vascular plant species within the plots were identified to species and their cover within the plot estimated as per the RECCE method (Allen, 1992) except that actual cover percentages were estimated, not Braun-Blanquet cover scales. Litter cover was also estimated visually as part of the categorisation of ground-cover.

2.2.3 Statistical analyses

Native and adventive species richness, relative cover and variety of growth forms as well as forest structure were tested against time since grazing exclusion (years), management regime, location (urban, peri-urban or rural), patch size and distance from a main road. Many of the variables were not normally distributed so non-parametric statistics were used as transformation did not add anything to the analyses. Correlations between the individual variables were explored using

Spearman's rank order correlation, from which significant interactions were identified for model building. Correlations within categories (i.e., within adventive tiers or within native tiers) will not be discussed because species often occurred in more than one tier and therefore the correlations are not independent. To test for recovery, the presence or absence of functional groups and 'characteristic species' of kahikatea-dominated forest was assessed against the aforementioned variables. Characteristic species were identified from literature by Clarkson *et al.* (2007) and Burns *et al.* (1999).

Homogeneity of slopes (HoS) analysis, a type of general regression modelling (GRM), was used to identify which of the independent variables best explained the variation in the measured variables and test for any interaction effects. Multiple regression analysis was then performed to determine the degree to which each model factor identified by HoS analysis contributed to apportioning variation in the dependent variable in question.

To make predictions about the effects of management regime and location on the patches, categorised scatter-plots with regression equations for a polynomial fit were used to determine percentage increase or decrease of native and adventive cover and growth form proportions over fencing time. All analyses were undertaken using Statistica v. 8. (StatSoft Inc, 2008).

2.3 Results

2.3.1 Species richness/biodiversity

Patch size was the best independent variable in predicting native species richness ($F_{1, 17} = 5.40$, $p<0.05$) accounting for 51% of the variation in native species richness between sites, with larger patches generally containing the highest numbers of native species.

Native species richness generally increased with increasing age at a rate of 7% per 10-year period (Figure 1). Urban sites showed a similar curve in species richness increase over time (14%; $r =0.933$, $p<0.05$), whereas peri-urban sites showed an increase of 19% every 10 years up to 20 years of fencing then a dramatic decrease after this point ($r = 0.089$, $p=0.793$). Rural areas showed an initial decrease in

native species richness until approximately 17 years of fencing then increased quickly after this point (18% every 10 years; r = 0.386, p=0.271) (Figure 2).

Sites with no management showed a rapid increase in diversity up to 25 years then a similarly quick decrease after this point (50% per 10-year period; r = 0.318, p=0.540). Sites with low management showed an initial decrease in richness until 15 years of fencing then increased quickly after this point (23%; r = 0.753, p=0.05). Sites with medium management showed a slow decrease in native species richness after fencing (-13%; r =0.174, p=0.680). Sites with high management showed a steady increase over time (29%; r =0.831, p=0.081) (Figure 3).
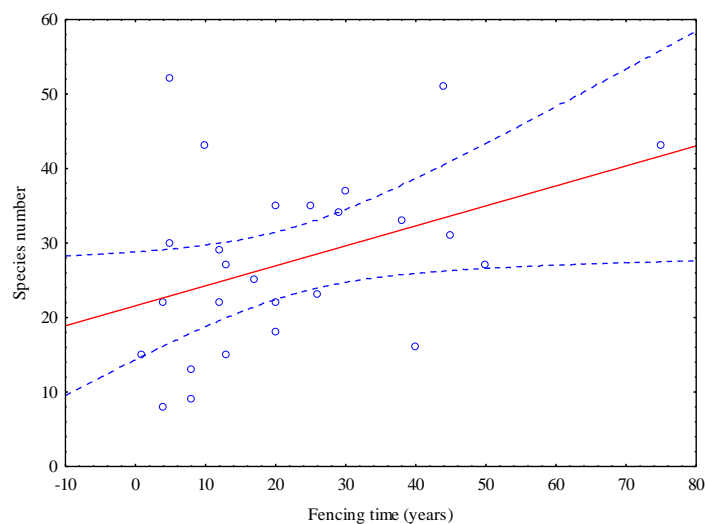
Figure 1. Native species richness scores across all patches compared with fencing time.
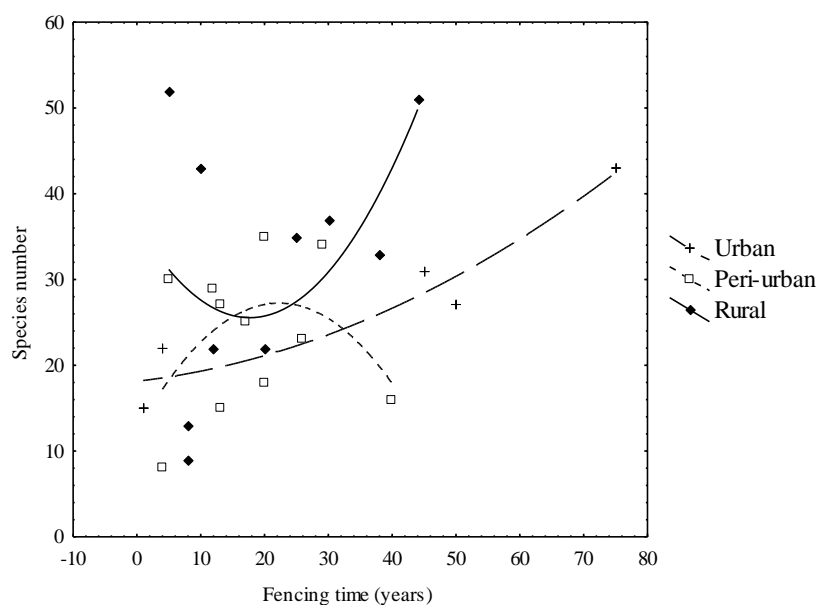
Figure 2. Native species richness scores across all patches as a function of patch fencing time categorised by location.
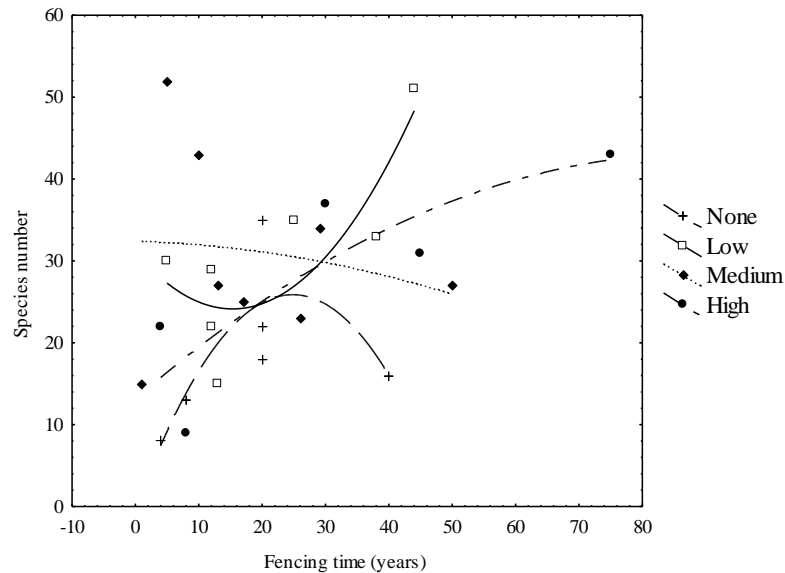
Figure 3. Native species richness scores for all patches against patch fencing time and categorised by management regime.

The interaction of patch size and fencing time was the best model in apportioning variation in adventive species richness ($F_{1, 17} = 6.46$, $p<0.05$) but even then was not particularly powerful only accounting for 29% of the variation between sites. Fencing time on its own was a fairly poor indicator with only a 2% reduction in adventive species richness per 10-year period evident (Figure 4) and a similar reduction with each hectare increase in patch size. Urban sites showed an increase in adventive species richness up to 30-40 years fencing time, after which, a decrease of 23% per 10-year period was observed ($r = 0.052$, $p=0.934$). Peri-urban sites showed a minor increase in adventive species richness until 15 years of fencing when a decrease of 4% per 10-year period was recorded ($r = -0.243$, $p=0.473$) and rural sites showed a steady decrease of 13% for every 10 year period ($r = -0.395$, $p=0.259$) (Figure 5).

Sites with no and low management effort showed a slight increase in adventive species richness until approximately 15 years after fencing then adventive species richness decreased by 5% ($r = -0.366$, $p=0.75$) and 12% ($r=-0.310$, $p=0.499$) respectively. Sites with medium levels of management showed a steady 15% decrease in adventive species richness per 10-year period after fencing ($r = -0.318$, $p=0.443$) and sites with high management showed an increase in adventive species richness up to 45 years when species richness started to decrease at 20% per 10-year period ($r = 0.313$, $p=0.608$) (Figure 6).

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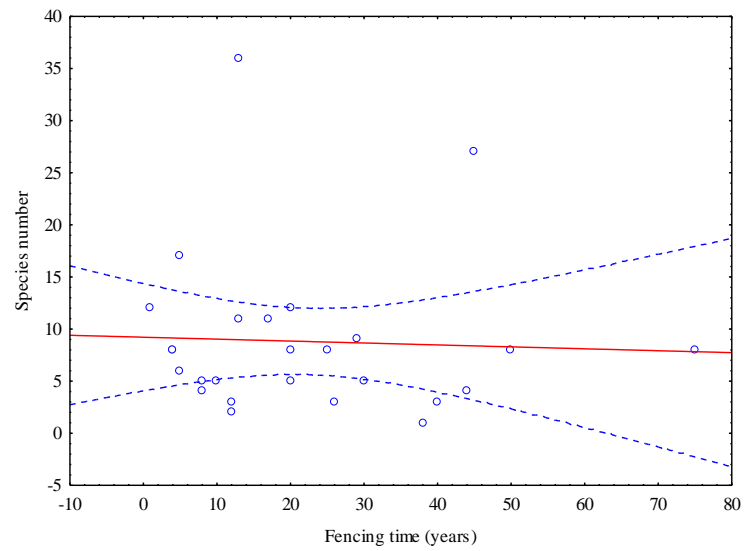
Figure 4. Adventive species richness scores for all patches as a function of patch fencing time.
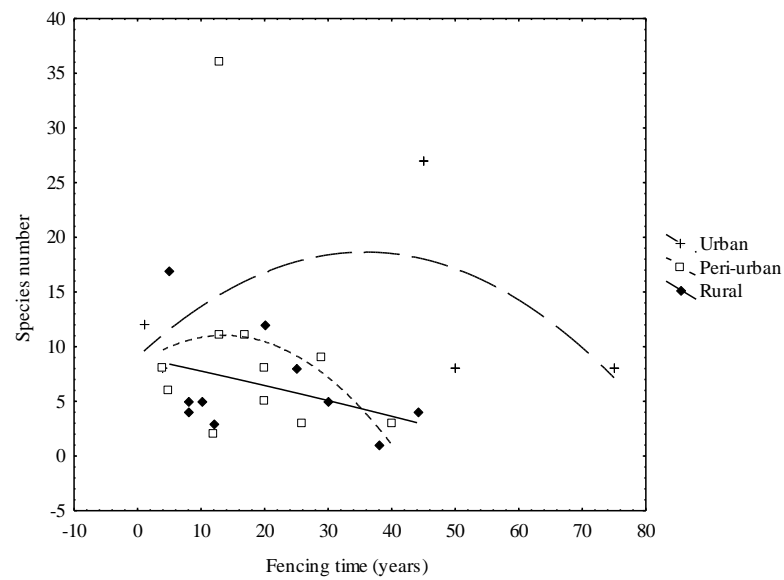
Figure 5. Adventive species richness scores for all patches as a function of patch fencing time and location.
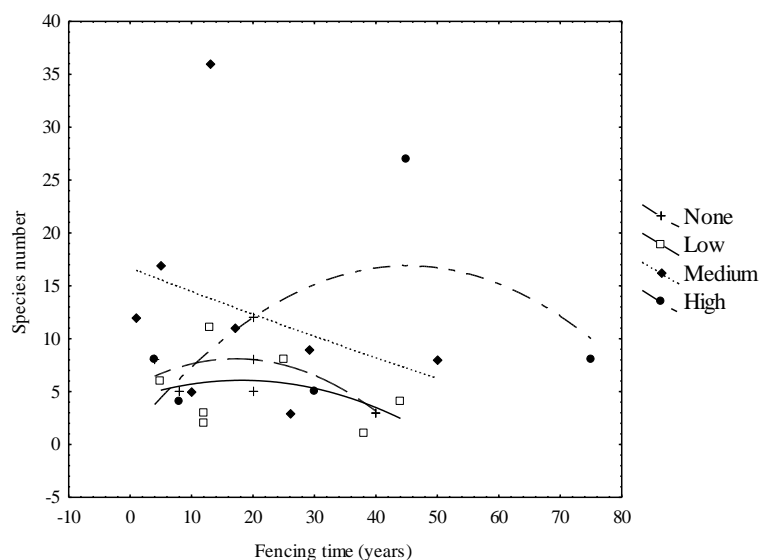
Figure 6. Adventive species richness scores for all patches as a function of patch fencing time and management regime.

2.3.2 Tier analyses

Table 3. Homogeneity of slopes best subset models for forest tiers. E = adventive species cover, N = native species cover and *=p<0.05, **=p<0.01 and ***=p<0.001.

	N 5-2	N 2-30	N <30	E 5-2	E 2-30	E <30
Location*Management*Size		6.335***	8.927***	14.656***	2.251*	
Location*Management*Age*Size	2.911**					28.491***

The interaction of location, management and size was the best model for apportioning variation in four of the six tiers under scrutiny with an explanatory power of 63% for native tier five ($F_{6, 189}$ = 6.335, p<<0.001) and 60% for native tier six ($F_{6, 189}$ = 8.927, p<0.001), 56% for adventive tier four ($F_{6, 189}$ = 14.656, p<<0.001) and 48% for adventive tier five ($F_{6, 189}$ = 2.251, p<0.05) (Table 2). For native species in the shrub layer, location explained 61% of the variation, fencing time 17%, management regime 15% and patch size 13%. For native species in the ground-cover, location explained 53% of the variation, fencing time 11%, management regime 10% and patch size 19%. For adventive species in the understorey, location explained 1% of the variation, fencing time 16%, management regime 17% and patch size 12%. For adventive species in the shrub layer location explained 22% of the variation, fencing time 20%, management regime 5% and patch size 21%.

For adventive tier six and native tier four, the interaction of location, management, fencing time and size was the best model in apportioning the

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measured variation between sites ($F_{6,\ 189}$=28.491, p<<0.001 and $F_{6,\ 189}$ =2.911, p<0.01 respectively) with explanatory powers of 73% and 40% respectively. Location explained 10% of the variation in adventive ground-cover, fencing time 2%, management regime 20% and patch size 17%. For native understorey cover, location explained 37% of the variation, fencing time 24%, management regime 18% and patch size 17%.
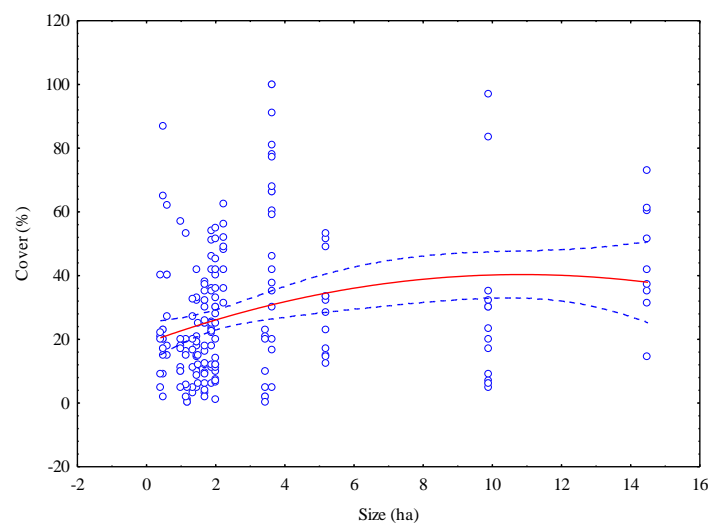
Figure 7. Native understorey cover scores as a function of patch size.
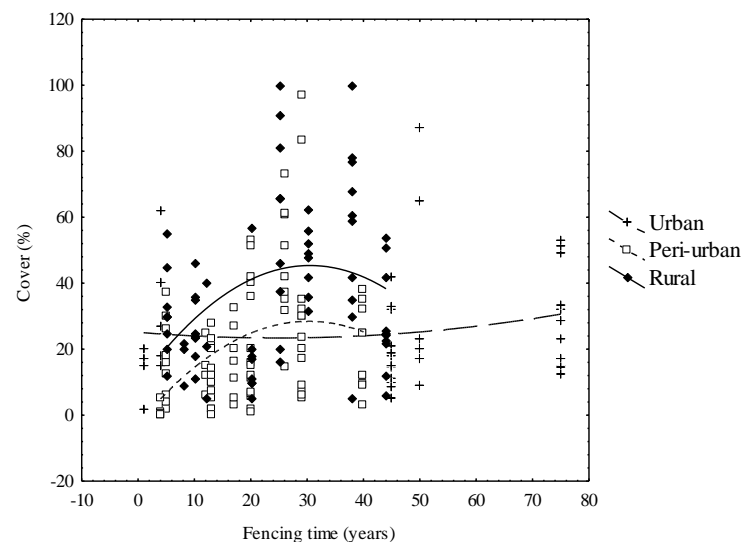
Figure 8. Native understorey cover scores as a function of patch fencing time categorised by location.
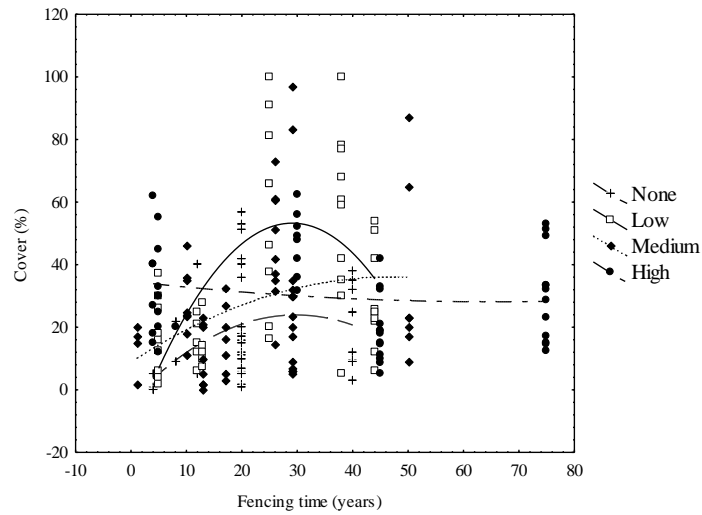
Figure 9. Native understorey cover scores as a function of patch fencing time categorised by management effort.

Native understorey cover was highly variable but increased at a rate of 12% per hectare increase in size until approximately 10 hectares of area where it began to plateau (r =0.235, p<0.001) (Figure 7). Native understorey cover in urban patches was initially moderate but increased at a very slow rate of 3% per 10-year period (r =0.112, p=0.479); Peri-urban patches by contrast, began with very low levels of native understorey cover but increased very quickly at 50% per 10-year period until approximately 30 years of fencing then began to decrease after this point (r =0.338, p<0.01); Rural patches varied substantially in their levels of native understorey cover but showed a general trend of increasing at 27% per 10-year period until approximately 30 years of fencing, at which point cover began to decrease again (r =0.267, p<0.05) (Figure 8).

In patches with no management, native understorey cover increased quickly from very low cover at 40% per 10-year fencing period up until approximately 30 years of fencing then began to decrease after this point (r =0.284, p=0.069). Patches with low management also began with very low native understorey cover which increased quickly up until approximately 30 years of fencing then decreased at a similar rate (39% per 10-year period; r =0.378, p<0.01). Patches with medium management effort began with low native understorey cover which increased at 36% per 10-year period up until approximately 50 years of fencing then started to plateau (r =0.303, p<0.05). Patches with high management effort began with high

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understorey cover which decreased gradually over time (3% decrease per 10-year period; r =-0.139, p=0.345) (Figure 9).

Figure 10. Native shrub-layer cover scores as a function of patch size.
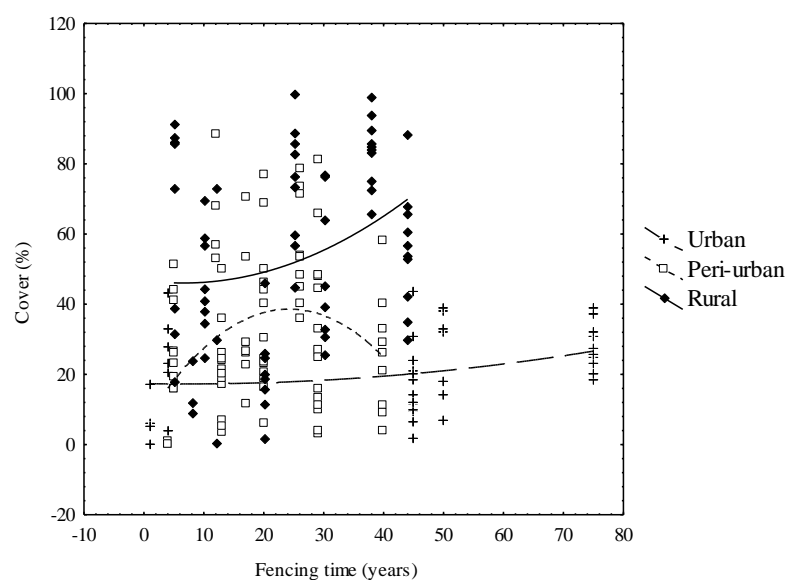
Figure 11. Native shrub-layer cover scores as a function of patch fencing time categorised by location.
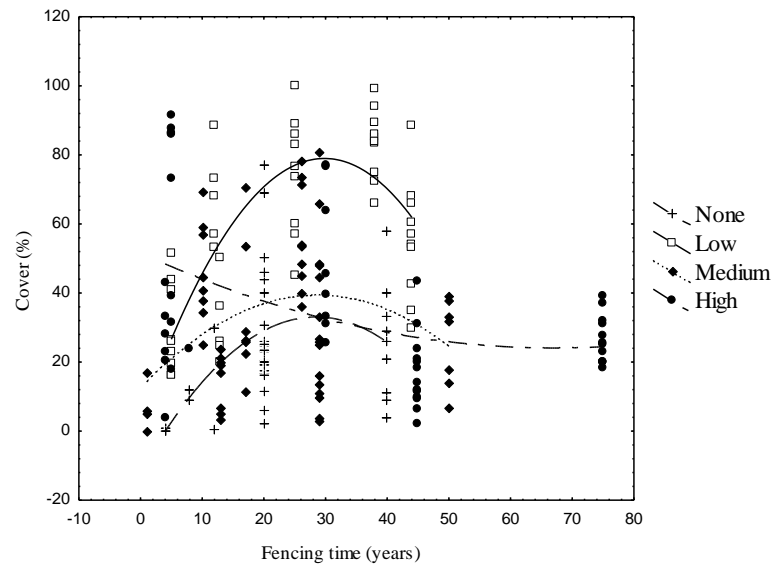
Figure 12. Native shrub-layer cover scores as a function of patch fencing time categorised by management effort.

Native shrub-layer cover between sites was highly variable but generally increased at a rate of 10% per hectare increase in size until approximately 10 hectares of area where it began to plateau (r =0.170, p<0.05) (Figure 10). Native understorey cover in urban patches was initially low and increased at a very slow rate over time (7% per 10-year period; r =0.284, p=0.069); cover in peri-urban and rural patches was highly variable between sites but showed quick increases in cover of 24% and 26% per 10-year period respectively (r =0.159, p=0.143 and r =0.302, p<0.05 respectively). In peri-urban patches however, cover in this tier began to decrease again after approximately 20 years of fencing whereas in rural patches no decrease in cover was evident after 45 years of fencing (Figure 11).

Patches with no management began with very low native shrub-layer cover which showed a sharp increase until approximately 30 years of fencing then a quick decrease in cover after this point (44% per 10-year period, r =0.342, p<0.05). Patches with low management showed a similar trend of change in cover but contained substantially more cover in this layer than patches with no management (30% increase per 10-year period; r =0.512, p<0.001). Patches with medium levels of management showed a similar trend of change in cover and also contained more cover than patches with no management, though this was not particularly higher (18% increase per 10-year period; r =0.084, p=0.532). Finally, patches with high levels of management showed a general trend of decreasing in

cover over time from high shrub-layer cover at 5 years of fencing to moderate levels at 70 years of fencing (r =-0.392, p<0.01) (Figure 12).

Figure 13. Native ground-cover cover scores as a function of patch size.
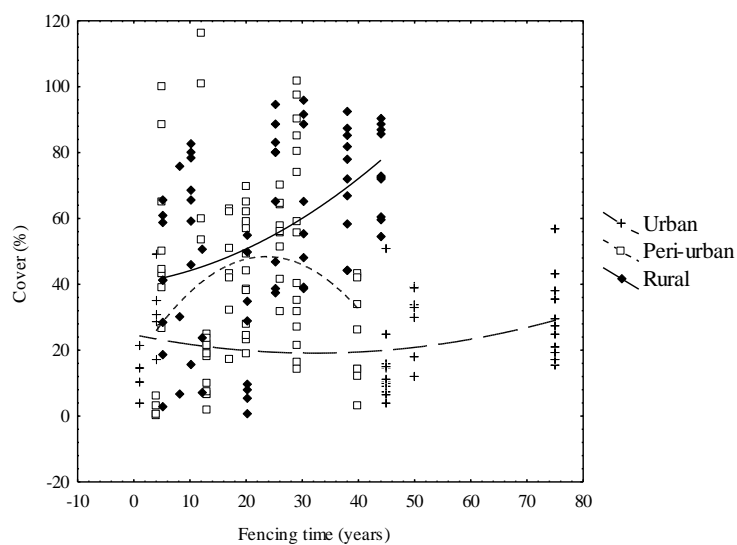
Figure 14. Native ground-cover cover scores as a function of patch fencing time categorised by patch location.
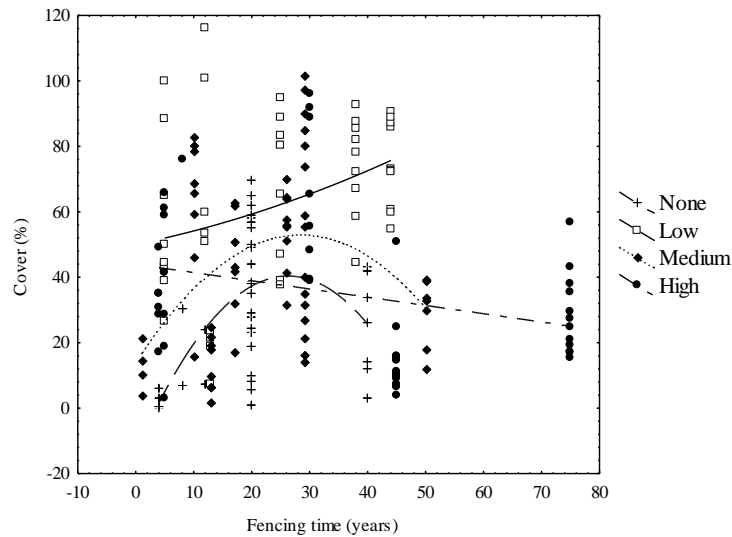
Figure 15. Native ground-cover cover scores as a function of patch fencing time categorised by management effort.

Native ground-cover between sites was highly variable but generally increased at a rate of 10% per hectare increase in size until approximately 10 hectares of area where it began to plateau (r =0.204, p<0.01) (Figure 13). Urban patches had very little native ground-cover and essentially stayed that way over time (r =0.124, p=0.436). Peri-urban patches were highly variable in their levels of native ground cover but began with moderate levels of cover which increased steadily until approximately 20 years of fencing, then decreased steadily after this point (20% change per 10-year period; r =0.106, p=0.331). Rural sites were also highly variable in their levels of native ground-cover but showed a rapid increase in cover over time with no evidence of a decrease (32% increase per 10-year period; r =0.462, p<<0.001) (Figure 14).

Patches with no management effort initially had low levels of native ground-cover but increased quickly until approximately 30 years of fencing then decreased at the same rate after this point (31% change in cover per 10-year period; r =0.250, p=0.110). Patches with low management effort initially contained high native ground cover which increased steadily showing no sign of decreasing (24% increase per 10-year period; r =0.360, p<0.05). Patches with medium management effort showed a similar trend to those with no management but continuously contained more native ground-cover than those with no management (18% change per 10-year period; r =0.090, p=0.501). Patches with high

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management effort initially had moderate levels of native ground cover but this declined over time at 9% per 10-year period (r =-0.286, p<0.05) (Figure 15).

Figure 16. Adventive understorey cover scores as a function of patch size.

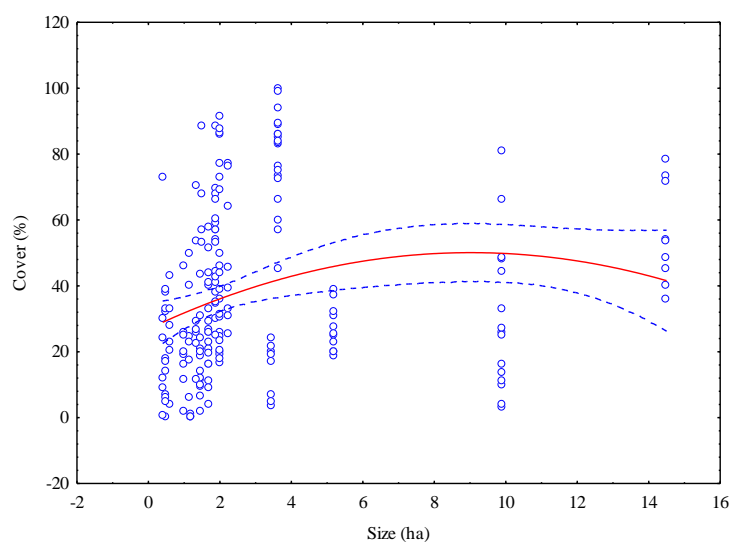**Figure 17. Adventive understorey cover scores as a function of patch fencing time categorised by patch location.**
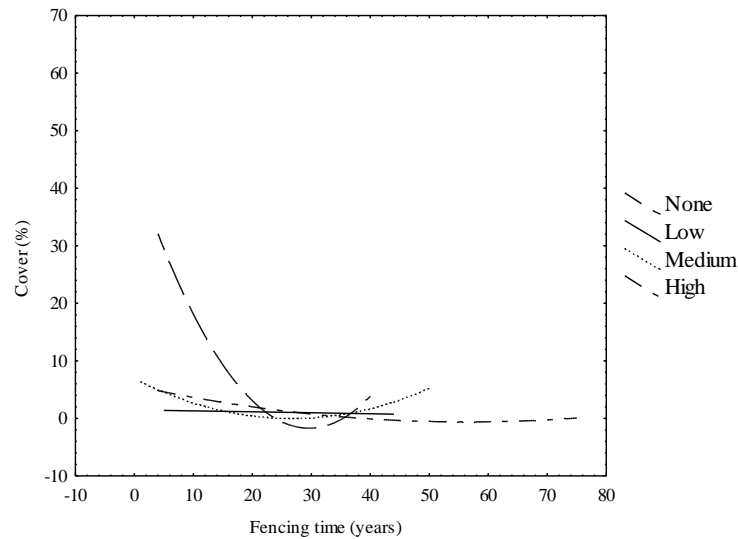
Figure 18. Adventive understorey cover scores as a function of patch fencing time categorised by management effort.

Adventive cover in tier four showed a general trend to decrease as patch size increased at a rate of 4% per hectare change in size (r =-0.192, p<0.01) reaching <5% cover in patches around 3 hectares in area (Figure 16). Urban patches contained very little adventive cover in this tier, staying below 5% cover and change in cover over time was very slow at 4% per 10-year fencing period (r =-0.1323, p = 0.4037). Conversely, peri-urban and rural patches showed a steady decrease in adventive understorey cover over time at 23% and 15% per 10-year period respectively, reaching <5% cover around 10 and 20 years of fencing respectively (r =-0.2491, p<0.05 and r =-0.389, p<0.01 respectively) (Figure 17).

Patches with no management effort had the largest initial cover of adventive understorey species but demonstrated a rapid decrease of 20% per 10-year period until approximately 25 years of fencing time reaching <5% cover at approximately 20 years, then increased at a similar rate after 35 years of fencing (r =-0.469, p<0.01); patches with low management effort had very little initial cover in this tier staying below 5% cover and showed a very slow decrease in adventive understorey cover of 3% per 10-year period (r =-0.0962, p=0.515); patches with medium levels of management began with a moderate level of adventive understorey cover which proceeded to decrease by 13% per 10-year period attaining less than 5% cover in under 10 years (r =-0.006, p=0.964). After 30 years however, cover started to increase at a similar rate. Finally, patches with high management effort also had moderate initial levels of adventive understorey

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cover which decreased at 3% per 10-year fencing period attaining <5% cover around 10 years of fencing time (r =-0.276, p=0.058) (Figure 18).
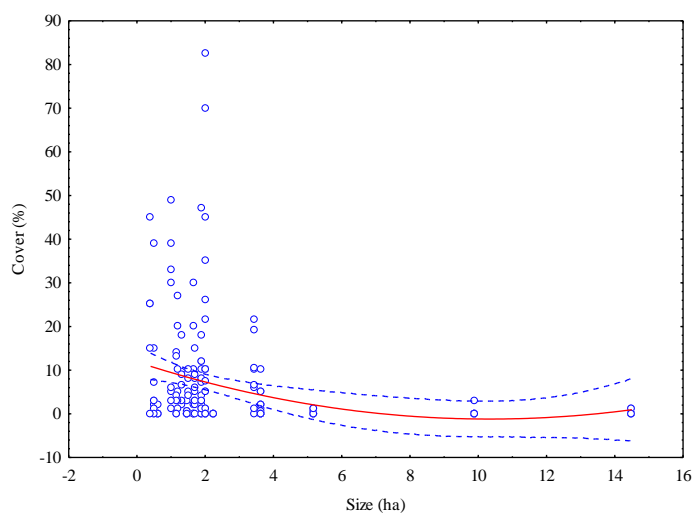
Figure 19. Adventive shrub-layer cover scores as a function of patch size.
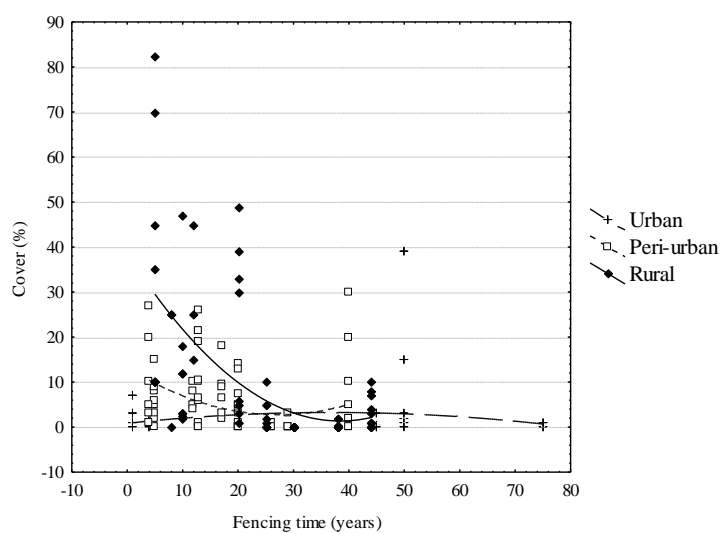
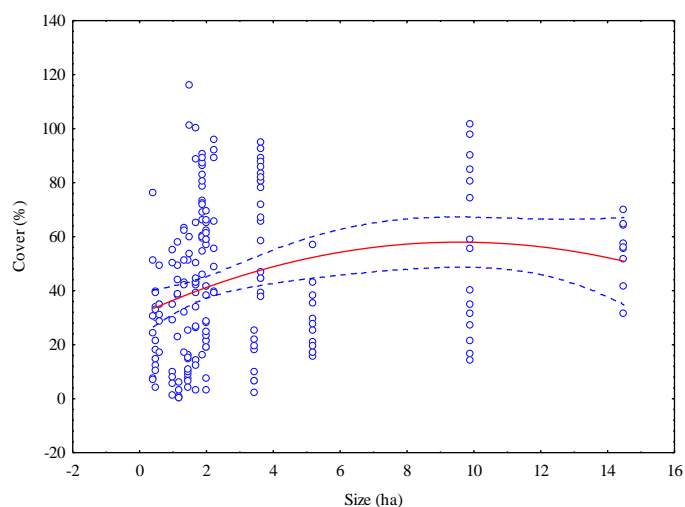**Figure 20. Adventive shrub-layer cover scores as a function of patch fencing time categorised by patch location.**
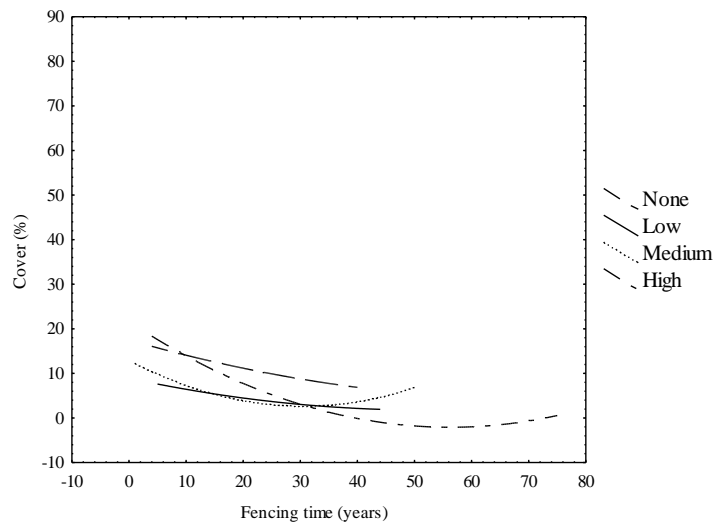
**Figure 21. Adventive shrub-layer cover scores as a function of patch fencing time categorised by management effort.**

Adventive cover in the shrub tier showed a general trend to decrease as patch size increased at a rate of 2% per hectare change in size (r =-0.244, p<0.001) reaching <5% cover in patches around 4 hectares in area (Figure 19). Although urban patches initially had the lowest adventive cover in this tier, cover increased slightly until approximately 40 years of fencing then declined at a similar rate after this point, but was always below 5% cover (13% per 10-year period; r =-0.015, p=0.926). Peri-urban patches initially contained moderate amounts of adventive cover in the shrub layer which gradually declined at 7% per 10-year period until approximately 25 years of fencing then began to increase at a similar rate after 30 years fencing time (r =-0.251, p<0.05) and attained <5% cover around 18 years of fencing. Rural patches by contrast, initially contained high adventive cover in the shrub layer which decreased at 14% per 10-year period reaching <5% cover around 30 years of fencing, but also showed a slight increase after this point (r =-0.537, p<<0.001) (Figure 20).

Patches with no management showed a decline in adventive shrub-layer cover at 1% per 10-year period not yet reaching <5% cover after 40 years of fencing (r =-0.209, p=0.183). Patches with low management, though initially containing moderate levels of adventive shrub-layer cover, showed a decline in this tier of 5% per 10-year period reaching <5% cover after 30 years of fencing (r =-0.372, p<0.01). Patches with moderate management also showed moderate adventive shrub-layer cover initially which decreased at 2% per 10-year period until

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approximately 30 years of fencing, at which point cover began to increase again (r =-0.126, p=0.348) and had attained <5% cover by 20 years of fencing. Patches with high management effort, though initially containing the highest adventive cover in this tier, decreased at 20% per 10-year fencing period, reaching <5% cover after approximately 30 years of fencing (r =-0.404, p<0.01) (Figure 21).
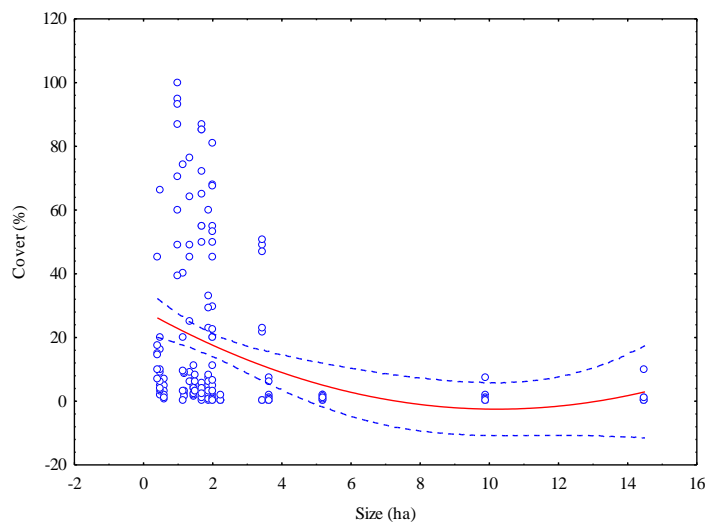
Figure 22. Adventive ground-cover cover scores as a function of patch size.
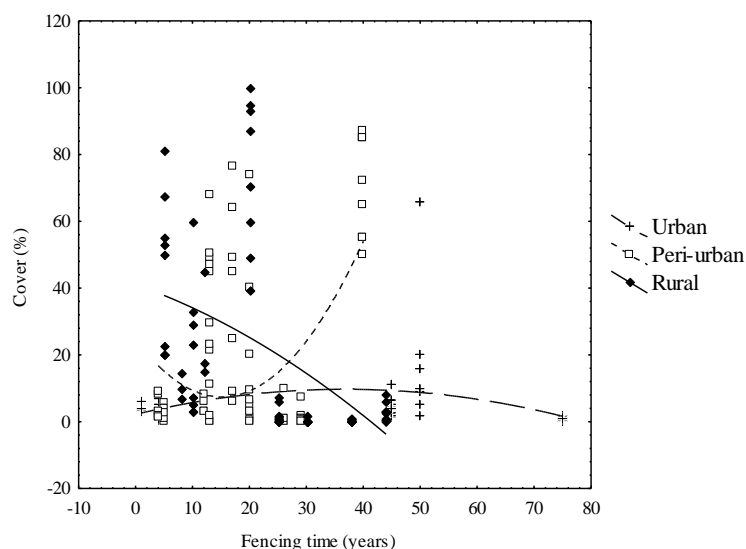
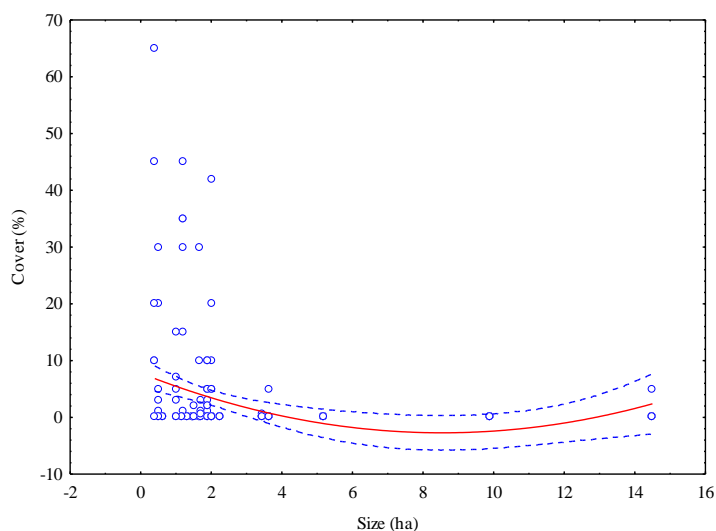**Figure 23. Adventive ground-cover cover scores as a function of patch fencing time and patch location.**

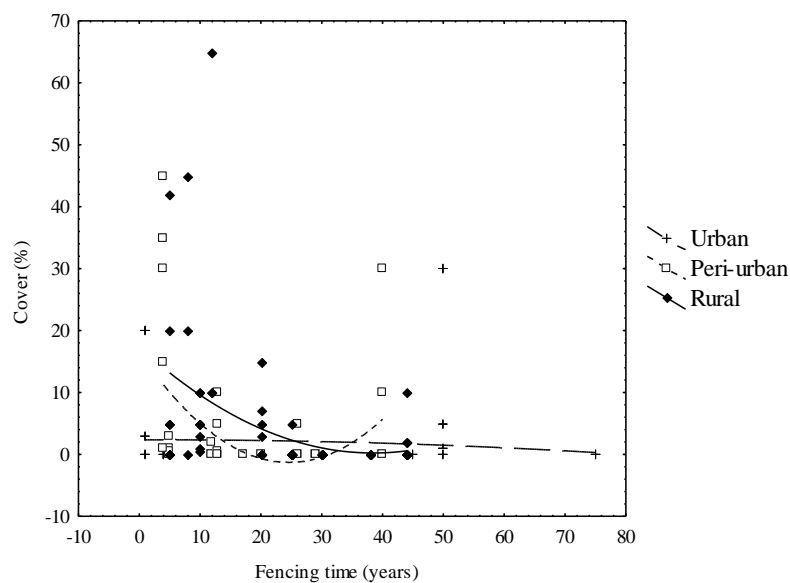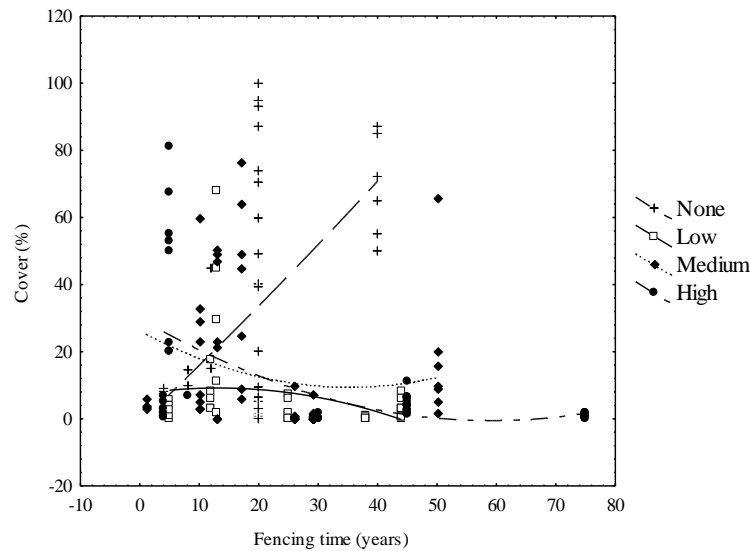**Figure 24.** **Adventive ground-cover cover scores as a function of patch fencing time and management effort.**

Adventive ground-cover showed a general trend to decrease as patch size increased at a rate of 2% per hectare change in size (r =-0.280, p<<0.001) reaching <5% cover in patches of approximately 6 hectares in area (Figure 22). Again urban patches had the lowest initial cover in this layer but showed an increasing trend up until approximately 40 years of fencing then a decline of the same magnitude after this point, attaining <5% cover around 65 years of fencing (13% per 10-year period; r =-0.030, p=0.853). Peri-urban patches initially contained high adventive ground cover and showed an initial decline in cover of 29% per 10-year period until 20 years of fencing, then increased at the same rate after this point reaching very high levels after 45 years (r =0.370, p<0.001). Rural patches again started with very high adventive cover in the ground layer which reduced quickly at 12% per 10-year period reaching <5% cover after approximately 38 years of fencing (r =-0.507, p<<0.001) (Figure 23).

Patches with no management began with moderate levels of adventive ground-cover which increased quickly at 31% per 10-year period (r =0.596, p<<0.001). Patches with low management effort started with low levels of adventive ground-cover which decreased slowly over time reaching <5% cover at about 35 years of fencing (7% decrease per 10-year fencing period; r =-0.282, p=0.052). Patches with medium management effort started with high levels of adventive ground-cover which decreased slowly until approximately 35 years fencing then began to increase at a similar rate (2% per 10-year fencing period; r =-0.152, p=0.255).

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Patches with high management effort began with moderately high levels of adventive ground-cover then decreased quickly at 19% per 10-year period reaching <5% cover after approximately 36 years of fencing (r =-0.499, p<0.001) (Figure 24).

2.3.2 Functional groups/species composition

The combination of patch location and fencing time was the best predictor of native species functional group increase ($F_{2, 16}$ =1000.11, p<<0.001) especially for herbaceous and shrub species – two of the subsets most affected by livestock grazing. Figures 25, 26 and 27 illustrate the contribution of the seven growth forms other than trees to the native biomass in each type of forest patch: urban, peri-urban and rural. Although urban patches contained cover in all tiers of the forest, Figure 25 illustrates that this is primarily composed of tree fern cover with little input from grasses/sedges, shrubs, lianes and epiphytes and none from herbaceous species. Ground fern species represent the next most numerous growth form cover but have four times less cover than tree ferns.
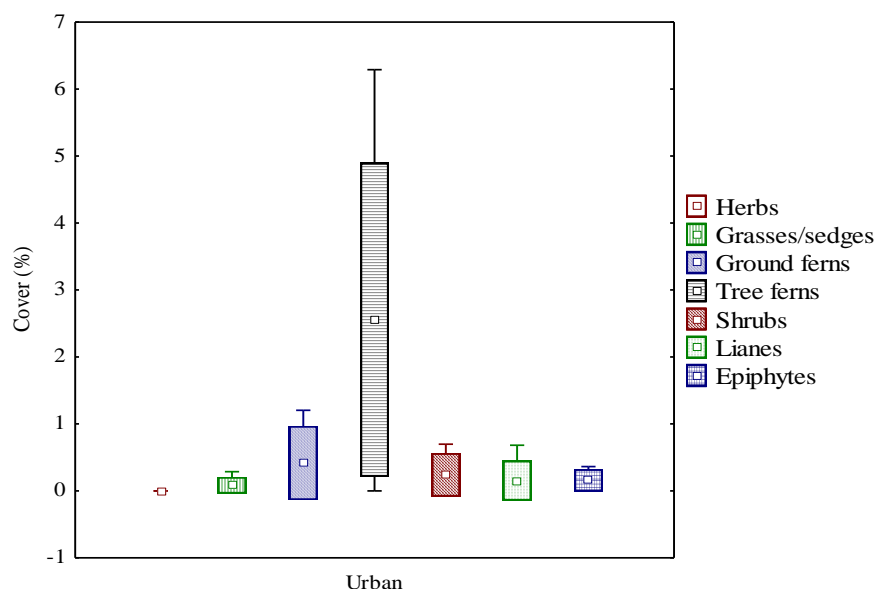
Figure 25. Native species cover scores in urban patches as a function of growth form.

Peri-urban forest patches, by contrast, contain similar levels of cover in ground fern and tree fern growth form categories, and contain substantially more cover of shrubs, lianes, epiphytes and grasses/sedges but also contain no herbaceous species.
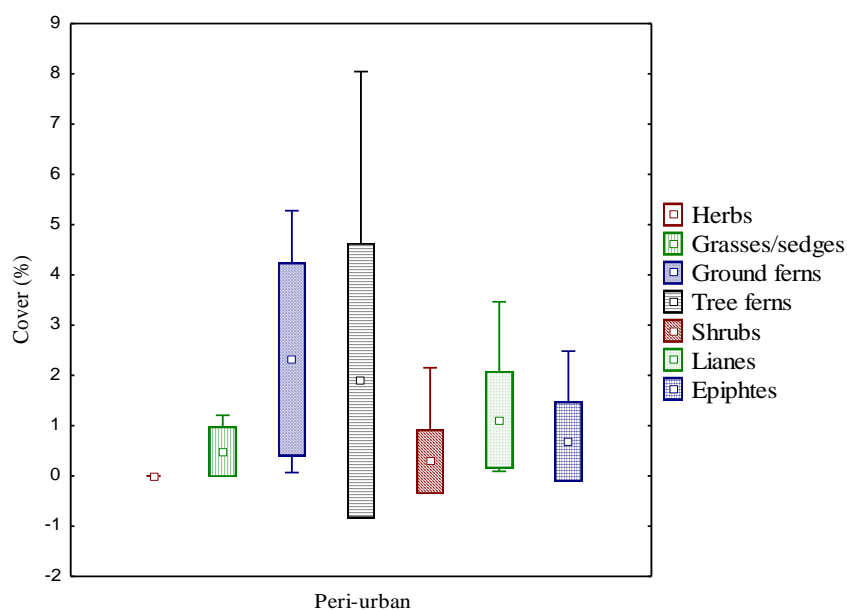
Figure 26. Native species cover scores in peri-urban patches as a function of growth form.

Rural patches have a much more even spread of cover over grasses/sedges, ground ferns, tree ferns and shrubs, were the only patches to contain herbaceous species and had moderate levels of liane and epiphytes.
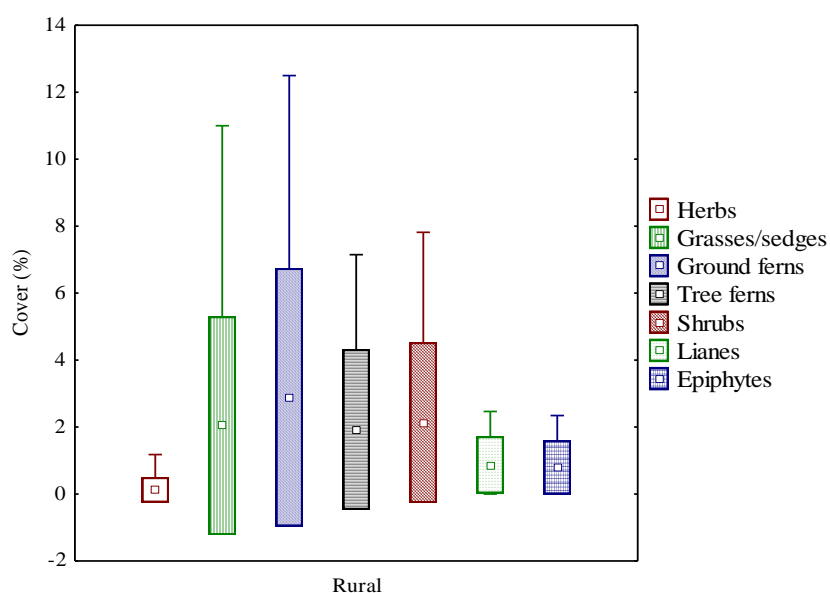
Figure 27. Native species cover scores in rural patches as a function of growth form.

2.3.3 Species characteristic of kahikatea-dominated forest

Of 36 species listed as characteristic in kahikatea-dominated forest by Clarkson *et al.* (2007) and Burns *et al.* (1999), five were completely absent from any of the forest patches in this study. These species were *Astelia fragrans*, *Carex lambertiana*, *Myrsine divaricata*, *Plagianthus regius* and *Syzygium maire*. The greatest number of characteristic species in any one fragment was 19 (53% of the 36 kahikatea characteristic species identified), the minimum was 4 (11%) and the average was 11 (29%). Patch age and management effort were identified as the only independent factors significantly correlated with the percentage of characteristic species in a patch together explaining 59% of the variation observed with explanatory powers of 53% and 47% respectively (p<0.001).

The number of characteristic species in a patch was positively correlated with time since grazing exclusion (age) and management regime with patches fenced for longer and patches with greater management effort containing more characteristic species (Figures 28 and 29).
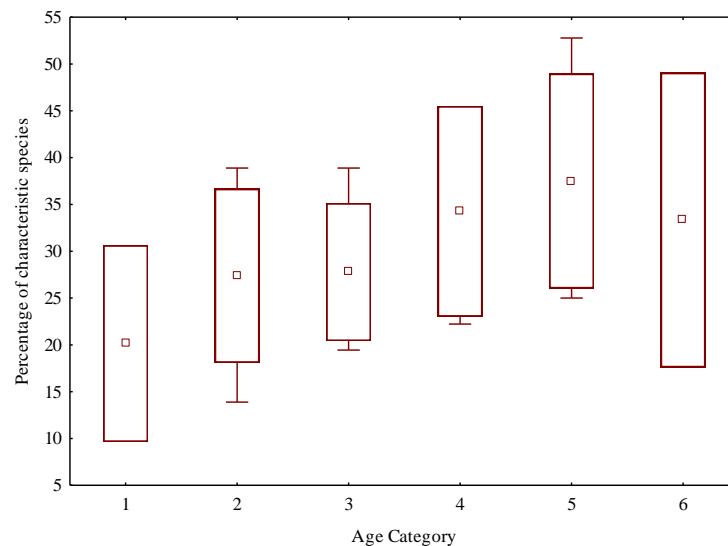
Figure 28. Percentage of characteristic species in a patch as a function of patch age. 1= <5 years, 2= 6-15 years, 3= 16-25 years, 4= 26-35 years, 5= 36-45 years, 6= >46 years.
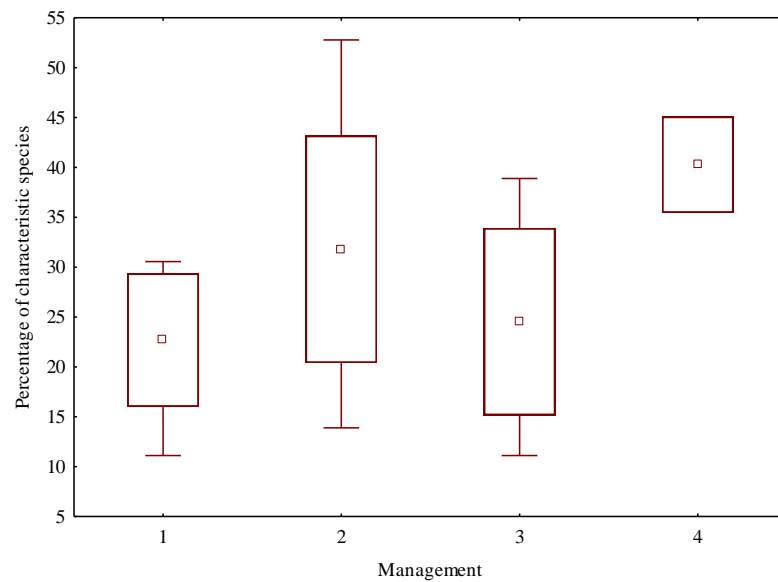
Figure 29. Percentage of kahikatea characteristic species in a patch categorised by management effort. 1= None, 2= Low, 3= Medium, 4= High.

17% of patches with no management effort contained one site with more than 30% of the characteristic species compared with 55% of sites with low management effort, 17% of sites with medium management effort and 100% of sites with high management effort.

2.4 Discussion

2.4.1 Patch size

Small patches essentially contain all edge habitat, which is characterised by high disturbance due to higher temperatures and wind speeds and therefore lowered soil moisture and greater daily temperature fluctuations (Fischer and Lindenmayer, 2007). In this study, patches below two hectares in area contained moderate to high levels of adventive species cover and native cover was moderate to low particularly in the understorey layer (although this was highly variable). Similarly, patches of mixed podocarp-broadleaf forest below one hectare in area contained no amelioration of temperature, vapour pressure deficit or photosynthetically active radiation and were consequently characterised by vegetation suited to a disturbed environment (Young and Mitchell, 1994).

In this study, six to seven hectares represents a critical size for native cover enhancement. Below this size native understorey, shrub layer and ground cover occupied less than 50% cover, and adventive species cover was greater than 5%. Likewise, Young and Mitchell (1994) found that nine hectares was a necessary size for patches to contain some interior environment sheltered from abiotic factors. Though there were some patches that did not conform to this pattern, these patches either had extensive planting of vegetatively dense native species such as flax (*Phormium tenax*/*P. cookianum*) around the perimeter for shelter and strategic weed management or had strategic weed management and had been fenced for longer than 30 years.

The benefits of *Phormium tenax* for native forest regeneration have been highlighted by Reay and Norton (1999) who found that flax 'clumps' play an important facilitative and nurse role for woody native species' regeneration. *P. tenax* establishes readily, is fast-growing, tolerates a wide range of environmental conditions and, once established, is able to tolerate low levels of grazing by livestock (Reay and Norton, 1999). In order to disentangle the relationship between patch size, native vegetation condition and management regime in the current study, note would need to have been made of the type of management utilised as well as its intensity separating out weeding and planting. Additional to deliberate planting of vegetatively dense native species on small patch margins, plantations of *Pinus radiata* adjacent to forest patches have been shown to act as

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edge buffers reducing temperature and PAR to interior-like conditions at forest patch edges and provide a nursing role for many native species (Denyer *et al.*, 2006; Ogden *et al.*, 1997).

2.4.2 Proximity to a main road

The proximity of a patch to a main road potentially exacerbates the effects of patch size as traffic can carry large numbers of propagules (Von der Lippe and Kowarik, 2007) and can change wind patterns (Forman and Alexander, 1998), an effect most keenly seen on the edges of fragments (Gelbard and Belnap, 2003). Indeed there were slightly higher numbers of adventive species in patches closer to roads and slightly fewer native species in patches closer to roads but this was not statistically significant (Table 11, Appendix 1). It is possible that these relationships would have been significant had note been made of the distance of each sampling unit (plot) to a main road and the relative contribution of native and adventive species to the plots makeup rather than one measure of distance to a road for the whole patch. Forest and scrub reserves in New Zealand that are closer to roads and railways have been shown to contain significantly higher numbers of problem weeds than those more distant from roads and railways (Timmins and Williams, 1991).

With respect to native and adventive cover, there was significantly more adventive ground cover in patches closer to main roads and significantly more native understorey, shrub-layer and ground cover in patches further from main roads (Table 11, Appendix 1). This result lends support to roadsides being disturbed areas that are not conducive to native species growth and acting as conduits for adventive species invasion and growth (Gelbard and Belnap, 2003). However, in order to truly test this relationship, note would need to have been made of the proximity of a sampling unit to a road and its cover characteristics. The results found here are potentially confounded however, by the fact that older patches tended to be further from main roads and tended to be larger than more recently fenced patches and rurally located patches also tended to be further from roads. Additionally, a forest patch located in close proximity to a road is more likely to be utilised for recreation than are patches further from roads which will subsequently increase weed propagule movement via clothes, animals and dumping of rubbish.

2.4.3 Location

For understorey native species, biomass increase was 9 and 16 times greater in rural and peri-urban locations than in urban patches. For native shrub-layer species, biomass increase was 3 or 4 times greater in peri-urban and rural environments than urban environments and in native ground-cover species, rural and peri-urban locations resulted in 10 and 7 times faster biomass increase than urban locations. Species richness was highest in rurally located remnants and rurally located remnants had higher proportions of characteristic species.

2.4.3.1Urban patches

Although urban patches were represented in the youngest and oldest categories of fencing time, any change in vegetation structure in the layers most affected by grazing were very slow compared to patches in other locations. For example, the average increase in native cover in any given tier was 16% per 10-year period compared to a 36% increase in peri-urban patches with the same management effort. This disparity in cover accrual as a result of location was particularly pronounced in the ground-cover layer where urban sites showed a maximal increase of only 12% per 10-year period as compared to a maximal increase of 28% in peri-urban patches and 16% in rural patches. Additionally, urban patches did not attain as high species richness scores as patches in the other locations fenced for similar (or lesser) periods of time, accrual of species was at a slower rate (9% per 10 year period as compared to 12% or 13% in peri-urban and rural patches) and ecosystem functional groups were highly skewed towards disturbance tolerant species such as tree ferns.

Although exotic cover was never high in urban patches, exotic species richness was generally higher in urban patches than patches in other locations. Exotic species cover and species richness also showed a very slow increase in cover over time in spite of the fact that most of the urban patches were highly managed. Because most urban patches were highly managed however, it is difficult to tease apart whether the lack of native biomass recovery is a result of location or intense management. Furthermore, assessment of change over time was difficult as there were few available sites in the urban environment for study. Ecosystems in urban areas suffer a disproportionate range of disturbance regimes compared to non-

urban systems from abiotic factors such as air and water pollution (McDonnell and Pickett, 1990) to biotic factors such as higher concentrations of introduced species and greater recreational use (Honnay *et al.*, 1999). The consequences of these concentrated changes include species composition change, increased morbidity, altered reproductive status, changes in growth rates and reduced species richness (McDonnell and Pickett, 1990) many of which are illustrated in this study.

These consequences are best illustrated in New Zealand by the changes in Claudelands bush, a kahikatea-dominated remnant located in Hamilton city and one of the patches in this study. This forest patch was once part of an extensive tract of mixed-podocarp forest adjoining a flax-manuka (*Leptospermum scoparium*)-cabbage tree (*Cordyline australis*) swamp (Gudex, 1955). Oral records indicate that the floor of this remnant was once so wet that there were holes between 1 and 1.5 metres deep filled with water throughout the forest (Gudex, 1955). Between 1954 and 1980 one third of the indigenous vascular flora became extinct with further fragmentation, grazing by livestock, desiccation from drainage and the smothering nature of *Tradescantia fluminensis* primarily responsible (Whaley *et al.*, 1997). Today, nearly half of the remaining original species are in small numbers with further losses likely, 60 adventive species are present, 17 of which are regarded as problem weeds and kahikatea are not naturally regenerating (Whaley *et al.*, 1997).

2.4.3.2 Peri-urban patches

Peri-urban patches had the least native and most exotic cover initially but showed largest changes in native cover over time. For example, native cover in peri-urban patches with moderate management averaged 36% increase in cover per 10-year period. As stated above however, peri-urban patches initially contained the least native cover in the layers under study and therefore the change in cover over time represented as a percentage is somewhat misleading. As is evidenced by the graphs, although the change in cover per 10-year period was highest in peri-urban patches, at no point did cover in these patches attain the same level as in rural patches. Additionally, after approximately 30 years of fencing time, all of the

layers in question showed a reduction in cover of the same magnitude as the previous increase.

It is possible that this effect is the result of maturation of the forest with early-successional shrub and tree species reducing in density as they mature and with latter stage species with slower-growth tendencies becoming more prominent. What is most likely however, is that this decline in cover is an artefact of the patches that were used in the study, as the oldest of the peri-urban patches (at approximately 40 years fencing) does not receive any pest plant management. This has resulted in a patch with little to no native understorey and shrub layer species, and a ground cover totally dominated by *Tradescantia fluminensis*, a serious weed of forest patches that suppresses native species regeneration (Standish *et al.*, 2001).

The trend in adventive species cover in peri-urban patches indicates that without moderate management, adventive species could come to dominate. This is supported by the characteristics of the patch mentioned previously which has been fenced for around 40 years but has little in the way of native species cover other than in the canopy and is dominated by adventive species in the other layers. Because the peri-urban environment is characterised by the land-use directly adjacent to urban areas it is possible that land use is more intensive and experiences a lot of traffic that flows into and out of cities and towns and is therefore subjected to similar volumes of weedy propagules as are urban areas (Simon, 2008). In contrast however, where the urban environment experiences high levels of abiotic stress, peri-urban environments are not so affected and may therefore be more at risk in developing serious weed infestations if management is not utilised.

This hypothesis is supported to a certain extent by the data collected on adventive species richness in sites in urban and peri-urban areas where peri-urban patches contain an average of 9.27 adventive species and urban patches contain an average of 12.6 adventive species (Table 9, Appendix 1). Once again however, it is difficult to project the results seen here into the future for peri-urban patches as there was only one patch in this environment that had been fenced for longer than 26 years which had no pest plant management whereas urban and rural sites had

three patches that fell into this category. Consequently, the data from this site heavily skews the trends seen in peri-urban patches.

2.4.3.3 Rural patches

Surprisingly, the magnitude of change over time in rural patches was slightly less than peri-urban patches change over time (average of 28% per 10-year period rather than 31% in peri-urban patches). However, the 'youngest' rural patch had only been fenced for 5 years but had accrued substantial cover during that time due to a high level of management effort, whereas the comparable peri-urban patch, which had been fenced for 4 years, contained little native cover has had no weed management. Therefore, when comparing growth over time, increases in cover were not as marked from youngest to oldest even though native biomass increase was probably greater in rural rather than peri-urban patches. This is evidenced by the fact that at no point did native cover in peri-urban patches exceed that present in rural patches of comparable fencing times. In order to directly test this hypothesis however, patches in both environments would need to have been fenced for the same period and have the same management effort expended upon them.

Of the tiers studied, the increase in cover of native ground-cover species was the most prominent in rural patches. In terms of the recovery of a patch, this is significant because ground-cover species are often the most sensitive to damage by stock and to habitat fragmentation, as a large proportion of ground-cover species constitute herbaceous and fern species which are prone to desiccation and do not tolerate disturbance (Hobbs, 2001; Lunt *et al.*, 2007; McLachlan and Bazely, 2001). Additionally, average native species richness was highest in rural patches and rural patches had the highest proportions of characteristic kahikatea-dominated forest species. This may be the result of rural patches being more likely to be close to a conservation reserve or large tract of native bush meaning that native species propagules are more readily available. In order to test this however, the distance of the patches to a large tract of native forest needed to be measured.

Although rural patches had the lowest average adventive species richness of the locations, adventive species cover was initially highest, occasionally exceeding

60% cover. This is most likely attributable to a set of woody adventive species that were heavily used as hedge-rows in early rural New Zealand like Chinese privet (*Ligustrum sinense*), tree privet (*Ligustrum lucidum*) and barberry (*Berberis glaucocarpa*) (McQueen, 1993). All of these species are capable of persisting below intact forest canopies and can form mono-specific stands excluding all other species (Champion, 1988; McQueen, 1993). Consequently, although the number of species is few, the ones that are present form dense cover, which is evidenced by the high cover in adventive understorey and shrub-layer tiers in patches fenced for less than 10 years.

The results here are obscured by the fact that rurally located patches tended also to be further from a main road and tended to be larger in size than their urban or peri-urban counterparts. This makes it difficult to ascertain whether the patterns seen here are as a result of their rural locality and potential proximity to native seed sources or are a result of their larger size and distance from the disturbance associated with road-sides.

2.4.4 Management

For understorey native species, medium to low management effort resulted in the greatest increases in biomass over time. For native shrub-layer species, low management resulted in twice as fast biomass increase as other management regimes. In native ground-cover species, low and medium management regimes were similarly effective in increasing ground-cover biomass and were 2 to 3 times more effective than high management effort.

2.4.4.1 No management

Although native cover initially increased and adventive cover initially decreased in patches with no management, the amount of native cover in any of the tiers never exceeded 40% and adventive cover did not reach a manageable level even after 40 years of fencing. Additionally, after approximately 30 years of fencing, native cover and species richness began to decrease and adventive cover increased indicating a shift towards a patch dominated by adventive species, particularly in the shrub and ground-cover layers. Sites with no management effort also contained a very low proportion of the species identified as characteristic of kahikatea-dominated lowland forest in the North Island. Consequently, if no

management effort is afforded these patches in the future, native regeneration will almost certainly cease and these patches will be lost from the landscape or any that do remain will no longer resemble kahikatea-dominated patches. However, because kahikatea and its associated canopy species are long-lived, it will take hundreds of years for this to eventuate. Therefore, if management intervention is supplied, these patches have the potential to recover.

2.4.4.2 Low management

Low management effort resulted in the greatest increases in native cover accrual over time particularly in the shrub and ground-cover layers and kept adventive species cover at a manageable level over time. Because this regime was characterised by irregular weeding, it may be that it better mimics natural disturbance and consequently is the regime to which native species are best suited. Natural disturbance regimes for mixed-podocarp forest however, are characterised by non-cyclic events such as extreme winds or storms and extreme flood events resulting in large-scale canopy gaps forming within the forest (Duncan, 1993; Wardle, 1974). Therefore, rather than being synonymous to natural disturbance regimes, low management effort may instead represent a threshold for management of adventive species, many of which are adapted to frequent soil disturbance.

Like patches with no management however, there was a tendency for native cover to decline after 30 years of fencing in the understorey and shrub-layer tiers. Unlike the patches with no management however, this pattern was not seen in the ground-cover layer. It may be that the pattern of decreasing cover in shrub-layer and understorey tiers is a result of self-thinning of shrubs and trees and represents a shift from an early successional state to a more mature forest structure. This is a more likely scenario than the one hypothesised for no management patches where adventive species out-compete natives for space because in low management patches, adventive cover continued to decline after 30 years of fencing. However, because shrub density was not recorded, this scenario is purely speculative.

There were no urban patches with this management regime so the conclusions drawn here cannot be extrapolated to encompass patches in urban locations.

However, for peri-urban and rural patches with low management effort, 30 years of fencing may represent a substantial recovery point.

2.4.4.3 Medium management

Medium management effort was also very effective in reducing adventive species cover and increasing native species cover but not to the same degree as a low management effort regime. The results presented here are however, highly skewed by patches located in urban areas. The decrease in native cover seen after 30 years of fencing in patches with medium management is the result of one very small urban site (<1 ha). If the urban patches are removed from the analyses (see Appendix 3), then medium management effort becomes more effective than low management effort in reducing adventive species cover but not quite as effective in promoting native species cover increase. However, by removing urban patches from the analyses, the oldest patch with medium management effort is only 29 years which makes extrapolation of trends past this point difficult.

2.4.4.4 High management

The negative impact of high management effort on native species cover was initially surprising but may be reasonable given that regular weeding will increase the frequency and intensity of disturbance of the ground layer, a disturbance regime to which native species are not adapted and may allow/facilitate the establishment of adventive species. This idea is supported by the multivariate regression analyses which show that low and medium levels of management were optimal in terms of increase in native cover and decrease of adventive species cover. Additional to this however, is the fact that most sites with high management effort occur in urban environments, a stress-prone environment. Therefore, the observed patterns may be the result of the location of the patch rather than the management regime.

Like patches with medium management effort, if we remove the urban patches from the analyses, the effectiveness of high management effort becomes much more pronounced and in some cases exceeds that of medium and low management effort. Also like the medium management patches with urban sites removed, the fragment with the longest fencing time is only 30 years which makes extrapolation difficult and by removing the urban patches from this analysis, the

sample size for testing medium management effort reduces from five to three patches.

2.5 Conclusions

Although fencing as a management tool was an important factor in promoting native species biomass recovery and ecosystem composition, the combination of the size of a fenced patch, its distance from a main road, and its location were better predictors of the observed variation in native species cover than fencing time alone particularly in the layers most affected by grazing. This result supports previous data on native forest species condition which indicates an area of nine hectares is necessary for the existence of interior vegetation conditions (Davies-Colley *et al.*, 2000; Young and Mitchell, 1994), roads (and other transport corridors) act as vectors for invasion by exotic species (Forman and Alexander, 1998; Timmins and Williams, 1991; Von der Lippe and Kowarik, 2007) and rurally located patches are more likely to be located in close proximity to larger tracts of forest and therefore species pools.

The combination of the same factors (size, distance from a main road and location) was also a better predictor of adventive species cover than fencing time alone. This is similarly supported by past studies which indicate that roads act as vectors for invasion by exotic species by increased propagule movement (Timmins and Williams, 1991; Von der Lippe and Kowarik, 2007) and providing conditions suitable for exotic species establishment (Forman and Alexander, 1998; Young and Mitchell, 1994), or conversely, unsuitable for native species establishment and growth.

However, the categorization and level of detail on management techniques and intensities used in this study and the deficiency of management effort, location and fencing time combinations did not allow robust comparison of management effort techniques on native species biomass and richness recovery or adventive species management. Consequently, these findings should be taken as potential outcomes only, not as a definitive statement on forest recovery. To enable robust comparison of vegetation recovery over time, and the range of management

techniques, permanent plots should be established in each patch and measured in 5 yearly intervals. Additionally, to enable robust comparison of management techniques, weed management and re-vegetation should be separated out to ascertain the degree to which these are important in native vegetation recovery/growth.

CHAPTER 3 –

Woody weed species distribution, density and control.

3.1 Introduction

Weeds have been identified as one of the leading factors worldwide in the suppression of native plants and consequently a major impediment to regaining healthy, functioning, indigenous ecosystems (Heywood, 1989; Prieur-Richard and Lavorel, 2000; Williams and West, 2000). In New Zealand, weed invasions have been identified as a serious threat to the preservation, conservation and continuation of native biodiversity; threatening ecosystem processes (Ehrenfeld *et al.*, 2000), already threatened plants (Miller and Duncan, 2004; Ogle *et al.*, 2000) and suppressing regeneration of native species (Ogle *et al.*, 2000; Standish *et al.*, 2001). There are approximately 24 700 adventive vascular plants resident in New Zealand (Williams *et al.*, 2002), of which 2 390 are considered naturalised or causal (Howell and Sawyer, 2006) and 328 are considered environmental weeds (Howell, 2008). Additionally, an estimated fourteen new plant naturalisations are identified each year (Landcare Research, 1996).

A conservative estimate of the economic costs of invasive weeds is around $100 million per annum with approximately $40 million of this lost annually from New Zealand's economic output and the remaining $60 million spent on defensive services to restrict the spread of current invasive plants and to control against new naturalisations (Williams and Timmins, 2002). Additionally, invasive/ environmental weeds directly threaten the survival of 61 threatened native New Zealand plant species and have an impact on at least another 16 species (Department of Conservation, 2000).

Environmental weeds are those species that are a threat to the health and functioning of ecosystems in that they alter or outright change existing ecosystem processes (Rejmánek, 2000). Compare this with species that rely on continual introduction by man to persist (causal weeds) or species that persist in the environment, have the capacity for reproduction and spread but are as yet localised in their effect (naturalised weeds) (Rejmánek, 2000). The damaging

effects that environmental weeds can have on native ecosystem function in New Zealand has been demonstrated by Standish *et al.*'s (2001) study of the impacts of *Tradescantia fluminensis* (wandering Jew/*Tradescantia*) on native forest regeneration. This weed can regenerate vegetatively from very small fragments and quickly carpets forest floors thereby suppressing native seedling establishment by reducing the amount of light reaching the forest floor. Consequently, only large-seeded, shade tolerant natives such as karaka (*Corynocarpus laevetigus*) and tawa (*Beilschmeidia tawa*) are able to germinate which could potentially alter ecosystem structure resulting in a forest composed only of large-seeded, shade-tolerant species. Similarly, *Clematis vitalba* (Old-man's beard) is a vine that rapidly invades disturbed forest systems and results in loss of forest structure and function as well as suppressing regeneration of other species by smothering trees and shrubs preventing reproduction and destroying their structure (Ogle *et al.*, 2000).

3.1.1 Woody weeds

Of the 328 species of plant that are considered environmental weeds in New Zealand, woody species comprise just over half (54%, 177) (Williams and West, 2000). Woody weed species have the potential to be particularly damaging to forest ecosystems as they can fundamentally change forest structure, nutrient and water cycling, fire or flood regimes and facilitate further weed invasion by acting as roosts and food sources for birds (Denslow and Hughes, 2004; Mack *et al.*, 2002). For example *Pyracantha angustifolia*, an invasive woody shrub in the central mountains of Argentina, acts as a locus of invasion for tree privet (*Ligustrum lucidum*) which establishes preferentially under this shrub than the co-dominant native shrub (*Condalia montana*) and contains densities of other bird-dispersed woody species eight times higher than under either *Condalia* canopy or in the absence of shrub cover (Tecco *et al.*, 2006). Gorse (*Ulex europaeus*) and buddleia (*Buddleia davidii*) displace woody and herbaceous native primary species and, though eventually facilitating a return to native forest, contribute to declines in indigenous coloniser species populations (Smale, 1990; Sullivan *et al.*, 2007).

3.1.2 Weeds in the Waikato

Although environmental weeds and declining biodiversity represent a threat to all ecosystems in New Zealand, the Waikato is particularly depauperate in native vegetation with only 1.6% of its original (pre-human) vegetation remaining (Clarkson and McQueen, 2004). Furthermore, the Waikato is dominated by lowland geography and 85 percent of lowland forests and wetlands in New Zealand have been converted to agriculture, exotic forestry plantations or residential buildings (Ministry for the Environment, 2007), therefore the vegetation that does remain is not only of regional, but national significance. Within the Waikato region, 85 species of plant have been identified as 'pest plants' requiring legislative action to reduce their spread and prevent further invasion (Environment Waikato, 2002). Of these 85, forty-one have the potential to disrupt lowland forest ecosystem processes, and of these forty-one, twenty-four are listed on the national pest plant accord (NPPA) as serious threats to New Zealand's biodiversity and economy (Ministry of Agriculture and Fisheries, 2008) and twenty-three are listed on the Global Invasive Species Database (GISD) of alien species that threaten native biodiversity worldwide (Table 13, Appendix 6).

In an agriculture-dominated environment like the Waikato, many of the most widespread environmental weeds of indigenous forest are those that were introduced as hedgerow species such as barberry (*Berberis darwinii* and *B. glaucocarpa*), gorse (*Ilex europaeus*), hawthorn (*Crataegus monogyna*), buckthorn (*Rhamnus alternus*) and privet (*Ligustrum lucidum* and *L. sinense*) (McQueen, 1993; Rahman and Popay, 2001). Hedgerows were traditionally planted as an alternative to fences as they not only provided a boundary but also shelter, wood for timber and fires, additional food sources or medicine (Baudry *et al.*, 2000). By the time European settlers moved to new lands, hedgerows were well established within the European psyche as of practical and cultural importance and so were replicated in the settled lands (Baudry *et al.*, 2000). Consequently, many of the species used in hedgerows in New Zealand have a much larger seed source advantage than many native species, whose range has been severely contracted. The widespread and prolific growth of these hedgerow species throughout the countryside is therefore unsurprising.

3.1.3 Weeds in Kahikatea-dominated forest

The small size and lack of connectivity to other forest patches threatens the persistence of kahikatea forest patches through altered hydrological regimes, competition from adventive species and continued grazing by livestock (Champion, 1988; Environment Waikato, 2007; Smale *et al.*, 2005; Walsh, 1898). An altered hydrological regime results in a drier substrate, which in turn provides more suitable conditions for shade-tolerant, drier substrate inclined species such as tawa and tītoki therefore potentially altering ecosystem competition (Champion, 1988). Invasion by adventive species can restrict and may even out-compete native species regeneration resulting in altered floristic trajectories (old man's beard, (Ogle *et al.*, 2000); gorse, (Sullivan *et al.*, 2007); buddleia, (Smale, 1990)). Additionally, continued grazing by livestock removes all vegetation but mature trees, facilitates weed invasion by spreading weed propagules and creating frequent disturbance and suppresses regeneration of native species (Atkinson, 2001; Champion, 1988; Cranwell, 1939; Walsh, 1898).

Smale *et al.*'s (2005) study of nine kahikatea forest patches found that only *Ligustrum sinense* was widespread, though *Berberis glaucocarpa* and *Hedera helix* were locally common. However, due the limited number of patches in this study, the present study seeks to determine whether this pattern holds throughout the Waikato Basin and aims to quantify the effects of fencing time, management regime and proximity to seed sources on woody weed invasion into kahikatea (*Dacrycarpus dacrydioides*)-dominated forest fragments in the Waikato region. It is expected that the results of the study will aid the understanding of the ecology of kahikatea-dominated forest fragments that have been released from grazing pressure, and will provide some guidelines on the most effective management regime for forest fragment owners and managers. Specifically this study aims to: 1) Identify the woody weed species present in kahikatea-dominated forest patches and determine the extent of their spread; 2) Identify the patch characteristics associated with each woody weed; 3) Determine the risk status the identified woody weeds pose to native forest patches; 4) Make recommendations for their control based on the aforementioned results.

3.2 Methods

3.2.1 Study sites

Twenty locations in the Hamilton basin with a total of twenty-eight forest patches of different fencing times each differing in size and distance to an urban area were chosen for this study. The Hamilton basin is a roughly oval-shaped depression around 80 kilometres long north to south and around 40 kilometres wide (McCraw, 1967) that was formed by parallel faulting and differential uplift of the land to the north, west and east, volcanism, erosion and many layers of alluvial deposits by the Waikato river (McCraw, 2002). The climate is warm-temperate and humid with a mean annual rainfall of 1186 mm, mean temperature range of 3.8 C (minimum July) to 23.8 C (maximum January), an average of 64 ground frost days per annum (New Zealand Meterological Service, 1998) and the prevailing wind direction is from the westerly quarter (New Zealand Meterological Service, 2009).

In the absence of long-term permanent plots, the best way to measure vegetation change over time is to use a chronosequence of patches in which all factors other than time are standardised (Burrows, 1990). Consequently, the current study employed a 'space-for-time' method with selection of patches primarily based on time since grazing exclusion and location within the Hamilton basin to achieve an even spread of ages and distance from an urban area and avoid confounding factors such as soil properties and climate. However, as is evident from Table 1, this was not completely possible due to the scarcity of certain patch age/location combinations.

Table 4. Distribution of forest patches based on patch fencing time and location.

Fencing time (years)	Urban	Peri-urban	Rural	Total
0-5	2	2	1	5
6-15	0	3	3	6
16-25	0	4	0	4
26-45	1	2	5	8
46+	2	0	0	2
Total	5	11	9	25

McDonnell and Pickett (1990) and McDonnell *et al.* (1997) conservatively define an urban area as one with a human population density greater than 620 individuals per square kilometre and a rural area as one with population density less than 10

individuals per km^2. These thresholds were followed to classify urban, peri-urban and rural zones for the present study. Other variables measured were the degree of management effort by landowners/caretakers and the distance (in metres) of the nearest edge of the forest to the nearest main road, because transport of seed by vehicles has been shown to be a significant vector in weed invasion (Timmins and Williams, 1991; Von der Lippe and Kowarik, 2007).

Patch size and distance to a main road were calculated using MapToaster Topo v. 4 (MetaMedia Ltd, 2006) using the track drawing tool. Current management regime was characterised after discussion with the land owners/caretakers of each patch. Patches were classified as having no, low, medium or high management according to the criteria listed in Table 2.

Table 5. Management categories used in this study and their definitions.

Management regime	Weed control	Pest Animal control
No management	None	None
Low management	- Sporadic - At most annual - Species-focussed - Low intensity	Occasional
Medium management	- Regular - Up to six-monthly - Species-focussed - Intensive	One of the following: - Traps - Bait stations - Hunting
High management	- Regular - More often than six-monthly - Removal of all weed species - Intensive	Two or more of the following: - Traps - Bait stations - Hunting

3.2.2 Data collection

Five metre by five metre sampling areas (hereafter 'plots') within the patches were chosen by use of a random number table and a transect running the entire length of the study area. For example, a number from the table of 59 12 L would result in a plot being placed 59 metres along the transect into the patch and 12 metres left of the transect. The number of plots within any given site was determined in-field by use of a running mean to ensure sampling adequacy was attained (Mueller-Dombois and Ellenberg, 1974). This resulted in the measurement of 196 plots totalling 4900 square metres – a sampling intensity of 0.80%.

All vascular plant species within the plots were identified to species and their cover within the plot estimated as per the RECCE method (Allen, 1992) except that actual cover percentages were estimated, not Braun-Blanquet cover scales. Vegetation cover was estimated in six tiers (Figure 30). Tier one comprised vegetation cover present at 25 metres height or over (canopy); Tier two comprised vegetation cover present between 25 and 12 metres in height (sub-canopy); Tier three comprised vegetation cover present between 12 and 5 metres in height (understorey 1); Tier four comprised vegetation cover present between 5 and 2 metres in height (understorey 2); Tier five comprised vegetation cover present between 2 metres and 30 centimetres height (shrub layer) and Tier six comprised vegetation cover present below 30 centimetres height (groundcover layer).
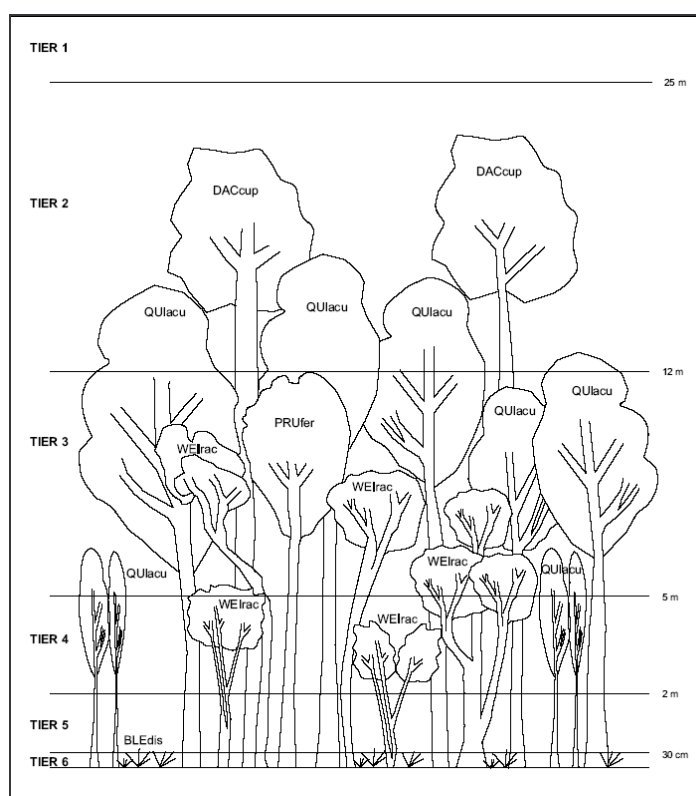
Figure 30. Diagram of RECCE forest tiers from Hurst and Allen (2007).

Native vegetative condition of each of the six tiers in a patch was then characterised as poor, moderate or good based on the amount of native vegetative cover in each tier (Table 3; adapted from Janssen, 2006) and scores for each tier summed to produce an overall vegetation condition score for each patch. Woody weed species were identified in each plot and their population structure and phenological status recorded. This population survey took the form of recording the number of seedlings in five categories (<15cm tall, 16-45cm tall, 46-75cm tall,

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76-105cm tall and 106-135cm tall), tallying the number of saplings (plants greater than 135 cm height but less than 2 cm diameter) and tallying and measuring the diameter (d.b.h) of trees (plants greater than 135 cm height and greater than 2 cm diameter) of each species in each plot (Allen, 1992). Phenological status was recorded as no flowers or fruit present, fruit or flowers present or old fruit or flowers present to determine reproductive status.

Table 6. Native vegetation condition categories by forest tier (adapted from Janssen 2006).

Vegetative tier	Good	Moderate	Poor
Emergent tier (>25m)	>20% cover	1-20% cover	No cover
Canopy (25-12m)	>90% cover	50-90% cover	<50% cover
Subcanopy (12-5m)	>40% cover	10-40% cover	<10% cover
Understorey (5-2m)	>40% cover	10-40% cover	<10% cover
Shrub layer (2m-30cm)	>40% cover	10-40% cover	<10% cover
Ground layer (<30cm)	>50% cover	10-50% cover	<10% cover

3.2.3 Statistical analyses

Woody weed species richness, proportion of plots per site containing woody weed species and woody weed species density were tested against time since grazing exclusion (years), management regime (none, low, medium and high), location (urban, peri-urban or rural), patch size, distance from a main road and native forest vegetation condition (good, moderate or poor). Many of the variables were not normally distributed so non-parametric statistics were used as transformation did not improve distributions or trends. Detailed analysis was only performed on those weed species that were present in >20% of patches (widespread weeds) and only *Ligustrum sinense* data was used for phenological comparisons as other species were either not flowering or fruiting in the patches studied (*Berberis glaucocarpa* and *Prunus* species) or there were too few data points to allow significant comparison (*Solanum nigrum* and *Solanum pseudocapsicum*). Additionally, of the widespread weeds only *Ligustrum sinense* is listed on the Waikato Regional Pest Management Strategy (Waikato RPMS) as a species of concern and therefore receives the most attention. Correlations between the individual variables were explored using Spearman's rank order correlation, from which significant interactions were identified for model building. T-tests were used to test for significant differences in treatment types, that is, differences between patch location, patch management regime and native vegetation condition on woody weed population structures.

Multiple regression analysis was performed to determine the degree to which each independent variable contributed to apportioning variation in the dependent variable in question. For box and whisker plots, categories with different letters indicate statistically significant difference based on t-tests. All analyses were undertaken using Statistica v. 8. (StatSoft Inc, 2008).

3.3 Results

3.3.1 Species richness

Table 7. Woody weed species richness descriptive statistics (n=28).

	Mean	σ	Min	Max
Woody species richness	3.23	1.68	1.00	6.00
% of plots per site with woody weeds	76.96	26.06	12.50	100.00

Although 50% of patches contained fewer than 3 woody weed species, all patches contained at least one woody species (mean 3.23, $\sigma = 1.68$). On average, 77% of plots in a given patch contained woody weed species though this was highly variable ($\sigma = 26.06$) and 38% of patches had 80% or fewer plots containing woody weed species.

Woody weed species richness was significantly negatively correlated with patch location (-0.492, $p<0.05$; $F_{(2, 23)} = 3.53$, $p<0.05$) and with native vegetation condition (-0.516, $p<0.01$; $F_{(2,23)} = 4.49$, $p<0.05$) with fewer woody species in rurally located patches than urban patches and fewer woody weed species in patches with better native vegetation condition (Figures 31 and 32). Peri-urban and rural patches did not differ significantly in the number of woody weed species they contained, nor did peri-urban and urban patches. However, urban patches containing significantly greater woody weed species richness than rural patches (t = 2.65, $p<0.05$).
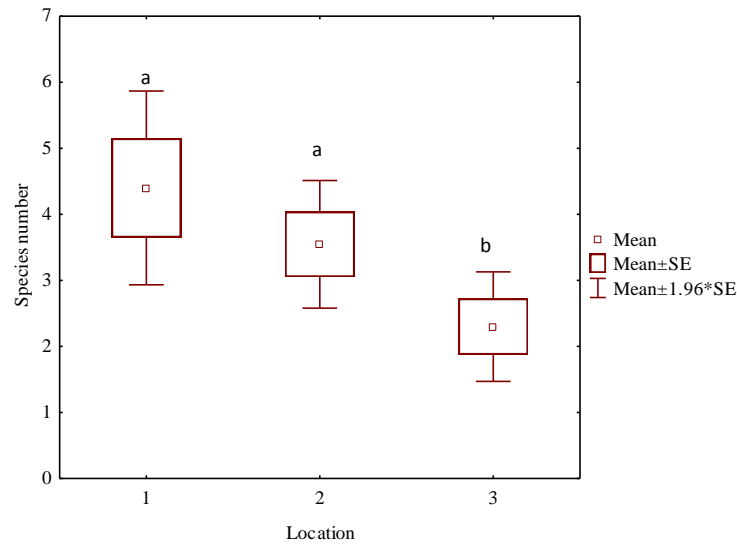
Figure 31. Number of woody weed species per patch based on patch location (n=28).
1=Urban, 2=Peri-urban, 3=Rural.

Woody weed species richness was significantly higher in patches with poorer native vegetation condition than patches with either moderate or good native vegetation condition (t=2.41 and t=2.55 respectively; p<0.05). Patches with moderate vegetation condition tended to have higher woody weeds species richness than patches with good native vegetation condition but this was not significant (Figure 32).
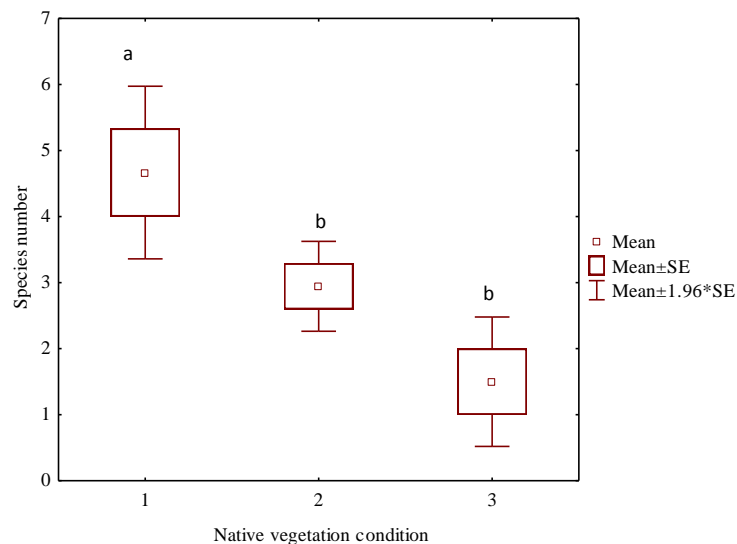
Figure 32. Number of woody weed species per patch based on vegetation condition (n=28).
1=Poor, 2=Moderate, 3=Good.

Fencing time and patch size were both significantly negatively correlated with the proportion of plots per site containing woody weed species (-0.575 and -0.625

respectively; both significant at p<0.01) with fewer plots per patch containing woody weed species in patches fenced for longer periods of time and in larger patches (Figures 33 and 34).
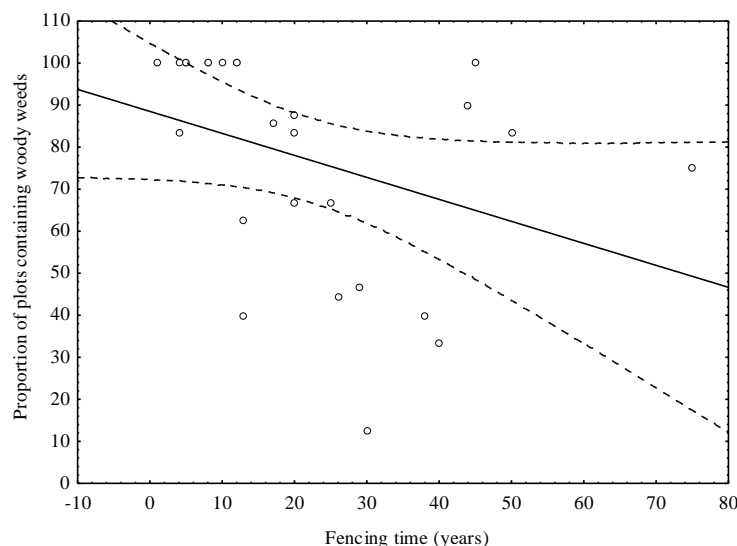
Figure 33. Proportion of plots per patch containing woody weeds as a function of patch fencing period (n=28).
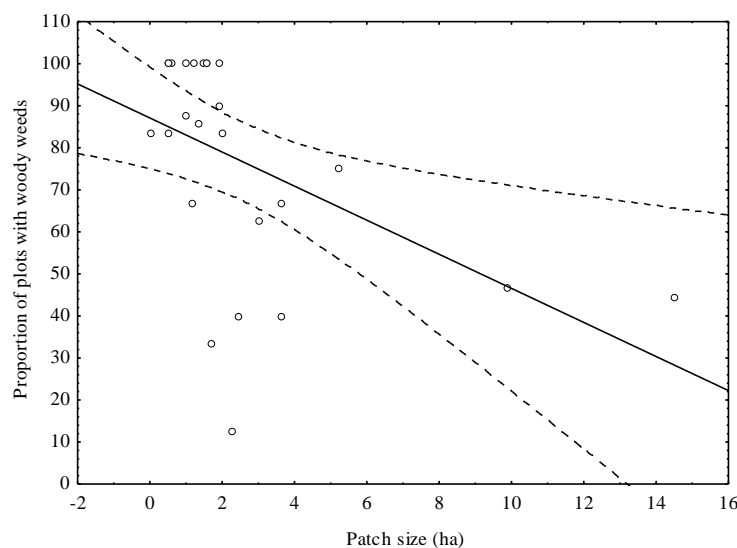
Figure 34. Proportion of plots per patch containing woody weeds as a function of patch size (n=28).

3.3.2 Weed spread, density and management

Of the 14 woody weed species found across the patches (Appendix 4), only five were common (present in >20% of patches). These weed species were: *Ligustrum sinense* (Chinese privet) found at all patches; *Solanum pseudocapsicum* (Jerusalem cherry), found at 50% of patches; *Prunus* species (flowering cherry

species) found at 39% of patches, *Berberis glaucocarpa* (barberry) found at 25% of patches; and *Solanum nigrum* (black nightshade) found at 21% of patches. Only one of these five – *Ligustrum sinense* – is listed on the Waikato Regional Pest Management Strategy as an environmental weed (Appendix 7).

Table 8. Top five woody weed species density per hectare descriptive statistics.

	Mean	σ
Ligustrum sinense	875.21	1770.24
Solanum pseudocapsicum	38.73	95.24
Prunus species	3.91	7.87
Berberis glaucocarpa	1.13	2.81
Solanum nigrum	2.91	13.09

Ligustrum sinense averaged 875 individuals per hectare but was highly variable with a standard deviation of 1770 and with 50% of patches containing fewer than 300 individuals per hectare. *Solanum pseudocapsicum* averaged 39 individuals per hectare but was also highly variable ($\sigma = 95$) with 65% of patches containing no *Solanum pseudocapsicum*. *Prunus* species averaged 4 individuals per hectare ($\sigma = 8$) with 76% of patches containing fewer than 5 individuals per hectare. *Berberis glaucocarpa* averaged 1 individual per hectare ($\sigma = 3$) with 83% of patches containing fewer than 2 individuals per hectare. *Solanum nigrum* averaged 3 individuals per hectare but was quite variable ($\sigma = 13$) with 89% of patches containing no *Solanum nigrum*.

Ligustrum sinense density was significantly positively correlated with patch location ($r = 0.528$, $p<0.01$) and significantly negatively correlated with patch fencing time ($r = -0.449$, $p<0.05$), management regime ($r = -0.387$, $p<0.05$) and patch size ($r = -0.516$, $p<0.01$) (Figures 38, 39, 40 and 41). With lower densities of *Ligustrum sinense* in urban patches, patches fenced for longer periods of time, patches with higher management effort and larger patches. *Solanum pseudocapsicum* was not significantly correlated with any of the independent variables but tended to be more abundant in peri-urban patches than rural and urban patches (Figure 35). *Solanum nigrum* density was only significantly correlated with patch location with higher density of *Solanum nigrum* individuals in peri-urban patches ($F_{(2, 26)} = 4.193$, $p<0.05$) (Figure 36). *Prunus* species density was significantly negatively correlated with patch location ($r = -0.573$, $p<0.01$) with lower *Prunus* densities in peri-urban and rural patches than urban patches ($F_{(2,26)} = 6.243$, $p<0.01$) (Figure 37). *Berberis glaucocarpa* density was

significantly positively correlated with *Solanum nigrum* density with patches containing high *Berberis glaucocarpa* density also tending to have higher densities of *Solanum nigrum* (Appendix 5). *Solanum nigrum* was not significantly correlated with any independent variable.
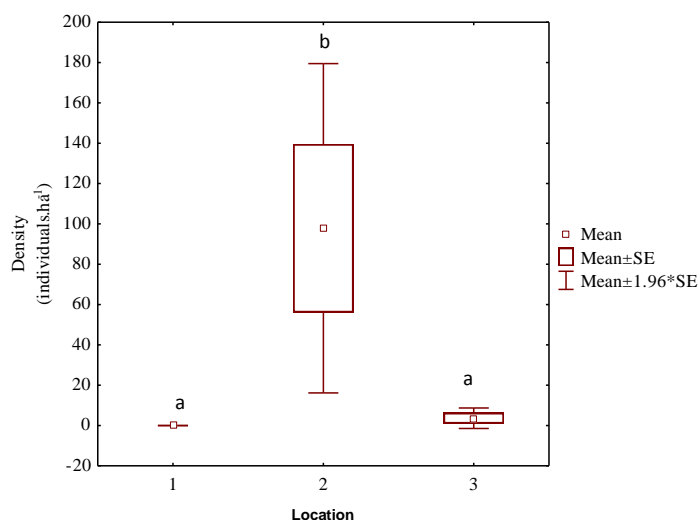
Figure 35. Density of *Solanum pseudocapsicum* individuals per hectare as a function of patch location. 1=Urban, 2=Peri-urban, 3=Rural.
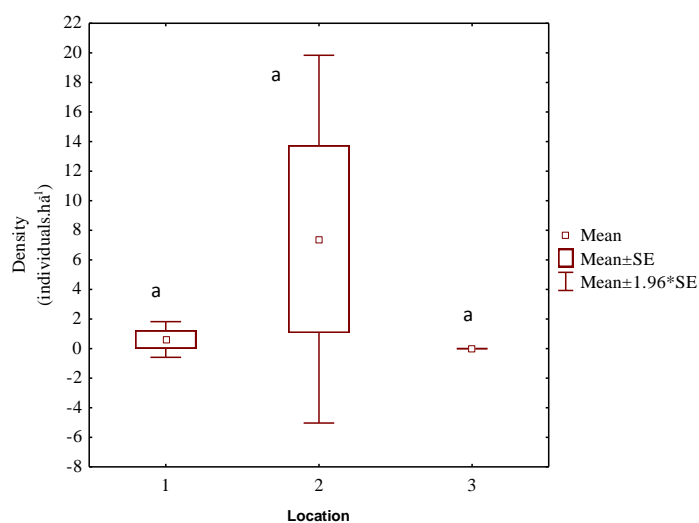
Figure 36. Density of *Solanum nigrum* individuals per hectare as a function of patch location. 1=Urban, 2=Peri-urban, 3=Rural.
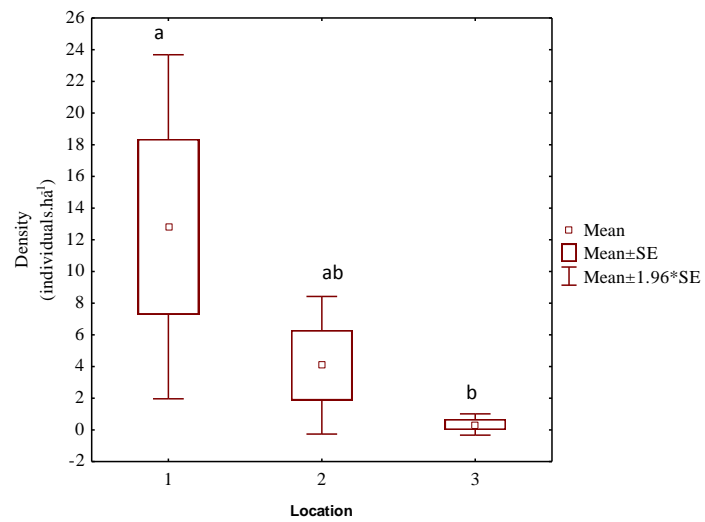
Figure 37. Density of Prunus species individuals as a function of patch location. 1=Urban, 2=Peri-urban, 3=Rural.
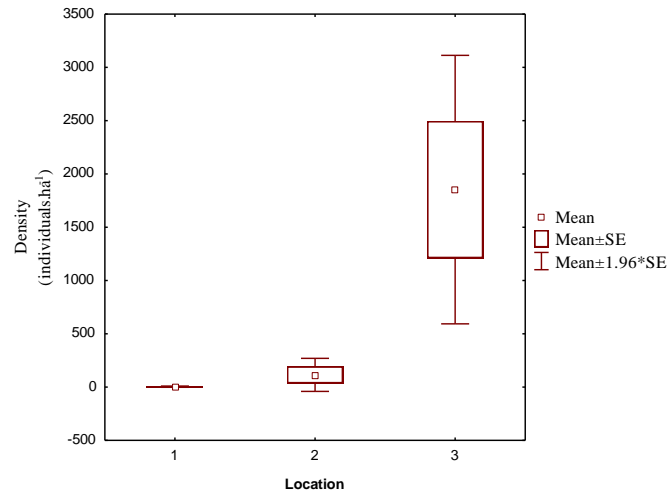
Figure 39. Density of *Ligustrum sinense* by patch location.
  1=Urban, 2=Peri-urban, 3=Rural.

Figure 38. Density of *Ligustrum sinense* by patch fencing time.

Figure 41. Density of Ligustrum sinense by patch management effort.
1=None, 2=Low, 3=Medium, 4=High.
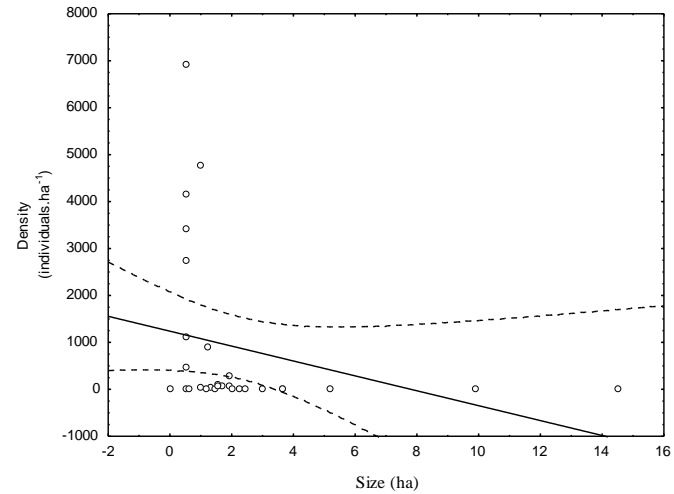
Figure 40. Ligustrum sinense density by patch size (ha).

3.3.4 *Ligustrum sinense* phenology

The average proportion of *Ligustrum sinense* individuals with flowers or fruits in a given patch was 16.45 (σ = 31.85) and was only significantly correlated with management effort (r = -0.381, p<0.05) with lower proportions of reproductively mature *Ligustrum sinense* individuals in patches with medium and high management effort than patches with low and no management effort (Figure 42).
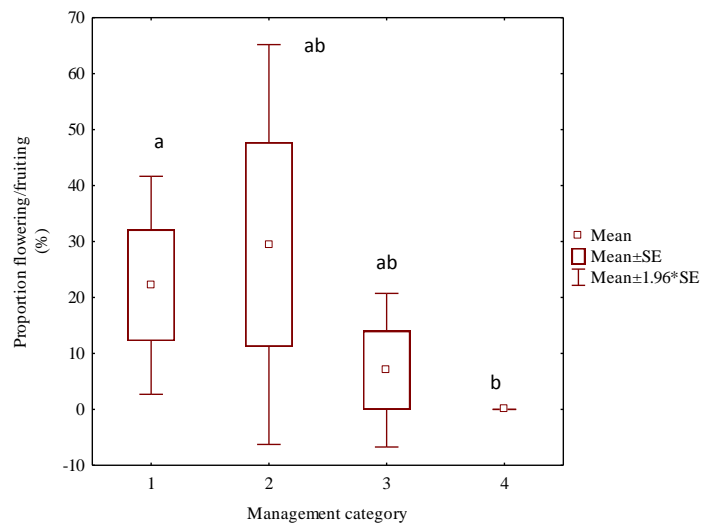
Figure 42. Proportion of *Ligustrum sinense* individuals flowering/fruiting categorised by management effort. 1=None, 2=Low, 3=Medium, 4=High.

Although not significantly correlated with patch location or native vegetation condition, no flowering or fruiting individuals were found in urban patches or patches with good native vegetative cover (Figures 43 and 44).
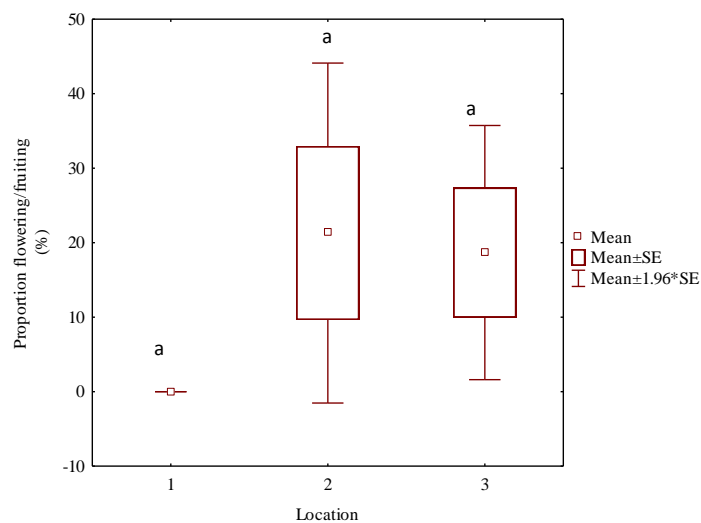
Figure 43. Proportion of Ligustrum sinense individuals flowering/fruiting as a function of patch location. 1=Urban, 2=Peri-urban, 3=Rural.
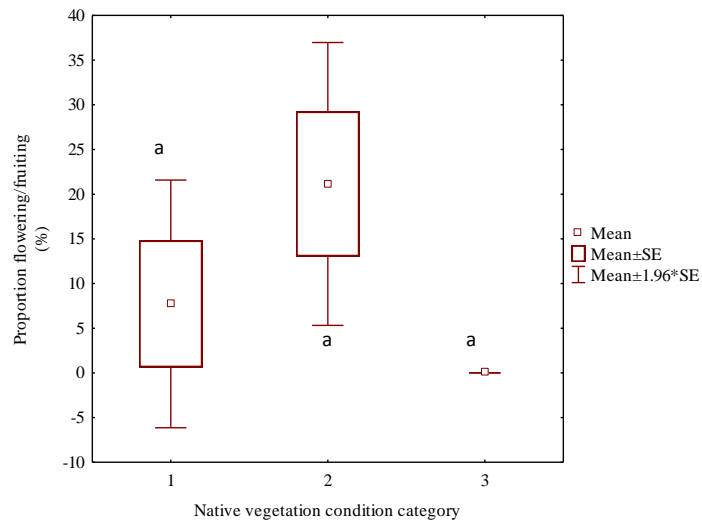
Figure 44. Proportion of *Ligustrum sinense* individuals flowering/fruiting as a function of patch vegetation condition. 1=Poor, 2=Moderate, 3=Good.

Additionally, there was no significant difference in the size of individuals with fruit or flowers than individuals that were not reproductively mature with averages of 3.31 (σ = 3.22) and 3.03 (σ = 1.84) centimetres diameter respectively (Figure 45).
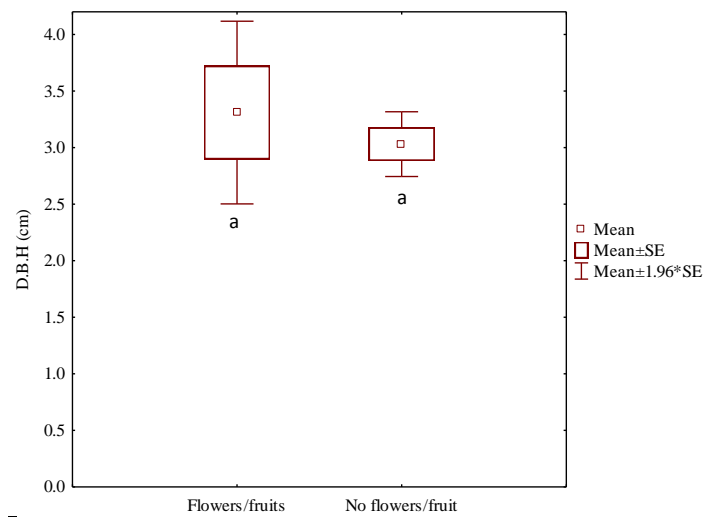
Figure 45. Diameter of *Ligustrum sinense* individuals as a function of reproductive maturity.

3.4 Discussion

Although woody weeds are widespread throughout the Waikato countryside and few forest patches are immune from invasion, only one widespread species in this study stands out as an immediate problem for kahikatea forest patch restoration and conservation - *Ligustrum sinense* (Chinese privet). Observational data from this research indicate that *Solanum pseudocapsicum*, *Solanum nigrum* and *Berberis glaucocarpa* were all restricted to patches with low native vegetative cover in the tiers of forest most affected by grazing from stock and human-related disturbance (ground, understorey and sub-canopy); therefore management focussed on the closure of these tiers and reduction in disturbance frequency will most likely result in exclusion of these species from forest patches. Chinese privet (*Ligustrum sinense*) and flowering cherry species (*Prunus* species) on the other hand, are species that have the potential to grow in closed-canopy forests and may therefore inhibit native forest restoration goals. Their potential effects are outlined in the remainder of this discussion.

3.4.1 Flowering cherry (Prunus species*)*

Patches affected and risk of spread

This study indicates that *Prunus* species are, as yet, restricted to Waikato kahikatea patches located in the urban environment, most likely due to their strong presence as ornamental species in residential gardens and as street trees. However, in spite of currently being restricted in their spread and population density in kahikatea forest patches, *Prunus* species have the potential to become more widespread, principally due to their attractiveness and dispersal by avian frugivores (Deckers *et al.*, 2008). Specifically, *Turdus merula* (European blackbird) is an important frugivore and dispersal agent of *Prunus serotina* in Belgium (Deckers *et al.*, 2008) and has similarly been shown to consume and disperse a wide range of indigenous, native and adventive species in New Zealand (Williams, 2006; Williams and Karl, 1996), where it may transport seed up to one kilometre or more to potentially develop new invasive loci (Williams, 2006).

Furthermore, several flowering cherry species have been identified as potential environmental weeds either in New Zealand or overseas: *Prunus serotina* (Europe), *P. serrulata* and *P. campanulata* (New Zealand) (Deckers *et al.*, 2005;

Howell, 2008). *P. serotina* is particularly invasive in European forests and agricultural landscapes where it reduces native floristic diversity and in some cases changes floristic community structure, with young forests on coarse-textured, dry soils identified as particularly susceptible to invasion by this species (Deckers *et al.*, 2005; Verheyen *et al.*, 2007). The concentrated presence of this species in agricultural hedgerows and its dispersal by birds are the main factors implicated in its invasive success (Deckers *et al.*, 2005).

In New Zealand, *Prunus serotina* is listed on the National Pest Plant Accord (NPPA) as a potential environmental weed due to its ability to invade intact forest and it's toxicity to domestic stock (Ministry of Agriculture and Fisheries, 2008). However, *P. serotina* is not listed in Howell's (2008) consolidated list of environmental weeds and only Taiwan cherry (*Prunus campanulata*) is recorded in any of the sixteen existing Regional Pest Management Strategies (RPMS) (Table 14, Appendix 7). Taiwan cherry is listed in Northland and Auckland's pest management strategies as a plant of potential biodiversity concern that requires further research due to its strong presence in gardens and as street amenity plantings, its high capacity for spread via bird dispersed seed or vegetatively via suckering and its ability to colonise bush margins, canopy gaps and clearings, where it competes with regenerating native plants (Auckland Regional Council, 2007).

In spite of the absence of *Prunus* species from pest management strategies in other regions, I believe that land owners, conservation managers and regional councils should incorporate *Prunus* species into their management plans in order to contain these potential pest plants and halt their spread. However, further research is required to establish the degree to which the species identified here are capable of inhibiting New Zealand native biodiversity goals.

Control

Effective control methods depend on the growth stage of the plant. Hand-pulling is effective for low-density infestations of seedlings, and herbicide or mycoherbicide (fungal herbicide) application to cut stumps can be utilised for older, woodier plants. A mycelium suspension of *Chondrostereum purpureum* (a basidiomycete responsible for silver leaf disease in fruit trees) is being trialled in

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Europe to control re-sprouting of cut stumps of *Prunus serotina* and has been made commercially available (BioChon™/Chontrol™) (De Jong, 2000).   This fungal herbicide promotes wood decay and in trials to date has an effectiveness rating of *c.*95% kill of stumps after two years; however, there is a risk of infection to neighbouring trees with fresh wounds as this is the entry point for the fungus (De Jong, 2000). *Chondrostereum purpureum* is being trialled in New Zealand as a potential control agent for non-native woody weeds in pastures and its commercial product, BioChon™/Chontrol™, is undergoing registration in New Zealand for trials in the large-scale control of gorse (*Ulex europaeus*) and broom (*Cytisus scoparius*) (Landcare Research, 2001).

Plate 3. *Prunus campanulata*.

3.4.2 Chinese privet (Ligustrum sinense)

Patches affected and risk of spread

In this study *Ligustrum sinense* was most abundant in rurally located forest patches and patches with low to medium management effort, and was least dense in patches fenced for long periods of time, urban patches, patches with high management effort and patches greater than two hectares in area. However, the potential for ecology related management conclusions are slim because most of the patches that are large are also those that have high management effort expended upon them and have been fenced for the longest period of time. However, these patches were probably identified early as patches of significant natural importance and therefore are likely to have had low weed density from the beginning.   Therefore, none of the patch characteristics listed here are good

indicators of low *Ligustrum sinense* density and subsequently are not good indicators of resistance to invasion.

Though *Ligustrum sinense* was found in all forest patches irrespective of indigenous vegetative condition, none of the privet individuals found in patches of good vegetative condition showed signs of reproductive maturity in spite of being of sufficient height and diameter to be reproductively mature in other stands. This indicates that whilst good vegetative condition of a forest may not prevent invasion by *Ligustrum sinense*, the presence of deeper shade cast by multiple layers of forest may not allow the sequestration of enough energy by *Ligustrum sinense* to become reproductively mature and this may therefore limit its spread within the forest. This observation is supported by Morris *et al.* (2002) who found that *Ligustrum sinense* individuals growing in high light conditions (glade edges) produced four times more flowers and fruits per ramet than *Ligustrum sinense* individuals growing in low light (woodland) conditions. Similarly, an Australian study showed that fruit production in Chinese privet, per unit canopy area, was progressively reduced as degree of shading increased (Westoby *et al.*, 1983).

Ligustrum sinense is not listed on the NPPA but has been identified in many other countries as a weed of concern for indigenous biodiversity due to its rapid growth, wide environmental tolerance, and dispersal by indigenous and exotic bird species (Matlack, 2002; Merriam and Feil, 2002). In New Zealand, *Ligustrum sinense* was probably brought into the country in the early to mid 1800s as hedging for farms and as an ornamental plant but was only recorded as naturalised around 1950 (though it probably established earlier) (Esler, 1988c). Though not present in the NPPA, it is listed in 9 of the 16 existing regional pest management strategies as a species of concern for biodiversity and/or human health (Table 14, Appendix 7) and is present on Howell's (2008) consolidated list of environmental weeds. In the Waikato region, *Ligustrum sinense* is still widespread in agricultural hedges and, as this study demonstrates, is abundant in native forest patches (*pers. obs.*).

Although *Ligustrum sinense* grows and establishes most quickly in disturbed high-light environments, it can also establish successfully in closed-canopy communities due to its high tolerance of low light levels, and can form

monospecific stands excluding all others (Grove and Clarkson, 2005; Morris *et al.*, 2002). Other attributes that contribute to its success include moderate ecological versatility, quick maturity, prolific seeding, efficient dispersal, fast recovery rate from coppicing, and strong competitive ability (Esler, 1988b). *Ligustrum sinense* has received less press in New Zealand as a potential environmental weed than other *Ligustrum* species, in part due to its smaller stature and shorter life-cycle (Esler, 1988c). However, it is just as widespread across the country and has the potential to be just as damaging as larger stature family members (for example *Ligustrum lucidum*) (Esler, 1988c).

Control

Common practice for privet management in New Zealand is to hand-pull seedlings or to cut individuals and paste the stumps with strong herbicide; for example any herbicide with either metsulfon or glyphosphate as the active ingredients (Escort®, Answer®, Vigilant® or Roundup®). However, both of these techniques are time and resource intensive and the literature is in-conclusive on the best chemical to use. Additionally, where large infestations occur, the battle with privet can seem insurmountable, and cutting of a stump can induce re-sprouting (Munger, 2003). Numerous international studies have attempted large-scale solutions including use of a foliar spray and water-logging with mixed results (Brown and Pezeshki, 2000; James and Mortimer, 1984; Miller, 1998; Mowatt, 1981) (Appendix 8).

Flooding

Brown and Pezeshki (2000) found that short-term flooding resulted in an 80% reduction in net photosynthesis and stomatal conductance and a reduction in height and diameter growth compared with non-flooded plants. However, recovery of gas exchange was observed after 20 days and treatment plants showed morphological adaptations (lenticels and adventitious root development) after three weeks of treatment.

Foliar sprays

The use of foliar sprays is advantageous because it is time and resource efficient and can be used for large areas and dense infestations. However, there is always the risk of non-target damage and the potential for persistence of the chemical in the soil (Cox, 2004). Of the multiple herbicides, application rates and timing of

application that have been tested, glyphosate application in spring emerges as the most consistent common chemical employed for privet control and need not be used in high concentrations (Harrington and Miller, 2005). 1.7 kg ae/ha applied in spring or autumn can reduce privet foliar cover by 93-100% of non-treatment individuals (Harrington and Miller, 2005) and application 0.36 a.i. % w/v in spring resulted in 87% mortality of treated individuals (James and Mortimer, 1984).

However, in spite of being marketed as benign (Harrington and Miller, 2005), laboratory studies have found adverse effects in all standard categories of toxicology testing including long- and medium-term toxicity, genetic damage, reproduction effects and carcinogenicity (Cox, 2004). Additionally, reduction in populations of beneficial insects, birds and small animals, increased susceptibility to disease, glyphosate resistance and half-lives of between 50 and 150 days have been reported (Cox, 2004).

Shading

There is evidence that deep shade reduces *Ligustrum sinense* fruit production (Morris *et al.*, 2002; Westoby *et al.*, 1983) and may reduce trunk number thereby decreasing shrub density (Esler, 1988a). Additionally, once dispersed, seeds are only viable up to 1 year after release from the parent tree (Grove and Clarkson, 2005; Panetta, 2000). Therefore, it is possible that if surrounding seed sources can be removed, high density populations of Chinese privet growing in deep shade may become self-thinning and eventually become manageable. However, given the prevalence of this weed throughout the landscape, complete eradication is unlikely. Rather, where large-scale herbicide application is acceptable, this method may prove very effective. In most cases however, hand pulling and cutting and pasting of stumps is still the most effective method for forest fragments.

Plate 4. A forest patch with an understorey composed of *Ligustrum sinense*.

3.5 Conclusions

The conclusions of this study support previous research on weed species richness, which indicate that high weed species richness is more likely to be found in forest patches located closer to urban environments (Sullivan *et al.*, 2005; Timmins and Williams, 1991), patches with low canopy and subcanopy cover (Teo *et al.*, 2003) and patches fenced for short periods of time (Smale *et al.*, 2005).

Woody weed species distribution and density however, reflects the historical use of the species. For example, *Ligustrum sinense* density is generally higher in rural patches, which likely reflects the use of this plant as a hedgerow species. Though capable of reproducing and persisting in shaded environments, the density of *Ligustrum sinense* seems to be negatively correlated with high native species cover, thereby limiting the effect of *Ligustrum sinense* on native biodiversity. *Prunus* species on the other hand, tend to occur more frequently, and at higher densities in the urban environment. This is likely a result of its common usage in gardens and as street trees. As yet however, the spread and impact of this species in the Waikato is low and consequently, there are too few data points available to make inferences about patch characteristics that may, or may not, influence *Prunus* species growth and spread.

Due to the prevalence of *Ligustrum sinense* in the Waikato dairying landscape, the most effective course of action is persistence with current control techniques and vigilance in identifying new invasion points. The spread of *Prunus* species however, has the potential to be contained due to its current low density and spread in the landscape and as such, should be brought to the attention of land holders and local government departments.

CHAPTER 4 –

Conclusions and management recommendations

4.1 CONCLUSIONS

The principle aim of this study was to determine whether fencing alone is a sufficient management tool for facilitating the recovery and persistence of indigenous flora in kahikatea-dominated forest patches in the Waikato region. The results demonstrate that, while fencing of a patch and time for native vegetation recovery are important factors in promoting native biomass recovery and ecosystem composition, the combination of patch size, distance of a patch from a main road, and patch location were better predictors of the observed variation in native species cover than fencing time alone; particularly in the layers most affected by grazing. This study indicates that patches less than seven hectares in area, regardless of location, will require continued human intervention to ensure their persistence; and patches in urban areas, irrespective of size, may never become self-sustaining. This result is supported by previous data on native forest species' condition that indicates an area of nine hectares is necessary for amelioration of the high disturbance regime associated with edge effects (Davies-Colley *et al.*, 2000; Young and Mitchell, 1994), transport corridors act as vectors for invasion by exotic species (Forman and Alexander, 1998; Timmins and Williams, 1991; Von der Lippe and Kowarik, 2007), and rurally located patches are more likely to be located in close proximity to larger tracts of forest and therefore larger species pools.

Chapter two specifically focussed on the effects of management regime and patch location along an urban-rural gradient on forest patch condition. Answers to the key research questions are as follows:

> 1) Patches located in urban environments do appear to have lower indigenous species diversity and slower indigenous species recruitment rates than their peri-urban and rural counterparts in spite of the urban study patches having been fenced for longer periods of time. However, extrapolation of this data is inadvisable due to the small number of patches located in urban environments used in this study (five patches);

2) High management effort does not result in greater indigenous species diversity and recruitment and lower adventive species cover and diversity than lower management effort regimes. In fact patches characterised by high management effort were generally only in better vegetative condition than patches that received no management effort;

3) While native species were still present in patches that did not receive management effort even after 40 years of fencing, the general trend indicated by the results was towards a forest patch characterised by an indigenous canopy with an understorey and ground cover dominated by adventive species, and;

4) Kahikatea forest patches closer to main roads did tend to contain higher numbers and covers of adventive species and fewer native species than patches further from main roads; however, this was not statistically significant.

Furthermore, the results indicate that with medium and low levels of management, 15-20 years of fencing represents an important stage where exotic species cover drops below 5%, a manageable level. Although all levels of management effort above zero reduced the impact of adventive species in the patches, low management effort was the most successful. It may be that regular disturbance by high management effort promotes adventive species growth whereas irregular weeding gives the chance for at least annual plants to be removed from the system by being shaded out. Alternatively, it may be that patches with low management effort are so characterised because they were less degraded from the outset.

Chapter three specifically investigated the impacts of woody weed species invasions in kahikatea forest patches and answers to the key research questions are as follows:

1) Woody weed species density and richness was significantly correlated with patch location and fencing time. Specifically, woody weed species richness was lower in rurally located kahikatea patches than urban or peri-urban patches; and woody weed density tended to decrease over time.

2) Intensive weed control (that is, high management effort) does not seem to be most effective management strategy with respect to the woody weed species

found in kahikatea-dominated patches in the Waikato. Rather, selective removal of shade-tolerant woody weeds from around regenerating native species appears the most effective method of encouraging native species regeneration and recruitment. The time-frames involved in the study patches were insufficient to determine whether populations of woody weeds will undergo self-thinning.

3) There was a slight negative correlation between indigenous species richness and adventive species richness but this was not significant.

To fully determine the effects of adventive species on kahikatea forest patches, the life history of adventive species needs to be taken into account and more detailed histories of grazing and management are needed. When combined with the native species information, low to medium levels of management appear to be the most effective methods that should be employed depending on location. For example, urban patches exist in a highly modified environment with large pools of exotic species propagules and so require intensive management if they are to persist over time let alone reach a self-sufficient state. Patches located in the peri-urban environment are increasingly being impacted by sub-division but are capable of swift regeneration with moderate to low management with 30 years representing a crucial turning point for native species recovery. Finally, rurally located patches appear to be most at risk from serious environmental weeds but also benefit from generally being larger, further from potential disturbance and closer to native seed sources with 25 to 30 years representing a turning point for native species recovery.

Underlying the interacting effects of location, management and fencing time is the effect of patch size, a confounding factor which imposes a "major limitation [on] the detailed analysis of the different variables [due to] the strong correlation between patch size and condition" (Hobbs, 2001, p. 1525). Even so, without some management effort applied to kahikatea forest patches, their future is certainly bleak. Fencing off a patch, at least one smaller than 7 hectares, and letting nature run its course is not sufficient to ensure their survival as indigenous dominated forest patches into the future and it is unlikely that in today's weedy, fast-changing environment, human intervention will ever be redundant. Patches greater than seven hectares may become self-supporting, however continued monitoring is necessary to indicate whether this statement will be supported.

Furthermore, each patch will have different species composition depending on the length of time since fragmentation, the length of time it has been grazed, how far it is from native seed sources and its surrounding landscape use. Consequently, the trajectory of floristic change will be different for each patch. Overall however, a management regime intermediate between low and medium management effort with planting of indigenous buffers to reduce edge effects and targeted weed management may be sufficient to reduce adventive species' impact to manageable levels after only 20 years of fencing. Similarly, such management should facilitate native plant species community recovery within 30 years of fencing irrespective of a patches initial state.

4.2 Management recommendations

1) **Reducing edge effects:**

All patches irrespective of size would benefit from the planting of vegetatively dense native species such as flax around the margins. This is particularly beneficial for patches situated in close proximity to roads and patches situated in urban environments. Patches less than 3 hectares in size should have all margins planted to effectively increase patch size and reduce the drying and disturbance effects of wind and solar radiation. Larger patches (>7 ha) would benefit from planting on the edge that experiences the prevailing wind and weather conditions. In the Waikato, this is usually the west to north-west edges (New Zealand Meterological Service, 2009).

Though planting solely native species is often advised, supplementation of the native species planted with exotic species that attract birds may facilitate plant species regeneration (Meurk and Swaffield, 2000). Species such as tree lucerne (*Chamaecytisus palmensis*) and banksia (*Banksia* species) attract important pollinator species such as tui and bellbird to forest patches (Meurk and Swaffield, 2000). However, if this strategy is utilised, restoration workers should obtain advice from qualified personell to ensure non-invasive species only are planted.

2) Keeping (or making) patches characteristic of kahikatea-dominated forest:

Identify species from the list of characteristic kahikatea-dominated forest (Appendix 2) that are missing in your patch and plant accordingly. Where possible, source species locally. If possible, block drains around the forest patch to increase the residence time of water within the forest system. This will improve kahikatea regeneration and will also effectively control many weed species. Without regular flooding, kahikatea are unlikely to recruit into the sapling stage and forest patches are likely to change to a forest dominated by broadleaved species such as tawa and titoki (Champion, 1988; Whaley *et al.*, 1997).

3) Adventive species management:

Identify adventive species that have the potential to alter ecosystem function and focus on their removal. For example, *Ligustrum sinense, L. lucidum, Berberis darwinii, B. glaucocarpa.* For herbaceous species other than *Tradescantia fluminensis*, only weed around regenerating or planted native species. The best method for controlling *Ligustrum sinense* is still cutting of trees/saplings and pasting with herbicide or handpulling of seedlings. However, where privet infestation is very dense, concentrating on the removal of individuals around regenerating native species is quite effective and much less time-consuming.

Plate 5. Whewell's bush southern boundary.

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6.0 Appendices

Appendix 1 – Species richness data

Table 9. Summary of patch species richness descriptive statistics (n=28).

	Mean	Minimum	Maximum	σ
Native Species Richness	27.500	8.000	52.000	11.857
Exotic Species Richness	8.808	1.000	36.000	7.725
Total species #	36.308	13.000	69.000	14.551

Table 10. Summary of patch species richness descriptive statistics categorised by patch location (n=28).
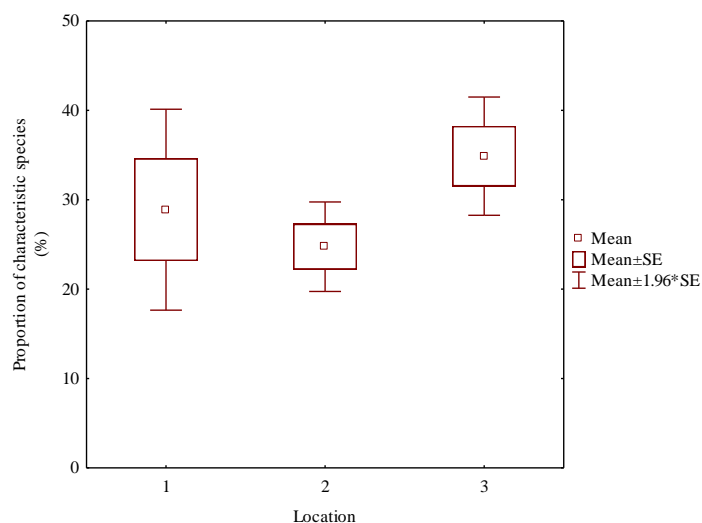
	Urban		Peri-urban		Rural	
	Mean	σ	Mean	σ	Mean	σ
Native Species Richness	27.600	10.479	23.636	8.535	31.700	14.930
Exotic Species Richness	12.600	8.234	9.273	9.403	6.400	4.766
Total species Richness	40.200	13.590	32.909	13.050	38.100	17.052

Table 11. . Spearman's rank order correlations of species richness per site and associated variables (n=28). Values in bold are significant at p<0.01, values not in bold are significant at p<0.05 and non-significant factors show the direction of the correlation

	Native Species Richness	Exotic Species Richness	Total Species Richness
Native Species Richness			
Exotic Species Richness	-		
Total species Richness	**0.877**	0.419	
Location	+	-	-
Age	0.431	-	+
Management	+	+	0.392
Size (ha)	0.412	-	+
Road	+	-	-
Town	+	-	-

Appendix 2 – Characteristic species list

Alectryon exclesus

Asplenium bulbiferum

Astelia fragrans

Astelia grandis

Beilschmiedia tawa

Carex dissita

Carex lambertiana

Collospermum hastatum

Coprosma areolata

Coprosma grandifolia

Coprosma rotundifolia

Cordyline australis

Cyathea dealbata

Dacrycarpus dacrydioides

Dacrydium cupressinum

Dicksonia fibrosa

Dicksonia squarrosa

Elaeocarpus hookerianus

Freycinetia banksii

Geniostoma ligustrifolium var. *ligustrifolium*

Hoheria sexystylosa

Hymenophyllum demissum

Knightia excelsa

Laurelia novae-zelandiae

Melicope simplex

Melicytus micranthus

Melicytus ramiflorus subsp. *ramiflorus*

Microlaena avenacea

Microsorum scandens

Myrsine australis

Myrsine divaricata

Oplismenus imbecillis

Pittosporum tenuifolium

Plagianthus regius

Prumnopitys taxifolia

Ripogonum scandens

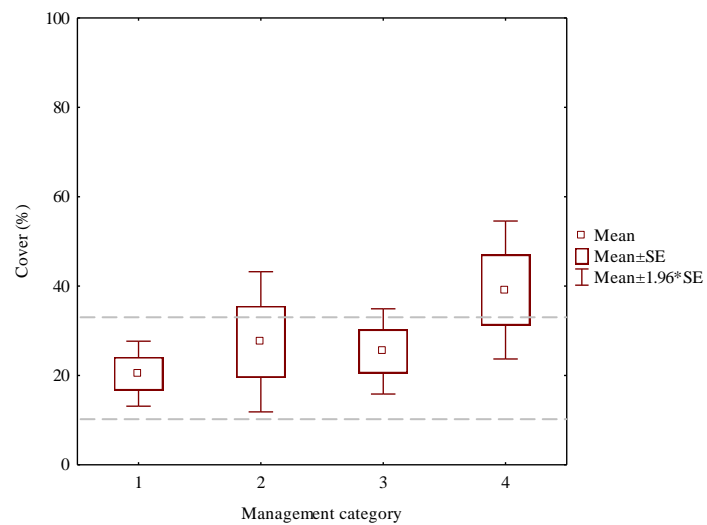*Schefflera digitata*

Streblus heterophyllus

Syzygium maire

**Figure 46. Percentage of characteristic species categorised by location. 1=Urban, 2=Peri-urban, 3=Rural.**

Appendix 3 – Effect of management regimes excluding urban patches
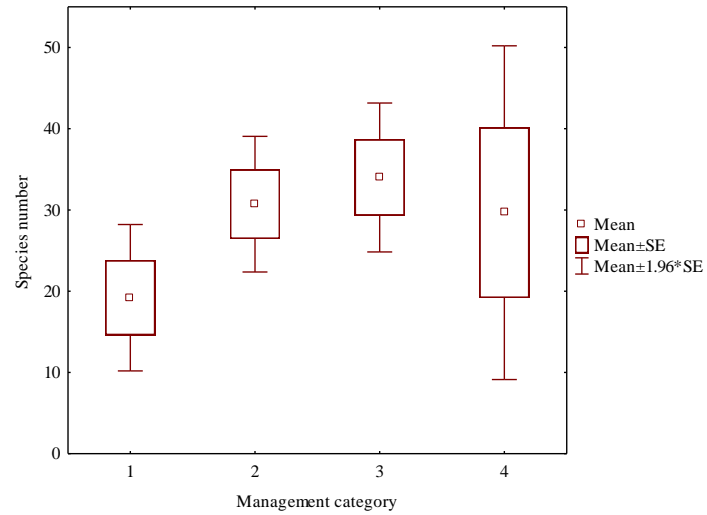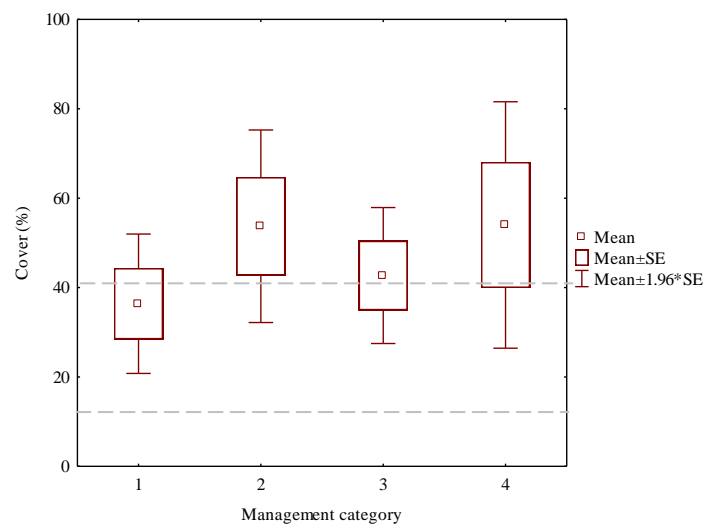

Figure 47. Native species richness scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.
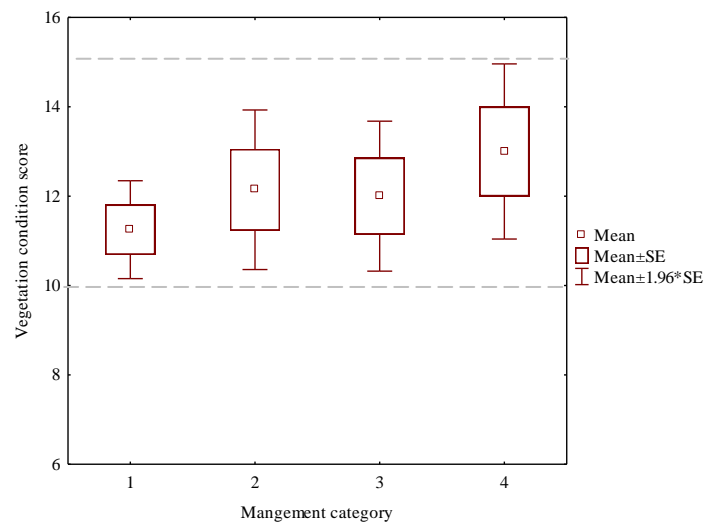

Figure 48. Native species understorey cover scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.
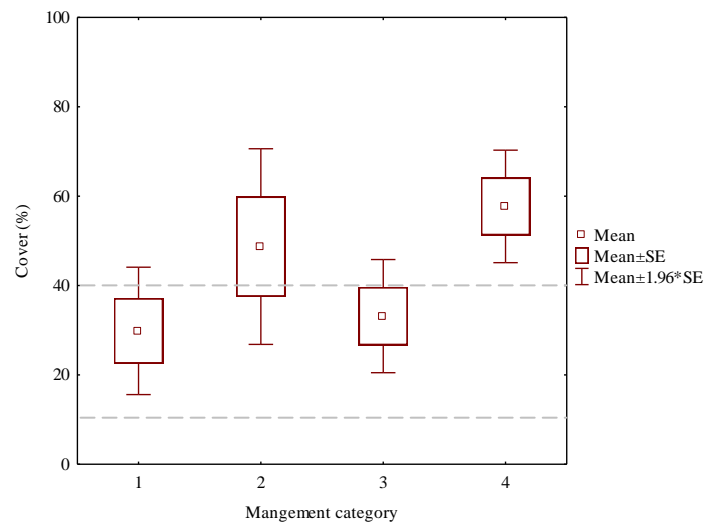
Figure 49. Native species shrub-layer cover scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.

Figure 50. Native species ground-cover cover scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.

Figure 51. Patch vegetation condition scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.

Figure 52. Proportion of kahikatea characteristic species in study forest patches as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.

Figure 53. Exotic species richness scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.

Figure 54. Exotic species understorey cover scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.
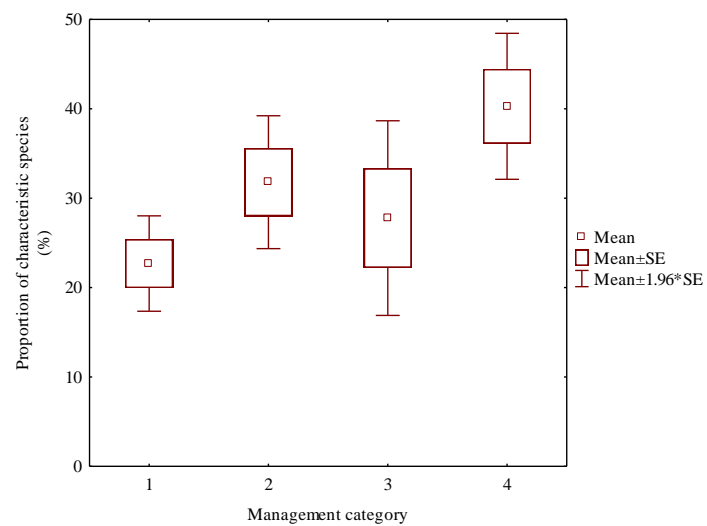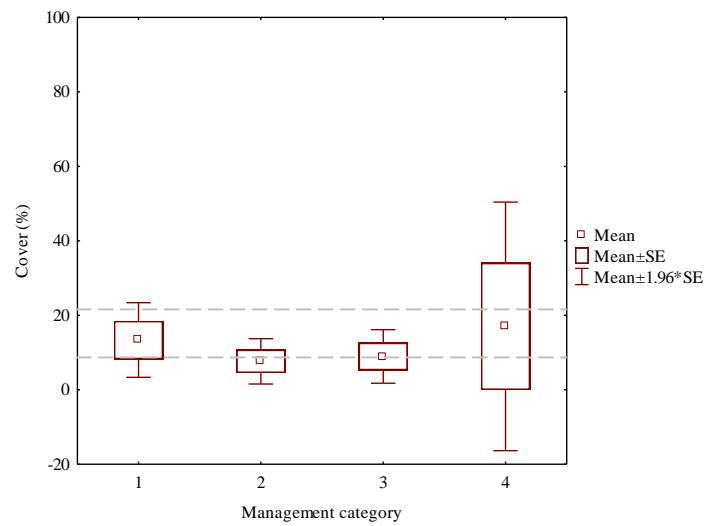
Figure 55. Exotic species shrub-layer cover scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.
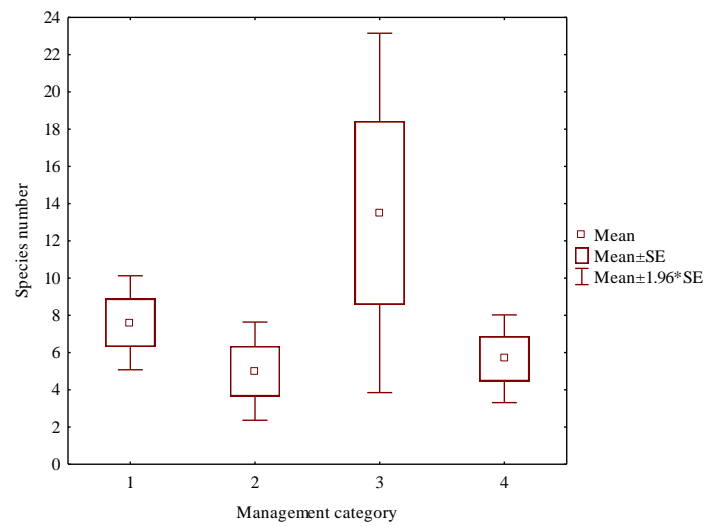

Figure 56. Exotic species ground-cover cover scores as a function of management effort. 1=None, 2=Low, 3=Medium, 4=High.
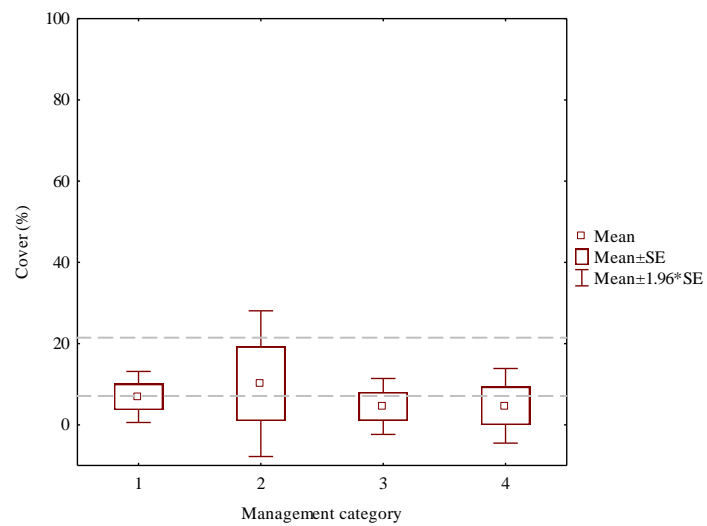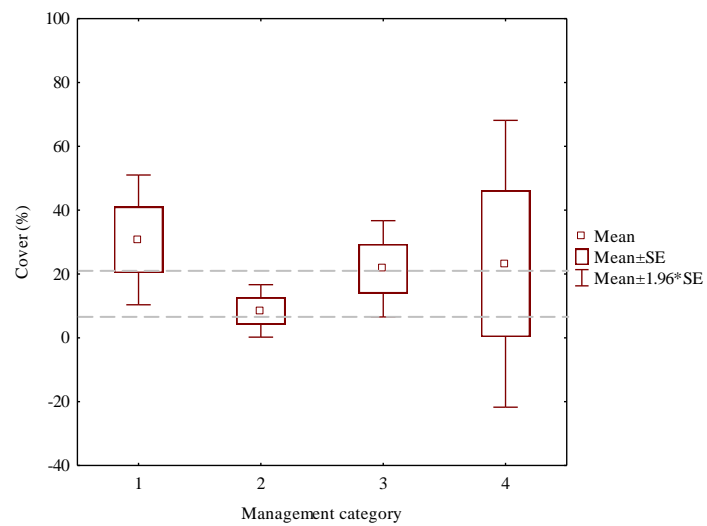
Appendix 4 – Woody weed species list

Barberry	*Berberis glaucocarpa*
Black nightshade	*Solanum nigrum*
Blackberry	*Rubus fruticosus*
Chinese privet	*Ligustrum sinense*
Chinese windmill palm	*Trachycarpus fortunei*
Spindle tree/Euonymus	*Euonymus japonicus*
Fatsia/Japanese Aralia	*Fatsia japonica*
Flowering cherry species	*Prunus* species
Ivy	*Hedera helix*
Jerusalem cherry	*Solanum pseudocapsicum*
Phoenix palm	*Phoenix canariensis*
Velvety nightshade	*Solanum chenopodioides*
Walnut	*Juglans* species
Woolly nightshade	*Solanum mauritianum*

Appendix 5 – Top 5 Woody Weeds Species Correlations

Table 12. Top five woody weed species statistical correlations.

	Ligustrum sinense	*Solanum pseudocapsicum*	*Prunus* species	*Berberis glaucocarpa*	*Solanum nigrum*
Ligustrum sinense					
Solanum pseudocapsicum	-				
Prunus species	-	+			
Berberis glaucocarpa	-	+	+		
Solanum nigrum	-	+	+	0.423	
Vegetation condition	-	-	-	+	-
Location	**0.528**	-	**-0.573**	-	-
Fencing time	-0.449	-	+	+	+
Management	-0.387	-	+	+	+
Size (ha)	**-0.516**	+	+	+	+

Appendix 6 – Pest Plants of Forest Ecosystems and their legislative and pest status

Table 13. Pest plants of forest ecosystems, their growth form, regional and national legislative status and presence in the ISSG database.

Species	Growth form	Waikato RPMS status	NPPA	ISSG database
Acmena smithii Monkey apple	Tree	Potential	Yes	No
Ageratina riparia Mistflower	Perennial herb	Containment	No	Yes
Anredera cordifolia Mignonette/Madiera vine	Liane	Containment	Yes	Yes
Araujia sericifera Moth plant	Liane	Containment	Yes	No
Asparagus asparagoides Smilax	Liane	Nuisance	Yes	No
Asparagus scandens Climbing asparagus	Liane	Containment	Yes	No
Asparagus setaceous Ferny asparagus	Liane	Potential	No	No
Berberis darwinii Darwin's barberry	Shrub	Potential	Yes	No
Bryonia cretica ssp. *dioica* White Bryony	Liane	Eradication	Yes	No
Buddleja davidii Buddleia	Shrub	Nuisance	No	Yes
Celastrus orbiculatus Climbing spindleberry	Liane	Eradication	Yes	Yes
Clematis vitalba Old man's beard	Liane	Eradication	Yes	Yes
Cobaea scandeis Cathedral bells	Liane	Nuisance	Yes	No
Cotoneaster glaucophyllus Cotoneaster	Shrub	Nuisance	No	No
Dipogon lignosus Mile-a-minute	Liane	Nuisance	Yes	Yes
Elaeagnus x reflexa Eleagnus	Shrub	Nuisance	No	No
Gunnera tinctoria Chilean rhubarb	Herb	Nuisance	Yes	Yes
H. flavescens Yellow ginger	Perennial herb	Containment	Yes	Yes
Hedychium gardnerianum Kahili ginger	Perennial herb	Containment	Yes	Yes
Ipomoea indica Blue morning glory	Liane	Nuisance	Yes	No
Juglans ailantifolia Japanese walnut	Tree	Nuisance	No	No
Lantana camara var. *aculenta* Lantana	Shrub	Nuisance	Yes	Yes
Leycesteria formosa Himalayan honeysuckle	Liane	Nuisance	No	No
Ligustrum lucidum Tree privet	Tree	Containment	No	Yes
Ligustrum ovalifolium Oval-leaved privet	Shrub	Containment	No	No
Ligustrum sinense Chinese privet	Shrub	Containment	No	Yes

Species	Growth form	Waikato RPMS status	NPPA	ISSG database
Ligustrum vulgare Common privet	Shrub	Containment	No	Yes
Lonicera japonica Japanese honeysuckle	Liane	Nuisance	Yes	Yes
Passiflora mixta Banana passionfruit	Liane	Nuisance	Yes	Yes
Passiflora mollissima Banana passionfruit	Liane	Nuisance	Yes	Yes
Pennisetum macrourum (African feather grass)	Perrenial grass	Eradication	Yes	Yes
Rhamnus alaternus Buckthorn	Shrub	Eradication	Yes	Yes
Rhododendron ponticum	Shrub	Potential	No	No
Salix cinerea Grey willow	Tree	Nuisance	Yes	Yes
Salix fragilis Crack willow	Tree	Nuisance	No	No
Senecio angulatus Cape ivy	Liane	Nuisance	No	No
Senecio mikanioides German ivy	Liane	Nuisance	No	Yes
Solanum mauritianum Wooly nightshade	Tree	Containment	No	Yes
Tradescantia fluminensis Wandering jew	Herb	Nuisance	Yes	Yes
Tropaeolum speciosum Chilean flame creeper	Liane	Eradication	Yes	No
Ulex europaeus Gorse	Shrub	Containment	No	Yes

Appendix 7 – Woody weeds in this study and their status in New Zealand Regional Council Pest Management Strategies.

Table 14. Top five woody weed species and their status in regional council pest management strategies.

Species	Details
Berberis glaucocarpa	Auckland – Surveillance Gisborne – Containment Horizons – Site led Greater Wellington – Site led Canterbury – Restricted pest Chatham Islands – Pest
Ligustrum sinense	Northland – boundary control Auckland – Surveillance Waikato – Containment Bay of Plenty – Surveillance Gisborne – Limited control Hawkesbay – Eradication Greater Welllington – Site led Canterbury – Restricted pest Chatham Islands – Pest
Prunus species	*P. campanulata* – Total control Northland and Auckland
Solanum nigrum	None
Solanum pseudocapsicum	None

Appendix 8 – Chemical control of privet

Table 15. Summary table of chemical control methods for *Ligustrum sinense*.

	Treatment method	Glyphosphate	Triclopyr	Imazapyr	Metsulfuron	Hexazinone	Picloram	Ticlopyr/ picloram
Mowatt (1981)	Injection	Variable	Consistently high			Consistently high		
Miller (1998)	Foliar spray	>90% Late summer-early spring	64% Late summer-early spring	>90% Late summer-early spring	89% Late summer		12% Late summer-early spring	
					73% Early spring			
	Cut and paste							92% @ 1/0.25 a.i. % w/v
James and Mortimer (1984)	Foliar spray Spring	87% @ 0.36 a.i. % w/v				35% @ 0.36 a.i. % w/v		8% @ 0.2/0.05 a.i. % w/v
		100% @ 0.72 a.i. % w/v				85% @ 0.72 a.i. % w/v		30% @ 0.4/0.1 a.i. % w/v
	Foliar spray Autumn	20% @ 0.49 a.i. % w/v	0% @ 0.26 a.i. % w/v					35% @ 0.27/0/07 a.i. % w/v
		15% @ 0.72 a.i. % w/v	5% @ 0.4 a.i. % w/v					38% @ 0.4/0.1 a.i. % w/v
Harrington and Miller (2005)	Foliar spray Spring and autumn	93-100% cover control	49-70% cover control					
	Foliar spray Summer	67-69% cover control	14-26% cover control					

Appendix 9 – Species list for all patches

* indicates adventive species

Alectryon excelsus

*Allium triquetrum**

Anarthropteris lanceolata

*Arcctotheca calendula**

Aristotelia serrata

Arthropteris tenella

Asplenium bulbiferum

Asplenium flaccidum

Asplenium oblongifolium

Asplenium polyodon

Astelia grandis

*Aster subulatus**

Beilschmiedia tawa

*Berula erecta**

*Berberis darwinii**

*Berberis glaucocarpa**

*Bidens frondosa**

Blechnum chambersii

Blechnum discolor

Blechnum filiforme

Blechnum fluviatile

Blechnum novae-zealandiae

Blechnum penna-marina

Calystegis species*

*Cardamine hirsuta**

Carex dipsacea

Carex dissita

Carex geminata

Carex ochrosaccus

Carex secta

Carex virgata

Carex species

Carpodetus serrata

*Cirsium arvense**

*Cirsium palustre**

*Cirsium vulgare**

Clematis cunninghamii

Collospermum hastatum

*Conyza albida**

Convolvulus species

Coprosma areolata

Coprosma grandifolia

Coprosma propinqua

Coprosma propinqua x robusta

Coprosma rhamnoides

Coprosma robusta

Coprosma rotundifolia

Coprosma spathulata

Coprosma taylorii

Cordyline australis

Corynocarpus laevigatus

*Crataegus monogyna**

*Crepis capillaris**

*Crocosmia x crocosmiiflora**

Cyathea cunninghamii

Cyathea dealbata

Cyathea medularis

Cyathea smithii

Dacridium cuppressinum

Dacycarpus dacridioides

*Daucus carota**

Deparia petersenii

Dicksonia fibrosa

Dicksonia squarrosa

Diplazium australis

Doodia australis

Drymoanthus adversus

Dysoxylum spectabile

Earina mucronata

Elaeocarpus hookerianus

Epilobium pedunculare

*Euonymous japonicus**

*Euphorbia lathyris**

*Euphorbia peplus**

*Fatsia japonica**

Freycinetia banksii

Fucshia perscandens

*Galium aparine**

*Galeobdolon luteum**

Geniostoma ligustrifolium var. *ligustrifolium*

*Geranium robertianum**

Hebe stricta

Hedycarya arborea

*Hedera helix**

*Helminthotheca echioides**

Histiopteris incisa

Hoheria populnea

Hoheria sexystylosa

Hymenophyllum bivalve

*Hypochoeris radicata**

*Iris foetidissima**

Isolepis reticularis

Isolepis sububtilissima

Knightia excelsus

Kunzea ericoides

*Lapsana communis**

Lastreopsis glabella

Lastreopsis microsorum

Laurelia novae-zelandiae

*Ligustrum lucidum**

*Ligustrum sinense**

Litsea calicarus

*Lonicera japonica**

*Lycopus europaeus**

Macropiper excelsum

Melicytus micranthus

Melicytus ramiflorus

Melicope simplex

*Mentha x piperita var. piperita**

Metrosideros colensoi

Metrosideros diffusa

Metrodieros perforata

Microlaena avenacea

Microsorum pustulatum

Microsorum scandens

Muehlenbeckia australis

*Myosotis arvensis**

Myrsine australis

*Nandina domestica**

Nestegis cunninghamii

Nestegis lanceolata

Oplismenus hirtellus subsp. *imbecillus*

*Oxalis species**

Parsonsia capsularis

Parsonsia heterophylla

Passiflora tetranda

Pellaea rotundifolia

Pennantia corymbosa

Persicaria hydropiper*

Phoenix canariensis*

Phormium tenax

Phytolacca octandra*

Pittosporum colensoi

Pittosporum crassifolius

Pittosporum euginoides

Pittosporum tenuifolium

Plantago lanceolata*

Pneumatopteris pennigera

Podocarpus hallii

Podocarpus totara

Prunus species*

Prumnopitys ferruginea

Prumnopitys taxifolia

Pseudopanax arborea

Pseudopanax crassifolius

Psedopanax crassifolius x lessonii

Pseudopanax lessonii

Psedopanax lessonii x arborea

Pseudopanax lessonii x crassifolius

Pteridium esculentum

Pteris cretica*

Pteris tremula

Pyrrosia eleagnifolia

Ranunculus repens*

Rhopalostylis sapida

Ripogonum scandens

Rosa rubiginosa*

Rubus cissoides

Rubus fruticosus*

Rumex crispus*

Rumex obtusifolius*

Schefflera digitata

Sellaginella kraussiana*

Senecio bipinnatisectus*

Solanum chenopodioides*

Solanum mauritianum*

Solanum nigrum*

Solanum pseudocapsicum*

Sonchus asper*

Sophora microphylla

Stachys sylvatica*

Streblus heterophyllus

Taraxacum officinale*

Tradescantia fluminensis*

Trachycarpus fortuneii*

Trifolium species*

Uncinia unciniata

Zantedeschia aethiopica