Understory restoration in Hamilton urban forests

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Abstract

Research was undertaken to determine how the understory vegetation of Hamilton urban forests compares with reference old-growth forests in rural locations, identify causes for differences, and develop methods to enhance species diversity. Understory vegetation was measured in five rural old-growth forests and compared with 20 urban forests, categorised into four age groups, to assess differences in richness, composition and density. Environmental profiling quantified soil type, nutrient levels, pH, moisture content, understory light transmittance, temperature and vapour pressure deficit of selected forests to identify variation across the forest categories and determine if environmental conditions were the chief cause for vegetative differences. Three native species (*Melicytus micranthus*, *Hedycarya arborea* and *Coprosma arborea*), absent from or less abundant in urban forests, were reintroduced into forest sites to assess their growth, survival and potential for becoming a prominent component in Hamilton City forests.

Urban forests displayed reduced native understory diversity comprising only 61.5% of the native understory species found in the rural old-growth forests. Native understory species richness and density decreased from the rural old-growth forest category to the youngest urban forest group. Rural old-growth forests averaged 7.96 species and 41.28 stems per 50 m^2 compared to 2.68 species and 8.20 stems per 50 m^2 in the youngest urban forests. The exotic understory stem density trend was reverse. Reduced understory diversity in second-growth and urban forests is widely reported overseas but has not been quantified in New Zealand previously.

Soil nutrients and acidity increased from the youngest to the oldest urban forest category. Light transmittance into the understory decreased with forest age from 18.43% in the youngest urban forest group to 4.17% in the rural old-growth forest category during winter and spring. Buffering patterns were evident in rural old-growth forests with higher temperatures and vapour pressure deficits outside the forests by as much as 3 ºC and 0.3 kPa respectively during the day, compared to the forest interior. Similar patterns were evident in urban forests during spring but the interior temperatures and vapour pressure deficits were not as low, compared to the rural forests. Environmental profiling proved there were significant differences in environmental conditions between the forest categories and that these were within the range of values reported elsewhere in New Zealand.

Survival and growth rates between 77.8 to 100% and 2.7 to 12.1 cm respectively, for the three translocated species over the measured seven months were on par with other New Zealand trials and suggest the selected species can grow and establish viable populations within urban forests. The reintroduction success further indicates that the reduced diversity of urban forests is likely to result from the effects of fragmentation and isolation and urban pressures.

Active reintroduction of missing or less abundant native understory species is the best method to improve diversity in Hamilton urban forests. The species experimented with should be included in forest enrichment planting plans as early as 5 to 15 years. Management plans should address active removal of exotic species including methods for manipulation of developing vegetation to favour enhancement of native understory diversity in urban forests.

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1 Chapter one: Introduction

1.1 Introduction

Many remnant and restored urban forests are hotspots of biodiversity, providing an ecosystem for many types of plants and animals that would otherwise be unable to survive in such a human modified environment (Croci et al. 2008). Restoration projects are increasingly occurring in urban environments, providing an array of ecosystem services such as pollution filtration, cultural benefits and educational and recreational opportunities (Vidra et al. 2007). While urban forests are able to recover some of their features through the natural regeneration process, many components become completely lost from the ecosystem (Toniato & de Oliveira-Filho 2004). One such component is understory vegetation. Disturbed and restored forests generally contain reduced diversity of understory vegetation compared to what could potentially survive and grow within the selected urban forest (Lehvavirta & Rita 2002).

The development and dynamics of understory forest communities have been mostly ignored and instead study has focussed on the forest canopy and the more economically important large tree species (Small & McCarthy 2002). Old-growth forests often contain an understory diversity that is relatively distinct compared to second-growth forests or old-growth forests that have been adversely affected by anthropogenic activities (D'Amato et al. 2009). Old-growth forests contain more sensitive understory species, which are often the first species to become extinct in forests that have undergone extensive disturbance (Spyreas & Matthews 2006). What species naturally regenerate in disturbed old-growth and second-growth forests is dependent on a variety of factors including the type and intensity of the disturbance, type of forest, appropriateness of microhabitats, the propagules present in the seed bank and the dispersion characteristics within and around the selected area (Toniato & de Oliveira-Filho 2004).

This thesis addresses the restoration of vegetative understory composition of urban forests within Hamilton. It identifies what native plant species are absent from urban forests relative to old-growth forests, in rural locations surrounding the city. The thesis investigates the environmental conditions of urban forests as a determinant for the absence of plant species which, theoretically, can exist in Hamilton City forests. Three species suitable for enrichment plantings are trialled and assessed as a method to restore urban Hamilton forests to a more natural state and develop underlying principles in restoration ecology. Recommendations and management options are provided for improving understory plant diversity within urban forests. In order to provide background and context the three major themes of the research are reviewed below. Objectives, experimental design and study locations are then outlined in detail.

1.2 New Zealand forests

New Zealand has a distinctive bio-geographic history, one which has been significant in shaping a diverse array of plant communities (Macdonald 2006). Major events contributing to the creation of New Zealand’s distinctive ecosystems include; breaking away from Gondwanaland approximately 80 million years ago and the creation of diverse habitats through continual climate changes and geological processes (Halkett 1991; McGlone et al. 2001; Macdonald 2006).

Vicariance and long-distance dispersal are both thought to have shaped New Zealand’s flora and along with isolation, has led to 80% of New Zealand’s tree and shrub species being endemic (Halkett 1991; McGlone et al. 2001; Macdonald 2006). Colonisation of New Zealand by Polynesians, and later Europeans, led to widespread deforestation, particularly lowland conifer/broadleaf forests, for agricultural, urbanisation and economic purposes (Wardle 1991; Walker & Wass 2006).

New Zealand now has just 34% of the original indigenous forest cover, while in the North Island alone only 20% remains (Halkett 1991). Remaining native forest is, in many places, restricted to hill country, where farming is impractical, as well as small vulnerable fragments, that remain in lowland areas (Wardle 1991).

1.3 Restoration ecology

1.3.1 Restoration ecology and ecological restoration

Restoration ecology is a relatively new area of science that is increasing in popularity around the world in a variety of diverse ecosystems (Roberts et al. 2009). Restoration ecology can be described as the study, understanding and development of theories, concepts, techniques and models in which to manipulate and aid the recovery of a degraded ecosystem. The theory is developed to restore an ecosystem to one which existed historically, or to a historical trajectory, before human intervention (Halle & Fattorini 2004; Society for Ecological Restoration International Science and Policy Working Group 2004). Restoration ecology provides the opportunity to study ecosystem structure and discover how ecosystems function and interconnect (Smith & Smith 2001).

Ecological restoration is the exercise of using ecological theory to improve a degraded ecosystem which is often caused through human activities, to an ecosystem which existed historically or to a historical trajectory. (Halle & Fattorini 2004; Society for Ecological Restoration International Science and Policy Working Group 2004). Ecological restoration can include a vast array of activities that aim to restore the function, composition and structure of the original ecosystem (Halle & Fattorini 2004).

1.3.2 Dilemmas of restoration ecology

Restoration ecology is still in developmental stages but is often criticised for having a limited set of theories and principles that can be used in more than one set of circumstances (Halle & Fattorini 2004). Most research has so far been based on restoration work undertaken and success achieved (or not) at individual sites and, although important, may not produce widely applicable theories (Hobbs & Norton 1996; Halle & Fattorini 2004). Other limitations include; the breakdown of communication between researchers developing the concepts and individuals/groups undertaking practical restoration, the use of appropriate reference sites to use as examples, the lack of information available to describe historical ecosystems and the abiotic and biotic facets, such as climate change and the introduction/extinction of species, which cannot be reversed (Halle & Fattorini

2004; Society for Ecological Restoration International Science and Policy Working Group 2004).

However, there are many other areas of science, in particular ecology, which already have developed concepts and these can be applied to restoration ecology. Such related disciplines include; paleoecology, succession, disturbance ecology and conservation biology (Crawley 1997a; Halle & Fattorini 2004; Jackson & Hobbs 2009).

1.4 Urbanisation

1.4.1 Urban and urbanisation

An urban area can be described as a human community that contains a high density of people with their associated dwellings and other constructed buildings and infrastructure (Niemela 1999). Urbanisation is the continual loss of natural and semi-natural habitats due to the increasing human population, the requirement for more infrastructures and associated growth and expansion of urban areas (Crawley 1997a). It is anticipated that human growth will predominantly occur in urban areas over the coming years and it is estimated 60% of the global human population will be found within urban areas by 2030 (Alberti 2005; Bernhardt & Palmer 2007).

1.4.2 Effects of urbanisation on forests

The main effects of urbanisation on forest ecosystems are a reduction in forest area and forest fragmentation as a result of land clearance. Reducing and fragmenting forest environments result in changes in overall composition, structure and function. This is primarily due to habitat removal and edge effects which subjects the forest to a contrasting environment often causing detrimental effects. When forest is removed it is replaced by urban infrastructures that are commonly inhospitable to forest plants, while the edge effect exposes remnant forests to natural and human disturbances and altered environmental conditions (Alberti 2005). The loss of connectivity between forests limits the spread of plants and animals between sites reducing the area available for species to maintain a viable population. In many land clearance scenarios, native plant diversity is reduced with many plant species becoming extinct or endangered (Godefroid & Koedam 2003; Alberti 2005).

Another effect of urbanisation is the pressure from introduced exotic species (Borgmann & Rodewald 2005). Invasive species affect urban forests all around the world, including New Zealand. The human induced environment is generally different to that which native plants and animals have evolved under where high levels of disturbance and stress promote establishment of invasive exotic species (Sullivan et al. 2009). Early succession and adventive species are generally first to colonise available ecological niches after a disturbance, often dispersed from areas such as residential yards and unmanaged sections by the wind or birds (McKinney 2002). Invasive plants are usually generalists that are able to exploit a variety of habitats and are also usually highly competitive (Porteous 1993).

Recent studies have shown the effect humans have on the nutrient cycling (Alberti 2005), soil properties (Rebele 1994), pollution levels (Rebele 1994; Lehvavirta & Rita 2002; Alberti 2005) and hydrology (Walsh et al. 2005) within urban forest ecosystems. Human altered biogeochemical processes, movements and transformations have created new sources of nutrients and created new vectors for nutrients to move to and from urban forests. Sources include fertilisers and nitrogen oxides from the burning of fossil fuel. Vectors include surface water and atmospheric exchange of carbon (Alberti 2005). In many urban areas, including forest remnants, the soil profile and qualities have been altered due to activities related to human settlement. Soil may be mixed or combined with foreign substrates, compacted, or more commonly, sealed over (Rebele 1994). Pollution deposits from various human activities can also build up in urban environments causing elevated levels of heavy metals in soil (Rebele 1994; Lehvavirta & Rita 2002; Alberti 2005). The hydrology of urban areas is altered by increased impervious areas, modified soil conditions, altered vegetation type and cover and the deliberate alteration of drainage. Subsequent effects include water table changes, erosion and many stream and stream process modifications (Walsh et al. 2005).

Human use and recreation are some of the most well-known and intensively managed threats to urban forests (Cole & Landres 1996; Lehvavirta & Rita 2002).

Public use of urban forests for recreational use is increasing and is a threat to forest regeneration, species richness and vegetation coverage. Trampling of the soil causes soil compaction which has subsequent adverse impacts on plants causing reduced soil porosity, increased root penetration resistance and changes in bulk density (Bhuju & Ohsawa 1998). Other recreational consequences include the importation of foreign materials and subsequent littering and the harvesting of plant material (Cole & Landres 1996). Intentional management practices such as creating defined pathways, also contribute to the adverse outcomes, although most activities become beneficial in the future (Cole & Landres 1996; Bhuju & Ohsawa 1998).

1.4.3 Benefits of urban forests

Urban forests provide similar benefits to rural old-growth forests. Urban forests hold important ecological values, often containing a high diversity of not only native plants but animals as well (Porteous 1993; Crawley 1997a). Urban forest remnants act as refuges and wildlife corridors, allowing native animals to pass between different regions to breed and forage (Porteous 1993; Fuhrer 2000). Urban ecosystems present the opportunity to study, not just the ecology of plants and animals, but also paleoecology, allowing a ‘snapshot’ into past ecosystems, and research directed towards environmental science (Porteous 1993; Jackson & Hobbs 2009).

There are ethical and moral reasons why we should protect and restore native forests (Crawley 1997a). Native forests contribute to New Zealand’s identity and support the quality of the environment, illustrating historical, pre-human settings (Porteous 1993). Forests provide an area for tourism and recreational activities including picnic backdrops, hiking and hunting as well as strong aesthetic values (Crawley 1997a; Fuhrer 2000; Dodd & Ritchie 2007). New Zealand’s indigenous forest is central to many cultural beliefs and uses (Porteous 1993). For example, many Maori legends are created around forests, including one of the most prominent gods, tane-mahuta, the god of the forest (Halkett 1991). Maori extensively used native plants for an array of activities including; food supply, medicines, tools and materials (Macdonald 1973).

Historically, New Zealand forests were harvested unsustainably, with native timber used primarily for infrastructure and export (Halkett 1991). It is now recognised that forests offer different economic values, including the removal of pollutants and nutrients from urban runoff and improving water quality (Rowntree 1986; Naiman & Decamps 1997). Mature forests effectively remove sediments from urban runoff which contain nutrients and pollutants. The finer the material the higher amounts of pollutants, nutrients and pesticides that can be adsorbed. Urban forests offer the ability to filtrate contaminated sediments through plant uptake resulting in long-term accumulation in woody plants (Naiman & Decamps 1997).

A consequence of urban environments is high levels of noise. Urban forests and soil help to modify the acoustic environment by masking and attenuating sound. Studies have shown that high frequency sound is scattered by trees while low frequency sound is reduced due to direct sound waves and reflected sound waves from the ground. Forest floors tend to absorb sound better than bark from trees as thicker litter layers cause greater attenuation (Rowntree 1986).

Although limited research has been conducted, early studies are suggesting that urban forests are potentially urban sinks for air pollution. Air pollution in urban centres is created primarily from the industrial sector and vehicles, which inject carbon monoxide, sulphur dioxide and hydrogen sulphide into the lower atmosphere on a daily basis. Various sized particles land on the leaves although the succeeding pathway of the particles is unclear. However urban forests can also act as a source of air particles, releasing pollen and in certain circumstances types of hydrocarbons into the atmosphere (Rowntree 1986).

Urban areas contain a variety of different sized buildings and structures placed in a mostly systematic pattern. This can cause strong winds at street level with variable velocity. It is possible for forests and low branching trees to alleviate problems caused by strong winds, by reducing wind intensity. Placed in the right location trees have the potential to decrease the effect of a cool wind, whilst maintaining ventilation in the summer period (Rowntree 1986).

1.4.4 Urban ecology and restoration

Urban ecology has previously been a neglected part of science due to the research area being deemed unnatural, dull and uninteresting (Lehvavirta & Rita 2002). Ecologists were reluctant to study urban ecology because of unknown disturbance regimes, multiple and unusual stressors and other unfamiliar processes operating in urban environments. Instead urban environments have more regularly been studied by geographers, social scientists, anthropologists and economists (McDonnell et al. 1997).

It is important to understand urban ecology in order to manage and sustain urban forests and their biodiversity (Lehvavirta & Rita 2002). The lack of knowledge regarding urban ecology has meant that documentation of biodiversity values within the city is poor and results in limited knowledge causing poor planning decisions (Niemela 1999). Urban ecology is an expanding science and there exist many opportunities to study this young area of research in particular the ecological impacts of urbanisation (Rebele 1994; McDonnell et al. 1997; Alberti 2005).

1.5 Research objectives

Forests within Hamilton City have been subject to many pressures associated with urbanisation. In particular, the forest understory has undergone major transformations in composition, structure and function. This research addresses three interlinked objectives relevant to the theory and practice of restoring the understory component of urban forests:

1) Compare and contrast the vegetative composition of the understory of different aged urban forests within Hamilton and that of rural forests surrounding the city. Determine which native understory plant species are present in forests surrounding the city, but absent or lower in abundance within urban forests.

2) Quantify the environmental differences between the understory of different aged urban forests within Hamilton and forests surrounding the city. Determine the extent to which these factors might contribute to the

differences in composition of understory vegetation between urban forests and rural forests surrounding the city.

3) Conduct a translocation experiment by reintroducing three native understory plant species into Hamilton urban forests that are abundant in forests surrounding the city. Measure each species performance (growth and survival rates) and their potential for becoming fully established in the understory of urban forests.

1.6 Experimental design

1.6.1 Methods

The first objective was addressed by conducting a survey to quantify the understory vegetation at the 25 listed sites (Figure 1.1 and 1.2) described below under section 1.7. The second objective was addressed by measuring and comparing environmental parameters, at a sub-sample of the 25 sites (except relative humidity and temperature) illustrated in Table 1.1 by the shaded boxes. The third objective was addressed by planting three species suitable for enrichment plantings into the same sub-sample of locations used in the second objective.

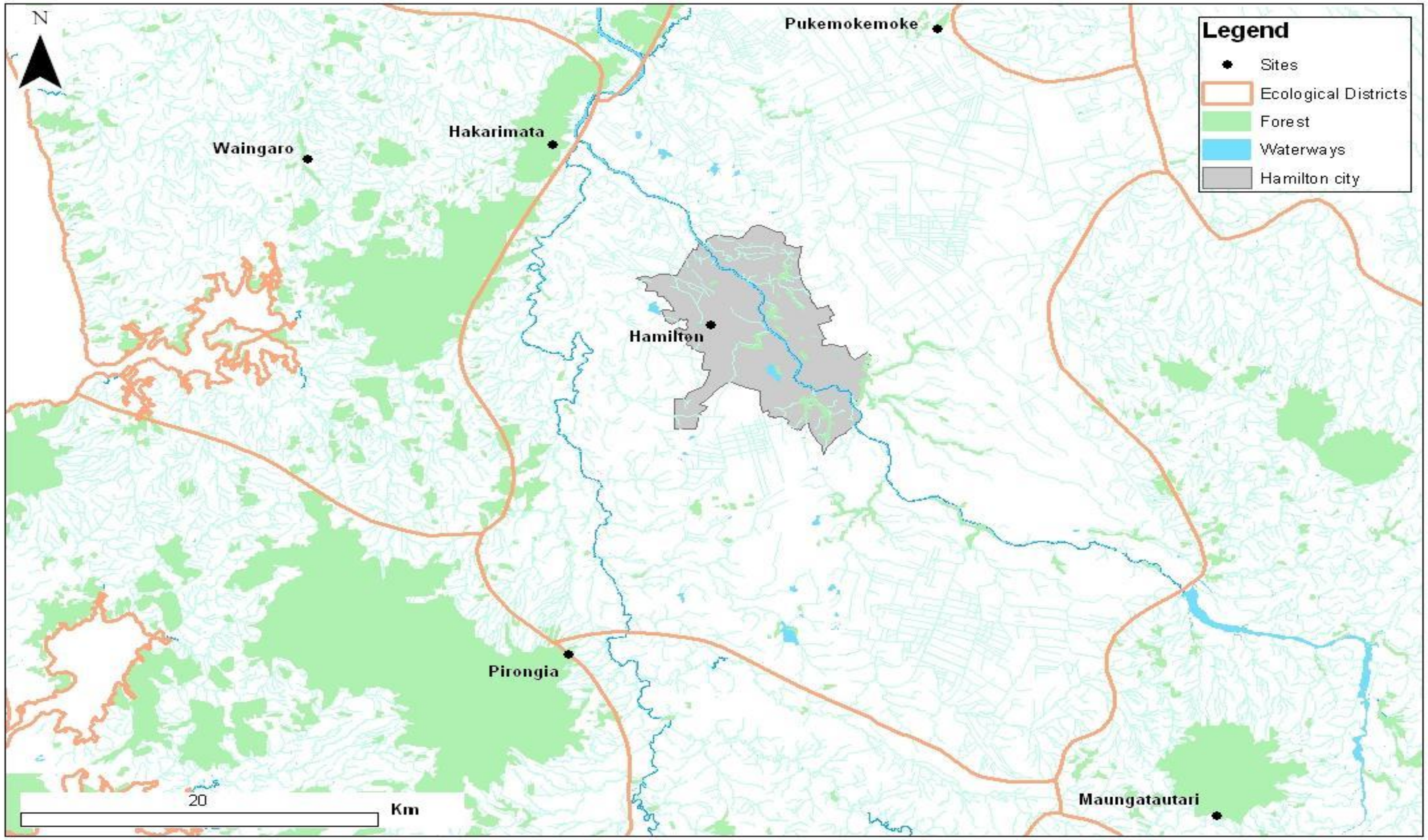

Figure 1.1: The location of the five rural forest sites that were used in this research (Map courtesy of T. Cornes).

Figure 1.2: The location of the 20 urban forest sites that were used in this research (Map courtesy of T. Cornes).

1.7 Study locations

1.7.1 General description of the Waikato Region

The Waikato Region, situated in the central North Island of New Zealand, has a midpoint of approximately 175º longitude and 38º latitude (Terralink 1998). The main physiographic units are the highlands, basins and extensive river networks (McCraw 2002). Prevailing winds are from the west and southwest, which create humid conditions. The average annual rainfall is 1250 mm; however mountainous areas often receive higher amounts while droughts can occur in the basins during the summer. The Waikato Region is well known for foggy and frosty conditions during cold periods (Environment Waikato 2010).

1.7.2 Rural old-growth forest sites

1) Hakarimata Scenic Reserve

This reserve (1850 ha) encompasses most of the Hakarimata Range, a low lying greywacke range (highest point 374 m above sea level) trending north to south, near Ngaruawahia, 15 km northwest of Hamilton. Forest entirely covers the reserve and there is also regenerating forest outside the reserve boundaries (Department of Conservation 2010). Forest types represented are conifer-broadleaf forest with *Dacrydium cupressinum** (rimu)-*Beilschmiedia tawa* (tawa), tawa and *Agathis australis* (kauri)-softwoods-hardwoods (Forest Research Institute 1979). Slopes are steep near the range crest but gentle at lower altitudes. There is a history of low level logging and fires (Department of Conservation 2010).

2) Mount Maungatautari Ecological Island

Mount Maungatautari is an extinct volcano (highest point 797 m above sea level) that is the focus of a restoration project (3363 ha) and is located approximately 35 km southeast of Hamilton City (Maungatautari Ecological Island Trust 2002). Forest types include conifer-broadleaf forest with rimu-tawa and tawa (Forest Research Institute 1979). The mountain summits are rugged but slopes are gentler at lower altitudes and forest has in the past come under threat from encroaching farmland (Maungatautari Ecological Island Trust 2002).

*Species nomenclature follows the NZ Plant Conservation Network (New Zealand Plant Conservation Network 2011)

3) Pirongia Forest Park

Mount Pirongia an extinct volcano (highest point 959 m above sea level) is the main feature of Pirongia State Forest Park (17 000 ha). The forest park is surrounded by agricultural land and located 25 km southwest of Hamilton City (Department of Conservation 2010). Main forest classes are conifer-broadleaf, rimu-tawa and tawa (Forest Research Institute 1979). The mountain contains steep slopes at the summit but ease with lower altitudes. The forest park has a history of forest clearance and logging (Department of Conservation 2010).

4) Pukemokemoke Bush Reserve

The bush reserve (40 ha) is located on a greywacke hill (highest point 166 m above sea level) 20 km northeast of Hamilton City (Department of Conservation 2010). The reserve comprises steep and gentle slopes throughout and is covered by conifer-broadleaf forest with tawa and kauri-softwoods-hardwoods-*Nothofagus* (beech) (Forest Research Institute 1979). The reserve and surrounding area has a history of tree logging and milling and clearance. There is a current restoration programme in place to enhance the reserve (Department of Conservation 2010).

5) Waingaro Forest

The private native bush patch (8 ha) adjoining the Waingaro Forest Park is situated on undulating greywacke topography (highest point 120 m above sea level) 25 km northwest of Hamilton City (Clarkson 2004, unpublished report). Conifer-broadleaf forests are found on the terrace and lower slopes while tawa forest covers the steeper slopes (Forest Research Institute 1979). Agricultural development and tree logging have reduced the reserve size. Extensive sheep and beef farming occur on the northern and eastern boundaries and plantations of *Pinus radiata* (pine) to the west and south (Clarkson 2004, unpublished report).

1.7.3 Urban sites: Old-growth forest remnants 100+ years

6) Berkley Bush

Berkley bush is a privately owned *Dacrycarpus dacrydioides* (kahikatea) stand (0.4 ha) located in the southeast of Hamilton City (Hillcrest). Part of the Mangaonua Gully, the patch has a kahikatea canopy ranging between 25 to 30 m. Slopes vary between 0 to 40º and have a predominant eastern aspect. *Tradescantia fluminensis* (wandering Jew) and *Lamium galeobdolon* (aluminium plant) have formed thick ground cover in places. Houses and residential buildings surround the area as well as large introduced trees. Grazing ceased more than 20 years ago when the bush boundary was fenced off (Downs et al. 2000b).

7) Claudelands Bush

Claudelands Bush (officially Jubilee Park) (5.4 ha) is a kahikatea stand located in Hamilton City (Claudelands). The stand contains 20 to 25 m tall kahikatea on an alluvial plain that has a minimal slope. The fenced off forest is surrounded by roads and a field. The forest is protected on the western margin by a wind break and there are raised boardwalks within the forest (Downs et al. 2000a). The surrounding land was drained first for agriculture and then for urban development, and also logged for economic purposes. Grazing ceased in the late 1920s when the remnant was fenced (Gudex 1955). The forest currently has an active community group working to enhance the forest (Downs et al. 2000a).

8) Hammond Bush

Hammond Bush (1.8 ha) flanks the northern riverbank of the Waikato River in the southeast of Hamilton City (Hillcrest). Slopes vary between 0 to 80˚ with a predominant southern aspect. The area contains three forest types; *Laurelia novae-zelandiae* (pukatea)-*Syzgium maire* (swamp maire), tawa-*Alectryon excelsus* (titoki) and *Melicytus ramiflorus* (mahoe)-*Kunzea ericoides* (kanuka) with *Weinmannia racemosa* (kamahi). The forest is surrounded by the Waikato River and residential properties. Raised boardwalks and paths are present throughout and a community group works to enhance the forest (de Lange 1996; Downs et al. 2000b).

9) Hillcrest Park

Located in the southeast of Hamilton City (Hillcrest), Hillcrest Park (1.3 ha) is a kahikatea stand on a flat alluvial plain. The kahikatea canopy is 20 to 25 m high and is supplemented by native plantings in the understory and on the sides. The fenced off forest is bisected by a path and raised boardwalk and there is a scouts hall in the centre. Grass fields surround the majority of the fenced-off forest with some residential housing (Downs et al. 2000b).

10) Mooney Park

Mooney Park (0.3 ha) is a kahikatea stand located in the northwest of Hamilton City (Nawton North). The 20 to 25 m tall kahikatea are on a flat alluvial plain supplemented with plantings around the edge. A road, field and residential area surround the forest. Prior to subdivision development the land was grazed (Downs et al. 2000a).

1.7.4 Urban sites: Forests aged between 25-100 years

11) Braithwaite Park

The kanuka forest patch is located within Braithwaite Park in the northern regions of Hamilton City. The forest (0.5 ha) contains slopes ranging from 30 to 60º in a north eastern direction. The kanuka canopy is between 15 to 20 m high with wandering Jew a prominent ground cover in places. A maintained field and the Waikato River surround the forest (Downs et al. 2000a).

12) Chelmsford Park, Mangaonua Gully

Chelmsford Park is located in the southeast of Hamilton City (Silverdale). The forest patch (0.7 ha) is predominantly flat however does contain a range of slopes from 0 to 80º. The gully floor is poorly drained in most places. The canopy is dominated by 10 to 15 m *Salix* (willow) species, with occasional *Pinus* (Pine) and *Eucalyptus* species present on the ridge. On the higher terrace are kanuka and *Pittosporum* species plantings. Area adjacent to the park includes a maintained field and beyond the boundaries are agricultural paddocks (Downs et al. 2000b).

13) Pukete Riverside Forest

The Pukete riverside forest is located in the northern regions of Hamilton City, adjacent to the Waikato River. The forest patch (1.2 ha) contains slopes that range between 10 to 70º with a predominant south western aspect. The dense and mostly continuous mahoe and tree fern canopy is approximately 10 m high. *Alnus* (Alder) and willow are more common in the canopy closer to the river. An informal track is present running parallel to the river and there is residential area above the slopes. Wandering Jew is becoming more abundant (Downs et al. 2000a).

14) Seeley’s Gully Reserve

Seeley’s Gully Reserve is found in eastern Hamilton City (Claudelands) and comprises a native forest planted by A. J. Seeley. The site (2.2 ha) has slopes from 0 to 40º facing an array of directions. The forest canopy is from 15 to 20 m in height and comprises many different species including kahikatea, rimu, kauri, titoki, *Podocarpus totara* (totara) and kanuka. Tree planting began over the past 50 years and the forest is the oldest example of reestablishment of native bush in Hamilton City. Wandering Jew is a common groundcover and residential areas surround the reserve (Downs et al. 2000a).

15) Tauhara Gully (Opposite River Road)

Tauhara gully (3.9 ha) is located in northern Hamilton City (Flagstaff) and supports several different aged forests. Slopes vary between 0 to 30º with variable aspects. Mahoe is the dominant species in the 10 to 15 m canopy, although exotics such as willow are present. The western edge of the gully has been supplemented with plantings, while wandering Jew is present. Residential areas and a main road surround this part of the gully.

1.7.5 Urban sites: Forests aged between 15-25 years

16) Howell Street, Hudson Gully

The Howell Street forest patch (0.2 ha) is privately owned and is part of the Mangaonua Gully system found in the southeast of Hamilton City (Hillcrest). Wandering Jew, aluminium plant and a *Convolvulus* species form a dense covering in places. The 10 to 15 m canopy contains tree ferns, *Juglans ailantifolia* (Japanese walnut) and a large *Corynocarpus laevigatus* (karaka). The gully contains a flat terrace and slopes between 0 to 25º. The surrounding area is residential housing.

17) Morrinsville Road, Mangaonua Gully

The Mangaonua Gully forest patch (0.4 ha) on Morrinsville Road is privately owned and well taken care of. The forest area contains a high diversity of native plants. The 20 m canopy is predominantly willow with some tree ferns. The slopes range between 0 to 20º with a south western aspect. The area surrounding the forest is residential.

18) Ranfurly Park

Ranfurly Park is a kanuka patch (0.3 ha) located in central Hamilton City (Fairfield). The gully contains slopes from 0 to 30º with variable aspects. The kanuka canopy is 10 to 15 m tall and contains a native understory that is both planted and naturalised. *Allium triquetrum* (three-cornered garlic) and *Hedera helix* (common ivy) form localised patches. The park is surrounded by residential areas with a field and footpath in the centre of the park (Downs et al. 2000a).

19) St. Andrews Riverbank

St. Andrews kanuka stand (2.2 ha) is located adjacent to Hamilton City's St. Andrews golf course and the Waikato River. The forest area contains steep slopes between 10 to 80º with eastern aspects. The kanuka canopy is 15 to 20 m high and also contains mahoe, tree ferns and some exotics such as alder. Wandering Jew and a *Convolvulus* species form dense patches in places. Residential housing is nearby and a paved walkway runs parallel along the river through the forest (Downs et al. 2000a).

20) Waitawhiriwhiri Gully, Whitiora

The Whitiora forest patch (0.6 ha) of the Waitawhiriwhiri Gully is located in central Hamilton City (Whitiora). Slopes range between 0 and 50º with variable aspects. The 10 to 15 m canopy consists of native and exotic species including *Hoheria sexstylosa* (lacebark), tree ferns, pine and Japanese walnut. Wandering Jew is abundant in the area forming dense mats. The area surrounding the gully is residential housing (Downs et al. 2000a).

1.7.6 Urban sites: Forests aged between 5-15 years

21) Ashmore Crescent, Onukutara Gully

The Onukutara Gully forest segment (3.1 ha) is found in northern Hamilton City (Rototuna). The forest contains a flat terrace with slopes between 0 to 15º and a 10 to 20 m alder and willow canopy. Wandering Jew and common ivy form dense patches in some places. Residential area and a main road surround the forest.

22) Clements Crescent, Tauhara Park

Tauhara Gully (3.1 ha) is located in northern Hamilton City (Flagstaff). Slopes range from 0 to 30º in a 10 to 15 m mixed native and exotic canopy that includes

Pittosporum species, totara, *Cordyline australis* (cabbage tree), pine and Japanese walnut. A sealed track runs through the centre of the forest. Residential area and a field surround the forest.

23) Pickering Crescent, Onukutara Gully

The Onukutara Gully forest segment (0.5 ha) is located in eastern Hamilton City (Queenwood). The 10 to 15 m canopy contains kanuka and willow on slopes that range between 0 to 15º with a predominant northern aspect. The surrounding area is residential housing with a primary school, where students have helped to plant native vegetation (Downs et al. 2000a).

24) Porritt Stadium

Porritt Stadium is located on the eastern side of Hamilton City (Chartwell). The gully, which is part of the Kirikiriroa Gully system, runs along the south western side of the stadium. The forest patch (6.5 ha) canopy is dominated by 15 to 20 m high exotic species that include pine and *Acacia* (wattle). Hill slope and terrace land forms are represented. Wandering Jew and a *Convolvulus* species form dense mats in places. Selected areas within the forest are aged between 5 to 15 years. Residential area and a maintained field surround the forest.

25) Sandford Park, Mangakotukutuku Gully

Sandford Park (8.3 ha) is located in southern Hamilton City (Fitzroy). The park supports a range of forest patch ages with the sampled forest area containing a 10 to 15 m canopy dominated by *Pittosporum eugenioides* (lemonwood), kanuka and *Aristotelia serrata* (wineberry). Slopes range between 0 to 20º with an eastern aspect. The area surrounding the park is residential housing and a recreational field. A community group operates in the park clearing exotics such as wandering Jew and planting native species.

Table 1.1: Sub-sample of locations (shaded boxes) used in the second, environmental profiling, and third, the reintroduction experiment, objectives.

Rural old-growth forests	Urban forests 100+ yrs	Urban forests 25-100 yrs	Urban forests 15-25 yrs	Urban forests 5-15 yrs
Hakarimata Scenic Reserve	Berkley Bush	Braithwaite Park	Howell St. Hudson Gully	Ashmore Cr. Onukutara Gully
Mt. Maungatautari Ecological Island	Claudelands Bush	Chelmsford Park	Morrinsville Rd. Mangaonua Gully	Clements Cr. Tauhara Park
Mt. Pirongia Forest Park	Hammond Bush	Pukete Riverside Forest	Ranfurly Park	Pickering Cr. Onukutara Gully
Pukemokemoke Bush Reserve	Hillcrest Park	Seeley's Gully Reserve	St. Andrews Riverbank	Porritt Stadium
Waingaro Forest	Mooney Park	Tauhara Gully (Op. River Rd)	Waitawhiriwhiri Gully, Whitiora	Sandford Park Mangakotukutuku

1.8 Thesis layout

The thesis comprises five chapters and addresses in turn the three objectives described in section 1.5.

1.8.1 Chapter one

Chapter one provides a literature review of the three main themes of the research topic; New Zealand forests, restoration ecology and the impacts of urbanisation. This is followed by a discussion of the objectives of the study and information on the overall layout of the thesis, the content in the five chapters and descriptions of the study sites.

1.8.2 Chapter two

Chapter two explores the differences in understory vegetation composition between the selected urban and rural forest sites. An introduction describes the history of the Waikato Region and remnant forests and the importance of understory vegetation. The chapter describes the survey methods in the investigation and displays and explains the results, before discussing the overall findings of the survey.

1.8.3 Chapter three

Chapter three explores the environmental differences between the selected forest sites. Introduction and methods sections describe how each of the environmental factors affects plant growth and how the variables were measured. The results and discussion sections presents the findings and evaluate the differences and similarities in environmental factors between sites.

1.8.4 Chapter four

Chapter four researches the potential of reintroducing three native understory plants into remnant and restored forests in Hamilton City. Introduction and methods sections describe the principles of enrichment plantings and explain the processes and systems used in the experiment. The results section provides an in-depth comparison between the plantings and the discussion assesses the potential of the three species becoming a more prominent component in Hamilton urban forests.

1.8.5 Chapter five

Chapter five synthesises the results for each of the three objectives and describes the results in terms of ecological theory. The chapter discusses overall conclusions of the research and provides management recommendations for improving native understory vegetation within Hamilton City forests. Finally the chapter presents potential future research options that are applicable to the present research.

2 Chapter two: Understory vegetation survey

2.1 Introduction

This chapter investigates the vegetative understory composition and diversity of Hamilton urban forests in relation to rural old-growth forests surrounding the city. Composition, richness and density of understory species were measured across five forest categories; rural old-growth; urban over 100 years; urban 25 to 100 years; urban 15 to 25 years; and urban 5 to 15 years. The analysis shows how understory composition, richness and density of the various urban forest age categories compare with reference rural old-growth forests. The results also provide a basis for chapter three, which examines if environmental factors are cause for compositional and density differences in urban forests, and chapter four, which assesses the performance of three understory species absent or less abundant from urban forests following their reintroduction.

2.1.1 Hamilton City and the Waikato Region

The landscape of the Waikato Region is characterised by three highland areas separating two lowland zones. Along the western side, adjacent to the Tasman Sea, are the Western Uplands, running from north to south. Parallel to the Western Uplands and separated by the Hamilton Basin, are the Central Hills which are narrower than that of the Western Uplands. Further east is the Hauraki Basin followed by the Eastern Ranges which makes up the Coromandel Region and the western side of the Bay of Plenty. Within the Hamilton and Hauraki basins are four extensive river networks, the Waipa and Waikato Rivers and the Piako and Waihou Rivers respectively (McCraw 2002).

The Waikato Region’s largest city, Hamilton City, is located in the centre of the Hamilton Basin, adjacent to the Waikato River (McCraw 2002). The Hamilton ecological district is made up of four main landforms; low rolling hills, alluvial plains, gullies and peatlands. Each landform contains different vegetation types, depending on environmental conditions (Clarkson et al. 2007).

2.1.2 History of the Waikato Region

The Waikato landscape has been shaped by New Zealand's long and turbulent geological history and by recent human settlement. Before human influence, the Waikato Region contained an array of ecosystems including extensive stream and river networks, large areas of bog and swamps and widespread high and low land forests. Within the diverse ecosystems were an equally varied animal biota that was, apart from bat species, mammal free (Jay 1997).

Studies of forest remnants and pollen analyses suggest that the dominant forest type of lowland Waikato has changed repeatedly over the past 20 000 years. Climates 20 000 years ago were representative of the late glacial period and dominant trees were beech, *Libocedrus bidwillii* (kaikawaka) and *Phyllocladus alpinus* (mountain toatoa), along with hardy shrubs such as *Myrsine* and *Coprosma* species. As the climate became more mild 10 000 to 14 000 years ago, the forest was dominated by *Prumnopitys taxifolia* (matai), rimu and broadleaved trees such as *Elaeocarpus dentatus* (hinau) and pukatea (Newnham et al. 1989; Nicholls 2002).

Change occurred again around 5000 years ago as the climate became cooler and drier, rimu became less abundant, whilst *Phyllocladus trichomanoides* (tanekaha) and kauri were more plentiful (Newnham et al. 1989; Nicholls 2002). Semi-swamp kahikatea forests also developed. These forests were subject to a regular flooding regime and contained species with similar habitat requirements including cabbage tree, *Phormium* (flax) and *Coprosma* species. Totara, matai and *Sophora* (kowhai) were characteristic of better drained sites while tawa and titoki occurred in built up alluvial areas where flooding did not occur (Newnham et al. 1989; Jay 1997).

The first human settlement in the Waikato Region was thought to be around 830 years ago. Pollen analyses show a sudden decrease of forest tree species and an increase in *Pteridium esculentum* (bracken fern) which becomes abundant after forest clearance (Newnham et al. 1989). By 1500 AD, Polynesian colonisers had settled throughout the Waikato Region and cleared some lowland forest in order to create living space and room for food crops. The arrival of European colonisers during the 19th century resulted in further land and forest clearance for agricultural farms and exotic plantations to be developed (Jay 1997; Nicholls 2002). The only indigenous forest remaining was on steep mountain slopes and small remnants scattered throughout the region (Leathwick et al. 1995). Exotic vegetation was also widely planted for shelter and amenity use along waterways (Jay 1997; Nicholls 2002).

As at 1840, forests below the treeline (the zone beyond where trees cannot grow) covered 54% of the Waikato Region. This forest cover was not uniform and ecological districts with highland and mountainous areas contained the most forest cover. In other ecological districts, particularly Hamilton, forest cover was significantly reduced. Present day primary forest cover is just 30% of the original cover with 40% of those remaining forests having undergone modification of some type. Of the original coastal and lowland forests just 6% remains unmodified (Leathwick et al. 1995).

Primary forests in ecological districts which occupy lowlands in the Hamilton and Hauraki Basins have been almost completely cleared. In some instances within, Hauraki, Hamilton, Hinuera and Waipa Ecological Districts, primary forest has been reduced to below 2% of the total area. In the Hamilton Ecological District only 1.6% of the original indigenous vegetation remains with at least 20% of the native flora extinct or endangered in the area (Leathwick et al. 1995). The remaining indigenous vegetation comprises small remnants scattered throughout the city and surrounding farmland. Fortunately, Hamilton City contains an extensive gully system that has the potential to be restored and contribute towards improving forest coverage within the region (Clarkson & McQueen 2004).

2.1.3 Understory vegetation

The understory of a forest can be described as the vegetation tier between the canopy and the ground layer and comprises shrubs, small trees, young canopy plants and ferns (Smith & Smith 2001). The definition of the understory level can be subjective with some researchers describing it as the zone which has less than 10% light transmittance (Parker & Brown 2000). In this investigation, I have used Smith and Smith's (2001) definition of forest understory but considered only small trees and shrubs that grow to a maximum of 12 m or less, excluding ferns and tree ferns.

The understory layer is one of the most important tiers in any forest ecosystem, often supporting the highest floristic diversity as it contains not only shrubs and small trees, but also young canopy plants (Smith & Smith 2001; Gilliam & Roberts 2003; D'Amato et al. 2009). The forest understory also provides an array of habitats for a variety of animal species (D'Amato et al. 2009). Forest succession and development are dependent on, and affected by the understory layer. It acts as a strong canopy filter, allowing only certain plant species through after a disturbance in the canopy and prior to any disturbance in a closed canopy (Royo & Carson 2006). Understory plants play an important part in nutrient cycling due to their smaller support structure, compared to canopy trees, which cause a more rapid turnover of biomass and nutrients (Chapin 1983).

The understory of old-growth forests is quite distinct to that of second-growth forests and old-growth forests that have been disturbed by anthropogenic activities in urban and rural locations (D'Amato et al. 2009). More sensitive understory plants are eliminated from disturbed and second-growth forests, creating reduced native species richness and density (Spyreas & Matthews 2006). Such losses in urban biodiversity could be attributed to unfavourable microenvironments (Spyreas & Matthews 2006; D'Amato et al. 2009), urban pressures such as recreational and educational activities (Guntenspergen & Levenson 1997; Lehvavirta & Rita 2002), fragmentation and isolation (Godefroid & Koedam 2003), the reproductive and dispersal characteristics of plant species (D'Amato et al. 2009), the stage of succession (Honnay et al. 1999b) and how the forest is managed (Bhuju & Ohsawa 2001). The differences in understory density and composition between rural old-growth forests (Figure 2.1) surrounding Hamilton City and urban forests within Hamilton (Figure 2.2) are noticeable. The rural old-growth forest contains a much thicker understory and diverse composition compared to the urban forest which commonly features a mat of wandering Jew (smothering exotic ground cover).

Figure 2.1: Understory of a rural old-growth forest (Waingaro Forest). Note the diverse range of understory species.

Figure 2.2: Understory of a Hamilton urban forest (Braithwaite Park). Note the reduced plant density and prolific growth of wandering Jew at ground level.

As of 2004 at least 20% of the indigenous plant life within the Hamilton Ecological District was endangered or extinct, including some significant understory plant species (Gudex 1955; de Lange 1996; Clarkson & McQueen 2004). It could be expected that within the boundaries of Hamilton City, the numbers of endangered and extinct indigenous plant species, is significantly higher. Urban forests often contain a lower diversity of plants and animals (Alvey 2006).

2.1.4 Objective

The aim of this investigation was to determine the composition and density of native understory vegetation focussing on which species are absent or depauperate in Hamilton urban forests compared to reference old-growth forests surrounding the city in rural locations. Differences in densities of exotic species between the various forest categories were also investigated. Further objectives included; determining how similar the 25 forest sites are in terms of plant composition and density and the factors controlling any relationships, as well as identifying elements responsible for reduced native composition and diversity.

2.2 Methods

The methods used to assess and measure the understory vegetation across the 25 forest sites are outlined below.

2.2.1 Site selection

Important factors taken into account for selecting forest sites included age, canopy type, and to a lesser extent, location. The age of the sites was important with regard to differentiating between stages of forest succession and development. Earlier stages of forest succession would be expected to have different understory vegetation to a forest in the later stages of succession. Differences between the stages of succession also lead to differences in environmental conditions in the understory, such as light penetration and soil nutrient balances (Crawley 1997b).

Where possible, sites with a native canopy were selected to enable comparisons to be made between the understory of sites from urban forests and reference forest sites outside urban boundaries. However, in some situations it was difficult to find an urban forest of the appropriate age with a native canopy. Instead a forest site with an exotic canopy was chosen. The location of forest sites was important to some extent as the reference forests had to be within a reasonable distance to enable comparisons to be made as there are often differences in vegetation between regions. Urban locations were also spread across the city to enable a fair representation of the understory vegetation.

Forest sites chosen for the research were largely selected based on previous research by Mackay (2006). Topographic maps (Hamilton City Council 2002) of the desired sites were reviewed and reconnaissance of potential sites was also undertaken. Forest ages were based on research conducted by Mackay (2006) as well as information obtained from land owners.

Increment cores were obtained, following procedures outlined by Grissino-Mayer (2003), from the largest trees of the 15 forest sites in the three youngest urban categories to further verify forest age. The tree corer was set at a slight upwards angle, to prevent water draining into the hole, and drilled into the tree at a height of approximately 1.35 m. The core was removed and placed into a straw for protection before being glued into a grooved piece of wood and sanded. The number of growth rings was counted to indicate the age of the tree and thus the minimum age of the forest at the site.

2.2.2 Surveys

Understory vegetation surveys were conducted over the summer period of 2009/10 and covered 25 forest sites. Initial preparation had to be undertaken in order to establish the parameters of the survey, before the surveying could be conducted. Five nested plots were measured in Waingaro Forest to determine the most appropriate sized quadrat to measure understory vegetation in terms of adequacy of data and efficiency.

Five plots were established throughout the reserve. The locations were selected randomly using a random number table to determine compass direction and the number of paces to the location. The nested plot method was adapted from Smith & Smith (2001). The nested plots were constructed using four 30 m tape measures and contained sub-plots measuring 2.5 by 2.5 m, 2.5 by 5 m, 5 by 5 m, 5 by 10 m and 10 by 10 m. Species were identified within each subplot and recorded. The mean number of species was plotted against the increasing area/size of the subplots (Figure 2.3). The number of species rose sharply before levelling off.

This suggested that a 50 m^2 plot was adequate for the purpose of sampling understory vegetation.

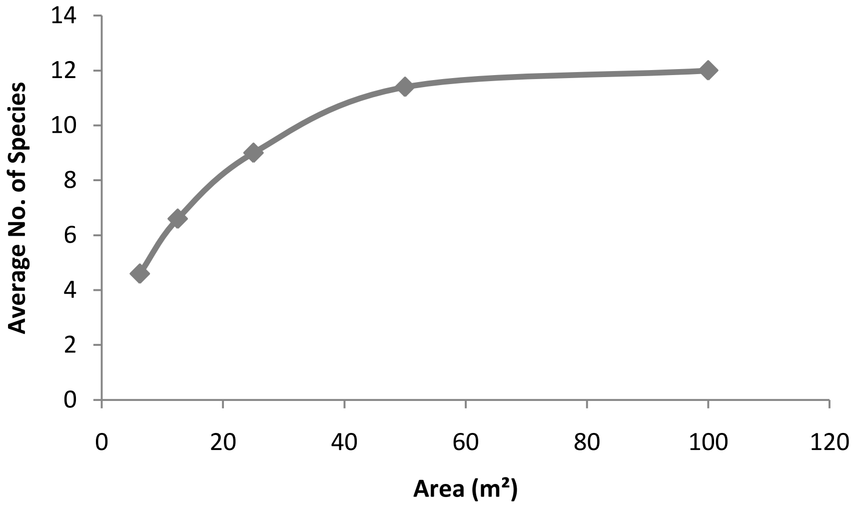

Figure 2.3: The mean number of species found compared with the increasing area/size of the subplots.

The number of plots measured at each of the 25 sites was determined by considering the area of the forest patch, its floristic diversity and the amount of time available to complete the survey. The sampling intensity (Table 2.1) varied between forest sites but was predominantly higher in urban forests. Ten plots were measured in each of the rural old-growth reference forest sites, five plots in each of the urban over 100 years forest sites and three plots were measured in each of the categorised urban forest sites aged between 5 to 100 years. All plots were positioned below 450 m to ensure they were within the zone of tawa dominance (Burns & Smale 2002) and therefore lowland forests in rural and urban locations were comparable.

Table 2.1: The sampling intensity (%) at each of the 25 forest sites surveyed. Only area that was below 450 m above sea level was used to calculate the sampling intensities.

Category	Location	Sampling intensity
Rural	Hakarimata Scenic Reserve	<0.01
	Mount Maungatautari Ecological Island	<0.01
	Pirongia Forest Park	<0.01
	Pukemokemoke Bush Reserve	0.13
	Waingaro Forest	0.63
Urban 100+ yrs	Berkley Bush	6.25
	Claudelands Bush	0.46
	Hammond Bush	1.39
	Hillcrest Park	1.92
	Mooney Park	8.33
Urban 25-100 yrs	Braithwaite Park	3.00
	Chelmsford Park, Mangaonua Gully	2.14
	Pukete Riverside Forest	1.25
	Seeley's Gully Reserve	0.68
	Tauhara Gully (Opposite River Road)	0.38
Urban 15-25 yrs	Howell Street, Hudson Gully	7.50
	Morrinsville Road, Mangaonua Gully	3.75
	Ranfurly Park	5.00*
	St. Andrews River Bank	0.68
	Waitawhiriwhiri Gully, Whitiora	2.50
Urban 5-15 yrs	Ashmore Crescent, Onukutara Gully	0.48
	Clements Crescent, Tauhara Park	0.48
	Pickering Crescent, Onukutara Gully	3.00
	Porritt Stadium	0.23*
	Sandford Park, Mangakotukutuku Gully	0.18*

Note: * underestimates as total includes older forests or grass fields.

At each of the 25 forest sites, the plot locations were paced out on a compass bearing using a random number table to determine the distance and direction. Metal rods and two tape measures were used to mark out a 10 by 5 m plot. A plot sheet (Appendix A) was completed and included; the date, location, altitude, geographic coordinates, plot size, slope and aspect of the plot and the canopy type and height.

The methodology was based on Hurst and Allen's (2007a&b) reconnaissance and permanent plot procedures which are widely used in New Zealand. The understory was divided into four tiers; tier one: less than 0.3 m, tier two: 0.3 to 2 m, tier three: 2 to 5 m and tier four: 5 to 12 m. For tiers two, three and four any vascular plant that had a height within those designated ranges was recorded for that tier using the first three letters of each of its binomial name (genus and species) e.g. *Melicytus ramiflorus* = MEL ram. For every subsequent plant of the same species and same height range a tally system was used. The vegetation within tier one was measured using three circular subplots with a diameter of 49 cm. Subplot locations within the plot was selected using a random number table. Similar to the other tiers, vascular plants within that height range were recorded and tallied. Hurst and Allen (2007a) use a much larger circular subplot in their permanent plot field procedures although they measure vegetation from 0 to 1.35 m.

Notes were made on other factors considered important, for example the number of tree ferns within the plot, drainage conditions and ground cover composition and abundance.

2.2.3 Analyses

Statistical analyses of the native understory species richness, native understory stem density and exotic stem density data was undertaken using the software package 'Statistica' version 9.0. The main analyses undertaken were descriptive statistics and one-way Analysis of Variance (ANOVA). Descriptive statistics was used to provide the mean, maximum, minimum, range and standard deviation. The native understory species richness and stem density were analysed using one-way ANOVA and included the following tests; the initial one-way ANOVA test for a significant difference, Levene's test for homogeneity of variance, test for normality, Shapiro-Wilk and post-hoc analysis fisher test. The exotic understory stem density was analysed using the non-parametric Kruskal-Wallis test as the data did not meet the assumptions of ANOVA. The ordination component of the results used the software 'PATN' version 3.03 and multiple linear regression analyses and ANOVA goodness of fit. In all statistical tests a difference was considered statistically significant if a p-value was obtained that was equal to or less than 0.05.

2.3 Results

2.3.1 General results

The rural forest category contained the highest number of total native understory species with 39, whilst the youngest urban forest category, aged between 5 to 15 years, contained the least with 13 (Table 2.2). The four urban forest classes each contained species normally found in the coastal zone along with native hybrids, while these were absent from the rural forest category. The urban groups also contained three native understory species absent from the rural forest sites; *Cordyline australis*, *Pittosporum tenuifolium* and *Streblus banksii*. Excluding the coastal species and hybrids as well as species only found within the urban categories, the total number of native understory forest species found was again highest in the rural forest group with 39 and lowest in the youngest urban forest group with just nine. Collectively, the urban sites contained just 61.5% of the native understory species found across the five rural old-growth forests. A list of the absent and less abundant species in the urban forest categories is in Table 2.3.

The rural forest group contained nearly twice as many native understory species as that of the next highest category, urban aged over 100 years, which contained 22 (Figure 2.4). Converted to a percentage, the urban sites; older than 100 years, 25 to 100 years, 15 to 25 years and 5 to 15 years each contained 56.4%, 28.2%, 35.9% and 23.1% of the native understory plants found in the rural forest category, respectively.

Table 2.2: Summary of the number and type of different native understory species found between 0 and12 m in height across the five different forest categories.

	Rural	Urban 100+ yrs	Urban 25-100 yrs	Urban 15-25 yrs	Urban 5-15 yrs
Total no. of native taxa.	39	30	16	18	13
Total no. of native coastal species and hybrid taxa.	0	5	3	2	2
Total no. of native species minus hybrid taxa and coastal species.	39	25	13	16	11
Urban species not found in the rural forests.	-	*C.australis* *P.tenuifolium* *S.banksii*	*C.australis* *P.tenuifolium*	*C.australis* *P.tenuifolium*	*C.australis* *P.tenuifolium*
Total no. of native forest species.	39	22	11	14	9
Total % of native forest species.	100	56.4	28.2	35.9	23.1

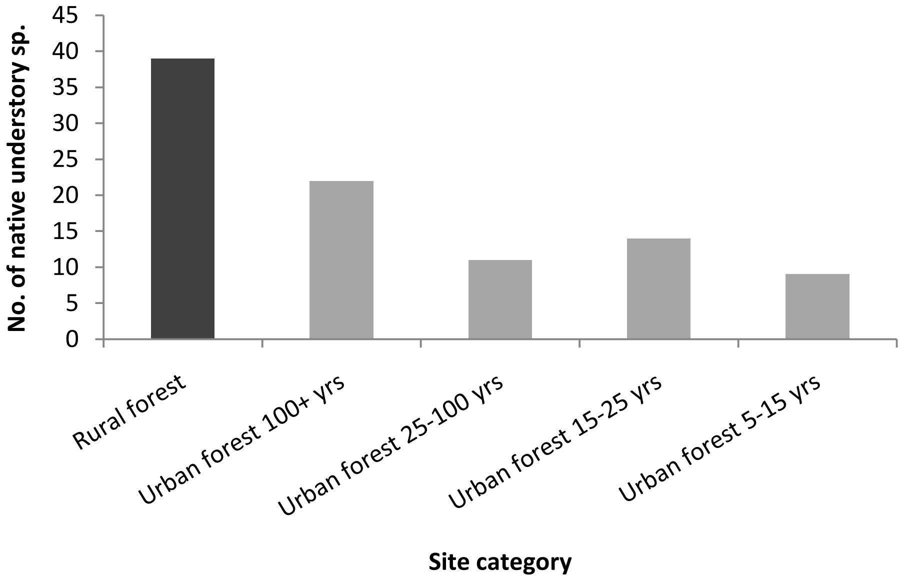

Figure 2.4: The number of native forest understory species found across the five forest categories (Rural group-black bar, Urban groups-grey bars).

Table 2.3: Native understory species absent or lower in abundance (ha^{-1}) in collective urban forests, compared to the rural old-growth forests (Number of species per hectare listed in appendix B).

Absent (tiers 0-12 m)	Less abundant (tiers 0.3-12 m)
Alseuosmia macrophylla	*Alseuosmia quercifolia*
Brachyglottis repanda	*Carpodetus serratus*
Carmichaelia australis	*Coprosma areolata*
Coprosma arborea	*Coprosma grandifolia*
Coprosma rhamnoides	*Coprosma lucida*
Coprosma spathulata	*Coprosma rotundifolia*
Dodonaea viscosa	*Coprosma tenuicaulis*
Hebe stricta	*Geniostoma ligustrifolium*
Leucopogon fasciculatus	*Hedycarya arborea*
Melicytus lanceolatus	*Macropiper excelsum*
Myrsine salicina	*Melicope simplex*
Olearia rani	*Melicytus micranthus*
Pennantia corymbosa	*Myrsine australis*
Pseudopanax arboreus	*Pseudopanax crassifolius*
Rhabdothamnus solandri	*Rhopalostylis sapida*
	Schefflera digitata

2.3.2 Native understory species richness

Mean species richness (Table 2.4 and Figure 2.5) decreased from the rural forest category, down through the four urban age forest categories. The minimum and maximum values followed a similar trend although the urban 15 to 25 age group had a slightly lower minimum than the youngest urban category and a higher maximum than that of the second oldest urban category. Subsequent effects included a larger range and higher standard deviation than the other classes. The statistical analyses (p-value <0.01) indicated that there was a statistically significant difference in native understory species richness between the five forest categories.

Seven statistically significant differences were revealed in the data (Table 2.5). The native understory species richness of the rural forest category was statistically different (higher) to all of the urban forest classes. The native understory species richness of the oldest urban category, aged over 100 years, was statistically different (higher) to the other three urban age groups. There were no significant differences in native understory species richness between the urban categories; 25 to 100 years, 15 to 25 years and 5 to 15 years.

Table 2.4: Descriptive statistics of the number of native understory species per 50 m^2 across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	7.96	5.70	9.20	3.50	1.46
Urban 100+ yrs	5.64	4.40	7.40	3.00	1.27
Urban 25-100 yrs	3.72	3.00	4.30	1.30	0.54
Urban 15-25 yrs	3.06	1.00	6.30	5.30	2.17
Urban 5-15 yrs	2.68	1.30	4.70	3.40	1.33

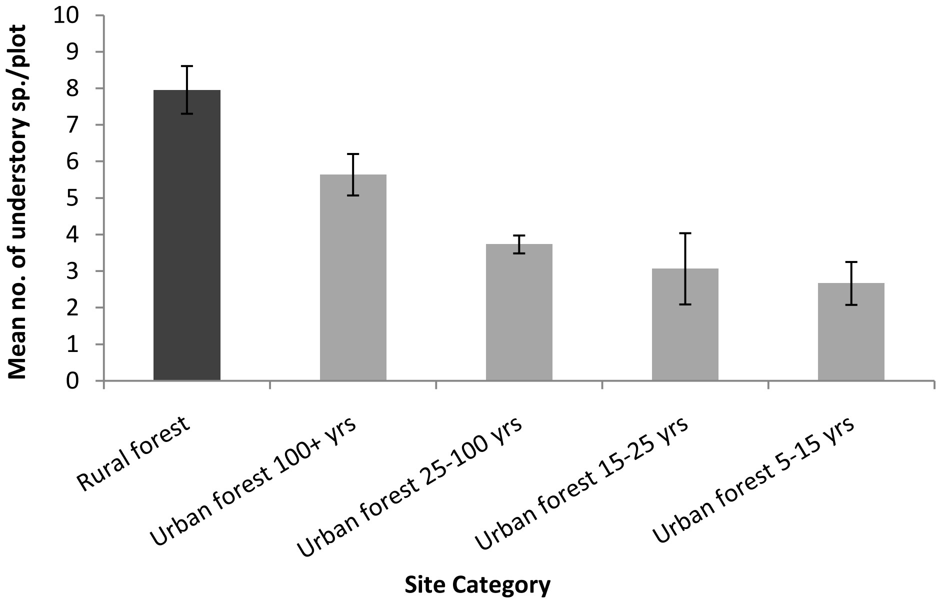

Figure 2.5: The mean number of native understory species per 50 m^2 across the five forest categories with standard error bars (Rural group-black bar, Urban groups-grey bars).

Table 2.5: Post-hoc analysis (Fisher test) for significant differences in the number of native understory species per 50 m^2 across the five forest categories.

	Rural	Urban 100+ yrs	Urban 25-100 yrs	Urban 15-25 yrs	Urban 5-15 yrs
Rural	-	0.02	<0.01	<0.01	<0.01
Urban 100+ yrs		-	0.05	0.01	<0.01
Urban 25-100 yrs			-	0.48	0.27
Urban 15-25 yrs				-	0.68
Urban 5-15 yrs					-

2.3.3 Native understory stem density

The following analysis was undertaken on plants between 0.3 and 12 m in height. Mean stem (Table 2.6 & Figure 2.6) densities decreased from the rural forest category, down through the urban age forest classes, except the two youngest aged urban categories were similar. The minimum and maximum stem densities followed the same trend, except that the oldest urban category had the highest maximum, and consequently the greatest range and standard deviation. The statistical analyses (p-value <0.01) indicated that there was a statistically significant difference in the native understory stem density between the five forest categories.

There were five statistically significant differences within the stem density data (Table 2.7). The native understory stem density of the rural forest category was significantly different (higher) to the three youngest urban classes. The native understory stem density of the oldest urban category, aged over 100 years, was significantly different (higher) to the two youngest urban age groups. There were no significant differences in native understory species richness between the urban categories; 25 to 100 years, 15 to 25 years and 5 to 15 years.

Table 2.6: Descriptive statistics of the number of native understory stems (0.3-12 m), per 50 m^2 across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	41.28	27.20	54.20	27.00	11.98
Urban 100+ yrs	29.76	10.40	59.60	49.20	18.61
Urban 25-100 yrs	17.02	12.70	22.00	9.30	4.10
Urban 15-25 yrs	7.92	1.30	18.30	17.00	6.47
Urban 5-15 yrs	8.20	2.70	17.00	14.30	5.69

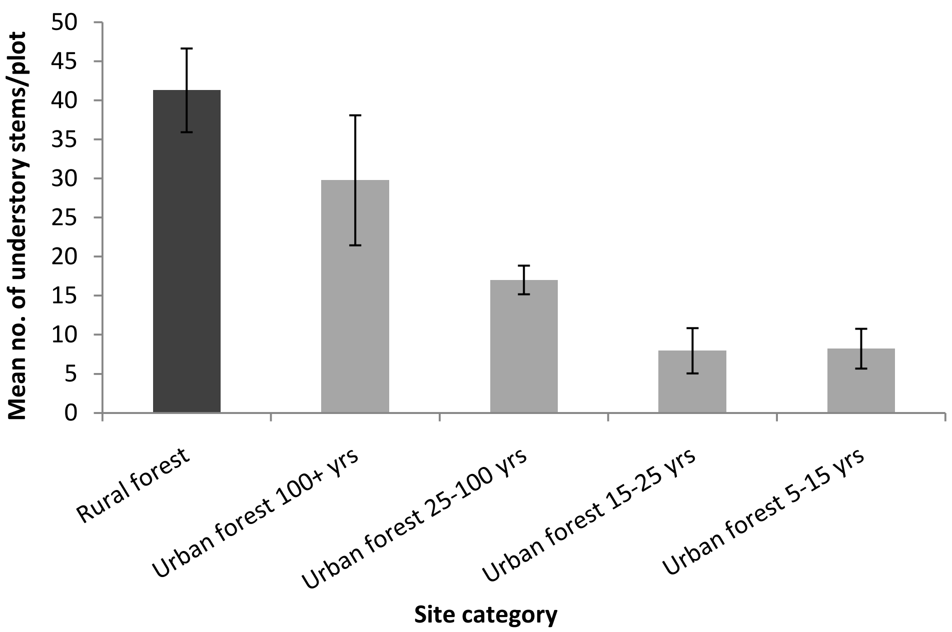

Figure 2.6: The mean number of native understory stems per 50 m^2 across the five forest categories with standard error bars (Rural group-black bar, Urban groups-grey bars).

Table 2.7: Post-hoc analysis (Fisher test) for significant differences in the number of native understory stems per 50 m^2 across the five forest categories.

	Rural	Urban 100+ yrs	Urban 25-100 yrs	Urban 15-25 yrs	Urban 5-15 yrs
Rural	-	0.11	<0.01	<0.01	<0.01
Urban 100+ yrs		-	0.08	<0.01	<0.01
Urban 25-100 yrs			-	0.92	0.21
Urban 15-25 yrs				-	0.97
Urban 5-15 yrs					-

2.3.4 Exotic understory stem density

The mean exotic understory stem density, between 0.3 and 12 m, increased from the rural forest category, excluding the Waingaro Forest data, to the youngest urban forest category (Table 2.8 and Figure 2.7). The minimum, maximum and range also followed the same trend. The rural category, including Waingaro Forest, contained the second highest mean, the highest maximum and the largest range and did not support the trend. The statistical analyses (p-value = 0.02) indicated that there was a statistically significant difference in the exotic understory stem density between the rural and urban forest categories. There was one statistical difference within the data (Table 2.9). The rural group, excluding the Waingaro Forest data, contained a significantly lower amount of exotic understory stems than the youngest urban category.

Table 2.8: Descriptive statistics of the number of exotic understory stems (0.3-12 m) per 50 m^2 across the five forest categories and a rural category excluding the Waingaro Forest data.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	10.42	0	50.8	50.8	22.58
Rural excl. Waingaro	0.33	0	1.3	1.3	0.65
Urban 100+ yrs	2.2	0	4.6	4.6	1.91
Urban 25-100 yrs	3.13	0.33	8.33	8.00	3.18
Urban 15-25 yrs	6.00	1.33	16.67	15.33	6.24
Urban 5-15 yrs	10.8	3.33	19.33	16.00	6.10

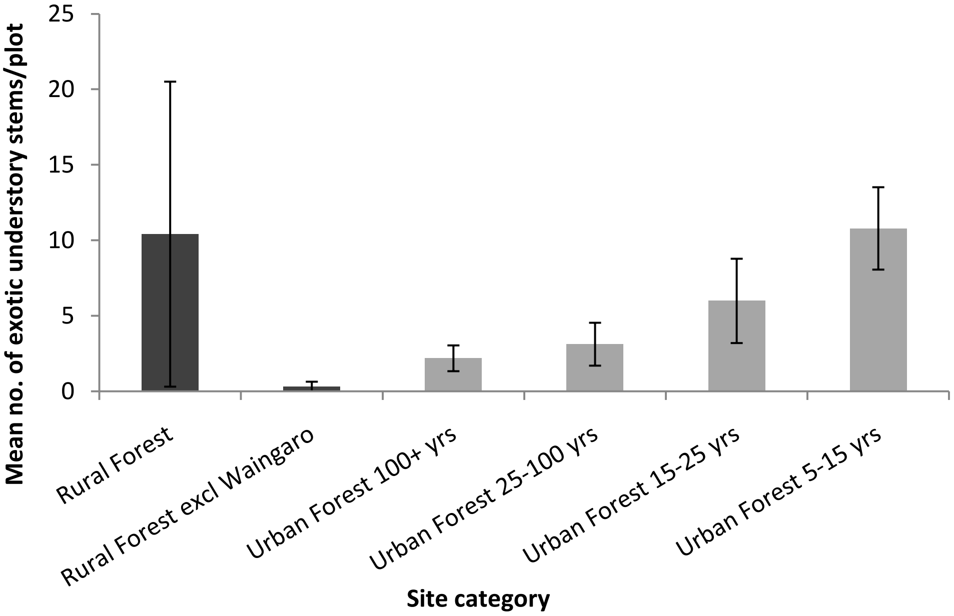

Figure 2.7: The mean number of exotic understory stems per 50 m^2 across the five different categories and a rural category excluding the Waingaro Forest data with standard error bars (Rural groups-black bars, Urban groups-grey bars).

Table 2.9: Non-parametric (Kruskal-Wallis) test for significant differences in the number of exotic understory stems per 50 m^2 across the five forest categories and a rural category excluding Waingaro Forest.

	Rural	Rural excl Waingaro	Urban 100+ yrs	Urban 25-100 yrs	Urban 15-25 yrs	Urban 5-15 yrs
Rural	-	1.00	1.00	1.00	1.00	0.18
Rural excl Waingaro		-	1.00	1.00	0.32	0.02
Urban 100+ yrs			-	1.00	1.00	0.74
Urban 25-100 yrs				-	1.00	1.00
Urban 15-25 yrs					-	1.00
Urban 5-15 yrs						-

2.3.5 PATN ordination

The PATN ordination (Figure 2.8) for the 25 forest sites showed a gradient of understory vegetation, between 0.3 to 12 m, across the sites and categories. The forest sites from the rural and two oldest urban groups were clustered within their respective categories and were found in a sequence of oldest to youngest from the top to the bottom of the ordination. This indicated that the understory vegetative components of the rural old-growth forest group were most closely related to the oldest urban forest group than any of the other categories.

Trend lines and R^2 values of the x and y coordinate regressions were calculated for a range of floristic and structural variables. Native understory species richness (Figure 2.9) showed a statistically significant relationship with the x and y coordinates of the PATN ordination ($R^2 = 0.32$, $p < 0.01$ and $R^2 = 0.55$, $p < 0.01$). Similarly understory stem density (Figure 2.10) revealed a statistically significant but lesser correlation with the x and y coordinates of the PATN ordination ($R^2 = 0.23$, $p = 0.02$ and $R^2 = 0.37$, $p < 0.01$).

On the PATN ordination the native understory species richness and understory stem density was highest at the top before decreasing further down the ordination. With some exceptions (Mooney Park) the two gradients correlated with the age of the urban forest sites. The forests at the top of the ordination were the oldest and became younger further down the ordination, until the youngest two urban categories were reached, where the locations then appear mixed. The ordination and regression analyses also showed that the rural forests contained higher native understory species richness and understory stem density than the urban forests aged over 100 years.

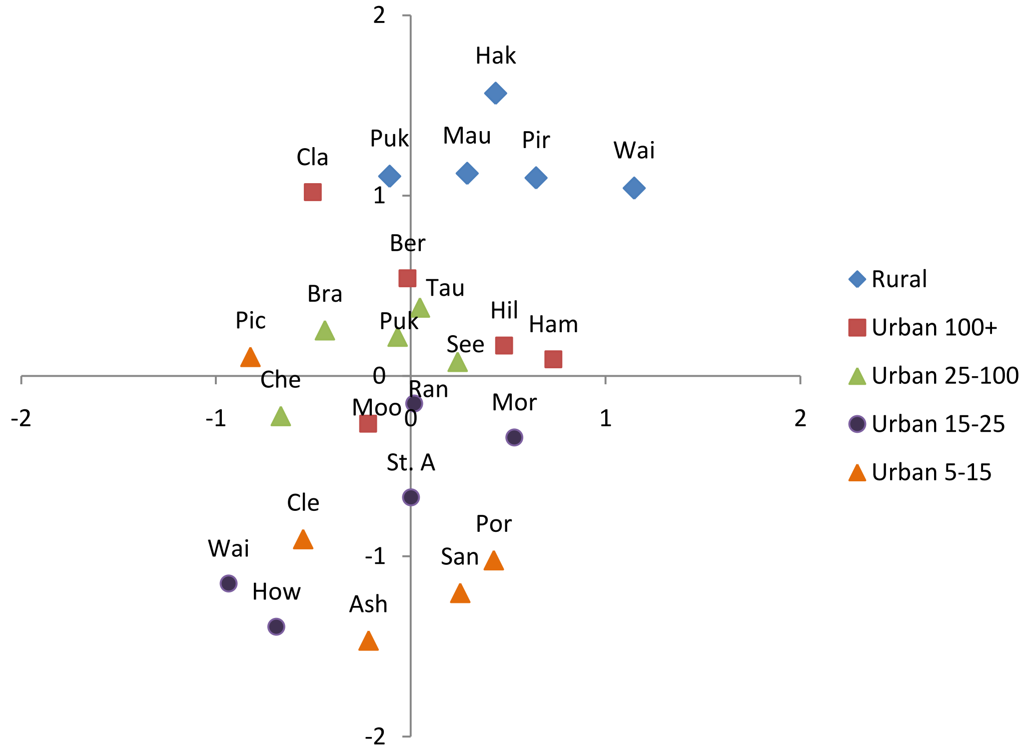

Figure 2.8: Ordination comparing the understory species (0.3-12 m) across the 25 forest sites surveyed.

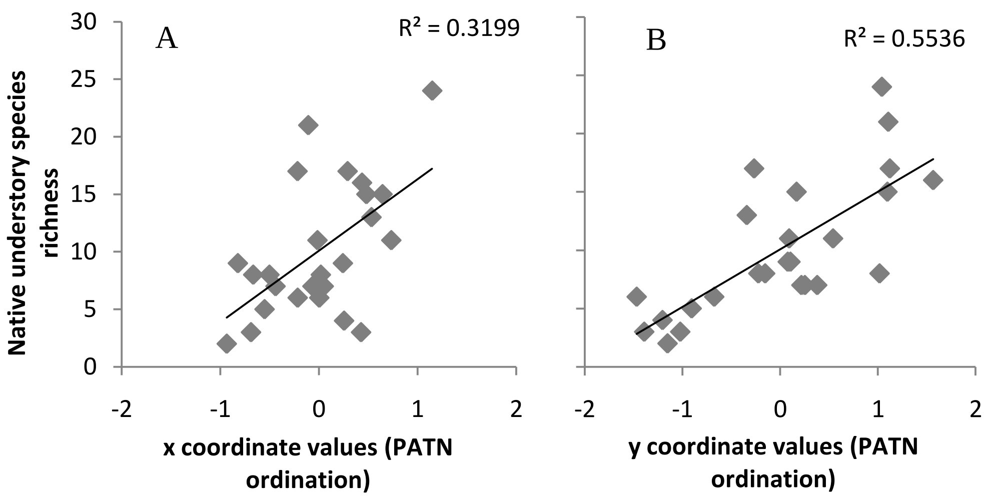

Figure 2.9: Regression of the native understory species richness of the 25 forest sites against the x (A) and y (B) coordinates from the PATN ordination.

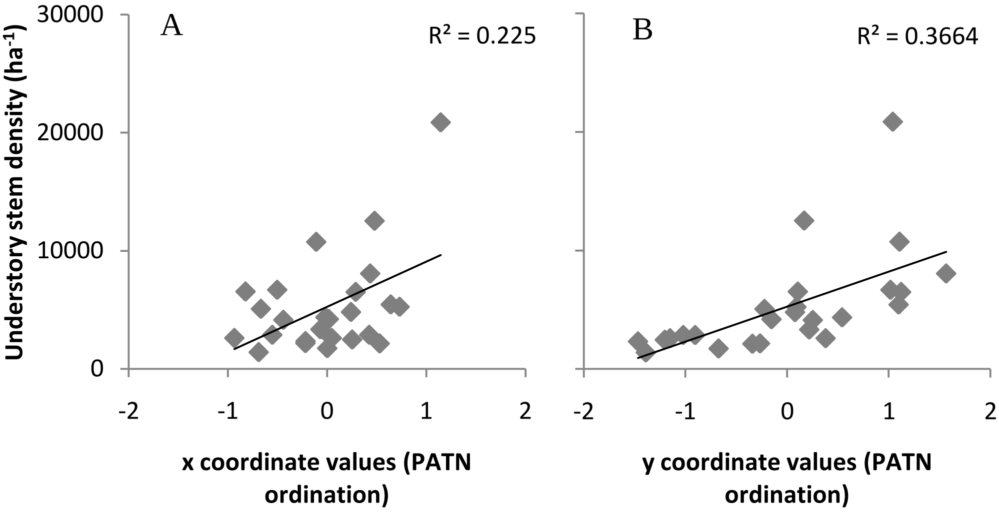

Figure 2.10: Regression of the understory stem (0.3-12 m) density (ha^{-1}) of the 25 forest sites against the x (A) and y (B) coordinates from the PATN ordination.

2.4 Discussion

2.4.1 General results and native understory species richness and density

Native understory species richness results showed that the collective urban forest sample contained just 61.5% of the native understory species present in the rural old-growth forests. Both native understory species richness and density decreased from the rural old-growth forest category down through to the youngest urban forest group. High species diversity cannot be expected in early forest development; however native species richness usually increases as a forest matures (Honnay et al. 1999b; D'Amato et al. 2009). This is evident in the results; however there is still an absence and lower abundance of native understory species from both early and late succession stages in the urban forests assessed. The understory native stem density was also reduced in the later succession urban forests compared to the rural old-growth forests.

Reduced native understory species richness and density in urban forests are caused by a combination of factors, mainly stemming from urbanisation (Guntenspergen & Levenson 1997). These factors include fragmentation and isolation (Godefroid & Koedam 2003), unfavourable environmental conditions (D'Amato et al. 2009), urban pressures such as vandalism and invasive exotic species (Lehvavirta & Rita 2002; Spyreas & Matthews 2006; D'Amato et al. 2009) and disturbance and forest management (Bhuju & Ohsawa 2001). All of these factors are likely to have contributed to the reduced native understory species richness and density observed in Hamilton urban forest fragments.

The Waikato Region consists of 8207 isolated and fragmented lowland forests with 95% of these forests less than 25 ha (Innes 2010). Forest remnants are scattered throughout the Hamilton Basin in both rural and urban areas (Leathwick et al. 1995). These urban forest fragments contain a higher edge to area ratio than the rural old-growth forests they are compared to in the present study. Reduced forest area in urban forests is a major cause for lower species diversity as fragmentation leads to area-dependent extinction through certain species requiring a minimum threshold of suitable habitat in order to sustain a viable population (Honnay et al. 1999b; Smith & Smith 2001; Godefroid & Koedam 2003). The urban forests are also isolated from the rural old-growth forests, separated by urban infrastructure and agricultural areas. Activity and densities of the organisms that plants rely on for pollination and dispersal (e.g. *Hemiphaga novaeseelandiae*, kereru) are reduced in isolated forests and this prevents the spread of plants and results in decreasing diversity (Ramos & Santos 2006; Spyreas & Matthews 2006).

The environmental conditions in fragmented urban forests, such as in the present study, are often modified, compared to undisturbed old-growth forests. Changes to soil often include the addition and removal of materials, compaction and increased impervious area (Pickett et al. 2001). Fragments also comprise more variable temperatures, humidity and soil moisture and light transmittance levels compared to old-growth forests (Chen et al. 1995). Altered environmental conditions in the urban forests prove unfavourable for the existence of some plant species and reduce species richness (D'Amato et al. 2009).

The urban forests in Hamilton are used extensively for recreation and educational purposes and trampling and other related activities are common (e.g. Seeley’s Gully). Trampling changes the composition and structure of forests through preventing regeneration and succession (Bhuju & Ohsawa 1998; Malmivaara et al. 2002). Vandalism, litter deposition and pollution are common in Hamilton urban forests (e.g. Seeley’s Gully and Hillcrest Park). They also have adverse effects on native plants, reducing forest diversity (Guntenspergen & Levenson 1997; Pauleit et al. 2002). Invasive species are problematic in many Hamilton urban forests investigated (e.g. Porritt Stadium contained extensive exotics including *Tradescantia fluminensis*, *Ligustrum* species and *Solanum* species). Adventive species change forest structure and reduce native diversity by out-competing and preventing the establishment and regeneration of native species (Honnay et al. 1999a).

Agricultural and tree felling practices have occurred in and around many of the Hamilton urban forests (Downs et al. 2000a; Downs et al. 2000b; Gudex 1955) and have subsequently reduced native understory plant richness and density (e.g. Claudelands Bush). Grazing reduces plant diversity both directly and indirectly, through livestock trampling and consuming plants and compacting soil and increasing erosion which retard regeneration and succession (Wassie et al. 2009). Tree harvesting compacts and erodes soil and alters the structure of forests through removing trees which effects subsequent plant regeneration. The adverse effects of agriculture and milling can last for centuries as the resulting changes are not easily reversed (Spyreas & Matthews 2006).

Management practices in Hamilton urban forests have been variable, with some sites benefiting from enrichment plantings (e.g. Mooney Park) (Downs et al. 2000a), weed control (e.g. Berkley Bush), the creation of defined paths and fenced walkways (e.g. Hillcrest Park) or combinations of all three and other actions (e.g. Claudelands Bush). Effective management is required to maintain plant diversity. The total effort of management practices and where the effort is directed contributes towards native species richness (Guntenspergen & Levenson 1997; Bhuju & Ohsawa 2001; McLachlan & Bazely 2001).

Related studies have found similar species richness results to the present study, where species richness is greatest in old-growth forests compared to disturbed and second-growth forests. D'Amato et al. (2009) compared the understory of an old-growth *Tsuga canadensis* (hemlock) forest with a second-growth forest that originated late in the 19^{th} century in western Massachusetts. Despite similarities in canopy composition, distinct differences between the understory of old-growth and second-growth hemlock forests were found. The old-growth forest contained higher species richness and diversity, while the second-growth forest contained only four (33%) of the 12 shrub species found in the old-growth forest. Qian et al.

(1997) evaluated the understory vascular vegetation of five different forest types that comprised both old-growth and 40 year old plantations on Vancouver Island, Canada. Of the five measured forest types, the second-growth forests contained between 23.0% and 62.2% of understory diversity compared with the corresponding old-growth forest. These studies document results which are lower and similar to what was observed in the present study where the old-growth and secondary-growth urban forests collectively contained just 61.5% of the understory plant species found in the rural old-growth forests.

Guntenspergen and Levenson (1997) assessed plant species composition changes in forest fragments along an urban to rural gradient in Milwaukee, Wisconsin. The objective of the study was to determine if understory vegetation was affected by landscapes with differing types of development including urban, urbanising and rural. Similar results to the present research were found in their study, with a noticeable absence of shrub species in urban forest stands. However, Toniato and de Oliveira-Filho (2004) contradict the observed trend in the present study of reduced plant diversity in second-growth forests with their studies comparing old-growth and second-growth submontane seasonal semi-deciduous forests in Brazil. Diversity was higher in the 40 year old second-growth forest, compared to the old-growth forest. The higher diversity in second-growth forests was thought to relate to the intermediate succession theory, where a forest in intermediate succession contains both early and late succession species (Horn 1974). However, there were fewer shade-tolerant and late succession species in the second-growth forest.

2.4.2 Exotic stem density

The trend of increasing exotic stems from the rural forest excluding Waingaro Forest to the youngest urban forest categories could be due to the low native plant diversity and higher availability of exploitable niches in younger forest ecosystems. Many non-native invasive exotics have characteristics which allow them to quickly exploit and dominate forest ecosystems (Cronk & Fuller 1995; Myers & Bazely 2003) such as *Ligustrum* (privet) and *Solanum* species.

The rural old-growth forests from the present study comprised a high diversity of native species in the understory compared to the young urban forests which provided more available ecological niches. Mature forests typically contain a high diversity of native species in a variety of habitats, preventing invasive exotics from becoming established. Newly restored forest ecosystems have not achieved a ‘stable’ state and provide the most favourable conditions for invasive exotics to take advantage of (Cronk & Fuller 1995; Myers & Bazely 2003).

Many of the urban forests in Hamilton are surrounded by residential areas, which contain exotic plant species. Humans are important vectors for the spread of exotic species as they provide a seed source (Hill 1977). The close proximity of the exotic species to the urban forests, which is evident in Hamilton, allows adventive species to spread more easily into younger forests that contain more available ecological niches (Cronk & Fuller 1995).

2.4.3 PATN ordination

The ordination clustered the forest sites of the rural and two oldest urban groups within their respective categories, indicating close relationships between understory vegetative components. There are a variety of options that could potentially explain the relationship and are not necessarily mutually exclusive. The linear regression analyses suggested that native understory species richness and understory stem density were the two factors that best explained the patterns in the PATN ordination.

These two factors produced gradients which correlated most strongly with the age of the forest sites. Generally, as forest age deceased, the native understory species richness and understory density decreased. This is characteristic of forest succession where understory stem density and native species richness increase as forests develop (Honnay et al. 1999b; D'Amato et al. 2009). The reduced native understory species richness and understory stem density in the oldest urban forests, compared to the rural old-growth forests, further emphasised the absent and reduced native understory species composition and density results obtained from the earlier analyses.

3 Chapter three: Assessment of environmental variation

3.1 Introduction

This chapter assess the relative importance of selected environmental variables with regard to the performance, growth and regeneration of understory species. Nine environmental attributes were measured and used to develop an environmental profile for selected study sites and forest classes. The results are used to examine how each of the variables might influence habitat conditions and explain the reduced native understory composition and density in urban forests compared to rural old-growth forests identified in chapter two.

3.1.1 Environmental conditions

Different environments support a range of species composition and diversity with environmental conditions being abiotic such as substrate, soil moisture and nutrient supply. Environmental conditions are an important influence on plant communities and are essential for plant processes such as regeneration and growth (Chen et al. 1999; Smith & Smith 2001). A change in one or more abiotic conditions can alter the composition and structure of plant communities. Plant species have specific tolerances or ranges in relation to environmental conditions, thus plants exploit and colonise environments which provide the most favourable surroundings (Crawley 1997b; Smith & Smith 2001). Key environmental variables include soil type, nutrient levels, pH and moisture content, light transmittance and temperature and vapour pressure deficit.

The Waikato Region including Hamilton City contains a wide variety of soil types, with distinctive soil properties and distribution patterns (Singleton 1991). Soil types exhibit variable properties, such as texture, drainage and mineral composition, which affect plant growth (McLaren & Cameron 1996). To grow successfully, plants require 17 nutrients and elements, including carbon, nitrogen and phosphorus (McLaren & Cameron 1996; Nabors 2004). Generally, as forests become more mature carbon, nitrogen and phosphorus levels increase (Grier et al. 1989) while the soil becomes more acidic (Brais et al. 1995). Soil nutrient status differs depending on factors such as parent material, formation, land management practices, adjacent land use and acidity (McLaren & Cameron 1996; Nabors 2004).

Different nutrients also have varying availabilities for plants depending on soil acidity or alkalinity. For example nitrogen is least available in acidic soils, while phosphorous is at maximum availability between a pH of 6 and 7. Above and below these values the phosphorous availability decreases. Excessively acidic or alkaline soils can result in nutrient deficiency or toxicity, depending on the mineral and species involved (McLaren & Cameron 1996).

The amount of water available to plants is dependent on soil water content and soil properties, including the soil structure and particle size distribution. Soil water content is influenced by the amount of precipitation in the area as well as drainage, evaporation, runoff and plant uptake of water (McLaren & Cameron 1996). On forest edges and in small forest patches the soil water content is often reduced, compared to the interior of larger intact forests, due to increased light and wind exposure and the resulting higher levels of evapotranspiration (Kapos 1989).

There are many factors which can affect light direction, intensity and duration, including the time of day and season, latitude, altitude and the topography. In forests, the majority of light is intercepted by the canopy, decreasing the amount of photosynthetically active radiation available for the understory vegetation. The amount and quality of light received in the understory depends on the density of the canopy, leaf properties (size, shape, optical properties and inclination angle) and the number and size of gaps in the canopy (Smith & Smith 2001).

Temperature and vapour pressure deficit of an environment can influence the distribution of plant communities depending on the ranges a species can tolerate (Nabors 2004). Forests often contain a distinctive temperature and vapour pressure deficit pattern compared to outside the forest. Within forests, the temperature and vapour pressure deficit fluctuate less, compared to outside the forest. This is due to the vegetation acting as a buffer, reducing the extremes of high and low temperature relative to outside the forest (Kapos 1989; Chen et al. 1999).

3.1.2 Environmental changes in urban forests

Urbanisation can affect the environmental conditions of urban forests through modifying the structural integrity of the forest ecosystem. For example, fragmentation of urban forests creates an edge effect, altering soil moisture and light transmittance levels due to increased exposure (Kapos 1989; Chen et al. 1999). Some environmental variables, such as temperature and vapour pressure deficit, are particularly susceptible to structural changes to forests and to the surrounding landscape, and can show marked temporal and spatial variation as a result (Vallet et al. 2010; Chen et al. 1999).

In undisturbed forest ecosystems the forest edge usually transitions gradually from a forest canopy to grassland over large distances and steadily decreases in vegetation height. Urban forest remnants often end abruptly and are subject to more pronounced edge effects that cause altered environmental conditions compared to the internal area of a larger, intact forest (Porteous 1993). Such changes in environmental conditions can cause differences in species composition on the forest edge and within remnants compared, to the interior of more extensive forests (Young & Mitchell 1994). Urbanisation also influences soil function of forests through physical disturbances, water, chemical and organic additions and increasing impervious material within and around the forest (Pickett et al. 2001).

3.1.3 Objective

This chapter identifies how selected environmental variables vary between different aged urban forests and rural old-growth forests. Soil types, nutrients, pH, and moisture levels, light transmittance into the understory, temperatures and vapour pressure deficits across the forest categories are quantified. Results are assessed to determine whether altered environmental conditions might be responsible for the reduced native understory composition, richness and density in urban forest categories compared to rural old-growth forests observed in chapter two. The likely relative importance of the different environmental variables is also assessed.

3.2 Methods

The methods used to assess and measure each of the nine key environmental variables are outlined below.

3.2.1 Selected study sites

A subsample (15) of the forest sites used in the understory vegetation research component was used to assess the environmental variables in this chapter and the plant reintroduction experiment in chapter four (Table 1.1, chapter one). Soil type, nutrient status, pH and moisture levels, and light transmittance into the understory were measured at the 15 forest sites. Temperature and vapour pressure deficit were measured at five of the 15 forest sites from the subsample.

3.2.2 Soil types

Soil types were obtained using soil maps of Hamilton City (Landcare Research 2002; McCormack 1979) and the Waikato Region (New Zealand Soil Bureau 1954b). The scale of the Waikato Region map used to identify the rural old-growth forest soil types was a limitation as the map was less precise than the Hamilton maps. Hewitt's (1998) 'New Zealand soil classification' was also used to clarify the New Zealand genetic soil classification and provide information on the soil types.

3.2.3 Soil nutrients and pH

One position from each location was chosen at random and 5 cm of the top soil was removed before the newly exposed soil was sampled using a trowel. Enough soil was collected to half-fill a 1400 cm^3 snap lock bag. The trowel was washed with water before each use between sites.

In the laboratory the soil samples were sieved through 2 mm mesh before being left to air dry. Samples were then divided so they could be used in the three different analyses; Olsen P tested for available phosphorous; soil pH measured acidity/alkalinity; and a Leco Truspec CN machine was used to measure total carbon and total nitrogen. The Olsen P test was based on the methods used in Blakemore et al. (1987) where the concentration is measured using a spectrophotometer at 880 nm.

Total carbon and total nitrogen were measured by staff from the Waikato Stable Isotope Unit using the Leco Truspec CN machine. Four grams of each soil sample were weighed out, crushed and ground down into a fine powder. Approximately 2.5 mg of each powder sample was weighed out and wrapped up in tinfoil. Analysis follows the theory of LECO Corporation (2006) where carbon was measured as carbon dioxide by a carbon dioxide detector while a thermal conductivity cell was used to determine the nitrogen content.

Soil pH was measured following methods described by Blakemore et al. (1987) by making a standard soil solution and using a calibrated pH meter. Calibration was undertaken using the standard procedures that came with the instrument and buffers of known pH.

3.2.4 Soil moisture

Volumetric soil water content readings (%) were obtained at one month intervals using a time domain reflectometry probe (Hydrosense CS 620 Campbell Scientific, Logan, Utah). Forty five soil moisture measurements were taken at each forest site. Measurements were taken in the vicinity of the reintroduced plantings, used in the chapter four research component (five measurements around each planting), to maintain accurate and precise repeated sample locations. The 20 cm probe was pushed vertically into the ground 20 to 50 cm from the plant. Caution was taken to avoid taking a measurement in the disturbed soil from the reintroduced plantings.

3.2.5 Light transmittance

Light transmittance through the canopy was estimated using hemispherical photography (Machado & Reich 1999) and image analysis software (Gap Light Analyzer') (Frazer et al. 2001). Photographs were taken twice, during the middle of winter and towards the end of spring to capture the effects of exotic deciduous tree species, present at eight of the nine forests within the three youngest urban forest groups, on seasonal changes in light transmittance.

Photographs were taken using a digital camera and hemispherical lens (Nikon Coolpix 990 and FC-E8) under uniform overcast sky conditions. Exposure was set at 2 F-stops (over exposed) above a metre reading on the sky only.

Between 14 and 19 photographs of the canopy were taken at each of the forest sites. Nine photographs were taken directly above each of the reintroduced plantings. A further five to ten photographs were taken at a location within the forests using a random number table to determine the distance and direction to the site. All the photographs were taken between 0.5 and 1.5 m above the ground.

Photographs were uploaded onto the computer and the Gap Light Analyzer software was used to analyse the total light transmittance of each photograph. Light transmittance was estimated for the whole year for both sets of photographs (winter and spring).

3.2.6 Temperature and vapour pressure deficit

Temperature and relative humidity were recorded in forests using micro data-loggers (iButton, DS1923, Maxim, Indiana). All data-loggers were cross calibrated before use by placing them in a range of stable temperature and relative humidity environments, enclosed in a glass chamber. Temperature environments included inside a fridge (4 ºC) and oven (36 ºC), and inside a room with set temperatures (18 ºC and 20 ºC). Relative humidity environments were obtained by placing different saturated salt solutions into the enclosed glass chamber in a room with a constant temperature of 20 ºC. Saturated salt solutions included sodium chloride (75% Relative Humidity), potassium chloride (85% RH), magnesium chloride (33% RH) and magnesium nitrate (54% RH). All subsequent measurements were corrected using regression equations obtained from the calibration exercise.

When deployed in the field, all data-loggers were shielded from direct solar radiation by enclosure in stacked plate Gill radiation shields (Tarara & Hoheisel 2007), either commercially produced (Radiation (Stevenson) Shield, HortPlus, Hastings) or custom made. Custom made shields were fabricated from six plates, three 8.2 cm lengths of thread and 15 12 mm plastic spacers. Tests confirmed that the two different types of solar radiation shields used had no significant effects on the results.

The number of sites that could be monitored for temperature and relative humidity was limited by the number of data-loggers available. Fifteen data-loggers were therefore positioned in five different sites; six data-loggers were deployed in two rural forest sites and nine data-loggers into three urban forest sites; old-growth over 100 years, 25 to 100 years and 15 to 25 years. Forest sites were selected to avoid the risk of the data-loggers becoming damaged or stolen. Sites included Waingaro Forest, Mount Maungatautari Ecological Island, Berkley Bush, Chelmsford Park and Hudson Gully on Howell Avenue. At each of the five sites two data-loggers were deployed within the forest and one in an exposed site outside the forest. A second rural forest site was chosen over an urban forest aged between 5 to 15 years because it was thought the rural site would show more variation and the urban forest would be similar to an urban forest aged between 15 to 25 years.

Data-loggers were placed into the solar radiation shields and tied from tree branches between 1 and 2 m off the ground. The data-logger serial number, location, geographic coordinates and a brief description of the area was recorded. Data-loggers were set to record hourly and were downloaded every two months. Their position was changed to a different random location within the forest when they were downloaded to minimise the impact of individual microsites on the overall result. Data was downloaded using the software package 'OneWireViewer'. Vapour pressure deficit values were obtained using the equation: VPD = $e_s - e$, where 'e_s' is the saturation vapour pressure and 'e' is the measured vapour pressure. Negative vapour pressure deficit values were occasionally estimated due to data-logger measurements of humidity above 100% (the accuracy of the humidity measurement was reduced in high relative humidity). Data was divided up into winter (1st of June to 31st of August) and spring (1st of September to 30th November) to produce mean diurnal temperature and vapour pressure deficit for the two seasons.

3.2.7 Analyses

Statistical analyses on all measured environmental parameters were undertaken using the software package 'Statistica' version 9.0. Descriptive statistics provided the mean, maximum, minimum, range and standard deviation. The carbon, nitrogen, Olsen P and pH data was analysed for differences between categories using the one-way Analysis of Variance (ANOVA) test. Soil moisture data was analysed using one-way ANOVA and repeated measures ANOVA tests. The temperature and vapour pressure deficit data was analysed using non-parametric sign tests. Light transmittance was analysed using one-way ANOVA, Levene's test for homogeneity of variance, Shapiro-Wilk test for normality, post-hoc analysis fisher test and the non-parametric sign test. A difference was considered to be statistically significant if a p-value was obtained that was equal to or less than 0.05 and the associated assumptions, if required for ANOVA, were met.

3.3 Results

3.3.1 Soil type

Seven major soil types were represented between the 15 forest sites (Table 3.1). All three rural forests were mapped as skeletal soils, whilst the urban forests comprised a variety of different soil types. The predominant soil type in urban forests was allophanic, occurring in 11 of the 12 locations. Gley and recent soils were the next most abundant soil types featuring in six and five of the 12 sites respectively. Pumice, anthropic and brown soil types were also present in three, two and one locations respectively.

Table 3.1: The 15 study locations in the five forest categories and their corresponding soil types.

Forest Category	Site	Soil Type
Rural	Waingaro Forest	Skeletal (Recent on terrace)
	Mount Maungatautari	Skeletal
	Pukemokemoke Bush Reserve	Skeletal
Urban 100+ yrs	Hammond Bush	Allophanic, Pumice
	Hillcrest Park	Allophanic, Gley
	Berkley Bush	Gley
Urban 25-100 yrs	Chelmsford Park	Allophanic, Gley
	Seeley's Gully Reserve	Allophanic, Gley, Recent
	Pukete Riverbank Forest	Allophanic, Pumice, Recent
Urban 15-25 yrs	Howell Street	Allophanic, Gley
	St. Andrews	Allophanic, Anthropic, Pumice
	Ranfurly Park	Allophanic, Recent
Urban 5-15 yrs	Sandford Park	Allophanic, Recent
	Porritt Stadium	Allophanic, Brown, Gley
	Pickering Crescent	Allophanic, Anthropic, Recent

3.3.2 Soil nutrients

The mean Olsen P values (Table 3.2) showed a trend of increasing concentrations from the youngest to the oldest urban forest category, although the value of the urban forest group aged 25 to 100 years did not fit this trend. The minimum and maximum concentrations followed the same trend. Hillcrest Park and Seeley's Gully Reserve produced inconclusive results and were not used in the analysis. According to the criteria in Blakemore et al. (1987) the oldest and third oldest urban forest categories contained a high Olsen P rating, the urban forest group aged 25 to 100 years was considered medium whilst the rural and youngest urban forest categories were judged as low. All the urban forest categories contained a higher mean Olsen P concentration than the rural forest group. No statistically significant differences were detected.

Mean total nitrogen values (Table 3.3) demonstrated a trend of increasing concentration from the youngest to the oldest urban forest category. The minimum and maximum concentrations followed a similar trend although there were some inconsistencies. In reference to the criteria in Blakemore et al. (1987), the rural and two oldest urban forest categories had high total nitrogen content while the two youngest urban forest groups were considered to have a medium rating. The rural forest category contained a mean nitrogen concentration that was lower than the two oldest urban forest groups. No statistically significant differences were detected.

The mean total carbon values (Table 3.4) displayed a trend similar to that of Olsen P and total nitrogen, where concentration increased from the youngest to the oldest urban forest category. The minimum and maximum concentrations followed a similar trend although with some discrepancies. The Blakemore et al. (1987) rating system suggested that the two oldest urban forest groups had a high carbon content while the rural and two youngest forest categories contained medium carbon content. The rural forest category contained a mean carbon concentration that was lower than the two oldest urban forest groups. No statistically significant differences were detected.

Table 3.2: Descriptive statistics of Olsen P concentrations (μg g^{-1}) across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	6.95	5.48	9.14	3.67	1.94
Urban 100+ yrs	26.86	19.92	33.79	13.87	9.81
Urban 25-100 yrs	14.80	3.99	25.61	21.62	15.28
Urban 15-25 yrs	22.67	17.60	30.50	12.90	6.87
Urban 5-15 yrs	9.99	3.99	17.32	13.33	6.77

Table 3.3: Descriptive statistics of total nitrogen concentrations (%) across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	0.65	0.41	1.08	0.67	0.37
Urban 100+ yrs	0.94	0.83	1.01	0.18	0.10
Urban 25-100 yrs	0.77	0.14	1.52	1.38	0.70
Urban 15-25 yrs	0.48	0.21	0.89	0.67	0.35
Urban 5-15 yrs	0.38	0.31	0.51	0.19	0.11

Table 3.4: Descriptive statistics of total carbon concentrations (%) across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	7.79	3.94	13.79	9.85	5.27
Urban 100+ yrs	15.77	12.76	19.06	6.30	3.16
Urban 25-100 yrs	10.85	1.19	21.54	20.35	10.22
Urban 15-25 yrs	6.18	1.94	11.22	9.28	4.69
Urban 5-15 yrs	4.21	3.19	5.95	2.76	1.51

3.3.3 Soil pH

The mean soil pH levels (Table 3.5) illustrated a pattern of increasing acidity from the youngest to the oldest urban forest category. The pattern was less obvious within the minimum and maximum data. The rating system, from Blakemore et al. (1987), suggested that the pH in the rural forest group was low, the two oldest urban forest groups were very low and the two youngest urban forest categories were medium. The two oldest urban forest categories were more acidic than the rural forest group. No statistically significant differences were apparent.

Table 3.5: Descriptive statistics of pH levels across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	5.24	4.87	5.82	0.95	0.51
Urban 100+ yrs	4.06	4.00	4.12	0.12	0.06
Urban 25-100 yrs	4.45	3.4	5.03	1.63	0.91
Urban 15-25 yrs	5.42	5.05	5.88	0.83	0.42
Urban 5-15 yrs	5.30	4.62	6.29	1.67	0.88

3.3.4 Soil moisture

Volumetric soil water content (Table 3.6) varied between the forest categories with no obvious relationship to forest age or urban and rural forests. No statistically significant differences were detected. Soil moisture levels at the five forest categories had comparable increases and decreases over time (Figure 3.1) but there were no statistically significant differences between groups.

Table 3.6: Descriptive statistics of soil moisture (%) over the measured period across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	46.61	35.63	61.73	26.09	13.53
Urban 100+ yrs	31.85	28.98	35.21	6.24	3.15
Urban 25-100 yrs	52.48	41.84	67.63	25.79	13.47
Urban 15-25 yrs	35.31	19.58	52.70	33.13	16.62
Urban 5-15 yrs	44.06	31.06	65.26	34.20	18.52

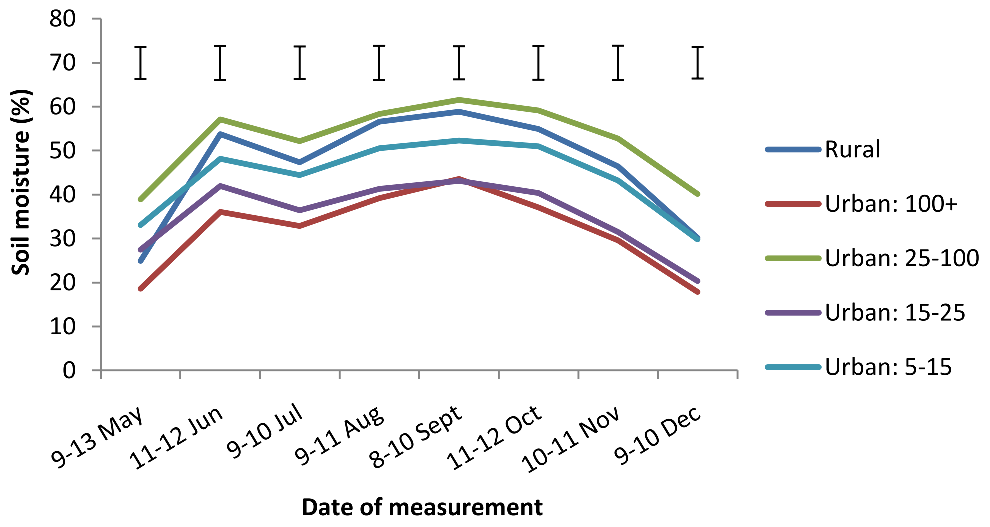

Figure 3.1: Soil moisture levels across the five forest categories from May to December with standard error bars.

3.3.5 Light transmittance

Mean total light transmittance of the five forest categories during winter (Table 3.7) showed an increasing trend from the rural old-growth forest group to the youngest urban forest category. The oldest urban forest group was only slightly higher than that of the rural forest category. The minimum and maximum values followed a similar trend although with some discrepancies. The p-value (0.02) suggested that there was a statistically significant difference in total light transmittance between the five forest categories. Further analyses (Table 3.8) revealed that the total light transmittance of the rural and oldest urban forest categories was statistically different (lower) to the youngest urban forest group.

Similarly, the mean total light transmittance of the five forest categories during spring (Table 3.9) showed a trend of increasing light transmittance from the rural old-growth forest group to the youngest urban forest category. The oldest urban forest group was only slightly higher than that of the rural forest category. The minimum and maximum values followed a similar trend although with some inconsistencies. The p-value (<0.01) indicated that there was a statistically significant difference in total light transmittance between the five forest categories. Further analyses (Table 3.10) revealed that the total light transmittance of the urban forest category aged between 15 to 25 years was statistically different (higher) to the rural and oldest urban forest groups and that the youngest urban forest group was statistically different (higher) to all the other categories.

The majority of locations contained higher light transmittance values during winter compared to spring (Figure 3.2). The differences between values across the locations varied in magnitude. The p-value (<0.01) indicated that there was a statistically significant difference between winter and spring mean total light transmittance values.

Table 3.7: Descriptive statistics of total light transmittance (%) across the five categories in winter.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	5.91	3.60	8.18	4.58	2.29
Urban 100+ yrs	6.46	4.72	7.60	2.88	1.53
Urban 25-100 yrs	11.45	3.58	17.99	14.41	7.29
Urban 15-25 yrs	13.11	11.59	14.06	2.47	1.33
Urban 5-15 yrs	18.43	15.66	23.61	7.95	4.49

Table 3.8: Post-hoc analysis (Fisher test) for significant differences in the total light transmittance across the five different categories in winter.

	Rural	Urban 100+ yrs	Urban 25-100 yrs	Urban 15-25 yrs	Urban 5-15 yrs
Rural	-	0.87	0.13	0.06	<0.01
Urban 100+ yrs		-	0.16	0.07	<0.01
Urban 25-100 yrs			-	0.62	0.06
Urban 15-25 yrs				-	0.14
Urban 5-15 yrs					-

Table 3.9: Descriptive statistics of total light transmittance (%) across the five categories in spring.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	4.17	3.34	4.69	1.35	0.72
Urban 100+ yrs	4.87	4.02	5.54	1.52	0.78
Urban 25-100 yrs	6.39	3.79	8.82	5.03	2.52
Urban 15-25 yrs	8.39	7.54	9.56	2.02	1.05
Urban 5-15 yrs	13.63	11.15	16.23	5.08	2.54

Table 3.10: Post-hoc analysis (Fisher test) for significant differences in the total light transmittance across the five different categories in spring.

	Rural	Urban 100+ yrs	Urban 25-100 yrs	Urban 15-25 yrs	Urban 5-15 yrs
Rural	-	0.63	0.15	0.01	<0.01
Urban 100+ yrs		-	0.31	0.03	<0.01
Urban 25-100 yrs			-	0.19	<0.01
Urban 15-25 yrs				-	<0.01
Urban 5-15 yrs					-

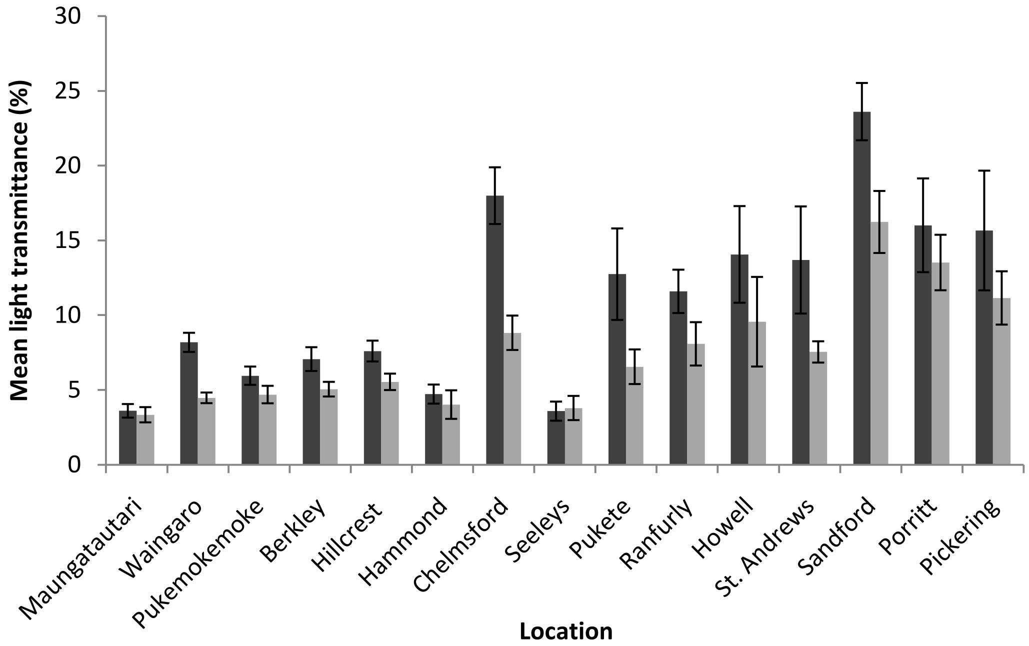

Figure 3.2: Mean light transmittance into the understory for winter (black columns) and spring (grey columns) across the 15 forest sites with standard error bars.

3.3.6 Temperature

Mean diurnal temperature results illustrated some distinctive patterns, with several sites displaying consistently warmer temperatures outside the forest compared to the forest interior during certain time periods. Mean diurnal temperature at the two rural old-growth forests (Figure 3.3) was higher outside the forest throughout the day, particularly in the afternoon, during both seasons.

Temperature in the two urban forest sites, Berkley Bush and Howell Street (Figure 3.4), had similar outside and inside temperatures in the winter but during the spring, a pattern similar to the rural old-growth forests was present.

Temperatures in the forest interiors of Berkley Bush and Howell Street were slightly higher than that of the two rural forests during spring. Chelmsford Park contained similar temperature patterns (Figure 3.4) to that of the rural old-growth forests during both seasons. However the outside forest temperatures, during both seasons, were higher than that of the outside forest temperatures at the other two urban locations.

The statistical analyses suggested there was a statistically significant difference between the mean diurnal temperature outside and inside the forest at Mount Maungatautari Ecological Island, during the winter (p-value <0.01), and Chelmsford Park, during both seasons (both p-values <0.01). Although there were diurnal temperature patterns in Waingaro Forest, during both seasons, and at Mount Maungatautari Ecological Island, Berkley Bush and Howell Street during spring, the differences were not statistically significant.

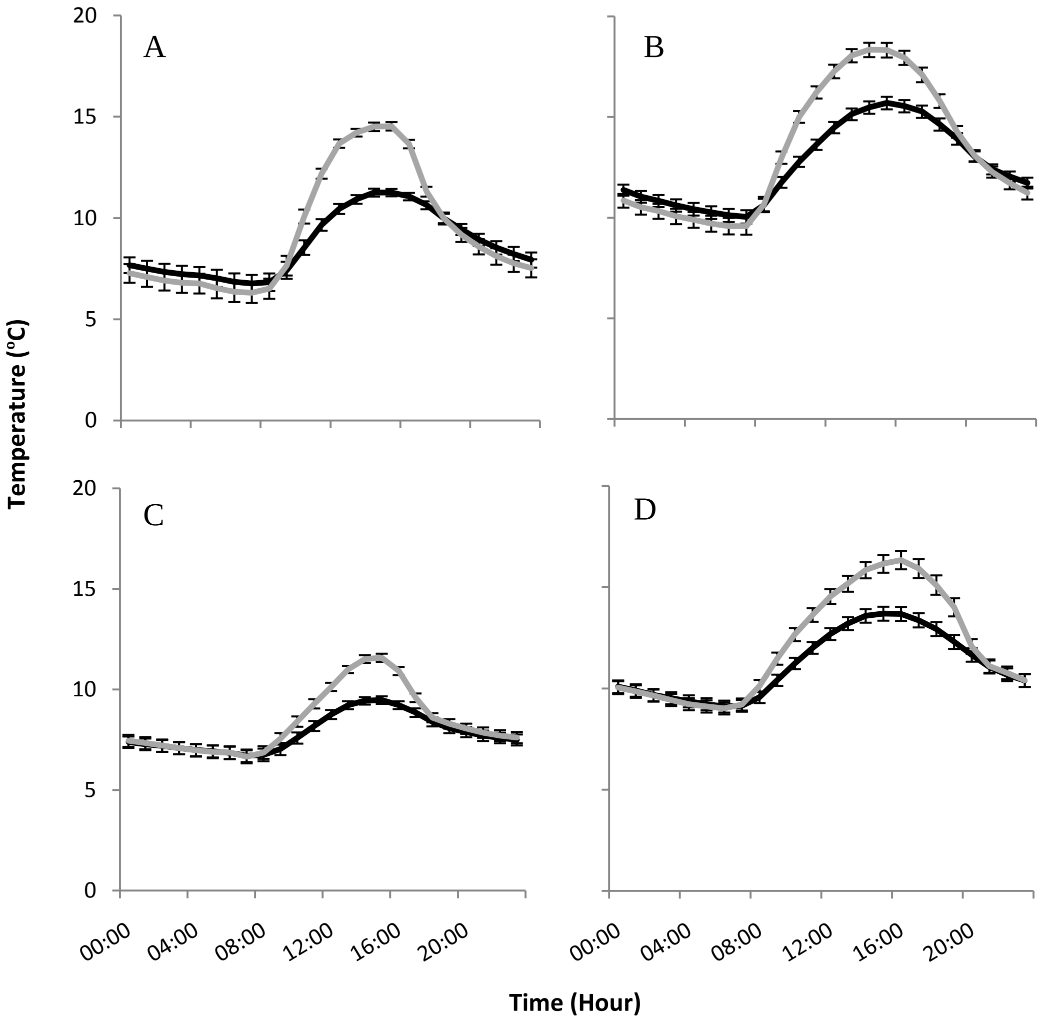

Figure 3.3: Mean diurnal temperature inside (black line) and outside (grey line) the rural old-growth forests of Waingaro Forest (A-winter B-spring) and Mount Maungatautari Ecological Island (C-winter D-spring) with standard error bars.

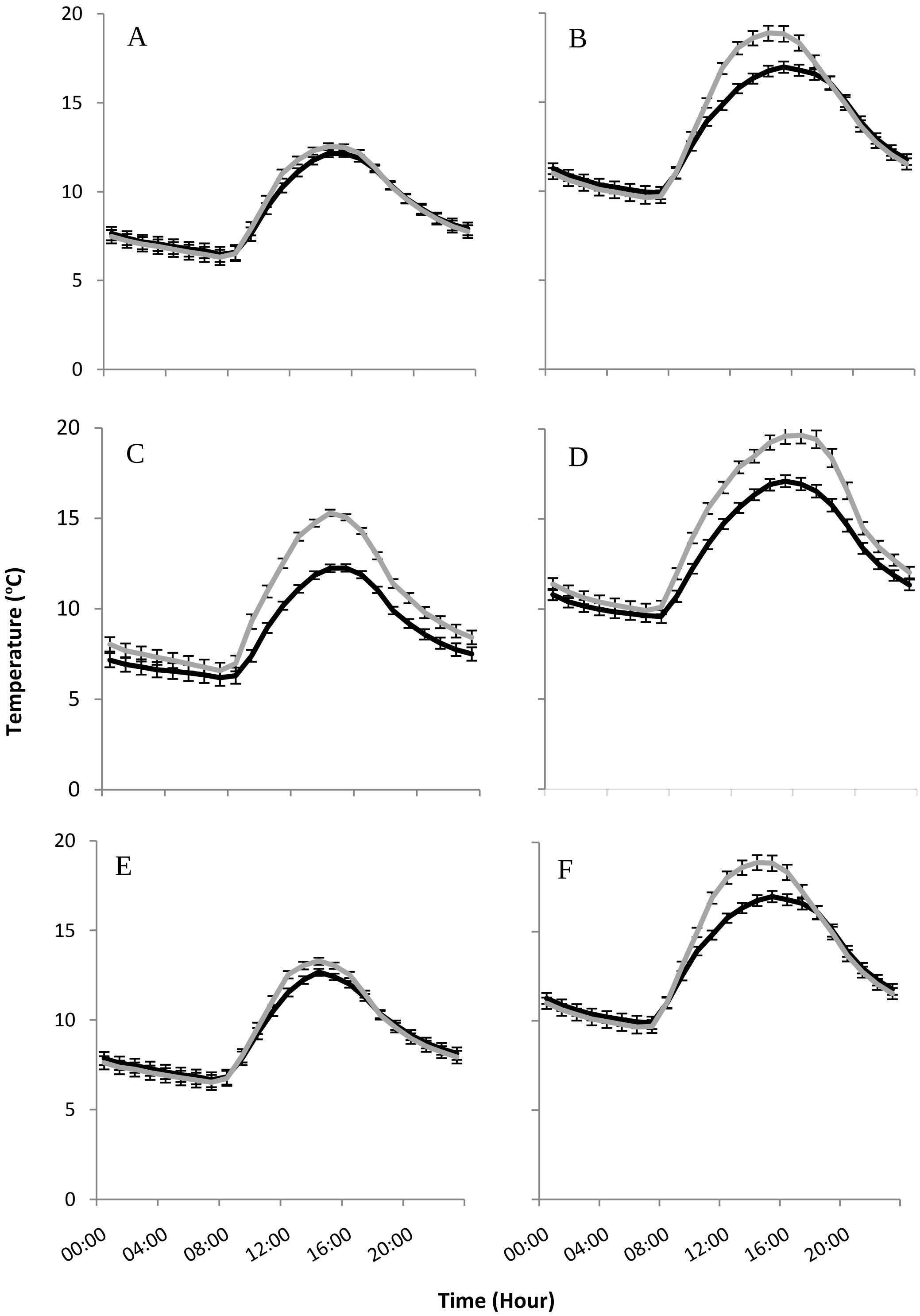

Figure 3.4: Mean diurnal temperature inside (black line) and outside (grey line) the urban forests of Berkley Bush (A-winter B-spring), Chelmsford Park (C-winter D-spring) and Howell Avenue (E-winter F-spring) with standard error bars.

3.3.7 Vapour pressure deficit

The mean diurnal vapour pressure deficit results displayed distinctive trends, with similarities to the temperature results. Several forest sites displayed consistently higher vapour pressure deficits outside the forest compared to the forest interior, during the day, particularly in the afternoon. The vapour pressure deficit at the two rural old-growth forests (Figure 3.5) was higher outside the forest throughout the day during both seasons. The vapour pressure deficit in the two urban locations, Berkley Bush and Howell Street (Figure 3.6), had similar outside and inside levels in the winter but during the spring, a pattern similar to the rural old-growth forests was present. Vapour pressure deficit was slightly higher in the forest interiors of Berkley Bush and Howell Street compared to the two rural forests. Chelmsford Park contained comparable vapour pressure deficit patterns (Figure 3.6) to that of the rural old-growth forests during both seasons. However the vapour pressure deficit levels outside the forest, during both seasons, were higher than the levels outside the forests at the other two urban locations.

The analyses showed that there were only statistically significant differences between the mean diurnal vapour pressure deficit outside and inside the forest at Mount Maungatautari Ecological Island, Waingaro Forest and Chelmsford Park during both seasons (all p-values <0.01). Although there were diurnal vapour pressure deficit patterns during spring at the two urban sites, Berkley Bush and Howell Street, the differences were not statistically significant.

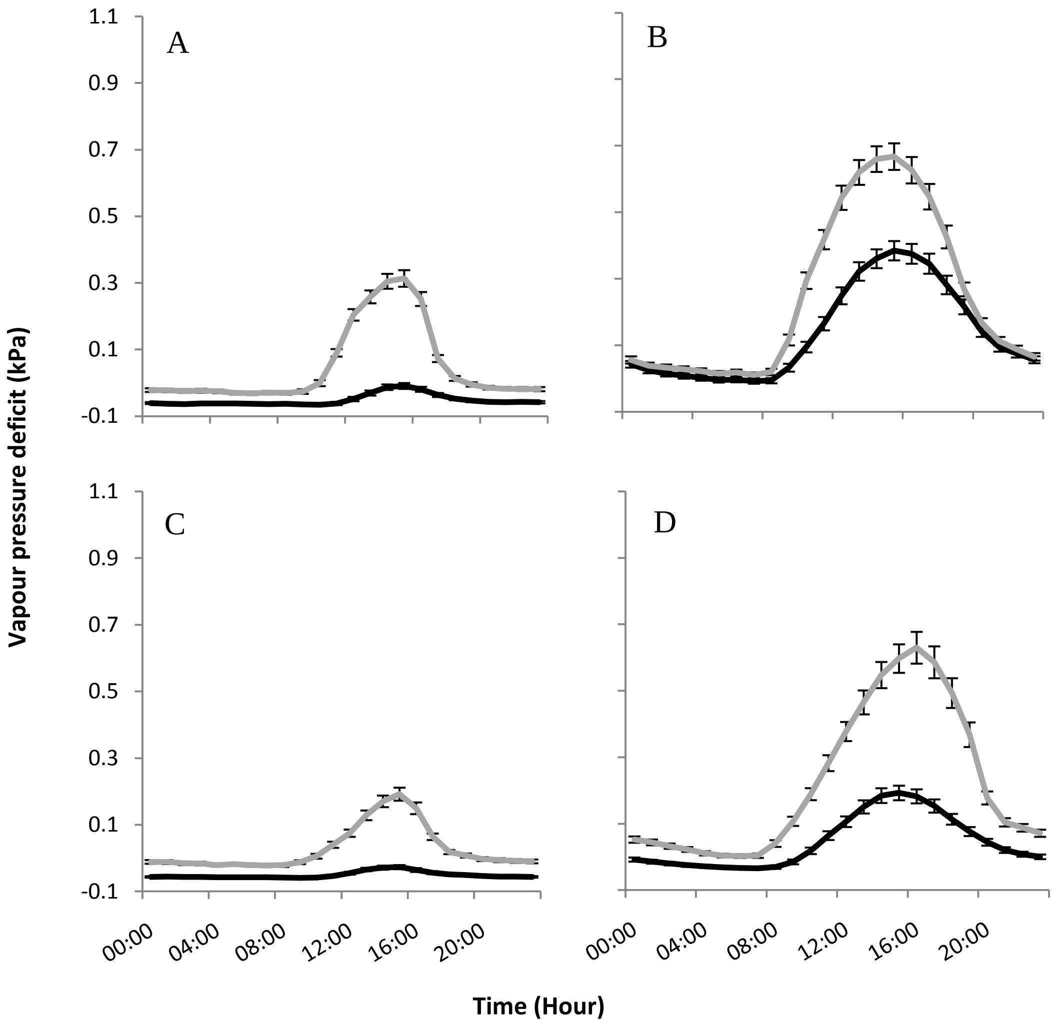

Figure 3.5: Mean diurnal vapour pressure deficit inside (black line) and outside (grey line) the rural old-growth forests of Waingaro Forest (A-winter B-spring) and Mount Maungatautari Ecological Island (C-winter D-spring) with standard error bars.

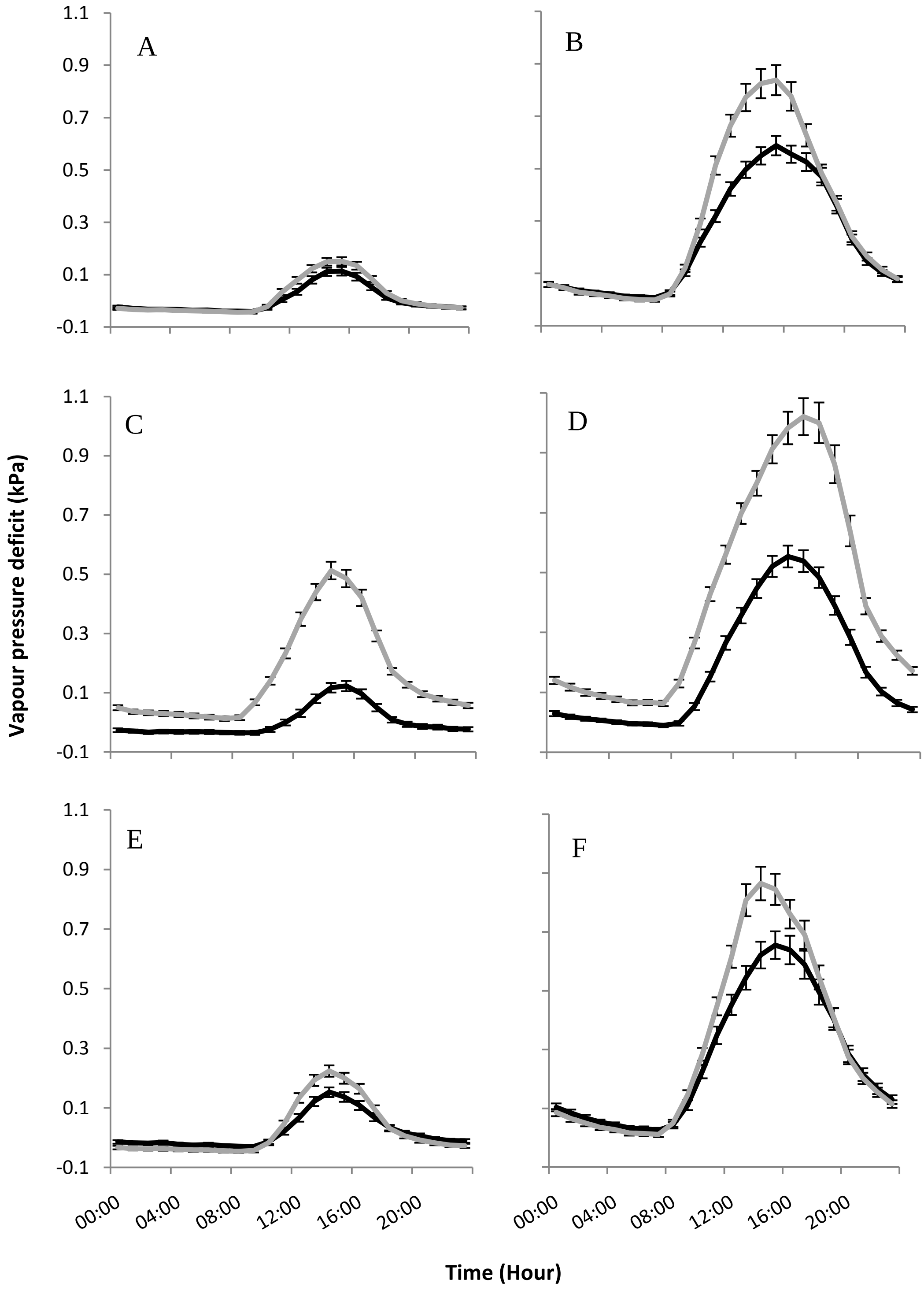

Figure 3.6: Mean diurnal vapour pressure deficit inside (black line) and outside (grey line) the urban forests of Berkley Bush (A-winter B-spring), Chelmsford Park (C-winter D-spring) and Howell Avenue (E-winter F-spring) with standard error bars.

3.4 Discussion

3.4.1 Soil type

The array of soil types found within the research locations is typical of the entire Hamilton City and Waikato Region (New Zealand Soil Bureau 1954a; Singleton 1991). Skeletal soils are generally common on steep slopes (Cutler 1983), including the three rural forest locations in the present study. Due to the steep slope they are relatively unstable and can contain varying soil properties (New Zealand Soil Bureau 1954a).

The majority of low-lying soils within Hamilton City have been produced from alluvium containing materials of volcanic origin with associations to the allophane mineral, creating allophanic soils (McLaren & Cameron 1996; Hewitt 1998; McCraw 2002). Gley soils, also predominant in Hamilton urban forests, are formed when soil saturation occurs over prolonged periods (McLaren & Cameron 1996; Hewitt 1998), which is common in Hamilton City with numerous springs and seepages (Wall & Clarkson 2006). Recent soils were found abundantly within the study locations. They are characterised by their weak soil formation due to the soils relatively young age and association with alluvial floodplains (McLaren & Cameron 1996; Hewitt 1998).

Pumice, anthropic and brown soil types were all represented within the study sites. Pumice soils are formed from volcanic deposits, whilst anthropic soils are those that have been modified through human activities. Brown soils are the most common soil type throughout New Zealand and are derived primarily from selected sedimentary and igneous rocks (McLaren & Cameron 1996). The forest sites in the present study covered the range of soils typical for the area.

3.4.2 Soil nutrients

Olsen P, total nitrogen and total carbon all exhibited similar trends of increasing concentrations from the youngest to the oldest forests within the urban categories in this study. This is expected as nutrients are continually being added to forest ecosystems through weathering, atmospheric input and the decomposition of organic matter. As forests become more mature the nutrient levels increase and the relative losses of nutrients, through leaching, decreases (Tilman 1985; Grier et al. 1989).

The late succession urban forest categories contained higher concentrations of Olsen P, total nitrogen and total carbon than the rural old-growth forests. This could be due to the management activities undertaken in Hamilton urban forests, where plant addition/removal and pest control are common. These activities can alter the nutrient balance through increasing and decreasing nutrient levels (Grier et al. 1989). Forest edges allow adjacent ecosystems to interact together (Guirado et al. 2006) and can add nutrients to urban forests through the combustion of fossil fuel, which is greatest in urban environments (McDonnell & Pickett 1990; Murcia 1995; Carreiro & Tripler 2005) such as Hamilton. Forest edges have also been documented to cause an increase in nutrient deposition through trapping air-borne nutrients from adjacent urban land (Weathers et al. 2001).

There have been many studies on nutrient and pH levels of New Zealand's native forests including in the central South Island (Chen et al. 2003), Nelson area (Hart et al. 2003), nationwide (Sparling & Schipper 2004) and in the Waikato Region (Stevenson 2004). Studies were conducted in an attempt to better understand nutrient dynamics, how they compare to other areas and what impacts they have. The rural forest category contained comparable concentrations of Olsen P, total nitrogen and total carbon with the other New Zealand studies, using the rating system from Blakemore et al. (1987). Olsen P concentrations in indigenous New Zealand forests have been recorded as low (Sparling & Schipper 2004) and very low (Chen et al. 2003; Stevenson 2004) compared to the very low levels found in the present study. Total carbon levels have been recorded as low (Hart et al. 2003) and medium (Chen et al. 2003) compared to the medium levels found in this study. Total nitrogen was considered high in the present research while in previous studies it has been recorded as medium (Chen et al. 2003) and very low (Hart et al. 2003).

3.4.3 Soil pH

Soil pH increased in acidity from the youngest to the oldest urban forest categories. This could be due to an increase and accumulation of organic plant material associated with forests becoming more mature (Brais et al. 1995). Soils also become more acidic with time (McLaren & Cameron 1996). The lower pH values present in the rural old-growth forests compared to the two oldest urban forest categories could also be influenced by adjoining land use. Urban environments often deposit nutrients and pollutants which can increase and decrease the pH of urban forest soils (Vallet et al. 2010).

The soil pH of the rural old-growth forests from the present study are comparable with other published soil pH levels from indigenous forests within New Zealand, using the rating system by Blakemore et al. (1987). Soil pH has been recorded as very low (Hart et al. 2003), low (Stevenson 2004) and medium (Sparling & Schipper 2004) compared to low levels found in the present research.

3.4.4 Soil moisture

There was no relationship between the mean soil moisture values across the five forest categories during the time period they were measured. The sites in each category contained variable soil moisture levels that displayed no correlation with site age or rural and urban location. Variable soil moisture levels between sites could relate to a variety of processes including anthropogenic activities or natural disturbances altering the soil moisture pattern, site aspect, forest structure and complexity (Ramos & Santos 2006; Chen et al. 1999) and different soil types and characteristics (Kapos 1989). Another factor, significant to soil moisture levels, is the numerous springs and seepages found sporadically throughout many of the study locations that are associated with the Hamilton City gully network, altering the water tables (Wall & Clarkson 2006; Collier et al. 2009).

There were no differences in the monthly soil moisture fluctuations between any of the forest categories. It was anticipated that soil moisture within urban forests would fluctuate more than that of the rural forests. Fragmented forests contain altered environmental conditions such as increased wind penetration and exposure which subsequently causes increased evapotranspiration and reduced soil moisture (Kapos 1989). Kapos (1989) measured the soil moisture between large and small forest reserves in the Amazon rainforest in Brazil. Soil moisture levels were consistently higher in larger forests compared to smaller forest fragments but differences were thought to be related to soil properties.

3.4.5 Canopy light transmittance

The total light transmittance results showed similar trends in winter and spring. Light transmittance increased from the rural and oldest urban forest categories through to the youngest urban group. The trend is likely to be primarily due to vegetation structure as more mature forests generally have more forest strata. It is the presence of emergent, canopy, sub-canopy and understory layers which restrict light transmittance into the understory (Denslow & Guzman 2000; Montgomery & Chazdon 2001; Smith & Smith 2001). During winter, the trend is also influenced by the presence of deciduous tree species. Deciduous trees lose their leaves over the winter period and allow more light to be transmitted through the canopy (Ramos & Santos 2006). Increased relative forest edge in urban fragmented forests also increases light transmittance due to reduced vegetative cover (Matlack 1993; Vallet et al. 2010).

Results suggested that forests between the ages of 25 to 100 years can have similar canopy development and light transmittance values to that of old-growth forests. Nicotra et al. (1999) further supports this idea, with their research suggesting that forests as young as 15 to 20 years can potentially contain similar understory light transmittance levels as old-growth forests. Other investigations have shown that rural and urban forests can contain similar light transmittance levels in the understory (Vallet et al. 2010), which correspond with the current results from the rural and oldest urban groups.

Reduced mean light transmittance values in spring in relation to winter results are likely related to the deciduousness of the canopies, particularly for the younger urban locations, as already discussed. Other possible causes for such differences include spring growth and the subjectivity, in relation to the methods used, in obtaining the images and how they are analysed (Robison & McCarthy 1999; Hymus et al. 2002). Marques et al. (2010) measured seasonal canopy openness on the edge and in the interior of semi-deciduous forest fragments in Brazil. No seasonal differences between canopy openness in the forests were found. In contrast, Ramos and Santos (2006) found seasonal variance in canopy openness between forest interiors in Brazil. However, no differences in canopy openness between anthropogenic forest edges and forest interiors were found.

3.4.6 Temperature and vapour pressure deficit

The temperature and vapour pressure deficit results showed that the forest interiors from the two rural locations contained distinctive temperature and vapour pressure deficit environments compared to those outside the forests over both the winter and spring seasons. The pattern showed relatively lower temperatures and vapour pressure deficits in the forest interiors during the day compared to outside the forests. The same pattern was also evident in Chelmsford Park (25 to100 years) during both winter and spring but only present in Berkley Bush (100 plus years) and Howell Street (15 to 25 years) during the spring.

Although the analysis did not reveal statistically significant differences in temperature within and outside Waingaro Forest during winter and spring and at Mount Maungatautari Ecological Island during the spring, the trends were still obvious. It is probable the trends are due to larger and undisturbed forests naturally buffering against the penetration of wind and radiation, creating stable air masses within the forest interior and forming a distinctive microclimate, different to that of the area surrounding the forest (Kapos 1989; Chen et al. 1999).

The two urban forest sites, Berkley Bush and Howell Street, contained interior forest temperature and vapour pressure deficit levels similar to that of the outside conditions during the winter. A distinctive interior microclimate with lower temperatures and vapour pressure deficits during the day was present in the spring. Disturbed and fragmented forests contain more variable temperatures and humidity than that of old-growth and undisturbed forests. The reduced vegetation structure, both vertically and horizontally, enables increased penetration of wind and radiation into the forest, creating conditions in the forest which are more similar to those outside the forest (Kapos 1989; Chen et al. 1999) which may explain the winter observations at Berkley Bush and Howell Street. The interior forest temperatures and vapour pressure deficits were not as low in Berkley Bush and Howell Street as they were in the two rural forests which could be due to the surrounding infrastructure. Moisture and heat fluxes are altered in urban environments depending on urban thermo-physical and geometrical characteristics, which can cause higher temperatures in urban areas, termed the ‘urban heat island’ (Taha 1997).

Denyer et al. (2006) examined differences in the microclimate of Waikato forest fragments that were adjacent to pasture and exotic pine plantations. Temperature and vapour pressure deficit were on average 2.2 ºC and 0.48 kPa higher on the edge of the forest, adjacent to pasture, than in the interior of the forest fragments during the afternoon. This is comparable with the present study where temperature and vapour pressure deficit was up to and over 3 ºC and 0.3 kPa higher outside selected forests than in the interior during the afternoon of certain seasons.

Ruiz-Jean and Aide (2006) investigated the temperature differences, during July 2002, between an urban pre-forested, reforested and reference forest site that were all less than 1 ha. They found distinctive microclimates in the interiors of the old-growth and 40 year old restored forests compared to the open site which contained fluctuating temperatures. They also found that differences in temperature and relative humidity between the forests and the exposed site were most pronounced during the day, which has also been found in the present study.

Young and Mitchell (1994) researched differences in temperature and vapour pressure deficit between the interiors and edges of five forest fragments in New Zealand and found both lower vapour pressure deficits and temperatures in the interior of the forests. They concluded that regularly shaped forests in New Zealand, less than 9 ha in size, are dominated by edge related effects and forests less than 1 ha cannot sustain interior forest conditions. Their results suggest that the interiors of small fragmented forests can contain distinctive microclimates during certain periods but become absent during the winter, which is evident in the present study. Kapos' (1989) studies of temperature and vapour pressure deficit in the forest edges of large and small forest reserves in the Amazon rainforest in Brazil further support the current results. Their study revealed that the margins of a one and 100 ha forest were, on average, 1.4 ºC and 2.8 ºC warmer than the respective interiors. Vapour pressure deficits and temperatures were higher in the interiors of 1 ha forests compared to 100 ha forests.

The temperature and vapour pressure deficit patterns at Chelmsford Park are unlikely to result from buffer zones which were evident at the two rural locations. The interior forest temperatures and vapour pressure deficits fluctuated to similar levels as the other two urban sites, both inside and outside the forests. However the temperature and vapour pressure deficit outside the forest at Chelmsford Park contained greater ranges. It is suspected that impervious surfaces surrounding the positioning of the data-logger were cause for the unusual results. Impervious surfaces can absorb solar radiation and alter the environmental conditions of the surrounding environment (Vallet et al. 2010). With unreliable data from outside the forest it is difficult to tell if a distinctive interior microclimate was present during winter or spring.

Young and Mitchell (1994) document different vegetative compositions associated with increases in temperature and vapour pressure deficit on the forest edge compared to the interior. The forest edge contained reduced leading forest dominants, which were replaced by other species. However, density remained the same and species richness increased. The same scenario could be applied to this study, where species richness and diversity could be altered due to higher vapour pressure deficits and temperatures in the urban forests during the day at certain times of the year. An alternative explanation could be that reduced plant diversity and structure in urban forests is causing the absence of distinctive diurnal temperature and vapour pressure deficit patterns during selected times of the year (Murcia 1995).

3.4.7 Overall evaluation of environmental variables

The results showed an increase in soil nutrients from the youngest to the oldest urban forest categories as well as a reduction in the amount of light transmitted into the forest understory. These results follow the resource-ratio pattern described by Tilman (1985), as forests become more mature the limiting resource, and subsequently the dominant vegetation, changes. The soil surface initially begins with a high level of light and low nutrient levels, as vegetation colonises and the structure and composition becomes more complex, light becomes reduced and the breakdown of plant material increases nutrient levels.

The increased nutrient status in the older urban forest categories compared to the rural old-growth forest group could arise due to the higher amount of nutrient deposition in urban areas, which also affects pH. The soil acidity also increased from the youngest to the oldest urban forest categories. This is a common occurrence as soils mature (McLaren & Cameron 1996). Diurnal temperature and vapour pressure deficit patterns were apparent where buffering occurred in the rural old-growth forests during both seasons but only during spring at two of the urban sites. Temperature and vapour pressure deficit buffering in fragmented forests during certain times of the year has also been documented by Young and Mitchell (1994).

These environmental differences between different aged and urban and rural forests suggest that they could be the critical factor causing reduced native understory diversity. The soil nutrient and light transmittance trends indicate that early forests may not be suitable for late succession forest species; however they theoretically do provide conditions suited to early succession species, of which some species were absent in urban forests compared to rural old-growth forests (Table 2.3, chapter two). Changes in species composition in forests may be caused by higher temperatures and vapour pressure deficits (Young and Mitchell 1994), which could explain the reduced native understory diversity in the present study. Alternatively, reduced diversity and structure of urban forests cause higher temperatures and vapour pressure deficits (Murcia 1995).

4 Chapter four: Plant reintroduction experiment

4.1 Introduction

This chapter investigates the success of reintroducing three native understory plant species into different aged Hamilton urban forests. The selected species (*Melicytus micranthus*, *Hedycarya arborea* and *Coprosma arborea*) are found abundantly in rural old-growth forests surrounding the city but are absent from or in relatively lower abundance within urban forests. Quantification of each plant species growth is provided across the five different aged forest categories. The results are analysed to determine the potential of the three understory species becoming a more prominent component within Hamilton City forests and to help elucidate whether environmental conditions or fragmentation, isolation and urban pressures are the critical factor/s causing reduced species diversity in urban forests.

4.1.1 Enrichment plantings

It is impractical to think that the restoration of a forest or any ecosystem will result in the passive reintroduction of all native species and the improvement of native biodiversity. Actively reintroducing propagules that are extinct or lower in abundance into second-growth or disturbed forests is a required management strategy for successful restoration (Menges 2008). Reintroduction or enrichment plantings can be described as the active movement of plants into habitat within its natural distribution range where it has previously reduced in numbers or become extinct due to human activities or a natural event (Maunder 1992). There are many different enrichment techniques; one common example includes raising seedlings in a nursery before planting out into the desired environment (Romell et al. 2008).

Enrichment planting is an important conservation strategy and an effective method of restoring forest ecosystems using endemic and native species (Maunder 1992; Kenzo et al. 2008). The technique of improving diversity aids in the long-term functioning of the forest ecosystem and fortunately this method has been realised as a means of which to decelerate biodiversity decline. There are many other reasons for reintroducing plant species including social, economic, recreational and cultural benefits (Alvey 2006).

The overall objective of plant reintroductions is to create a viable, self-sustaining population that is resilient, persistent and contains sufficient genetic resources to maintain diversity. Because the activity is high risk and high cost, reintroductions should first be undertaken as a scientific experiment (Maunder 1992; Guerrant & Kaye 2007). All plant reintroduction experiments are distinct in terms of species used, purposes of the experiment and the circumstances it is carried out. However, there are often similarities including reintroduced propagule type, where the plants are sourced from, reintroduction site/s and the timing of the experiment (Guerrant & Kaye 2007).

It is difficult to gauge the success of plant reintroductions as it can take hundreds of years to determine if a viable population has been established (Maunder 1992; Guerrant & Kaye 2007). However, there are various short and long-term techniques for which to assess planting success. Short-term measures include measuring plant survival, growth and fecundity while long-term measures can include population growth, plant dispersal and life-cycle completion (Guerrant & Kaye 2007; Menges 2008). Failure of reintroduced plant species is comparatively easier to measure and is not necessarily a negative outcome. A failed experiment can prevent wasting time, money and resources as well as provide practical information and value on future reintroductions (Guerrant & Kaye 2007).

4.1.2 Objective

The aim of this research is to assess the success of reintroducing three native understory species that are absent or less abundant within Hamilton urban forests compared to rural old-growth forests surrounding the city. The growth and survival of these species will determine their ability to become a more prominent component of Hamilton urban forests and also help to elucidate on whether environmental conditions are the fundamental cause for reduced native understory plant diversity in Hamilton urban forests.

4.2 Methods

The methods used to assess and measure each of the three reintroduced native understory plant species are outlined below.

4.2.1 Species selection

Melicytus Micranthus, H. arborea and *C. arborea* were selected as suitable species to undertake a growth and survival reintroduction experiment. The selections were based on data from the initial survey (chapter two) and availability from nurseries. The survey conducted over the summer period showed that these three plant species were abundant in the rural forests but less so within the urban forest environments. *Melicytus micranthus* was found only once in the urban survey but abundantly in two rural forests. *Hedycarya arborea* was found at just two urban sites but in all of the rural forests. *Coprosma arborea* was found in three of the rural forests but was not found at all within the urban forests. Several other species would also have been appropriate (e.g. those species listed in Table 2.3, chapter two) however adequate numbers of plants were not available from nurseries to use these species.

4.2.2 Reintroduced plant species

Melicytus micranthus

Melicytus micranthus (Figure 4.1) is a small shrub belonging to the Violaceae family; common names include swamp mahoe and manakura. It can grow up to 2 m high and contains small, stiff, interlacing branches. The alternate leaves are 1 to 2.5 cm long and 1 to 2 cm wide with up to ten small rounded teeth. The flowers are approximately 2 mm in diameter and vary in colour from light yellow to dark purple. The berries are between 2 to 4 mm in diameter and of dark purple colour. Flowering occurs from spring through to autumn whilst berries can be present from late spring to late autumn (Allan 1982; Dawson & Lucas 2000).

Melicytus micranthus is found in lowland forests throughout both North and South Islands and on many offshore islands (Allan 1982; Dawson & Lucas 2000). The plant had several uses including as a perfume addition (flowers) to titoki oil and an ingredient (bark) in a concoction to cure and aid a variety of ailments (Brooker et al. 1987; Riley 1994).

Figure 4.1: Reintroduced *Melicytus micranthus* planting in Hillcrest Park.

<u>*Hedycarya arborea*</u>

Hedycarya arborea (Figure 4.2) is known commonly as pigeonwood or porokaiwhiri and is a small tree from the Monimiaceae family. It can grow up to 12 m tall and the trunk to a diameter of 50 cm, encased in smooth bark. The opposite leaves are distantly toothed and measure between 5 to 12 cm long and have a width of 2.5 to 5 cm. The stem is dark, while new growth and the leaf midribs contain a red tinge. The flowers are small, usually between 6 to 10 mm, and green while the berries are bright orange. Flowering occurs during spring and the berries are produced from the end of spring and through summer (Allan 1982; Dawson & Lucas 2000).

Hedycarya arborea is predominantly found in lowland forests throughout New Zealand. The species is found on both North and South Islands but is absent from south of Banks Peninsula on the eastern coast of the South Island. It is found on many offshore islands (Allan 1982; Dawson & Lucas 2000). *Hedycarya arborea* has flowers which are aromatic and have been used in vapour baths (Brooker et al. 1987).

Figure 4.2: Reintroduced *Hedycarya arborea* planting in Chelmsford Park.

<u>*Coprosma arborea*</u>

Coprosma arborea (Figure 4.3) is a small tree that can grow up to 10 m high with a trunk up to 40 cm in diameter. It is a member of the Rubiaceae family and is also known as mamangi and tree coprosma. *Coprosma arborea* has distinct juvenile and adult leaves, both of which are opposite and winged. The juvenile leaves are 1.5 to 2.5 cm long and 1 to 2 cm wide, whilst the adult leaves are 5 to 6 cm long and 3.5 to 4 cm wide. The adult leaves are thin and glossy and are often wine coloured underneath. Flowering occurs during spring, followed by fruiting in the summer period (Allan 1982; Dawson & Lucas 2000).

Coprosma arborea is found in coastal to montane forests, although is most dominant in coastal and lowland forests. The species is distributed in the North Island from near North Cape down as far south as Waitomo in the west and Gisborne in the east as well as on several offshore islands (Allan 1982; Dawson & Lucas 2000). The genus *Coprosma* belongs to the same family as the coffee plant and it is possible to make a 'Coprosma coffee' using the seeds (Dawson & Lucas 2000).

Figure 4.3: Reintroduced *Coprosma arborea* planting in Hillcrest Park.

4.2.3 Site selection

The species suitable for enrichment plantings were trialled in the subsample of locations (15) listed in chapter one (Table 1.1). Selection of the forest sites was based on topographic similarities between the sites and avoiding forests where the plant species and experiment might be subjected to intentional disturbance from human interference. To ensure consistency within the experiment, the sites had to exhibit similar topographic units for the plantings. It was determined that the most similar topographic unit between all the sites was the foot slope within a slope range of 10 to 25º. The sites selected all contained appropriate foot slopes within the desired slope angles except Hillcrest Park. Hillcrest Park only had a flat terrace available but was the most appropriate option.

4.2.4 Planting design

A total of 135 plants, 45 of each species, were purchased from Oratia Native Plant Nursery and Taupo Native Plant Nursery. *Melicytus micranthus* and *C. arborea* were obtained from Oratia Native Plant Nursery and sourced from the Waitakere Ranges, west of Auckland. *Hedycarya arborea* were obtained from the Taupo Native Plant Nursery and were sourced from within the Taupo Region.

Three plants of each species were planted into the 15 different forest sites. Foot slopes with the appropriate slope angle were located in the forests and a random number table was used to determine the distance and direction of the planting position, whilst ignoring numbers which located the planting site outside the designated foot slope area.

All the planting was carried out between the 9th and 13th of May 2010, the period preceding these dates was a drought and inappropriate for planting. An appropriate sized hole was dug and plants were removed from the plastic bags and placed into the hole. Soil was then lightly compacted around the plant and between 2 and 3 L of water was poured on the soil surrounding the plant.

Two measures of plant height were recorded; from the ground to the upper most point of the plant in its natural position and from the ground to the apex of the *H. arborea* and *C. arborea* and to the tip of the longest branch of the *M. micranthus*. The number of leaves on each *H. arborea* was counted. The cover of *M. micranthus* and *C. arborea* species was measured. The widest crown diameter was measured first and a second measurement of the crown diameter was taken at a right angle to the first measurement. The plants were measured immediately after being planted, then again after four months and finally after approximately seven months.

Other environmental aspects were recorded such as the slope angle, aspect, canopy height and type and geographic coordinates.

4.2.5 Analyses

The statistical analyses on all the reintroduced planting results were undertaken using the software package ‘Statistica’ version 9.0. Descriptive statistics provided the mean, maximum, minimum, range and standard deviation of all the measured growth variables. One-way Analysis of Variance (ANOVA) was used to determine if there were any significant differences in survival and growth variables of the enrichment plantings between the forest categories. Statistically significant differences were proved if a p-value was obtained that was equal to or less than 0.05 and the associated assumptions, if required, were met.

4.3 Results

4.3.1 Plant survival

The survival rate results after four months proved to be of little value in the current analyses as the timeframe was not long enough to show distinct results and were not presented. The survival rate, after seven months, of the three reintroduced native plant species was high, both across the five forest categories and in total (Table 4.1). The lowest categorical survival rate was 77.8% obtained by *H. arborea* in the rural forest category, while 100% was achieved by all three species in a variety of categories. *Melicytus micranthus* had the highest overall survival rate with 97.8%, followed by *C. arborea* with 93.3% and *H. arborea* with 91.1%. No statistically significant differences were apparent in the survival rates and between forest categories.

Table 4.1: The relative abundance (%) of plants from the three species that survived during the experiment across the five forest categories and in total.

	M. micranthus	*H. arborea*	*C. arborea*
Rural	100	77.8	100
Urban 100+ yrs	100	88.9	100
Urban 25-100 yrs	88.9	88.9	77.8
Urban 15-25 yrs	100	100	100
Urban 5-15 yrs	100	100	88.9
Total	97.8	91.1	93.3

4.3.2 Plant growth

The growth of the reintroduced plantings after four months proved to be of little value in the current analyses as the timeframe was not long enough to show distinct results and instead just the seven month growth data was presented and analysed in this research.

Melicytus micranthus increased in height (Table 4.2) and volume (Table 4.3), across the five forest categories. However there were no consistent trends evident within the measured height from the ground to the longest point or the volume across the forest categories. The height growth was greatest in the oldest urban forest sites, however the minimum, maximum and range showed it was variable across all the forest groups. The varied mean volume values further emphasised this result with the greatest increase in mean volume in the youngest urban category, but again with variable minimum, maximum and range components. There were no statistically significant differences apparent in height or volume between the forest categories.

All *H. arborea* increased in height (Table 4.4) and in the number of leaves grown (Table 4.5), across the five forest categories. However there were no consistent trends evident within the measurements of growth from the ground to the apical meristem or the number of leaves grown, across the forest categories. The minimum, maximum and range were variable across all the forest categories. The urban category aged between 25 to 100 years showed both the greatest mean height growth and mean leaf number increase. There were no statistically significant differences apparent in height or leaves grown between the forest categories.

Coprosma arborea increased in size, both height (Table 4.6) and volume (Table 4.7), across the five forest categories. However, there were no consistent trends apparent within the growth of the height from the ground to the apical meristem or volume across the five forest categories. The rural forest category had the greatest increase in mean height growth, whilst the youngest urban forest group displayed the highest mean volume increase. The minimum, maximum and range were variable across all the five forest categories. There were no statistically significant differences apparent in height or volume between the forest categories.

Melicytus micranthus

Table 4.2: Descriptive statistics of the growth, from the ground to the longest point (cm), of *Melicytus micranthus* across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	5.2	4.0	6.8	2.8	1.5
Urban 100+ yrs	6.7	1.8	11.0	9.2	4.6
Urban 25-100 yrs	3.0	2.8	3.1	0.3	0.2
Urban 15-25 yrs	2.7	1.5	3.9	2.4	1.2
Urban 5-15 yrs	3.3	2.8	4.1	1.3	0.7

Table 4.3: Descriptive statistics of the growth, in volume (cm^3), of *Melicytus micranthus* across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	30091.4	18903.2	44582.3	25679.2	13154.3
Urban 100+ yrs	30154.2	28554.0	32164.7	3610.7	1840.0
Urban 25-100 yrs	31915.1	11721.8	51108.2	39386.4	19712.3
Urban 15-25 yrs	20936.2	16794.8	24688.7	7893.8	3961.3
Urban 5-15 yrs	39205.2	18990.3	62479.5	43489.2	21905.4

Hedycarya arborea

Table 4.4: Descriptive statistics of the growth, from the ground to the apical meristem (cm), of *Hedycarya arborea* across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	9.8	6.0	13.3	7.3	3.7
Urban 100+ yrs	9.0	4.9	15.9	11.0	6.0
Urban 25-100 yrs	12.1	8.5	14.1	5.6	3.1
Urban 15-25 yrs	9.4	8.2	10.3	2.1	1.1
Urban 5-15 yrs	9.7	5.2	12.7	7.5	4.0

Table 4.5: Descriptive statistics of the number of leaves grown, by *Hedycarya arborea*, across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	10.8	7.5	16.3	8.8	4.8
Urban 100+ yrs	15.0	6.0	28.0	22.0	11.5
Urban 25-100 yrs	17.1	10.5	22.7	12.2	6.1
Urban 15-25 yrs	13.4	10.7	18.0	7.3	4.0
Urban 5-15 yrs	14.1	12.0	16.0	4.0	2.0

Coprosma arborea

Table 4.6: Descriptive statistics of the growth, from the ground to the apical meristem (cm), of *Coprosma arborea* across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	5.5	2.8	7.5	4.7	2.4
Urban 100+ yrs	3.9	3.1	4.5	1.4	0.7
Urban 25-100 yrs	3.3	1.7	5.5	3.8	2.0
Urban 15-25 yrs	3.1	2.3	4.6	2.3	1.3
Urban 5-15 yrs	4.6	3.6	6.4	2.8	1.5

Table 4.7: Descriptive statistics of the growth, in volume (cm^3), of *Coprosma arborea* across the five forest categories.

	Mean	Minimum	Maximum	Range	Std. Dev.
Rural	17695.8	13987.5	19643.5	5656.0	3212.9
Urban 100+ yrs	14690.7	8491.0	24475.7	15984.7	8574.3
Urban 25-100 yrs	14382.2	6561.0	19085.3	12524.3	6819.6
Urban 15-25 yrs	14724.6	7848.5	25716.3	17867.8	9618.7
Urban 5-15 yrs	24991.4	20278.0	30279.0	10001.0	5025.2

4.4 Discussion

4.4.1 Plant survival

Measuring the survival rate of reintroduced plants is an important, and commonly used, method in determining the early success of enrichment plantings as it is straight forward and informative (Guerrant & Kaye 2007; Menges 2008). The similarities between survival rates of the three species across the five forest categories suggested there were no significant differences in survival rates between the forest groups. A high survival rate in the early stages of reintroduced plantings is a positive and successful start; however the overall success will further be determined by long-term measurements of life-cycle completion and population dynamics (Menges 2008) but this is beyond the scope of this thesis. The high survival rate in the early stages in the present study for all three reintroduced species is a positive indication that all three species have the potential to continue to exist within Hamilton urban forests.

In a similar overseas study, Vora et al. (2008) investigated the potential of restoring diversity in disturbed Minnesota forests by measuring the growth and survival of 19 native tree and shrub species. Survival rates varied from 20 to 100% across the different species with a mean of 68%, in sites that had received no preparation after five years. The survival rates from the present study are comparatively higher, although the time period is significantly less. Wendelberger et al. (2008) monitored the growth and survival of a rare native shrub population, *Amphora herbacea* var. c*renulata*, which had been planted into a restored urban forest. Their one to seven year old nursery plants had a survival rate of a little more than 90% after seven months, which was similar to the results obtained from the present study. However, after 33 months the survival rate was less than 80%.

The survival rates from studies conducted in New Zealand show variable survival rates for different species over one, two and four years. Sullivan et al. (2009) experimented with restoring vegetation in an urban environment to explore challenges affecting urban forest restoration. They document *Myrsine australis* (mapau) and mahoe with survival rates between 0 and 80% over four years. Two studies, experimenting with early succession plants and different planting techniques, document survival rates of between 49 to 99% after one year (Bergin 2008) and similar plant species with a survival rate of between 10 to 80% after two years (Langer et al. 2008). Survival results in the present study were generally higher compared to those documented in the literature. This is positive, but again the time period of monitoring in the present study was significantly less. It must also be noted that the trials from these three New Zealand studies were conducted in open environments with the objective of establishing new forest cover, whereas the present study aimed to enhance plant diversity under a canopy.

The present study had relatively higher survival rates than Standish (2002), who measured the survival and growth rates of native plant seedlings planted into a mat of wandering Jew and under a developed canopy. Survival rates ranged from 40 to 100%, although after 2.5 years. It must also be noted that the species used in the four New Zealand studies reported are early succession species (Sullivan et al. 2009); whilst in the present study the three species used are considered mid to late succession (Landcare Research n.d.).

It is difficult to pin-point the exact cause of failed reintroduced plantings in the present study. In some instances there was evidence of vandalism, with the presence of litter and other foreign materials in the vicinity, and trampling, with some plants found lying on the ground with a snapped stem. It is not uncommon for enrichment plantings to be affected by vandalism or trampling (Bhuju & Ohsawa 1998; Pauleit et al. 2002). In some observations from the present study, plants gradually wilted and died which is likely due to transplant shock. Transplant shock is when the plants are unable to become established in the new environment (Close et al. 2005). Continued monitoring of the survival of reintroduced plantings is required as plant condition can often change (Vora et al. 2008).

4.4.2 Plant growth

Plant growth is another clear cut and useful method for determining the early success of enrichment plantings and one that is widely used (Guerrant & Kaye 2007; Menges 2008). The similarities in the measured growth components for the three species across the five forest categories suggested there were no significant differences in the reintroduced planting growth between the forest groups. Growth of reintroduced plantings in the early stages indicates initial success; however further growth monitoring and long-term measurements of life-cycle completion and population dynamics will determine if the reintroduced plantings are successful (Menges 2008). The increase in growth and volume of *M. micranthus* and *C. arborea* as well as the positive growth and increase in the number of leaves of *H. arborea* across the five forest categories indicated that the early stage of the reintroduced plantings had been successful and that these species have the potential to survive and grow within Hamilton urban forests.

It is difficult to compare plant growth between species as different plant species have diverse growth forms and partition and allocate resources differently (Cornelissen et al. 1996; McKenna & Shipley 1999). Wendelberger et al. (2008) recorded an increase in mean plant growth and volume for the reintroduced plantings of *A. herbacea* of approximately 3 cm and 15000 cm^3 respectively after seven months. However, *A. herbacea* drops part or all of its leaves over a season. These results, although from a markedly different species, are similar to those obtained from the present study.

The growth rates recorded from other New Zealand studies are comparable with the present study, when they are converted to annual growth. Pratt (1999) obtained growth rates of 5.2 to 9.6 cm yr^{-1} for *Coprosma* species, 2.0 to 9.5 cm yr^{-1} for *Pseudopanax arboreus* (five-finger) and 6.6 to 11.0 cm yr^{-1} for mahoe. The present study revealed mean growth rates, across the five forest categories, of 7.2, 17.1 and 7.0 cm yr^{-1} for *M. micranthus*, *H. arborea* and *C. arborea* respectively. Growth rates from the present study were similar although the annual growth rates have been extrapolated from seven months of data.

Studies documenting the growth of similar earlier succession species are quite variable ranging from 17 to 36 cm yr^{-1} (Bergin 2008) and 0 to 14 cm yr^{-1} (Langer et al. 2008), which are higher in the first case and similar in the second case to the present study. Annual volume increments of 4 750 000 cm^3 yr^{-1} and 2 925 000 cm^3 yr^{-1} for mapau and mahoe respectively (Sullivan et al. 2009) are much higher than the volume increments of 52 218 cm^3 yr^{-1} and 29 652 cm^3 yr^{-1} obtained for *M. micranthus* and *C. arborea* in the present study.

As noted earlier, the species measured in the other New Zealand studies referenced are early succession species (Sullivan et al. 2009) while the three species used in the present study are mid to late succession species (Landcare Research n.d.). Early succession species are faster growing than late succession species (Crawley 1997b) which explains why the annual growth and volume increments are generally higher in the New Zealand studies compared to the present study.

The slight differences in the species growth across the five forest categories could be attributed to a variety of factors and their interactions. Such factors include; inter and intra-specific interference from other plants, differences in the age of the reintroduced plantings, the presence/absence and differential effects of herbivores, parasites and pathogens, genetic differences including maternal effects and environmental heterogeneity between sites (Weiner 1988). The growth and survival of reintroduced plantings is dependent on these factors which are stochastic between sites (Menges 2008).

4.4.3 Overall evaluation of reintroduced plantings

All three species had positive growth and high survival rates over the measured seven month period. The results suggested that the early stages of the reintroduced plantings had been successful and, although further long-term analyses need to be undertaken in order to determine the overall success, the three species have the potential to grow and survive within Hamilton urban forests. Comparisons with other New Zealand studies have shown relatively higher survival rates for the present study although the results were collated from only seven months of data. The slightly lower growth results in the present study compared to New Zealand studies are likely due to specific differences in environmental factors between experiments and the different succession types of the selected species.

Enrichment plantings are high risk and high cost and should initially be undertaken as a scientific experiment (Maunder 1992; Guerrant & Kaye 2007). The present reintroduced planting results suggested no obvious reasons why the other species absent and lower in abundance within Hamilton urban forests (Table 2.3, chapter two) could not be successfully reintroduced in urban forests, albeit within an appropriate habitat. The results suggested that the three species can survive within Hamilton urban forests as young as 5 to 10 years even though they are considered mid to late succession species (Landcare Research n.d.).

Chapter three proved that there were environmental differences between the different aged and urban and rural forest categories and could potentially be the cause for reduced native understory diversity in urban forests evident from chapter two. This chapter however, showed that three species absent and less abundant in Hamilton urban forests, compared to rural old-growth forests surrounding the city, can grow and survive under the various environmental conditions present. This would suggest that the variance in the environmental conditions are not the critical factor preventing the colonisation of absent species, but urbanisation and the associated effects, fragmentation and isolation and urban pressures, are the underlying cause.

Fragmented forests, such as those in Hamilton City, can cause area-dependent extinctions (Honnay et al. 1999b), reduce environmental corridors isolating forest fragments (Godefroid & Koedam 2003) and decrease the number and activity of animals, plants rely on for pollination and dispersal, further isolating species to their current distributions (Spyreas & Matthews 2006). Urban pressures, such as those evident in Hamilton urban forests, have the potential to alter forest diversity (Lehvavirta & Rita 2002). The urban matrix surrounding Hamilton City forests continually applies pressure through the immigration of invasive exotics. Invasive exotics alter the composition and structure of forests by out-competing the native species and preventing regeneration (Honnay et al. 1999a; Borgmann & Rodewald 2005).

5 Chapter five: Final conclusions

5.1 Final conclusion

The present study has demonstrated that the understory of Hamilton's isolated urban forests are characterised by reduced native understory species diversity in comparison to rural old-growth reference forests surrounding the city (Figures 2.4, 2.5 and 2.6, chapter two). Urban forests contained a comparatively higher abundance of invasive exotic species which have out-competed many of the native species. Most of the Hamilton urban forests featured a developed native canopy with a sparse understory and a groundcover comprised of a mat of wandering Jew (Figure 2.2, chapter two).

5.1.1 Understory vegetation survey

The understory vegetation survey showed that Hamilton urban forests were relatively depauperate in understory species compared to old-growth forests surrounding Hamilton City. Numerous species were absent from the urban forests, while many more were of lower abundance, including both early and late succession species. Native species richness and density, per 50 m^2, decreased from the rural old-growth forest group down through the four urban categories to the youngest aged forest group, 5 to 15 years. It is expected that early succession forests would contain lower species diversity than old-growth forests as forests generally increase in diversity naturally overtime (Honnay et al. 1999b). This provides an explanation for the lower species richness in the youngest urban categories relative to more natural old-growth forests.

However, there were still many early and late succession species absent from the urban sample that should have been present when compared to the rural old-growth forests. The older urban forests contained reduced species richness compared to surrounding rural forests. Based on the environmental profiling and reintroduction planting experiment results, reduced species diversity was due to urbanisation and its associated effects of fragmentation and isolation and urban pressures.

The colonisation of New Zealand by Polynesian and European settlers resulted in the clearance of the majority of lowland forests (Wardle 1991; Walker & Wass 2006). Within the Hamilton Ecological District just 1.6% of the original vegetation remains as small fragmented and isolated forests scattered throughout rural and urban areas (Leathwick et al. 1995; Clarkson & McQueen 2004; Innes 2010). The small size of forest patches eliminates species which cannot survive in such a small area (Godefroid & Koedam 2003). Fragmentation of urban forests, such as those present in Hamilton, also reduces the number and activity of animals within and between forest fragments which plants rely on for pollination and dispersal, isolating plant populations and species and preventing migration between forests (Ramos & Santos 2006; Spyreas & Matthews 2006). Evidence from within Hamilton urban forests illustrated that the forests are exposed to vandalism, pollution and recreational pressures. Such pressures can affect forest diversity (Lehvavirta & Rita 2002). The high density of exotic plants in urban environments surrounding urban forests further exposes forests to invasive exotics which alter the composition and structure by out-competing native species (Honnay et al. 1999a; Borgmann & Rodewald 2005).

Exotic understory stem density increased from the rural old-growth forest category, excluding data from Waingaro Forest, to the youngest urban forest category. Invasive exotic species exploit the high availability of ecological niches in the younger, less developed forests (Myers & Bazely 2003). A high population, such as in Hamilton City, further increases the likelihood of elevated exotic stem densities in urban forests (Cronk & Fuller 1995) as humans are important vectors in the migration of exotic plants (Hill 1977). Invasive exotic species are also often the only plants that can successfully survive in highly disturbed urban landscapes (Borgmann & Rodewald 2005).

The ordination further illustrated the differences in native understory species diversity between each of the forest categories. Forest age correlated with a decrease in native understory species richness and stem density, further supporting the results from the understory vegetation survey.

5.1.2 Assessment of environmental variation

Soil analyses of the forest sites produced results typical forest succession documented by Tilman (1985). The Olsen P, total nitrogen and total carbon levels increased from the youngest to the oldest urban forest category while the pH levels decreased from the youngest to the oldest urban forest categories. However the oldest urban forest category contained unexpectedly higher nutrient levels, and was more acidic, than the rural old-growth forest group. This could arise from the deposition of relatively higher amounts of nutrients, which also effects pH, from an urban matrix into forests (Vallet et al. 2010) through the addition and removal of materials from management activities in and around the forest (Grier et al. 1989), additions from the combustion of fossil fuels (Carreiro & Tripler 2005) and forest edges trapping air-borne nutrients (Weathers et al. 2001). The light transmittance trend across the five forest categories was also characteristic of forest succession according to Tilman (1985). Light transmittance through the forest canopy decreased from the youngest to the oldest urban category. Mature forests normally contain more forest strata which prevent light from penetrating into the understory (Denslow & Guzman 2000; Montgomery & Chazdon 2001). The total light transmittance results suggested that urban forests, aged between 25 to 100 years, can develop canopy structure and have similar light transmittance levels to rural old-growth forests.

The light transmittance and soil nutrient levels in the younger forest categories suggested that they are not yet suitable for late succession plant species and are a potential cause for the lack of native understory forest species richness and density. However the understory vegetation survey found an absence and lower abundance of earlier forest succession species, compared with the rural old-growth forests, which would be expected to grow within the younger forests that exhibited early succession conditions. This suggested that environmental differences were not the underlying cause for reduced native understory diversity.

Investigation into forest temperatures and vapour pressure deficits showed that the urban forests were in equilibrium with their surroundings during the winter but contained distinctive microclimates during spring. Temperatures and vapour pressure deficits were higher during the day outside the forests compared to within the forests. The larger rural old-growth forests contained the same distinctive microclimate patterns, found in the urban forests in spring, during both winter and spring likely arising from buffer zones. Young and Mitchell (1994) showed variation in species composition associated with increases in temperature and vapour pressure deficit from a forest interior to the edge. This could suggest that in the present study the lack of buffering, leading to higher temperatures and vapour pressure deficits, within the urban forests was the cause for reduced native species diversity due to conditions not suiting certain species. Conversely, reduced plant diversity and structure in urban forests might cause the absence of buffering during the winter (Murcia 1995).

5.1.3 Plant reintroduction experiment

The three species used in the reintroduction experiment displayed high survival rates across all the forest categories after seven months. Survival rates ranged from 77.8% for *H. arborea* in the rural old-growth forest category to 100% for all three species for the majority of the other forest categories. The high survival rate in the early stages of reintroduced plantings is a positive indication for successful reintroduction; however long-term measures would be required to determine the overall success. Causes for the demise of reintroduced plantings were difficult to determine but evidence found at planting sites; litter and foreign materials, snapped stems and plants lying on the ground and witnessing plants wilting slowly, suggested that vandalism, trampling and transplant shock were the main causes for failure.

Further supporting the survival results was the positive growth increments of the reintroduced plantings for each of the three species. The three species all displayed similar growth measurements, within species, across the five forest categories. Several forest categories did reveal some differences in growth components. Such differences could be attributed to environmental heterogeneity between sites, inter and intra-specific interference, differential effects of herbivores, parasites and pathogens and genetic differences (Weiner 1988). The results displayed no significant differences in growth increments between any of the three selected species across the five forest categories. Overall, the positive growth rates were an early indication for successful reintroduced plantings however, further long-term monitoring would need to be undertaken to determine the ultimate success.

The results from environmental profiling had suggested that differences in environmental variables may be cause for the absence or lower abundance of some native understory species. However the positive survival rate and growth over the (measured) seven month period of the three reintroduced plant species showed that environmental conditions were not limiting and were therefore not the primary cause for the absence or lower abundance of selected plant species. The results indicated that urbanisation and its associated effects, fragmentation and isolation and urban pressures, were more crucial in reducing understory native species richness in Hamilton urban forests. Fragmentation of urban forests is responsible for area-dependent extinctions (Godefroid & Koedam 2003) and isolates species and forests through reduced forest connectivity and activity of animals which plants rely on for pollination and dispersal. Urban environments expose forests to vandalism, pollution and recreational pressures as well as high levels of invasive exotic species which can all alter the forest structure and diversity (Lehvavirta & Rita 2002).

5.2 Implications for urban forest restoration and management

The findings of this research are beneficial to public and private gully owners and restoration organisations within Hamilton wishing to restore indigenous forest ecosystems as they assist in enhancing native understory diversity. The underlying principles are also transferable to other regions nationally. The vegetation survey method used in the present research could also be applied in other urban centres to identify absent native understory vegetation in the urban forests. Following this, missing species could be experimentally reintroduced into the urban forests and the progress monitored. As in the present study, any variation (unless extreme) in environmental conditions between urban forests and rural old-growth forests is unlikely to be the critical factor for the absence of native understory species.

The research verified that there was reduced understory plant diversity within Hamilton urban forests compared to rural old-growth forests surrounding the city. There were numerous native understory species found in the old-growth control sites outside the city that were not present within the urban sample or were in comparatively lower densities. The absent urban forest species found outside in the rural old-growth forests, were in reasonably similar habitats to that which exist in Hamilton urban forests.

The environmental profiling investigation showed that there were differences between the different aged and urban and rural forest categories. However, the early positive survival and growth of the three reintroduced species in the plant reintroduction experiment proved that urbanisation and its associated effects, fragmentation and isolation and urban pressures, were the critical factors for reduced native understory diversity in Hamilton urban forests. Urbanisation, as already discussed, reduces native species diversity and prevents the natural migration of native species between forests which would improve diversity. Passive re-introduction of absent species can take centuries to occur (Spyreas & Matthews 2006) and the best option is to actively reintroduce the appropriate species.

Further monitoring of the reintroduced plantings should be undertaken to further evaluate if the reintroduced native plant species can successfully survive in Hamilton urban forests. If the long-term results prove to be positive then the three species can be reintroduced in larger numbers. Reintroducing other missing native understory species or those less abundant from the present study (Table 2.3, chapter two) should initially be experimentally reintroduced to prevent any waste of resources. From the present research there was no obvious reason why other absent and less abundant species should not be able to grow and survive within Hamilton urban forests.

The similar positive survival and growth of the reintroduced plantings across the five forest categories indicated they can be reintroduced into urban forests as young as 5 to 15 years. Mackay (2006) suggested that Hamilton urban forests benefitted from enrichment plantings after 10 to 20 years of establishment. The present study refines that suggestion by noting which species are missing or in lower abundance and suggests that these species should be incorporated into any enrichment planting plans for forests established over 10 years to enhance urban forest diversity. Wandering Jew is abundant in many Hamilton urban forests and Standish et al. (2001) documents some native species tolerances and height requirements to escape smothering from the invasive exotic species. If management plans do not include the removal or clearing of wandering Jew around enrichment plantings then tolerance to smothering of wandering Jew should be taken into account when selecting native understory plant species.

Management plans should be implemented to control and remove invasive understory exotic species. Active removal of exotic species is required in some of the urban forests, where as in other scenarios methods involving forest development should be used to manipulate the demise of pest plant species. The high density of understory exotic stems in the younger urban forests correlates with the more abundant vacant ecological niches

5.3 Recommendations for future research

This research illustrated how urban forests in Hamilton are relatively depauperate in native understory species composition and density. Successfully reintroducing this component of forest ecosystems would improve native understory diversity and richness within Hamilton urban forests and contribute to their restoration. To achieve full restoration it is important to reintroduce all elements (Halle & Fattorini 2004). Further detailed research into what other vegetative elements, including large canopy trees, ground covers and ferns, are depauperate within Hamilton urban forests would aid in the restoration of the urban forests.

The present research proved that differences in environmental conditions of urban forests, compared to rural old-growth forests, were significant but unlikely to be the critical factor for reduced native understory diversity. Instead urbanisation and its associated effects of fragmentation and isolation and urban pressures were considered to be the underlying causes. Further research into the interaction between fragmentation, isolation and urban pressures in urban forests is required to determine how and to what degree the pressures affect diversity.

The positive results from the reintroduced plantings suggested that the three species, *M. micranthus*, *H. arborea* and *C. arborea* can grow and survive within Hamilton urban forests. Continued monitoring of short and long-term growth and survival of the three species would contribute further to determining how successful the reintroduced plantings have been. Further research is necessary on the long-term population dynamics and restoration success of large scale enrichment plantings in Hamilton urban forests.

References

Alberti M 2005. The effects of urban patterns on ecosystem function. International Regional Science Review 28(2): 168-192.

Allan HH 1982. Flora of New Zealand: Volume 1. Wellington, P.D. Hasselberg, Government Printer.

Alvey AA 2006. Promoting and preserving biodiversity in the urban forest. Urban Forestry and Urban Greening 5(4): 195-201.

Bergin D 2008. Establishment and early performance of planting trials with indigenous woody shrub species in the Rotorua District. Rotorua, Central Plateau Sustainable Farming Fund project 03/065. Scion. 37 p.

Bernhardt ES, Palmer MA 2007. Restoring streams in an urbanizing world. Freshwater Biology 52(4): 738-751.

Bhuju DR, Ohsawa M 1998. Effects of nature trails on ground vegetation and understory colonization of a patchy remnant forest in an urban domain. Biological Conservation 85(1-2): 123-135.

Bhuju DR, Ohsawa M 2001. Patch implications in the maintenance of species richness in an isolated forest site. Biological Conservation 98(1): 117-125.

Blakemore LC, Searle PL, Daly BK 1987. Methods for chemical analysis of soils. Lower Hutt, NZ Soil Bureau Scientific Report 80. Department of Scientific and industrial research. 103 p.

Borgmann KL, Rodewald AD 2005. Forest restoration in urbanizing landscapes: interactions between land uses and exotic shrubs. Restoration Ecology 13(2): 334-340.

Brais S, Camire C, Bergeron Y, Pare D 1995. Changes in nutrient availability and forest floor characteristics in relation to stand age and forest composition in the southern part of the boreal forest of northwestern Quebec. Forest Ecology and Management 76(1-3): 181-189.

Brooker SG, Cambie RC, Cooper RC 1987. New Zealand medicinal plants. Auckland, Reed.

Burns B, Smale M 2002. Lowland forests. In: Clarkson B, Merrett M, Downs T ed. Botany of the Waikato. Hamilton, Waikato Botanical Society Inc. Pp. 73-81.

Carreiro MM, Tripler CE 2005. Forest remnants along urban-rural gradients: examining their potential for global change research. Ecosystems 8(5): 568-582.

Chapin FS 1983. Nitrogen and phosphorus nutrition and nutrient cycling by evergreen and deciduous understory shrubs in an Alaskan black spruce forest. Canadian Journal of Forest Research 13(5): 773-781.

Chen CR, Condron LM, Davis MR, Sherlock RR 2003. Seasonal changes in soil phosphorus and associated microbial properties under adjacent grassland and forest in New Zealand. Forest Ecology and Management 177(1-3): 539-557.

Chen J, Franklin JF, Spies TA 1995. Growing-season microclimate gradients from clearcut edges into old-growth douglas-fir forests. Ecological Applications 5(1): 74-86.

Chen J, Saunders SC, Crow TR, Naiman RJ, Brosofske KD, Mroz GD, Brookshire BL, Franklin JF 1999. Microclimate in forest ecosystem and landscape ecology. Bioscience 49(4): 288-297.

Clarkson BD, Clarkson BR, Downs TM 2007. Indigenous vegetation types of Hamilton Ecological District. Hamilton, CBER Contract Report No. 58. The University of Waikato. 18 p.

Clarkson BD, McQueen JC 2004. Ecological restoration in Hamilton City, North Island, New Zealand. Proceedings of the 16th International Conference, Society for Ecological Restoration, Victoria, British Columbia, August 2004. 6 p.

Close DC, Beadle CL, Brown PH 2005. The physiological basis of containerised tree seedling 'transplant shock': a review. Australian Forestry 68(2): 112-120.

Cole DN, Landres PB 1996. Threats to wilderness ecosystems: impacts and research needs. Ecological Applications 6(1): 168-184.

Collier KJ, Aldridge BMTA, Hicks BJ, Kelly J, Macdonald A, Smith BJ, Tonkin J 2009. Ecological values of Hamilton urban streams (North Island, New Zealand): constraints and opportunities for restoration. New Zealand Journal of Ecology 33(2): 177-189.

Cornelissen JHC, Castro Diez P, Hunt R 1996. Seedling growth, allocation and leaf attributes in a wide range of woody plant species and types. Journal of Ecology 84(5): 755-765.

Crawley MJ 1997a. Biodiversity. In: Crawley MJ ed. Plant ecology. Second ed. Malden, Blackwell Publishing. Pp. 595-632.

Crawley MJ 1997b. The structure of plant communities. In: Crawley MJ ed. Plant ecology. Second ed. Malden, Blackwell Publishing. Pp. 475-531.

Croci S, Butet A, Georges A, Aguejdad R, Clergeau P 2008. Small urban woodlands as biodiversity conservation hot-spot: a multi-taxon approach. Landscape Ecology 23(10): 1171-1186.

Cronk QCB, Fuller JL 1995. Plant invaders. London, Chapman and Hall.

Cutler EJB 1983. Soil classification in New Zealand: a review of soil classification systems and proposals for a New Zealand soil classification. Lincoln, Occasional report No. 2. Lincoln College. 148 p.

D'Amato AW, Orwig DA, Foster DR 2009. Understory vegetation in old-growth and second-growth *Tsuga canadensis* forests in western Massachusetts. Forest Ecology and Management 257(3): 1043-1052.

Dawson J, Lucas R 2000. Nature guide to the New Zealand forest. Auckland, Random House New Zealand.

de Lange PJ 1996. Floristics and microclimate of Hammond Bush, a Hamilton basin forest remnant. Wellington Botanical Society Bulletin 47: 65-80.

Denslow JS, Guzman SG 2000. Variation in stand structure, light and seedling abundance across a tropical moist forest chronosequence, Panama. Journal of Vegetation Science 11(2): 201-212.

Denyer K, Burns B, Ogden J 2006. Buffering of native forest edge microclimate by adjoining tree plantations. Austral Ecology 31(4): 478-489.

Department of Conservation 2010. Waikato tracks and walks. http://www.doc.govt.nz/parks-and-recreation/tracks-and-walks/waikato/a-z-of-waikato-tracks-and-walks/ (accessed 21 April 2010).

Dodd M, Ritchie H 2007. Farming with native trees: A guide for farmers from Northland to Waikato. Rotorua, New Zealand Forest Research Institute Limited.

Downs TM, Clarkson BD, Beard CM 2000a. Key ecological sites of Hamilton City: Volume 2. Hamilton, CBER contract report 5. The University of Waikato. 39 p.

Downs TM, Clarkson BD, Beard CM 2000b. Key ecological sites of Hamilton City: Volume 3. Hamilton, CBER contract report 5. The University of Waikato. 32 p.

Environment Waikato 2010. Our climate. http://www.ew.govt.nz/environmental-information/About-the-Waikato-region/Our-natural-environment/Our-climate/ (accessed 8 October 2010).

Forest Research Institute 1979. Waikato. 1:250 000. Wellington, P.D. Hasselberg Government Printer.

Frazer GW, Fournier RA, Trofymow JA, Hall RJ 2001. A comparison of digital and film fisheye photography for analysis of forest canopy structure and gap light transmission. Agricultural and Forest Meteorology 109(4): 249-263.

Fuhrer E 2000. Forest functions, ecosystem stability and management. Forest Ecology and Management 132(1): 29-38.

Gilliam FS, Roberts MR 2003. Conceptual framework for studies of the herbaceous layer. In: Gilliam FS, Roberts MR ed. The herbaceous layer in forests of Eastern North America. New York, Oxford University Press.

Godefroid S, Koedam N 2003. How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? Global Ecology and Biogeography 12(4): 287-298.

Grier CC, Lee KM, Nadkarni NM, Klock GO, Edgerton PJ 1989. Productivity of forests of the United States and its relation to soil and site factors and management practices: A review. Portland, General technical report PNW-GTR-222. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p.

Grissino-Mayer HD 2003. A manual and tutorial for the proper use of an increment borer. Tree Ring Research 59(2): 63-79.

Gudex MC 1955. The native flora of Claudelands Bush. Transactions of the Royal Society of New Zealand 83(2): 310-319.

Guerrant EO, Kaye TN 2007. Reintroduction of rare and endangered plants: common factors, questions and approaches. Australian Journal of Botany 55(3): 362-370.

Guirado M, Pino J, Roda F 2006. Understorey plant species richness and composition in metropolitan forest archipelagos: effects of forest size, adjacent land use and distance to the edge. Global Ecology and Biogeography 15(1): 50-62.

Guntenspergen GR, Levenson JB 1997. Understory plant species composition in remnant stands along an urban-to-rural land-use gradient. Urban Ecosystems 1(3): 155-169.

Halkett J 1991. The native forests of New Zealand. Upper Hutt, GP Publications Limited.

Halle S, Fattorini M 2004. Advances in restoration ecology: Insights from aquatic and terrestrial ecosystems. In: Temperton VM ed. Assembly rules and restoration ecology: Bridging the gap between theory and practice. Washington DC, Island Press. Pp. 10-33.

Hamilton City Council 2002. Hamilton City green map. No scale. Hamilton, Hamilton City Council.

Hart PBS, Clinton PW, Allen RB, Nordmeyer AH, Evans G 2003. Biomass and macro-nutrients (above- and below-ground) in a New Zealand beech (*Nothofagus*) forest ecosystem: implications for carbon storage and sustainable forest management. Forest Ecology and Management 174(1-3): 281-294.

Hewitt AE 1998. New Zealand Soil Classification. Second ed. Lincoln, Manaaki Whenua Press.

Hill TA 1977. The biology of weeds. London, Edward Arnold Limited.

Hobbs RJ, Norton DA 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology 4(2): 93-110.

Honnay O, Endels P, Vereecken H, Hermy M 1999a. The role of patch area and habitat diversity in explaining native plant species richness in disturbed suburban forest patches in northern Belgium. Diversity and Distributions 5(4): 129-141.

Honnay O, Hermy M, Coppin P 1999b. Effects of area, age and diversity of forest patches in Belgium on plant species richness, and implications for conservation and reforestation. Biological Conservation 87(1): 73-84.

Horn HS 1974. The ecology of secondary succession. Annual Review of Ecology and Systematics 5: 25-37.

Hurst JM, Allen RB 2007a. A permanent plot method for monitoring indigenous forests – expanded manual: version 4. Lincoln, Landcare Research Contract Report LC0708/028. Landcare Research. 103 p.

Hurst JM, Allen RB 2007b. The recce method for describing New Zealand vegetation – expanded manual: version 4. Lincoln, Landcare Research Contract Report: LC0708/029. Landcare Research. 62 p.

Hymus GJ, Pontailler J, Li J, Stiling P, Hinkle CR, Drake BG 2002. Seasonal variability in the effect of elevated CO_2 on ecosystem leaf area index in a scrub-oak ecosystem. Global Change Biology 8(10): 931-940.

Innes JG 2010. Forest fragments research. Unpublished PowerPoint presentation, Landcare Research, Hamilton.

Jackson ST, Hobbs RJ 2009. Ecological restoration in the light of ecological history. Science 325(5940): 567-569.

Jay M 1997. A short history of the landscapes of central Waikato. In: Stokes E, Begg M ed. Te hononga ki te whenua belonging to the land: people and places in the Waikato Region. Hamilton, New Zealand Geographical Society. Pp. 60-68.

Kapos V 1989. Effects of isolation on the water status of forest patches in the Brazilian Amazon. Journal of Tropical Ecology 5(2): 173-185.

Kenzo T, Yoneda R, Matsumoto Y, Azani MA, Majid NM 2008. Leaf photosynthetic and growth responses on four tropical tree species to different light conditions in degraded tropical secondary forest, Peninsular, Malaysia. Japan Agricultural Research Quarterly 42(4): 299-306.

Landcare Research 2002. Soil map of Hamilton City. 1:20,000. City unknown, Landcare Research.

Landcare Research (n.d.). New Zealand Indigenous Plant Species Selector. http://www.landcareresearch.co.nz/services/greentoolbox/gtbweb/default.asp (accessed 7 January 2011).

Langer L, Ledgard N, Henley D 2008. Establishing native plants alongside the Sherry River, with particular emphasis on initial weed control. The Bavin's trial. Christchurch, Information Technology Sherrytrialrep 108.doc. Scion. 9 p.

Leathwick JR, Clarkson BD, Whaley PT 1995. Vegetation of the Waikato Region: current and historical perspectives. Hamilton, Landcare Research Contract Report: LC9596/022. Landcare Research. 66 p.

LECO Corporation 2006. TruSpec CN carbon/nitrogen determinator. Saint Joseph, LECO Corporation.

Lehvavirta S, Rita H 2002. Natural regeneration of trees in urban woodlands. Journal of Vegetation Science 13(1): 57-66.

Macdonald C 1973. Medicines of the Maori. Auckland, William Collins (New Zealand) Limited.

Macdonald J 2006. In the beginning - geological history and origins of New Zealand biota. In: Parsons S ed. Biology Aotearoa: unique flora, fauna and fungi. Auckland, Pearson Education New Zealand.

Machado JL, Reich PB 1999. Evaluation of several measures of canopy openness as predictors of photosynthetic photon flux density in deeply shaded conifer-dominated forest understory. Canadian Journal of Forest Research 29(9): 1438-1444.

Mackay DB 2006. Ecology of restored gully forest patches in Hamilton ecological district. Unpublished MSc thesis, University of Waikato, Hamilton, New Zealand.

Malmivaara M, Lofstrom I, Vanha-Majamaa I 2002. Anthropogenic effects on understorey vegetation in *Myrtillus* type urban forests in southern Finland. Silva Fennica 36(1): 367-381.

Marques TED, Beijo LA, Ramos FN 2010. Are biotic and abiotic factors and seedling mechanical damage in forest-edge fragments always different from the interior? Australian Journal of Botany 58(4): 241-247.

Matlack GR 1993. Microenvironment variation within and among forest edge sites in the Eastern United States. Biological Conservation 66(3): 185-194.

Maunder M 1992. Plant reintroduction: an overview. Biodiversity and Conservation 1(1): 51-61.

Maungatautari Ecological Island Trust 2002. Maungatautari Ecological Island Trust. http://www.maungatrust.org/. (accessed 21 April 2010).

McCormack, DD 1979. Soil map of Hamilton City, North Island, New Zealand. 1:20,000. Wellington, New Zealand soil bureau.

McCraw J 2002. Physical environment. In: Clarkson B, Merrett M, Downs T ed. Botany of the Waikato. Hamilton, Waikato Botanical Society Inc. Pp. 13-22.

McDonnell MJ, Pickett STA 1990. Ecosystem structure and function along urban-rural gradients: an unexploited opportunity for ecology. Ecology 71(4): 1232-1237.

McDonnell MJ, Pickett STA, Groffman P, Bohlen P, Pouyat RV, Zipperer WC, Parmelee RW, Carreiro MM, Medley K 1997. Ecosystem processes along an urban-to-rural gradient. Urban Ecosystems 1(1): 21-36.

McGlone MS, Duncan RP, Heenan PB 2001. Endemism, species selection and the origin and distribution of the vascular plant flora of New Zealand. Journal of Biogeography 28(2): 199-216.

McKenna MF, Shipley B 1999. Interacting determinants of interspecific relative growth: empirical patterns and a theoretical explanation. Ecoscience 6(1): 286-296.

McKinney ML 2002. Urbanization, biodiversity, and conservation. Bioscience 52(10): 883-890.

McLachlan SM, Bazely DR 2001. Recovery patterns of understory herbs and their use as indicators of deciduous forest regeneration. Conservation Biology 15(1): 98-110.

McLaren RG, Cameron KC 1996. Soil science: sustainable production and environmental protection. Second edition. Melbourne, Oxford University Press.

Menges ES 2008. Restoration demography and genetics of plants: when is translocation successful. Australian Journal of Botany 56(3): 187-196.

Montgomery RA, Chazdon RL 2001. Forest structure, canopy architecture, and light transmittance in tropical wet forests. Ecology 82(10): 2707-2718.

Murcia C 1995. Edge effects in fragmented forests: implications for conservation. Trends in Ecology and Evolution 10(2): 58-62.

Myers JH, Bazely D 2003. Ecology and control of introduced plants. Cambridge, Cambridge University Press.

Nabors MW 2004. Introduction to botany. San Francisco, Benjamin Cummings.

Naiman RJ, Decamps H 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28: 621-658.

Newnham RM, Lowe DJ, Green JD 1989. Palynology, vegetation and climate of the Waikato lowlands, North Island, New Zealand, since c. 18 000 years ago. Journal of the Royal Society of New Zealand 19: 127-150.

New Zealand Plant Conservation Network 2011. NZ Plant Conservation Network http://www.nzpcn.org.nz/ (accessed January 2011).

New Zealand Soil Bureau 1954a. General survey of the soils of North Island, New Zealand. Wellington, Government Printer.

New Zealand Soil Bureau 1954b. Soil map of New Zealand: Sheet 3. 1:250,000. Wellington, Department of Scientific and Industrial Research.

Nicholls J 2002. History of the vegetation. In: Clarkson B, Merrett M, Downs T ed. Botany of the Waikato. Hamilton, Waikato Botanical Society Inc. Pp. 23-32.

Nicotra AB, Chazdon RL, Iriarte SVB 1999. Spatial heterogeneity of light and woody seedling regeneration in tropical wet forests. Ecology 80(6): 1908-1926.

Niemela J 1999. Ecology and urban planning. Biodiversity and Conservation 8(1): 119-131.

Parker GG, Brown MJ 2000. Forest canopy stratification - is it useful? The American Naturalist 155(4): 473-484.

Pauleit S, Jones N, Garcia-Martin G, Garcia-Valdecantos JL, Riviere LM, Vidal-Beaudet L, Bodson M, Randrup TB 2002. Tree establishment practice in towns and cities - results from a European survey. Urban Forestry and Urban Greening 1(2): 83-96.

Pickett STA, Cadenasso ML, Grove JM, Nilon CH, Pouyat RV, Zipperer WC, Costanza R 2001. Urban ecological systems: linking terrestrial, ecological, physical, and socioeconomic components of metropolitan areas. Annual Review of Ecology and Systematics 32: 127-157.

Porteous T 1993. Native forest restoration: A practical guide for landowners. Wellington, Queen Elizabeth the Second National Trust.

Pratt C 1999. Factors affecting the establishment, growth and survival of native woody plant communities on the Canterbury Plain, New Zealand. Unpublished MSc thesis, Lincoln University, Lincoln, New Zealand.

Qian H, Klinka K, Sivak B 1997. Diversity of the understory vascular vegetation in 40 year-old and old-growth forest stands on Vancouver Island, British Columbia. Journal of Vegetation Science 8(6): 773-780.

Ramos FN, Santos FAM 2006. Microclimate of Atlantic forest fragments: regional and local heterogeneity. Brazilian Archives of Biology and Technology 49(6): 935-944.

Rebele F 1994. Urban ecology and special features of urban ecosystems. Global Ecology and Biogeography Letters 4(6): 173-187.

Riley M 1994. Maori healing and herbal: New Zealand ethnobotanical sourcebook. Paraparaumu, Viking Sevenseas NZ.

Roberts L, Stone R, Sugden A 2009. The rise of restoration ecology. Science 325(5940): 555.

Robison SA, McCarthy BC 1999. Potential factors affecting the estimation of light availability using hemispherical photography in oak forest understories. Journal of the Torrey Botanical Society 126(4): 344-349.

Romell E, Hallsby G, Karlsson A, Garcia C 2008. Artificial canopy gaps in a *Macaranga* spp. dominated secondary tropical rain forest - effects on survival and above ground increment of four under-planted dipterocarp species. Forest Ecology and Management 255(5-6): 1452-1460.

Rowntree RA 1986. Ecology of the urban forest - introduction to part 2. Urban Ecology 9(3-4): 229-243.

Royo AA, Carson WP 2006. On the formation of dense understory layers in forests worldwide: consequences and implications for forest dynamics, biodiversity, and succession. Canadian Journal of Forest Research 36: 1345-1362.

Ruiz-Jean MC, Aide TM 2006. An integrated approach for measuring urban forest restoration success. Urban Forestry and Urban Greening 4(2): 55-68.

Singleton PL 1991. Soils of Ruakura: A window on the Waikato. Hamilton, DSIR Land Resources Scientific Report No. 5. Department of Scientific and Industrial Research. 127 p.

Small CJ, McCarthy BC 2002. Spatial and temporal variation in the response of understory vegetation to disturbance in a central Appalachian oak forest. Journal of the Torrey Botanical Society 129(2): 136-153.

Smith RL, Smith TM 2001. Ecology and field biology. Sixth ed. San Francisco, Benjamin Cummings.

Society for Ecological Restoration International Science and Policy Working Group 2004. The SER International Primer on Ecological Restoration. www.ser.org and Tucson: Society for Ecological Restoration International.

Sparling G, Schipper L 2004. Soil quality monitoring in New Zealand: trends and issues arising from a broad-scale survey. Agriculture, Ecosystems and Environment 104(3): 545-552.

Spyreas G, Matthews JW 2006. Floristic conservation value, nested understory floras, and the development of second-growth forest. Ecological Applications 16(4): 1351-1366.

Standish RJ 2002. Experimenting with methods to control *Tradescantia fluminensis*, an invasive weed of native forest remnants in New Zealand. New Zealand Journal of Ecology 26(2): 161-170.

Standish RJ, Robertson AW, Williams PA 2001. The impact of an invasive weed *Tradescantia fluminensis* on native forest regeneration. Journal of Applied Ecology 38: 1253-1263.

Stevenson BA 2004. Changes in phosphorus availability and nutrient status of indigenous forest fragments in pastoral New Zealand hill country. Plant and Soil 262(1-2): 317-325.

Sullivan JJ, Meurk C, Whaley KJ, Simcock R 2009. Restoring native ecosystems in urban Auckland: urban soils, isolation, and weeds as impediments to forest establishment. New Zealand Journal of Ecology 33(1): 60-71.

Taha H 1997. Urban climates and heat islands: albedo, evapotranspiration, and anthropogenic heat. Energy and Buildings 25(2): 99-103.

Tarara JM, Hoheisel GA 2007. Low-cost shielding to minimize radiation errors of temperature sensors in the field. HortScience 42(6): 1372-1379.

Terralink 1998. Hamilton. 1: 50,000, Wellington, Land Information New Zealand.

Tilman D 1985. The resource-ratio hypothesis of plant succession. The American Naturalist 125(6): 827-852.

Toniato MTZ, de Oliveira-Filho AT 2004. Variations in tree community composition and structure in a fragment of tropical semideciduous forest in southeastern Brazil related to different human disturbance histories. Forest Ecology and Management 198(1-3): 319-339.

Vallet J, Beaujouan V, Pithon J, Roze F, Daniel H 2010. The effects of urban or rural landscape context and distance from the edge on native woodland plant communities. Biodiversity and Conservation 19(12): 3375-3392.

Vidra RL, Shear TH, Stucky JM 2007. Effects of vegetation removal on native understory recovery in an exotic-rich urban forest. Journal of the Torrey Botanical Society 134(3): 410-419.

Vora RS, Lerol S, Danz NP 2008. Multi-species planting and other practices to restore forest diversity in northeastern Minnesota. Ecological Restoration 26(4): 340-349.

Walker M, Wass R 2006. He kimihanga turangawaewae - finding our place. In: Parsons S ed. Biology Aotearoa: unique flora, fauna and fungi. Auckland, Pearson Education New Zealand.

Wall K, Clarkson B 2006. Gully restoration guide: A guide to assist in the ecological restoration of Hamilton's gully systems. Hamilton, Hamilton City Council.

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan II RP 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24(3): 706-723.

Wardle P 1991. Vegetation of New Zealand. Cambridge, Cambridge University Press.

Wassie A, Sterck FJ, Teketay D, Bongers F 2009. Effects of livestock exclusion on tree regeneration in church forests of Ethiopia. Forest Ecology and Management 257(3): 765-772.

Weathers KC, Cadenasso ML, Pickett STA 2001. Forest edges as nutrient and pollutant concentrators: potential synergisms between fragmentation, forest canopies, and the atmosphere. Conservation Biology 15(6): 1506-1514.

Weiner J 1988. Variations in the performance of individuals in plant populations. In: Davy AJ, Hutchings MJ, Watkinson AR ed. Plant population ecology. London, Blackwell Scientific Publications. Pp. 59-82.

Wendelberger KS, Fellows MQN, Maschinski J 2008. Rescue and restoration: experimental translocation of *Amorpha herbacea* Walter var. *crenulata* (Rybd.) Isley into a novel urban habitat. Restoration Ecology 16(4): 542-552.

Young A, Mitchell N 1994. Microclimate and vegetation edge effects in a fragmented podocarp-broadleaf forest in New Zealand. Biological Conservation 67(1): 63-72.

Appendices

Appendix A: Sample understory vegetation survey recording sheet

Date:	**Location:**
Coordinates: E	**Altitude:**
N	**Physiography:** Ridge, Face, Gully, Terrace
Slope / Aspect:	**Plot size:**
Canopy Height:	**Canopy Type:**

Height Class	**Species**	**Tally**

<u>Notes:</u>

Ground Cover

Appendix B: Native understory vegetation densities (0.3-12 m)

Species	Rural forests (ha^{-1})	Urban forests (ha^{-1})
Alseuosmia macrophylla	8	
Alseuosmia quercifolia	184	3
Aristotelia serrata	12	20
Brachyglottis repanda	104	
Carmichaelia australis	4	
Carpodetus serratus	168	3
Coprosma arborea	180	
Coprosma areolata	132	129
Coprosma grandifolia	412	34
Coprosma lucida	280	14
Coprosma repens		11
Coprosma rhamnoides	68	
Coprosma robusta	40	489
Coprosma rotundifolia	620	23
Coprosma spathulata	64	
Coprosma tenuicaulis	32	29
Cordyline australis		66
Dodonaea viscosa	4	
Fuchsia excorticata	8	34
Geniostoma ligustrifolium	1012	117
Griselinia lucida		3
Hebe stricta	4	
Hedycarya arborea	748	6
Hoheria sextylosa	120	340
Kunzea ericoides	8	80
Leucopogon fasciculatus	148	
Macropiper excelsum	784	143
Melicope simplex	16	6
Melicytus lanceolatus	4	
Melicytus micranthus	224	3
Melicytus ramiflorus	1088	1426
Myrsine australis	492	117
Myrsine salicina	4	
Olearia rani	468	

Pennantia corymbosa	108	
Pittosporum crassifolium		31
Pittosporum tenuifolium		46
Pseudopanax arboreus	4	
Pseudopanax crassifolius	84	6
Pseudopanax hybrid		137
Rhabdothamnus solandri	48	
Rhopalostylis sapida	316	14
Schefflera digitata	180	126
Sophora chathamica		6
Streblus banksii		6
Streblus heterophyllus	48	77