Vegetation change and native forest restoration in urban environments: Management options for post-earthquake Christchurch

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by

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Global biodiversity is threatened by human actions, including in urban areas. Urbanisation has removed and fragmented indigenous habitats. As one of the 35 biodiversity 'hot spots', New Zealand is facing the problems of habitat loss and indigenous species extinction. In New Zealand cities, as a result of the land clearance and imported urban planning precepts, many urban areas have little or no original native forest remaining. Urbanisation has also been associated with the introduction of multitudes of species from around the world.

Two large earthquakes shook Christchurch in 2010 and 2011 and caused a lot of damage. Parts of the city suffered from soil liquefaction after the earthquakes. In the most damaged parts of Christchurch, particularly in the east, whole neighbourhoods were abandoned and later demolished except for larger trees.

Christchurch offers an excellent opportunity to study the biodiversity responses to an urban area with less intensive management, and to learn more about the conditions in urban environments that are most conducive to indigenous plant biodiversity.

This study focuses on natural woody plant regeneration of forested sites in Christchurch city, many of which were also surveyed prior to the earthquakes. By repeating the pre-earthquake surveys, I am able to describe the natural regeneration occurring in Christchurch forested areas. By combining this with the regeneration that has occurred in the Residential Red Zone, successional trajectories can be described under a range of management scenarios. Using a comprehensive tree map of the Residential Red Zone, I was also able to document minimum dispersal distances of a range of indigenous trees in Christchurch. This is important for planning reserve connectivity. Moreover, I

expand and improve on a previous analysis of the habitat connectivity of Christchurch (made before the earthquakes) to incorporate the Residential Red Zone, to assess the importance for habitat connectivity of restoring indigenous forest in this area. In combination, these data sets are used to provide patch scenarios and some management options for biodiversity restoration in the Ōtākaro-Avon Red Zone post-earthquake.

Keywords: Forest regeneration, Indigenous biodiversity, Habitat connectivity, Residential Red Zone, Urban ecology.

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Chapter 1

Native plant biodiversity in New Zealand urban areas: an introduction

1.1 Global Biodiversity

Global biodiversity refers to the full extent of the diversity of life on earth. Biodiversity includes all terrestrial, freshwater, marine and subterranean organisms, which includes plants, animals, fungi and microbes. The scale of diversity ranges from genetic to species to community diversity (Sala et al., 2000). Now, around 30 million different species of organism live on this planet with us (Reid, 1998; Strain, 2011; Stork, 2018). Moreover, there are 25 terrestrial biodiversity 'hotspots' globally (Subsequent revisions now list 35 terrestrial (Mittermeier et al., 2004; Mittermeier, Turner, Larsen, Brooks, & Gascon, 2011; Hopper, Silveira, & Fiedler, 2016), and 10 marine biodiversity hotspots (K. Williams et al., 2011; Hopper et al., 2016)), including New Zealand, which contain large numbers of endemic species found in relatively small areas that are facing significant threats of habitat loss (Reid, 1998). As many as 44% of all vascular plants and 35% of all four vertebrate groups species live in these 25 hotspots which comprise only 1.4% of the land surface of the Earth (Myers et al., 2000). However, many of these areas are under threat from land-use changes that will reduce biodiversity. Because of a biodiversity crisis caused by human actions, species extinction rates are up to 1000 times higher than historical background (Barnosky et al., 2011; De Vos et al., 2015). Based on these rates, by the mid-21st century, 30% of all species may become extinct (Brooks et al., 2006; Novacek & Cleland, 2001).

1.2 Biodiversity in New Zealand

New Zealand has a high level of endemic biodiversity. There are at least 52,517 known native species in New Zealand, of which a minimum of 27,380 (52%) species are endemic, including 78% of vascular plants and 91% of terrestrial animals (Gordon, 2013). However, native biodiversity has declined considerably since NZ was first settled. About 63% of New Zealand's land area has been converted from forest in the last 700–800 years (Proce et al., 2006; Ewers et al., 2006; Christchurch City Council, 2008). Today, nearly 42% of New Zealand is agricultural and horticultural land. Native forest covers about one-quarter of this country while wetlands are reduced 90% of their original extent (Ministry for the Environment & Statistics New Zealand, 2015). On one hand, because of human impacts, many native species have lost their habitats, while on the other hand, large numbers of exotic species were introduced into New Zealand though human activities, both purposefully and accidentally. Predation and competition from exotic species have subsequently reduced the numbers of native species (Christchurch City Council, 2008). In the last 700-800 years, about 32% of land and freshwater birds, 18% of sea birds, and 11 plants have become extinct (Department of Conservation, 2000). By the year 2000, there were about 1000 species known to be under threat (Department of Conservation, 2000).

1.3 An urban role for biodiversity

1.3.1 Urbanisation: Implications for people

Urbanisation is an inexorable trend of global development, and cities have become an important kind of novel ecosystem. More than half of the world's population are now living in urban areas and this number is projected to grow at a rate of 67 million per year (Pickett et al., 2011). Urbanisation not only influences human wellbeing, but also creates both problems and opportunities for nature (Grimm et al., 2008; Newman, Beatley, & Boyer, 2009).

1.3.2 Urban biodiversity: urban roles for species habitats

Urbanisation has reduced the number of native habitats and fragmented the remainder (Wu, 2014). However, urban areas, which have been highly modified, have added a different range of landscapes and habitats, including green or open spaces for humans and wildlife (Angold et al., 2006; Pickett et al., 2008; Pickett, Cadenasso, & Grove, 2004).

They provide suitable living conditions and habitats for some wildlife and can have a high level of species richness when both native and introduced communities and species are counted (Pyšek, 1993; Tallamy, 2009). Importantly, the environmental conditions offered by urban areas comprise both artificial and naturalistic habitats, providing a wide range of opportunities for both native and exotic species.

1.4 Urban biodiversity in New Zealand

1.4.1 Urban areas in New Zealand

Compared with the extensive historical human influence in 'Old World' countries, which can date back thousands of years, it is only about 750 years since people settled in New Zealand (Wilmshurst, Anderson, Higham, & Worthy, 2008). However, species introductions since European settlement in the 19th century have caused New Zealand's urban areas to become homogenised with those from around the world. As a result of land clearance and imported urban planning precepts, many urban areas have little native forest remaining. Because exotic planting began over the past 150 years ago in many New Zealand cities, a large number of introduced species have become established in and around human settlements and urban areas are dominated by exotic species which were imported for production and amenity (Brockie, 1997; Esler, 2004; Clarkson et al., 2007; Freeman & Buck, 2003).

1.4.2 Native and Exotic species

Both native and exotic species are important components of urban species richness, but while total species richness may go up, native biodiversity may drop. Therefore, planting exotic plants may increase local species richness, but it can reduce existing native biodiversity (Given & Colin, 2000) when those plants displace or compete with native species. As more exotic species are introduced, more native species will be displaced, meaning that the well-adapted genetic forms and distinctive ecosystems can be disrupted or lost (Norton & Miller, 2000; Christchurch City Council, 2008).

1.4.3 Biodiversity in New Zealand urban areas

As shown in Table 1.1, the total number of vascular plants in New Zealand's main cities and comparable cultural landscapes range from 350 to 559 (Given & Colin, 2000). The plants in these sites account for 14–22% of New Zealand' total vascular plants (Given &

Colin, 2000). There are still a lot of native species living in New Zealand urban areas and the challenge is how to make their populations sustainable and more abundant (Given & Colin, 2000).

Table 1.1: Minimum estimates for vascular plant biodiversity in New Zealand cities or cultural landscape from Given & Meurk (2000). (Nat. = native vascular species richness, Adv. = adventive vascular species richness)

	Auckland	**Rotorua**	**Manwatu**	**Christchurch**	**Dunedin**	**New Zealand**
Nat.	559	540	500	350	470	2500
Adv.	615	545	525	>500	211	2500
Area(ha)	265 200	195 000	105 200	40 000	37 500	35 700k
Nat/1000ha	2.11	2.77	4.75	8.75	12.5	0.07
Nat.%of. NZ	22.4	21.6	20	14	18.8	100

1.5 Habitat Connectivity of Cities

1.5.1 Biodiversity decline in fragmented habitats

Urbanization is one of the major drivers of biodiversity loss (Mimet, Kerbiriou, Simon, Julien, & Raymond, 2019). It causes habitat loss and habitat fragmentation (Spellerberg, 1995; Fahrig, 2003) both of which will reduce species biodiversity (McDonald, Kareiva, & Forman, 2008; Olden, Poff, & McKinney, 2006; Newbold et al., 2015; Pellissier, Mimet, Fontaine, Svenning, & Couvet, 2017; Threlfall, Law, & Banks, 2012), as well as leading to biotic homogenisation (McKinney, 2006). The effects of habitat loss and habitat fragmentation are due to several inter-related processes: reduction in total habitat area and accompanying species-area effects, reduction in patch size and accompanying edge effects (Devictor et al., 2008; Devictor, Julliard, Couvet, Lee, & Jiguet, 2007), and increasing distance between patches (Spellerberg, 1995; Clauzel, Jeliazkov, & Mimet, 2018). These processes in combination are called 'fragmentation' (Fahrig, 2003).

Habitat connectivity is important to help ecosystem to overcome habitat loss and fragmentation (Z. Zhang, Meerow, Newell, & Lindquist, 2019; P. L. Thompson, Rayfield, & Gonzalez, 2017). Connectivity allows habitat patches to maintain an exchange of individuals, and gene flow, with other habitat patches (Taylor, Fahrig, Henein, & Merriam, 1993). To enhance urban landscape connectivity, habitats should be linked by corridors which can form a network facilitating dispersal and movement (Vergnes, Le Viol, &

Clergeau, 2012) and quantifying habitat connectivity has become an important management method to identify key conservation areas and maintain connectivity (Trapp, Day, Flaherty, Zollner, & Smith, 2019).

Lots OF work has been done about quantifying habitat connectivity in the literature (Ernst, 2014; Tischendorf & Fahrig, 2000; Zeller, McGarigal, & Whiteley, 2012; Rayfield, Fortin, & Fall, 2011; Moilanen & Nieminen, 2002; Adriaensen et al., 2003). Among these, graph theory, network theory, and circuit theory are increasingly being used to quantify multiple aspects of habitat connectivity and protected areas (Rayfield et al., 2011). These graph-based measures were either designed specifically for assessing the habitat connectivity (Saura & Pascual-Hortal, 2007) or from other fields such as social science, transportation theory and communication theory (Fortuna, Gómez-Rodríguez, & Bascompte, 2006).

1.5.2 Increasing connectivity leads increases of urban biodiversity

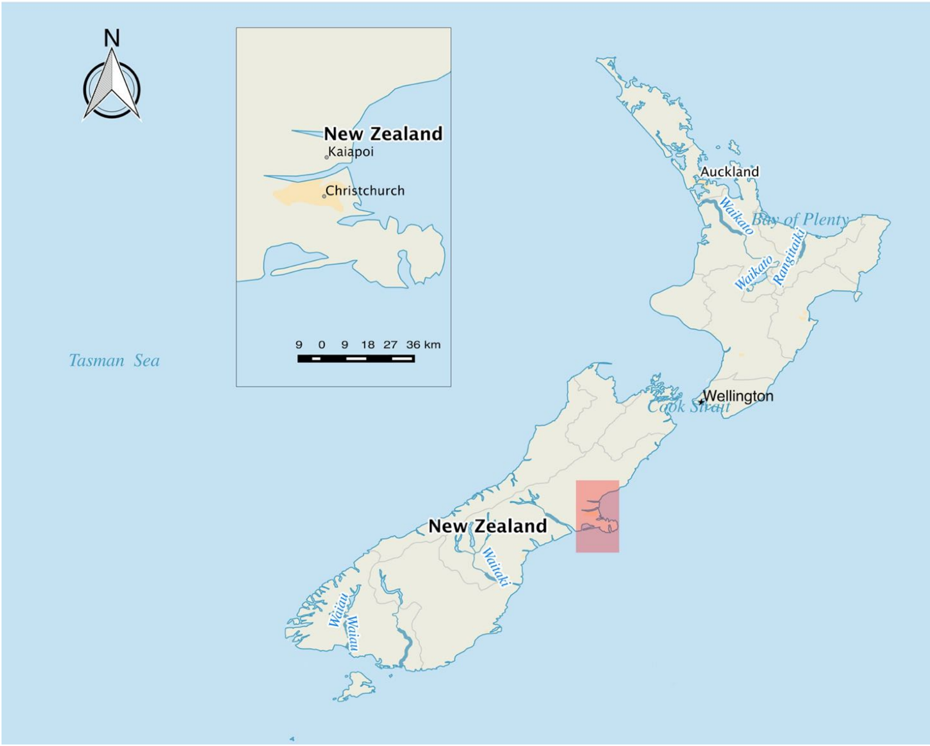
Most conservation work focuses on maximizing biodiversity in protected areas. However, in urban areas, where reserve space is limited, habitat corridors become an important feature of habitats which can increase the species richness, gene flow and reduce the extinction in the habitats by facilitating movement of organisms between these habitats (Brudvig et al., 2009; Levey et al., 2005). A lot of studies have demonstrated that corridors have positive effects on increasing species biodiversity (Beier & Noss, 1998; Tewksbury et al., 2002; Haddad et al., 2003; Gilbert-Norton et al., 2010; Damschen et al., 2006).

1.6 The background and ecological history of Christchurch

1.6.1 Christchurch City characteristics

Christchurch is the third largest city in New Zealand, and is located on the relatively dry eastern coast of the South Island (Fig. 1.1). Internationally, Christchurch is a young city, founded in 1850 (Wilson, 1989a). The climate is cool temperate and oceanic. January is the warmest month with a mean daily maximum of 21.4 °C and July is the coldest month with a mean daily maximum temperature of 10.2 °C (McGann, 1983). Average annual rainfall in Christchurch varies between 600 and 700 mm (McGann, 1983). Christchurch has average 1985 hours of sunshine every year and average 89 ground frosts and 37 air frosts (McGann, 1983). Although hot and desiccating foehn winds are common in summer, this city still gets moderate summer rainfalls. Christchurch has a low mean annual rainfall of around 660 mm but rain falls all year round (McGann, 1983). There are a range of natural habitats in the city, including wetlands, coastal habitats, grasslands, drylands, hills and a small patch of remnant forest (Christchurch City Council, 2000).

FIGURE 1.1: Map showing location of Christchurch, New Zealand.



1.6.2 Vegetation changes in Christchurch

Pre-human settlement

Christchurch has two dominant landscape types (Wilson et al., 2005). They are the flat of the Canterbury Plains, the volcanic Port Hills to the south (Wilson et al., 2005). The city was mostly built on a mosaic of shingle lobes which was deposited by the Waimakariri River interspersed and overlaid with swamplands, waterways, and sandhillst (Wilson et al., 2005). Two small spring-fed rivers, the Heathcote and Avon Rivers, drain the city swamplands into the Avon-Heathcote estuary (Wilson et al., 2005). In north of the city, the Styx River flows into the Brooklands lagoon at the mouth of the Waimakariri River, and to south of the city, the Halswell River flows into Te Waihora Ellesmere lagoon (Wilson et al., 2005)..

Christchurch was originally covered by three main natural vegetation types: swamplands (flax and rushes), grasslands with shrubs (e.g., kanuka, matagouri, ribbonwood and cabbage trees) and patches of kahikatea and totara dominated evergreen forests (Wilson et al., 2005). The Port Hills were covered by evergreen forests (Wilson et al., 2005).

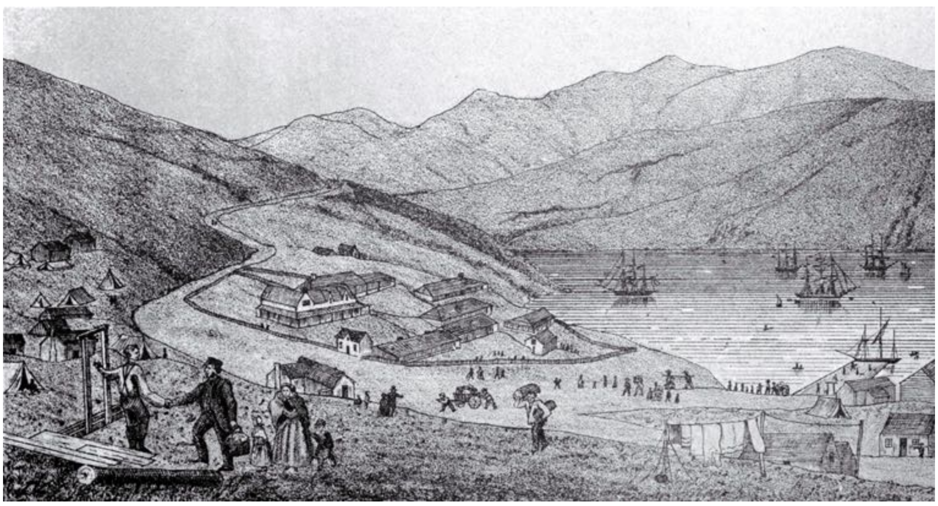
Māori times

The wetlands in the city's area provided abundant food resources for Māori (Wilson et al., 2005). Most forest in Christchurch and the adjacent Canterbury Plains was destroyed by Māori fires before European's settlement (Wilson et al., 2005) . These Māori fires reduced the forest cover on the Port Hills by between 30 and 50 percent (Wilson et al., 2005). The short tussock areas which are now common on the hills developed in Māori times (Wilson et al., 2005). By the time Europeans arrived, there was only between 15 and 20 percent forest cover on the east Hills and between 50 and 75 percent left on the westside of the hills (Wilson et al., 2005). There was almost no remaining forest left on the Platins in early European times (Wilson et al., 2005).

European colonisation

At the beginning of European settlement (Fig. 1.2), the swampland and forest vegetation was seen as a barrier for the early settlers to overcome. A lot of records about the untamed things on the roads were given by early surveyors. For example, "Moorhouse Avenue was covered with tall flax, so tall that lanes had to be cut through it, to survey the street lines and section pegs (Dobson, 1924)".

Figure 1.2: Port Lyttelton, showing the first four ships and emigrants landing from the Cressy, December 28th 1850. Christchurch City Libraries



Botanist J. B. Armstrong showed his foresight in his paper of 1879 on the flora of Canterbury. He wrote that the native vegetation of Christchurch would change through the introduction of foreign weeds and the exotic plants would fill in the places where native plants die out.

A large number of plants were introduced to Christchurch during that time, for many reasons: agriculture, horticulture, colonial sense of order, nostalgia, familiarity, good growth performance and shelter needs (Gabites & Lucas, 2007).

Becoming the "Garden City"

In the 150 years since the founding of Christchurch in 1850, this planned English settlement grew into a city of over 350,000 people (Stewart, Ignatieva, Meurk, & Earl, 2004). After forest clearance, wetland draining, and exotic planting, this site, once a flat, waterlogged, forested floodplain, had become the self-proclaimed "Garden City" of New Zealand. Suburbs were filled with English gardens and parks with the added influences of immigrants from other countries (Gabites & Lucas, 2007). Although many exotic plants have established in Christchurch (Mahon, 2007), and almost all of the original vegetation had been cleared (Fig. 1.3), most of the pre-colonial native plant species still can be found in small patches, mostly on the more wild fringes of the city (Christchurch City Council, 2000).

In the late 20th century, there was a reawakened focus on the City's natural heritage and native plants. Today, most of exotic trees in Christchurch are European and Asian deciduous hardwoods, North American conifers, and Australian gums (Meurk, 2008). However, there has been a shift towards planting native plants in recent decades, at least in parts of the city. Indeed, it has been estimated that the total native forest cover in Christchurch now, largely from plantings in forest restoration projects, is greater than at the time the city was founded by European settlers (Robin Stove, Trees for Canterbury, pers. comm.).

FIGURE 1.3: The stump of the last tree from Papanui Bush, an old growth forest fragment that was cleared as Christchurch was first built. This stump was later removed to tidy up this park. Image source: (Molloy, 1995).

Christchurch damaged by earthquakes

An earthquake measuring 7.1 on the Richter scale shook Christchurch on 4 September 2010. Although the epicentre was 50 km away from Christchurch city, the earthquake and its aftershocks still caused substantial damage across Christchurch, especially in the east. Much of the area suffered from soil liquefaction or the related effects of lateral spreading (Vallance & Tait, 2013).

Another earthquake, of magnitude 6.3, shook the city on 22 February 2011. This quake was shallow and within the city limits and caused much more damage. It brought further liquefaction to the eastern suburbs of Christchurch. After this quake, a new Government Department was established, the Canterbury Earthquake Recovery Authority (CERA), which replaced the Canterbury Earthquake Recovery Commission (CERC) (Vallance & Tait, 2013). CERA was tasked with managing the demolition and rebuild of the damaged parts of the city.

Post-earthquake

The main damage of the two earthquakes and their aftershocks to the city's trees were caused by mass soil movement, soil liquefaction, rock falls and land slips. By June 2011, 384 trees directly impacted by the earthquakes had been removed from city parks (Christ & Shane, 2009). That's a direct result of the earthquakes in the short time. However, this is a relatively small impact as, in the long term, the number of trees removed from public lands typically ranged from 800–900 in a 'normal' year before earthquakes (Christ & Shane, 2009). The bigger impact on city trees, both through removal and planting, has occurred in the years subsequent to the earthquakes, as land use decisions have been made and implemented that have modified the city landscape.

1.6.3 Restoration activities in Christchurch

Most nature restoration activities in Christchurch focus on the sites which are more natural, and less modified, and less densely populated. The waterways that run through the city are good examples although the city's waterways have all been substantially modified. The waterways of Christchurch are made up of two main rivers, the Avon River and Heathcote River, and other minor waterways. The restoration activities of waterway enhancement have been about contouring the banks and planting native vegetation.

Waterway plantings can improve the ecological functions and build habitats for native wildlife (Alastair & Rachel, 2002). For example, the city transitioned in the 1980s from its historical practise of closely mowing river banks to allowing/planting denser riparian wetland vegetation. This is credited as the likely main cause of the extraordinary increase in the native scaup (*Aythya novaeseelandiae*), which increased from less than 100 birds in 1985 to many thousands in subsequent decades (Hartley, 2003).

1.6.4 Christchurch Biodiversity Strategy

The Christchurch City Council's Christchurch Biodiversity Strategy gave a vision, goals and objectives for the protection and enhancement of native biodiversity in Christchurch by 2035 (Christchurch City Council, 2008). The Strategy formally recognised that native biodiversity is an important part of a healthy ecosystem, and many ecosystems are dominated by exotic species (Christchurch City Council, 2008). In this Strategy, in order to combine the natural heritage and human heritage, the Christchurch City area aimed to continue having a capacity to support both native and exotic species, with the objective of enhancing biodiversity (Christchurch City Council, 2008).

The key conservation actions in the Strategy (Christchurch City Council, 2008) were as follows.

- Establish recovery plans for each of the vulnerable or locally rare species in Christchurch region.
- Expand and improve the range and species used in restoration plantings.
- Improve restoration and amenity planting planning and maintenance templates to reduce losses of plants and ensure better outcomes.
- Plan for optimal configurations of sanctuary, patches of habitat and native tree groves throughout the city and surrounding districts to enhance the overall long term sustainability of nature in the city, and provide areas of habitat and species suitable for customary use and wider utility.

1.7 Residential Red Zone in Christchurch

1.7.1 A brief view of Residential Red Zone

The Residential Red Zone was a public exclusion zone in eastern Christchurch created on 23rd June 2011 (Fig. 1.4). The area is 832.87 hectares. In this area, infrastructure was so destroyed by the earthquakes that whole neighbourhoods had to be bought up by the government and demolished (Fig. 1.6). Now most people have moved out of residential red zone with more than 7,000 properties being purchased by The Crown under a voluntary yet coercive scheme. CERA coordinated the individual property clearances in Residential Red Zone (Fig. 1.7) (Vallance & Tait, 2013). This research will focus on the part in eastern Christchurch which is 443.90 hectares.

While earthquake damaged houses were demolished, CERA did its best to retain the surviving trees of these properties. This area is the main focus of my research.

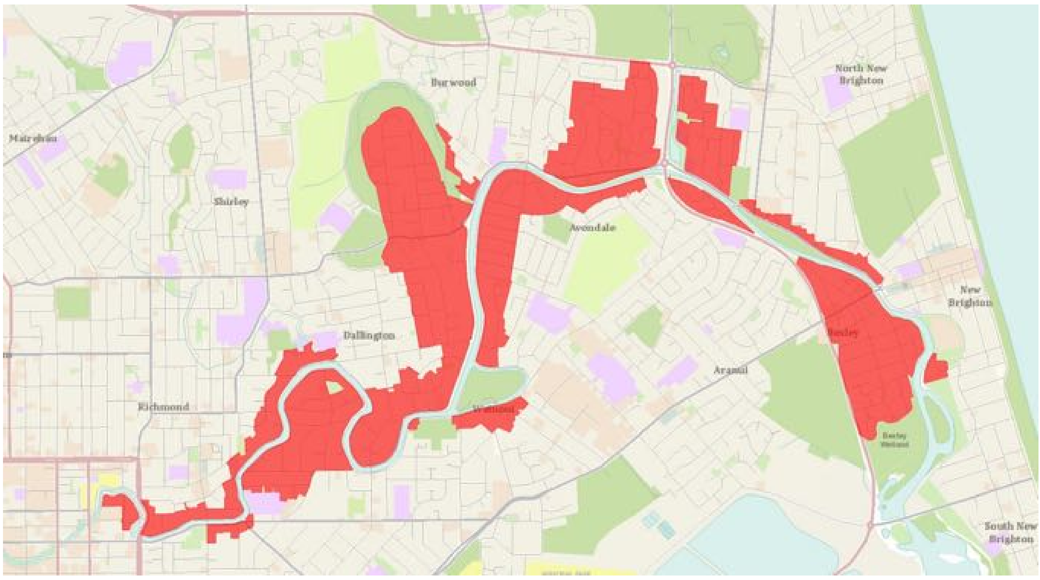


FIGURE 1.4: The Residential Red Zone of eastern Christchurch. The red areas were residential properties badly damaged by the earthquakes and purchased by the NZ Government. Buildings were demolished but most garden trees were retained. Source: CERA.

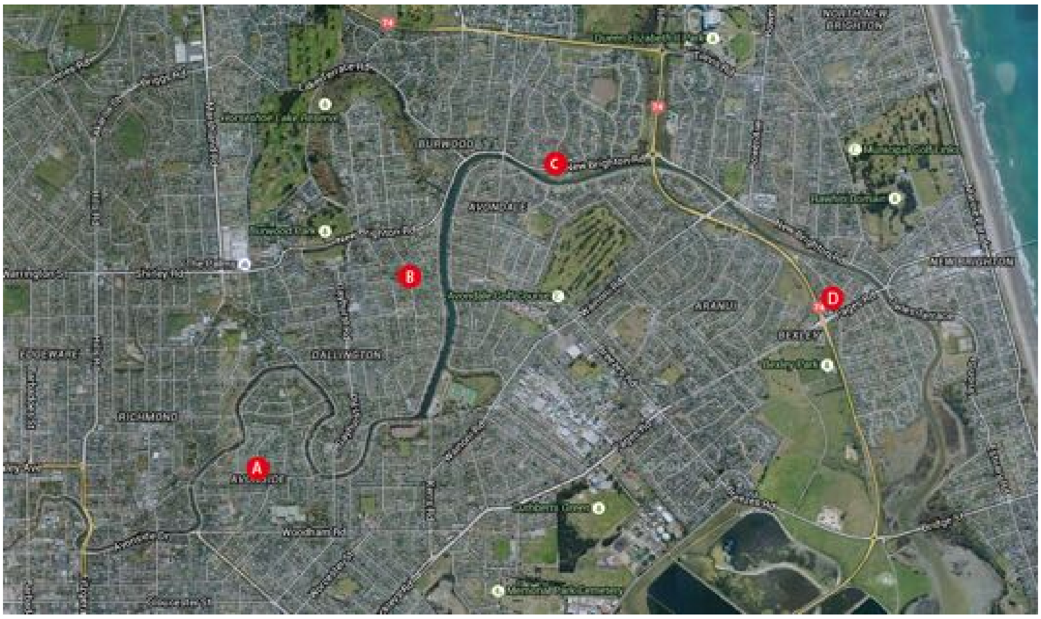


FIGURE 1.5: The Residential Red Zone map shows four sites locations. This map was taken in August 2012 from Google Earth.

Site A

Site B

Site C

Site D

FIGURE 1.6: The map of sites which show the RRZ before (December 2007) and after the earthquakes (August 2012) from Google map

1.7.2 Recent Residential Red Zone

Figure 1.7: Photos show the house removing in the Residential Red Zone. The top photo was in 2004 and the bottom one was in 2019. Source: Google Earth.

1.7.3 Why choose the Residential Red Zone?

The Residential Red Zone used to be a place of residential housing. After the earthquakes, CERA cleared the buildings from individual properties and left some vegetation "prior to the larger scale block clearances" (Vallance & Tait, 2013). The remaining vegetation includes the great majority of the larger trees planted in the private gardens and essentially all pre-earthquake vegetation from parks and reserves. These trees are a mixture of species with a large number and diversity of exotic plants but also many native plants.

The Residential Red Zone is therefore a good place to do research about vegetation change in urban areas. The Residential Red Zone gives unparalleled access to a large area of accessible urban habitat where trees and seedlings can be surveyed. This would have been prohibitively difficult across the many private properties before the earthquakes. This therefore provides an unusually comprehensive large-scale view of the trees that had been planted in private gardens. The ease of access and reduced intensity of human management also make it a useful place to investigate the potential for natural tree regeneration, by both exotic and native species, in urban environments. Also, there is the hope that much of the Residential Red Zone can be restored to native forest become an important part of the habitat networks of Christchurch to enhance the city's urban biodiversity. My work can inform these restoration plans.

1.8 Aim and Objectives

1.8.1 Aim

The aims of my research are to better understand natural forest regeneration processes in urban Christchurch and to provide some ecological restoration guidelines and a range of broad options for restoration in the Residential Red Zone area of eastern Christchurch.

My expectation, and hope, is that this detailed insight into vegetation dynamics in urban Christchurch will also be of relevance to ecological management and restoration in other urban centres in New Zealand and internationally.

1.8.2 Objectives

My research was a combination of field work, data collection and analysis, with the following objectives.

1. To describe the planting history of trees in eastern Christchurch private gardens. Do more recently established gardens have a higher proportion of native trees? How does the age and wealth of a neighbourhood affect the proportion of trees are native? (See **Chapter 2**.)
2. To assess the effects of a range of environmental factors, including suburb age, nearby adult trees, soil, and hydrology, on the natural tree regeneration occurring in urban Christchurch. (See **Chapter 3**.)

3. To assess how far away from parent trees wild seedlings typically establish in urban landscapes. This knowledge is important for assessments of habitat connectivity. (See **Chapter 4**.)

4. To investigate natural woody plant regeneration of forested sites in Christchurch by repeating and expanding on a previous survey. How quickly does the amount, and diversity, of tree regeneration increase with time after native tree planting? (See **Chapter 5**.)

5. To assess the landscape connectivity for forest species in urban Christchurch, including the Residential Red Zone area. (See **Chapter 6**.)

6. Based on the results above, to propose scenarios of ecological restoration in the Residential Red Zone area. (See **Chapter 7**.)

These objectives are addressed consecutively in each of the subsequent chapters of this thesis.

Chapter 2

Factors affecting on the distribution and abundance of tree species planted in Christchurch gardens

2.1 Introduction

Urban growth over the past century has been unprecedented. Since 2008, over half of the world's population now live in cities and this number is growing at a rate of 67 million per year (Pickett et al., 2011; Montgomery, 2007). About 4% of global land area (more than 471 million ha) is presently covered by urban environments (Gaston et al., 2005). In developed countries, more than 80% of people are living in urban areas (Montgomery, 2007). Although urban areas remain a relatively small fraction of the land surface, urbanisation is an inexorable trend of global development (Antrop, 2004) and cities have become an important kind of novel ecosystem (Goddard, Dougill, & Benton, 2010).

It is well known that green space and plants in urban areas bring benefits to people living there. For example, plants reduce the summer high temperature and air pollution (Rowntree & Nowak, 1991; McPherson & Rowntree, 1993). Also, some studies show that urban vegetation influences human well-being (Attwell, 2000; Ulrich, 1984; Ulrich et al., 1991). Well vegetated green space improves people's physical and emotional health (Fraser, Kenney, & Andrew, 2000). Moreover, plants provide wildlife with habitats and protection and so both directly and indirectly increase species diversity (Fraser et al., 2000).

However, urbanisation is also associated with the loss of natural habitats and native species by fragmenting, reducing, and degrading native vegetation, which consequently results in native wildlife declines (Collinge, 1996). Because of human settlement and land modification, wild natural habitats get smaller and more isolated. Many studies show that city construction and expansion decreases native plants and promotes exotics (McKinney, 2006; Duncan & Young, 2000; Esler, 2004). New York City is a good example: for the historical and modern periods combined, it lost 578 native plant species in the metropolitan area (42.6% of 1357 native species) while gaining 411 wild exotic plants (DeCandido, Muir, & Gargiullo, 2004).

Two main factors promote exotic plants in cities. One is the importation of exotic plant species, and the other is suitable environments for their establishment. The first may occur due to ecological invasiveness or for cultural reasons. Human settlements import exotic plants for many reasons, such as trade, cultivation for human or livestock food, amenity, garden decoration (Mack & Lonsdale, 2001) and accidentally as contamination in purposeful imports.

Specifically, there are three broad categories for exotic plants movement: accidental, utilitarian and aesthetic (Mack & Lonsdale, 2001). Although accidental introductions of exotic plants still happen today, they are less common because of seed cleaning techniques and quarantine inspections (Mack & Lonsdale, 2001). A good example is the naturalised species which arrived in Australia between 1971 and 1995. Only 2% of these are known to have been introduced as accidental contaminants (Groves, 1997). In New Zealand, there were 2,252 wild exotic species found by the year 2000 (Gatehouse, 2008). About 75 % of them were deliberately introduced and 77% were used for ornamental horticulture only (Gatehouse, 2008). Utilitarian introductions predominated during early colonisation in New Zealand and continues to this day. Colonists introduced more and more useful species in order to provide reliable sources of food, fibre and fodder (Mack & Lonsdale, 2001). When colonists came to a new place, they also began to import ornamental plants from their home countries and elsewhere (Mack & Lonsdale, 2001). On one side they tried to make the colonies to be like their homelands; on the other side people desired to collect novel and even bizarre plants for in their gardens from the 'new' lands (Mack, 1991).

Human settlement by definition creates frequently disturbed habitat conditions that are favourable for many weedy exotic plants to establish and flourish. These habitat conditions are usually quite different from pre-existing local habitats. For example, increased disturbances tends to help weedy exotic plants to establish (D'antonio & Meyerson, 2002) and disadvantages many slow growing local native plants of later succession.

As a dominant land use in cities, domestic gardens are an important component of urban green space and have the potential to make contributions to maintaining local biodiversity (Gaston et al., 2005; Doody et al., 2010). City gardens are usually not big but they are numerous, therefore they are a major component of urban nature conservation strategies (Goddard et al., 2010; Loram, Tratalos, Warren, & Gaston, 2007). Some ecological research of urban gardens include investigations of biodiversity of individual gardens (Sullivan et al., 2009; Thompson et al., 2003), land cover composition and changes at various spatial scales (Mathieu, Freeman, & Aryal, 2007) and the contribution urban gardens may make to sustaining and conserving biodiversity (Cameron et al., 2012; Goddard et al., 2010; Ignatieva, Stewart, & Meurk, 2008; Doody et al., 2010). Domestic gardens can also act as important sources of exotic plants (Duguay, Eigenbrod, & Fahrig, 2007; Sullivan, Timmins, & Williams, 2005a) and some garden species have or could become invasive (Reichard & White, 2001; Crooks, 2005; Sullivan, Williams, Cameron, & Timmins, 2004). These (former) are referred to as 'garden escapes'.

In New Zealand, forest of drier parts of the country was burnt off after Polynesian arrival (Wilmshurst, McGlone, & Partridge, 1997; Perry, Wilmshurst, & McGlone, 2014a) and, during the last 150 years, cleared more intensively for the establishment of exotic plant based agriculture and forestry. As J. B. Armstrong wrote in 1880: "No account, however short, of the plants of Canterbury would be complete without some reference to those plants which have been introduced through the agency of colonisation. Wherever settlement extends the native plants rapidly die out, and their places are filled by British and other exotic plants, mostly of a very weedy nature... There can, I think, be no doubt whatever that the native vegetation will eventually be almost, if not entirely exterminated, and the floral features of the country altogether changed through the introduction of these foreign weeds" (Winterbourn et al., 2008).

In 2000, the total number of fully naturalised vascular plants in each of the cities of New Zealand ranged from 350 to just over 550 species which represents 14–22% of New Zealand's total flora at each site (Given & Colin, 2000; Esler, 2004). The number of naturalised exotic plant species now outnumber the total number of native species (Stewart et al., 2004) and only small areas of native dominated vegetation remain within built-up areas, such as the old growth forest fragment, Riccarton Bush, in Christchurch (Molloy, 1995).

As urban areas increase globally, private gardens play an increasing important role as they can potentially make contributions to urban biodiversity (Smith, M, Gaston, Warren, & Thompson, 2005; Stewart, Meurk, et al., 2009), ecosystem functioning (Sperling

& Lortie, 2010) and providing habitats for native wildlife (Cameron, 2012). Private gardens are common in urban areas (van Heezik, Freeman, Porter, & Dickinson, 2013) and comprise a substantial proportion of the urban area (van Heezik et al., 2013). The estimated proportions of private garden area in cities ranges from 16% in Stockholm, Sweden (Colding, Lundberg, & Folke, 2006), through to around 25% in UK (Loram et al., 2007) and 36% in Dunedin, New Zealand (Mathieu et al., 2007). Private gardens are therefore a large proportion of all urban green space of urban area, such as 35% in Edinburgh and 47% in Leicester (Loram et al., 2007). Considering that private gardens are probably the biggest single contributor to urban green space (Gaston et al., 2005), they may also be the largest source of planted trees (Smith, Hodgson, & Gaston, 2006).

Ggardening practice is one of the most common and popular urban past times (van Heezik et al., 2013). In UK, about 52% of householders do gardening (Bhatti & Church, 2004) while in USA 78% of the householders practice gardening (Clayton, 2007). However, the fact is exotic plants are often more popular than native plants in urban gardens. In developed countries, the private gardens are filed with exotic ornamental plants because of their popularity (Shaw, Miller, & Wescott, 2017; Burghardt, Tallamy, & Gregory Shriver, 2009). A good example is in UK, where about 70% of the garden plants are exotic (Loram et al., 2008).

A number of factors influence planting choices and are therefore critical in the context of enhancing native plant biodiversity in urban areas (Shaw et al., 2017; van Heezik et al., 2013). These factors include social patterns (Caldicott, 1997), marketing influences (Shaw et al., 2017), environmental knowledge (Head & Muir, 2005), and economic conditions (Daniels & Kirkpatrick, 2006). The vegetation composition and structure are related to the householder socio-economic status, as well as their motivations and attitudes (van Heezik et al., 2013). Several studies have been made regarding environmental attitudes on gardens and planting (Head & Muir, 2004, 2005; Zagorski, Kirkpatrick, & Stratford, 2004; Lohr & Pearson-Mims, 2005). One study showed a strong relationship between gardeners' values and the species composition of their gardens with the gardeners who have pro-environmental views more likely to have more native plants in their gardens (Zagorski et al., 2004).

The objective of this chapter was to find out what environmental and social factors best explain planting patterns during the last 60 years in Christchurch. The Residential Red Zone of eastern Christchurch provided a unique opportunity to explore the composition of trees in private gardens across a large area of a New Zealand city. The trees of this area were surveyed for the Canterbury Earthquake Recovery Authority (CERA) prior to the demolition of houses. This tree map was analysed to assess the effects of environmental factors and social factors on the native and exotic tree species composition of private gardens. Specifically, the following questions were addressed.

1. What is the composition of residential garden trees in eastern Christchurch?
2. Do younger suburbs have higher native tree abundance and richness than older suburbs?
3. Does soil versatility have a positive effect on native tree abundance and richness?
4. Do social factors (human population density and economic deprivation) affect tree abundance and richness, and the proportion of native to exotic trees?
5. What environmental and social factors have driven people's planting choice during the last 60 years?

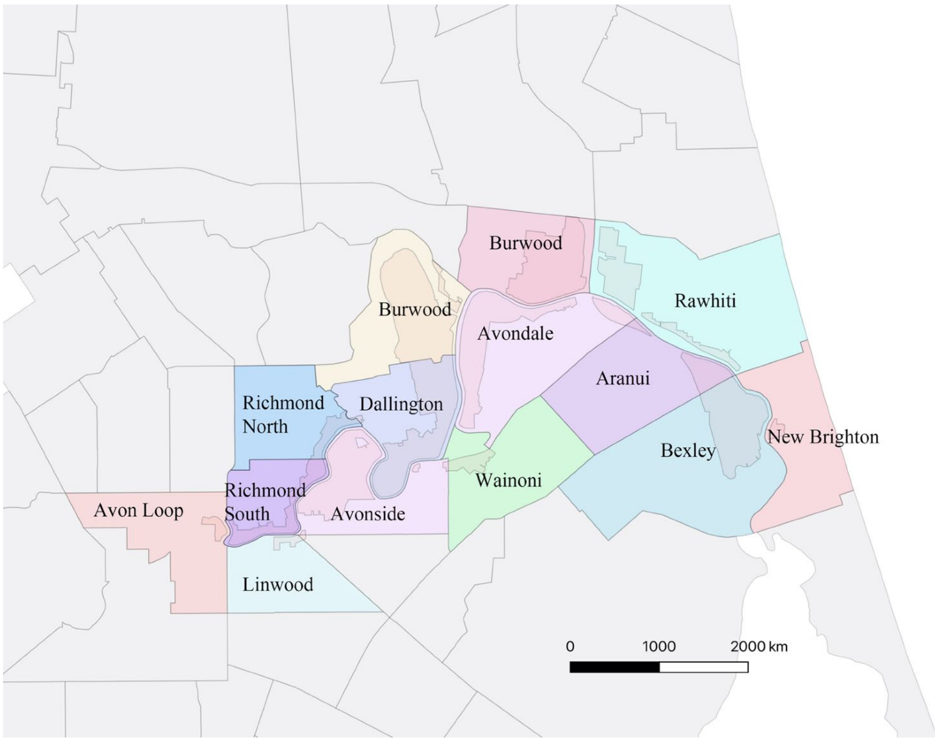
2.2 Methods

2.2.1 Study sites

2.2.1.1 Suburbs of the Residential Red Zone

The Residential Red Zone area covers several suburbs within eastern Christchurch city, New Zealand (see Section 1.7). In this research, 14 suburbs along the Avon River were chosen as the research area (Fig. 2.1). After the earthquakes, CERA contracted Treetech Specialist Treecare Ltd. to carry out a tree inventory in the residential area. All houses in the most damaged areas were directed to be removed by CERA and, as much as possible, the garden trees were saved. The remaining vegetation includes most of the larger ornamental trees planted in the private gardens, parks and reserves.

Figure 2.1: Map showing the 14 suburbs in the Residential Red Zone area.



2.2.2 Data sources

Tree map

Tree records of eastern Christchurch were obtained from the Residential Red Zone tree map provided by CERA. This map contains 27,698 mapped trees and large shrubs, identified to species (or, in some cases genus) from the ¿7,000 private properties acquired by CERA. The list was compiled prior to building demolition. As many of these large trees as possible were saved during house demolition.

Most of the trees (18,925, 97%) had recorded DBH data and were used. However, some values were unrealistically big, or small, indicating data entry errors. Unrealistic data values were removed and the data with DBH values between 5 cm and 2 m, inclusive, were used in the analysis.

Population data

The human population data came from the 2013 Census from Statistics NZ (`https://stats.govt.nz`). It was collected by delivering census forms to every person in New

Zealand. It includes a wide range of metrics, such as income, education, work, housing and personal information (Statistics New Zealand, 2013).

For this and other environmental factors, a grid layer of 100 m × 100 m was applied to standardise the scale for analyses. When more than one census meshblock overlapped a grid cell, an average value was calculated proportional to the area that each meshblock occupied in the grid square. Figure 2.2 shows the distribution of population densities across the study area.

Economic Deprivation data

Economic deprivation data came from the New Zealand Index of Socio-economic Deprivation for Individuals (NZiDep) which was made in 2013 (Atkinson, Salmond, & Crampton, 2014a). This index is applied to the same meshblocks as the population census data.

NZDep2013 deprivation scale is from 1 to 10 in which 1 is least deprived and 10 is most deprived. This scale divides the New Zealand population into tenths of the first principal component score of a multivariate analysis of deprivation (Atkinson, Salmond, & Crampton, 2014b). Figure 2.2 shows the distribution of deprivation values in the study area.

Soil data

Soil data was obtained from the soil map of Christchurch City from the NZ Soil Survery Report 16 held by Manaaki Whenua-Landcare Research (Webb & Trangmar, 2006).

We used the soil versatility rating as a measure of the overall quality of the soil conditions. The definition of versatility here is the ability of land to support the production and management of a range of crop plants on a sustained yield basis and is mainly assessed in terms of soil physical characteristics (Webb & Trangmar, 2006). It assumes that nutrient and soil moisture limitations are overcome by fertiliser application and irrigation (Webb & Trangmar, 2006).

Rating of land versatility for horticultural production is based on the system of Wilson and Giltrap (Wilson, 1984). The system uses six classes to classify the soil versatility in the range of horticultural crops with appropriate soil management techniques in accordance with soil conservation principles. The limitations to economically productive use increases from Class1 to Class 6 (Webb & Trangmar, 2006). This data set uses 5 soil versatility classes:

–Class 1 soils are very highly versatile

–Class 2 soils are highly versatile

–Class 3 soils are moderately versatile

–Class 4 soils are low versatile

–Class 5 soils are very low versatile

Figure 2.2 maps the range of soil versatility values across the study area.

2.2.3 Housing maximum age

To assess the range of garden ages in the study area, all the grids were assigned to the year in which houses were first built. This was extracted manually for every 100 m $\times$ 100 m grid square from the historical aerial photography layers of the Canterbury Maps website (`https://canterburymaps.govt.nz`). The time range available in the aerial photography are from 1940-2010 excluding 1950-1954. The earliest year in which more than three houses were established was used as the maximum garden age for each grid. This avoided the bias created by single old farm houses that were present in rural parts of the city prior to suburban house subdivisions being built.

Figure 2.2 maps the range of housing ages across the study area.

Figure 2.2: Maps showing human population, economic deprivation, soil versatility and maximum age of suburban housing in all 100 m× 100 m grid squares across the Residential Red Zone study area in eastern Christchurch. A: Estimated human population per grid cell, B: Mean economic deprivation per grid cell, C: Mean soil versatility per grid cell, D: Maximum age of suburban housing per grid cell.

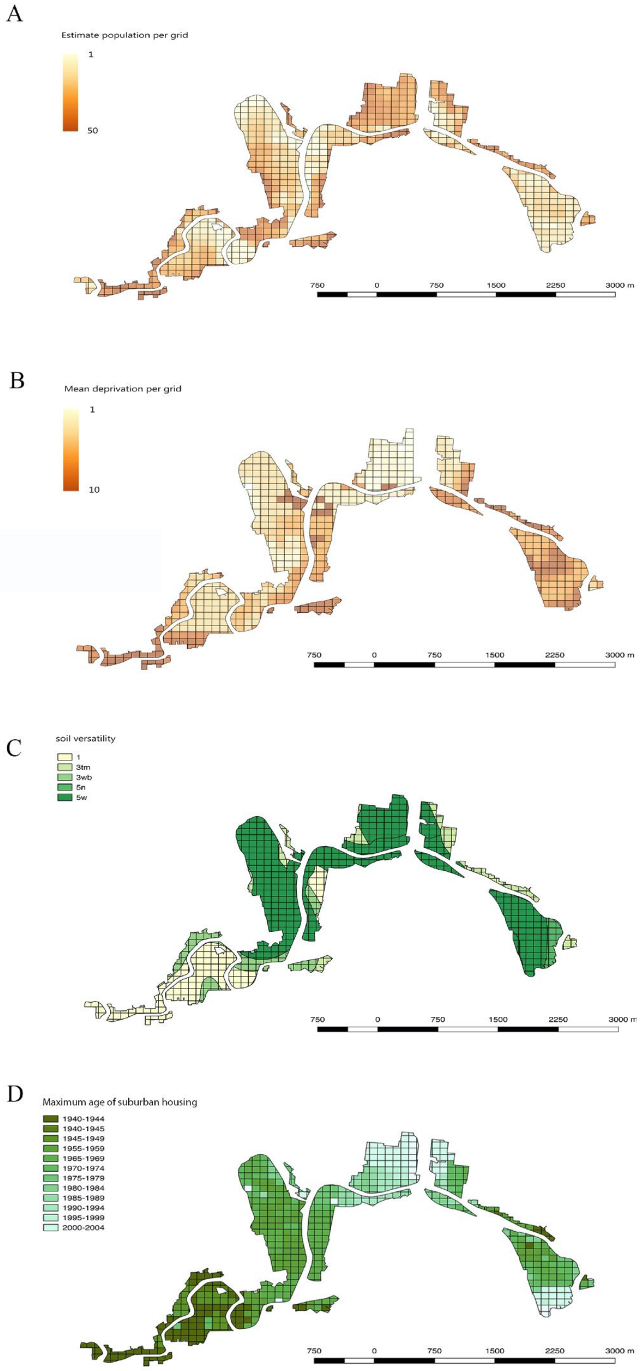
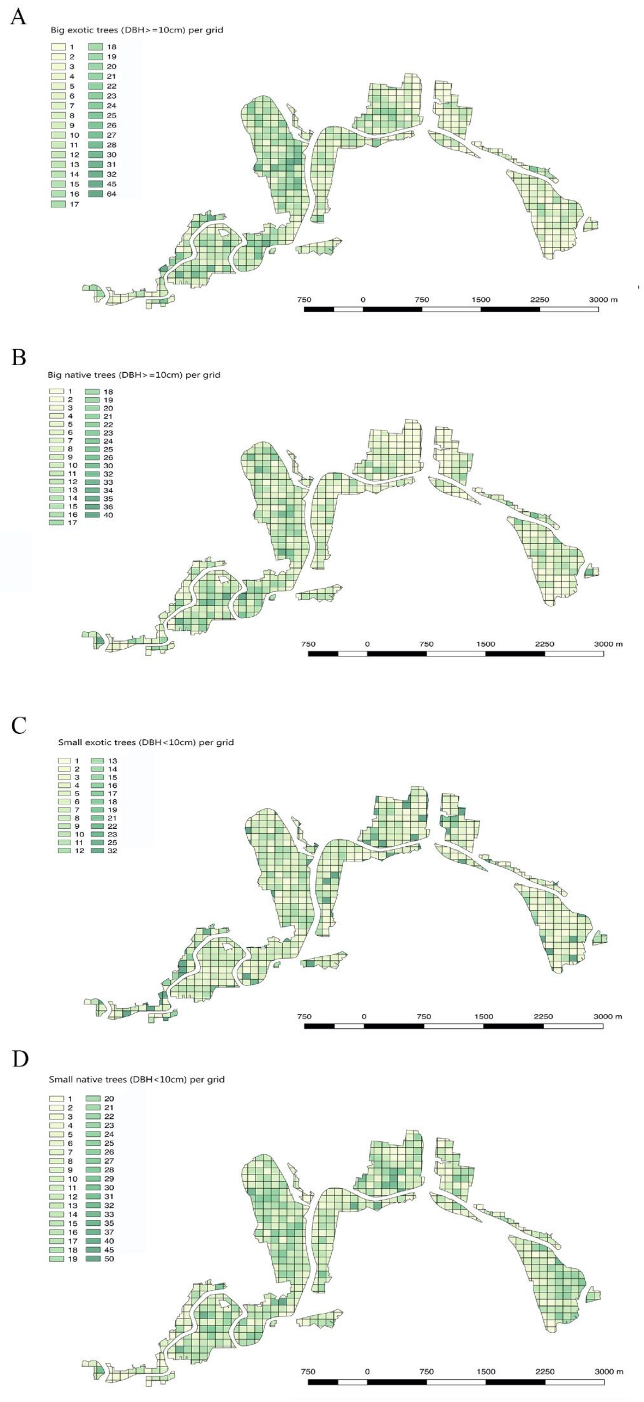


FIGURE 2.3: Maps showing big exotic trees, big native trees, small exotic trees and small native trees in Red Zone. A: Big exotic trees per grid map, B: Big native trees per grid map, C: Small exotic trees per grid map, D: Small native trees per grid map.



Plant nomenclature

Because the tree data were surveyed by different contractors, several common rules needed to be applied before analysing the data. Some taxa were recorded as both 'sp' and 'cultivar', for example, "*Prunus* sp" and "*Prunus* cultivar". For analysis, all 'cultivar' were changed to 'sp'. Another case is *Tilia* × *europaea*, where two names were applied for the same species, that is *Tilia* × *europaea* and *Tilia europaea*. In this case, both names were included as *Tilia* × *europaea* in the analysed data. In general, plant names were made consistent with Ngā Tipu o Aotearoa, the New Zealand Plant Names Database (`https://nzflora.landcareresearch.co.nz`).

A group of plants (766 individuals, 2.77%) which could not be identified by the surveyors were named "other sp" in the database. Because native plants in gardens could be reliably identified by local botanical contractors, it is assumed that the biostatus of unidentified plants was 'exotic' for the analysis.

To analyse the data, all unknown species recorded as 'Other sp' were conservatively treated as one species.

Plant biostatus

Plant biostatus describes whether a plant is native or exotic. New Zealand-wide plant biostatus data came from the New Zealand Organisms Register (`http://www.nzor.org.nz`) and has three biostatus categories, 'Endemic(Native)', 'Exotic', and 'Naturalised'. The tree map data was merged with the New Zealand plant biostatus database in R (R Core Team, 2016). All the trees categorised as 'Native' or 'Exotic', and then further split into local native categories: "Native to Christchurch", "Non-native to Christchurch", "Naturalised" and "Exotic"[1] (D. J. Mahon, 2007; Gatehouse, 2008).

For trees only identified to genus where the genus contained no native species, biostatus was "Exotic". Similarly, if the genus only contained native species, the biostatus was conservatively assigned to "Non-native to Christchurch". Where the genus contains species that are found in other countries as well as New Zealand, but in which 75% of the species known to be in NZ (wild or cultivated, based on the Plants Biosecurity Index (Version: 2.0.0, 2014) list of cultivated plants from what was then the Ministry of Agriculture and Forestry) are native, they were recorded as "Non-native to Christchurch". Otherwise they were recorded as "Exotic".

[1] Parts of the biostatus data came from Jon J. Sullivan and Colin D. Meurk's personal comments.

2.3 Analysis

The tree map database was analysed using R (R Core Team, 2017) to get the total tree list, native tree list and exotic tree list for each grid square. Each list includes the botanical names of the species, suburbs, and biostatus.

Package 'AICcmodavg' (Mazerolle, 2017) was used in R (R Core Team, 2017) to compare plausible generalised linear models (GLMs) involving the factors human population, economic deprivation, soil versatility, age of suburban housing, and total tree number (all measured per 100 m$\times$ 100 m grid square). This package includes functions to implement model selection and multi-model inferences based on Akaike's Information Criterion (AIC) and the second-order AIC (AICc) (Akaike, Petrov, & Csaki, 1973). When the difference between AICc values is ≥ 2, the model with smaller AICc value is considered the best model (Arnold, 2010). However, the compute results shows most of the AIC values are very close. So several models were compared in the analyses of both native and exotic big (DBH $\geq$ 10 cm) and small (DBH $<$ 10cm) trees, such as an all three interactions model, all two interactions model, two interactions without population model, two interactions without versatility model, two interactions without established year model, and a no interactions model.

Another package 'MuMIn' was applied in R (R Core Team, 2017) to average the best models. This package averages models based on model weights derived from AICc.

For plotting model predictions, near minimum and maximum values of each factor were selected. For human population, this was a minimum population was 0 and a maximum was 36 per grid cell. For tree number, it was 7–21 trees per grid cell, for soil versatility 3–5, and for economic deprivation, 3–7. The age of oldest suburban housing was plotted for 1940 and 2000.

2.4 Results

2.4.1 Species composition

There were 413 identified taxa (species or genus) recorded in the 14 suburbs of the Christchurch Residential Red Zone area. Exotic plants (naturalised species and exotics only cultivated) made up 80.6% (333) of taxa, while only about 11% were native to Christchurch (Fig.2.4). However, for the individual trees, over half of them were native, mostly trees native to Christchurch (Fig.2.4).

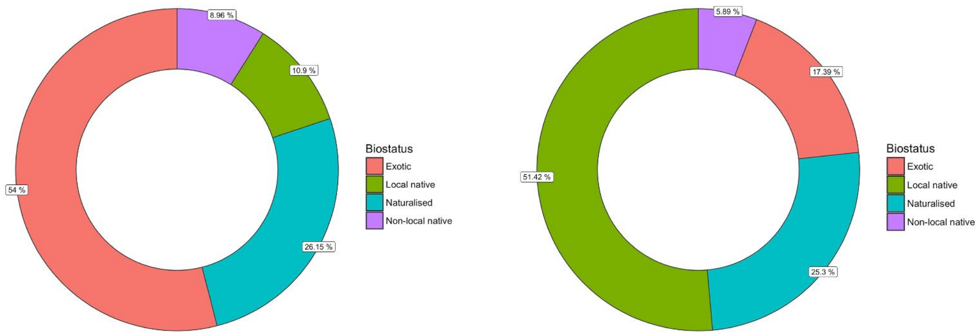


FIGURE 2.4: Percentage of native species and native individual trees with different biostatus in the Residential Red Zone area. Left: the percentage of native species. Right: the percentage of individual trees

Of the different suburbs, Avonloop had the most plants per hectare (21.8 trees per hectare, including 7.9 native trees and 13.9 exotic trees), New Brighton followed with 19.5 trees per hectare and third was Linwood with 17.8 trees per hectare. Dallington, Avondale, Bexley, Burwood and Travis suburbs, all of which contain areas of relatively recent housing subdivisions, had a low TPH which were all under 4 trees per hectare (Fig. 2.3, Table 2.1).

TABLE 2.1: Total tree numbers/tree densities of the residential red zone areas in different Christchurch suburbs. TPH is trees per hectare.

Suburb	**Area(ha)**	**Native/TPH**	**Exotic/TPH**	**Total/TPH**
Linwood	2.8	14/4.9	36/12.8	50/17.8
Richmond North	8.3	30/3.6	78/9.4	108/13
Aranui	14.4	40/2.8	68/4.7	108/7.5
Wainoni	7.8	30/3.8	65/8.3	95/12.1
Richmond South	18.7	38/2	92/4.9	130/7
Avonloop	3.7	29/7.9	51/13.9	80/21.8
Avonside	50	54/1	153/3	207/4.1
Dallington	62.2	62/1	158/2.5	220 3.5
New Brighton	2.7	20/7.3	33/12.1	53/19.5
Avondale	57.3	50/0.9	149/2.6	199/3.4
Rawhiti	35.3	45/1.3	115/3.3	160/4.5
Bexley	52.6	49/0.9	116/2.2	165/3.1
Burwood	71.3	67/0.9	195/2.7	262/3.7
Travis	56.1	56/1	127/2.3	183/3.3

2.4.2 Planting changes in Red Zone

2.4.2.1 DBH (Diameter at breast height) of exotic and native trees

Comparing the DBH distributions of all exotic and all native trees showed that trees with large DBH were more likely to be exotic (Figure2.5). The DBH of most native trees was under 50 cm. There were 13% more native trees than exotic trees for trees whose DBH was under 30 cm. In contrast, for trees with DBH over 30 cm, there were 7.8% more exotic than native ones. This suggests that native trees in these gardens smaller stature as adults than the exotics, and/or that a higher proportion them are of more recently planted (they are younger than the exotics).

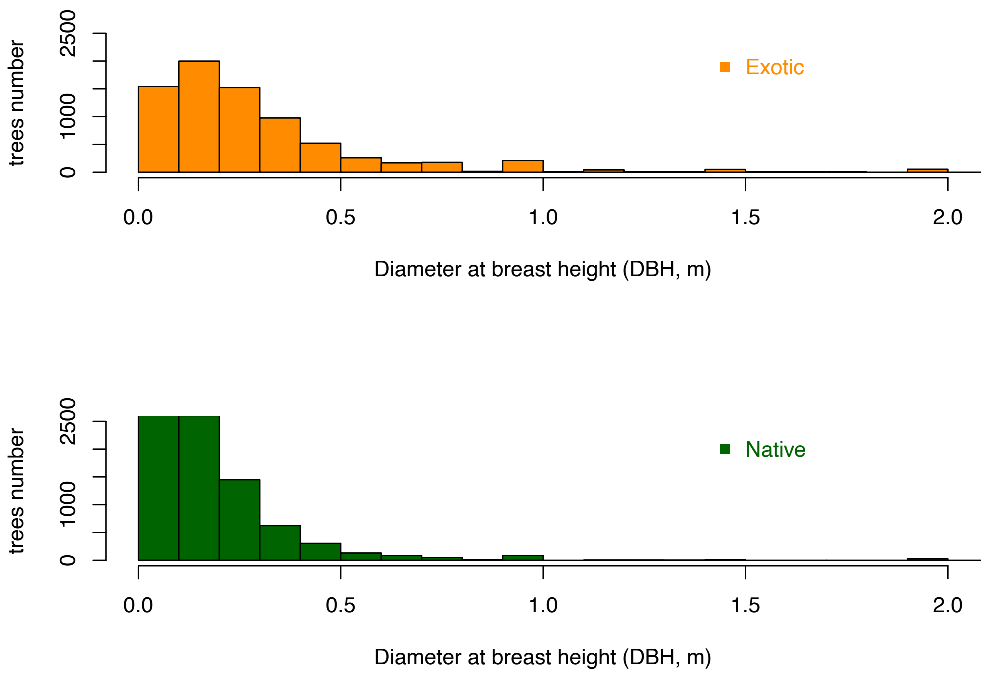


FIGURE 2.5: The DBH (Diameter at breast height) distribution of both exotic and native trees. Only DBH values between 5 cm and 2 m are included.

2.4.2.2 Changes of native species and native individual trees

Overall, the proportion of plant species that were native changed little regardless of housing age (Figure 2.6). In the oldest areas of the city, natives made up 55% of the garden tree species. In the most recently established suburbs, this was 60%, an insignificant difference.

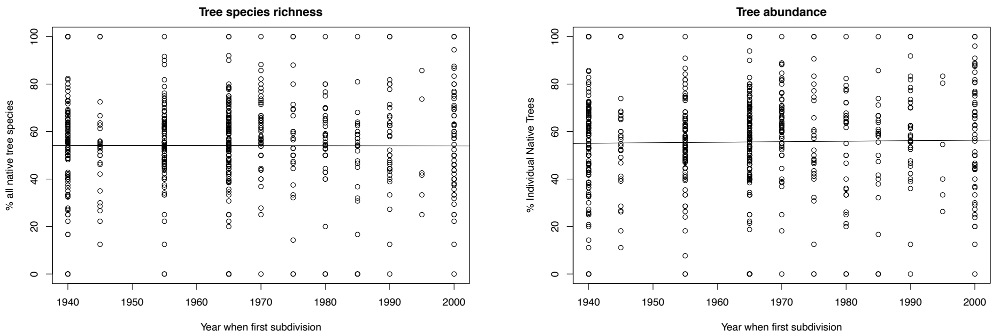


FIGURE 2.6: Percentage of tree species and individuals that were native, plotted against the decade in which each 100 m by 100 m grid square was first developed for suburban housing. The left graph shows the percentage of native species, and the right graph shows the percentage of all individual trees. Neither relationship is statistically significant.

The same result was seen for individual trees. Overall, there were no big changes from 1940s to 2000s. The percentage of individual native trees in 2000 was still was under 60%(Figure 2.6).

2.4.2.3 Different sizes of native trees

Greater differences were seen when I divided the trees into large trees (DBH≥10cm) and small trees (DBH<10cm). The percentage of big tree species that were native dropped from ca. 50% to ca. 40% from older to younger subdivisions (Figure 2.7). The percentage of big individual trees that were native showed a similar tend (Figure 2.7).

In comparison, for small trees, the percentage of both native species richness and individual trees increased about 10% in the past 60 years.

2.4.3 Environmental factors affecting garden tree composition

Tables 2.2 and 2.3 show the results of my generalised linear models assessed the combined effects of human population density, soil versatility, economic deprivation, housing age, and total tree density on the number of native and exotic trees per 100 m × 100 m grid square. Large trees (DBH≥10cm) and small trees (DBH<10cm) were analysed separately. All factors were included in some or all of the best models (within 2 AICc values of the best fitting model). The next sections explore the trends in more detail.

TABLE 2.2: Model Selection for big trees(DBH≥10cm). In this table, Pop=Estimated human population in grid in 2013, Estab=Subdivision establishment year (first year with housing development per grid), Dep=Mean economic deprivation from the New Zealand Index of Socioeconomic Deprivation for Individuals in 2013, Vers=Mean soil Versatility,Total=Total number of trees in grid. The table shows the model parameters for the 17 models compared. The binomial response variable was the total number of individual native and exotic trees.

No.	(Intercept)	Pop	Estab	Dep	Vers	Totaltrees	Pop:Estab	Pop:Dep	Pop:Vers	Pop:Total
1	63.3146279	0.02808691	-0.0330635	-7.7873588	0.30054103	-0.968914	–	–	–	-0.0013111
2	59.6948347	0.02548032	-0.031291	-7.2648	0.30478016	-0.9004362	–	–	–	-0.0011968
3	59.4627635	0.44493699	-0.0310599	-7.631372	0.28356915	-0.9366715	-0.000212	–	–	-0.0013735
4	56.8464619	0.02744566	-0.029757	-7.5553328	1.60723539	-0.9565062	–	–	–	-0.0013102
5	62.9109287	0.03348604	-0.0328476	-7.6798677	0.29838176	-0.9724266	–	–	-0.0014477	-0.0013471
6	63.6905248	0.03071133	-0.0332708	-7.828571	0.30413823	-0.9713035	–	-0.0004894	–	-0.0013032
7	56.8782613	0.3646779	-0.0298163	-7.1832044	0.29062445	-0.8802423	-0.0001724	–	–	-0.0012575
8	51.8774045	0.02461164	-0.027298	-6.9681225	1.85267825	-0.8826984	–	–	–	-0.0011909
9	57.670053	0.85906952	-0.0301877	-7.6915227	0.2855962	-0.9176291	-0.0004157	-0.0024982	–	-0.001393
10	59.4926156	0.02987725	-0.0311775	-7.1964537	0.30292342	-0.9054667	–	–	-0.0011551	-0.0012294
11	60.1188125	0.02889267	-0.0315293	-7.3088892	0.30959755	-0.9022065	–	-0.0006457	–	-0.0011841
12	51.2497022	0.48248401	-0.0268582	-7.3355547	1.86947287	-0.9185266	-0.0002315	–	–	-0.0013781
13	63.919885	0.05090548	-0.0334126	-7.7109065	0.31018642	-0.9865493	–	-0.0019697	-0.0032887	-0.0013608
14	59.5042158	0.43887688	-0.0310812	-7.6293316	0.28373586	-0.9372978	-0.0002088	–	-5.90E-05	-0.0013741
15	56.5268971	0.03278121	-0.0295843	-7.4521762	1.58924694	-0.9601076	–	–	-0.0014294	-0.0013457
16	57.3180217	0.02983878	-0.0300118	-7.5973633	1.58414067	-0.958934	–	-0.0004438	–	-0.0013031
17	55.2163206	0.7677857	-0.0290072	-7.2541496	0.29235047	-0.8634141	-0.0003707	-0.0024166	–	-0.0012799

Table 2.2 continued

Estab:Dep	Estab:Vers	Estab:Total	Dep:Vers	Dep:Total	Vers:Total	df	logLik	AICc	delta	weight
0.00405167	–	0.00050819	-0.0348023	–	-0.0058152	11	-1145.8603	2314.20942	0	0.18312268
0.00379817	–	0.00047652	-0.0340247	-0.0012121	-0.0061268	12	-1145.2897	2315.15818	0.94875194	0.11395235
0.00396457	–	0.00049184	-0.0319959	–	-0.0057741	12	-1145.5601	2315.69913	1.48970102	0.08694762
0.00393477	-0.0006665	0.00050195	-0.0355281	–	-0.0058983	12	-1145.7444	2316.0677	1.85827088	0.07231423
0.00399152	–	0.00051025	-0.0326501	–	-0.0059097	12	-1145.7464	2316.07167	1.86225019	0.07217049
0.0040761	–	0.00050936	-0.0356593	–	-0.0058008	12	-1145.8332	2316.24526	2.03583243	0.06617086
0.00374928	–	0.00046601	-0.0318126	-0.0011062	-0.0060663	13	-1145.0956	2316.86782	2.6583968	0.04847062
0.00364911	-0.0007894	0.00046771	-0.0348536	-0.0012634	-0.006238	13	-1145.1282	2316.93305	2.72362531	0.04691529
0.0040054	–	0.00048197	-0.0336692	–	-0.0056604	13	-1145.1332	2316.94306	2.73363346	0.04668111
0.00375863	–	0.00047919	-0.0323349	-0.0011719	-0.0061917	13	-1145.2178	2317.11223	2.90280164	0.04289501
0.00382541	–	0.00047742	-0.0351392	-0.0012359	-0.0061135	13	-1145.2428	2317.16213	2.95271015	0.04183784
0.00381475	-0.0008097	0.0004827	-0.0326195	–	-0.0058711	13	-1145.3917	2317.46004	3.25061448	0.03604792
0.00401432	–	0.00051759	-0.0333735	–	-0.0059721	13	-1145.4926	2317.66169	3.45226177	0.03259065
0.00396343	–	0.00049217	-0.0319503	–	-0.0057785	13	-1145.56	2317.79661	3.58718925	0.03046449
0.00387688	-0.0006584	0.00050405	-0.033395	–	-0.0059905	13	-1145.6334	2317.94336	3.73393967	0.02830919
0.00395927	-0.000653	0.00050314	-0.0362909	–	-0.0058835	13	-1145.7222	2318.12101	3.91158247	0.02590316
0.00379489	–	0.00045719	-0.0334352	-0.0010733	-0.0059469	14	-1144.6967	2318.17553	3.96610906	0.0252065

Table 2.3: Model Selection for small trees(DBH<10cm). In this table, Pop=Estimated human population in grid in 2013, Estab=Subdivision establishment year (first year with housing development per grid), Dep=Mean economic deprivation from the New Zealand Index of Socioeconomic Deprivation for Individuals in 2013, Vers=Mean soil Versatility,Total=Total number of trees in grid. The table shows the model parameters for the 22 models compared. The binomial response variable was the total number of individual native and exotic trees.

No.	(Intercept)	Pop	Estab	Dep	Vers	Totaltrees	Pop:Estab	Pop:Dep	Pop:Vers	Pop:Total
1	-3.2376608	-1.346392944	0.001872311	-	-	0.01960859	0.00068351	-	-	-
2	-3.4878808	-1.542969062	0.001951612	0.018514269	-	0.0201027	0.00078323	-	-	-
3	-3.1494251	-1.3175469	0.001848282	-	-	0.01717368	0.00066581	-	-	0.00039363
4	5.12075594	-1.454621433	-0.002423261	-1.958297259	-	0.02020707	0.00073961	-	-	-
5	-1.8896779	-1.382708055	0.001164929	-	0.01084209	0.01950888	0.00070221	-	-	-
6	-5.5521165	-1.326817545	0.003048823	-	-	0.13469083	0.00067365	-	-	-
7	-3.396077	-1.514950675	0.001925747	0.018621467	-	0.01763974	0.0007659	-	-	0.00039885
8	6.35253016	-1.40776984	-0.003022383	-2.212669156	-	0.01704185	0.00071198	-	-	0.00051575
9	-1.8350241	-1.598019809	0.001081605	0.019460301	0.013394377	0.02000364	0.0008115	-	-	-
10	-5.4442482	-1.523510437	0.002946841	0.01822069	-	0.11769886	0.00077342	-	-	-
11	-3.7705315	-1.432845511	0.0020831	0.023400897	-	0.02004648	0.00072958	-0.0008435	-	-
12	-3.4910325	-1.548381283	0.001937339	0.025237361	-	0.02171796	0.00078587	-	-	-
13	-1.9042599	-1.35174435	0.001194314	-	0.010032079	0.01714133	0.00068348	-	-	0.0003841
14	-5.5815456	-1.295402027	0.003084863	-	-	0.13773179	0.00065461	-	-	0.00039802
15	5.60665727	-1.490075227	-0.002687	-1.856179737	0.007506551	0.02014609	0.00075773	-	-	-
16	3.34464984	-1.438236105	-0.00151974	-1.945893849	-	0.10614133	0.00073135	-	-	-
17	5.0874383	-1.459663223	-0.002420001	-1.944926315	-	0.02159814	0.00074208	-	-	-
18	4.90915497	-1.412763712	-0.002320413	-1.933527552	-	0.02018438	0.00071921	-0.000326	-	-
19	-4.2482841	-1.363314439	0.002363377	-	0.011085533	0.13826579	0.00069245	-	-	-
20	0.07589424	-1.400522402	0.000155906	-	-0.42640131	0.01949183	0.00071136	-	-	-
21	-1.7282096	-1.410248393	0.001079999	-	0.012290886	0.01952362	0.00071673	-	-0.0002916	-
22	-1.8935623	-1.382009772	0.001165488	-	0.011553222	0.0196436	0.00070186	-	-	-

Table 2.3 continued

Estab:Dep	Estab:Vers	Estab:Total	Dep:Vers	Dep:Total	Vers:Total	df	logLik	AICc	delta	weight
-	-	-	-	-	-	5	-1102.8224	2215.74652	0	0.15588725
-	-	-	-	-	-	6	-1102.3029	2216.7485	1.00198142	0.09445677
-	-	-	-	-	-	6	-1102.541	2217.22467	1.47815179	0.07444474
0.00100431	-	-	-	-	-	7	-1101.5655	2217.32157	1.57505205	0.07092386
-	-	-	-	-	-	6	-1102.727	2217.59655	1.85002484	0.06181342
-	-	-5.85E-05	-	-	-	6	-1102.7596	2217.66177	1.91525134	0.05983001
-	-	-	-	-	-	7	-1102.0155	2218.22143	2.47490957	0.04522627
0.0011336	-	-	-	-	-	8	-1101.1009	2218.4471	2.70057824	0.04040057
-	-	-	-	-	-	7	-1102.1585	2218.50748	2.76096051	0.03919906
-	-	-4.96E-05	-	-	-	7	-1102.2579	2218.70623	2.95971226	0.03549092
-	-	-	-	-	-	7	-1102.2669	2218.72432	2.9777969	0.03517145
-	-	-	-	-0.000347383	-	7	-1102.2801	2218.75068	3.00415966	0.03471088
-	-	-	-	-	-	7	-1102.4594	2219.10923	3.36270637	0.02901404
-	-	-6.13E-05	-	-	-	7	-1102.4722	2219.13485	3.38832464	0.02864476
0.00095269	-	-	-	-	-	8	-1101.5222	2219.28979	3.54326602	0.02650942
0.00099787	-	-4.37E-05	-	-	-	8	-1101.5304	2219.30621	3.55968552	0.02629267
0.00100046	-	-	-	-0.000299654	-	8	-1101.5485	2219.34224	3.5957202	0.02582319
0.00099268	-	-	-	-	-	8	-1101.5603	2219.36593	3.61940972	0.02551912
-	-	-6.04E-05	-	-	-	7	-1102.6599	2219.51032	3.76380208	0.02374168
-	0.00022419	-	-	-	-	7	-1102.7162	2219.62293	3.87640911	0.02244187
-	-	-	-	-	-	7	-1102.7247	2219.63987	3.8933511	0.02225257
-	-	-	-	-	-3.51E-05	7	-1102.7268	2219.64411	3.89758769	0.02220548

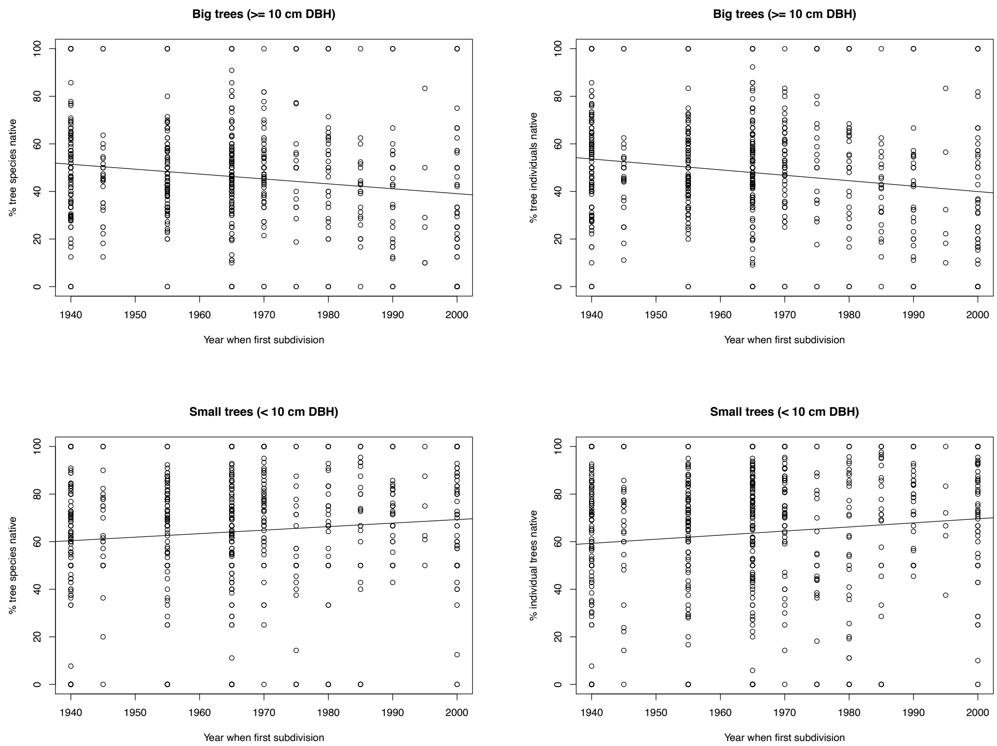


FIGURE 2.7: Percentage of different sized native trees and native individual trees

Proportion of small native trees

The proportion of small native trees increased from old to new suburbs in both low and high deprivation areas (Figure 2.8). For small native trees, human population density is an important factor which increased their percentage especially in younger subdivisions. For economic deprivation, at least in the last 40 years, high deprivation areas had a higher proportion of small native trees than low deprivation areas.

When the value for resident human population density is 0, the sites can be treated as public parks or reserves. These areas had the highest percentage of small native trees compared with areas with higher population density (meaning more private gardens). The percentage of small native trees in these areas of public parks/reserves was lower in recently established low deprivation areas than recent higher deprivation areas.

Proportion of big native trees

In the oldest areas of housing, higher human population had a higher proportion of big native trees than low population areas (Figure 2.9). As population density increased, the proportion of big native trees in high deprivation areas started to decrease. As soil

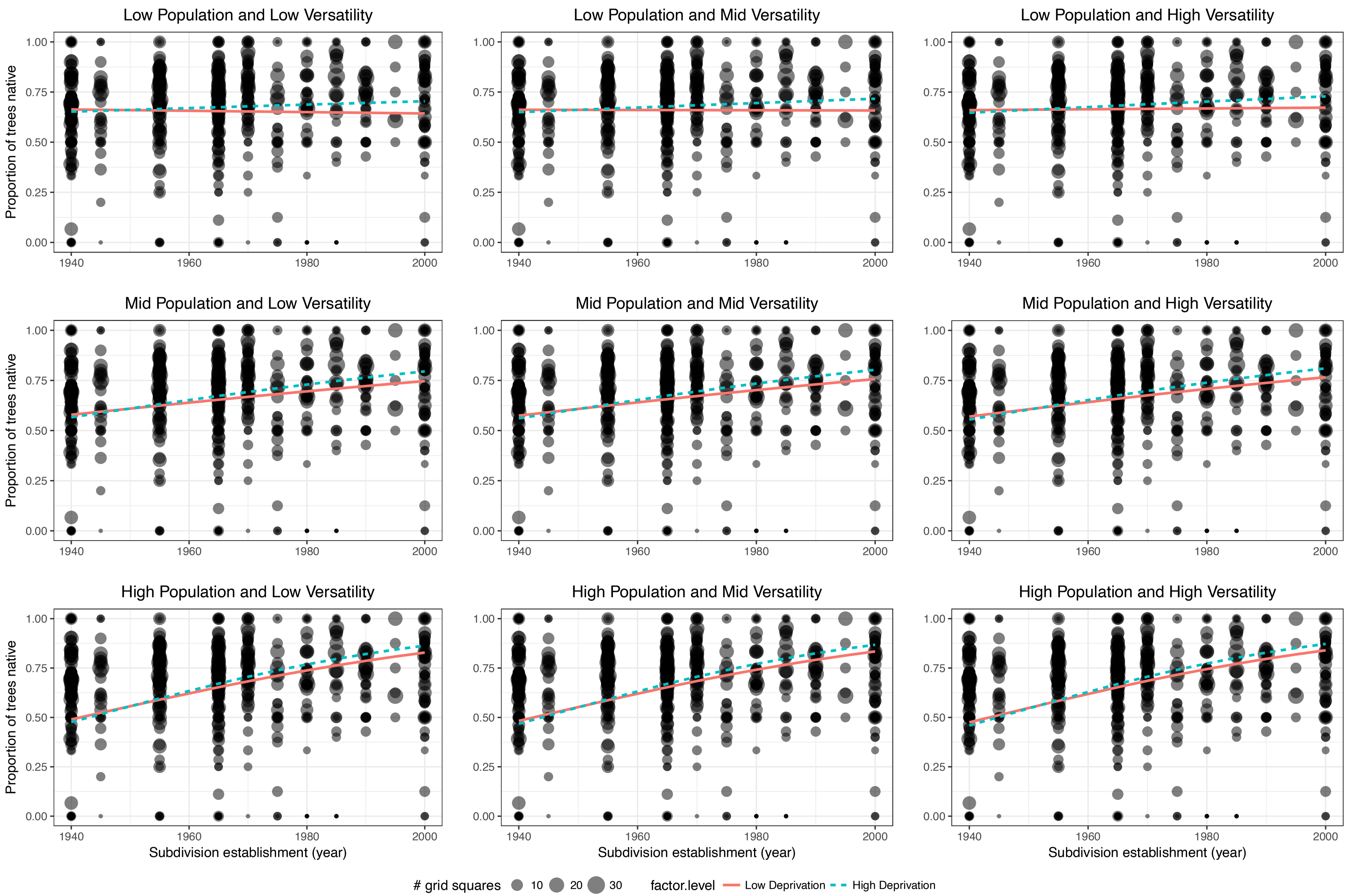


FIGURE 2.8: The generalised linear models' predicted effects of different levels of factors on the proportion of small native trees (DBH<10cm) in 100 m × 100 m grid squares. Total trees was of less interest than other four factors so it was set as an average value which is 14 throughout.

versatility increased, the proportion of big native trees also increased in low deprivation area.

Overall there was a decline in the proportion of big trees that were native in new subdivisions. This decline was most pronounced in low deprivation areas.

2.5 Discussion

From our results we can see the percentage of native tree species in surveying Red Zone private garden areas is currently 55% during the last 60 years. Elsewhere there is research showing 32% of native species in gardens in Belfast, 29% in Cardiff, 30% in Edinburgh, 29% in Leicester and 29% in Oxford in UK (Loram et al., 2008).

In Christchurch, the proportion of big trees that were native is less in recent housing subdivisions than older areas of housing. One possible reason could be that there are many more choices of exotic garden plants and nurseries are in the business of trying to find new plant fashions that attract buyers. Most native tree species sold in plant nurseries are seedlings or saplings, and, compared with native trees species, exotic trees tend to be bigger and more expensive, so younger housing aged and wealthier areas would be expected to have more big exotic trees initially planted.

For big native trees, both human population and soil versatility had significantly different effects in high and low deprivation areas. The difference between high and low deprivation areas in the percentage of big trees that were native was greatest in poor versatility soils and lower population density. In most soil conditions and human population densities, there were proportionately more big native trees in richer areas of the oldest housing. Interestingly, this reversed in more recently established suburbs, with proportionately more big native trees in poorer areas.

Several studies have shown a positive association between wealth of suburbs and vegetation biodiversity, in USA (Hope et al., 2003; Kinzig, Warren, Martin, Hope, & Katti, 2005) and in Australia (Luck, Smallbone, & O'Brien, 2009). Here I document the association between wealth of suburbs and native biodiversity. In contrast to the big native trees, more small native trees are growing in Christchurch's urban areas, especially in areas of higher economic deprivation. Human population (density) was a key factor for small native trees. In old sites, areas with high human density had fewer small native trees while younger sites had more small native trees. Soil conditions did not affect small native trees. Compared with exotic tree species, native tree seedlings can grow in different types of soil. As mentioned by Stewart et al., (2004), recently people in Christchurch

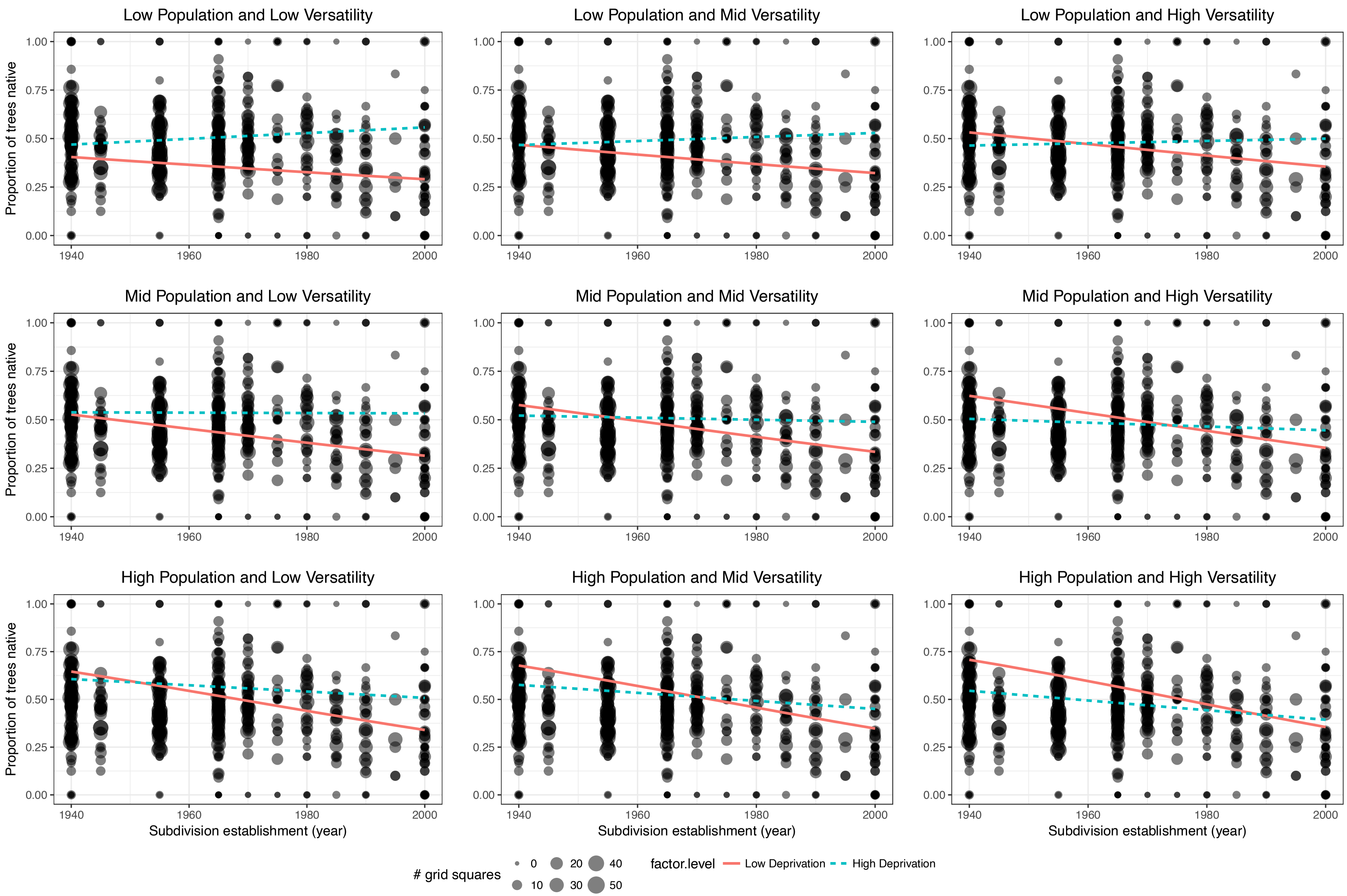


FIGURE 2.9: The generalised linear models' predicted effects of different levels of factors on the proportion of big native trees (DBH≥10cm) in 100 m × 100 m grid squares. Total trees was of less interest than other four factors so it was set as an average value which is 14 throughout.

have realised the values of native species. More small native trees or seedlings were kept or planted in the gardens, in areas of a high population density.

We set the minimum value of the population to be 0 when we graphed the model predictions and these can interpreted as public parks or reserves. In the oldest areas of Christchurch, public spaces in high deprivation areas had more big native trees than low deprivation areas, while private gardens in high deprivation areas had fewer big native trees than low deprivation areas. Public spaces had more small native trees than private gardens in older suburbs. However, in newer suburbs, private gardens had more small native trees than public spaces.

Exotic trees make up a big proportion of the forest canopies of Christchurch's urban public green spaces. Big native tree species were not common in public green space (Stewart, Meurk, et al., 2009). In my research, the proportion of native big trees was under 37.5% in recent housing in low deprivation areas, but for recent housing in high deprivation areas that figure was around 50%. In Auckland city, the percentage of garden trees that are native is around 25% (McDonnell, Hahs, & Breuste, 2009), surprisingly much lower than Christchurch.

Generally speaking, the proportion of small trees that are native increases in younger suburbs. That suggest that more people, both gardeners and landscape architects, are realising the importance of the native trees in our urban ecosystem (or that they require generally less effort to maintain). Doody and colleagues (Doody et al., 2010) found 54% of surveyed Christchurch residents in the suburb of Riccarton would like to plant native species which can be found in local urban forest in their gardens and van Heezik et al., (2013) found in Dunedin about 40% of garden holders in their research have a preference for planting native species in their gardens. However, in Australia, almost 90% of the respondents indicated they would like to plant native plants in their garden in the future, and the most preferred garden type was a lawn with native plants from the six choices (Shaw et al., 2017). That brought another question: why they don't plant more native plants in their garden currently (Shaw et al., 2017)? It was found in the research of relationship between attitudes and behaviors that having an intention to plant native plants and planting native plants is not a straight-forward relationship (Kollmuss & Agyeman, 2002).

Economic deprivation was an important social factor correlated with native garden trees but in complex ways. I expected exotic trees to be more abundant in the wealthier areas. One reason is that tree species sold in plant nurseries are expensive, and a diversity of garden plants is not affordable for poorer people (Bigirimana, Bogaert, De Cannière, Bigendako, & Parmentier, 2012). However, the reality turns out to be different. It can be found in the prediction of big and small native tree proportions. In recently established areas, wealthier areas typically have more native tree species than poor areas, whereas in older areas of the city, wealthier areas have similar or often fewer native trees than poorer areas. This may signal a changing attitude towards native trees in private gardens, with wealthier people now being more likely than in the past to invest in native trees when establishing their gardens.

2.6 Conclusion

Private gardens are an important kind of urban green space, holding much of the city's tree diversity. After European settlers founded Christchurch, almost all native vegetation was cleared and new exotic tree species were imported. Now nearly half of the city's trees are exotic species including most of the big trees.

My results are consistent with an increasing realisation among Christchurch citizens of the values of native tree species, and they are planting more native trees in their gardens. However, even if there are more native species in urban gardens than before, the percentage of the native tree species remains low. The number and diversity of exotic trees being planted both by the public and the council has increased along with native trees planting. About a quarter of trees in Christchurch gardens are exotic species that have naturalised in New Zealand and are capable of regenerating wild in the city as woody weeds.

Generally, wealthy people's gardens had more native trees than poorer people's garden in newer suburbs. This is however different in oldest suburbs, where poorer peoples' gardens had more native big trees than wealthy peoples' gardens. Wealthy people can accept and afford planting more native trees in their gardens. This is an encouraging sign that Christchurch residents are placing more value on having native trees in their neighbourhoods.

Chapter 3

Effects of suburb age, planting choices, soil and hydrology on woody species regeneration in urban gardens

3.1 Introduction

Urban environments are complicated habitats for wild plants, and urbanisation is one driver of species declines and extinctions (Duncan & Young, 2000; McKinney, 2008; Faeth, Bang, & Saari, 2011; McKinney, 2002; Marzluff, 2001; Seabloom, Dobson, & Stoms, 2002). As a consequence of land modifications and human activities, cities are losing native species (Drayton & Primack, 1996; Thompson & Jones, 1999; Duncan & Young, 2000; DeCandido, 2004; Whelan, Roberts, England, & Ayre, 2006; N. S. Williams et al., 2009) while gaining exotic species, both planted and naturalised. For example, about 70% of the garden flora (1056 species) in UK are exotic (Loram et al., 2008). On the other hand, some native species are taking advantage of the suburban and urban fringe habitats (McKinney, 2006; Kowarik, 2008; Kearns & Oliveras, 2009). Exotic species which are pre-adapted to disturbed or stressed urban environments can establish and spread easily while native forest species often decline and are found only on the fringes of cities (Kowarik, 2008; Pyšek, 1998; Olden & Poff, 2003; Kühn, Brandl, & Klotz, 2004). The net result is that internationally, urban biota is becoming increasingly diverse and abundant (McKinney, 2006, 2008) but biodiversity is decreased by homogenisation and species extinctions.

There is furthermore an increasing interest in planting native species in urban areas worldwide (Seidlich, 1997; Town & Association, 2004; Ignatieva, Meurk, van Roon, Simcock, & Stewart, 2008). For conservation work, a challenge now is to bring the missing native plants back into urban areas (Sawyer, 2005; Davies & Christie, 2001; Meurk, 2003; Doody, 2008; Behrens, 2011). Native plants can always be planted in gardens and parks but it is more difficult to establish sustainable, genetically diverse wild populations of these species in urban areas. Which planted species go on to form wild populations in urban environments, and in what conditions? Little is known about the factors influencing wild woody seedlings in urban environments.

Two main factors drive increasing exotic plant richness in urban areas. One is the large scale importation and propagation of exotic plants and the other is the high disturbance human environments that often better suite some exotic plants over local natives (Lozon & MacIsaac, 1997; D'antonio, Dudley, & Mack, 1999; Mack & Lonsdale, 2001; Parendes & Jones, 2000; D'antonio & Meyerson, 2002). There are several reasons that humans bring in exotic plants, such as for food, shelter, and beauty. As an immigrant country, New Zealand's urban flora is becoming homogenised with species from around the world (Thomson, 1922b; Cockayne, 1967; Allen & Lee, 2006). Settlers from other countries, especially from Europe, imported their familiar landscape designs like lawns, woodlands, shrubberies, hedges and flowerbeds, as well as the plants from their home countries that best suited those designs. After several centuries of land clearances and plants importation, there are few remaining 100% native patches within New Zealand cities and the forests and woodlands in cities are typically a mix of native and exotic species which came from around the world (Esler, 2004; Stewart et al., 2004).

Researchers have shown that urban forests have a higher species richness than the surrounding countryside (Stewart et al., 2004; Bertin, Manner, Larrow, Cantwell, & Berstene, 2005; Alvey, 2006). However, this high species richness is contributed by numerous exotic species while native tree species are not regularly planted (Clemants & Moore, 2003; Hitchmough, 2011; Schlaepfer, Sax, & Olden, 2012). Exotic species typically dominate the plant species richness in New Zealand urban areas (Given & Colin, 2000) and compete with indigenous species. A large number of introduced species have naturalised, and naturalisations continue. About 80% of naturalised herbaceous flora of Auckland have been introduced deliberately and most of them are garden escapes (A. Esler, 1988; Given & Colin, 2000).

In Christchurch, there has been extensive land modification with human settlement and, with very few exceptions (Molloy, 1995), all of the original native vegetation has been cleared. However, during the decade from 1994–2004, Stewart et al., 2004 estimated that over a million native plants had been propagated and planted in Christchurch, and if anything the subsequent rate of native plantings has increased. About 75% of the planted natives had survived (Stewart et al., 2004). Even in some gardens can be found native podocarp trees that were planted around a century ago (Stewart et al., 2004). These native plants can now be seed sources from which seeds dispersal and regeneration can happen in the city (Stewart et al., 2004). Therefore, there is an increasing seed rain from native plants providing the potential for a rolling succession of native tree species throughout the city.

Natural regeneration of tree species in forests plays an important role the maintenance of biodiversity, including in urban areas (Moore & Allen, 1999; A. Cameron, Mason, & Malcolm, 2001). However, many tree species struggle to regenerate, or cannot, in urban environments (Fredericksen, 1999). In New Zealand, most native trees do not form long-lived seeds bank, and most of the seeds waiting in urban soil are exotic species (Stewart et al., 2004). Also, most of the native tree species need shade to help them establish (Meurk, 1995).

Many factors have been shown to affect the natural regeneration of tree species in forests, such as site age, canopy density and height (Oberhauser, 1997), site elevation (Germino, Smith, & Resor, 2002; Cierjacks, Rühr, Wesche, & Hensen, 2008), landform (Eilu & Obua, 2005), land aspect (Masaki et al., 2004), and seed dispersal distances (Parrotta, 1995; Utsugi et al., 2006). In particular, understory light is an important environmental factor influencing the growth and survival of many tree species (Chazdon, Pearcy, Lee, & Fetcher, 1996; Whitmore, 1996) and understory light levels have a big impact on forest regeneration (Nicotra, Chazdon, & Iriarte, 1999). Tree leaf litter depth (and composition) is also an important factor which alters the composition of herbaceous species of forest floors and affects tree seedling establishment (Sydes & Grime, 1981). Soil moisture is considered to be another key factor for the seedlings regeneration (Ceccon, Sánchez, & Campo, 2004). Soil moisture and soil nutrients can interact to affect tree seedling regeneration (Ceccon, Huante, & Rincón, 2006). In urban ecosystems especially, human activities can alter natural regeneration (Ceccon et al., 2006) such as by planting, mowing and weeding. Moreover, habitat edges can have effects on urban forest regeneration (Hamberg, Lehvävirta, & Kotze, 2009; Hamberg, Lehvävirta, Minna, Rita, & Kotze, 2008).

Most research on forest regeneration has been on seedling regeneration in natural and rural habitats, like on tropical pasture land (Elgar, Freebody, Pohlman, Shoo, & Catterall, 2014), in mountain areas (Yu et al., 2013), in tropical forests (Swaine, 1996; Nicotra et al., 1999) and in oak-pine forest (Collins & Good, 1987). Much less research has been done on the seedling regeneration in urban area (Lehvävirta & Rita, 2002; Nowak, 2012; Zipperer, 2002; Lehvävirta et al., 2004).

In my research, I wanted to know which wild tree seedlings could be found in the Christchurch Residential Red Zone area and what environmental and historical factors determine seedling establishment. Specifically, I addressed the following questions:

1. What tree species are naturally regenerating in the Red Zone?
2. How important is the presence of planted adult trees (in abandoned Red Zone gardens) for explaining the diversity and abundance of wild tree seedlings?
3. How does species origin (biostatus) affect seedling regeneration? Are local native species most likely to regenerate?
4. What kind of factors affect seedling regeneration in the Red Zone?

This knowledge can inform future Red Zone management by identifying which tree seedlings will naturally regenerate, and where, which species will need assistance in establishing, and which exotic trees should be removed.

3.2 Methods

3.2.1 Study Sites

The study sites include six suburbs (Figure 4.2) with different ages within the Christchurch Residential Red Zone (now called the Ōtākaro Avon River Corridor) (Chapter 2). This area of the city had housing demolished after the earthquakes of 2010 and 2011, leaving most of the garden trees and shrubs, which were mapped out prior to demolition (Chapter 2). In 2017, I surveyed the wild tree seedlings naturally regenerating in this area.

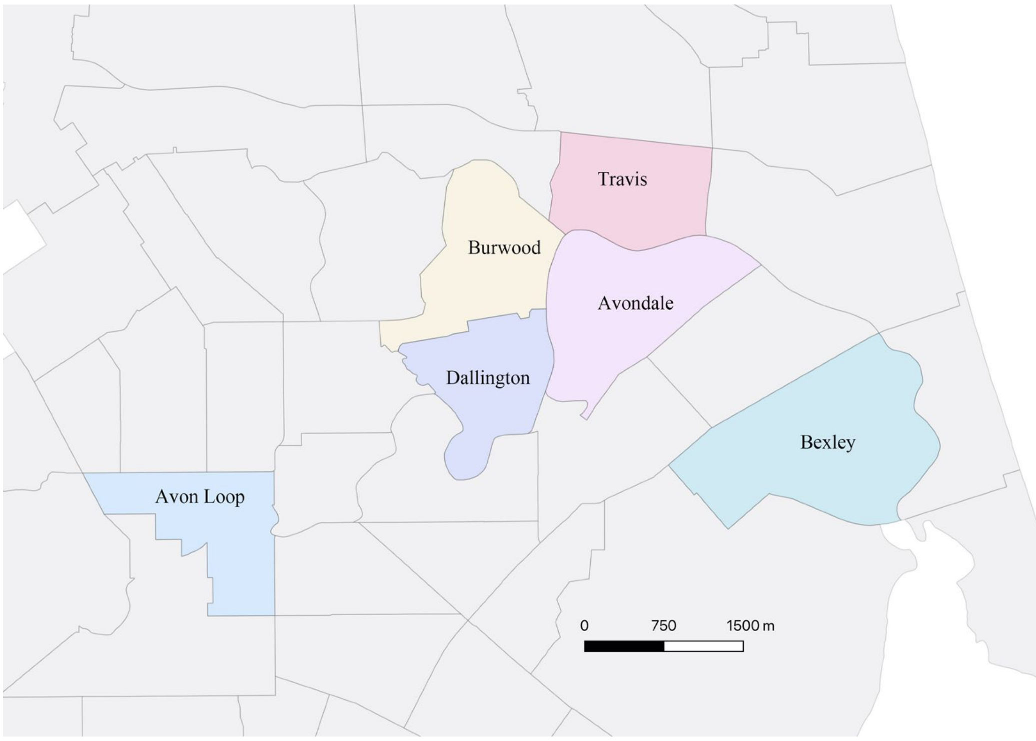


FIGURE 3.1: The location of the six suburbs in the Residential Red Zone area where tree seedling regeneration was sampled.

TABLE 3.1: Patch and quadrant numbers for the six suburbs.

Site name	Avondale	Avonhead	Bexley	Burwood	Dallington	Horseshoe
Patch number	14	12	9	31	62	20
Quardrat number	16	19	12	39	74	32

3.2.2 Data collection

Suburb Survey

A walk-in survey was done in six suburbs represented across the Residential Red Zone. These were walked transects located in stratified random locations in each suburb. In order to make the survey more time efficient, I only stopped to survey patches that included five or more seedlings of each of at least two tree species. Anything that did not match these minimum requirements was not considered a good microhabitat for woody seedling establishment. Most of the Residential Red Zone land did not meet these conditions since it had been converted to and maintained as mown grass and/or was being kept tidy with frequent applications of herbicide. Such sites therefore could not reflect a relationship between seedling establishment and spatial or environmental conditions. 1 m $\times$ 1 m quadrats were placed randomly in the suitable microhabitats and surveyed for their tree seedling composition.

For each quadrat I recorded the canopy cover (%) and GPS location. For each tree species present in the quadrat, I recorded the size categories present (S= seedling which is taller than 15 cm and under 1.4 m, SS= small seedlings which is under 15cm), the count category for each size category, the presence of adult(s) of the same species in the patch, and the presence on any reproductive structures on these adults (Flower buds/Flowers/Old Flowers/Fruit/Old Fruit). Also, the presence of seedlings of any additional species that were not present in the quadrats but in a patch were also recorded, along with the size categories present, presence of adult(s) of that species in the patch, and the any reproductive structures on those adults. As above, the proximity of adult trees of each species outside of the patch were calculated from the adult tree data from Residential Red Zone tree database.

The soil versatility data came from the soil map of Christchurch City from the New Zealand Soil Survery Report 16 held by Landcare Research New Zealand Ltd (Webb & Trangmar, 2006). For details see 2.2.2. The soil moisture data came from the Soil Moisture Spatial Data by Landcare Research (`http://ecan.maps.arcgis.com`). Canopy cover (%) and ground cover (%) were calculated from photos by a fish eye lens, taken facing north at 1 meter height. The canopy and ground photos were input into ImageJ (ImageJ version 1.50i, 2017) to get the sky area and ground vegetation area, then subtracted the value of sky area from the value of the lens view to get the % canopy area.

Species accumulation curves were used to estimate the proportion of all regenerating tree species detected in my surveys of each suburb. This was done with the specaccum and specpool functions of the vegan package in R (Oksanen et al., 2019), to graph the species accumulation curves and estimate the total species pool, respectively. The Chao estimate (Chao, 1987), as calculated by specpool, was used to estimate the likely total number of tree species regenerating in a suburb. I continued to sample in each suburb until at least 80% of regenerating tree species have been detected. The species accumulation curves at the completion of my survey are shown in Figures 3.3 and 3.4.

FIGURE 3.2: An example of the survey patch in Red Zone area. The trees and shrubs are the remains of suburban gardens, with the houses and other build structures demolished and removed. Bare areas were graded, sown with grass seed, and maintained as lawn with regular mowing. The fringes of the woody patches have been regularly maintained with sprayed herbicide. The only opportunities for tree seedling regeneration are under larger patches of woody plants away from the mowing and spraying.

Plants biostatus

All species found were assigned to one of four types of plant biostatus: 'Native to Christchurch', 'Non-native to Christchurch' (native to New Zealand but not Christchurch), 'Naturalised' (wild plants of species introduced to New Zealand), or 'Unnaturalised' (planted species introduced to New Zealand and not wild). New Zealand plant biostatus data came from the New Zealand Organisms Register (`http://www.nzor.org.nz`), with the species native to Christchurch additionally categorised by Jon Sullivan and Colin Meurk. The tree map data and seedling data were merged with the plant biostatus database in R (R Core Team, 2016).

Housing maximum age

The suburb age data was gotten for every 100 m $\times$ 100 m grid from the historical aerial photography layers of the Canterbury Maps website (`https://canterburymaps.govt.nz`). For details see 2.2.3.

3.3 Data analysis

All seedling data was combined with biostatus, first year of house, soil data in R (R version 3.3.3,2017) (R Core Team, 2017). A full model was built with all factors biostatus, canopy cover, first year of house, soil versatility, soil moisture, nearest conspecific adults as well as two possible plausible combinations: Biotatus-canopy and first year of house-nearest conspecific adults. However the result showed that the full model was to big and to slow to run. So package 'AICcmodavg' (Mazerolle, 2017) was used in R to get a subset model with all models within 4 AICc and these models were averaged to get one best average model. In order to make the factors influence and interactions clearer, another package 'MuMIn' was applied in R (R Core Team, 2017) to get the top model as the best model for analysing. The best model contained species and patch as random effects. The effects of canopy cover and biostatus were explored further in generalised linear models explaining the species richness of native and exotic seedlings in quadrats.

Table 3.2: Component models for the best average model.

	df	logLik	AICc	delta	weight
1/3/4/5/10	8	-999.08	2014.19	0.00	0.24
1/2/3/4/5/10	9	-998.86	2015.77	1.57	0.11
1/3/4/5/8/10	9	-998.92	2015.90	1.70	0.10
1/3/4/10	7	-1000.99	2016.02	1.82	0.10
1/3/4/5/7/10	10	-998.01	2016.07	1.87	0.09
3/4/5/10	7	-1001.41	2016.86	2.66	0.06
1/2/3/4/5/9/10	10	-998.58	2017.22	3.03	0.05
1/2/3/4/5/8/10	10	-998.71	2017.48	3.28	0.05
1/2/3/4/10	8	-1000.78	2017.60	3.40	0.04
1/3/4/5/6/10	10	-998.82	2017.70	3.51	0.04
1/3/4/5/7/8/10	11	-997.86	2017.78	3.59	0.04
1/2/3/4/5/7/10	11	-997.88	2017.83	3.63	0.04
1/3/4/7/10	9	-999.93	2017.90	3.71	0.04

Table 3.3: Code terms of the best average model.

Code	Terms
1	I(scale(Canopy)ˆ2)
2	scale(Canopy)
3	scale(first.year.with.houses)
4	scale(nearest.adult.meters)
5	Biostatus
6	moisture
7	versatility
8	I(scale(Canopy)ˆ2):Biostatus
9	scale(Canopy):Biostatus
10	scale(first.year.with.houses):scale(nearest.adult.meters)

Only native and exotic these two main types of biostatus were analysed in this research. We also did a further anaylsis with four types of biostatus: Christchurch native, Non-Christchurch native, naturalised and unnaturalised in the plants biostatus section.

Package 'multcomp' was applied in R (R version 3.3.3,2017) to do the pos-hoc test to compare which biostatus species will be more abundant in the Residential Red Zone area.

3.4 Results

3.4.1 Summary of model selection

TABLE 3.4: Summary of generalized linear mixed model. The results show that canopy cover and biostatus, as well as the interaction between housing maximum age and nearest adult tree, have significant effects on species presence in patch.

	Estimate	**Std. Error**	**z value**	**Pr(>|z|)**	
(Intercept)	-3.60291	0.54345	-6.630	3.36e-11	***
I(scale(Canopy)^2)	-0.09953	0.04698	-2.119	0.0341	*
scale(housing.maximum.age)	0.07345	0.05967	1.231	0.2183	
scale(nearest.adult)	-0.17380	0.07986	-2.176	0.0295	*
BiostatusNative	1.41324	0.68958	2.049	0.0404	*
scale(housing.maximum.age) :scale(nearest.adult)	-0.18700	0.06233	-3.000	0.0027	**

Table 3.4 gives the result of the model selection. In this result, species presence is impacted by canopy cover and biostatus. The interaction between housing maximum age and nearest adult also has a significant effect on the species presence.

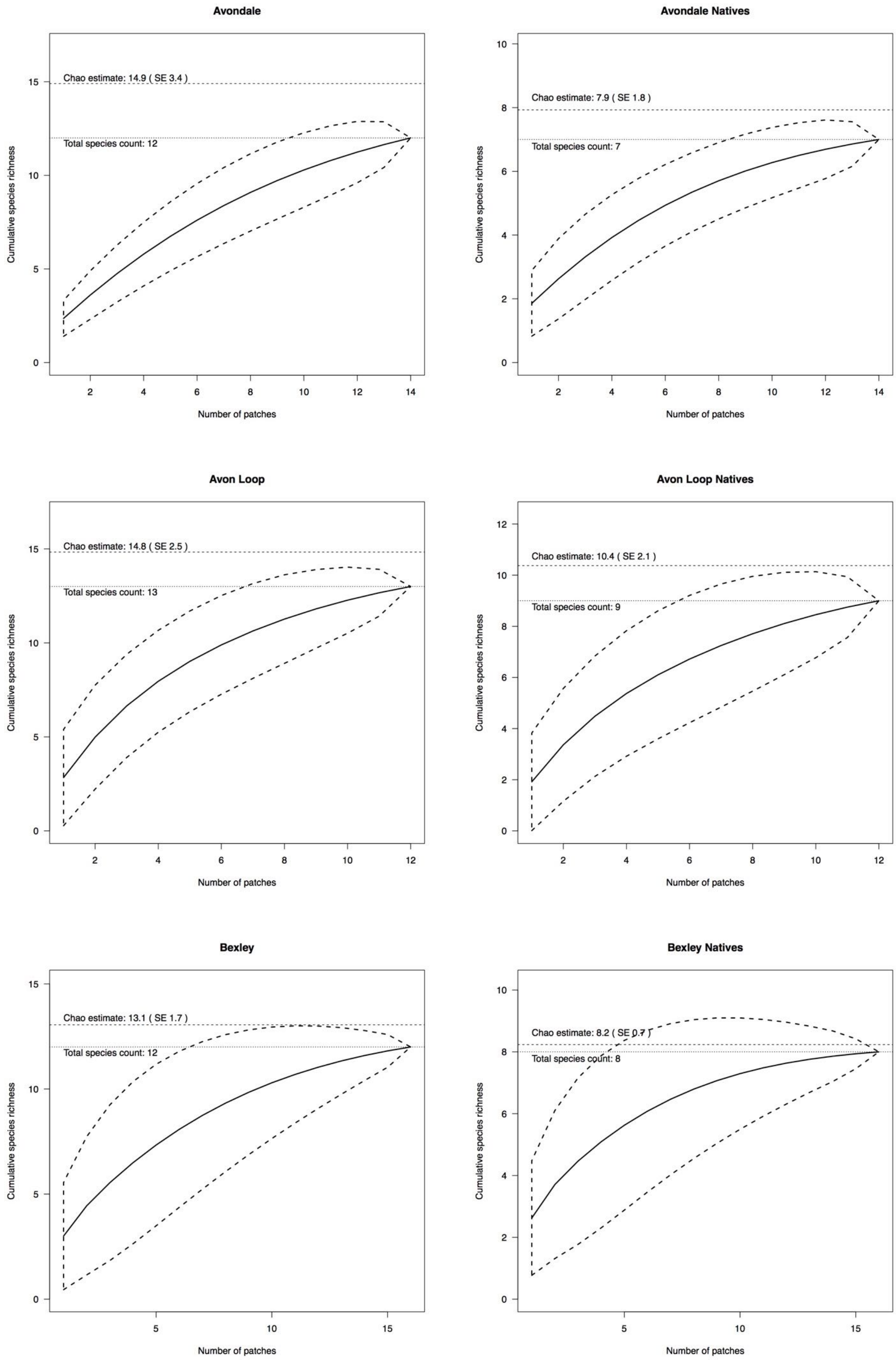


FIGURE 3.3: The results of species accumulation analysis of tree seedling species richness in each suburb. Graphs in left columns are showing the cumulative species richness of all seedlings and graphs in right columns are showing the cumulative species richness of native seedlings.

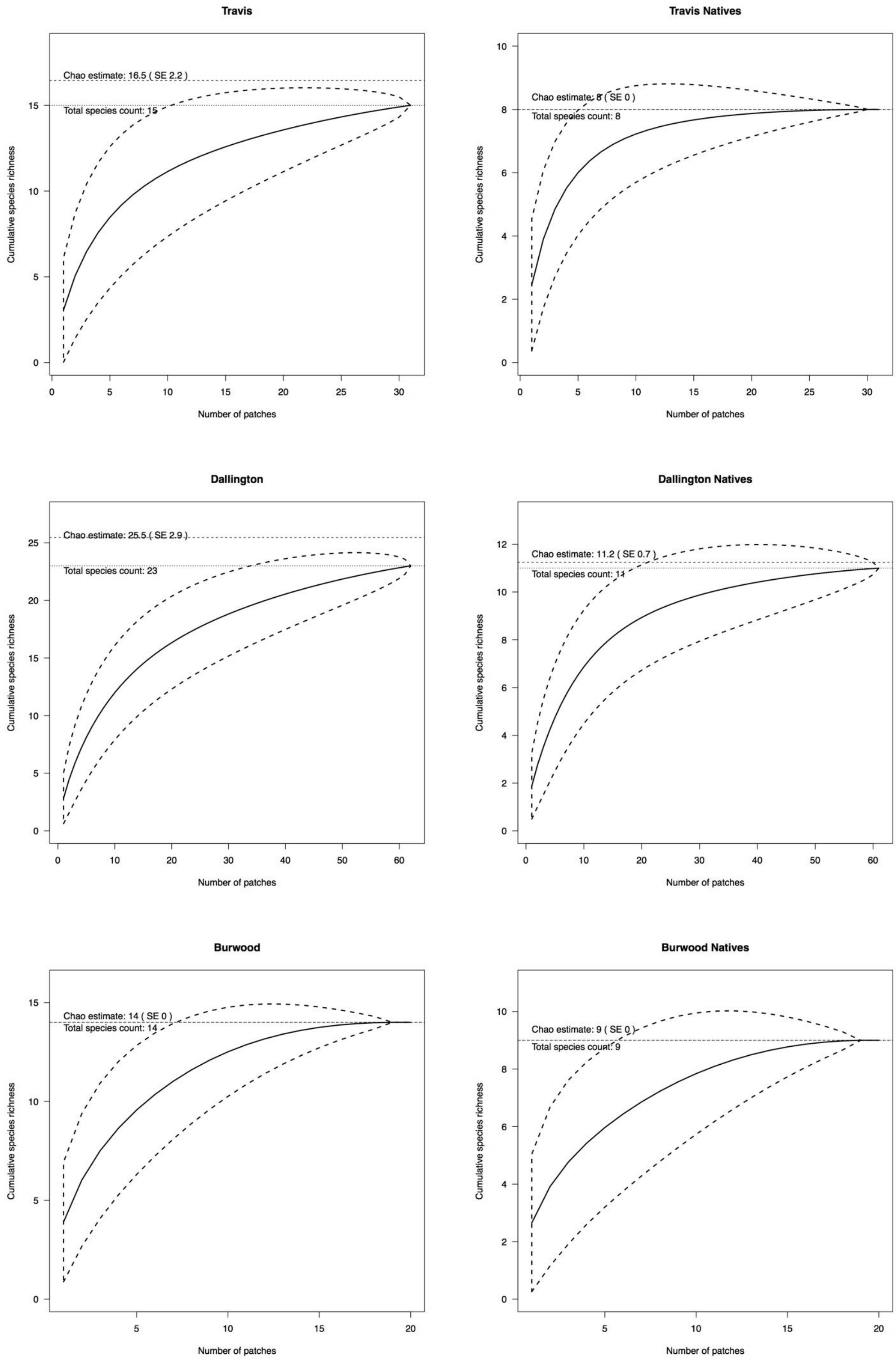


FIGURE 3.4: The results of species accumulation analysis of tree seedling species richness in each suburb. Graphs in left columns are showing the cumulative species richness of all seedlings and graphs in right columns are showing the cumulative species richness of native seedlings.

FIGURE 3.5: A dense patch of regenerating *Plagianthus regius* seedlings under a tree canopy. There were two adult trees next to this patch and the wind-dispersal seeds were stopped by this plant community and established under the canopy.

3.4.2 Seedling composition

Table 3.6 shows the tree species naturally regenerating in the Christchurch Residential Red Zone. *Cordyline australis* was the most common seedling in the area and I found 1926 seedlings totally. All top five species in the list are native species.

Some species only appeared in a few quardrats but made a lot seedling such as *Sophora microphylla* and *Plagianthus regius*. The seedlings of these species were found in the survey growing together by clusters (Figure 3.5).

3.4.3 Plants biostatus

Naturalised species dominate the species richness of planted trees in the Residential Red Zone area. The total species richness of exotic trees, including naturalised and unnaturalised species, was 331, which was 83.4% of all planted tree species. There are only 42 (10.6%) tree species native to Christchurch planted in the Residential Red Zone (Table 3.7).

Table 3.5: The seedling species found in the seedling survey, sorted from least to most quardrat numbers. There are 198 quardrats totally. Species names with "*" are naturalised exotic species.

Species	**Frequency**	**Species**	**Frequency**
**Aucuba japonica*	1	**Laurus nobilis*	7
**Camellia* spp	1	*Myoporum laetum*	9
**Cotoneaster* spp	2	**Euonymus europaeus*	11
**Ilex aquifolium*	2	*Pseudopanax* spp	21
Pittosporum eugenioides	3	**Hedera helix*	25
Podocarpus tōtara	3	*Sophora microphylla*	29
**Rubus fruticosus*	3	*Dodonaea viscosa*	32
Veronica spp	3	*Coprosma repens*	35
**Prunus* spp	4	*Pittosporum tenuifolium*	35
**Quercus robur*	4	*Plagianthus regius*	35
Solanum laciniatum	5	**Sambucus nigra*	50
Pittosporum crassifolium	6	*Coprosma robusta*	56
**Acer pseudoplatanus*	7	*Cordyline australis*	156

Table 3.6: The seedlings found in the seedling survey, sorted from least to most total numbers. Species names with "*" are naturalised exotic species.

Species	**Abundance**	**Species**	**Abundance**
**Aucuba japonica*	3	**Laurus nobilis*	34
**Camellia* spp	3	*Myoporum laetum*	40
**Cotoneaster* spp	6	**Euonymus europaeus*	53
**Ilex aquifolium*	6	*Pseudopanax* spp	81
Podocarpus tōtara	9	**Hedera helix*	88
**Rubus fruticosus*	9	*Coprosma repens*	141
Solanum laciniatum	15	*Dodonaea viscosa*	168
Pittosporum crassifolium	18	**Sambucus nigra*	175
**Prunus* spp	20	*Pittosporum tenuifolium*	193
Veronica spp	27	*Coprosma robusta*	327
**Acer pseudoplatanus*	29	*Sophora microphylla*	409
**Quercus robur*	30	*Plagianthus regius*	483
Pittosporum eugenioides	31	*Cordyline australis*	1926

The vegetation in Red Zone area is dominated by local native species, which is more than the total amount of exotic species (7836 species). As a residential garden area, exotic species especially naturalised species are very common (Table 3.7).

The 9367 local native species made 3709 seedlings which accounted for 85.8% of all seedlings (Table 3.7). The naturalised species made only 456 seedlings (10.5%, Table 3.7). Seedlings native to Christchurch were significantly more common than seedlings of other types biostatus (Table 3.10, Fig. 3.6).

Table 3.7: Amount of tree species and total seedlings numbers with different biostatus recorded planted in the Christchurch Residential Red Zone. Introduced (exotic) tree species are separated into naturalised species and unnaturalised species and all introduced seedlings are naturalised. N.A.= not applicable

	Christchurch native	**Non-Christchurch native**	**Naturalised**	**Unnaturalised**
Tree species	42	22	108	223
Seedling species	12	2	12	N.A.
Tree amount	9367	703	5011	2825
Seedlings amount	3709	159	456	N.A.

Table 3.8: Analysis result of Deviance Table. The result shows both Simple_Biostatus (native/exotic) and adult tree presence have significant effect on seedling presence in plots.

	Df	**Deviance**	**Resid. Df**	**Resid. Dev**	**Pr(>Chi)**	
NULL			394	4439.2		
Simple_Biostatus	3	1831.33	391	2607.9	<2e-16	***
treeFreq	1	1049.40	390	1558.5	<2e-16	***
Simple_Biostatus:treeFreq	3	9.78	387	1548.7	0.02053	*

Table 3.9: Summary of Generalized Linear Model of biostatus. All the components except the Naturalised:treeFreq affect the seedling regeneration in Residential Red Zone.

	Estimate	**Std. Error**	**z value**	**Pr(>|z|)**	
(Intercept)	-5.075e+00	7.597e-02	-66.805	<2e-16	***
Non-local native	-3.499e+00	6.432e-01	-5.441	5.31e-08	***
Naturalised	-1.776e+00	1.496e-01	-11.875	<2e-16	***
Unnaturalised	-6.328e+00	7.639e-01	-8.283	<2e-16	***
treeFreq	1.675e-03	4.838e-05	34.621	<2e-16	***
Non-local native:treeFreq	1.159e-02	5.014e-03	2.312	0.02077	*
Naturalised:treeFreq	-7.097e-04	1.117e-03	-0.635	0.52530	
Unnaturalised:treeFreq	1.544e-02	5.620e-03	2.748	0.00599	**

Table 3.10: The result of post hoc test comparing the different levels of biostatus. Christchurch native seedlings were proportionally much more abundant than seedlings of other types biostatus ($p<0.05$).

Biostatus	**Estimate**	**Std. Error**	**z value**	**Pr(>|z|)**
Non-Chch native - Chch native	-3.4994	0.6432	-5.441	<0.001
Naturalised - Chch native	-1.7761	0.1496	-11.875	<0.001
Unnaturalised - Chch native	-6.3276	0.7639	-8.283	<0.001
Naturalised - Non-Chch native	1.7233	0.6516	2.645	0.0324
Unnaturalised - Non-Chch native	-2.8282	0.9928	-2.849	0.0178
Unnaturalised - Naturalised	-4.5514	0.7710	-5.904	<0.001

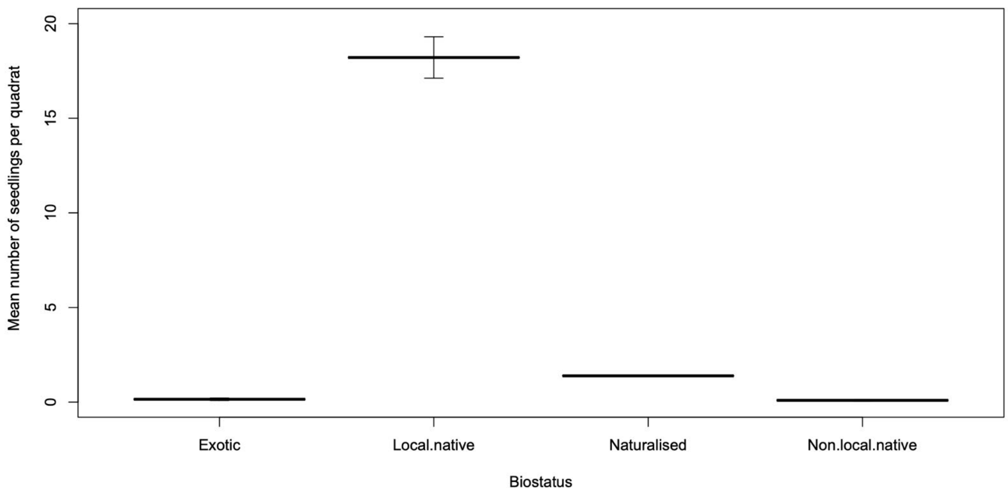


FIGURE 3.6: Model result shows the relationship between biostatus and mean number of seedling per quardrat. Species native to Christchurch have ability producing more seedlings than other three biostatus. Here local.native represents native to Christchurch, non.local.native represents non-Christchurch native.

Model result shows that species native to Christchurch have advantage producing more seedlings than other other biostatus (Table 3.6 & Figure 3.8)

3.4.4 Canopy area

TABLE 3.11: Summary of Generalised Linear Model of canopy cover. The result shows canopy has a significant effect on total seedling richness.

	Estimate	**Std. Error**	**t value**	**Pr(>|z|)**	
(Intercept)	5.622e-01	2.486e-01	2.262	0.0248	*
Canopy	2.341e-02	1.003e-02	2.334	0.0206	*
I((Canopy)ˆ2)	-2.052e-04	9.886e-05	-2.075	0.0393	*

The result of the canopy model shows that canopy cover has a significant influence on seedling richness. (Table 3.11 & Figure3.9).

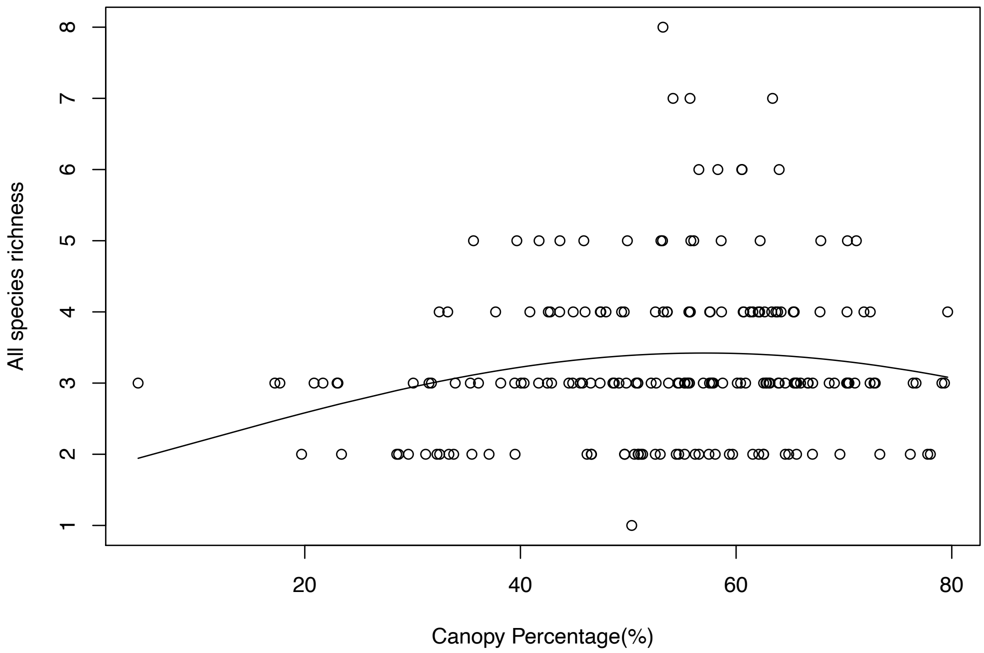


Figure 3.7: Prediction of all seedling species recorded in the survey regenerating under different canopy covers. The circles in the background show the species richness of different quadrats. The curves predicted the native species generating under different canopy cover in generalized linear model (GLM).

Seedlings mainly were found established under canopy covers ranging from 20% to 80%, only a few seedlings were found in the plot with low level canopy covers (<20%). The prediction shows that seedling species richness is getting higher in the plots with higher canopy cover, peaking around 70% (Fig. 3.7). Similar patterns were found for native (Fig. 3.8) and exotic species (Fig. 3.9).

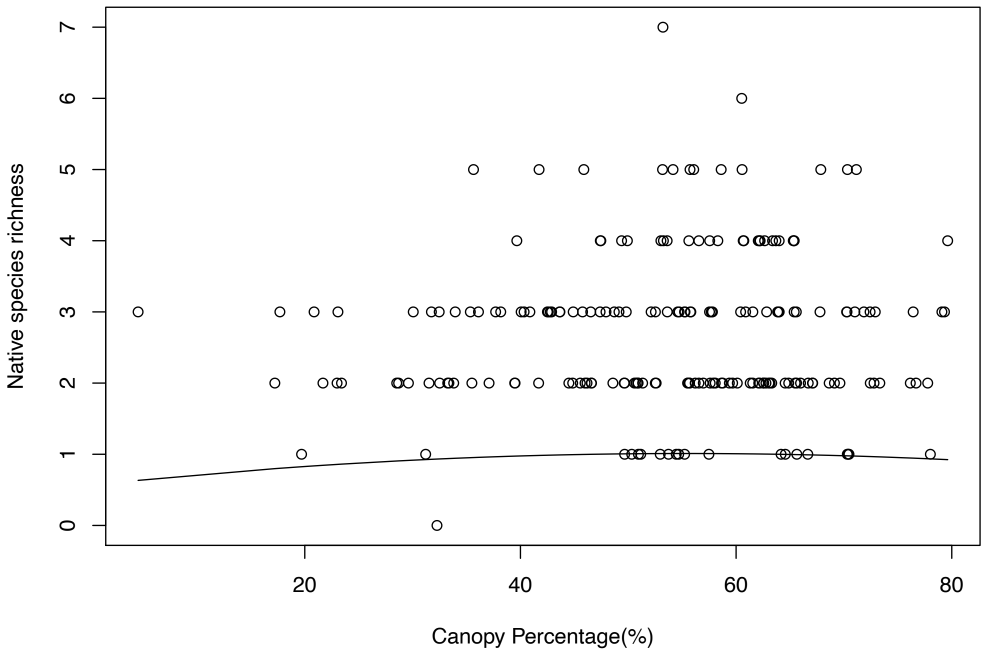


FIGURE 3.8: Prediction of native seedling species recorded in the survey regenerating under different canopy covers. The circles in the background show native species richness of different quadrats. The curves predicted the potential of native species generating under different canopy cover in generalized linear model (GLM).

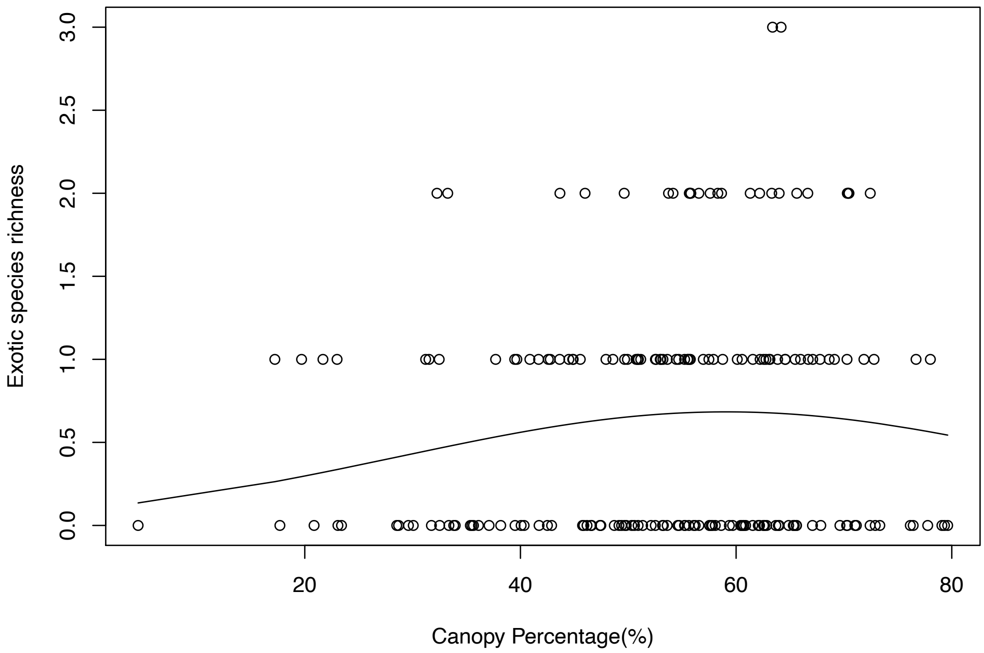


FIGURE 3.9: Prediction of exotic seedling species recorded in the survey regenerating under different canopy covers. The circles in the background show exotic species richness of different quadrats. The curves predicted the potential of exotic species generating under different canopy cover in generalize linear model (GLM).

3.4.5 Housing maximum age

Figure 3.10 shows that housing maximum age has a significant influence on total seedling amount. Mostly, older sites have more seedlings than the younger ones. The oldest sampled patch first subdivided at 1940 has the biggest seedling amount which is around 1400, however, the total seedling amounts of the younger patches followed it dropped down to the bottom under 100 at 1980. After that year, seedling total amount is getting bigger. Native (Fig. 3.11) and exotic seedling (Fig. 3.12) per quadrat per grid have similar patterns.

For mean exotic seedling amount, it had a slightly difference. In 1940 it was just around 9 and grew to 15 in 1955. That was a highest point. After that, there was a minor fluctuation in the next 30 years. It dropped to 5 in 1975 and went back to around 7 in 1980 and dropped to 1 in 1985. Finally it went up to 8 in 2000 (Figure 3.11).

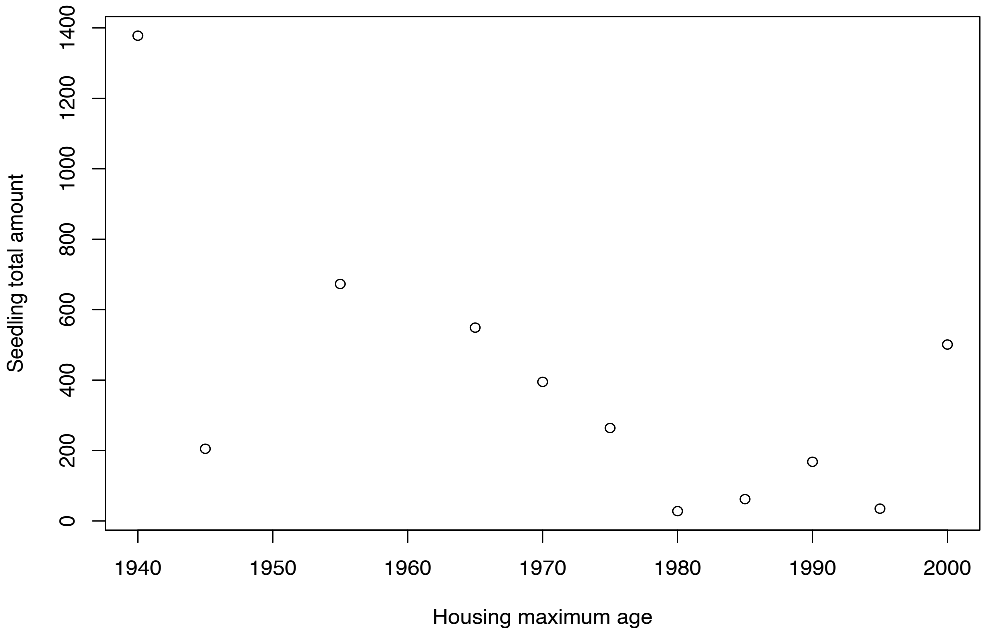


Figure 3.10: The relationship between total seedling abundance and the maximum age of the garden plants in the area, as measured by the maximum age of the houses in the surrounding 100 m × 100 m. Shown is the total seedling abundance.

Table 3.12: The relationship between soil versatility and total seedling abundance. PPQ= per patch per quadrat

Versatility level	**Number of quadrats**	**Total seedlings PPQ**
1	76	19.45
3	21	21.62
5	101	23.09

Table 3.13: The relationship between soil versatility and native seedling abundance. PPQ= per patch per quadrat

Versatility level	**Quadrat amount**	**Native seedlings PPQ**
1	75	15.29
3	21	15.33
5	101	20.15

3.4.6 Soil versatility

As shown in Table 3.12, there is no significant difference between total seedling per patch per quadrat of three versatility levels. Native (Fig. 3.13) and exotic seedlings (Fig. 3.14)

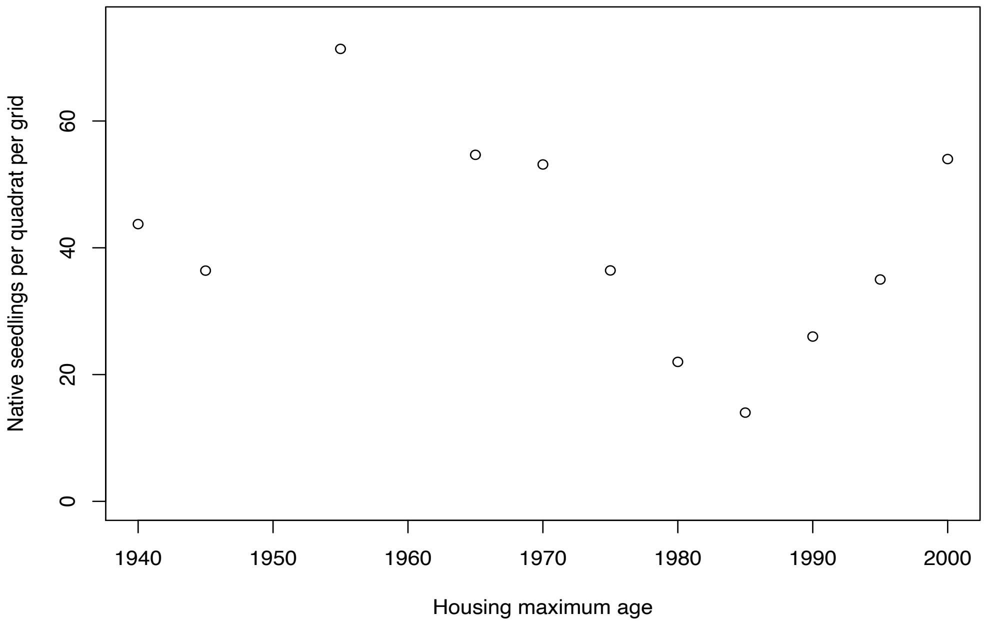


FIGURE 3.11: The relationship between native seedling abundance per quadrat per grid and the maximum age of the garden plants in the area, as measured by the maximum age of the houses in the surrounding 100 m × 100 m. Shown is the native seedling abundance per quadrat per grid.

TABLE 3.14: The relationship between soil versatility and exotic seedling abundance. PPQ= per patch per quadrat

Versatility level	**Quadrat amount**	**Exotic seedlings PPQ**
1	45	4.82
3	10	3.90
5	39	5.13

have similar results. Soil versatility level, at least as it was measured, does not have a big effect on seedling abundance.

3.4.7 Soil moisture

Unlike soil versatility, high and very high soil moisture have distinct effects both on total seedling (Table 3.15) and native seedling (Table 3.16), but there is no big difference between high and very high level. Soil moisture has no significant effect on exotic seedling abundance (Table 3.17).

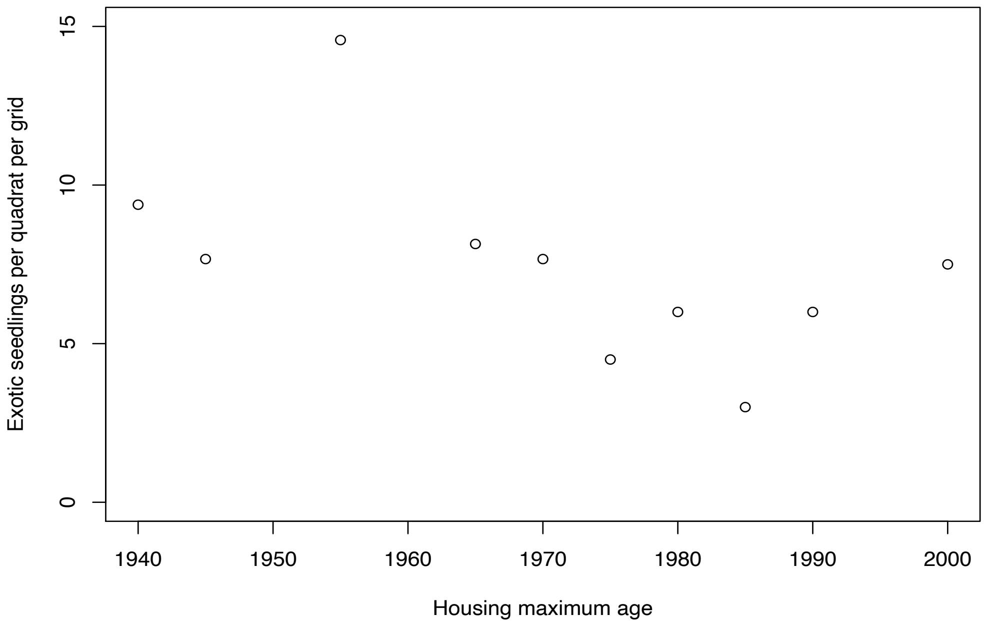


FIGURE 3.12: The relationship between exotic seedling abundance per quadrat per grid and the maximum age of the garden plants in the area, as measured by the maximum age of the houses in the surrounding 100 m × 100 m. Shown is the exotic seedling abundance per quadrat per grid.

TABLE 3.15: The relationship between soil moisture and total seedling abundance. PPQ= per patch per quadrat

Soil moisture level	**Plot amount**	**Total seedlings PPQ**
Moderate to High	6	9.83
High	114	19.78
Very High	82	24.48

TABLE 3.16: The relationship between soil moisture and native seedling abundance. PPQ= per patch per quadrat

Soil moisture level	**Plot amount**	**Native seedlings PPQ**
Moderate to High	6	8.33
High	113	17.64
Very High	82	22.22

TABLE 3.17: The relationship between soil moisture and exotic seedling abundance. PPQ= per patch per quadrat

Soil moisture level	Plot amount	Native seedlings PPQ
Moderate to High	3	3.00
High	56	4.68
Very High	35	5.29

3.4.8 The effect of the presence of adult trees in patches on seedlings

For exotic plants in Table 3.18 and Table 3.19, *Acer pseudoplatanus*,*Rubus fruticosus* and *Sambucus nigra* were the three species with seedlings found in patches both with adult trees and without adult trees. *Camellia* sp. and *Aucuba japonica* were only found in the patches with adult trees. The rest of the exotic species were only found in the patches without adult trees (*Cotoneaster* sp.,*Prunus* sp.,*Quercus robur* and *Laurus nobilis*).

Adult tree presence has a significant influence on native species seedlings. There are 10 native species (totally 14 species) were found having more seedlings in the patches with adult trees more than the patches without adult trees. However, exotic species are not influenced by adult tree presence, only four exotic species (totally 10 species) were found having more seedlings with adult trees in the same patches.

3.4.9 Minimum distances between seedlings and the nearest adults

As expected, fewer seedlings were found in Residential Red Zone area with increasing distance from the nearest conspecific adult tree. Most of the seedlings were located within 500 metres from the closest possible parent. *Sambucus nigra* seedlings can be found more than 4 kilometres away and this possible because we did not map the closer parents.

For the native species, most of the seedlings located within around 100 metres from the nearest adult trees. There were only three *Coprosma robusta* seedlings found more than 2 kilometres away from the adults and we did not map the closer parent trees.

3.4.10 Effects of nearest conspecific adults and site ages

The effect on native seedlings of the distance to the nearest conspecific adult tree was strongest in the youngest parts of the Residential Red Zone (Figure 3.16). Only for oldest parts, the distance to the nearest conspecific adult has a positive effect on the native seedlings.

TABLE 3.18: Part 1. The distribution of adult trees in patch or no. PPQ= per patch per quadrat. Species names with "*" mean exotic.

Species	Adult in patch	Seedlings PPQ
**Acer pseudoplatanus*	NO	4.25
**Acer pseudoplatanus*	YES	3.00
**Aucuba japonica*	NO	−−
**Aucuba japonica*	YES	3.00
**Camellia* spp	NO	−−
**Camellia* spp	YES	3.00
Coprosma repens	NO	3.33
Coprosma repens	YES	4.67
Coprosma robusta	NO	3.57
Coprosma robusta	YES	10.95
Cordyline australis	NO	8.33
Cordyline australis	YES	8.93
**Cotoneaster* spp	NO	3.00
**Cotoneaster* spp	YES	−−
Dodonaea viscosa	NO	3.66
Dodonaea viscosa	YES	5.8
**Euonymus europaeus*	NO	4.67
**Euonymus europaeus*	YES	3.00
**Hedera helix*	NO	3.00
**Hedera helix*	YES	4.00
**Ilex aquifolium*	NO	3.00
**Ilex aquifolium*	YES	3.00
**Laurus nobilis*	NO	4.07
**Laurus nobilis*	YES	−−
Myoporum laetum	NO	3.00
Myoporum laetum	YES	5.50
Pittosporum crassifolium	NO	3.00
Pittosporum crassifolium	YES	−−
Pittosporum eugenioides	NO	3.00
Pittosporum eugenioides	YES	14.00
Pittosporum tenuifolium	NO	3.19
Pittosporum tenuifolium	YES	4.58
Plagianthus regius	NO	5.79
Plagianthus regius	YES	11.75
Podocarpus totara	NO	3.00
Podocarpus totara	YES	3.00
**Prunus* spp	NO	4.25
**Prunus* spp	YES	−−
Pseudopanax spp	NO	3.00
Pseudopanax spp	YES	3.83
**Quercus robur*	NO	4.67
**Quercus robur*	YES	−−

Table 3.19: Part 2. The distribution of adult trees in patch or no. PPQ= per patch per quadrat. Species names with "*" mean exotic.

Species	**Adult in patch**	**Seedlings PPQ**
**Rubus fruticosus*	NO	3.00
**Rubus fruticosus*	YES	3.00
**Sambucus nigra*	NO	3.27
**Sambucus nigra*	YES	3.65
Solanum laciniatum	NO	3.00
Solanum laciniatum	YES	3.00
Sophora microphylla	NO	10.08
Sophora microphylla	YES	11.42
Veronica spp	NO	8.00
Veronica spp	YES	8.00

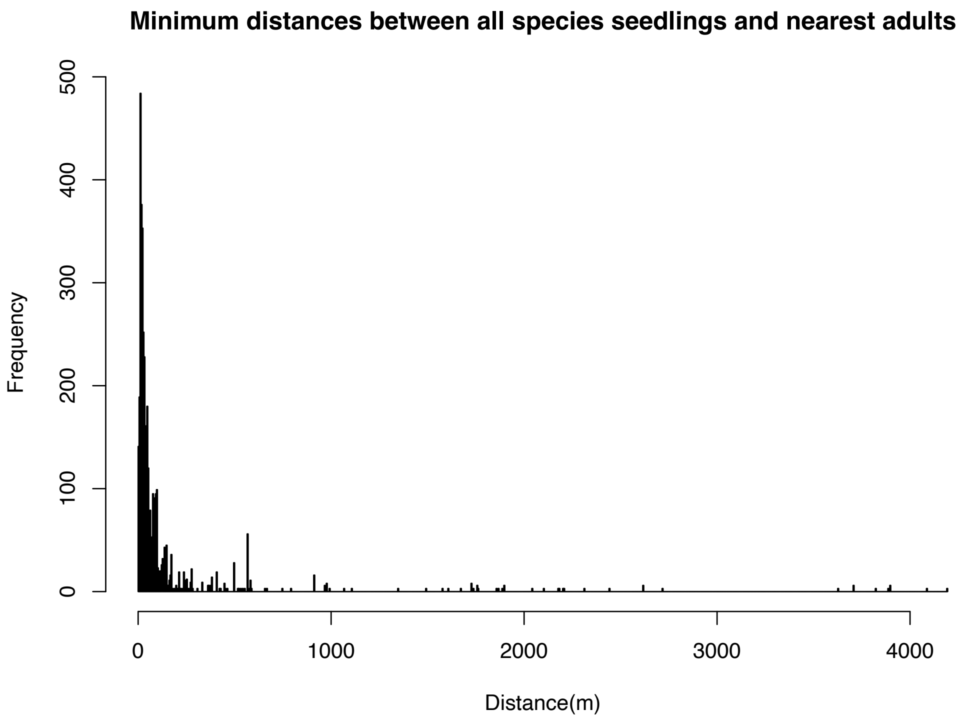


Figure 3.13: Minimum distances between all species seedlings and nearest adults

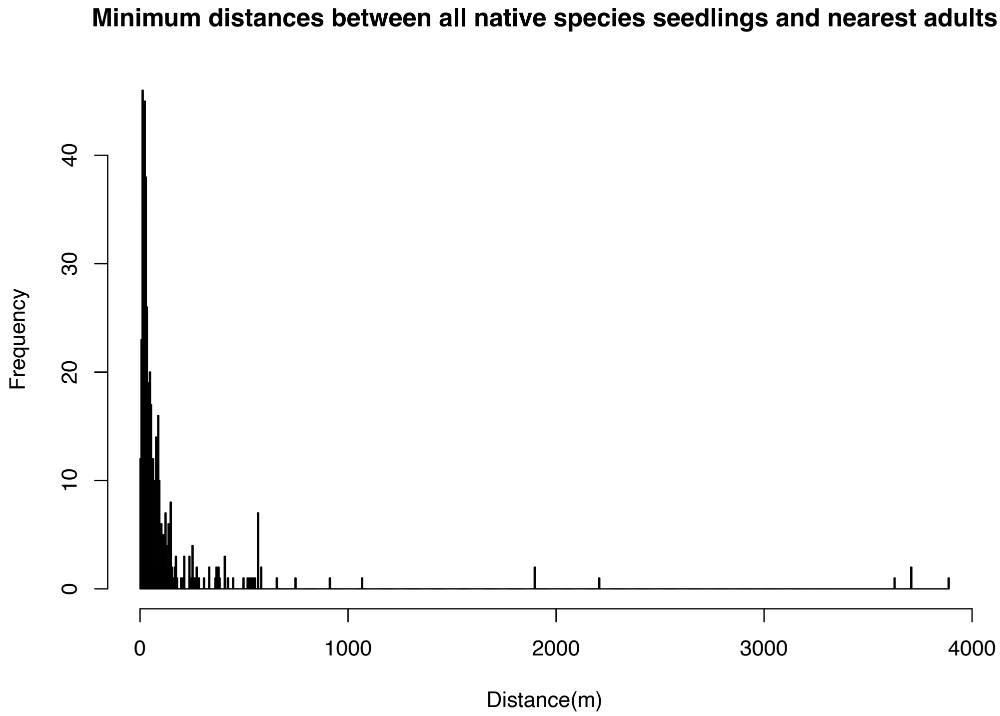


FIGURE 3.14: Minimum distances between all native species seedlings and the nearest recorded conspecific adults.

3.5 Discussion

The high proportion of Christchurch native species seedlings, especially relative to the number of adults present, shows they are doing much better than non-Christchurch native species and exotic species in this regeneration. From the result, it is clear that Christchurch native species can survive well under bright and dark environment. In other words, Christchurch native species have strong tolerance to light stress under canopy. Most native species are shade tolerant to some degree while most naturalised exotic species are much less so (Stewart et al., 2004). In New Zealand, many degraded urban forested are invaded by exotic deciduous trees which will senesce leaves in autumn causing a bright and drier seasonal environment (Wallace, Laughlin, & Clarkson, 2017; Heneghan et al., 2006). These changes will contribute to the invasion of exotic herbaceous weeds (McQueen, Tozer, & Clarkson, 2006) which in turn can prevent the regeneration of native species (Standish, Robertson, & Williams, 2001). Thus, restoring evergreen canopy will help native shade-tolerant species to regenerate in urban forests (Wallace et al., 2017).

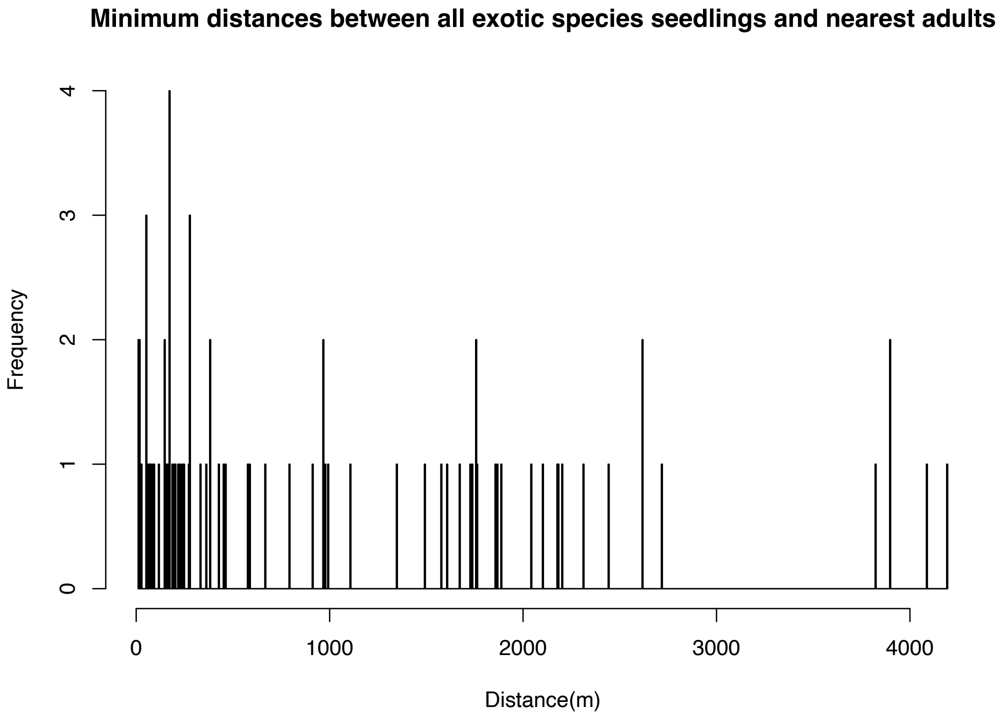


FIGURE 3.15: Minimum distances between all exotic species seedlings and nearest adults.

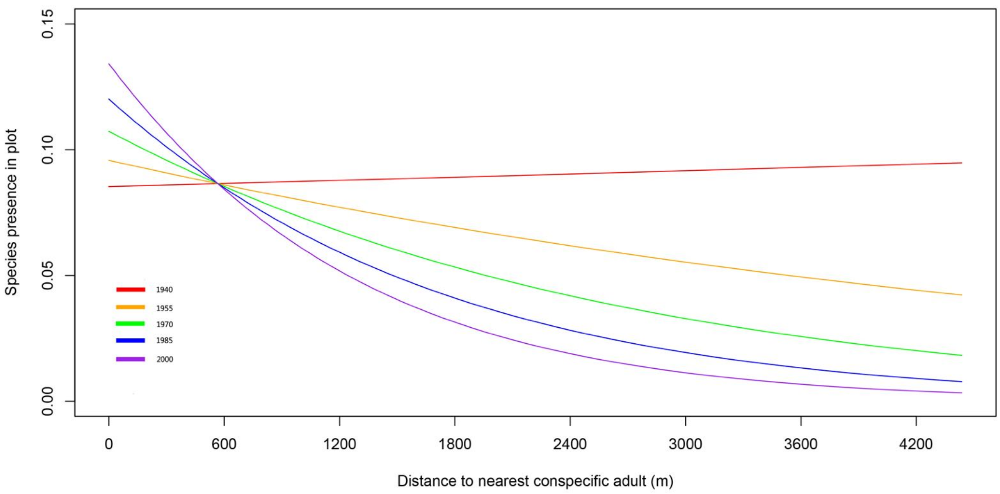


FIGURE 3.16: Effects of nearest conspecific adults and site ages on native species seedling presence. The plotted lines are the predictions from the generalised linear model. The effects of differing housing ages are represented by curves for the earliest year of housing in 1940, 1955, 1970, 1985, and 2000.

Recent studies show that invasive species can be one of the major factors which have negative effects on seedling regeneration in urban forest (Johnson & Handel, 2016; Labatore, Spiering, Potts, & Warren, 2017). Many native forest have been impacted negatively by exotic species (Biggerstaff & Beck, 2007). It was highly successful to do the restoration work in New York City forests with cleaning of invasive species and planting native species (Johnson & Handel, 2016). We do find when compared with native species, exotic ones do not rely as strongly on the adult trees which can be good seed resources and their seedlings can establish far away from the adult trees. Mostly because they are bird dispersed species. Among them, English Ivy(*Hedera helix*) is a good example which can form a dense ground cover to limit regeneration (Biggerstaff & Beck, 2007; Massad et al., 2019).

The occurrence of human activities can be reflected by vegetation even after natural succession has happened (LaPaix & Freedman, 2010). There has been a significant growth of native seedlings in the patches first subdivided from 1945 to 1965. Some historical records may show this reflection between human activities and seedlings presence. Helen Leach noted that a garden expert called David Tannock stated in 1934 that 'a native section is now an accepted feature of most large gardens' (Tannock, 1934). Also, Paul Walker claimed that one change of New Zealand gardening was an increasing interest in natives between 1940 and 1960 (Morris, 2006). That change in planting choices may now be responsible for the abundance and diversity of wild native seedlings in suburbs of this age.

The regeneration of exotic and native species is affected by seeds dispersal ability (Stewart et al., 2004). It has been found that exotic seedlings are less common under native tree canopies (Stewart et al., 2004). Some exotic species, such as *Quercus robur*, are recorded having limited dispersal ability in New Zealand (Stewart et al., 2004). Dispersal ability is also influenced by fruit types. Bird-dispersal is an important dispersal mechanism for most of the species in this research, both exotic and native species. As stated in the previous study, birds eat all native species producing fleshy fruits and disperse the seeds of these species (Burrows, 1994b). Seeds of native species which produce dry fruits are typically dispersed by wind or gravity only (Burrows, 1994b). Also, a high proportion of native species produce small size fruits which are between 2 and 8 mm in diameter (Burrows, 1994b). Compared with native species, a lot of planted exotic species produce dry fruit which have little value high-energy food sources for native birds (Burrows, 1994c; Stewart et al., 2004).

The ability to produce seeds is also important, and for dioecious trees this makes the spatial arrangement of parent trees particularly important. *Podocarpus totara* is a good example. Surprisingly, I did found some *Podocarpus totara* seedlings in the research area (one is in the survey plot and some others are located outside the plot)(Figure 3.17). It is well-known that tōtara has a 2-year reproductive cycle from strobilus initiation through to seed maturation and dispersal (Wilson & Owens, 1999). Moreover, male and female tōtara trees must live close enough to finish the reproductive cycle. I have measured the distance between the female tree that had seedlings and the several male trees around it, and the shortest distance is 21 meters. The suggestions for future planting is keeping all exist trees and planting as many tōtara trees as we can around these exist trees to make more seeds as we indeed cannot identify female and male tōtara trees at their juvenile stage.

FIGURE 3.17: *Podocarpus totara* seedling found in the survey

Combining seed dispersal distance and presences of adult tree in patches, adult trees in the patches can promote the seedlings in the regeneration. Most of the seedlings could be found very close to the nearest adult plants. We also found lots of *Pittosporum eugenioides* seeds and seedlings under a big canopy in the survey (Figure 3.18 and Figure 3.19). It has been mentioned that *Pittosporum eugenioides* can supply fleshy seeds to the birds and birds will help to disperse the seeds. However, as can be seen from seedling dispersal graph, gravity probably still the main way to disperse the seeds for *Pittosporum eugenioides*.

FIGURE 3.18: *Pittosporum eugenioides* seeds found in the survey

FIGURE 3.19: *Pittosporum eugenioides* seedling found in the survey

Interestingly, I found that the effects of distance to conspecific adults on seedling abundance was much weaker in older areas of housing that young areas of housing. This could be contributed by several factors. These factors may affect regeneration through seed limitation, or establishment limitation. It suggests that the old areas have enough older trees, which can make many more seeds than younger trees and can build up a larger seed bank. Previous studies have suggested seed limitation and seedling limitation have significant impacts on forests, potentially working on composition, structure and diversity (Hurtt & Pacala, 1995; McEuen & Curran, 2004). Old areas also have enough older trees which can make a better canopy cover, the microclimate may fluctuate less and light availability will be reduced, hence allowing shade-tolerant native species growing (Wallace et al., 2017). These forest with better canopy cover can supply the food and habitats for birds and in turn, birds can help to disperse the seeds.

Disturbance is generally treated as an important factor which can promote the invasion of exotic species (Byers, 2002). However, resource availability is a key determinant of the ability of exotics to invade a habitat (Alpert, Bone, & Holzapfel, 2000; Davis, Grime, & Thompson, 2000). As the result in this research, most exotic species likely prefer a mild canopy, which is not too bright nor too dark. If long dispersal and shade-tolerant exotic species can be controlled, this should help the urban native-dominant forests to regenerate in Christchurch.

3.6 Conclusion

Native species frequently present as seedlings are suggested as targets for restoration: enhancing the conditions for these species to naturally regenerate looks to be an effective way to increase urban forest biodiversity. Among these species, local native species play a dominant role in the regeneration of urban vegetation. These species can be good seed resources at the early stage of restoration as they can build the vegetation quickly.

Based on the presence of adult trees and seedling dispersal result, most of the seedlings are found very close to their parents trees, although birds especially can help some of them to establish much further away. In other words, parent plants in the patch can help the regeneration and seedling dispersal can help recolonisation. Management through maintaining the existing parent trees and enriching the local native biodiversity is necessary in the future restoration in Residential Red Zone area.

We also found the effects of distance to conspecific adults on seedling abundance is much weaker in older areas of housing that young areas of housing. That indicates the important value of older trees in urban forest regeneration. It's very important that the these old trees should be cared for, and not damaged, when restoration planting or other land use changes occur around them.

Chapter 4

Dispersal distances of trees establishing wild in urban Christchurch

4.1 Introduction

Global biodiversity is threatened for several reasons. The most important reasons at this moment are land modification and invasive species (Sala et al., 2000). Land modification has become one challenging problem for ecosystems globally (Mayer et al., 2016). It reduces habitat area and fragments habitat patches and, consequently, it reduces the connectivity between patches (DeFries, Hansen, & Turner, 2007). Reduced habitat area and connectivity are associated with biodiversity loss. The fragmented patches make plant communities spatially isolated from each other; most distances between these urban plant communities are many hundreds metres or longer (Cain, Milligan, & Strand, 2000). For the plant species in these communities, seed dispersal is the way that plant communities can exchange species and colonise new habitat patches (Cain et al., 2000).

In New Zealand, the land modification started from 12th century when the Polynesian settlers arrived (M. S. McGlone, 2001; Wilmshurst et al., 2008). They set fires and cleared the vegetation from extensive areas, especially in the drier east (M. S. McGlone, 2001; Perry, Wilmshurst, & McGlone, 2014b). In 19th century, European settlers arrived in New Zealand and continued removing the forests for settlements and agriculture (Wardle, 1991).

Dispersal defined

Dispersal is the movement of organisms, their propagules, or their gametes (e.g., pollen) away from the source area (Bullock, Kenward, & Hails, 2002; Nathan, Safriel, Noy-Meir, & Schiller, 2000; Petit, 2004; Stenseth & Lidicker, 1992). Seed dispersal is the movement of seeds away from their parent plants. It is an important ecological process which is typically the only way for plants to move in response to land modifications (Howe, 2016).

Dispersal mechanisms

There are several types of dispersal mechanism used by plants (Howe & Smallwood, 1982) (Table 4.1, Fig. 4.1). During the seeds' movement from one patch to another, animals may help to transport seeds between different patches (Lundberg & Moberg, 2003). In New Zealand, 70% of 240 common woody trees produce succulent fruits which are suitable for bird dispersal (Burrows, 1994a; Clout & Hay, 1989), such as *Coprosma* and *Podocarpus* species. Some other woody trees use wind to dispersal seeds with specialised structures, such as *Fuscospora* and *Dodonaea* which produce winged seeds, or like *Kunzea* and *Plagianthus* species produce very light seeds (Wardle, 1991). Other forms of dispersal are less common. Dispersal by water occasionally distributes *Fuscospora* and *Sophora* seeds and some *Pittosporum* species can produce sticky seeds with fat to attract birds.

TABLE 4.1: Different types dispersal mechanism used by plants (Howe & Smallwood, 1982)

Dispersal agent	**Animal**	**Wind**	**Water**	**Self-dispersal**
General adaptation	fleshy nutrient	size reduction	resistance to sinking	explosive fruits
	chemical attractant	high surface/volume ratio	uses surface tension	creeping diasporas
	clinging structures	tumbleweeds	low specific gravity	

Advantages of dispersal

A lot of work had been done to explore the advantages of plant seed dispersal (Howe & Smallwood, 1982). First of all, dispersal can help seeds and seedlings to escape from their parents (Connell, 1971; Janzen, 1970). Secondly, plants can colonise new empty habitats by seeds dispersal (Baker, 1974). Also, seed dispersal can find suitable microhabitats for establishment and growth (Howe & Smallwood, 1982). In all cases, seed dispersal has the potential to increases a parent plant's fitness through increasing the representation of its genes in the next generation.

FIGURE 4.1: Photos of different native tree species seed types. Upper-left: *Coprosma propinqua*, upper-right: *Dodonaea viscosa*, lower-left: *Pittosporum eugenioides*, lower-right: *Podocarpus totara*

Dispersal distance

Most seeds do not dispersal far from their parent trees; in many areas most seeds fall only one or several metres away (Cain et al., 2000). A lot of studies have focused on the different seed dispersal distances and the importance of dispersal for different plant growth forms (Cain, Damman, & Muir, 1998; Cheplick, 1998; Willson, 1993; Howe & Smallwood, 1982).

While most seeds typically fall close to their parents, a few seeds also can disperse long distances. This long-distance travel can be achieved in several ways (Chambers & MacMahon, 1994; Sauer, 1991; Sorensen, 1986), such as animal dispersal and wind dispersal (Cain et al., 2000). For many species, dispersal distances of 1–20 km have been recorded (Sauer, 1991; Cheplick, 1998; Nathan, 2000). In exceptional circumstances, some seeds can disperse across oceans, such as over 2,000 km from Australia to New Zealand (Close, Moar, Tomlinson, & Lowe, 1978).

Traits effects on dispersal

It is usually assumed that small seeds should travel further than large seeds (Thomson, Moles, Auld, & Kingsford, 2011; Greene & Johnson, 1993). Plants which make small seeds can increase their survival probability by making more seeds and dispersing them further from the parent plants than plants which make large seeds (Hyatt et al., 2003). Small and light seeds produced by small-seeded plants can increase the possibility of rare long-distance dispersal events, which will result in dispersal curves with fat, long tails and increased mean and maximum dispersal distances (Thomson et al., 2011).

Plant height is another important trait which affects plants' dispersal abilities (Falster & Westoby, 2005). Taller plants can release seeds at greater heights than shorter plants, and for wind-dispersed species, greater heights result in increased dispersal distances (Tackenberg, Poschlod, & Bonn, 2003; Soons, Nathan, & Katul, 2004; Travis, Smith, & Ranwala, 2010).

The objective of this study is to explore how far native tree seedlings establish from parent trees in urban Christchurch. After the devastating earthquakes of 2010–2011, only big garden trees were left remaining in the Residential Red Zone, with basic managements such as monthly mowing of the grass and spraying the weeds. This created an excellent opportunity to study tree dispersal across a large urban area. This knowledge will help to inform a habitat connectivity map of Christchurch (Chapter 6), and also helps to inform urban habitat restoration. There were two objectives in this chapter:

1. What are the dispersal distances for the naturally regenerating trees species in the Christchurch residential red zone?
2. Which tree species are likely to be currently dispersal limited in the residential red zone?

4.2 Methods

4.2.1 Study sites

Six suburbs with different ages were chosen to do the seedlings survey in the Residential Red Zone area (Figure 4.2). In these six suburbs, all houses had been removed and only larger garden trees were left. Now, the area is maintained as a temporary parkland. For details see Section 1.7.

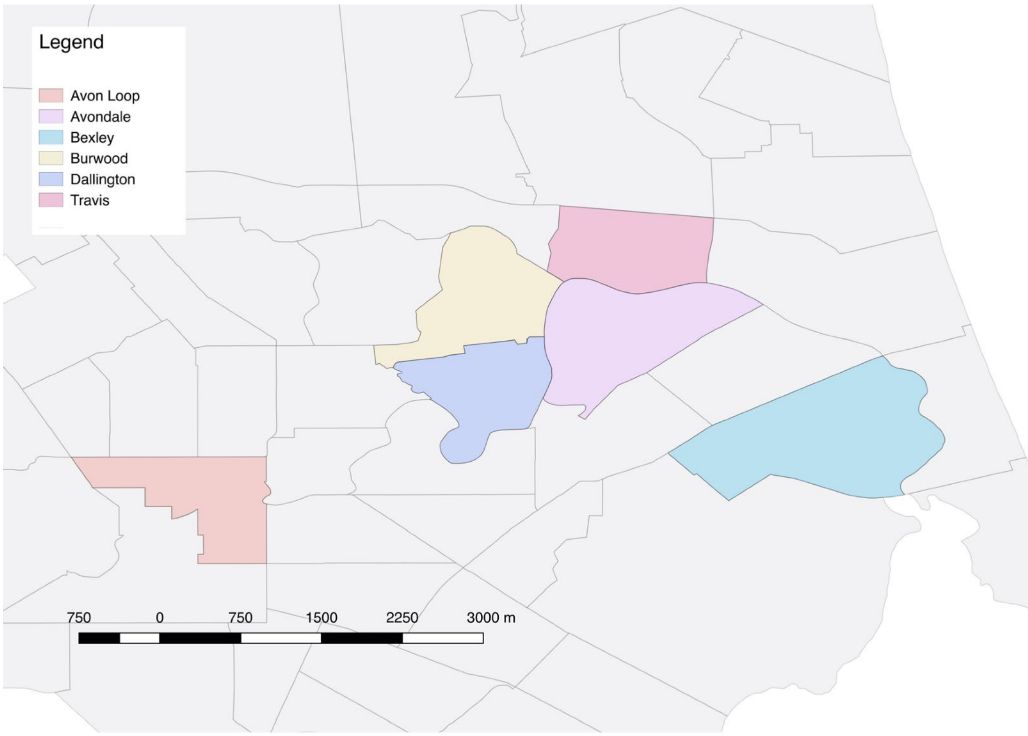


FIGURE 4.2: Map of six suburbs with different ages in Residential Red Zone.

4.2.2 Data collection

A walk-in survey (walked transects located in stratified random locations in each suburb) was done in the six suburb areas. Canopy data were measured and soil data was obtained from available GIS layers. For data collection details see Section 4.2.2.

4.2.3 Minimum seedling dispersal distance

A database of adult trees was created using the Red Zone tree maps created after earthquakes by using the software package R (R Core Team, 2016). It was based on tree survey data collected by Treetech for the Canterbury Earthquake Recovery Authority, and provided to me by the Canterbury Earthquake Recovery Authority for use in this study. The tree information included botanical names and GPS information for all the surveyed trees.

The minimum distance between every seedling found in my survey and the nearest mapped adult of the same species from whole Red Zone tree map was calculated in R.

Note that I did not have a tree map for areas of the city outside of the Residential Red Zone and for rarer species, it remains possible that the nearest conspecific adult was in a garden outside of the Residential Red Zone.

4.2.4 Random sample dispersal distance

Random points were set in the red zone area to simulate random seedling distribution. This was done so that I could assess what the distribution of seedling distances from parent trees would be within the long, thin, irregular shape of the Residential Red Zone in the absence of dispersal limitation. I created 10,000 random points within the residential red zone polygon in QGIS (QGIS version 2.18.13) by running the 'random points inside polygon' algorithm. The dispersal distances of these random points to all mapped adult trees were calculated in R.

4.2.5 Interpreting seedling dispersal distributions

I used t-tests to compare the distances of real seedlings from the nearest adult trees of the same species, with random points from the nearest adult trees. Tree species with seedlings closer to adult trees than random points are constrained in their dispersal. The bigger this difference, the more restricted seedlings are to just areas near adults. If there is no difference, then seedling establishment in this area is assumed to not be constrained by dispersal.

Seedling dispersal curves in this chapter are the result of the combined processes of dispersal distributions and seedling establishment. Seedling establishment will be determined by both propagule supply (dispersal) and the microhabitat conditions. The study site is uniform in its topography, management and general habitat structure and the upper soil layer is all the remains of suburban gardens. As such, I am assuming that the seedling dispersal curves primarily reflect the seed dispersal distributions, which are expected to be leptokurtic in shape (Westcott et al., 2005).

4.2.6 Seedling species

There are some species in the seedling survey which were not present in the adult tree data, and I assume too low statured to be included in this survey. These species therefore cannot be analysed in the chapter. They are *Aucuba japonica*, *Coprosma repens*, *Euonymus europaeus*, *Hedera helix*, *Rubus fruticosus* and *Solanum laciniatum* (Table 4.2). Other species had to be excluded because their seedlings could not be reliably identified to species, on account of widespread hybridisation in the area (Table 4.2). This left ten native species and five exotic species available for dispersal analyses.

TABLE 4.2: Seedling data which were not used for analysis. Species with * are exotic

Species name	**Seedlings no.**	**Reasons for not using**
*Aucuba japonica**	1	No records in tree data
Camellia spp*	1	Cannot identify species
Cotoneaster spp*	2	Cannot identify species
*Rubus fruticosus**	3	No records in tree data
Veronica spp	4	Cannot identify species
Prunus spp*	5	Cannot identify species
Solanum laciniatum	5	No records in tree data
*Euonymus europaeus**	11	No records in tree data
Pseudopanax spp	22	Cannot identify species
*Hedera helix**	26	No records in tree data
Coprosma repens	37	No records in tree data

4.3 Results

As expected, seedlings of most species declined with distance away from the nearest adult conspecific tree (Figure 4.5, (Figure 4.6)). More than half of the seedlings were located within 1 kilometre of the nearest possible parents. On average, seedlings of all species were within 100 m of the nearest conspecific adult. The maximum seedling dispersal distance was around 4 kilometres, for *Coprosma robusta*(see below).

4.3.1 Native species

Of the native species assessed, only two species, *Pittosporum eugenioides* and *Dodonaea viscosa*, did not have seedlings significantly closer to conspecific adults than the random seedling distribution (Table 4.3, Figure 4.5). For *D. viscosa*, there was no detectable difference between the mean seedling distances and mean random points distances. Surprisingly, seedlings of *Pittosporum eugenioides* were found significantly further from mapped conspecific adults than expected from a random seedling distribution. However, both *D. viscosa* and *P. eugenioides*differences were abundant as adults and the median seedling distance and the median random point distance from adults was less than 100 m in both cases (Figure 4.5).

The eight other native species had seedlings significantly closer than random points to nearest adult trees (Table 4.3). Two of these eight species, *Cordyline australis* and *Coprosma robusta*, had dispersal curve shapes close to the shapes of random point dispersal curves (Figure 4.5).

Most of the native seedlings were found within 500 meters from a possible parent. Only *Coprosma robusta* had seedlings also scattered substantially further away from mapped adults, with seedlings found a maximum distance of around 4 km from the nearest mapped adult (Figure 4.5).

For *Cordyline australis*, the dispersal curve shows it disperses almost as well as random points. The mean seedling dispersal distance was 31.9 m and the mean random points dispersal distance was 43.8 m. It may because *Cordyline australis* is very common in Red Zone, and they are close to each other (Figure 4.3).

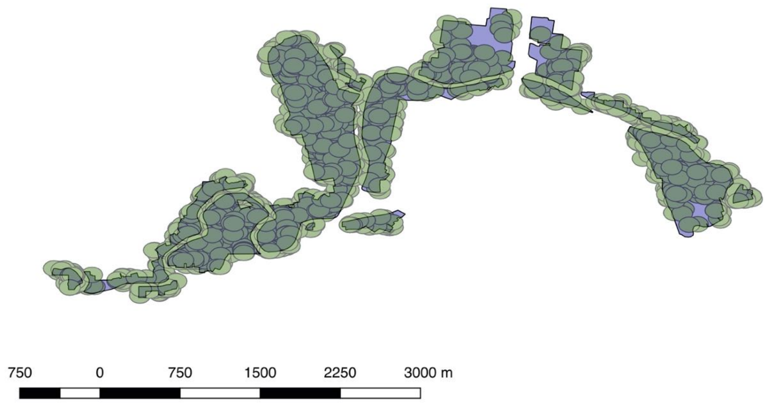


FIGURE 4.3: *Cordyline australis* buffer zone map. The buffer zone distance for *Cordyline australis* is 100 meters (Figure 4.5).

4.3.2 Exotic species

Generally, as shown in Figure 4.6, exotic seedlings were not as common as native seedlings in the survey area and most of them were found close to the adult trees.

Seedlings of four of the five exotic species seedlings in the survey were only found within 500 meters of mapped adults.

Of the five exotic species, only *Sambucus nigra* seedlings were not significantly closer to adults than random points (Table 4.4, Figure 4.6). *Sambucus nigra* had the most seedlings compared with other exotic species, and its seedlings could be found from very close to the adult trees until further than 4 km (Figure 4.6).

4.3.3 Fruit types and seedling distances

There were two species with maximum seedlings dispersal distances are nearly 4000 meters, *Coprosma robusta* and *Sambucus nigra*. Both produce copious amounts of fleshy bird dispersed fruits that are popular with local birds (personal observation) (P. A. Williams, Karl, Bannister, & Lee, 2000; Ferguson & Drake, 1999; Debussche & Isenmann, 1994). However, most other species in this study were also bird dispersed, and yet their seedlings maximum dispersal distances were all less than 500 meters. This will be an artefact of the varying abundance of the mapped adults of these species. It is not possible for seedlings with the more abundant adults to be kilometres away from the nearest adult.

Notably, there were only eight *Coprosma robusta* trees and three *Sambucus nigra* trees mapped in the Residential Red Zone tree data, much less abundance than the other species with seedlings (Table 4.3, Table 4.4). Much of this is because most of the adults of these two species remaining in Red Zone were shrubs or low-statured trees that were not mapped (personal observation). *Coprosma robusta* grows as a shrub to small tree and I found smaller adults that were missing from the tree map. *Sambucus nigra* is a small weedy tree that is not often purposefully grown but can be present in more wild and woody gardens. Since these two species were not as thoroughly mapped as the other, taller tree species, this will account at least in part for their apparently greatly increased seeds dispersal distances.

Wind can help seeds to disperse long distances (Lake & Leishman, 2004). Wind dispersed species can produce small, light seeds that are readily transported by wind (Lake & Leishman, 2004). In this study the wind dispersal species were *Dodonaea viscosa*, *Plagianthus regius*, and *Acer pseudoplatanus*. The maximum seed dispersal distance of *Dodonaea viscosa* was 169 meters and for *Plagianthus regius* it was 105 meters. However, I found the landscape management methods affected seeds dispersed by wind. Figure 4.4 gives a good example in which *Plagianthus regius* seedlings appear to have been stopped in large numbers by a garden fence.

FIGURE 4.4: *Plagianthus regius* seedling found around Fairway Park, Red Zone, Christchurch.

4.4 Discussion

Human activities are the main factor determining environmental conditions in urban areas (Maurer, Peschel, & Schmitz, 2000; Sukopp, 2004). In this study, the survey area was being maintained with basic management, such as lawn mowing and weed spraying. This kept the area open and "tidy" and provided few opportunities for tree seedlings to establish.

High densities of seeds fall around parent plants and long distance dispersal events will happen over larger scales which are infrequent and unpredictable (Moody & Mack, 1988). The spread of plants through a combination of local short-distance dispersal and infrequent long-distance dispersal has been called 'infiltration invasion' (Wilson, 1989b). For urban environments such as my study area in Christchurch, both processes are important. My research shows the value of short-distance dispersal from existing, mostly planted, adult trees, for fuelling wild tree regeneration in cities. All of the seedlings I found were of species present as adults in the city. The natural arrival and establishment of species currently absent from the city but present in neighbouring wildlands would require long-distance dispersal and subsequent successful establishment.

Which species are most likely to have sufficiently frequent long distance dispersal events to result in long-distance establishment? This is very hard to answer as long distance dispersal events are rarely recorded (Minor & Gardner, 2011). However, animal-dispersed seeds are thought to be likely to disperse seeds further than the seeds dispersed by other ways (Nathan & Muller-Landau, 2000; Clark et al., 2005; Minor et al., 2009). In other words, plants with large seeds or seeds dispersed by gravity, water, or wind may be less likely to disperse long distances (Minor & Gardner, 2011). This likely gives New Zealand native trees an advantage as the majority of them have bird dispersed seeds (P. A. Williams & Karl, 1996). At least 260 native species in New Zealand can produce fleshy fruits which are attractive to birds (Timmins & Williams, 1987).

Exotic species have become big problems in managing urban green spaces and frequently make restoration projects more complicated (D'antonio & Meyerson, 2002). There are typically more wild exotic plant species in the habitats close to the settlements than the habitats far from settlement (Sullivan, Timmins, & Williams, 2005b; Timmins & Williams, 1991). I did find twelve exotic species in the survey and all of them were common garden plants in the Residential Red Zone.

Exotic species often not easy to eradicated from established locations (Minor & Gardner, 2011). It is very important to stop or slow their spread into new areas. Therefore, several exotic species management strategies have been mentioned (Sterling, Thompson, & Abbott, 2004; Flory & Clay, 2009). A popular alternative is focusing control on large populations rather than the locations of all of the satellite patches, because these large populations contribute the greatest number of dispersed seeds to the next generation (Shmida & Ellner, 1984). However, in urban environments where source parents of weeds can be abundant in private gardens, ongoing control in value habitat patches is essential. Catching new weeds soon after they arrive in new patches is cheaper and has less impacts than allowing them to establish.

For those native species which take more time to be mature, the presence of mature adult trees in the Residential Red Zone is very important. An example of this is *Podocarpus totara*, which is a long-lived and relatively slow growing native *Podocarpus* tree. I only found tōtara seedlings nearby to a pair of large male and female trees. It is both likely that seedlings will have established near older trees, and more likely that they will have established further from older trees. This would be an interesting extension to my study to age parent trees in Christchurch, assess how their seed production increases with age, and explore how this influences seed dispersal and seedling establishment.

4.5 Conclusion

Generally speaking, most species' seeds do not travel far from their parent plants. In this research, I found that most seeds travelled less than 100 meters from the nearest conspecific adults. Because of this, seedling number decreases as the distance from parents gets further.

For several tree species, adult plants were sufficiently widely distributed and dispersal was adequate enough for wild seedlings to be regenerating throughout much of the Residential Red Zone. For example, *Cordyline australis* and *Coprosma robusta* seedlings have a near random distribution in the Red Zone, so they are the easiest and best species for regeneration. When planned native forest restoration begins in the Red Zone, species like these won't need to be widely planted as they are already regenerating naturally. Removing the current mowing and weeding will greatly assist with their natural regeneration.

In contrast, *Podocarpus totara* is one of the hardest species for regeneration. That is because male and female trees need to be growing in close proximity and only older trees fruit. The planting suggestion for this kind of species is to plant as close as possible to the adult trees. It is important that established adult trees of species like this should be kept in Red Zone as seeds resources for future forest restoration.

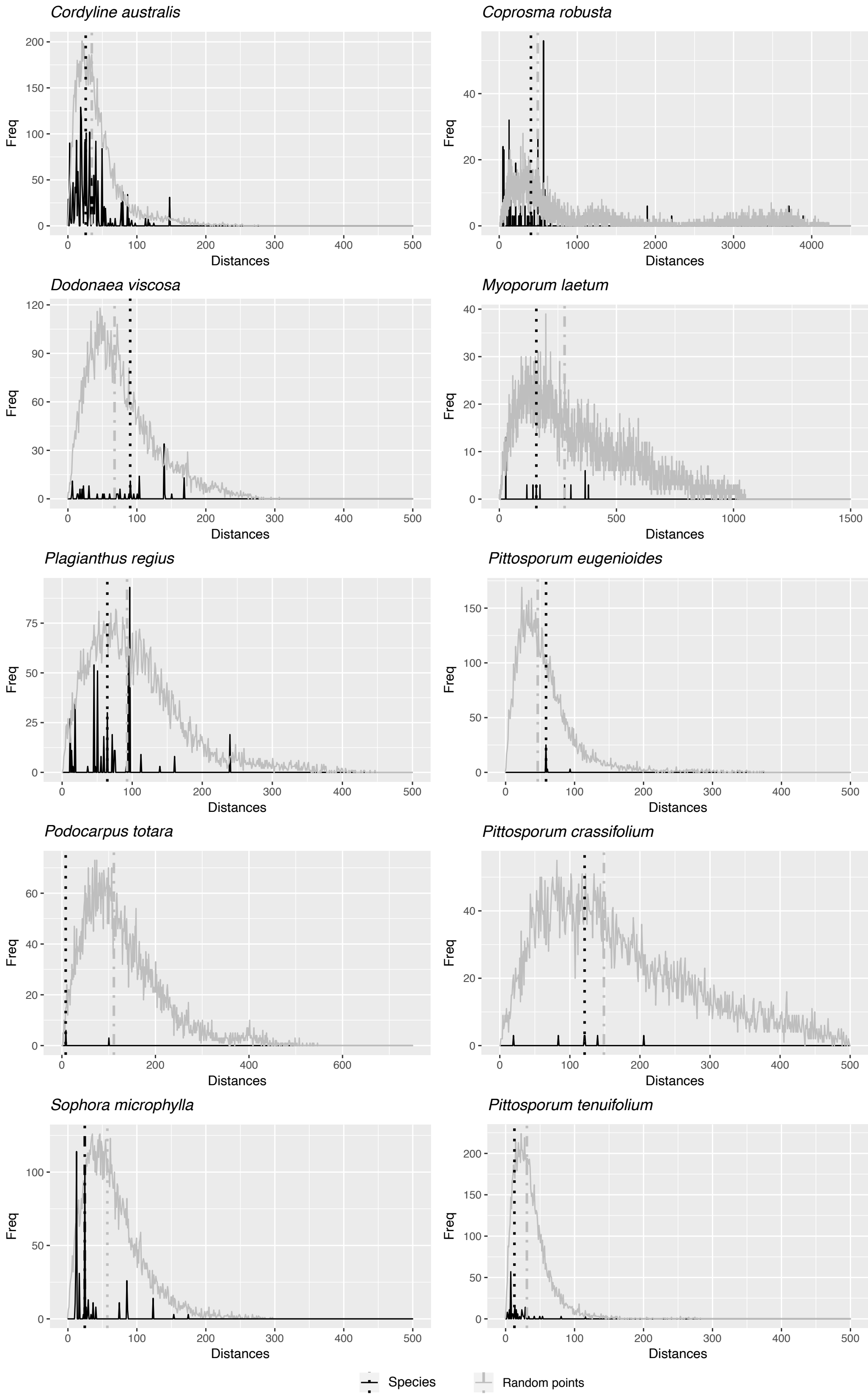


FIGURE 4.5: Seedling dispersal distances (black) and random points dispersal distance (grey) of native species. Vertical lines are the median distances.

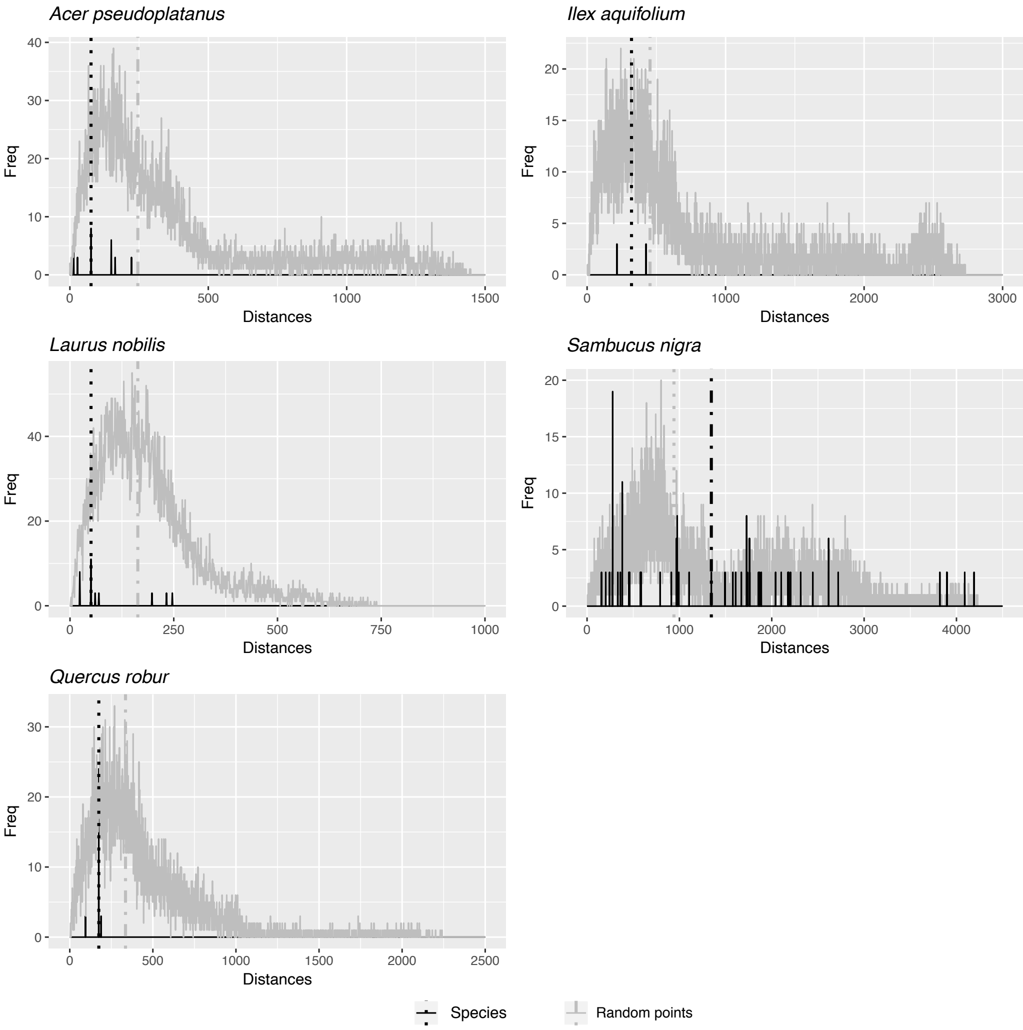


FIGURE 4.6: Seedling dispersal distances (black) and random points dispersal distance (grey) of exotic species. Vertical lines are the median distances.

Table 4.3: The t-test results comparing the distances of seedlings of native species and random points to the nearest conspecific adults. "TreeFreq" is the number of adult trees of each species mapped in the study area. "SeedlingsPlots" is the percentage of seedling plots that contained each species.

			Seedlings Dispersal Distances(m)		Points Dispersal Distances(m)				
Species	TreeFreq	SeedlingsPlots(%)	Average	Maximum	Average	Maximum	t	df	p-value
Coprosma robusta	8	9.063	510.2	3889	1019	4217	-12.3	380	<2.2e-6
Cordyline australis	2091	33.33	31.92	147.3	43.85	276.9	-17.2	3487	<2.2e-6
Dodonaea viscosa	463	5.529	84.39	168.9	78.72	306.4	1.39	172	0.167
Myoporum laetum	76	1.536	180.5	380.5	335.7	1051	-7.16	39.9	1.154E-08
Pittosporum crassifolium	98	0.923	115	205.7	174.5	633.9	-4.34	17.2	0.0004291
Pittosporum eugenioides	1007	0.461	62.13	93.03	56.81	374.7	2.79	33.6	0.008485
Pittosporum tenuifolium	2293	6.298	18.14	115.9	36.44	282.9	-13.5	2091	<2.2e-6
Plagianthus regius	358	7.373	73.2	239.8	105.4	446.4	-14.5	589	<2.2e-6
Podocarpus totara	177	0.461	38.93	100.6	130.1	546.3	-5.9	8.05	0.0003515
Sophora microphylla	593	5.837	31.32	174.7	65.74	296.3	-21.1	469.6	<2.2e-6

TABLE 4.4: The t-test results comparing the distances of seedlings of exotic (naturalised) species and random points to the nearest conspecific adults. Column labels follow Table 4.3.

			Seedlings Dispersal Distance(m)		Points Dispersal Distance(m)				
Species	TreeFreq	SeedlingsPlots(%)	Average	Max	Average	Max	t	df	p-value
Acer pseudoplatanus	56	1.229	97.54	222.68	369.6	1447	-20.3	32	<2.2e-16
Ilex aquifolium	18	0.307	320.6	425.6	729.4	2730	-8.61	5.22	0.0002812
Laurus nobilis	78	1.229	92.98	246.7	186.7	738.3	-6.58	33.5	1.625e-07
Quercus robur	29	0.768	168.1	188.5	417.7	2240	-44.23	65.25	<2.2e-16
Sambucus nigra	3	7.68	1430	4191	1284	4230	1.77	178	0.07923

Chapter 5

The rate and diversity of urban forest natural regeneration in Christchurch restoration sites

5.1 Introduction

Urban forests and trees are becoming critically important for urban biodiversity conservation and well-being of human societies (Botzat, Fischer, & Kowarik, 2016) by providing ecological, economic, social, health and aesthetic services (Pataki et al., 2011). Based on these functions of urban forests and trees, many cities start to restore degraded forests and try to make those native forests self-sustained (Sullivan et al., 2009; Stewart et al., 2004; B. Clarkson, Bryan, & Clarkson, 2012; Doroski et al., 2018). In recent decades, there has been a growing awareness of the importance of restoring degraded and lost native habitats both in New Zealand (McQueen et al., 2006; B. Clarkson & Meurk, 2004; Stewart et al., 2004; Sullivan et al., 2009) and elsewhere (Gillespie et al., 2017; Endreny et al., 2017; Ossola & Hopton, 2018; Guo, Morgenroth, & Conway, 2018; Guo et al., 2018).

Unlike most of other degraded lands, urban ecosystem has a lot of unique circumstances, such as frequent human disturbance (Rebele, 1994; Grimm, Grove, Pickett, & Redman, 2000), soil modification (Pavao-Zuckerman, 2008) and the invasive exotic species (Alston & Richardson, 2006; Cadotte, Yasui, Livingstone, & MacIvor, 2017). Several biotic and abiotic factors found in literature can limit the regeneration of forests which include canopy cover (Nakamura, Morimoto, & Mizutani, 2005), ground cover (Rawlinson, Dickinson, Nolan, & Putwain, 2004; Ruiz-Jaén & Aide, 2006), soil conditions like nutrients and moisture (Rebele, 1994; Pavao-Zuckerman, 2008; Oldfield et al., 2015), seeds dispersal (Dalling, Hubbell, & Silvera, 1998) as well as edge effect (Young & Mitchell, 1994; Murcia, 1995; Laurance et al., 2007; Harper et al., 2005).

The objective of ecological restoration is to expand native species habitat, recover native biodiversity and improve ecosystem function, and it is often measured by vegetation structure changes and species diversity (Ruiz-Jaen & Aide, 2005; Jackson & Hobbs, 2009). It is a common process in forest restoration that natural colonisation happens after the initial planting of early successional three species (McClain, Holl, & Wood, 2011). Previous studies have shown for native species restoration, it will be more helpful if restoration is located adjacent to a mature forest which can be a seed source (Jacquemyn, Butaye, & Hermy, 2003; MacKay, Wehi, & Clarkson, 2011), although this is often not possible. There are several factors can affect natural colonisation rate: site disturbance, species life history traits as well as the availability of seed sources (Wang & Smith, 2002). Seed dispersal mechanisms also is a key factor which can directly work on species distributions and natural vegetation recovery (Honnay, Bossuyt, et al., 2002).

Disturbance keeps the patches highly fragmented with reducing area, increasing isolation and a proliferation of edges (Kupfer, Malanson, & Franklin, 2006). These small and isolated patches have fewer plant species (Guirado, Pino, & Roda, 2006). Edges can create gradients of disturbance (Harper et al., 2005), availability of resource (Gehlhausen, Schwartz, & Augspurger, 2000), human activity (Guirado et al., 2006) and abundance of seeds (Cadenasso & Pickett, 2001). As a result of these influences, risks of native species extirpation and exotic species invasion will increase (Honnay, Verheyen, & Hermy, 2002; Guirado et al., 2006).

A lot of research has been done about the ecology of urban natural spaces in New Zealand. Glenn et al. examined the exotic and native woody species components in the urban area of Christchurch and also presented data of native and exotic seedlings regeneration in the city and discussed the potential for the expansion of native forest(Stewart et al., 2004). He also has several articles about Urban biotopes of New Zealand: from urban lawns to the residential and public woodlands(Stewart, Ignatieva, et al., 2009; Stewart, Meurk, et al., 2009). Jon et al. did a research about restoring native ecosystems in urban Auckland (Sullivan et al., 2009) and other countries(Crane & Kinzig, 2005; McDonnell & Pickett, 1990; Pickett et al., 2008).

In the last 150 years, Christchurch experienced widespread forest clearance, creating a period of open, largely tree-less landscape, followed by the growth of planted trees in parkland and gardens. However, although most of the pre-colonial species are endangered today, they still can be found in small fragmented patches (Christchurch City

Council, 2000). By the late 20th century, people had realised the importance of indigenous plants and started doing restoration work in the urban area. More than one million indigenous trees have been propagated and planted in Christchurch over the last decade and most of them have survived (Stewart et al., 2004). The oldest indigenous trees in gardens and parks, which were planted around a century ago, have also formed seed sources, and because of these, seeds dispersal and forest regeneration now occur in parts of the city (Stewart et al., 2004). The expectation has been that tree planting will be followed by the natural establishment of other native forest species and a return of natural forest ecological processes. Which forest species naturally colonise these restored sites, and how quickly this recovery occurs, are largely unknown and are now being researched.

However, the natural regeneration of urban forest has been a largely neglected research topic mostly because of the prevailing sense that urban forests need management. They are planted and then weeded to be kept tidy. It is important to understand the natural processes, including succession, that are happening in urban forests, as this will help people to manage them sustainably. Some of the main advantages of allowing natural regeneration in urban forests include maintaining the diverse genetics of locally indigenous plant populations, maintaining the natural distributions of species, matching plants with the most suitable sites, and creating natural plant densities and plant community structures (Mountford, Savill, & Bebber, 2006).

To examine the rate and diversity of urban forest natural regeneration, we surveyed the abundance and composition of natural regeneration in urban regenerated sites. This study will get the result of which species are regenerating most successfully in Christchurch urban forests. Several factors include site age, plant biostatus, soil, site management, canopy, adult tree presence, site vegetation area and width will be examined to see what factors can drive the seedling presence in the patch in the urban environment and if the seedling present in the patch what factors will have effects on the abundance. In addition, we will examine the seedling regeneration changes between survey in 2007 and 2015 and the factors impact on these changes.In this study, we will ask these questions:

1. which species are regenerating most successfully in urban area?
2. What factors have significant effects on seedling presence in the patch?
3. If the seedling present in the patch, what factors will impact on the seedling abundance?
4. What factors drive the seedling regeneration changes between 2007 and 2015?

5.2 Methods

5.2.1 Study Sites

Ten study sites in and near Christchurch city, New Zealand, were chosen for surveying in 2007 by Glenn Stewart and Jinbao Chang. Seven of them were located with the Christchurch city limits and three of them were in nearby towns of Lincoln and Rangiora (Figure 5.1). These sites were a selection of accessible ecological restoration projects that spanned a range of different sizes and ages. More details on the age, soils, and management of each site and include in section 5.3 below.

FIGURE 5.1: Ten ecological restoration sites in and near Christchurch city were surveyed in 2007–2008 and again in 2015–2016. The point colors help to distinguish the sites. Survey effort was approximately proportional to site area, with the size of each point proportional to the number of 5 m × 5 m quadrats that were surveyed at each site.

5.2.2 Vegetation survey

Sampling was done in ten urban restoration sites, all of which were planted with trees and have an understorey suitable for seedling growth. All sites surveyed in 2007 and 2008 by Jinbao Chang in a collaboration with Glenn Stewart and Colin Meurk. (unpublished data). I repeated this sampling at the same sites in the summer–autumn of 2015–2016 and new sites were surveyed at the same time.

At each site Chang placed 3–6 (mean 3.3) transects of typically eight consecutive 5 m × 5 m quadrats. Fewer transects and plots were made at the smallest sites. In each quadrat was recorded the mean canopy height, any evidence of human disturbance (e.g., walking tracks, evidence of weeding), drainage (on a scale of 1–5) and distance to forest edge. The percentage of ground cover was recorded that was woody plants, ferns, leaf litter, grasses, other herbaceous plants, and bare ground. The percentage canopy cover was estimated visually. For mature trees (free standing woody plants with a diameter>5cm), the diameters and abundance of each species were recorded. For saplings (diameter<5cm and height>1.4m), the abundance of each species were recorded. For seedlings (15cm<height <1.4m) and small seedlings (height <15cm), abundance per species was recorded.

For the tree diameter measurement, DBH (Diameter at Breast Height) was made at a height of 1.4m. If the tree was shorter than 1.4 m, its diameter was measured at the ground diameter instead of breast height. Only trees with DBH > 5 cm were recorded. If the tree had multiple trunks, the diameter of just biggest trunk was measured. All trees with DBH < 5 cm were recorded as saplings.

I redid the survey in 2015–2016. I used the same survey methods as Chang with the exception of measuring canopy cover using a fish eye lens adapter on my smart phone, taken at 1 meter height looking directly up with the phone always orientated to the north. Chang instead just visually estimated the percentage cover with his eyes.

Seedlings were assessed visually without digging them up. There could be mistakes also, such as the *Hydrangea*. The *Hydrangea* seedlings I recorded could be the small plants grow from rhizomes. Uncertain seedling identifications were checked by uploading photographs to iNaturalist NZ (`https://inaturalist.nz/people/adonis_wei`).

5.3 Data sources

5.3.1 Site age

Restoration plantings were identified spanning 20-100 years since initial planting so that we could document succession of urban forest regeneration (Table 5.1). Because some restoration plantings were made under established (exotic) tree canopies, we made two separate estimates of site age: canopy age and native restoration age. These data came from different places. The history of Matawai Park came from Waimakariri District Council website (`https://www.waimakariri.govt.nz/leisure-and-recreation/facilities/`

parks-and-playgrounds). The information of Wigram Retention Basin, Ashgrove Reserve, Ashgrove beech and Ernle Clarke Reserve came from Christchurch City Council website (https://cccgovtnz.cwp.govt.nz/parks-and-gardens/explore-parks/). Ilam garden A and B have records in the garden history book by (Strongman, 1984). For Christchurch Botanic Garden, I talked with the staff to get some historical information. Lincoln University Orchard Car Park information was from the staff's personal experience. Liffey Domain details were in the reserve's management plan by Selwyn District Council (Selwyn District Council, 2007).

Table 5.1: Maximum canopy ages and maximum native restoration ages of the ten study sites. Ilam Garden A is the ornamental garden part of the grounds and Ilam Garden B is planted native forest. Ages were calculated until 2016.

Site name	**Canopy Age(year)**	**Native Restoration Age(year)**	**Quardrats (Jinbao)**	**Quardrats (Wei)**
Matawai Park	38	38	32	32
Wigram Retention Basin	18	18	19	19
Liffey Domain	100	15	24	24
Ashgrove Reserve	90	90	24	24
Ashgrove beech	90	90	8	8
Ilam garden A	<100	<100	15	16
Ilam garden B	<100	<100	8	8
Lincoln Car Park	14	14	25	25
Botanic Garden	<100	<100	28	28
Ernle Clark Reserve	100	10	32	36

5.3.2 Plant biostatus data

I use four types of plant biostatus in this chapter: 'Native to Christchurch', 'Non-native to Christchurch', 'Naturalised' and 'Unnaturalised'. New Zealand plant biostatus data came from the New Zealand Organisms Register (http://www.nzor.org.nz). See section 2.2.3 for details. The tree map data and seedling data were merged with the plant biostatus database in R (R Core Team, 2016).

5.3.3 Soil

In addition to the on-site qualitative soil drainage assessment made by Chang, I also used available soil map data including soil drainage, depth to hard soil and soil moisture. These came from the S-map, a new digital soil spatial information system for New

Zealand created by Manaaki Whenua-Landcare Research (`https://smap.landcareresearch.co.nz`).

5.3.4 Site management

The study sites are managed by different councils and organisations and managed by different people. Consistent and accurate historical data on the past management of each site could not be obtained. Instead, while planning and doing my surveys I spoke with staff and volunteers managing each site about their management for each sites. To summarise these details, I created a set of categories for how sites were managing exotic and native species (Table 5.2 and Table 5.3).

Table 5.2: Categories for the broad types of exotic species management at sites.

Code	Management for exotic plants	Code	Management for exotic seedlings
A	Never plant any garden plants	a	Often remove exotic seedlings
B	Plant some garden plants	b	Sometimes remove exotic seedlings
C	Plant all garden plants	c	Never remove any exotic seedlings

Table 5.3: Categories for the broad types of native species management at sites.

Code	Management for native plants	Code	Management for native seedlings
D	Never plant any new native plants	d	Never remove any native seedlings
E	Plant native plants sometimes	e	Remove/cut some native seedlings
		f	Remove/cut most/all native seedlings

Table 5.4: How exotic and native species are managed at each study site. Ilam Garden A is the garden part and Ilam Garden B is native bush. "Exoplants" means management for exotic plants. "Exoseedlings" means management for exotic seedlings. "Natplants" means management for native plants. "Natseedlings" means management for native seedlings.

Site name	Exoplants	Exoseedlings	Natplants	Natseedlings
Matawai Park	A	b	D	d
Wigram Retention Basin	A	b	D	d
Liffey Stream	B	b	D	e
Ashgrove Reserve	B	b	D	d
Ashgrove beech	A	b	D	e
Ilam garden A	B	b	D	f
Ilam garden B	A	b	D	e
Lincoln Car Park	B	c	D	d
Botanic Garden	A	a	E	d
Ernle Clarke Reserve	B	b	E	d

5.3.5 Site area and width

All site areas and widths were measured in Google Earth Pro (version 7.3.2.5491 2018.07). Site area only included the contiguous areas of tree canopy cover, excluding grass clearings. The widest section of contiguous tree canopy cover was measured as the site width.

5.4 Analysis

5.4.1 PCA for soil factors

Principal Component Analysis (PCA) was used in R (R Core Team, 2016) to find out how similar the different soil factors were and whether all needed to be used (Tables 5.5, 5.6, 5.7, 5.8). The result suggests that half of the soil variation (51.27%) can be captured by the first axis (Table 5.9). For the first PCA axis, soil moisture correlates 0.88. For the second PCA axis, which explains a further 44.39%, is mostly soil drainage (correlation -0.905). Therefore, soil drainage and soil moisture were choose for our model.

Table 5.5: Distribution of different soil moisture levels.

Soil Moisture	**Very High**	**High**	**Moderate to high**
Number	647	2976	355

Table 5.6: Distribution of different soil drainage levels.

Soil Drainage	**Well Drainage**	**Moderate Well**	**Poor Drainage**
Number	1675	193	2110

Table 5.7: Distribution of different depth to hard soil levels.

Depth To Hard Soil	**Deep**	**Shallow**
Number	3623	355

Table 5.8: Result of rotation

	PC1	**PC2**	**PC3**
Soil Drainage	0.3588110	-0.9049625	0.2286865
Depth To Hard Soil	-0.3213294	-0.3497831	-0.8799995
Soil Moisture	-0.8769573	-0.2422698	0.4162971

Table 5.9: Importance of components

	PC1	PC2	PC3
Standard deviation	0.5439	0.5061	0.15836
Proportion of Variance	0.5127	0.4439	0.04346
Cumulative Proportion	0.5127	0.9565	1.00000

5.4.2 Seedling regeneration model

Several models were tested in R (R Core Team, 2017) with package 'lime4' (Bates, Sarkar, Bates, & Matrix, 2007). All of them with plot and species variables, and each with only one set of site variables, such as age, size, management or soil. They are site_plotonly, site_soil, site_agearea and site_management and a simple model just with canopy and biostatus variables. Five models were tested just with one variables. They are justlitterdepth, Ground.cover.litter, justbiostatus, justcanopy and justtree. These models were running for analysing the association between seedling presence in quadrats and different factors. The parameter family was binomial and models were compared based on their AICc values. Package 'car' (Fox et al., 2007) was used for an Anova of a glmer with P-values.

5.4.3 Seedling regeneration-change model

Several models were tried with package 'lme4' (Bates et al., 2007) in R (R Core Team, 2017). Species model has only biostatus variable. Site model has only site variable. Plot model just has plot variable. Plot.interactions model has plot information and interactions. Presence model has no interactions while presence.interactions has interactions. Plot.biostatus.interactions and plus age and area also were tested. which contained several factors as well as the interactions between different factors. These models were running for analysing the association between seedling presence in quadrats and different factors and interactions. Package 'car' (Fox et al., 2007) was used for an Anova of a glmer with P-values.

In order to make sure what determines how abundant a species is in a plot, a count model was tested with year, biostatus, canopy age, restoration age, patch width, site vegetation area, adult tree presence and canopy cover. The parameter family was poisson.

5.4.4 Ordination analysis

To determine community level responses to the effects of the environmental factors and human management, species composition of each site was assessed using non-metric multidimensional scaling (nMDS). The package 'vegan' (Oksanen et al., 2019) was used in R to analysis the ordination analysis on seedling data. Bray-Curtis with no wisconsin transformation was used as the coordinate system in the ordination analysis.

After examining a plot of minimum stress versus number of dimensions, a four-dimensional NMDS ordination was required to adequately summarise the data. All four axes of the 4D solution (stress = 0.149) can represent meaningful trends in species composition with different environmental factors.

In order to test the significance of how much the site variables affect the seedling composition per plot, an adonis_model was running in R (R Core Team, 2017) with site vegetation area, site width, canopy age, restoration age younger than canopy, both exotic and native management, soil drainage, canopy cover, ground cover litter, litter depth. Adonis model was running for more permutations (9999) and the method is bray.

5.5 Results

5.5.1 Species composition

Adult tree species richness did not change much during the eight years between surveys (Table 5.10). In both surveys, about half of the planted trees and about half of the wild seedlings were of local native species.

I found fewer native (both local native and non-local native) and naturalised species of adult tree in my 2015–2016 survey than Chang found in 2007–2008, but two more exotic species (Table 5.10). As recorded in Table A.1, most of the local native species that I did not find in 2015 are not common in the Christchurch urban area, such as *Phyllocladus trichomanoides*, *Prumnopitys ferruginea*(miro), *Olearia paniculata* (akiraho) and *Coriaria arborea*(tutu). There could be several reasons, mostly these plants were planted in somewhere in the city like Christchurch Botanic Garden. However, I have checked the species records from iNaturalist NZ[1] and most of records are not in the survey sites. Chang collected no specimens and took no photos so it is possible that there were some identification mistakes (although he worked alongside local botanists in most of his surveys).

In contrast to adult tree species, more seedlings species were found in the 2015–2016 survey than the 2007–2008 survey (Table 5.11). This increase was greatest for the

[1]iNaturalist NZ, https://inaturalist.nz, is a species identification system and an organism recording tool.

Table 5.10: Overall tree species richness (percentage) for each biostatus in each survey.

	Local native	**Non-local native**	**Naturalised**	**Exotic**	**Total**
Quan's survey(2015)	31(50%)	14(22.58%)	11(17.74%)	6(9.68%)	62
Chang's survey(2007)	40(53.33%)	17(22.67%)	14(18.67%)	4(5.33%)	75

Table 5.11: Overall seedling species richness (percentage) for each biostatus in each survey. There are no seedlings in in the "Exotic" category from Table 5.10 since all wild exotic species are in the "Naturalised" category.

	Local native	**Non-local native**	**Naturalised**	**Total**
Quan's survey(2015)	50(48.54%)	19(18.44%)	34(33%)	103
Chang's survey(2007)	47(55.95%)	18(21.43%)	19(22.62%)	84

naturalised species. A few species only were recorded one time or two times. For example, *Berberis* spp, *Celastraceae* spp and *Cotoneaster* spp.

Comparing the data of 2007 and 2015, I found more non-local native individuals but less naturalised individuals than 2007 as well as exotic individuals. The local native individual percentages were similar as 2007. For seedlings, the result changed from 2007 to 2015. Both of the local native and naturalised individual percentages went down while the non-local native individual percentage went up.

I found small percentage non-local native individual trees (8.91%) and naturalised individual trees (5.41%) (Figure 5.2) made lots non-local seedlings (19.75%) and naturalised seedlings (33.45%) in 2015 (Figure 5.3). In Chang's survey data (2007), the percentage of naturalised individual trees was 6.1% (Figure 5.4) and the percentage of naturalised seedlings was 42.21% (Figure 5.5). Some naturalised species were popular/abundant trees in some of our survey sites and made a good number of seedlings, such as *Acer pseudoplatanus*(Sycamore), which had most seedlings both in 2007 survey and 2015 survey.

5.5.2 Summary of the regeneration model

We ran several combined-factor models and simple-factor models to find the best mode. As showing in Table 5.12, site-simple model has the smallest AICc number which is the best model for this analysis. The site-simple model result shows that tree presence, canopy cover and biostatus these three factors affect on seedling regeneration 5.13.

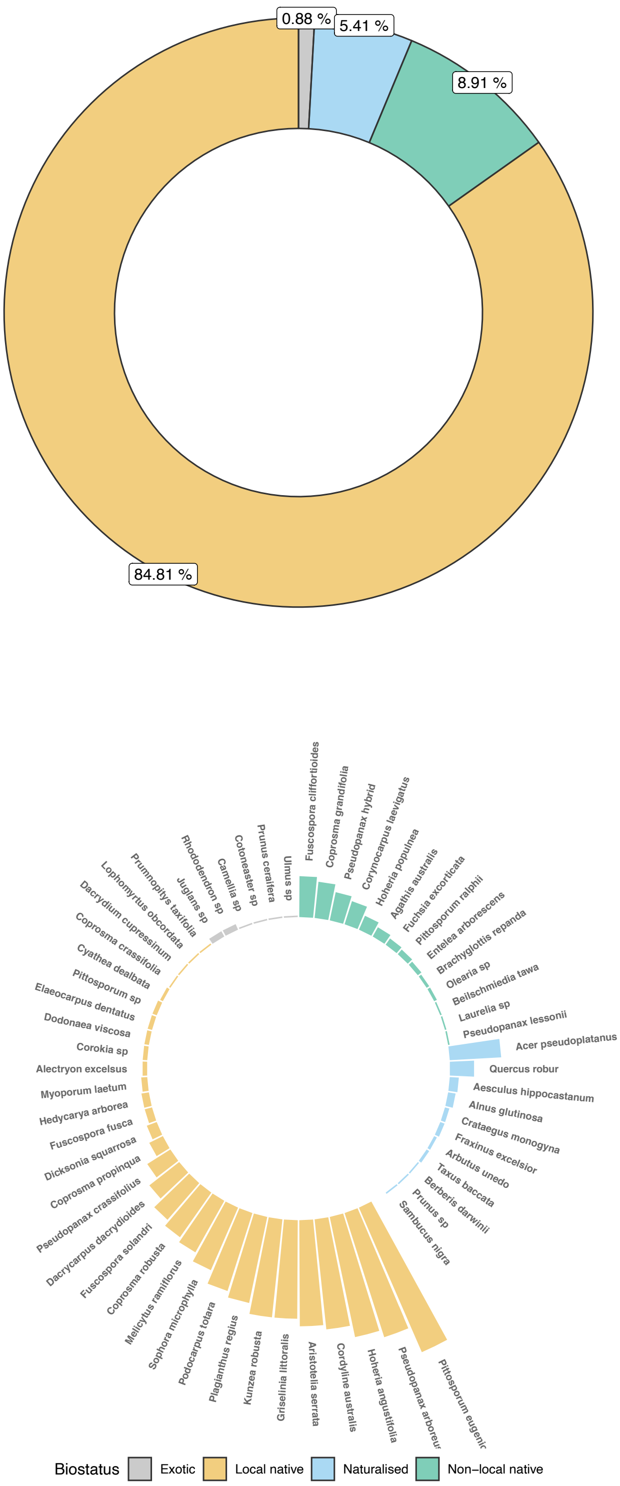


FIGURE 5.2: Composition of individual trees with different biostatus surveyed in 2015. The top one shows the percentage of individual seedlings with different biostatus. The bottom one shows the orderings of individual seedlings by biostatus.

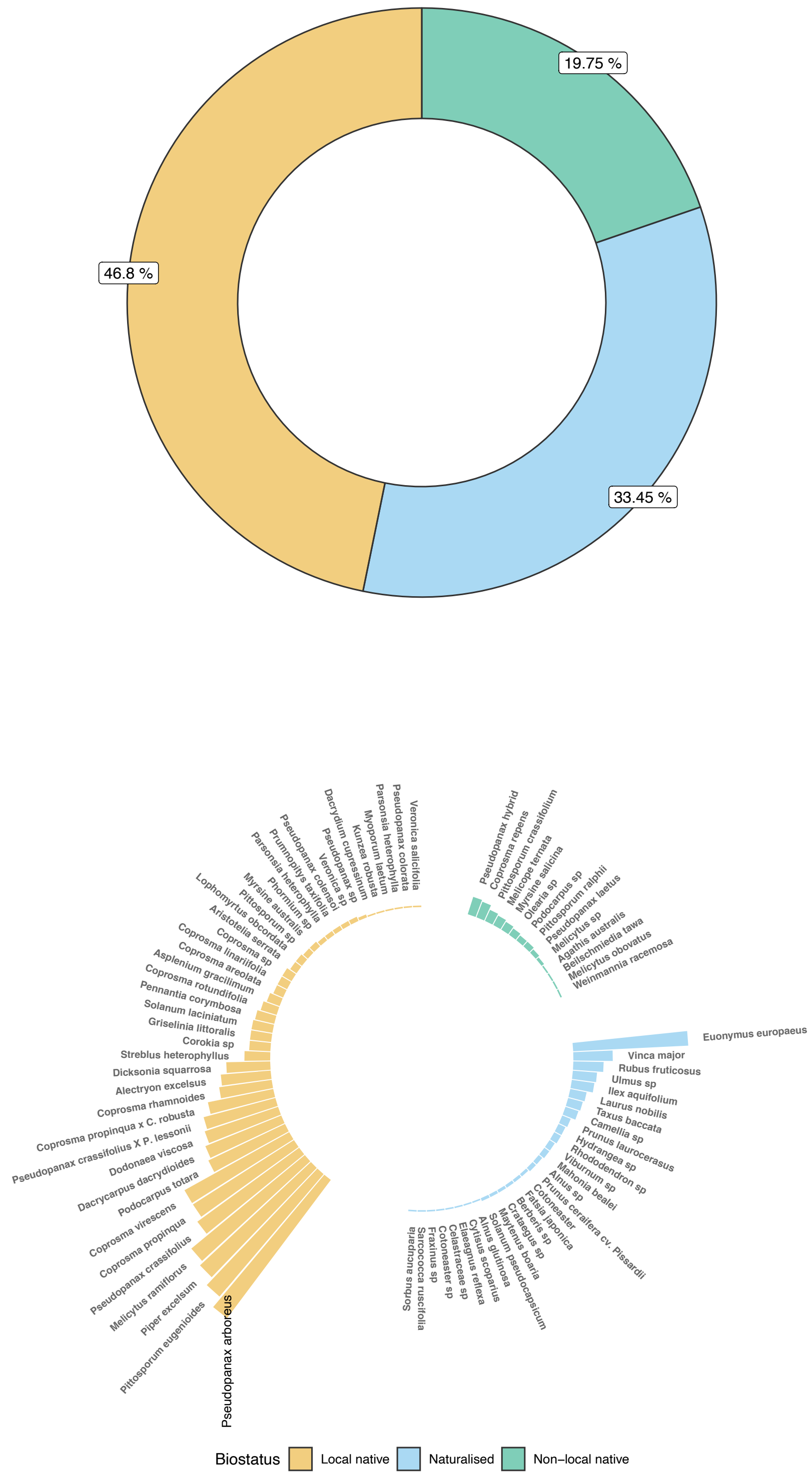


Figure 5.3: Composition of individual seedlings with different biostatus surveyed in 2015. The top one shows the percentage of individual seedlings with different biostatus. The bottom one shows the orderings of individual seedlings by biostatus.

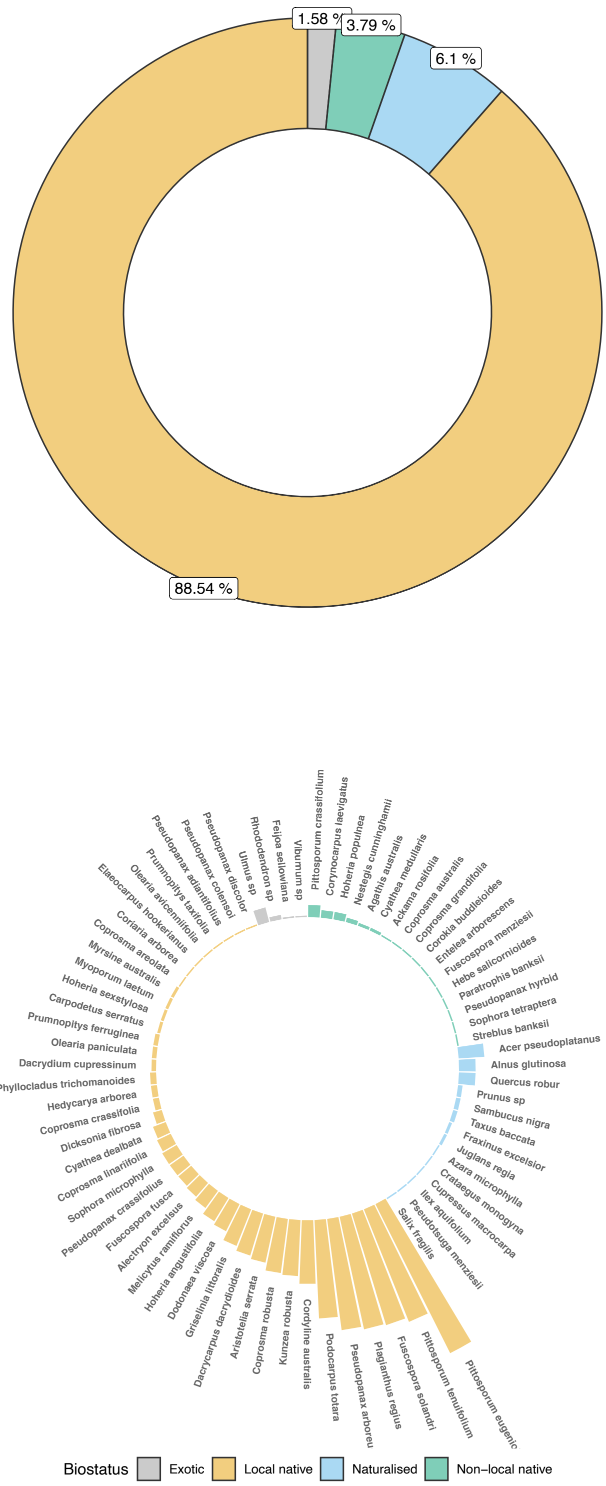


FIGURE 5.4: Composition of individual trees with different biostatus surveyed in 2007. The top one shows the percentage of individual trees with different biostatus. The bottom one shows the orderings of individual trees by biostatus.

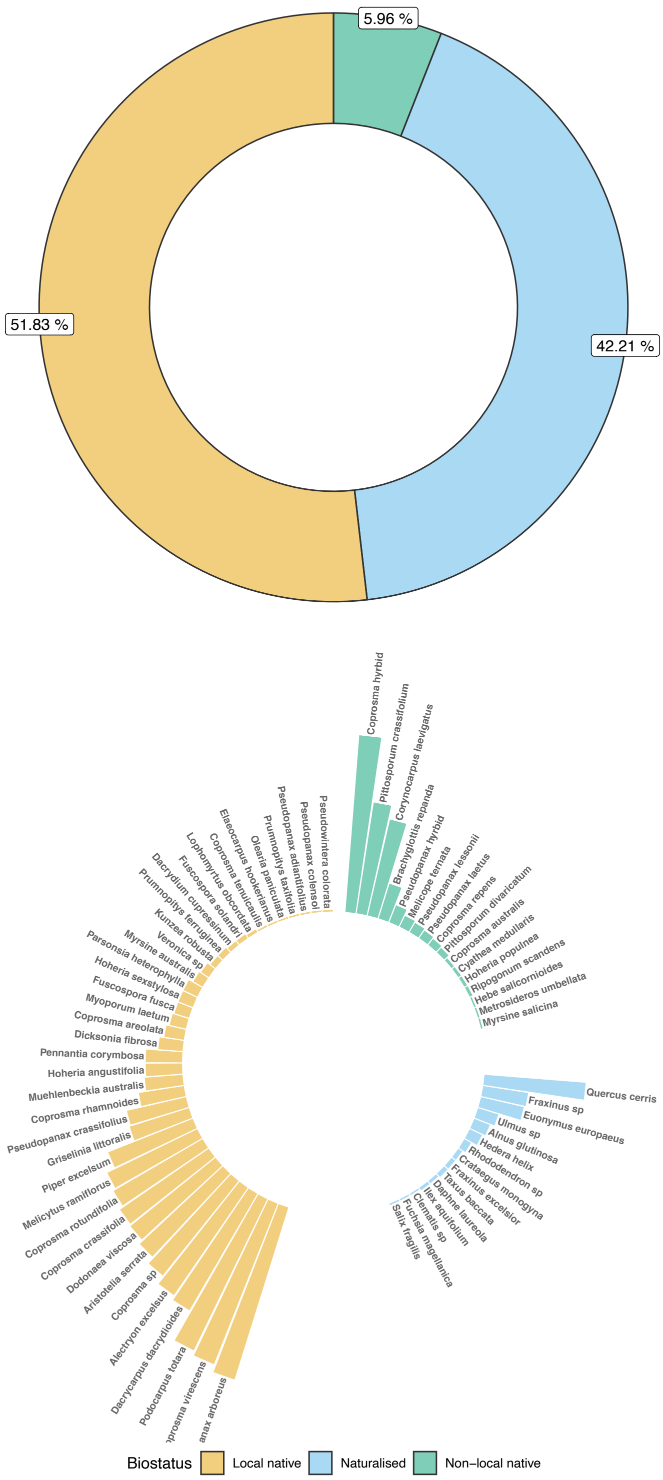


FIGURE 5.5: Composition of individual seedlings with different biostatus surveyed in 2007. The top one shows the percentage of individual seedlings with different biostatus. The bottom one shows the orderings of individual seedlings by biostatus.

TABLE 5.12: AICc numbers of different combine-factor and simple-factor models. The result shows simple model has the lowest AICc number and it is the best model to show what factors effect on seedling regeneration in urban regeneration sites.

Model name	AICc	Delta_AICc	AICcWt	Cum.Wt
justlitterdepth	7998.34	137.21	0.00	0.00
Ground.cover.litter	7997.75	136.63	0.00	0.00
justbiostatus	7994.35	133.23	0.00	0.00
justcanopy	7990.02	128.90	0.00	0.00
site_management	7872.09	10.97	0.00	0.00
justtree	7870.57	9.45	0.01	0.01
site_soil	7867.43	6.30	0.03	0.04
site_agearea	7865.68	4.56	0.08	0.12
site_plotonly	7864.69	3.57	0.13	0.25
simple	7861.12	0.00	0.75	1.00

TABLE 5.13: Anova of model Site_simple. Signif.codes:0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' 1

	Chisq	Df	Pr(>Chisq)	
Tree.presence	134.182	1	<2.2e-16	***
Canopy.cover	10.270	1	0.001352	**
Biostatus	25.409	3	1.268e-05	***

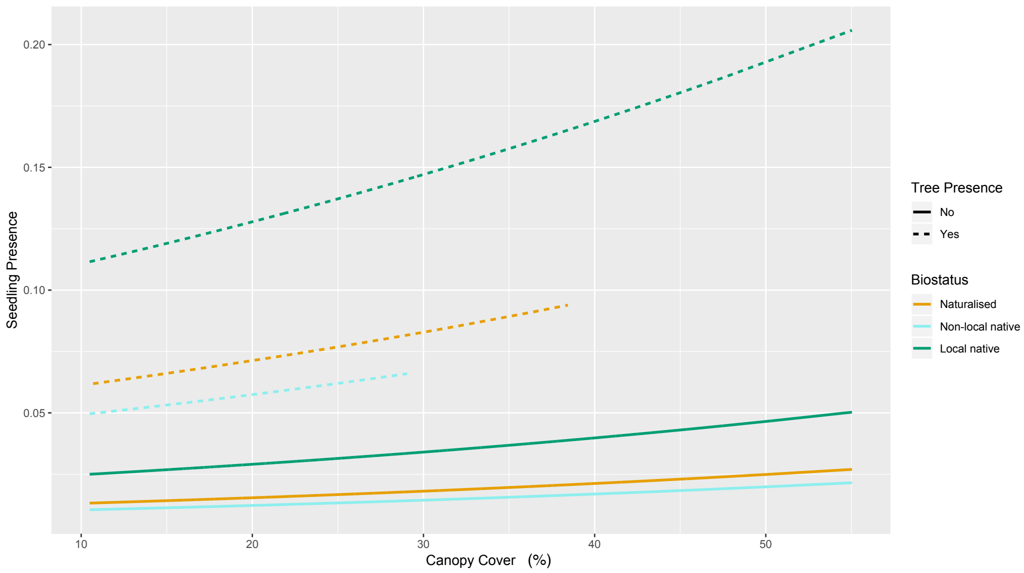


FIGURE 5.6: The effects of canopy cover on seedling presence at native restoration sites. The lines are the predictions from the model (Table 5.13). Shown are the trends for species with different biostatus, with and without conspecific trees present in the same quadrat.

The presence of seedlings of different species increased as the canopy cover increased (Fig. 5.6). This was regardless of the presence of adult conspecific trees, and biostatus.

Under the same canopy, among three biostatus, local native species had the highest level of seedling presence, followed by naturalised species.

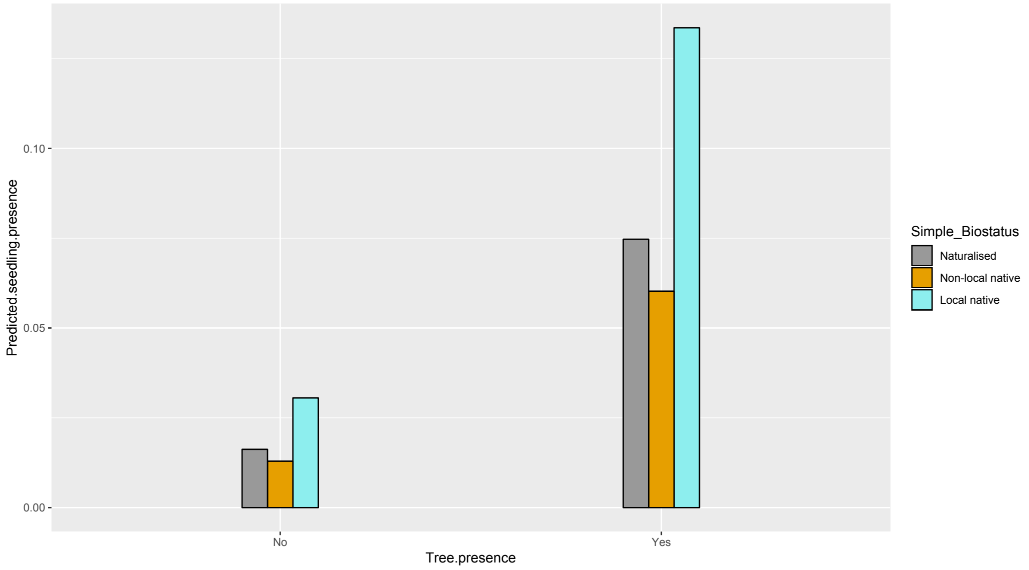


FIGURE 5.7: Prediction of tree presence effects on seedling presence.

Not surprisingly, the presence of adult trees significantly and substantially increased the probability of finding conspecific seedlings (Fig. 5.7). Across all three biostatus categories, seedling presence was more likely when the adult conspecific trees were in the quadrat.

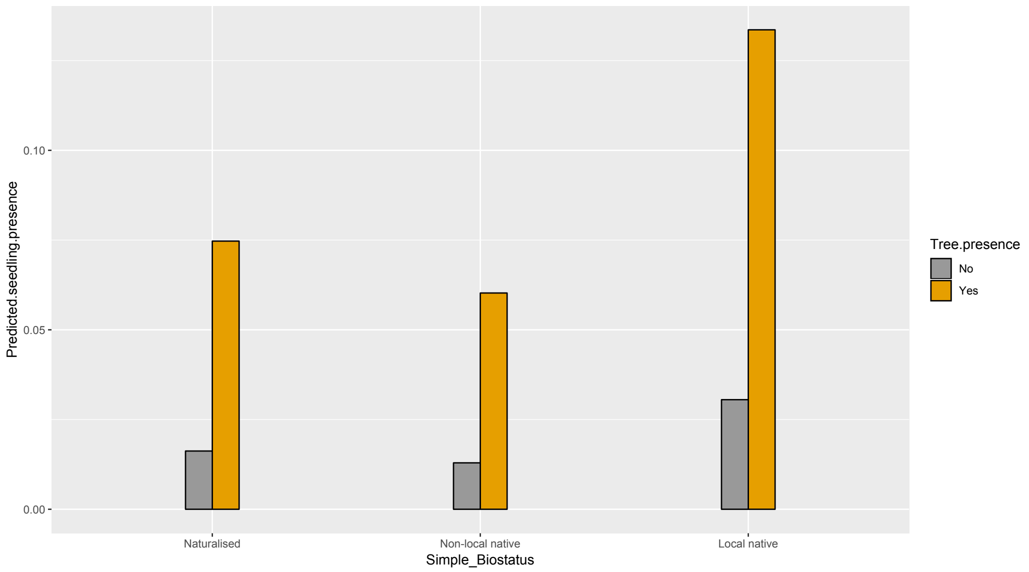


FIGURE 5.8: Prediction of tree biostatus effect on seedling presence.

Local native species were most likely to be present as seedlings, followed by naturalised species, then non-local natives (Fig. 5.8). This was true regardless of whether or not conspecific adult trees were in the same quadrats.

5.5.3 Summary of the regeneration change model

TABLE 5.14: AICc numbers of different models show presence.interactions has the lowest AICc number and it is the best model to analysis the seedling regeneration changes between two survey data.

Model name	AICc	Delta_AICc	AICcWt	Cum.Wt
species	-3214.37	1269.54	0	0
site	-3386.99	1096.92	0	0
plot	-3854.01	629.90	0	0
plot.interactions	-3872.33	611.58	0	0
presence	-3999.67	484.24	0	0
plot.biostatus.interactions	-4038.13	445.78	0	0
plot.biostatus.interactions.age	-4081.49	402.42	0	0
plot.biostatus.interactions.area	-4164.54	319.37	0	0
presence.interactions	-4483.91	0.00	1	1

As shown in Table 5.14, the model_all_presence_interactions which include all factors and interactions has the minimum AICc number. In the deviance analysis result, four single factors have a significant impacts on seedling regeneration changes: first subdivison year, minimum native restoration age, site width and adult tree presence. The interactions between some factors also have significant effects on seedling regeneration too 5.15.

As we expected, survey year is one of the most important factors for the seedling regeneration changes between 2007 and 2015. From Table 5.15, it shows biostatus is not the factor effect on seedling regeneration changes but the interactions between it and other factors have significant effects on seedling regeneration changes.

Generally speaking, adult tree presence has a positive effect on seedling presence. From Figure 5.9, for the species which have adult trees in the plots, more seedlings are found in 2015 than 2007. However, the species without adult trees in the plots have different results. More local native seedlings were found in 2007 while more naturalised seedlings were found in 2015 (Table 5.15). Non-local native species have similar seedling presences of this two years.

Site width has a significant influence on the seedling presence too (Table 5.15). For local native species, more seedlings can be found in the wider patches while for non-local native and naturalised species, the seedling presence increase in the young native restoration sites and decrease in the old native restoration sites.

Table 5.15: Anova of model_all_presence_interactions. Significance codes: '***' P <0.001, '**' P <0.01, '*' P <0.05, '.' P <0.1

	Chisq	Df	Pr(>Chisq)
Year	11.5869	1	0.0006642 ***
Simple_Biostatus	2.5702	2	0.2766274
min_canopy_age	1.8807	1	0.1702563
min_native_restoration_age	10.6640	1	0.0010924 **
width	36.8389	1	1.283e-09 ***
Site.vegetation.area	0.0546	1	0.8152698
Tree.presence	603.4087	1	<2.2e-16 ***
Canopy.cover....	0.7426	1	0.3888246
Year:Simple_Biostatus	64.5348	2	9.693e-15 ***
Year:min_canopy_age	11.0621	1	0.0008811 ***
Year:min_native_restoration_age	0.0988	1	0.7532983
Year:width	0.6486	1	0.4206167
Year:Site.vegetation.area	2.9080	1	0.0881428 .
Year:Tree.presence	3.7674	1	0.0522615 .
Year:Canopy.cover....	11.1262	1	0.0008512 ***
Simple_Biostatus:min_canopy_age	123.8395	2	<2.2e-16 ***
Simple_Biostatus:min_native_restoration_age	91.5102	2	<2.2e-16 ***
Simple_Biostatus:width	84.4017	2	<2.2e-16 ***
Simple_Biostatus:Site.vegetation.area	57.0396	2	4.112e-13 ***
Simple_Biostatus:Tree.presence	112.4186	2	<2.2e-16 ***
Simple_Biostatus:Canopy.cover....	0.9520	2	0.6212573
min_canopy_age:min_native_restoration_age	6.4780	1	0.0109220 *
min_canopy_age:width	3.4779	1	0.0621926 .
min_canopy_age:Site.vegetation.area	20.3399	1	6.484e-06 ***
min_canopy_age:Tree.presence	6.1959	1	0.0128043 *
min_canopy_age:Canopy.cover....	13.1880	1	0.0002817 ***
min_native_restoration_age:width	5.9313	1	0.0148743 *
min_native_restoration_age:Site.vegetation.area		0	
min_native_restoration_age:sTree.presence	20.9853	1	4.628e-06 ***
min_native_restoration_age:Canopy.cover....	14.7827	1	0.0001206 ***
width:Site.vegetation.area	7.6169	1	0.0057823 **
width:Tree.presence	3.9994	1	0.0455175 *
width:Canopy.cover....	1.0672	1	0.3015864
Site.vegetation.area:Tree.presence	1.1499	1	0.2835699
Site.vegetation.area:Canopy.cover....	2.8935	1	0.0889381 .
Tree.presence:Canopy.cover....	1.9655	1	0.1609268

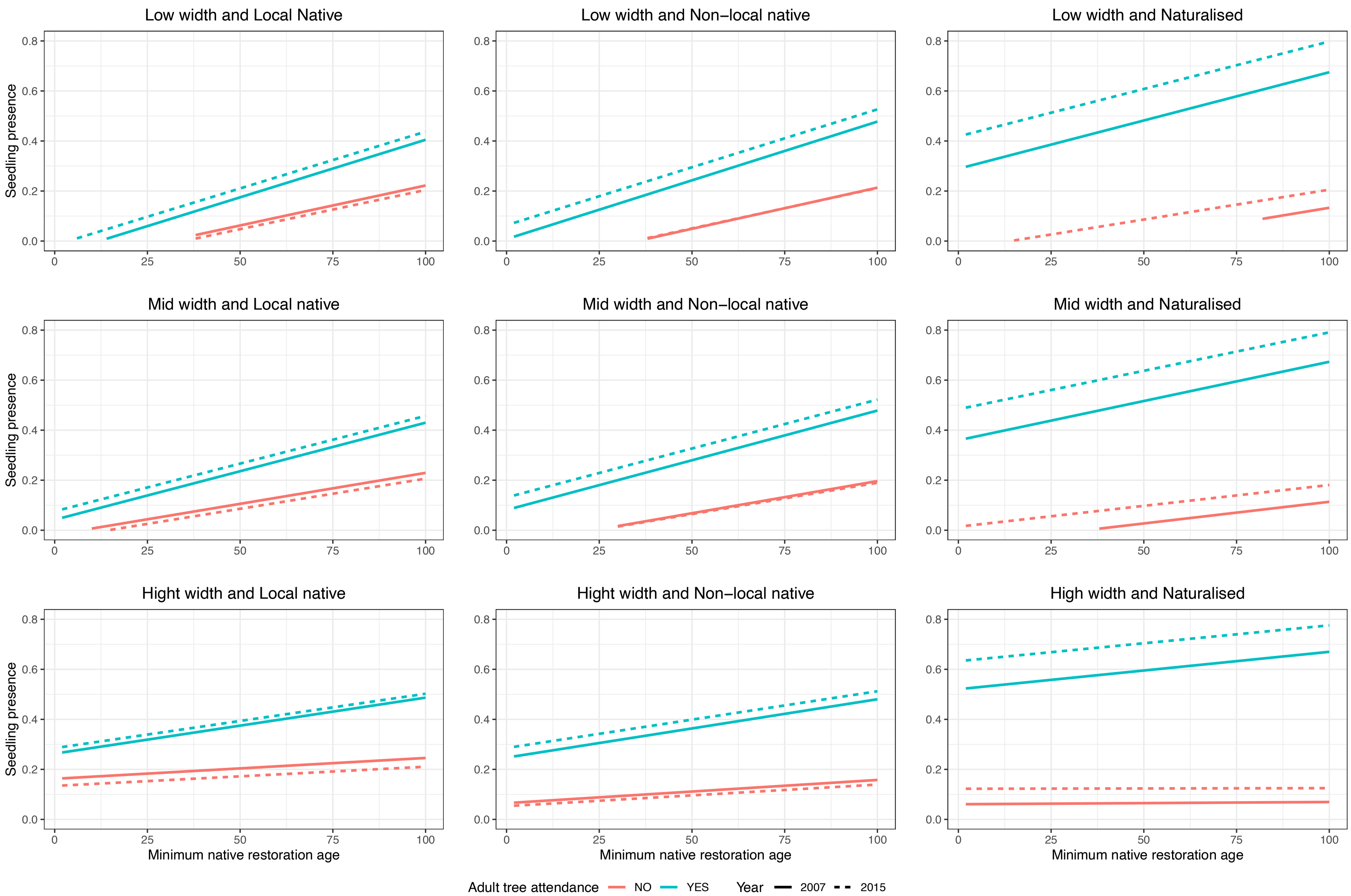


FIGURE 5.9: Different levels of factors affecting seedling presence for species in quadrats. Ranges of values representing the data were selected to plot the model predictions.

For different biostatus, in narrow sites, naturalised species have the highest seedling presence which followed by non-local native ones. Local native species have the lowest seedling presence. However, in wider sites, when there are no adult trees in plots, local native species has the highest seedling presence.

Minimum native restoration age is another important factor in the prediction. In the young restoration sites, the presence of the adult trees is more important for seedling presence. In the old restoration sites, seedling presence will drop down, especially for natualised species.

5.5.4 Summary of Seedling count model

Table 5.16: Anova of model_all_Count. Signif.codes:0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' 1

	Chisq	Df	Pr(>Chisq)
Year	0.6365	1	0.42499
Simple_Biostatus	2.4596	2	0.29236
scale(min_canopy_age)	5.6530	1	0.01743 *
scale(min_native_restoration_age)	0.0395	1	0.84250
scale(width)	1.5874	1	0.20770
scale(Site.vegetation.area)	0.2481	1	0.61844
scale(Tree.presence)	5.2678	1	0.02172 *
scale(Canopy.cover....)	0.8221	1	0.36456

Two factors had significant effects on seedling abundance: minimum canopy age and tree presence (Table 5.16). Adult tree presence has a positive effect on seedling count and minimum canopy age has a negative effect on seedling count (Figure 5.10). Less seedlings can be found in the quadrats with old canopy ages.

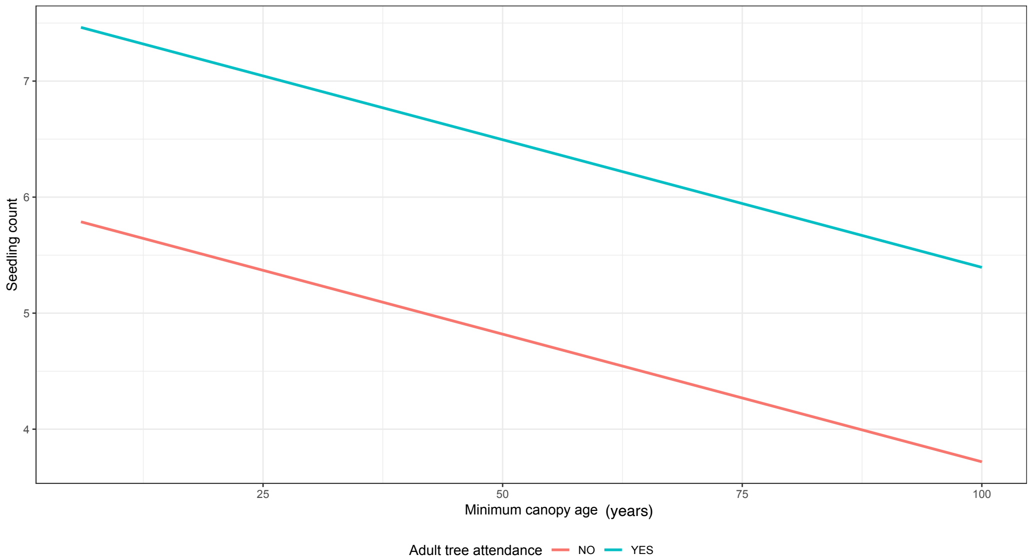


FIGURE 5.10: Important factors showing in the model determine how abundant the seedlings are.

5.5.5 Ordination analysis

5.5.5.1 The result of Adonis analysis

TABLE 5.17: Result of Adonis analysis

	Df	**SumsOfSqs**	**MeanSqs**	**F.Model**	**R^2**	**Pr(>F)**
Site.vegetation.area	1	5.716	5.7161	21.3500	0.07087	0.0001 ***
width	1	4.388	4.3882	16.3904	0.05441	0.0001 ***
min_canopy_age	1	4.596	4.5964	17.6181	0.05699	0.0001 ***
restoration.age	1	3.862	3.8620	14.4249	0.04789	0.0001 ***
exoticplantmanage	1	1.984	1.9843	7.4116	0.02460	0.0001 ***
exoticweedmanage	2	4.253	2.1265	7.9427	0.05273	0.0001 ***
nativeplantmanage	1	1.300	1.3005	4.8574	0.01613	0.0001 ***
nativeseedlingmanage	1	1.299	1.2994	4.8533	0.01611	0.0001 ***
canopy.cover	1	0.851	0.8513	3.1796	0.01056	0.0001 ***
Ground.cover.litter	1	0.608	0.6079	2.2694	0.00753	0.0001 ***
littler.depth	1	0.655	0.6549	2.4460	0.00812	0.0002 ***
Residuals	191	51.137	0.2677		0.63405	0.0034 **
Total	203	80.650			1.00000	0.0022 **

As shown in Table 5.17, all factors have significant influences on seedling composition (Pr>0.005). R^2 values indicate the strength of the effects.

5.5.5.2 Site vegetation area

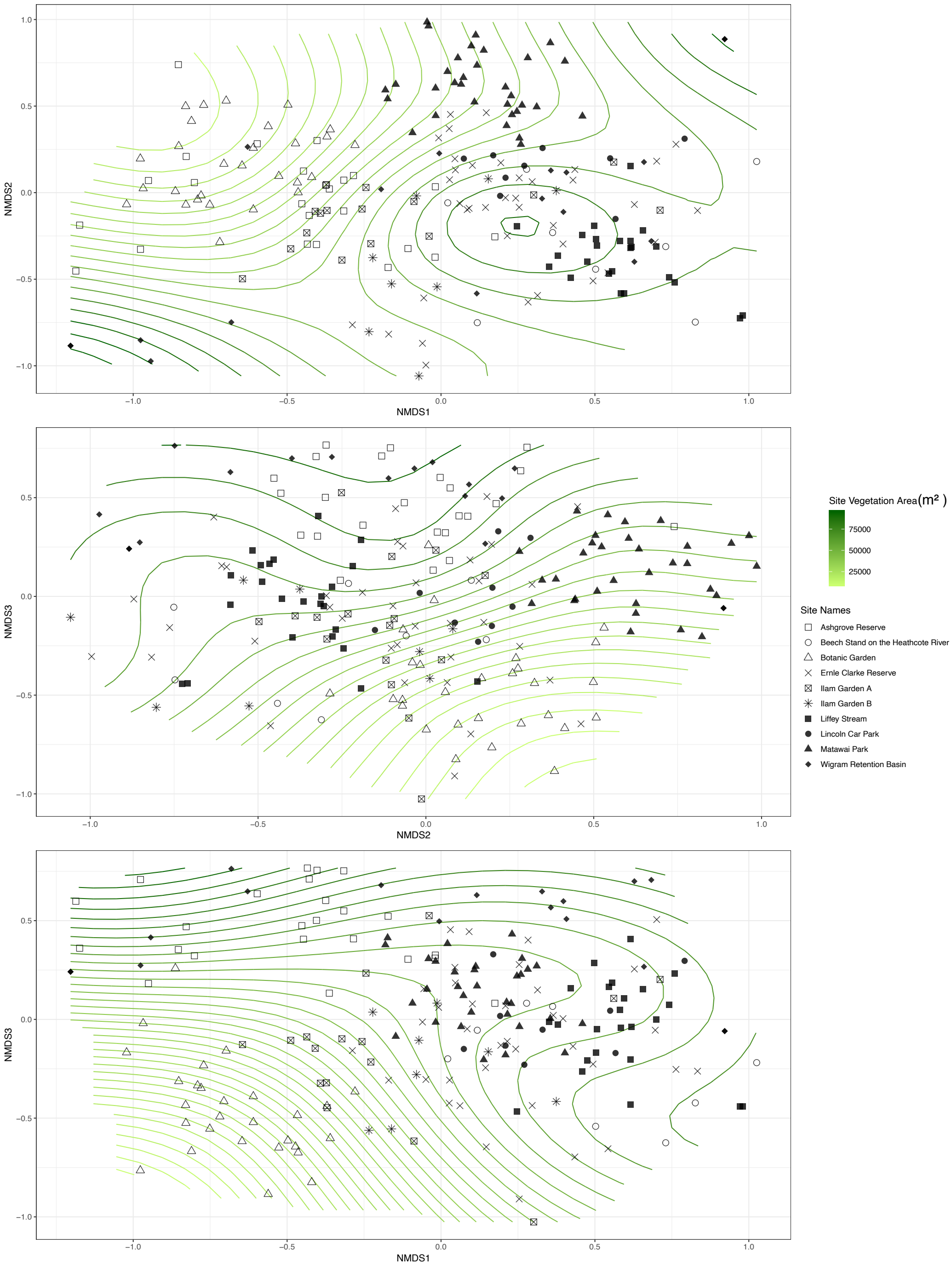


FIGURE 5.11: NMDS ordination in four dimensions, showing sites based on the site vegetation area gradients with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Site vegetation area ranges from 2500 to 7500 m^2. Liffey Stream and Matawai Park sites show a certain degree of clustering.

5.5.5.3 Width

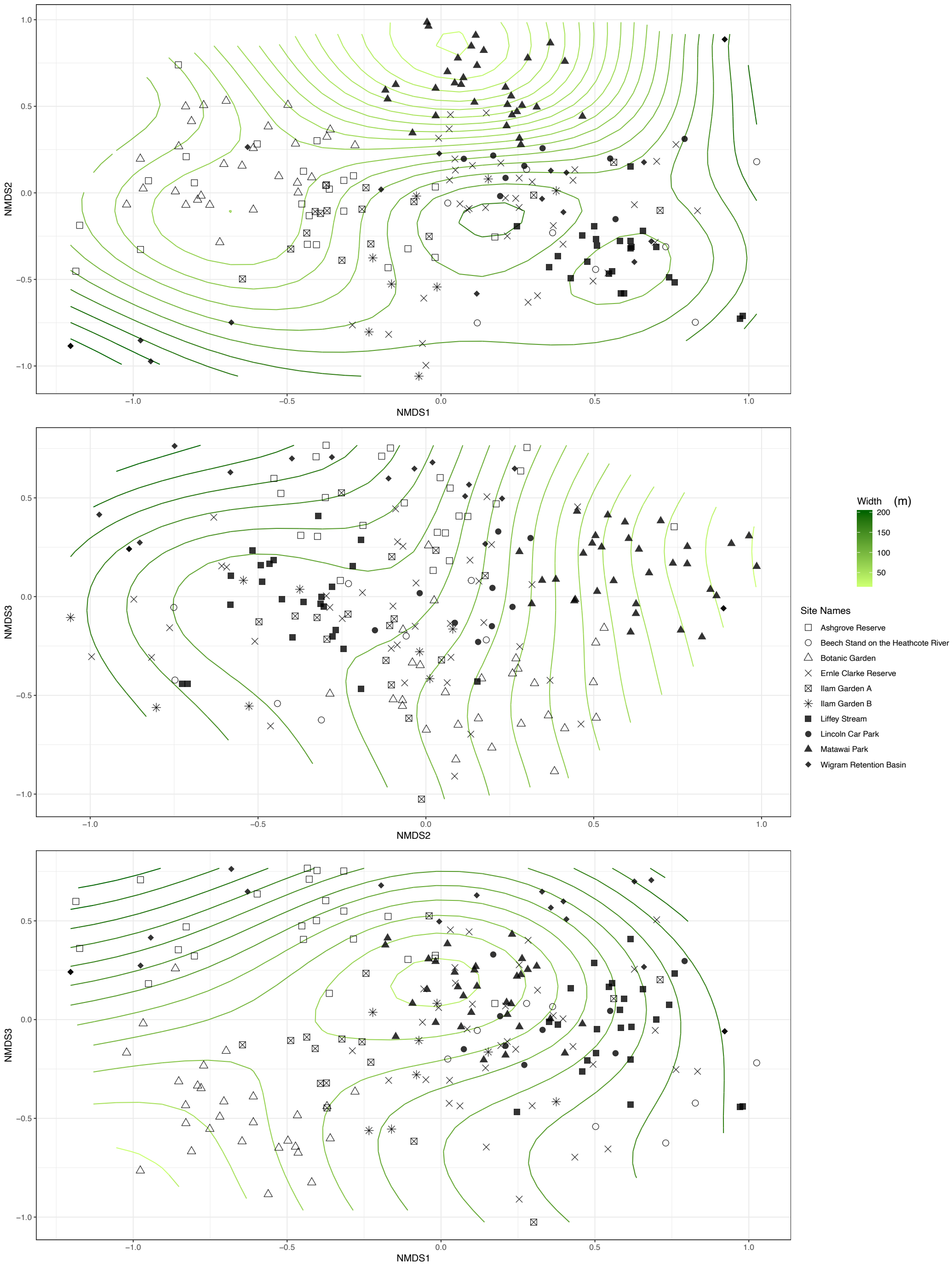


FIGURE 5.12: NMDS ordination results, showing sites and site width gradients with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Site width ranges from 20 to 160 m.

5.5.5.4 Minimum canopy age

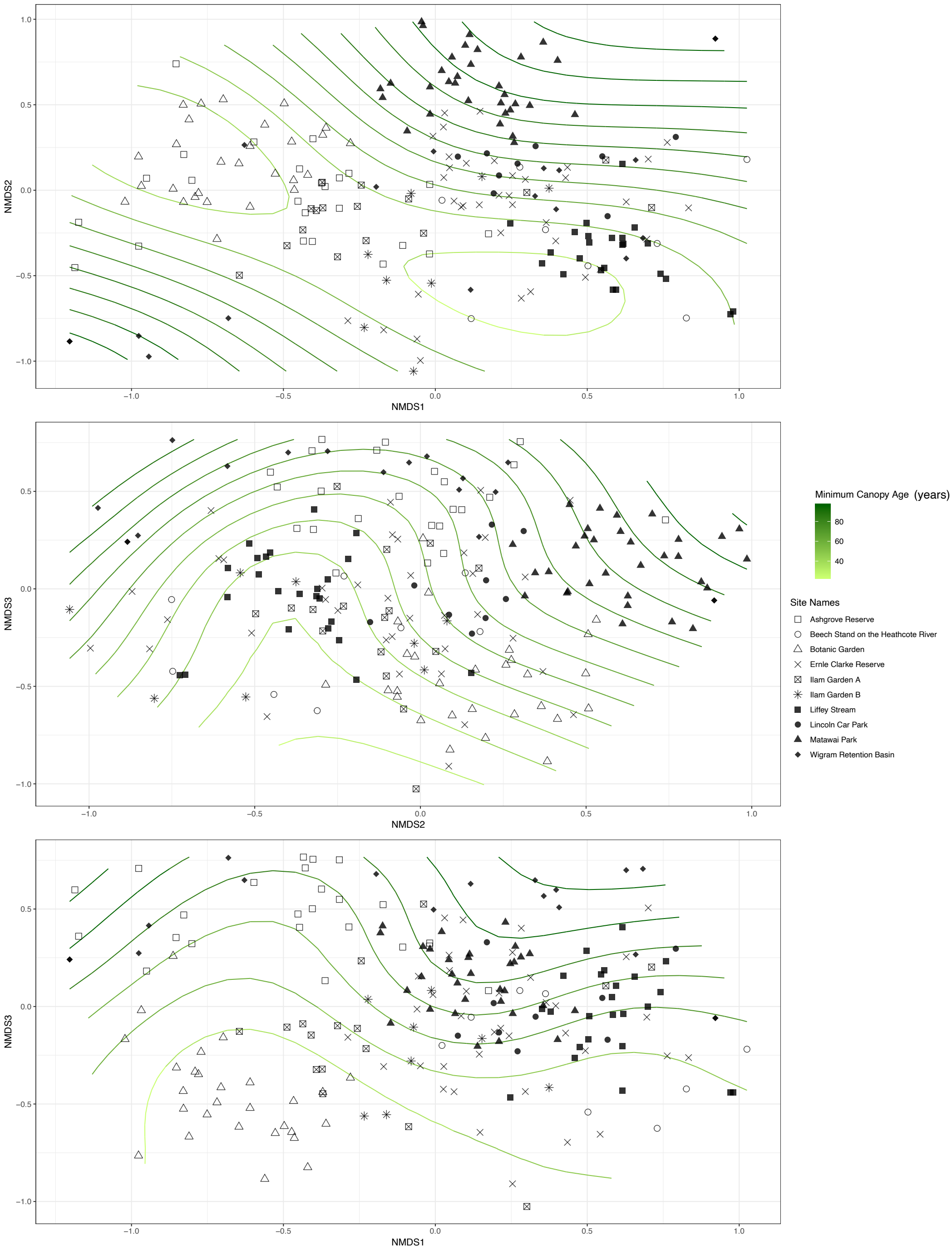


FIGURE 5.13: NMDS ordination results, showing sites and minimum canopy age gradients with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Minimum canopy age ranges from 30 to 110 years old.

5.5.5.5 Restoration age younger than canopy

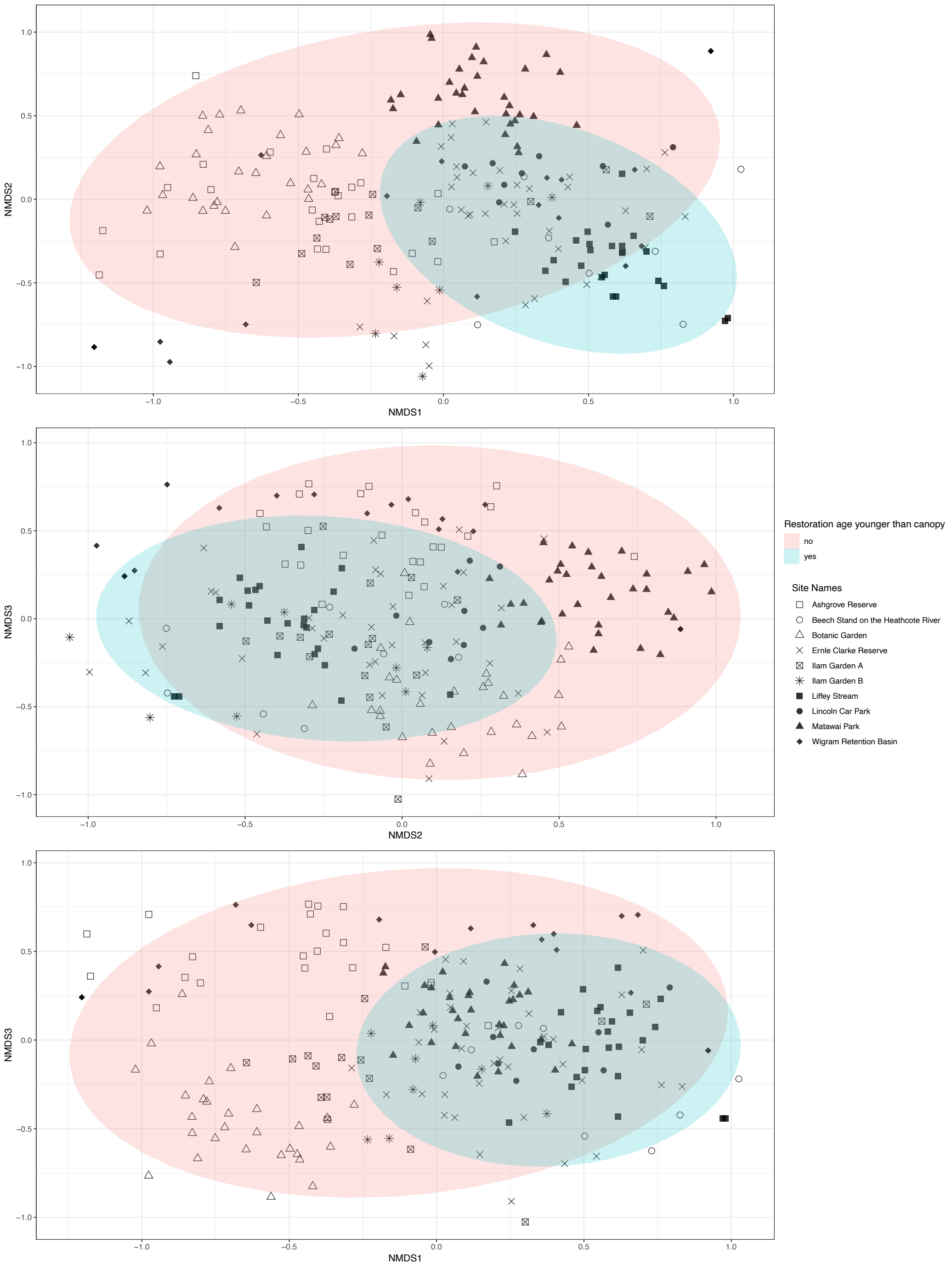


FIGURE 5.14: NMDS ordination results, showing sites and factor of restoration age younger than canopy clusters with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Yellow colour means restoration age older than canopy age and blue colour means restoration age younger than canopy age.

5.5.5.6 Exotic plant management

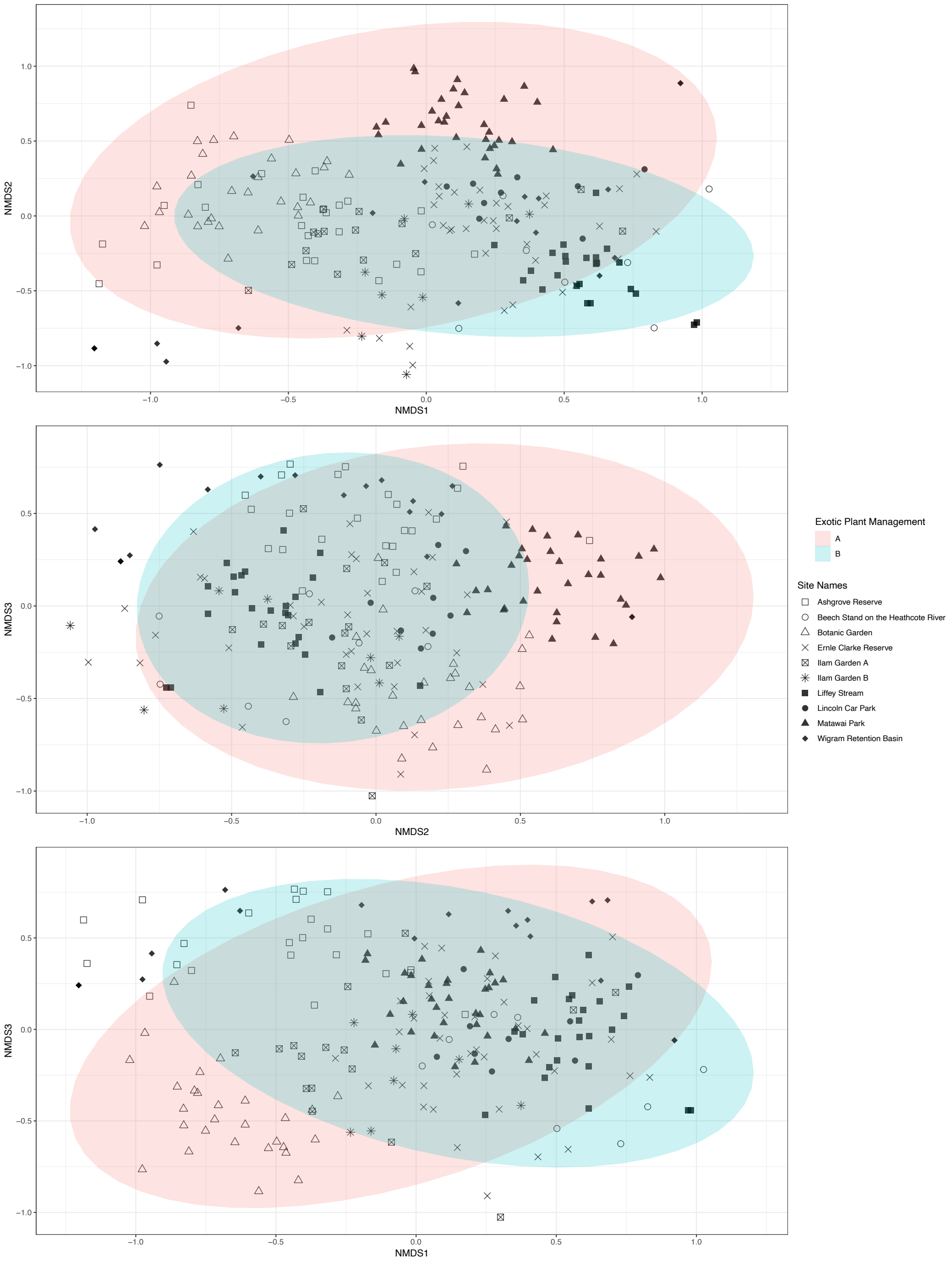


FIGURE 5.15: NMDS ordination results, showing sites and factor of exotic seedling management clusters with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Yellow colour with A means often remove exotic seedlings. Blue colour with B means planting some garden plants.

5.5.5.7 Exotic seedling management

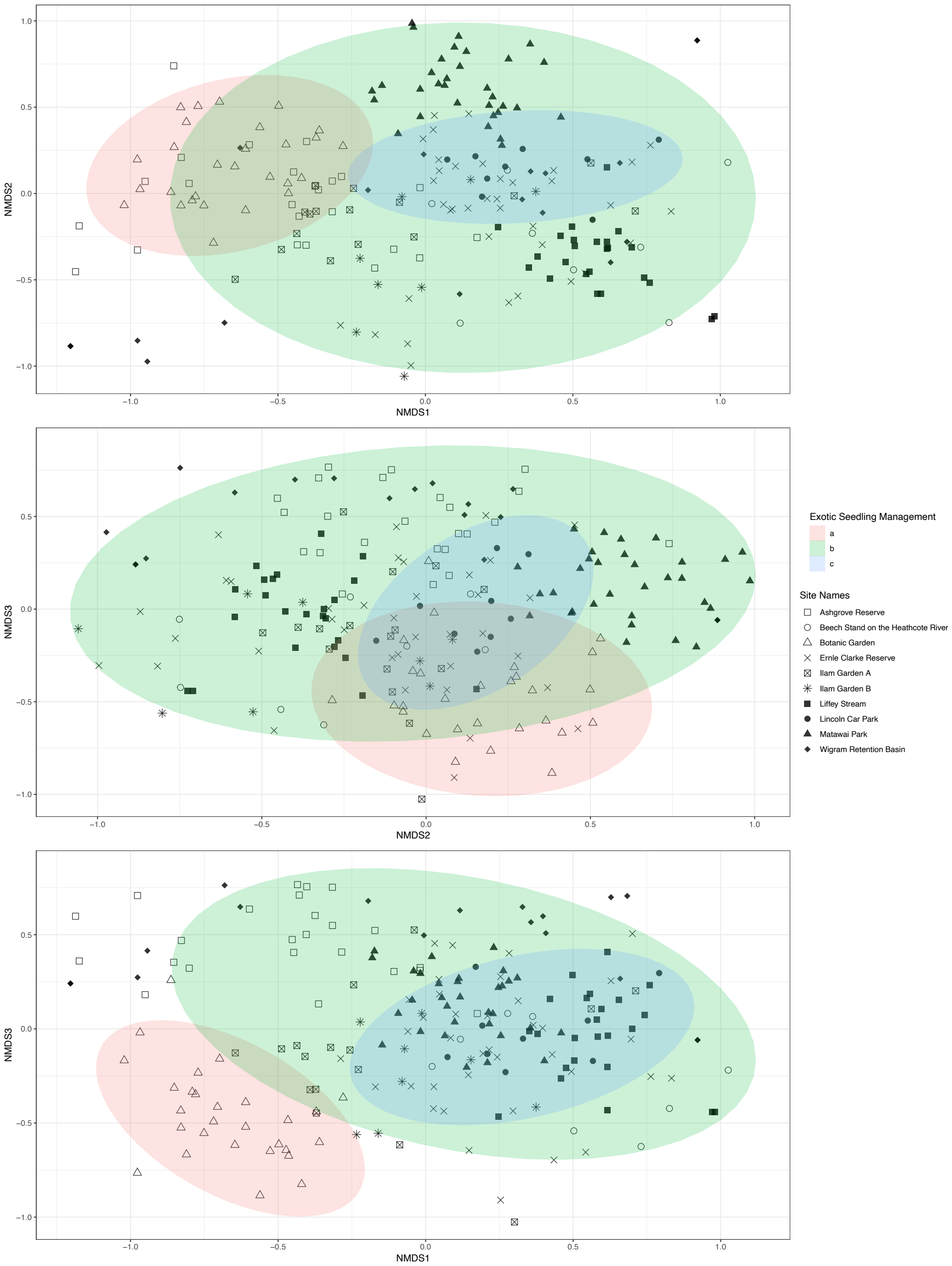


FIGURE 5.16: NMDS ordination in four dimensions, showing sites and factor of exotic seedling management clusters with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Yellow colour with a means often remove exotic seedlings. Blue colour with b means sometimes remove exotic seedlings. Green colour with c means never remove any exotic seedlings.

5.5.5.8 Native plant management

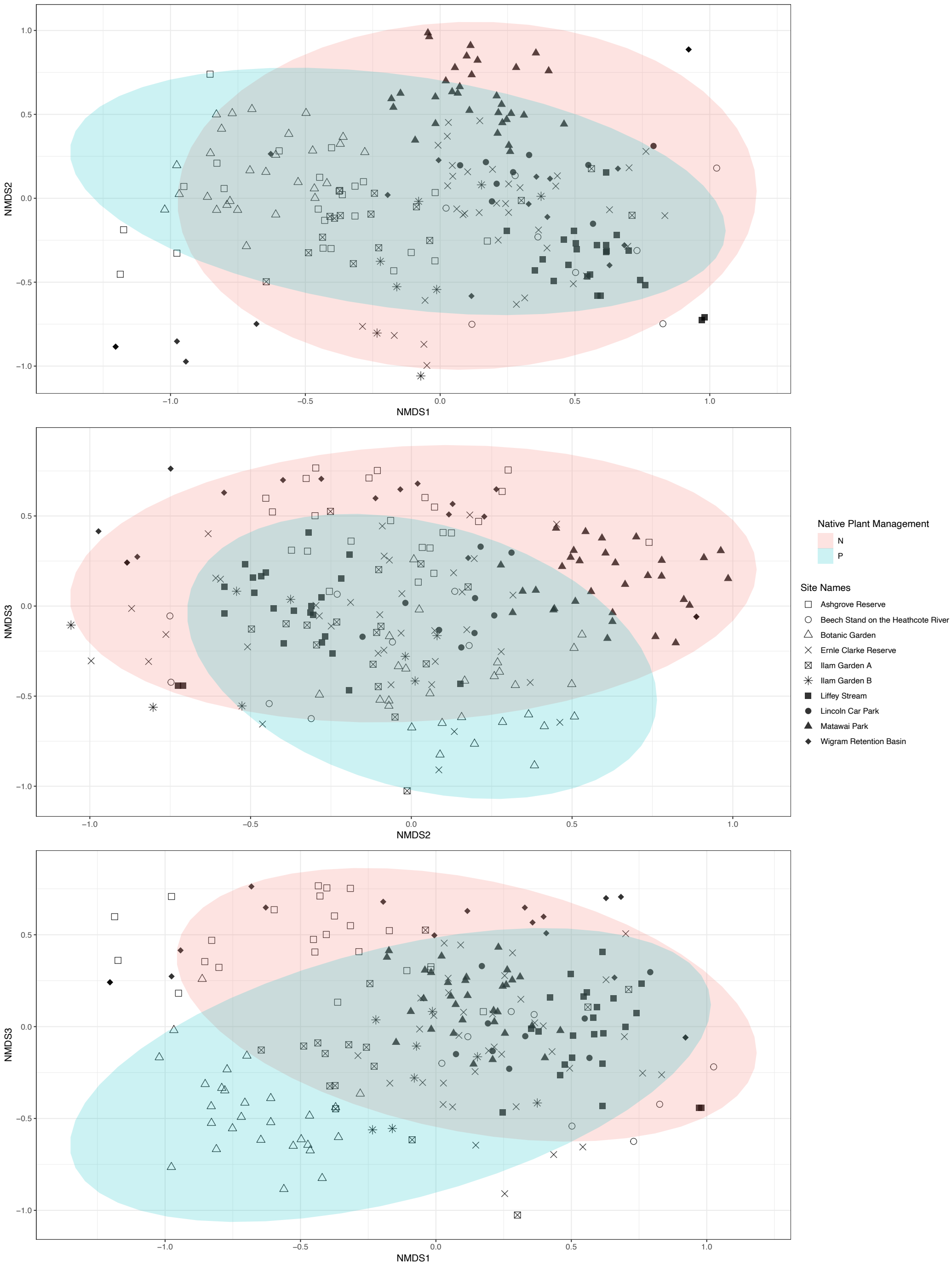


FIGURE 5.17: NMDS ordination results, showing sites and factor of native plant management clusters with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Yellow colour with N means never plant any new native plants and blue colour with P means plant native plants sometimes.

5.5.5.9 Native seedling management

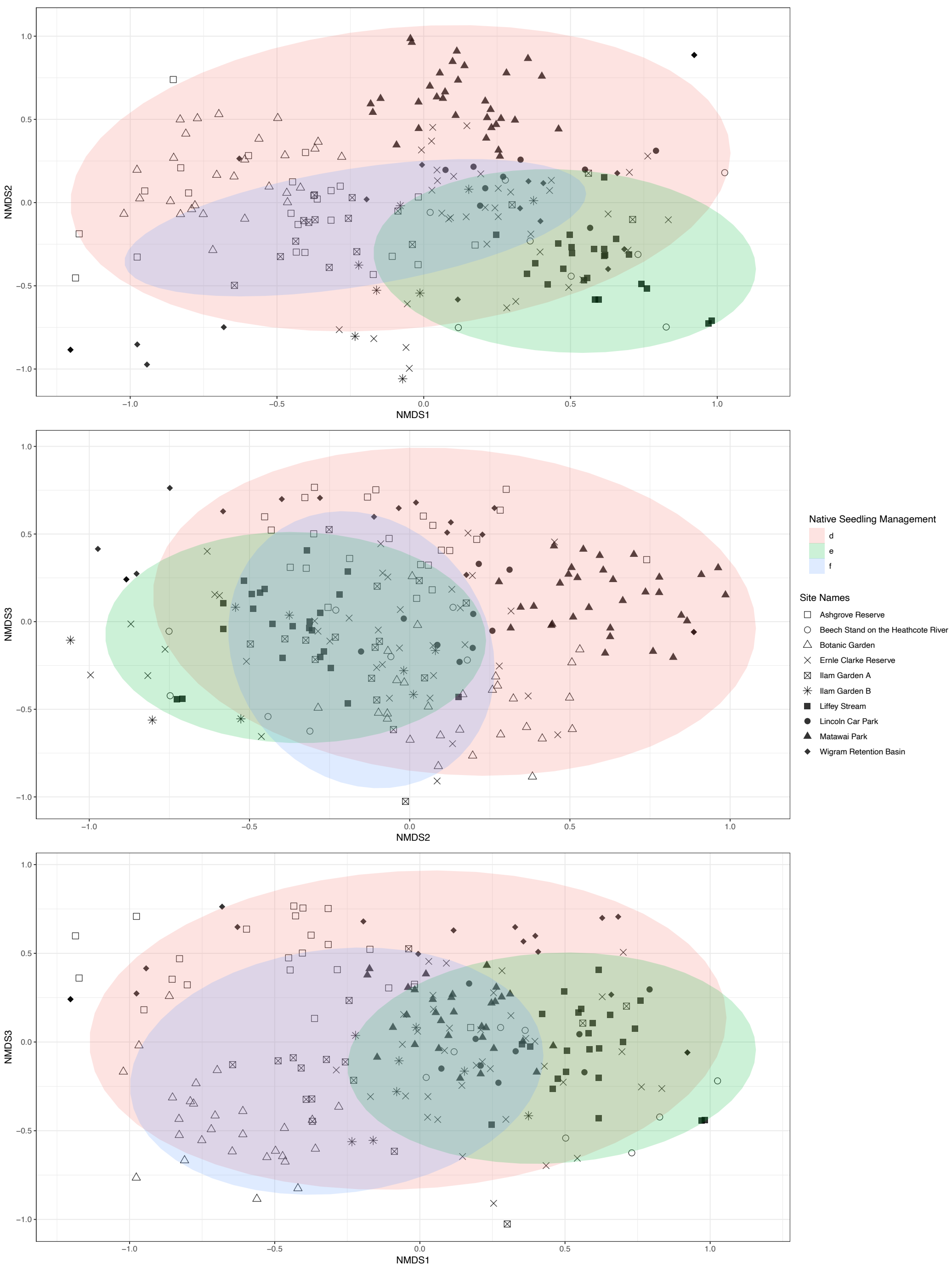


FIGURE 5.18: NMDS ordination results, showing sites and factor of native seedling management clusters with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Yellow colour with d means never remove any native seedlings. Blue colour with e means remove/cut some native seedlings. Green colour with F means remove/cut most/all native seedlings.

5.5.5.10 Canopy cover

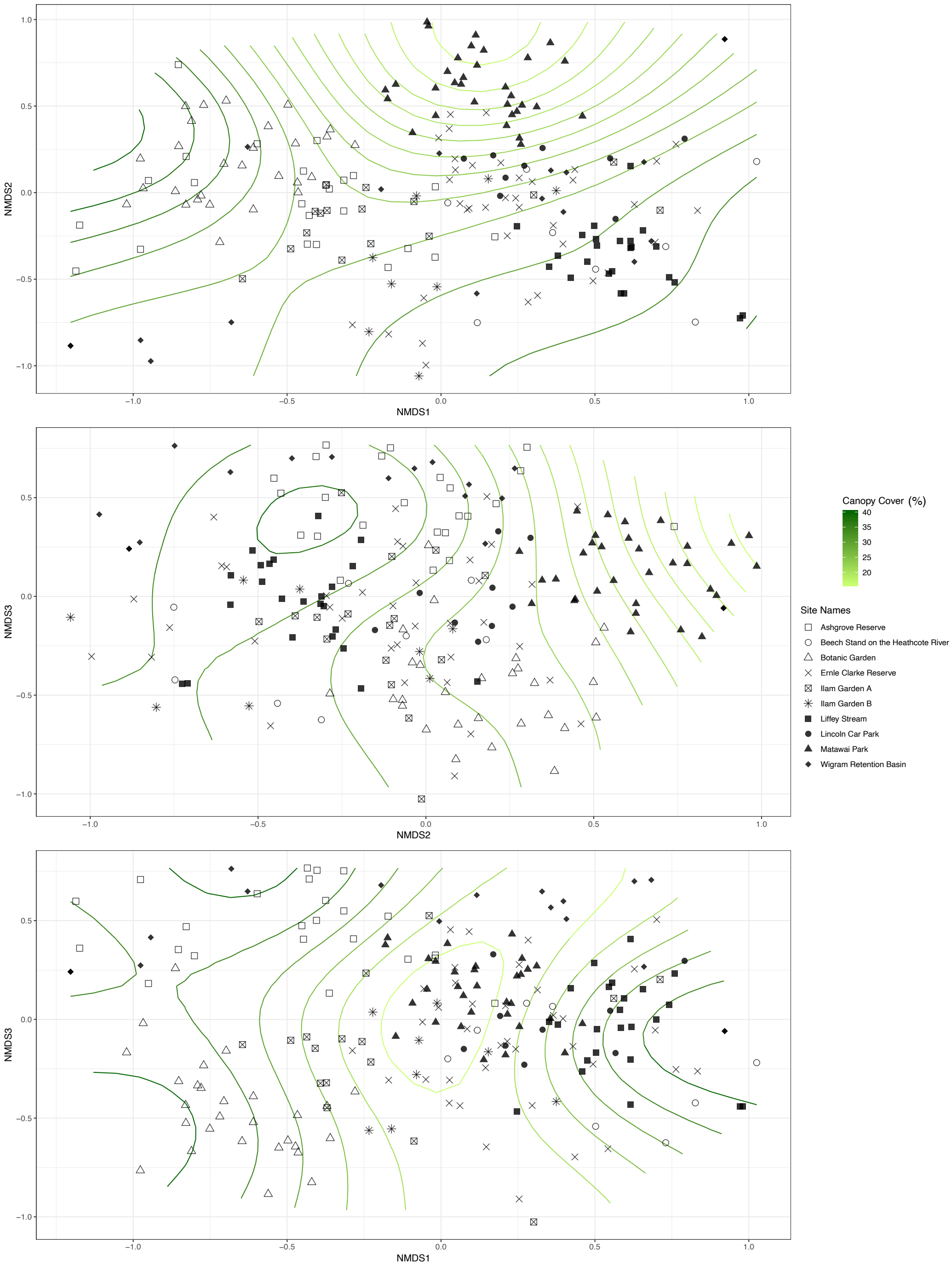


FIGURE 5.19: NMDS ordination results, showing sites and canopy cover gradients with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Canopy cover ranges from 16 to 38%.

5.5.5.11 Ground cover litter

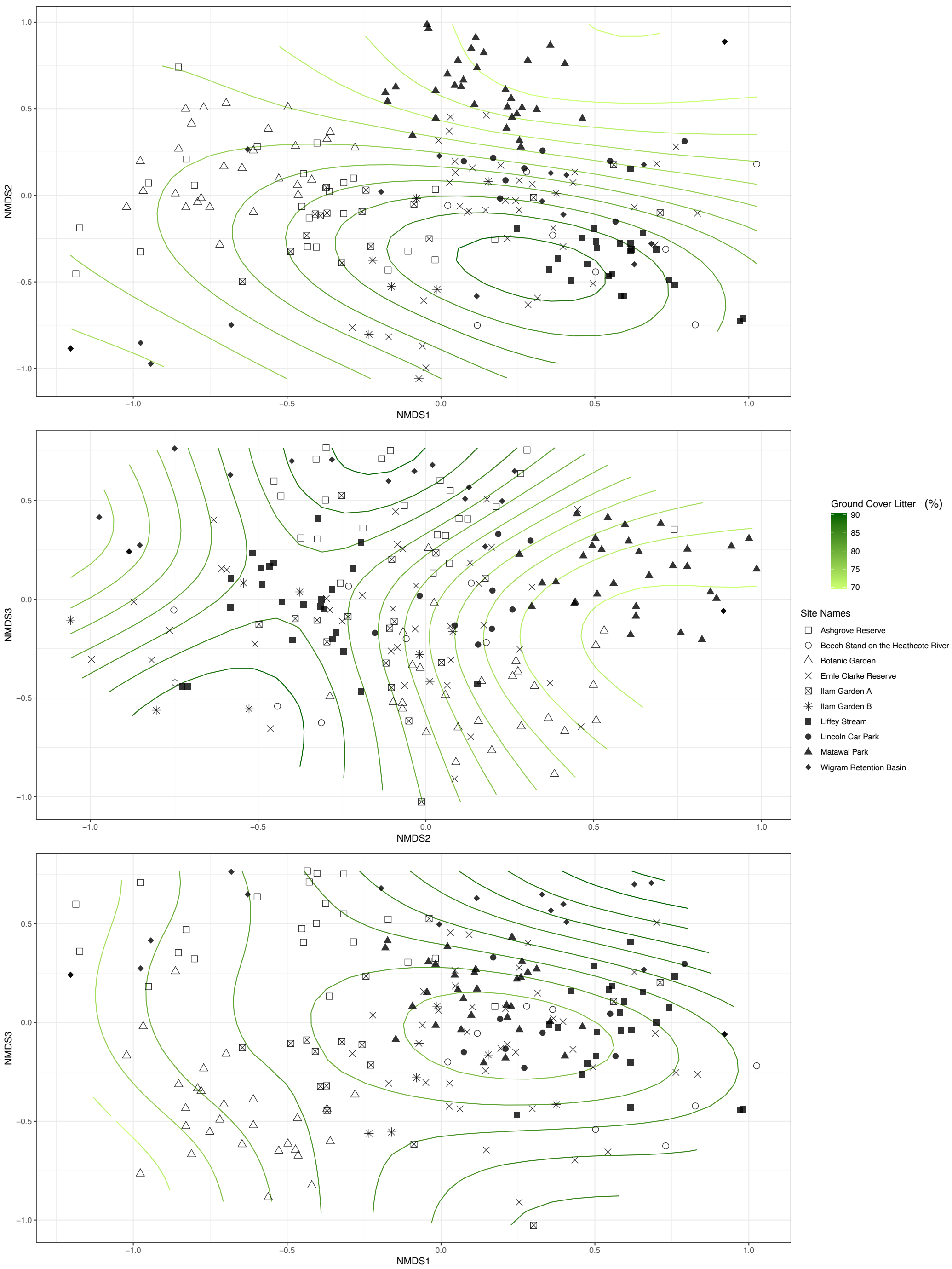


FIGURE 5.20: NMDS ordination results, showing sites and ground cover litter gradients with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Ground cover litter ranges from 70 to 88%.

5.5.5.12 Litter depth

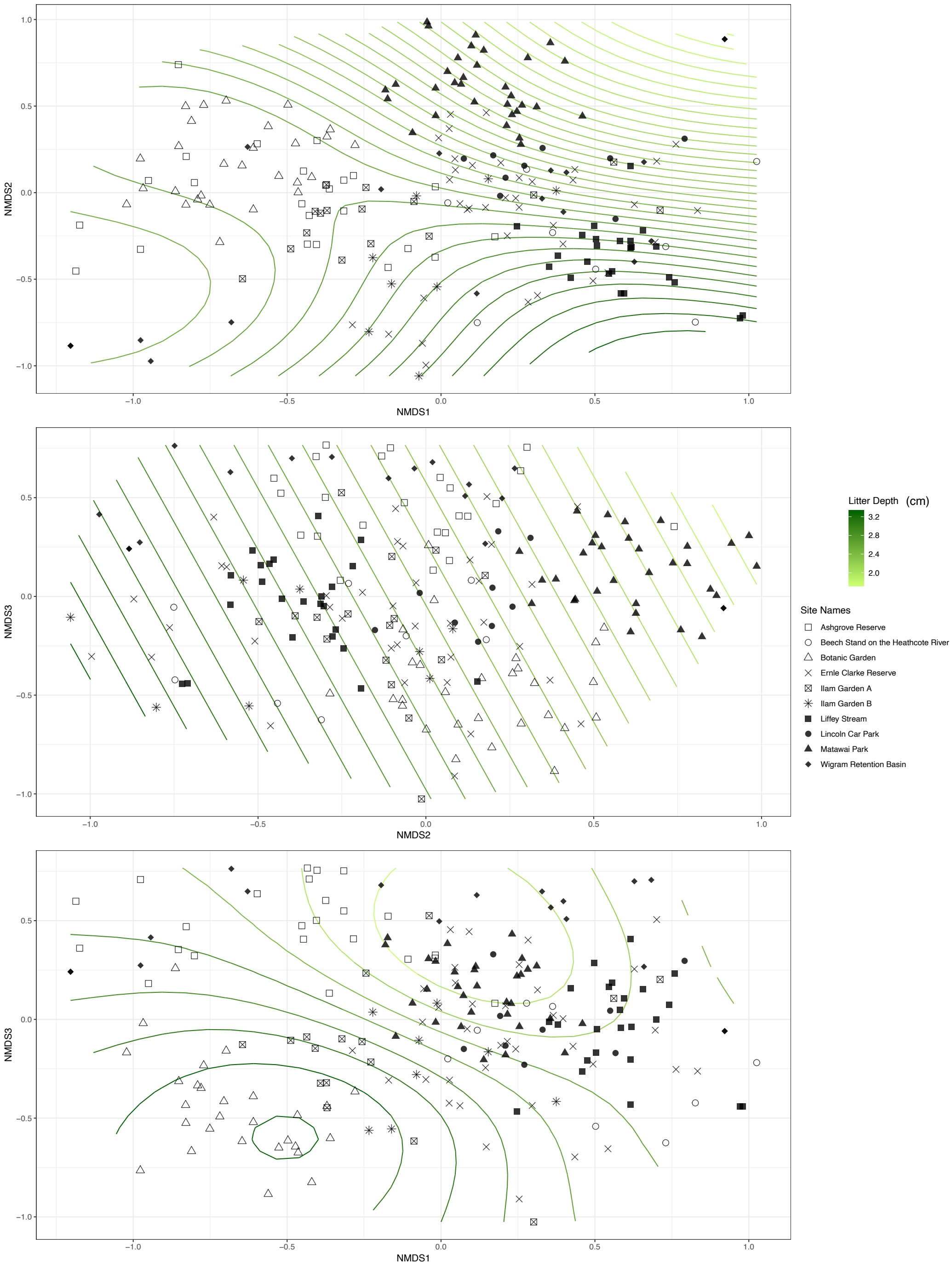


FIGURE 5.21: NMDS ordination results, showing sites and little depth gradients with respect to axes 1 versus 2, 2 versus 3, and 1 versus 3. Litter depth ranges from 1.8 to 3.2.

Non-metric multidimensional scaling (NMDS) on the data set of 2015 shows all the factors have significant effects on the vegetation composition of the sites and make them become several clusters.

The ordination diagram of site vegetation area resulted in a gradient on the first three axes from the Christchurch Botanic Garden to Lincoln's Liffey Domain, with a distinct clustering of Liffey Domain and Matawai Park (Figure 5.11). There is also a clear distinction between the Christchurch Botanic Garden and Ilam Garden A. Wigram Retention Basin has a much more dispersed pattern than other sites. For axis 1 versus 2, Matawai Park, Liffey Streem and Lincoln Car Park are in a cluster of large site areas as well as parts of the Ernle Clarke Reserve and Wigram Retention Basin. Only a few of Wigram Retention Basin sites are in the cluster of small site area. For axis 2 versus 3, Ashgrove Reserve and Wigram Retention Basin are in the large site area group while Botanic Garden is in the small site area group. For axis 1 versus 3, similarly, Matawai Park, Ernle Clarke Reserve, Liffey Streem are in the group of large site area.

The ordination diagram of width shows a similar picture. On axis 1, Matawai Park and Lincoln Car Park and Liffey Stream are in the cluster of large width value. While Botanic Garden is in the low width value cluster. On axis 2 Liffey Stream and Wigram Retention Basin are in the cluster of mid width value which is higher than the cluster value of Matawai Park and Botanic Garden. For axis 1 versus 3, the low width value cluster includes Matawai Park, Botanic Garden, Ilam Garden A, Ernle Clarke Reserve and Ashgrove Reserve. The mid width value cluster includes Ernle Clarke Reserve, Lincoln Car Park, Liffey Stream and Matawai Park.

The NMDS for minimum canopy age shows a different picture (Figure 5.13). There is clearly a cluster of young minimum canopy age which includes Botanic Garden and Matawai Park on the first axis. Ilam Garden A and Lincoln Car Park can be another cluster has a older minimum canopy age. Botanic Garden, Ilam Garden A and Matawai Park are in the similar minimum canopy age gradient on second axis. For axis 1 versus 3, Matawai Park can be a group while Liffey Stream and Ilam Garden A are in another group.

There are two clusters in the ordination diagrams of restoration age (Figure 5.14). It shows Botanic Garden and Matawai Park clearly separate out from the rest of sites in the cluster of older restoration age. Liffey Stream, Ilam Garden A, Liffey Stream fell within the cluster of younger restoration age. For axis 2 versus 3, most of the Matawai Park sites are in the cluster of older restoration age while Liffey Stream, Ernle Clarke Reserve, Ilam Garden A and Lincoln Car Park are in both two clusters. For axis 1 versus 3, Botanic Garden, Ashgrove Reserve and Ilam Garden A are only in the cluster with older restoration age while other sites are both in the clusters of two different restoration ages.

In the ordination diagram of exotic plant management(Figure 5.15, there are clearly two clusters. For first diagram (axis 1 versus 2), most of the sites in cluster B are also in cluster A. Only parts of the Liffey Stream sites are in cluster B. The second diagram (axis 2 v. 3) is different. Matawai Park is only in the management A cluster and Ilam Garden A, Ashgrove Reserve and Lincoln car park are in both management A and B clusters. The third diagram (axis 1 versus 3) shows only Botanic Garden is in cluster A and most of other sites are in the overlapping area.

For exotic seedling management ordination diagrams(Figure 5.16), there are three clusters. For first diagram (axis 1 versus 2), only Botanic Garden is in the cluster of management a which means often remove exotic seedlings and parts of the Ernle Clarke Reserve, Lincoln Car Park and Wigram Retention Basin are in the cluster of management c which means never remove any exotic seedlings which fells into the cluster of management b totally. The second diagram (axis 2 versus 3) shows a common area of three clusters which only includes a few of sites. Most of the sites are just in the cluster of management b. For the third diagram (axis 1 versus 3), Botanic Garden in cluster a and parts of the sites in the cluster b are also in the cluster c.

In the ordination diagrams of native plant management(Figure 5.17, three diagrams all show most of two different management clusters overlap. Among all sites, Ilam Garden A, Ernle Clarke Reserve, Ilam Garden B, Ashgrove Reserve, Lincoln Car Park and Beech on the Heathcote River are in the common area.

Native seedling management ordination diagrams(Figure 5.18) shows a similar result. For first diagram (axis 1 versus 2), cluster f is in cluster d with Ilam Garden A inside. Matawai Park and Botanic Garden are just in cluster d and Liffey Stream is in cluster e. For second diagram (axis 2 versus 3), Ernle Clarke Reserve, Liffey Stream and Lincoln Car Park are in the common cluster of three managements while Matawai Park is just in cluster d. For the third diagram (axis 1 versus 3), most of the cluster f area is in the cluster d and Wigram Retention Basin is cluster e only.

The NMDS for canopy cover shows the sites distribution in the gradient of canopy cover(Figure 5.19). First diagram (axis 1 versus 2) shows clearly Botanic Garden and Matawai Park are in the large canopy groups and followed by Ashgrove Reserve. Ilam Garden A and Liffey Stream can be same group of similar canopy cover. For second diagram (axis 2 versus 3), it is very clear that Liffey Stream, Ashgrove Reseve can be one group while Ernle Clark Reserve, Botanic Garden, and Ilam Garden A are in another group with smaller canopy cover. Matawai Park can be in another group. For the third diagram (axis 1 versus 3), Botanic Garden and Ilam Garden A can be one group with similar canopy cover. Matawai Park and Ernle Clarke Reservecan as well as Lincoln Car Park can be another group with the smallest canopy cover.

First diagram (axis 1 versus 2) of the NMDS for ground cover litter shows Matawai Park in the center of the ground cover litter gradient while other sites just located around it. The second diagram (axis 2 versus 3) shows Matawai Park, Botanic Garden and Ernle Clark Reserve can be one group which have bigger litter cover and Ilam Garden A and Liffey Stream can be another group. For the third diagram (axis 1 versus 3), Matawai Park, Lincoln Car Park and Liffey Stream are in the similar litter cover gradient. Botanic Garden and Ilam Garden A are in another similar litter cover gradient.

For litter depth, three diagrams are totally different. All sites in the first one(axis 1 versus 2) can be almost three groups: Mataiwai Park can be one group. Botanic Garden and Ashgrove Reserve can be another group. Most of other sites can be the third group. The second diagram (axis 2 versus3) shows a litter depth gradient from deep to light in the second axis. Two groups are shown clearly in third diagram (axis 1 versus 3) . The deepest group includes Botanic Garden and Ilam Garden A and the lightest one which includes Mataiwai Park, Ernle Clarke Reserve, Lincoln Car Park, Wigram Retention Basin and Liffey Stream.

5.6 Discussion

This study provides valuable results about the natural regeneration in urban forests and the factors which affect the regeneration procession. Of the ten sites I surveyed, the Christchurch Botanic Garden had the best native vegetation (Table 5.18). My survey data shows that local native species were a large percentage of the total. However, around 2,589 (44 species) trees were planted in Christchurch city parks by local government and most of them (around 70%) are exotic plants (Stewart et al., 2004). The good thing is attitudes of public and administrators towards the native species have been changing (Stewart et al., 2004) although we still need more work to promote the use of native tree species in our urban area.

It has been found that some native species in New Zealand regenerate well in urban environments (Smale & Gardner, 1999; Stewart et al., 2004), however, we still face the problem of losing other less adaptable species (A. E. Esler, 1991; Whaley, Clarkson, & Smale, 1997; Duncan & Young, 2000).

Urban system are more complex for seedling regeneration than most natural environments as it has unique circumstances such as frequent human disturbance (Rebele, 1994;

Grimm et al., 2000), modified soils (Pavao-Zuckerman, 2008) and exotic plants invasion (Alston & Richardson, 2006; Cadotte et al., 2017). I have some different results comparing the research of seedling regeneration in Red Zone without any management (see chapter 3). The seedling result shows naturalised species can make more seedlings in urban area. 5.41% of total seedling amount are naturalised species which had 33.45% of the seedlings in the survey data of 2015 and 6.1% of the total tree had 42.21% of the seedling in the survey data of 2007.Surprisingly, people's management did not have a big effect on the amount of the local native seedlings. Moreover, most of our survey sites have suitable canopy covers which are good for naturalised species to regenerate without people's management. From my results it is clear that local native species can do better than naturalised and non-local native species as the canopy cover thickens. Most New Zealand native tree species can tolerate shade to some degree (Stewart et al., 2004). Tree presence and biostatus analysis results also show that local native species have the highest seedling presence with adult trees in the plots. While naturalised species were not the most seedlings, they have considerable ability to produce seedlings and are worth controlling.

It was not easy to collect management information in this research as my research sites belong to city council and different organizations and most of them did not have management records. Based on conversations I had with some of the gardeners, I was able to summarise in a simple way the variation in management across the sites. My model results showed the management is not the important factor work on the seedling regeneration. As Stewart et al., (2004) suggested, if we can control the most aggressive, long-lived and shade-tolerant exotic species then it may help the urban forest to transfer to a new kind of indigenous-dominant forest (Stewart et al., 2004). Modelling work about various management and disturbance regimes was conducted to better understand the

Table 5.18: Total tree species richness at each of the ten sampled sites, separated by species' biostatus.

Site name	**Total Tree**	**Local native**	**No-local native**	**Naturalised**	**Exotic**
Ashgrove Reserve	24	18	6	0	0
Heathcote Beech Forest	4	3	0	1	0
Botanic Garden	29	19	10	0	0
Ernle Clarke Reserve	24	10	1	9	4
Ilam Garden A	19	12	2	3	2
Ilam Garden B	11	7	2	1	1
Liffey Stream	9	6	1	2	0
Lincoln Car Park	9	6	2	1	0
Matawai Park	12	11	1	0	0
Wigram Retention Basin	12	12	0	0	0

seedling successions dynamics in previous study (Meurk & Hall, 2000). It was mentioned that naturally regenerating sites in urban areas had greater native species richness in the understorey and it was suggested that reducing the understorey exotic vegetation will encourage for native restoration in the research based on Hamilton City, New Zealand (Overdyck & Clarkson, 2012).

Canopy cover is one of the important factors affecting seedling regeneration. As shown in Figure 5.6, seedling presence in a quadrat is higher as the canopy cover increases. Lower light transmittance in older sites may deter the establishment of some early-successional exotic species (Overdyck & Clarkson, 2012). There is another factor, site canopy age, that appears to covary with canopy cover, and may be driving part of the canopy cover effects. A great range of canopy covers in the sites with different canopy ages which will have influence on the germination and establishment of later successional species (White, Vivian-Smith, & Barnes, 2009).

There are significant differences between the surveys of 2007 and 2015. My models show several factors and the interactions associated with the differences among the surveys. The model results (Figure 5.9) show that seedling presence was higher in 2015 than 2007 in more combinations of factors. Several things may have changed in these sites during these eight years: more younger trees reached maturity, and more old trees were cut/removed or just died. Also there has been volunteer work in some of these sites, especially removing exotic weeds but also planting.

Several issues need to be noted about comparing the survey data from 2007 and my survey in 2015. Unfortunately, there was no GIS information for the sites nor any photos left from the survey in 2007. The 2007 survey also archived no evidence (collections, photos) that could be used to verify plant identifications. In the 2007 data I found some rare local native species which are not common in Christchurch urban area. As I mentioned before, I did not find these species in the re-survey and also I did not find lots records in iNaturalist.NZ, so I decided it was prudent to treat these as likely identification mistakes.

It was surprising that the soil factors were not important in the models. In previous studies, degraded soil conditions such as soil nutrients, beneficial microbes and moisture have been confirmed can limit regeneration (Rebele, 1994; Pavao-Zuckerman, 2008; Oldfield et al., 2015; Pregitzer, Sonti, & Hallett, 2016) as well as soil compaction (Craul, 1985). In this research, I was able to obtain a general urban soil map for Red Zone area and found no significant differences in seedling regeneration among soil types. It is likely that this general soil map did not capture the details of the soil conditions present at each site, especially the upper soil conditions experienced by germinating seedlings. Direct soil surveys at these sites would be informative.

The NMDS ordination results show a range of factors which are causing the sites to differ from one another in their species composition. The factors include environmental factors and management. This is particularly the case where the environmental factors are indirect and correlated with each other. Among these factors, site vegetation area and width are the two factors that have the strongest influence on the species composition. This can be explained by the edge and area effects. They are more important in small fragments (Yang et al., 1986; Wilcove, McLellan, & Dobson, 1986). Comparing area, fragment shapes are an important determinant of edge effect (Laurance & Yensen, 1991; Young & Mitchell, 1994; Murcia, 1995; Laurance et al., 2007; Harper et al., 2005).

In order to achieve the goals of the New Zealand Biodiversity Strategy (Department of Conservation, 2000) and Christchurch City Council Biodiversity Strategy (Christchurch City Council, 2008), a functional urban ecosystem with healthy urban forests as well as native species needs to be built. Because of the vulnerability of our flora to invasive species, the focus must keep on native biodiversity rather than species richness (Clarkson et al., 2007). For New Zealand, we do not have a history of managed urban woodlands (Clarkson et al., 2007), so urban forest and restoration site management is still relatively new (Clarkson et al., 2007). However, it has been accepted that native and exotic mixed urban forest is becoming acceptable rather than extermination of exotic species (Stewart et al., 2004).Inspiration for what can be achieved in Christchurch can be found in the management of the urban forest in Canberra, Australia. In Canberra, major tree planting started from 1920's and there are about 400,000 trees (over 200 species) in the urban forest today. Now urban forests have an equal mix of native and exotic species.A tree database and modelling system were used to help to monitor the develop the urban forest changes and supervise the planing for future landscapes (Banks & Brack, 2003).

5.7 Conclusion

Natural regeneration in urban areas are influenced by a mix of factors. In this study, we analysed several environmental factors as well as management factors. Adult tree presence, canopy cover, and biostatus had significant influences on the seedling regeneration in my 2015 survey. Both adult tree presence and canopy cover had positive effects on the seedling regeneration and local native species were most frequently found as seedlings.

Species composition has changed in the eight years from 2007 to 2015. Generally, local native trees dominate in the restoration sites in urban area. Compared with local native species, some naturalised and non-local species have stronger regenerative abilities. Less naturalised species were found in the survey in 2015 and those naturalised trees produced more seedlings in 2015. Furthermore, in 2015, we found more seedling species of all three biostatus than 2007.

Site age, minimum native restoration age, site width and adult tree presence these four single factors and a few interactions work on seedling presence changes between two surveys. Two factors which are minimum canopy age and adult tree presence have a significant effect on seedling abundance when seedlings were present.

NMDS ordination on the data set of 2015 revealed that several environmental factors, as well as site management, have significant effects on the seedling vegetation composition of the sites.

In conclusion, local native species regenerate successfully in the restoration sites under the environmental conditions and management. However, some of the naturalised species have strong regenerate abilities which can produce large number of seedlings, such as Sycamore (*Acer pseudoplatanus*). That will suggest in the restoration work, these naturalised species be should under controlled. Despite this, many local native species are naturally regenerating, especially in the larger and older sites.

Chapter 6

Urban forest patch connectivity in Christchurch

6.1 Introduction

Fragmentation and isolation, though dramatic loss of natural habitats, especially in urban areas, is recognised as a critical threat to biodiversity (Drinnan, 2005; Saunders, Hobbs, & Margules, 1991). Because of fragmentation, many species become confined to small areas of remnant vegetation (Loyn, 1987), typically surrounded by unsuitable human landscapes (Drinnan, 2005). Patches of secondary growth vegetation are typically referred to as 'patches' and a patch is a place with homogeneous conditions relative to other types of patches (Forman, 1995).

Connectivity has somewhat different meanings in landscape ecology and metapopulation ecology. In a review by Tischendorf and Fahrig, they clarified the concept of land connectivity. In their opinion, connectivity has different meanings in different contexts (Tischendorf & Fahrig, 2001). For example, "landscape connectivity" refers to the landscape ecology which covers an entire landscape. "Patch connectivity" is used in metapopulation which is an attribute of a patch (Tischendorf & Fahrig, 2001). There are also some other contexts such as "connectedness" and "habitat connectivity". They suggest in the article using "patch connectivity" and "landscape connectivity" since other terms are ambiguous (Tischendorf & Fahrig, 2001). Atte and Ilkka had a different idea and proposed that connectivity should have fundamentally the same meaning in both landscape ecology as well as metapopulation ecology (Moilanen & Hanski, 2001). In this study, we use the patch-based description to evaluate the connectivity of the patches in Christchurch urban area and "Patch connectivity" will be preferred. Here, patch connectivity is about the functional aspects of the real connections among the different patches, from energy to information and matter. It includes everything from pollen dispersion of flora and movements of fauna (Mallarach & Marull, 2006).

Patch connectivity also can be considered a measurement of the extent to which an individual's movement occurs between patches of habitat. There are two aspects of patch connectivity, structural and functional (Hilty, Lidicker Jr, & Merenlender, 2012; Lechner, Doerr, Harris, Doerr, & Lefroy, 2015; Ranius & Fahrig, 2006). Structural connectivity measures the arrangement of landscape elements in space. Functional connectivity characterises how the movement of species between patches is affected by the spatial arrangement of patch patches.

There are two common conservation practices to reduce the impacts of patch fragmentation (Beier & Gregory, 2012). One is increasing patch area, which reduces the species-area effect and edge effects, therefore, more species are able to sustainably exist in a single patch. Another one is rebuilding connections between patches, which lets small patches function together to work as a bigger patch. Based on this, species in those patches form functional metapopulations (Beier & Gregory, 2012). However, it is not easy to increase the patch size in urban areas, and increasing patch connectivity sometimes is the only option for reducing the effects of patch loss and fragmentation (Lechner & Lefroy, 2014). A major challenge for conservation is that indigenous ecosystems are getting smaller and more fragmented (Rutledge, 2003). There is a need to understand and assess the effects on biodiversity of existing patches or habitats, and to restore vegetation in locations that best support dispersal and population connectivity of species between existing patch patches (Lechner et al., 2015).

Identifying and evaluating functional connectivity between habitat patches is important for ecological network design (Prugh, 2009; Moilanen & Nieminen, 2002) and it has been recognized as a fundamental factor determining species distribution (Doak, Marino, & Kareiva, 1992; Taylor et al., 1993; Lindenmayer & Possingham, 1996; Hanski, 1998; Moilanen & Nieminen, 2002). There are two approaches to assessing connectivity. One is tracking the migration of animals, and using dispersal models to simulate the success of different ecological network designs (Bender, Tischendorf, & Fahrig, 2003; Gardner & Gustafson, 2004; Matter, Roslin, & Roland, 2005; Ranius, Johansson, & Fahrig, 2010; Duggan, Schooley, & Heske, 2011). The other is based on the least-cost model. The least-cost model is normally used to determine the movement routes of wildlife for the purposes of optimising conservation of meta-populations in wild environments (Adriaensen et al., 2003; Sawyer, Epps, & Brashares, 2011; Adriaensen et al., 2003; Piemontese et al., 2015; Balbi et al., 2019). Normally, the parameters of least-cost models should come from field data of a specific organism. However, it is difficult and time-consuming to quantify these parameters for a large number of species and many studies are needed in order to get the data. Moreover, animal movements are influenced not just by the biology of the species but also by the landscape.

Several methods have been suggested for calculating indices of habitat connectivity (Hanski, 1994; Bunn, Urban, & Keitt, 2000; Ricotta, Stanisci, Avena, & Blasi, 2000; Urban & Keitt, 2001; Moilanen & Nieminen, 2002; Jordán, Báldi, Orci, Racz, & Varga, 2003; Calabrese & Fagan, 2004). It has been found that many of these methods are "short of comprehensive understanding of their sensitivity to pattern structure and their behaviour to different spatial changes, which seriously limits their proper interpretation and usefulness" (Pascual-Hortal & Saura, 2006).It is really important to have robust methods for evaluating the importance of spatial elements such as patches and corridors in measuring connectivity (Jordán et al., 2003). Moreover, in forest management, it has been recommended that the spatial scales assessed should be broadened to a landscape scale to effectively integrate connectivity (Wiens, 1997; Tischendorf & Fahrig, 2001; Raison, Brown, & Flinn, 2001).

Pascual-Hortal and Saura (2006) have promoted a methodology which is based on using graph structures and habitat availability indices for analysis of forest habitat connectivity (Pascual-Hortal & Saura, 2006). This methodology can capture the landscape changes affecting connectivity; it can also detect the most important landscape elements most influencing connectivity (Pascual-Hortal & Saura, 2008; Saura & Pascual-Hortal, 2007). A graph contains a set of nodes (or vertices) and links between them. Each link connects two nodes (Pascual-Hortal & Saura, 2008). The links between each pair of patches can be used to represent the potential ability of an organism to disperse directly between these two patches (Pascual-Hortal & Saura, 2006). Sometimes they can be obtained as a dispersal distance to represent the functional connection (Pascual-Hortal & Saura, 2006). The distance between patches is compared with dispersal distances of the organism to assign or not a link between these patches (Pascual-Hortal & Saura, 2006). In the graph theory no node can be visited more than once; that means the link from one node to another only be charged once (Pascual-Hortal & Saura, 2006). A component in the graph theory is a set of nodes in which there always is a link between each pair of the nodes (Pascual-Hortal & Saura, 2006) but there is no functional relation (link) between nodes of different components (Pascual-Hortal & Saura, 2006). Because of the landscape change, the component numbers may go up once the links between the components are missing (Pascual-Hortal & Saura, 2006). If a node or a link removal can cause the disconnection of components, then the node is called cut-node and the link is cut-link (Pascual-Hortal & Saura, 2006).

Chundi et. al (2015) did research about the ecological networks of the Christchurch City area in New Zealand. In her paper, she used the Landscape Development Intensity (LDI)

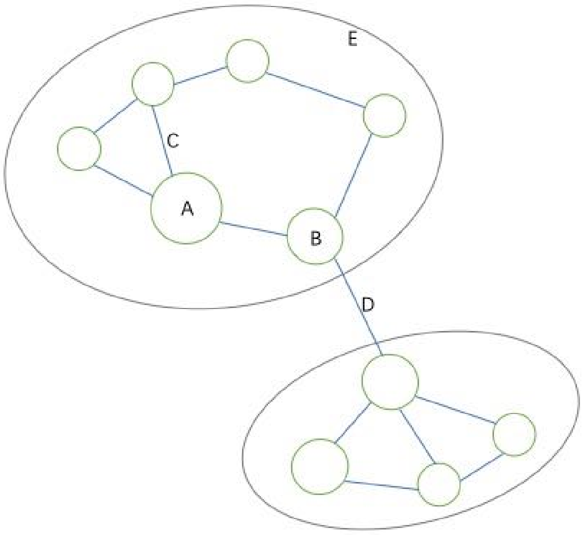


FIGURE 6.1: A simple graph based on graph theory. A: Patch(node), B: Cut-node, C: link, D: Cut-link, E: Component. This figure is modified from a figure from (Pascual-Hortal & Saura, 2006)

index which is a measure of human disturbance to ecosystem, to quantify the relative cost of land use/cover types to build the cost surface for least-cost model in ArcGIS. She used the maximum seed dispersal distance of *Dacrycarpus dacrydioides* as the threshold distance for network analysis in the software Conefor 2.6. The result showed that 408 links were simulated in the study area under the 1,200 m threshold distance for dispersal. Additionally, the study also found no linear relationship between the link importance value and the total area of habitats (Chundi et al., 2015). Based on results of Chapter 4, most of the species we found in the Red Zone are are dispersed by birds and wind and most of the dispersal paths would not be through the urban land and we found most species seedlings were found within less than 1000 meters. Further more, *Dacrycarpus dacrydioides* is not a common species in the Christchurch urban area. In this study, we more focus on the Red Zone area and the connection between it and other patches in the urban area. Combined the connectivity map got from Chundi et. al, it is worthy to try a different method which based on the the Integral Index of Connectivity (IIC) (Pascual-Hortal & Saura, 2006) to assess the importance of the patches in Christchurch urban area and test some path design for the Ōtākaro-Avon Red Zone.

In this study, we will apply this methodology to analysis the urban native forest habitats in Christchurch urban area, New Zealand. Specifically, we will use the Integral Index of Connectivity (IIC) of (Pascual-Hortal & Saura, 2006) to characterise the patches and their connectivity. The objectives of this study are:

1. Update the forest habitat connectivity analysis for Christchurch city used improved habitat data and methods.

2. Assess the affects on forest habitat connectivity in Christchurch city of different forest restoration scenarios in the residential red zone of eastern Christchurch, which was cleared of houses following the 2010–2011 earthquakes.

3. Identify priority areas for forest restoration in the Christchurch city area that would most improve forest habitat connectivity.

6.2 Methods

6.2.1 Study sites

This research focus on the urban area of Christchurch city, New Zealand (Figure 6.2). For details on the ecosystems and history of the city, see Section 1.6.

Figure 6.2: Map of Christchurch urban area.

6.2.2 Data collection

A feature layer map of public parks in the Canterbury Region was acquired from the Canterbury Maps website (`http://canterburymaps.govt.nz`) which originally came from Environment Canterbury, Canterbury's Regional Council (http://gis.ecan.govt.nz). This layer was updated on March 27, 2017. This Canterbury region parks map was clipped in QGIS (Version 3.6.0) to restrict it to just the Christchurch city area (Figure 6.3).

For the purposes of this study, Christchurch City refers to the Christchurch District north of, and including, the Christchurch Port Hills. This area includes the built area of Christchurch city, and excludes the rural landscape of Banks Peninsula. Banks Peninsula is also part of the Christchurch District and is under the jurisdiction of the Christchurch City Council, but it differs in many geographical and ecological ways from the built city landscape.

FIGURE 6.3: Park map of Christchurch urban area. Note that the habitats in these parks vary greatly, including native forest, open sports fields, exotic conifer plantations, and grasslands.

6.3 Analysis

6.3.1 Source patch(node)

Patches were picked up from the Christchurch parks data base as the source patches. Each source patch needed to be covered with native vegetation or have a high native plant cover under an open or closed tree canopy (Figure 6.4). The largest areas of native forest are on Port Hills at the south of the city. The large area of habitat to the northeast (Figure 6.4) is Bottlelake Plantation, mostly dominated by a productive plantation of *Pinus radiata* (a North American native species) but including patches of both planted and wild indigenous-dominated vegetation.

FIGURE 6.4: Source patch (node) map of Christchurch urban area, restricted to just the patches containing considerable areas of indigenous vegetation.

6.3.2 Link threshold distance

As the results of Chapter 4 showed, 9 of the native species seedlings (totally 10 species) are found within 1000 meters of adult conspecifics (Figure 3.14). Therefore, 1000 meters was used as the preferred threshold distance, although other distances were also tried (100m, 300m, 500m, 800m, 1000m, 1100m, 1200m, 1300, 1500m, 2000m, 3000m) to assess the sensitivity of conclusions to dispersal distance.

6.3.3 Minimum distance

Edge-to-edge minimum distance between patches was calculated in QGIS (version 3.6) by using the 'Distance matrix' tool. In this study, links are obtained by comparing the edge-to-edge distance between nodes with the threshold distance. If the distance between two patches was less than the threshold, a link was assigned between these nodes, otherwise there was no direct link (Pascual-Hortal & Saura, 2006).

6.3.4 Connectivity indices

In this study, we used several connectivity indices to quantity and characterise the importance of individual forest habitat patches. Details of the connectivity indices are below.

NL: Number of Links

NC: Number of Components

IIC: Integral Index of Connectivity

$$IIC = \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i . a_j}{1+nl_i j}}{A_L^2} \tag{6.1}$$

where n is the total number of the source patches (nodes), a_i and a_j are the areas of patch i and j, nl_{ij} is the number of links in the shortest path (topological distance) between patch i and j. A_L is the area of study area (Christchurch, 1,426 km^2) (Devi, Murthy, Debnath, & Jha, 2013).

Because the IIC is too small to make a graph result, here we use IICnum:

$$IICnum = \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i . a_j}{1 + nl_i j}. \tag{6.2}$$

dIIC: Source patch (node) importance index

$$dIIC = 100 \frac{IIC - IIC^{'}}{IIC} \tag{6.3}$$

where IIC and IIC' are the IIC value before and after the loss of a source patch (node) respectively. A higher $dIIC$ value indicates a higher source patch importance (Pascual-Hortal & Saura, 2008). All dIIC value were computed through the Conefor Sensinode 2.6 software (Saura & Torne, 2009).

6.3.5 Red Zone Scenarios

In this study, Residential Red Zone data from eastern Christchurch were computed with Christchurch patch data through Conefor Sensinode 2.6 to tested if making all or part of the Red Zone area into native forest would make a contribution to improve the habitat connectivity in Christchurch urban area. Three Red Zone scenarios were tested, only the banks of the Avon River restored to forest, scattered patches of native forest restoration in Red Zone, and all of the Red Zone restored to native forest.

6.4 Result

6.4.1 Fragmentation status

There were 175 source patches in the study area. In total these patches made up just 30.39 km^2, 2.1% of the Christchurch city study area. These 175 patches were categorised into 4 classes based on their area: small patches (<0.01km^2), medium patches (0.01-0.1km^2), large patches (0.1-1km^2) and super large patches (>1km^2). Among these patches, 87 patches were small patches which covered 0.27 km^2, 59 patches were medium patches which covered 2.29 km^2, 22 patches were large patches which covered 8.59 km^2 and only 7 patches were super patches covering 19.24 km^2 (Table 6.1).

Figure 6.4 shows that almost all of the 175 patches were located in the centre and eastern of the city. It is nearly blank to the west of the city (a landscape dominated by housing and farmland). Most of the large and super patches are located along coastal area and in the Port Hills, while the patches in the city center are quite small.

Table 6.1: Patches with substantial indigenous vegetation in the Christchurch City wider urban area, categorised by area.

Patch Type	Number	Proportion	Area(km^2)	Proportion of total area
Small patches	87	49.71	0.27	0.87
Medium patches	59	33.71	2.29	7.53
Large patches	22	12.57	8.59	28.27
Super patches	7	4	19.24	63.31

6.4.2 Optimal threshold distances

The optimal threshold distance analysis shows that the link number (NL) increases linearly as the threshold distance increasing (Figure 6.5). In contrast, component number decreases when the threshold distance increases (Figure 6.6). Short threshold distance make more components become apart and the links are getting less. Based on the dispersal distance result, 1000 is the optimal distance as most of the native species seedlings can establish within 1000 meters and in the NC index result there are 19 components for 1000 meters.

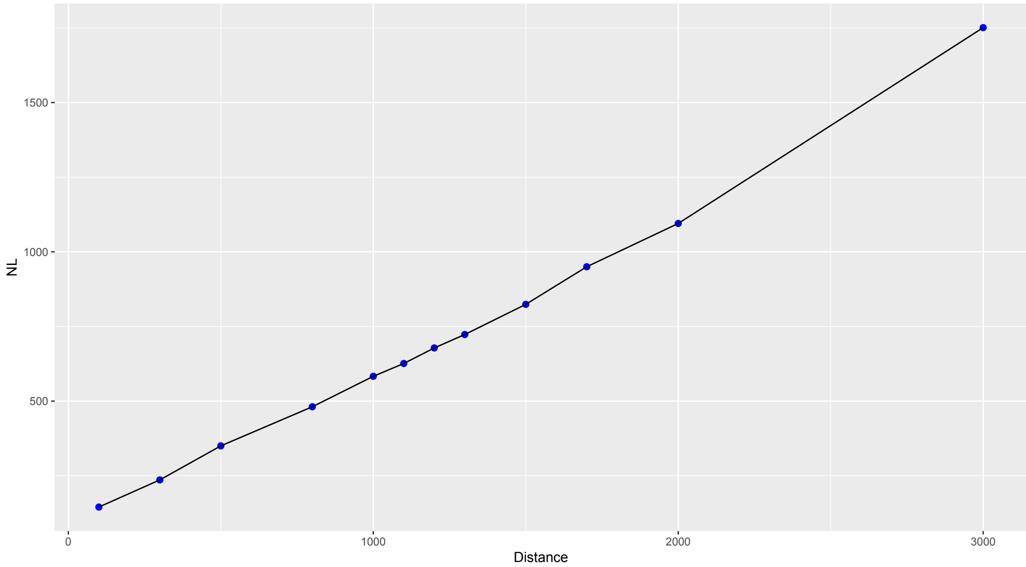


FIGURE 6.5: The Number of Links index (NL) with different threshold distances. As the dispersal distance threshold increases, patches get connected to more patches.

6.4.3 Assessment of components

There were 19 components found for the threshold distance 1000 meters. As we can see from Figure 6.8, component 2 has 102 patches which almost includes all patches along coastal area in eastern part of the city and the southern area of the Port Hills. Component 2 also had the largest area and made up over 75% of the total patch area in the landscape (Figure 6.9). Component 1 was located in and near to the city centre and had 29 patches but a small total area. Another two notably components are components

TABLE 6.2: List of 7 super large(>1 km^2) patches in Christchurch urban area with areas and the dIIC value.

Patch ID	Patch Name	Total Area(km^2)	dIIC
20	Bottle Lake Reserve	10.32	77.27
1379	Bowenvale Park	2.01	8.01
1456	Hoon Hay Valley	1.68	3.15
1457	Living Springs	1.54	1.80
1408	Montgomery Spur Reserve	1.28	10.56
1443	Travis Wetland	1.26	20.68
1452	Kennedys Bush	1.15	2.15

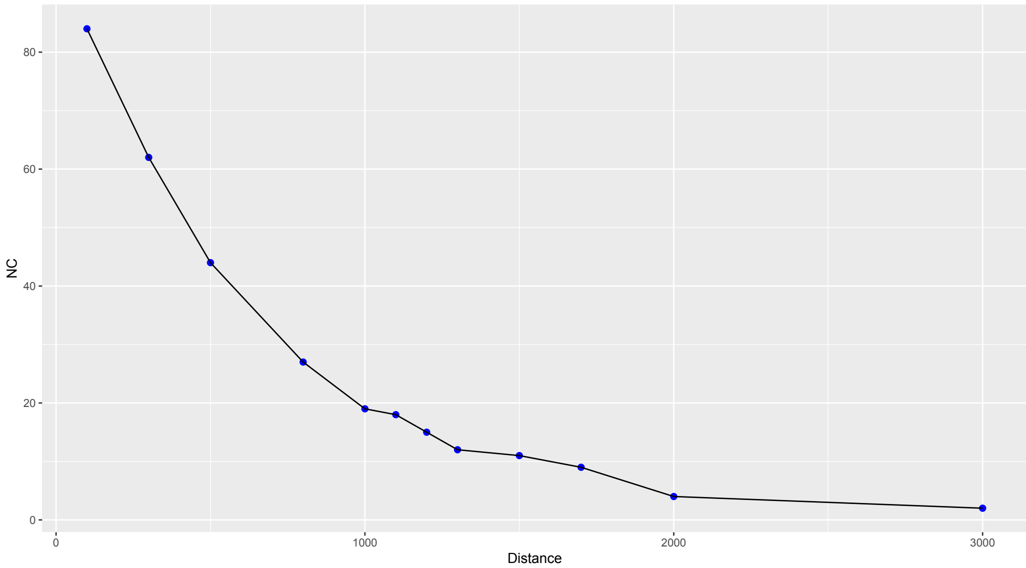


FIGURE 6.6: The Number of Components (NC) with different threshold distances. As the dispersal distance threshold increases, "components" of interconnected habitat patches get connected together to form fewer, larger components.

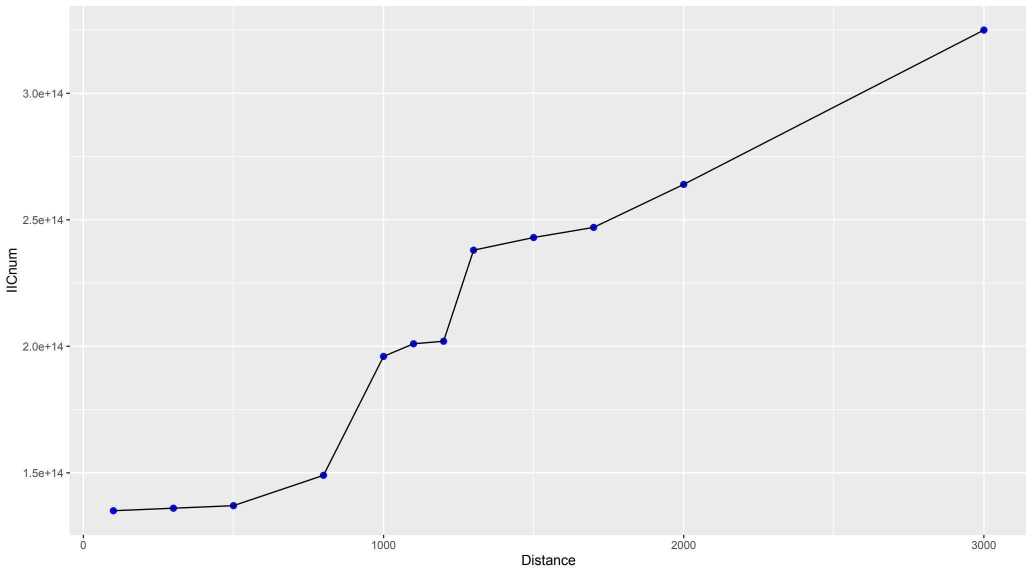


FIGURE 6.7: Result of the connectivity index IICnum with different threshold distances.

6 and 15 (Figure 6.10). Component 6 is in northern of the city just next to component 2 and component 15 is at the southwestern end of the Port Hills. Although component 1 has a few patches, they all are small size and have a low importance value. Thus, component 2 is the dominant component covering a large area and also containing many habitats, including coastal, estuarine, and hill country.

The result of this component analysis changed after putting the Avon River area of the Residential Red Zone as a potential new habitat corridor. This tested the extent to which restoring this area would help to improve habitat connectivity in the city (Table 6.3). Component number of the patch data with Avon river is 2 less than the component number of just patch data, which means the Avon river can work as a corridor which can help to improve the habitat connectivity of the city. Figure 6.11 shows if we can plan Avon river bank area as a corridor with native vegetation, it can link to component 1 which has all patches in the city centre.

Table 6.3: The connectivity indexes values of Christchurch Patches without and with patches from the Red Zone. Result shows patches from the Red Zone can promote the importance value of patches in Christchurch.

Source Patch	NL	NC	IICnum	EC(IIC)	IIC
Chch Patches	583	19	1.960398E14	1.400142E7	0.0000964
Chch Patches with Avon River	718	17	2.001355E14	1.414693E7	0.0000984
Chch Patches with Red Zone Patches	778	17	2.37766E14	1.541966E7	0.0001169
Chch Patches with whole Red Zone	974	17	2.561506E14	1.60047E7	0.0001260

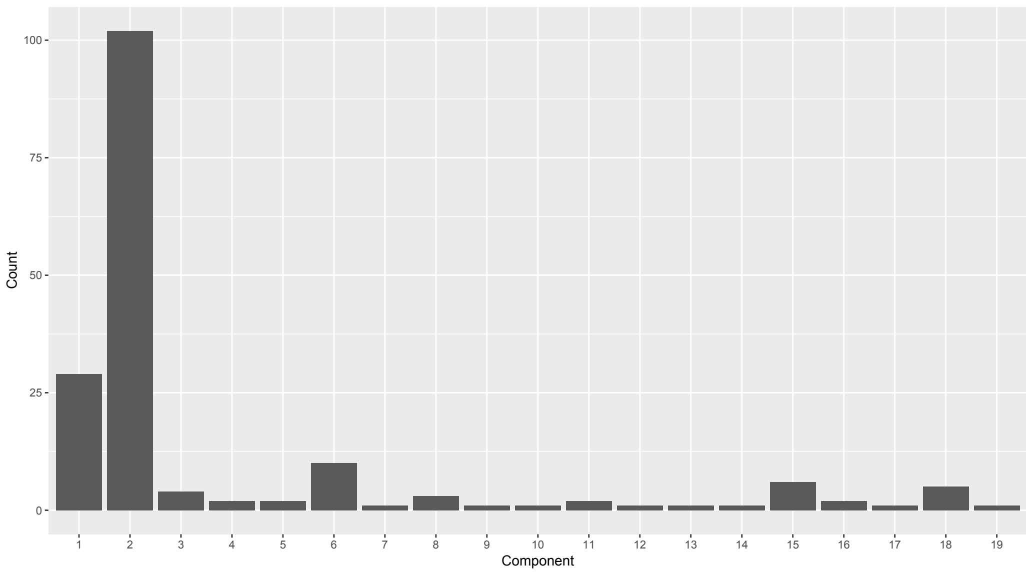


Figure 6.8: Numbers of patches in each component.

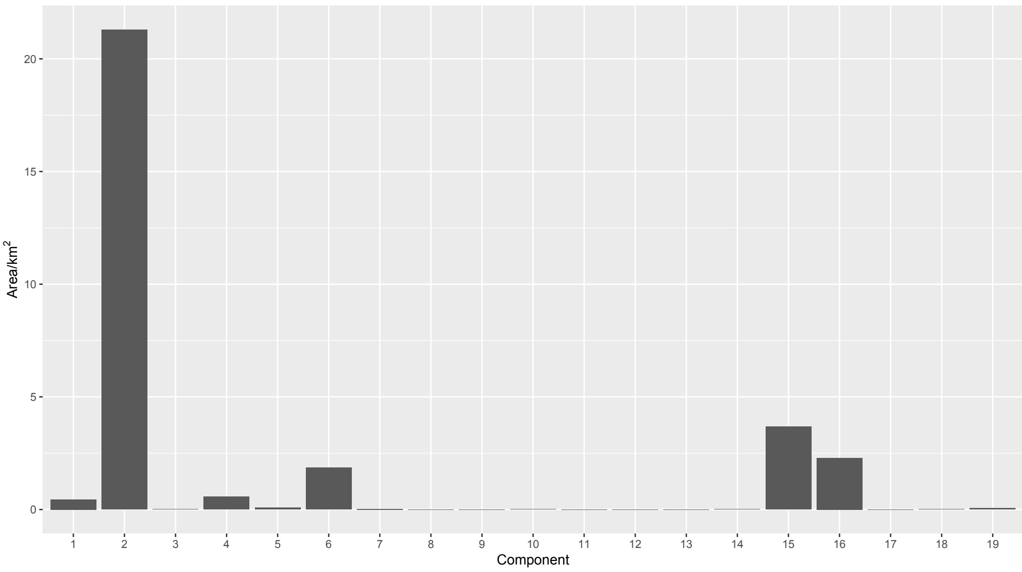


FIGURE 6.9: Total patch area of each component.

6.4.4 Patch Importance

TABLE 6.4: List of some important patches in Christchurch urban area with areas and their dIIC values.

Patch ID	Patch Name	Total Area (km^2)	dIIC
20	Bottle Lake Reserve	10.32	77.27
1443	Travis Wetland	1.26	20.68
1386	Cockayne Reserve	0.06	11.28
1474	Avon-Healthcote Estuary	0.27	10.81
1408	Montgomery Spur Reserve	1.28	10.56
1379	Bowenvale Park	2.01	8.01

Figure 6.12 shows the importance of patches for overall habitat connectivity across the landscape gets smaller from east to west. The bigger patches have a higher importance values. Among them, Bottle Lake Forest, which is the biggest patch by area in this study, has the highest importance value (although this does not take into account that its vegetation is dominated by exotic pines). Travis wetland also has a high importance value because it has a relatively large area but also is an important connector between other patches. There are a few patches such as Avon-Heathcote Estuary which plays a role as a corridor connecting the patches in urban area with the patches on Port Hills. On Port Hills there are several large patches have a high importance value.

I then added patches in Residential Red Zone area, as either a narrow riparian strip along the Avon River (Figure 6.14) or forest restoration across the whole eastern Residential

Figure 6.10: Map of the components based on Christchurch urban patches which contains more than two patches.

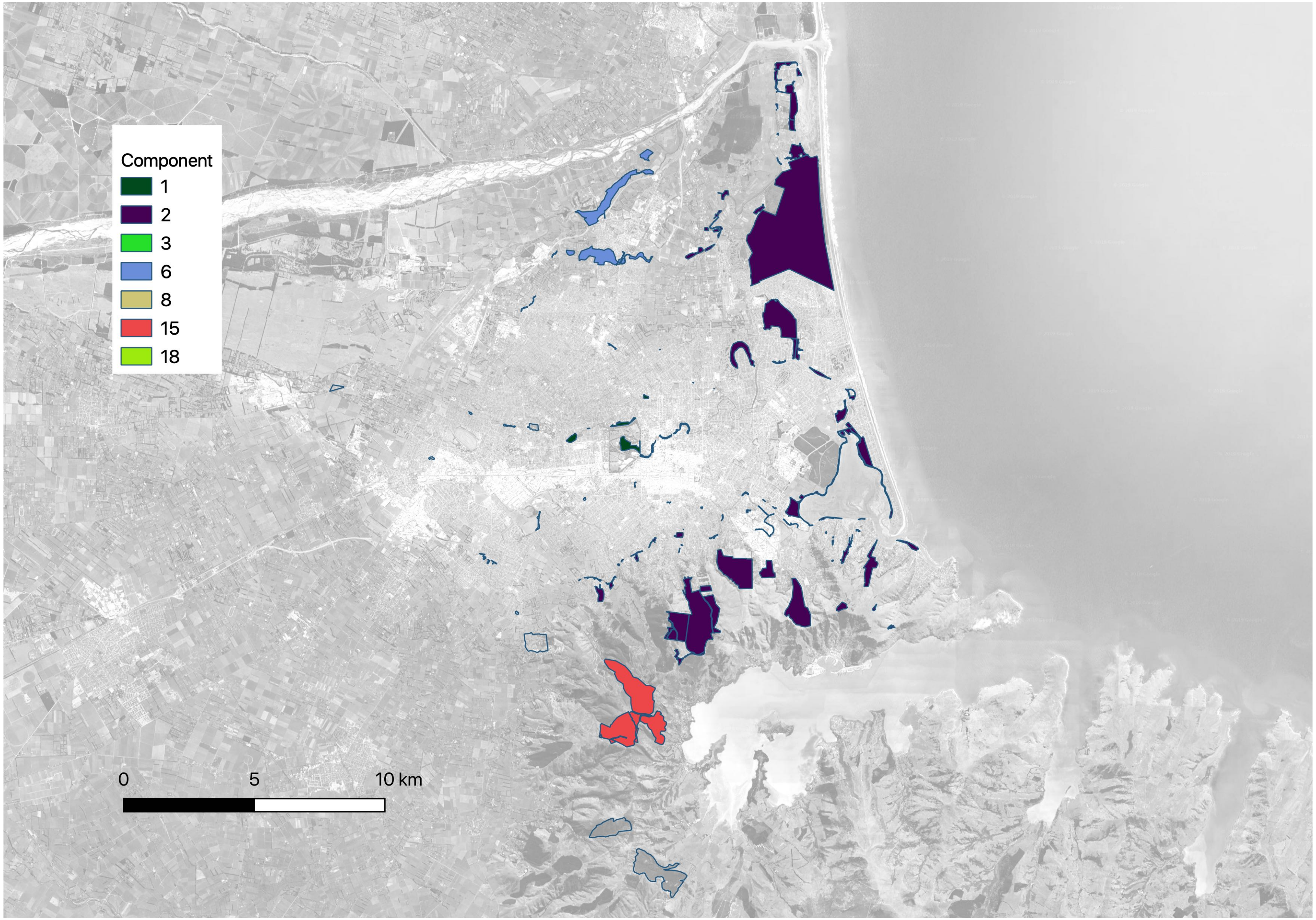
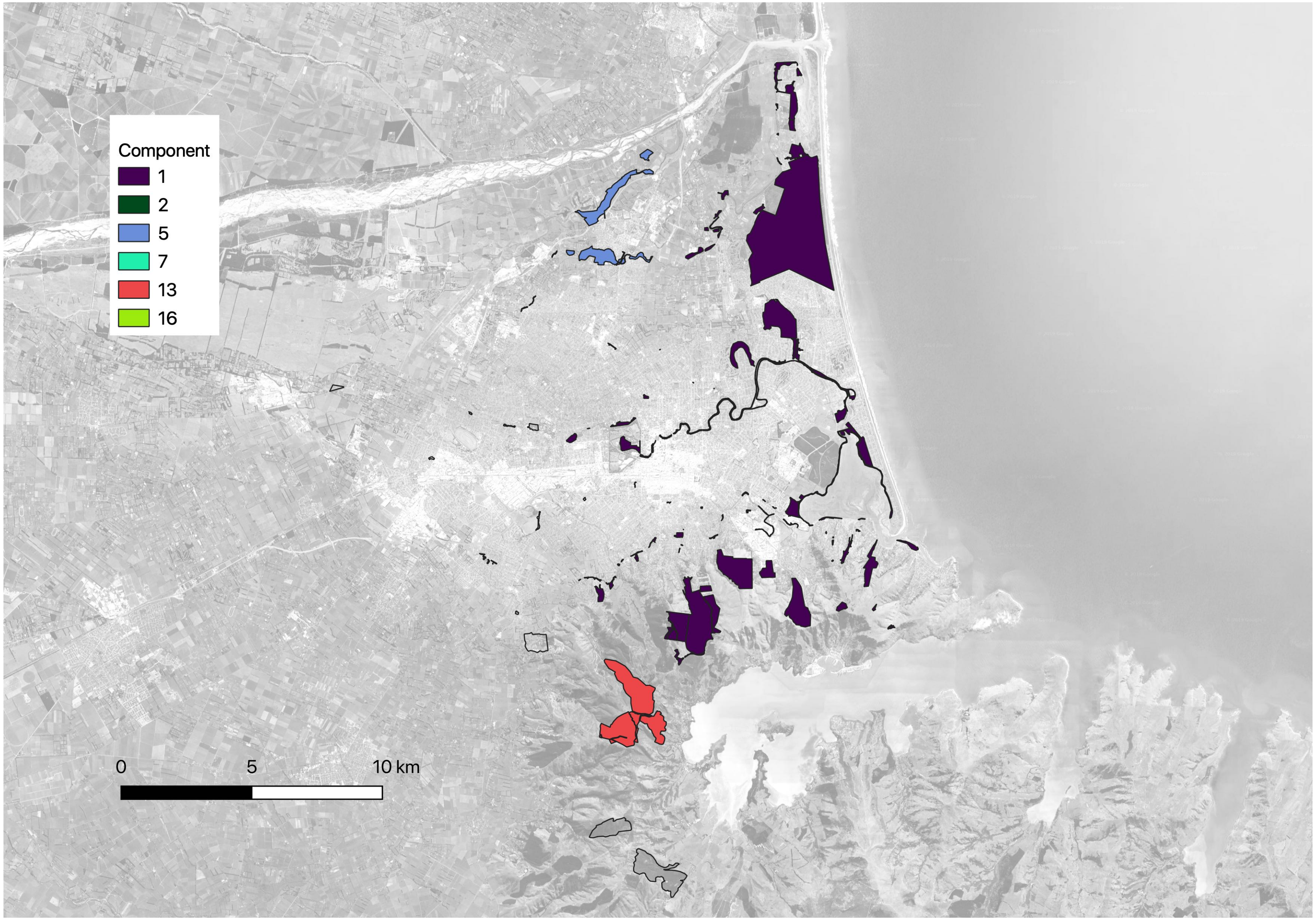


Figure 6.11: Map of the components based on Christchurch urban patches and Avon river which contains more than two patches



Red Zone area (Figure 6.15). The result shows the four patches (Dallington, Horseshore-Lake, Avondale and New Brighton) of Red Zone put into the analysis all have high important values (Figure 6.14). Similarly, in Figure 6.15, different patches of whole Red Zone all have a higher importance value than most of other patches.

Significant changes in the dIIC value of source patches from component 2 happened after I put the Avon River bank as a patch in the analysis. The importance value of Botanic Garden in component 2 jumped from 0.0452 to 0.9309. The importance value of Riccarton Bush jumped from 0.01 to 0.1484. The importance value of Little Hagley Park jumped from 0.0211 to 0.3562. Thus, Avon River is a cut-node which can work as a corridor linking component 1 and 2 (Figure 6.10)and helping the patches in component 2 to maximise their ecological functions.

6.4.5 Discussion

Habitat patches containing substantial indigenous vegetation are typically few, small and scattered in urban landscapes. The degree of patch connectivity from their spatial arrangement can make a big difference to the ecological functioning of these patches. I have shown here that Christchurch City has less than 2.5% of its area in woody areas with substantial indigenous vegetation. Yet, fortuitously, the eastern and southern patches, making up over three quarters of this area, are arranged close enough to together (within 1 km) to maintain functional ecological connections for many species. The challenge for the city, as more forest is restored, is to connect the isolated patches in the centre, northwest and west of the city. The restoration of forest into the Residential Red Zone offers the city an opportunity to do this.

Patch area is a factor which has a significant effect on the importance value(dIIC). As Table 6.1 shows, only seven patches of the 175 patches had areas more than 1 km^2 (Table 6.2). All of these have a high dIIC values compared to most of other small size patches. However, as we can see from Table 6.4, some small patches also have high importance value, such as Cockayne Reserve and the Avon-Healthcote Estuary. Especially Cockayne Reserve, it is only 0.06 km^2 but has a high dIIC value which is 11.28. In areas of the city with little natural habitat, these small patches provide a vital connectivity role. Without them, large components of functionally connected habitat patches would be disconnected. These smaller patches are therefore of high priority for conservation management, and for investment in expansion.

Although there are a lot of parks and reserves in Christchurch urban area (Figure 6.3), most of them are small size and distributed in the eastern part, city center as well as on the Port Hills. Very few of them are in the west of the city, although the Waimakariri

Figure 6.12: Distribution of the patches with different connectivity importance values (dIIC) in the Christchurch urban area.

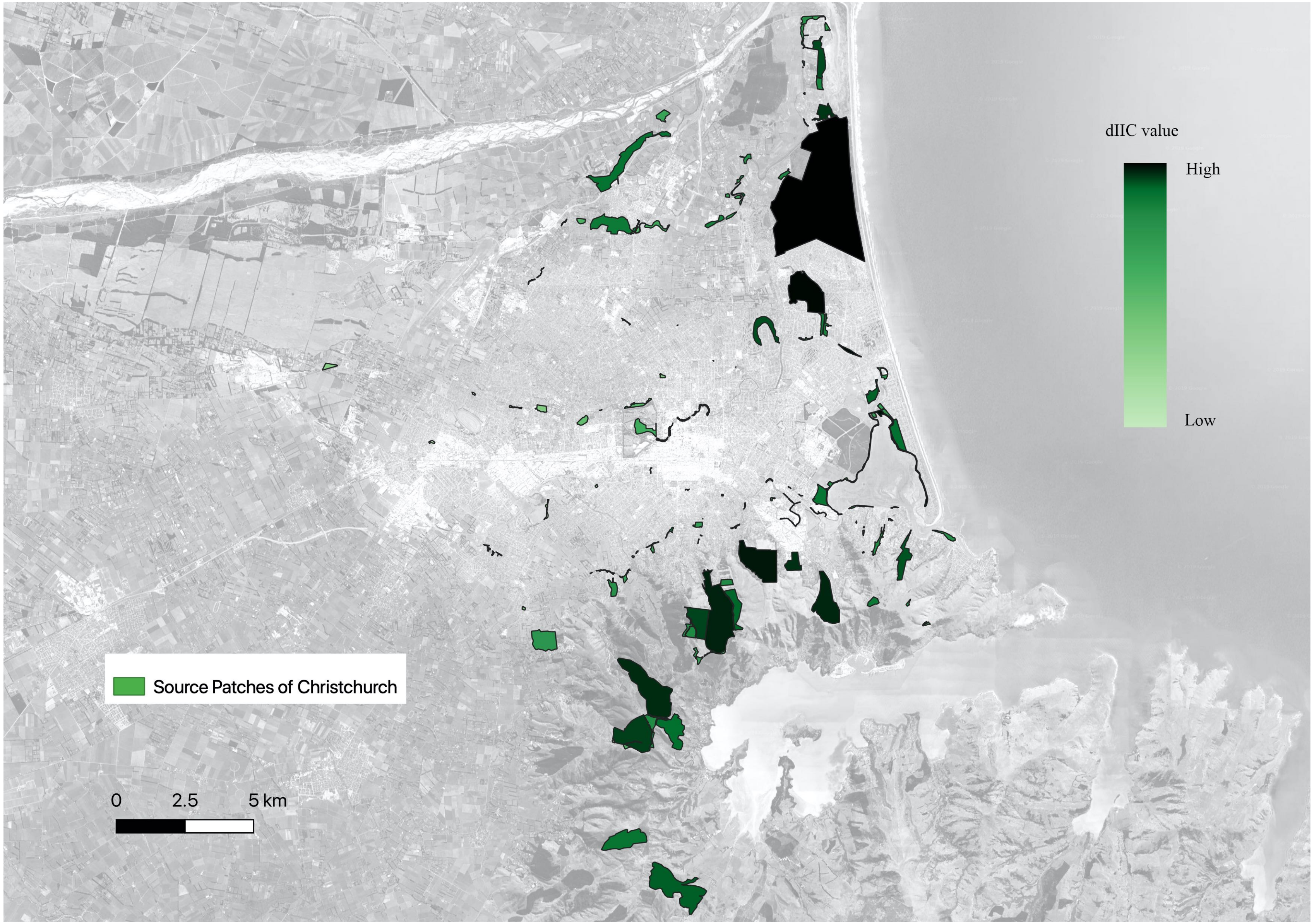


Figure 6.13: Distribution of the patches with different connectivity importance values (dIIC) in the Christchurch urban area, with the addition of a thin riparian habitat corridor restored along the eastern Avon River.

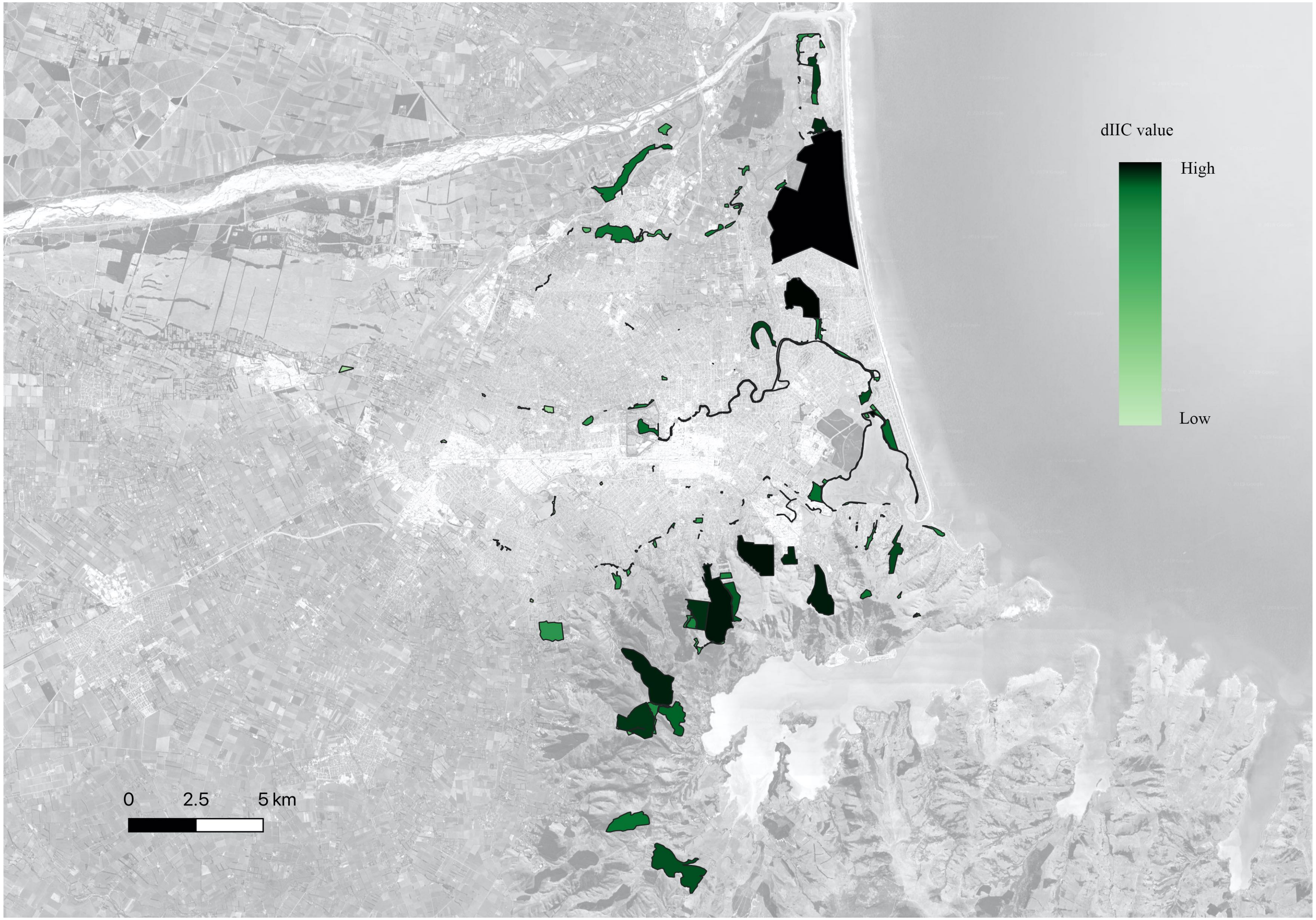


FIGURE 6.14: Distribution of the patches with different connectivity importance values (dIIC) in the Christchurch urban area, with the addition of a thin riparian habitat corridor restored along the eastern Avon River and four forest restoration patches established in the eastern Residential Red Zone.

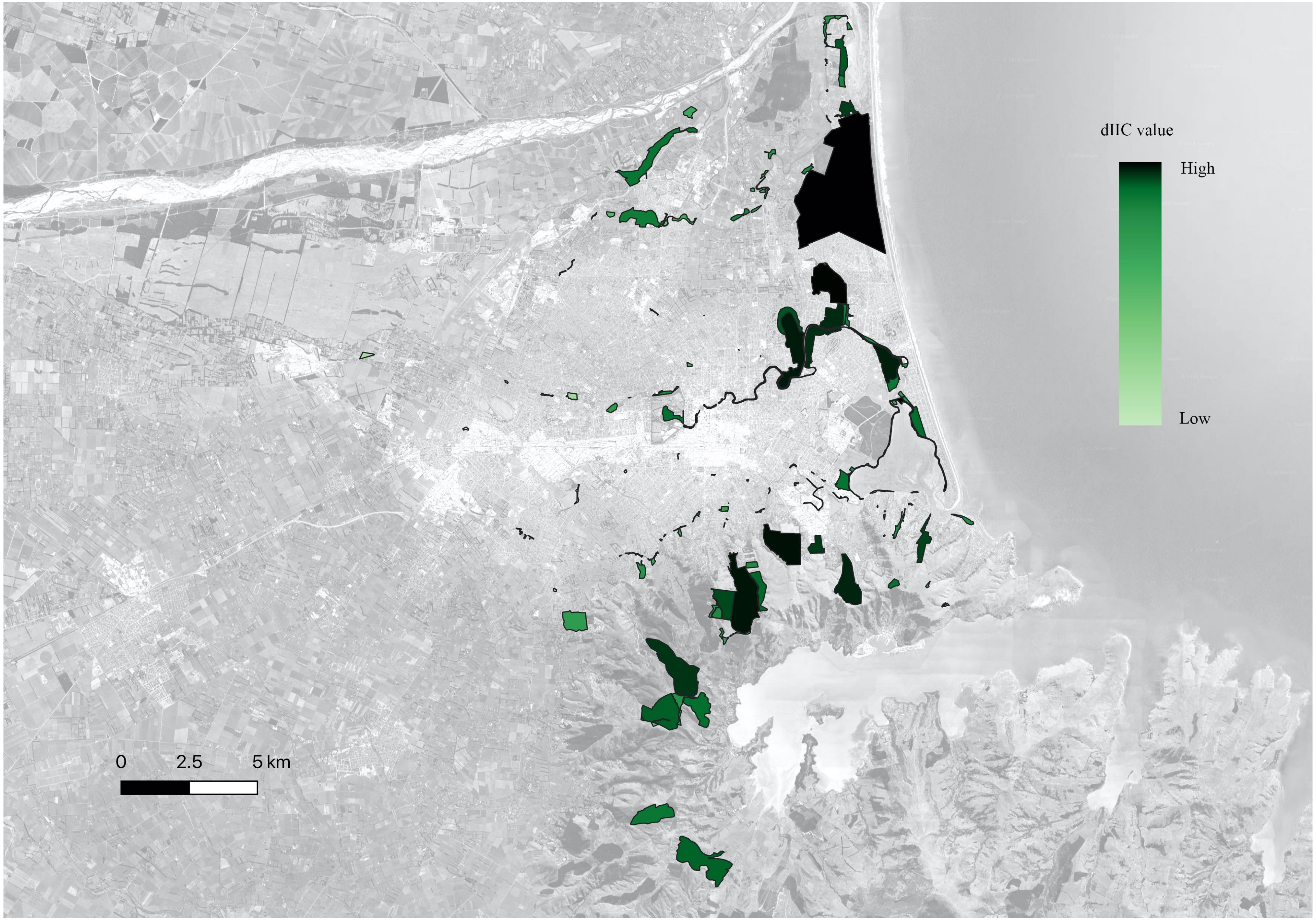
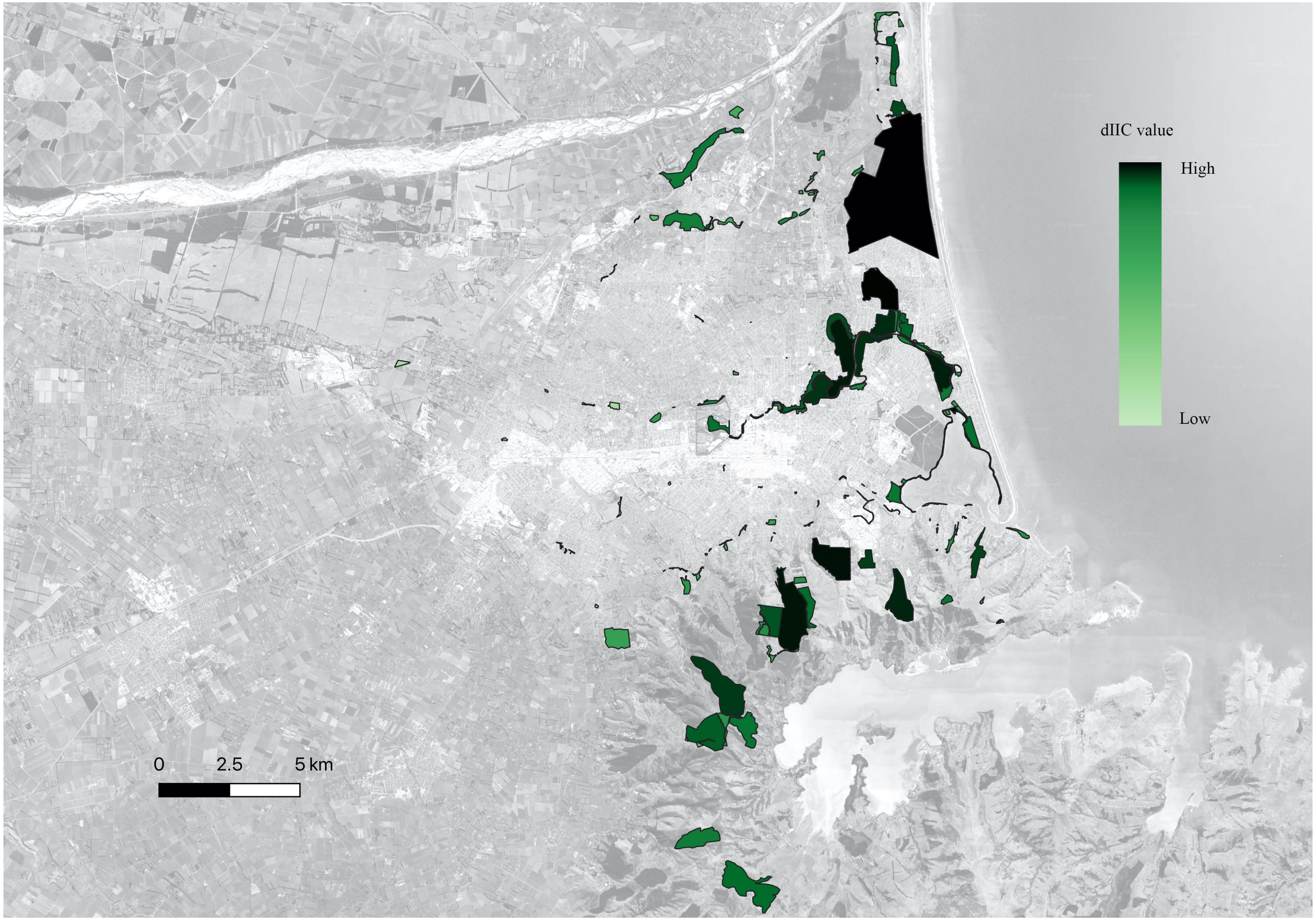


FIGURE 6.15: Distribution of the patches with different connectivity importance values (dIIC) in the Christchurch urban area, with the all of the eastern Residential Red Zone restored as forest.



Regional Park and McLeans Island are present to the northwest just outside my study area. All the patches ranked of high importance based on their area and role in connectivity were in the eastern and southern parts of the city. These are both a problem and an opportunity for the city. From this study, we know it is necessary to build more large source patches in the urban area especially connecting across the eastern part of the city to the city centre and westwards. Based on this, cutnodes are needed to increase habitat connectivity and link together the currently isolated components. There are a few potential patches, such as Marleys Hill, which can be a patch link component 2 and component 15, and the current Port Hills Hub concept of the Banks Peninsula Conservation Trust (`https://www.bpct.org.nz/bpct-2050-ecologicial-vision`) aims to restore these connections. There is no cutnode between component 2 and component 6; the Styx River can be a good corridor link these two component and forest restoration in this area is ongoing (`https://www.thestyx.org.nz/`).

The Christchurch Botanic Garden (especially its Cockayne New Zealand native garden) and Riccarton bush are important patches fully covered by native forest in the city centre and are the important source patches. However, component 1 which includes these patches is disconnected with component 2. My results show that building the eastern Avon River area of the Residential Red Zone as a habitat corridor can help connect component 1 and component 2 and it will make those patches in the city centre more important. Also the patches in Red Zone area all have high important values. They can be important patches to enhance the habitat connectivity. Therefore, the Red Zone offers an important opportunity to improve habitat connectivity in Christchurch urban area.

Graph theory is an efficient tool for landscape connectivity (Pascual-Hortal & Saura, 2008). It is a heuristic framework working with very little data that can provide valuable results (Bunn et al., 2000). By identifying the important patches, it provides helpful guidance for biodiversity conservation planning and connectivity improvement (Devi et al., 2013). Here, patches that are not physically linked can still provide functional connections between patches (Pascual-Hortal & Saura, 2006). That works for New Zealand native tree species which are mainly dispersed by birds and also wind. A connectivity index also gives information guiding where it is possible to connect between patches; the connectivity importance value of patches can be increased through both the restoration of new patches and enlarging existing patches (Devi et al., 2013).

Based on my results, regardless of measurements of link numbers and patch sizes, the management of the urban forest should also consider the composition and structure of the vegetation in and between patches (Pascual-Hortal & Saura, 2008). 1,000 meters was the threshold dispersal distance I used, but that is sufficient for dispersal by both many native species and many naturalised species. Enhancing the connectivity and enlarging the patch size can help some of the source patches with fully native bush cover to exchange species and regenerate well. For some of the large patches infested with weeds, it is important to manage the native species regeneration and suppress weed spread.

Habitat availability is one of the characteristic of the landscape which integrates both habitat area and connectivity (Pascual-Hortal & Saura, 2006) and it is suggested that this habitat availability concept should be used in the conservation decision making to get a successful integration of connectivity considerations (Pascual-Hortal & Saura, 2006).

Habitat quality of source patches were disregarded in the connectivity metrics in this study. However, several researchers have recognized that quality of source patches can impact the species extinctions, coloniztions and occupancy (C. Thomas, 1994; Hokit, Stith, & Branch, 1999; Fleishman, Ray, Sjögren-Gulve, Boggs, & Murphy, 2002; Armstrong, 2005; Thornton, Branch, & Sunquist, 2011) and have a better prediction for most systems (Schooley & Branch, 2011). Visconti and Elkin (2009) found connectivity metrics with patch quality worked better in all scenarios. When environmental factors become important to habitat quality, the weighting method can be applied to convert patch areas to effective patch areas (Hanski, 1994; Schooley & Branch, 2011) and this have been applied in several researches (Hokit, Stith, & Branch, 2001; Moilanen, n.d.; G. Rabasa, Gutiérrez, & Escudero, 2007; Jaquiéry et al., 2008; Cosentino, Schooley, & Phillips, 2010).

6.4.6 Conclusions

Urban forests are important for urban ecosystem. In order to improve native biodiversity and its regeneration, it is necessary to incorporate habitat connectivity in current urban forest planning and management. Based on graph theory, the integral index of connectivity (IIC) and the analysis method can evaluate the contribution of the individual patches and identify priority areas for making new patches.

The result of my source habitat connectivity analysis shows that the Christchurch urban area has a low degree of connectivity with lots of patches are small, isolated and fragmented. Most of the source patches with high importance values are in eastern part of the city as well as south on the Port Hills; together these form one habitat component. In contrast, patches with good native vegetation in the city centre are isolated and need to be reconnected.

Patch importance value is related to patch area and the number of links to other patches. Large size patches always have a higher importance values but some small patches are also important as they are cutpatches. Thus, it is very important to keep these large patches as healthy native bush but it is also necessary to manage and build important small patches between large patches to be the cutpatches linking between large patches.

Greening the Residential Red Zone offers a good chance to enhance the urban habitat connectivity in Christchurch city. As the results have shown, the eastern Red Zone around the Avon River can be a corridor which connects two currently isolated components in the city. Because of this, restored patches in Red Zone area will have high importance value for the city's habitat connectivity.

Chapter 7

Scenarios of open space and vegetation management for the Ōtākaro-Avon Red Zone

7.1 Introduction

New Zealand has a high level of endemic city in its native biodiversity, with around 80% of the approximately 2,500 native vascular plant species present in New Zealand being endemic (Wilton & Breitwieser, 2000). However, in the last 200 years, there has been a big group of exotic flora and fauna imported by Europeans to New Zealand (Druett, 1983; Dawson, 2010) and they have had deeply impacted New Zealand native biodiversity (Thomson, 1922a; Allen & Lee, 2006). Since humans settlement, approximately 70% of the New Zealand land environment has been substantially modified, causing the extinction of 32% of native land and freshwater birds, 18% of endemic seabirds, three of seven species of frogs, and at least 12 invertebrates (Statistics, 2002; Holdaway, 1999; Duncan & Young, 2000; Duncan & Blackburn, 2004; Collins et al., 2014). At least six plant species have become extinct and more than 184 species are naturally threatened (De Lange et al., 2018).

New Zealand native forest cover area ha reduced since human arrival. 80%-90% of New Zealand was covered by native forest before human arrival (McWethy et al., 2010). This had dropped to around 70% of the land area by 1800 (Ewers et al., 2006). More than 40% of the native forest had been destroyed by the 1850s (McGlone, 1983, 1989; McWethy et al., 2010). Moreover, between 1847 and 1909, more than 60% of the remaining forest (4.5 million ha) was destroyed (McGlone, 1983).

Native forest plays an important role in the New Zealand ecosystems. It provides native fauna with primary habitat and food sources and in turn, native fauna can help to pollenate native plants and disperse the seeds (Rastandeh, 2018). Loss of native forest in urban areas is a threat to the long-term sustainability of native urban biodiversity. Now only 1.96% of current urban New Zealand is covered by native forest (Clarkson et al., 2007). Therefore, it is important to restore as much native forest in urban areas as possible in order to maximise the support and provision of ecosystem services (Rastandeh, 2018).

Studies of urban flora changes over more than 100 years suggest that while the total species richness increases, the native species richness declines (Kowarik, 2011; Knapp, Kühn, Stolle, & Klotz, 2010). The world's cities have become hotspots of exotic species expansion. On average 40% of urban flora are exotic in 54 European cities and the proportion ranges from 20% to 60% (Pyšek, 1998). There is a similar range in some North American cities and an average of 35% (Clemants & Moore, 2003).

One of the main reason for enhancing conservation work in urban areas is that it can contribute to biodiversity conservation (McKinney, 2002; Miller & Hobbs, 2002). Urban areas can actually harbour some rare and endangered species, where they do not just occur but can also become self-sustaining. It has been found in Berlin that some Red List species can establish and self-sustain in urban areas (Kowarik, 2011). However, urban areas cannot act as habitats for a broad range of native species, especially rare species which are highly sensitive to fragmentation (Kowarik, 2011). This finding underscores that it is really necessary to protect, expand, and re-connect remnant patches to conserve biodiversity.

Ecological restoration in urban areas can be a solution to recover degraded ecosystems (Likens & Cronon, 2012) and also to reconnect people to nature (Miller, 2006). Urban ecological restoration is becoming regarded as a major strategy for increasing the provision of ecosystem services and reverse biodiversity loss (Bullock, Aronson, Newton, Pywell, & Rey-Benayas, 2011).

For example, Riccarton Bush is the only forest remnant in the built area of Christchurch City, but by the 1970s it had been degraded by inappropriate management, including mowing and weeding in the forest understorey (Molloy, 1995). To repair the structure and composition of the bush, a planting program began in1975 (Molloy, 1995). In the rehabitation areas, all introduced trees were removed and some fast-growing native species were selected to planted there. These native species include evergreen species hardwoods such as Black Matipo (*Pittosporum tenuifolium*), Lemonwood (*Pittosporum eugenioides*), Karamu (*Coprosma robusta*), Narrow-leaved lacebark (*Hoheria angustifolia*) and Māhoe (*Melicytus ramiflorus*), the deciduous Ribbonwood (*Plagianthus regius*

) and the semi-deciduous Wineberry (*Aristotelia serrata*) (Molloy, 1995). The planting programme was highly successful (see Figure 7.1 and Figure 7.2) because the planted native trees have grown up to naturally regenerate and most of the introduced tree gaps were filled by native trees (Molloy, 1995).

FIGURE 7.1: The planting work in south-eastern boundary of Riccarton Bush in 1978. Left: the exotic trees were removed in 1975 and young native trees were planted in 1978. Middle: five year growth of planted Lemonwood, Matipo, Karamu, Ribbonwood and Narrow-leaved Lacebark. Right: the same view in 1989 (Molloy, 1995)

FIGURE 7.2: The planting work in another place of Riccarton Bush in 1978. Left: an extensive grassy clearing in 1978. Middle: the same view in 1982 with rapid growth of Matipo, Lemonwood and Karamu planted in 1979. Right: the same view in 1989 with marked changes (Molloy, 1995).

Christchurch has an opportunity to create a substantial new area of indigenous forest within the city, in the Residential Red Zone of eastern Christchurch City. After the 2010–2011 earthquakes, this area had its houses removed, with all larger garden trees retained, and it has been maintained as an open woodland space. Decisions are now being made on what to do with this land. In this chapter, I use my findings from my previous chapters to suggest several restoration and management scenarios for the Residential Red Zone.

7.2 Foundations of the scenarios

7.2.1 A framework for applying the study results

Urban ecosystems are complex (Tjallingii, 1995) and in order to understand the condition and trends and to reverse degradation, it is important to identify the interactions among the many components of the system (Y. Zhang, Yang, & Yu, 2006). This includes ecological and sociological factors.

Figure 7.3 gives an overview of the questions my research can answer to help inform appropriate scenarios and management options for the Residential Red Zone. In Chapter 2, I explored some of the historical and social factors that influence which trees are planted in suburban gardens. This creates a foundation from which wild regeneration can occur (of both indigenous and exotic species). In Chapters 3 and 5 I documented the ecological (and historical) factors affecting seedlings regeneration in Christchurch forests. This shows what can be planted in the future Red Zone, and what species can be left to naturally plant themselves. In Chapter 4, I document the dispersal distances being achieved by trees in urban Christchurch, which helps to inform how far apart restored forest patches should be in planned forest restoration. In Chapter 5, I repeated a past survey of restoration sites which tells us how future Red Zone regeneration might vary with different environmental conditions and management scenarios, and how slowly natural forest processes like seedling regeneration appear in planted urban forests. Lastly, in Chapter 6, I assessed the patch connectivity of the Christchurch urban area and identified priority areas for forest restoration in the Christchurch city area. This connectivity analysis highlighted the importance of restoring forest in the Residential Red Zone for reconnecting the currently isolated forest patches of central Christchurch with the better connected network of patches along the coast and into the hills.These results in combination work together to give a design frame for future Red Zone regeneration patterns.

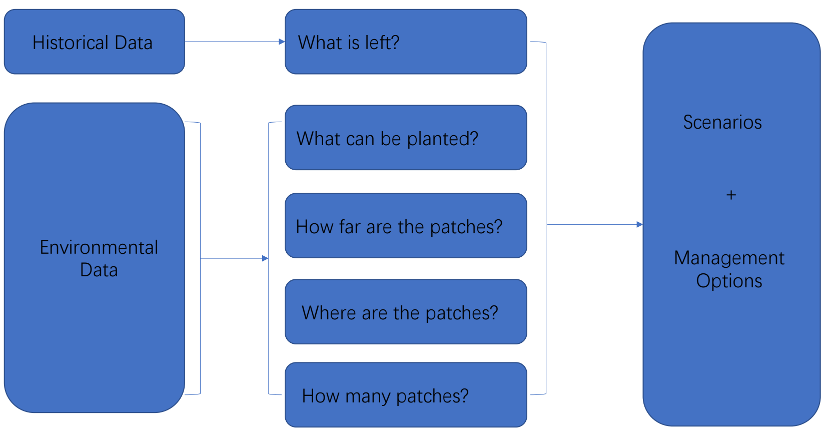


FIGURE 7.3: Overview of how my research can be used to inform the management of the Ōtākaro-Avon Red Zone.

This overview sets up my use of social/historical, biological and environmental data to answer key questions about plant and patch ecological dynamics and informs the interactions between the three main components in urban ecosystem. My framework in Figure 7.4 summarises how I see these factors coming together. (Fungi are an additional element but the details of their role in urban NZ forests are poorly understood and so I have excluded them here.) In this framework, humans, wildlife and native plants are three components that feature in design targets. The overlapping parts are the functional elements which include more than one component. For humans and wildlife the "discover" area facilitates people watching and enjoying wildlife in the city, and includes such things as the bird observation tower in Travis Wetland. For native plants and humans, "education" elements will can help humans to get close to the nature. It could be education center and native plants museum. The area for wildlife and native plants is "supply". Native patches can supply the food sources and habitat to wildlife and in turn, wildlife can help pollinate the flowers and disperse the seeds of native plant. The core of the diagram is the native patch, which includes all the functions of all three components needed.

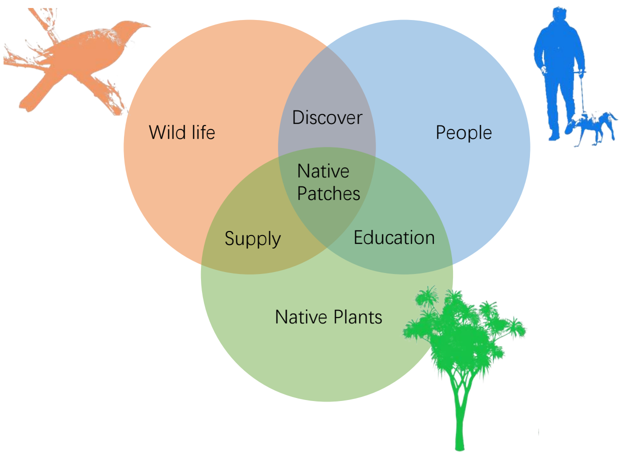


FIGURE 7.4: Three interacting components of the urban ecosystem in this study. Blue colour represents people, orange colour represents wildlife, green colour represents native plants.

7.3 Guiding ecological principles for restoration plan in the Red Zone

Planning native habitats in urban areas is a primary task for urban biodiversity restoration. It is far easier to plan and establish connectivity before the houses are built rather than retrofit ecology into suburbia. Because of urbanisation, habitats in urban areas become fragmented and isolated. As can be seen from Figure 7.5, most of the area in a city centre is building hard surface. Moreover, native biodiversity restoration needs to be based on the species which are appropriate to the environmental conditions. Based on past ecological research and my thesis research, I list below five fundamental ecological principles about site, tree species and patch patterns necessary for effective native patch restoration.

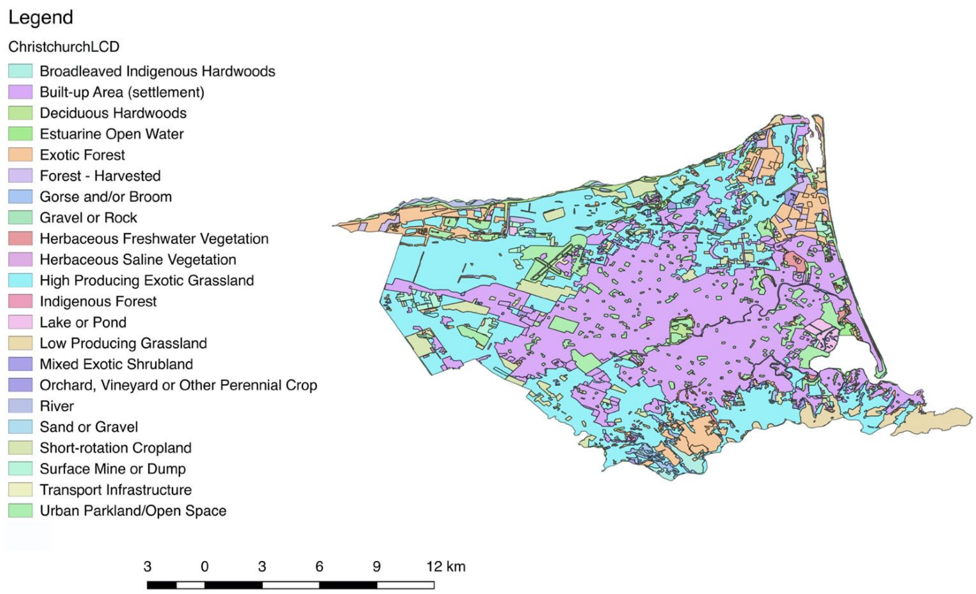


Figure 7.5: Land cover classes present in Christchurch based on Land Cover Database, LCDB4 (Landcare Research, 2015).

7.3.1 Principle one: Simplify patch shape and maximise patch area

Edge effects

Patch size and shape are important factors affecting species diversity (Fahrig, 2003; Fletcher, Ries, Battin, & Chalfoun, 2007; Yamaura, Kawahara, Iida, & Ozaki, 2008).

The positive relationship between patch area and species richness is well known (Peay, Bruns, Kennedy, Bergemann, & Garbelotto, 2007). Large patches will have greater population sizes which reduces the extinction rates (Russell, Diamond, Reed, & Pimm, 2006) and provides potential colonists (Lomolino, 1990). Large patches are also more complex, which helps increase species number (Russell et al., 2006). Because of edge effects, the densities of core-dwelling species will decrease in the patches with narrow linear shape (Ries, Fletcher Jr, Battin, & Sisk, 2004; Yamaura et al., 2008).

Soga et al., (2013) made several excellent suggestions about the patch size in urban habitats. In general, a single large habitat patch is more valuable than several small habitats of equal total area (Soga, Kanno, Yamaura, & Koike, 2013). However, it is difficult in urban area as most of the native patches are small sizes and isolated. Thus, simplifying the shapes of forest patches to minimize edge effects and to maximize the core areas could be the most realistic work for restoring native biodiversity in urban area (Soga et al., 2013).

Maximise the core area

In this principle, I use a circle graph to represent the three types of land use (vegetation type) planned for the Red Zone (Figure 7.6). The red circle means a core zone, which is restored native forest. The green border is a buffer zone and may be exotic-native mixed forest/woodland garden. The blue one is the open spaces with infrastructure and facilities. The importance of the zones is getting smaller from inside to outside. Native bush core is the most important for indigenous urban biodiversity and it should be kept as the threshold area and location. Appropriately positioned buffer zones can make the restored native forest functionally larger.

7.3.2 Principle two: Choose the ecologically appropriate plant species

Native or exotic?

Species selection is critical to functioning ecosystem supporting native wildlife. In order to restore a self-sustaining ecosystem which focus native biodiversity, native species are always chosen rather than exotic species (E. Thomas et al., 2014). There is a belief that native species from that ecological area can adapt to the local environmental conditions better and they can support native biodiversity and ecosystem services better than exotic species (Tang et al., 2007). Exotic species may also play a role in restoration process, especially as the first species to recolonize after initial vegetation removal (D'antonio & Meyerson, 2002). They can then act as nursery for native regeneration, such as nectar

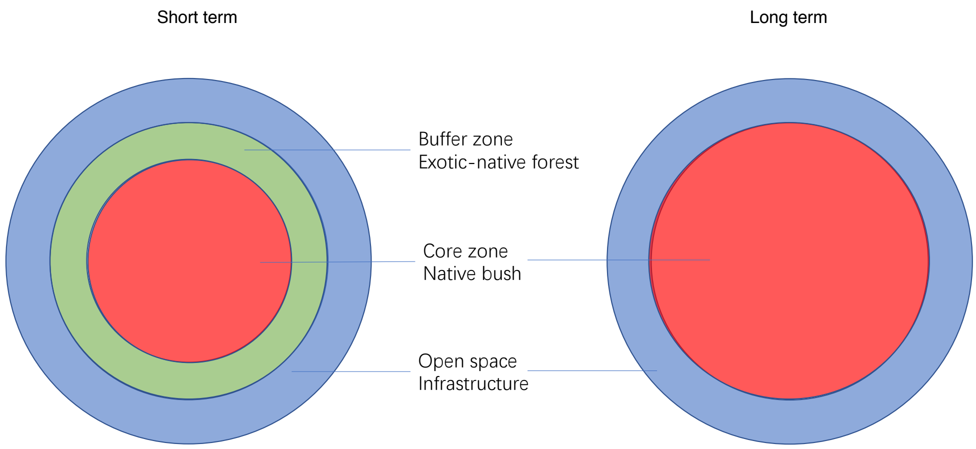


FIGURE 7.6: The circle graph shows the vegetation patch design for the Ōtākaro-Avon Red Zone. Red color represents core zone which is native bush. Green color represents the buffer zone which is exotic-native mixed forest. Blue one is the space for facilities and infrastructures. In the long term plan, the buffer zone will become to be the native bush.

supply for honey-eating bush birds (e.g. Australian gum trees). They even can be used as a "nurse" crop to restore particular functions if native species are not available (D'antonio & Meyerson, 2002). However, species known to be 'noxious' or invasive should never be used and/or should be eliminated before they taking hold.

Local native or non-local native?

Any non-local species, even non-local native species, all have risks (E. Thomas et al., 2014). There is a risk of genetic pollution if the planted species are the non-local species which are closely related to the species in the habitat but have difference genetically(same genus or cultivation) (Ellstrand & Schierenbeck, 2000; Ferkins, 2001; Rogers & Montalvo, 2004; McKay, Christian, Harrison, & Rice, 2005; Millar, Byrne, Nuberg, & Sedgley, 2012).

7.3.3 Principle three: Nature models dictates vegetation composition and structure

Ecological restoration has been defined as the 'process of repairing damage caused by humans to the diversity and dynamics of indigenous ecosystems' (Jackson, Lopoukhine, & Hillyard, 1995). As mentioned for ecological restoration, the best thing is to choose species which grow fast and can provide early cover for other species establishing, and then nature will sort out later community composition and structure (Norton, 1997). Planting should be planned to facilitate the natural processes of forest formation, not replace them.

Use succession theory in the restoration trajectory

Ecological succession is the progression of community composition structure and dynamics over time (Putnam, Leonardi, & Nanetti, 1994). Restoration should be working as a system with predictable directional change in structure during community development (Palmer, Ambrose, & Poff, 1997). In this scenario, we could intervene in natural succession processes. Manipulations can bypass some of the natural succession stages, and can also accelerate natural succession (Palmer et al., 1997).

Specific disturbance regimes can increase species richness

Specific disturbances in the restoration work are required to help some native species establish (Hobbs & Norton, 1996). In some cases, restoration just responds to the disturbance that has already happened. In other cases, disturbance may be needed to help the restoration move from one meta-stable state to another (Hobbs & Norton, 1996). A good example is that managements will be necessary to stop non-native species invading the restoration site (Hobbs & Norton, 1996). More specific disturbances may also be needed to promote germination of native species at some stages of regeneration (Hobbs & Norton, 1996).

Native species colonisation

Natural colonisation is one of the main processes that aids restoration (Luken, 1990), and can occur if a seed bank persists in the soil (Putwain & Gillham, 1990; Bellairs & Bell, 1990). Further colonisation will be influenced by the surrounding areas, especially for species dispersed by birds (McClanahan & Wolfe, 1993; Robinson & Handel, 1993). Natural sources of some species may be so far away that translocation/planting is necessary to return them to a restoration project.

Reference sites for restored patches and restored patches becoming reference sites

It is usual practice to identify reference sites (models) from existing sites before the colonisation of humans, or settlement by Europeans (Hobbs & Norton, 1996). Remnants of historic natural areas or naturally restored areas are often chosen as reference sites (Choi, 2004). Increasingly, long-established, planted vegetation (>30 years), which has achieved functional stability, can be used as reference sites for what will grow where.

7.3.4 Principle four: Forest patch pattern

Reserve design theory shows that big and compact patches are preferable for biodiversity conservation. However, in urban areas, different land uses constrain the total patch area (Meurk & Hall, 2006) and smaller patches are more realistic. Because most of the New Zealand wildlife are either small or vagile, small patches can supply ecological services to them (Henle, Davies, Kleyer, Margules, & Settele, 2004). The wood pigeon (*Hemiphaga novaeseelandiae*) is recorded as an example (Meurk & Hall, 2006). They now nest in a large exotic parkland in the Christchurch Botanic Garden which as about 30 ha and they visit Riccarton Bush, a smaller indigenous forest remnant, for food (Meurk & Hall, 2006). Thus, Meurk and Hall (2006) promote a greenspace planning pattern for cultural landscapes (Meurk & Hall, 2006).

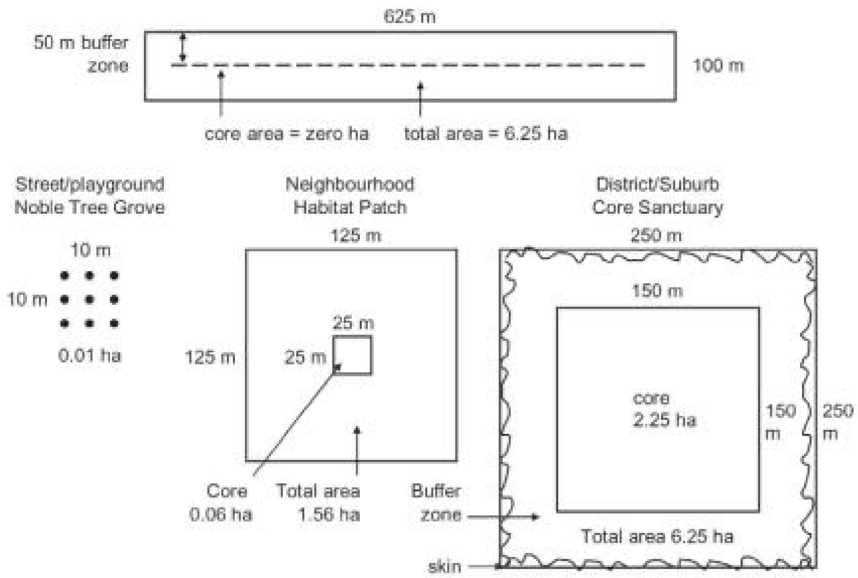
Forest patch shape

Several reasons have been listed to determine the patch sizes. The distance of edge effect penetrating into a forest patch in Auckland is typically at least 50 meters (Young & Mitchell, 1994; Davies-Colley, Payne, & Van Elswijk, 2000). The minimum necessary patch size for sustaining populations of sensitive native forest core species is therefore close to 2 ha (Meurk & Hall, 2006). Total area of 5 ha patches at 5 km spacing amount to 0.29% and when the smaller patches are added (Figure 7.7) the total cover area represents ca. 2% cover. Local government sets a target of less than 10% of subdivision areas for reserve contributions (Meurk & Hall, 2006). So this fits easily into this expectation given that the remarked (8%) may be available for open space and sports fields.

Based on these, figure 7.7 shows the effects of different patch shapes. There are four types of patches with different sizes. The first one is a core sanctuary (6.25 ha) with a core area (2.25 ha) and a 50 m wide buffer zone. The second one is a neighbourhood habitat patch which can be a habitat for most native species. The total area of it is 1.56 ha and the core area is 0.06 ha. The third one is a noble tree grove with the area of

0.01 ha and the distance between trees of 10 meters. The last one, a linear patch, has no core forest habitat.

Figure 7.7: Patch size pattern. Linear patch without core are has a 50 meters wide buffer zone. Three patch sizes are accommodated from left to right by the increased population and sensitivities of native species (used with permission) (Meurk & Hall, 2006). The point is that the linear patch (top) and the large compact patch (right)have the same area (not drawn to scale) but shows the effect of shape on core area.



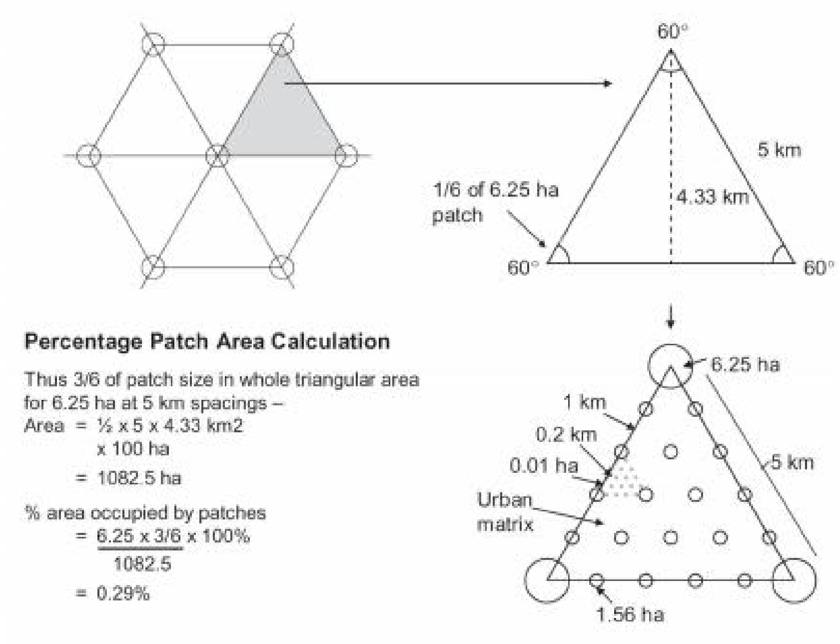
Nested forest patch configuration

A nested forest patch configuration that accommodated these metrics was proposed by (Meurk & Hall, 2006). Their proposed patch configuration shows the distances between different sized patches across a landscape (Figure 7.8). The distance between large patches with substantial core sanctuaries is 5 km. However, in-between these are neighbourhood habitat patches, spaced 1 km apart, and between these are tree groves spaced 0.2 km apart. This kind of patch-corridor-matrix pattern can provide the major ecological service for most native biodiversity (C. D. Meurk & Swaffield, 2000).

7.4 Native patch structure and scenarios for the Red Zone

An objective of restoration in the Ōtākaro-Avon Red Zone is to restore native biodiversity by building a range of quality native patches, enhancing habitat connectivity, establishing native species and improving ecosystem functions. The research here and

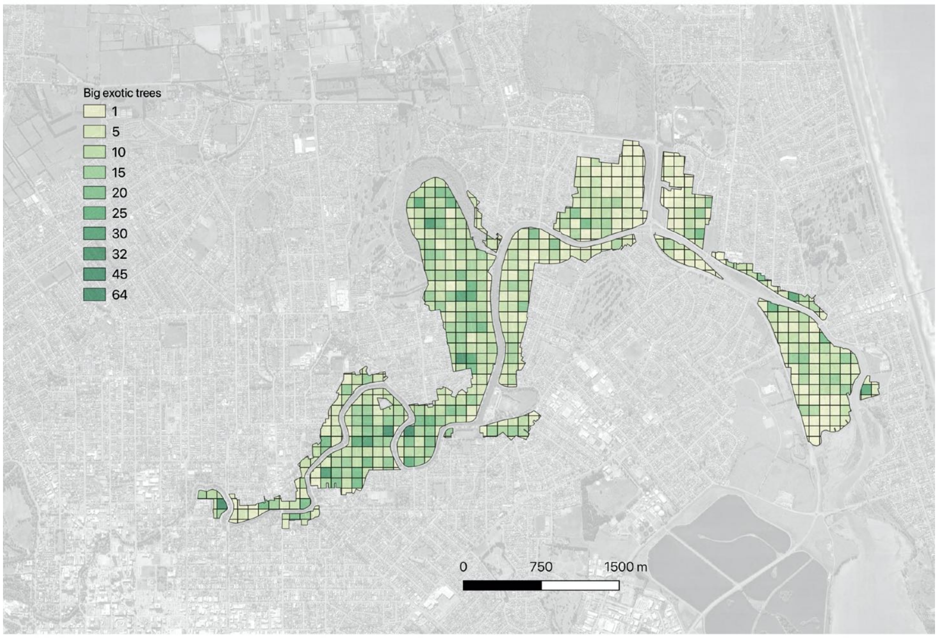
FIGURE 7.8: A nested forest patch configuration of three patch sizes (used with permission) (Meurk & Hall, 2006).



from the literature is combined to draw up a series of scenarios for the future of Ōtākaro-Avon Red Zone. The results of my previous chapters can be brought together to propose restoration scenarios for this landscape. My habitat connectivity study (Chapter 6) shows that the native patches in the Christchurch urban area are fragmented and isolated. The eastern Red Zone has the potential to work as an important habitat corridor and four sites have been identified as important patches. My seedling dispersal study (Chapter 5) gives the threshold seedling dispersal distance of 1000 meters, and most of native seedlings establish within this distance. This compares with the wider limit of 2.5 km described in Meurk and Hall (2006). My planting history study (Chapter 2) tells the story about changes of garden planting during last 80 years and discusses the social factors affecting on seedling regeneration. My Red Zone seedling regeneration study (Chapter 4) documented what species are already regenerating well under those conditions. Chapter 6 is the research on the restored sites in Christchurch urban area. I analysed the vegetation composition and the factors that have influenced seedling regeneration. Based on my results, I have made native patch options with five scenarios for the Red Zone.

In this patch matrix, there are four native patch types and one exotic-native mixed forest type (Figure 7.10). As the result shows in chapter 7, four patches and the Avon River are candidates for native patches with high importance value for habitat connectivity (Figure 6.14). The Avon River is planned to be a riparian reserve which can link important native bush in the city centre to existing habitat patches near the coast. The patches (blue area) under Travis wetland can be mixed forest-wetland habitat to expand and complement Travis Wetland in the future. Horseshoe Lake and Dallington area have a high tree density (Figure 7.9) which includes a large group of both native and exotic species can be the exotic-native mixed forest patch (with phased removal of woody weed species). The part in Bexley (the yellow area) is close to the coastal line which can be the saltmarsh reserve with species can grow in the salty soil. This is particularly important when considering the predicted sea level rise over the next decades. We have found some tōtara (*Podocarpus totara*) seedlings and kahikatea (*Dacrycarpus dacrydioides*) seeds already coming from established old garden trees; thus we should keep these areas as core native patches (the red area).

FIGURE 7.9: Map showing the exotic tree density by 100 m × 100 m polygons in Red Zone.



Scenario one: Avon River Riparian

As shown in the reserve patch pattern of (Meurk & Hall, 2006), any patch 100 m wide or less has no forest core habitat, regardless of how long the patch is. As shown in Figure 7.12, after putting a linear zone with 100 meter width along the Avon river, the whole the Avon River will then be covered by potential sites. Most of the linear zone include the vegetation along the river and two trails on both sides. In some parts, only one side

FIGURE 7.10: Five patch types developed and the distribution in Red Zone.

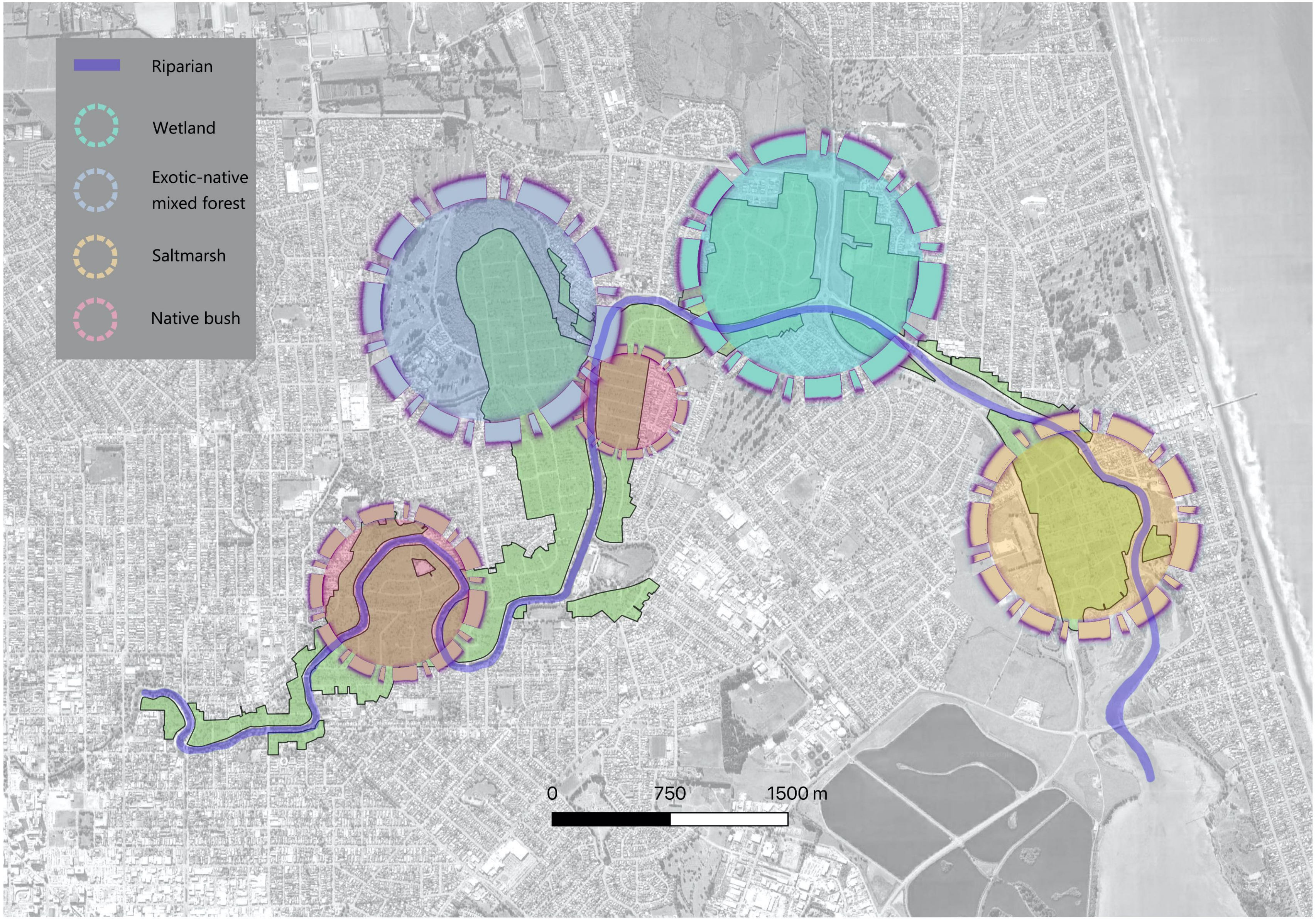
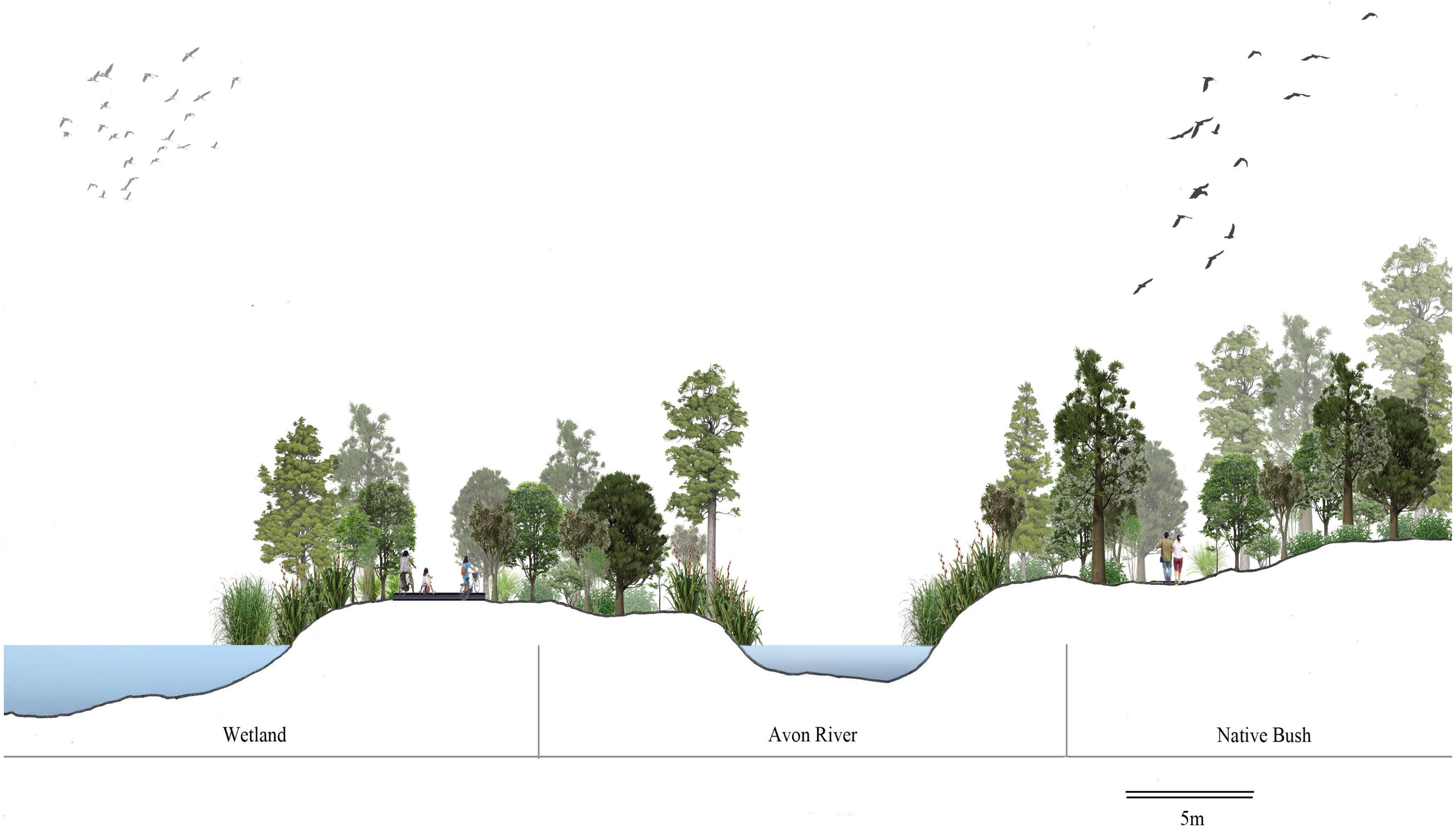


FIGURE 7.11: The section of the landscape for future Red Zone area which includes Wetland, Avon River and Native bush.



will be available to do forest restoration. I use Avonside as an example (Figure 7.13). One side of the reserve can only be 50 meters wide.

Figure 7.12: Map showing Avon river riparian reserve with 100 m width buffer and the Red Zone.

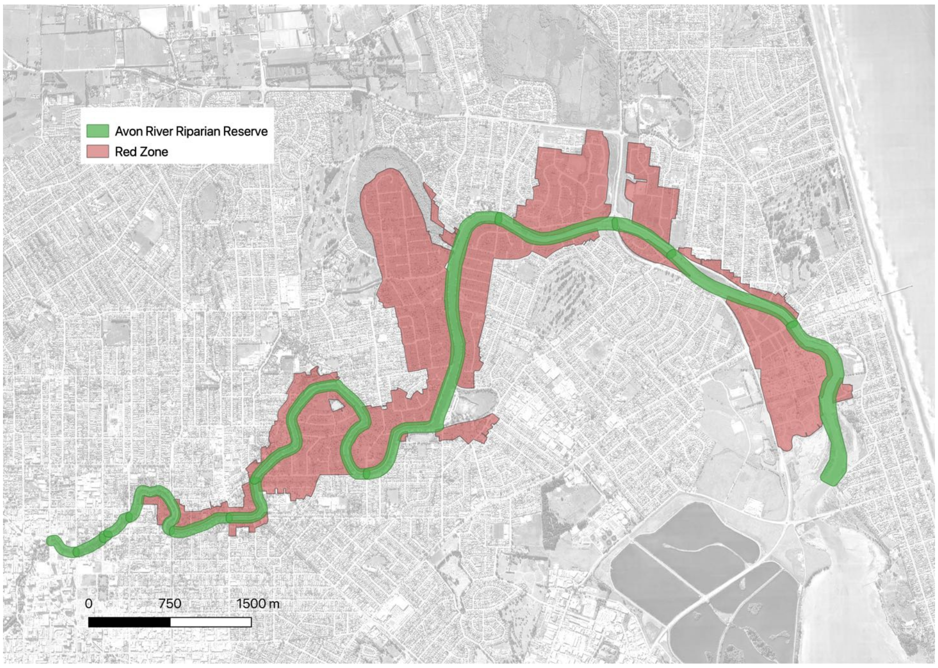
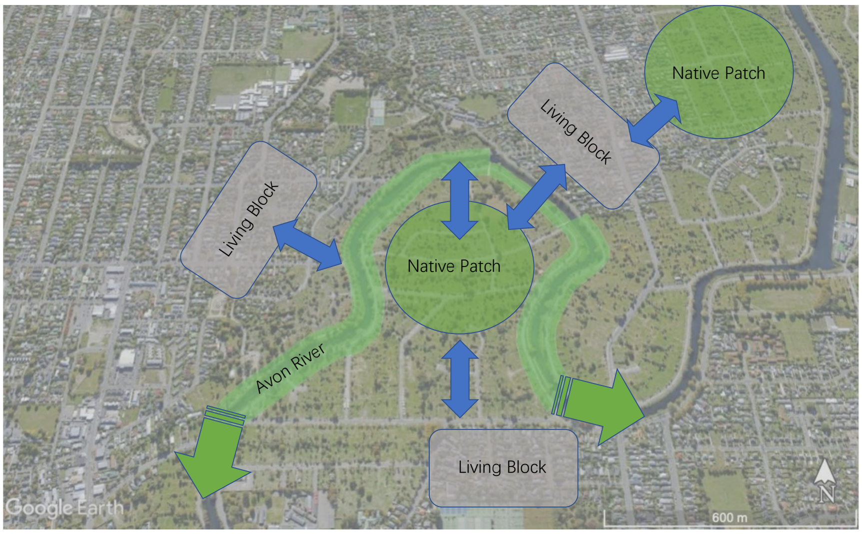


Figure 7.13: The network of Avon river reserve and other patches. Avon river can link patches in the city to the patches along the coastal area. Also species can exchange between patches and residential blocks through Avon river.



The Avon River riparian zone in the Red Zone includes creeks, riverbanks, wetlands, aquifers and floodplains. Figure 7.13 shows the connection between the Avon River riparian reserve and other patches. Basically, as a corridor, the Avon river will link native bush to the current habitat patches along the coast. Also, it links to native patches next to the river as well as the living blocks (suburban housing). Species will be able to disperse from native patches to living blocks or from living block to the patches. This will make a full network for the Avon River as an important part of the Red Zone restoration project.

Scenario two: Exotic-native forest patch

In the Horseshoe Lake-Dallington area there are a large group of adult exotic trees remaining, with a higher tree density than other areas. Based on this situation, this place can be an exotic-native forest patch in the restoration plan which will act as a transition zone. The vegetation of this patch can develop from exotic-native forest to pure native forest through regeneration with management. The main management will be eliminating invasive exotics and weeding out exotic regeneration. The existing exotic trees will form an important component of the habitat, at least over the first decades while native trees mature.

FIGURE 7.14: Views of the vegetation in Horseshoe Lake-Dallington area. This area has a high density of trees and it could be a exotic-native forest patch in the future.

In this case, we use Ernle Clark Reserve as a reference site. Ernle Clark Reserve has a long canopy age (100 years old) which means there are a few old exotic trees such as Oak tree(*Quercus robur*) but a young restoration age (more than 10 years old) (Table 5.1). Restoration work in Ernle Clark Reserve just started 10 years ago, but there has been strategic weeding, encouragement of nature regeneration, and enrichment planting by the local community for at least 10 years. Now it is covered by a nice exotic-native vegetation mix and ecological management is still ongoing in the reserve.

Scenario Three: Native patch

Because the rare tōtara (*Podocarpus totara*) seedlings were found in the Avondale area (Figure 3.17), this place is recommended as a native bush zone to protect and build on the regeneration already happening. Indeed, tōtara trees are commonly distributed in the Christchurch urban area but it is rare to record seedlings. One reason could be the reproductive cycle (Wilson & Owens, 1999). There is an extreme case we recorded in Ilam Garden. Here there were several tōtara trees living close together and a female tree bearing a lot of fruit (Figure 7.15). However, there were no seedlings nearby. I assume that is because there was too much human disturbance in this area, such as the path way and too much leaf litter on the surface of ground (Figure 7.16). Also we found one kahikatea (*Dacrycarpus dacrydioides*) tree with fruits in Avonside (Figure 7.17) but similarly, no seedlings as well. By restoring the right forest microclimate around these established trees, it is likely that regenerating would occur.

FIGURE 7.15: Female tōtara (*Podocarpus totara*) tree with fresh fruits found in Ilam Garden

Here I use the Botanic Gardens (the Cockayne New Zealand Garden area) as a reference site for this native patch. In the survey data of the restored sites in the urban area, the Botanic Garden had the best native vegetation and most of the native species were regenerating naturally in the patch. This site has a long history of restoration work since it was established. After a long period of regeneration with basic management, Botanic Garden has become very close to the original native bush of Riccarton Bush.

Figure 7.16: Photo shows the landscape of the tōtara (*Podocarpus totara*) grove in Ilam Garden. A walking path goes through it and the ground is covered with compost.

Figure 7.17: Kahikatea (*Dacrycarpus dacrydioides*) tree with fruits found in Avonside.

Scenario Four: Forest-Wetland

Parts of the Red Zone near Travis Wetland are suitable to be forest and wetland, which can enlarge the Travis wetland site. Larger patches are capable of sustaining more biodiversity and this is an excellent opportunity to secure the biodiversity of Travis Wetland. This is particularly important when sea level rise is considered, as the current Travis Wetland site could become substantially modified is salt water is allowed to extend this far inland. From previous findings, this place currently has a low density of trees as it was a young suburb before being irrevocably damaged by the earthquakes (Figure 7.18).

FIGURE 7.18: View of the place near Travis Wetland which could be a wetland in the restoration plan. There is a low tree density and most of the area is covered by grassland.

The reference sites for this area are Travis Wetland and Riccarton Bush. Travis Wetland is the largest freshwater wetland in the Canterbury Plains and one of only two freshwater wetlands in urban areas in New Zealand (Morgan, 2002). Now Travis Wetland has become one of the important patches in the Christchurch urban area. 76% of all native wetland birds in lowland Canterbury are supported by Travis wetland. Moreover, it also contains rare and vulnerable plant species (Crossland, 1996; Morgan, 2002). Riccarton Bush was a kahikatea swamp forest by the Avon River but its water table was dropped considerably with the establishment of the surrounding suburban housing

(Molloy, 1995). It still provides an unequalled reference for what a Christchurch swamp forest can contain.

Figure 7.19: Views of Travis Wetland.

Scenario Five: Saltmarsh patch

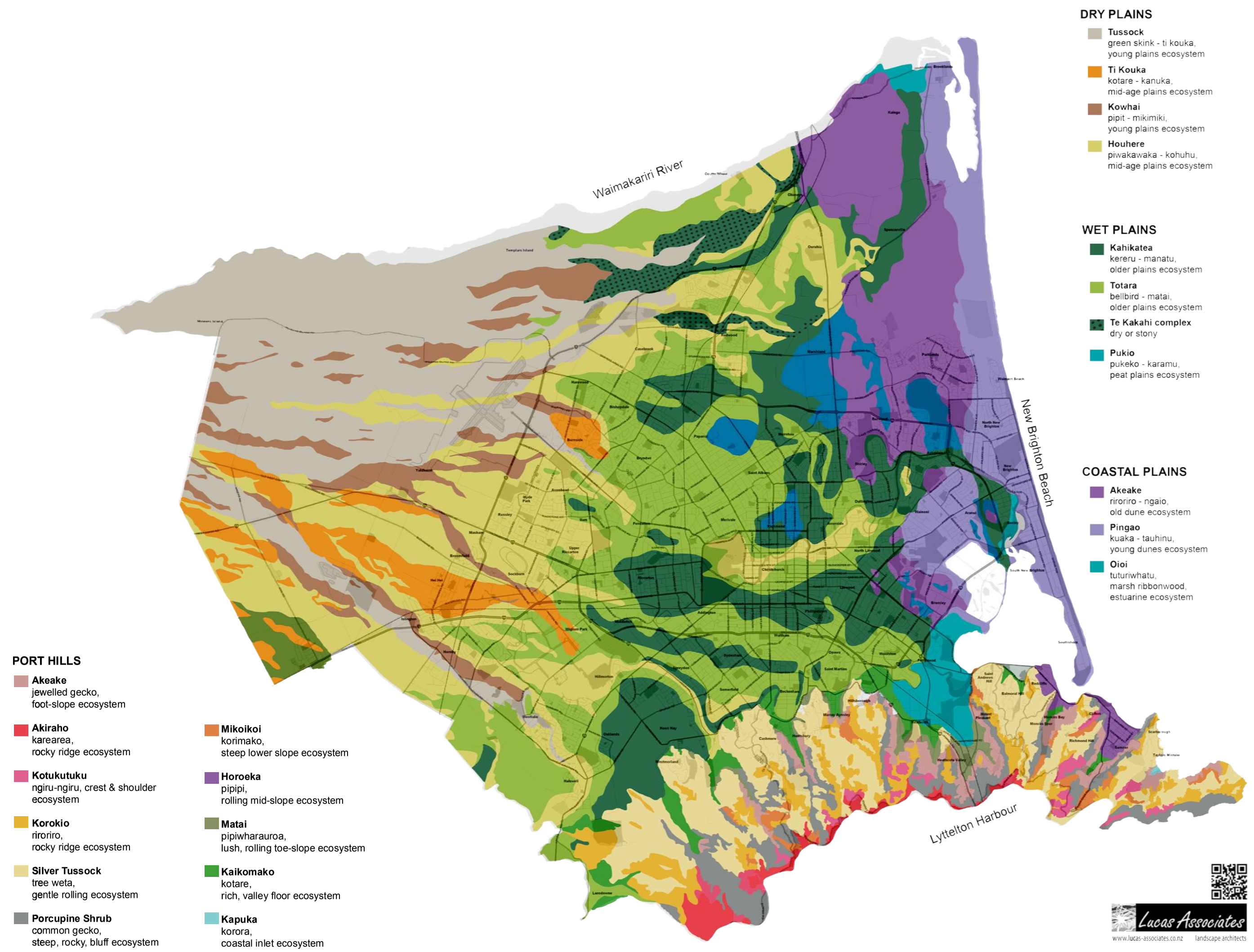
Due to the costal location of the site in this case, it is ideal to be a saltmarsh for native species, which are an important part of the estuary ecosystem. This area was historically saltmarsh before being drained and raised for suburban housing. With that housing now gone, and sea level rise projected for the coming decades, this is the best place in the city to plan for saltmarsh migration. The removal/repositioning of the current stop-bank alone would likely put in motion the succession of this site back to indigenous dominated saltmarsh.

7.5 Planting and management options

The first step to do restoration planting is to choose the ecologically appropriate species. The Otautahi Christchurch Indigenous Ecosystems guide from Lucas Associates (`www.lucas-associates.co.nz`) updated the 1995-1997 booklets to guide the city on what plant communities are appropriate where (Figure 7.20). The guide includes several ecosystem types of Christchurch urban area as well as the native plant lists for different ecosystems. This guide provides the planting guidelines for the native patch restoration across the city.

However, species like the cabbage tree(*Cordyline australis*) are already commonly distributed in the Red Zone area and my dispersal distance result shows the seedling of cabbage tree can establish everywhere in the Red Zone (Figure 4.3). Species like this do not need to be planted any more. Furthermore, 50 meters has been used as a common

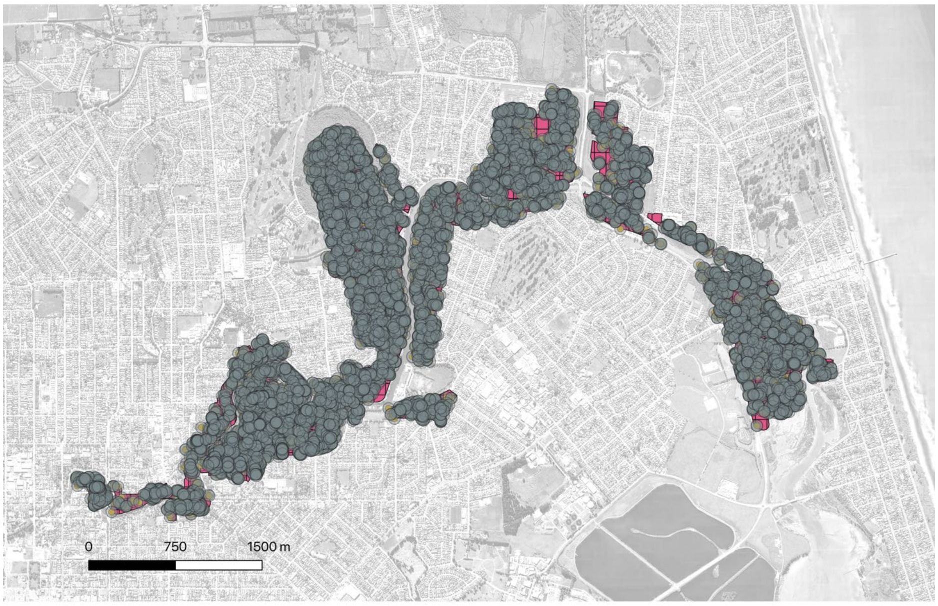
Figure 7.20: Otautahi Christchurch Indigenous Ecosystems map from Lucas Associates (`http://lucas-associates.co.nz/christchurch-banks-peninsula/christchurch-ecosystems/`).



native seedling establishment distance to make a buffer zone map to show the possible area for seedling establishment. Figure 7.21 shows a 50 m buffer around all of the existing native trees retained in the Red Zone; almost the whole Red Zone area is covered by naturally establishing native species. In other words, whole Red Zone could be covered by native seedlings naturally. Indeed, this is exactly what was happening in the years immediately following the earthquakes (Glenn Stewart, Colin Meurk, Denise Ford, and Jon Sullivan, unpublished data), before the area was "tidied" by laying down top soil and lawn seed then regularly mowing the lawn and herbiciding the edges sprayed to keep down weeds and other wild plants. My seedling study shows that the potential for this natural regeneration remains in the Red Zone, just waiting for the land management to change.

Two factors here affect the seedling regeneration process: seed resources and dispersal ability. Although most of the Red Zone can be covered by native species, those two factors still need to be considered when choosing native plants. Species with fewer seeds and weak dispersal ability such as *Podocarpus totara* and *Dacrycarpus dacrydioides*, those species currently not represented by adults in the Red Zone, and those species not represented by adequate local genotypes, may need to be planted more to encourage their establishment.

FIGURE 7.21: Native seedling establish buffer zone with 50 meters establish distance in Red Zone. It shows even from the existing scattered trees, virtually the entire Red Zone can over time be seeded with nature trees.



Genetic diversity is an important part of the biodiversity mix to consider when planning restoration. There are three approaches to enhance the genetic diversity (E. Thomas et al., 2014). The first one is to get the forest reproductive material from wild populations or remaining patches such as Riccarton Bush. The second one is using local native species and never using cultivars. The last one is increasing the connections between patches to help species exchange and gene flow (E. Thomas et al., 2014).

Basic management is necessary to create the appropriate conditions in restoration to foster native species re-establishment. At the early stage of restoration, in order to increase the canopy for seedling establishment, faster growing, bird dispersed species are ideal to be the pioneer trees. Tree planting work will become less needed as the restoration age getting older. Patches such as the Botanic Garden don't need to do planting anymore. However, exotic seedling removal will be needed all the time; frequent low-intensity weeding is still required even in the mature core forest of Riccarton Bush. Shade tolerant, fast growing and bird dispersed exotic species will be the priority to be removed. For the exotic-native mixed forest, as (Stewart et al., 2004) suggested, if we can control the most aggressive, long-lived and shade-tolerant exotic species then it may help the urban forest to transfer to a new kind of indigenous-dominant forest.

7.6 Discussion

Different management options for exotic-native forest patch and pure native forest patch mean that the goals for both can be the same; they can become pure native bush like Riccarton Bush or other Podocarpus forest types no longer occurring on the lower Canterbury Plain over time. Exotic-native transitional forest patches supply more opportunities for people exploring nature. Facilities and open space inside assist with the landscape transition. By using appropriately timed management, such as planting selected native trees to replace the dead exotic trees, these exotic-native forest patches can slowly transition to native-dominated forest. Planning for a multi-use, mixed-species transition to native dominated forest, making use of all of the existing adult trees, potentially allows for sites that can tolerate more initial disturbance and human use. Thus could provide a socially acceptable transition to more intact urban native bush, instead of planting and management an all-native forest from the start that would be sensitive to disturbance and require careful management. Core areas of all-native planting would likely need to be fenced with management such as planting native trees, pest control, and removing all exotic weeds.

As important source patches, Riccarton Bush and the Cockayne Garden in the Botanic Gardens are isolated patches (Chapter 7), which is a threat to the long-term sustainability of their biodiversity. Several studies have confirmed that when the remaining forest patches near the restoration sites get isolated and fragmented, tree species in the patches may be inbred, have reduced fitness or have other negative results of a small population size (Aguilar, Quesada, Ashworth, Herrerias-Diego, & Lobo, 2008; Breed, Gardner, Ottewell, Navarro, & Lowe, 2012; Eckert et al., 2010; Vranckx, Jacquemyn, Muys, & Honnay, 2012; Szulkin, Bierne, & David, 2010; E. Thomas et al., 2014). Hence, the eastern Red Zone is especially important to become a corridor from those source patches to other restoration sites and habitat patches.

Climate change is a factor that needs to be considered for planning modern ecological restoration projects. It will have a significant effect on many restoration sites (Hobbs, Higgs, & Harris, 2009). A previous study suggests that the tree species dispersal mode influences tree persistence under climate change (Bhagwat, Nogué, & Willis, 2012). Compared with the species with few large seeds and long regeneration time, those which have high fecundity, small seeds with long dispersal distance and short generation times have been able to adapt and migrate more quickly (Aitken, Yeaman, Holliday, Wang, & Curtis-McLane, 2008). Thus, the dispersal mode needs to be considered when planning connectivity networks and strategies (E. Thomas et al., 2014).

It is also important to control disturbance which can reduce the effective size and native biodiversity of the habitats (Baschak & Brown, 1995). Management actions such as trampling of shrub layers should be curtailed (Baschak & Brown, 1995) by the careful design of paths. Exotic seed resources need to be paid attention to in the neighbouring blocks as our previous study shows exotic plants still are the popular preference in Christchurch gardens, and many of these are woody weeds (Chapter 2). A public education program could teach the residents living near Red Zone (and other important habitat patches like Riccarton Bush and Travis Wetland) about the principles of ecological restoration and this could enhance public care and reduce the rate of exotic species invasion (Thorne & Huang, 1991; Baschak & Brown, 1995).

In summary, the devastating earthquakes have given Christchurch City what is arguably a once-in-a-lifetime opportunity to make substantial changes to a large area of the built city. The city's indigenous biodiversity would likely benefit greatly by the transitional restoration of this area into vibrant indigenous forest.

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Appendix A

Species lists from Chang and Quans surveys

Table A.1: Tree Species list of Quan's Survey Data

Species	Count	Simple_Biostatus
Pittosporum tenuifolium	258	Local native
Pittosporum eugenioides	104	Local native
Pseudopanax arboreus	86	Local native
Hoheria angustifolia	81	Local native
Cordyline australis	73	Local native
Aristotelia serrata	70	Local native
Griselinia littoralis	65	Local native
Kunzea robusta	65	Local native
Plagianthus regius	58	Local native
Podocarpus totara	54	Local native
Sophora microphylla	45	Local native
Melicytus ramiflorus	39	Local native
Coprosma robusta	36	Local native
Acer pseudoplatanus	34	Naturalised
Fuscospora solandri	32	Local native
Fuscospora cliffortioides	27	Non-local native
Dacrycarpus dacrydioides	25	Local native
Coprosma grandifolia	24	Non-local native
Pseudopanax hybrid	19	Non-local native
Pseudopanax crassifolius	18	Local native

Table A.1 continued

Species	Count	Simple_Biostatus
Corynocarpus laevigatus	16	Non-local native
Quercus robur	16	Naturalised
Coprosma propinqua	10	Local native
Hoheria populnea	10	Non-local native
Dicksonia squarrosa	7	Local native
Agathis australis	7	Non-local native
Aesculus hippocastanum	6	Naturalised
Fuscospora fusca	5	Local native
Hedycarya arborea	5	Local native
Fuchsia excorticata	5	Non-local native
Alnus glutinosa	5	Naturalised
Myoporum laetum	4	Local native
Pittosporum ralphii	4	Non-local native
Juglans spp	4	Exotic
Rhododendron spp	4	Exotic
Alectryon excelsus	3	Local native
Corokia spp	3	Local native
Dodonaea viscosa	3	Local native
Elaeocarpus dentatus	3	Local native
Pittosporum spp	3	Local native
Entelea arborescens	3	Non-local native
Crataegus monogyna	3	Naturalised
Fraxinus excelsior	3	Naturalised
Cyathea dealbata	2	Local native
Brachyglottis repanda	2	Non-local native
Olearia spp	2	Non-local native
Arbutus unedo	2	Naturalised
Taxus baccata	2	Naturalised
Coprosma crassifolia	1	Local native
Dacrydium cupressinum	1	Local native
Lophomyrtus obcordata	1	Local native
Prumnopitys taxifolia	1	Local native
Beilschmiedia tawa	1	Non-local native
Laurelia spp	1	Non-local native
Pseudopanax lessonii	1	Non-local native
Berberis darwinii	1	Naturalised
Prunus spp	1	Naturalised
Sambucus nigra	1	Naturalised
Camellia spp	1	Exotic
Cotoneaster spp	1	Exotic
Prunus ceraifera	1	Exotic
Ulmus spp	1	Exotic

Table A.2: Seedling Species list of Quan's Survey Data

Species	Count	Simple_Biostatus
Acer pseudoplatanus	1355	Naturalised
Corynocarpus laevigatus	994	Non-local native
Sophora microphylla	906	Local native
Quercus robur	668	Naturalised
Hedera helix	439	Naturalised
Pittosporum tenuifolium	416	Local native
Cordyline australis	317	Local native
Coprosma grandifolia	311	Non-local native
Hedycarya arborea	287	Local native
Coprosma robusta	281	Local native
Plagianthus regius	263	Local native
Muehlenbeckia australis	255	Local native
Prunus spp	196	Naturalised
Pseudopanax arboreus	186	Local native
Coprosma crassifolia	140	Local native
Brachyglottis repanda	137	Non-local native
Hoheria populnea	137	Non-local native
Nestegis cunninghamii	133	Non-local native
Sambucus nigra	121	Naturalised
Hoheria angustifolia	114	Local native
Pittosporum eugenioides	106	Local native
Piper excelsum	100	Local native
Melicytus ramiflorus	97	Local native
Pseudopanax crassifolius	83	Local native
Coprosma propinqua	79	Local native
Coprosma virescens	79	Local native
Euonymus europaeus	76	Naturalised
Podocarpus totara	56	Local native
Dacrycarpus dacrydioides	54	Local native
Dodonaea viscosa	51	Local native
Pseudopanax crassifolius X P. lessonii	49	Local native
Coprosma propinqua x C. robusta	44	Local native
Coprosma rhamnoides	35	Local native
Alectryon excelsus	33	Local native
Dicksonia squarrosa	29	Local native
Vinca major	26	Naturalised
Rubus fruticosus	20	Naturalised
Streblus heterophyllus	17	Local native
Ulmus spp	16	Naturalised
Ilex aquifolium	15	Naturalised
Corokia spp	14	Local native
Griselinia littoralis	14	Local native
Solanum laciniatum	14	Local native
Pennantia corymbosa	13	Local native
Pseudopanax hybrid	12	Non-local native

Table A.2 continued

Species	Count	Simple_Biostatus
Laurus nobilis	11	Naturalised
Coprosma repens	11	Non-local native
Coprosma rotundifolia	11	Local native
Taxus baccata	10	Naturalised
Camellia spp	9	Naturalised
Asplenium gracilimum	9	Local native
Pittosporum crassifolium	8	Non-local native
Melicope ternata	7	Non-local native
Coprosma areolata	7	Local native
Coprosma linariifolia	7	Local native
Coprosma spp	7	Local native
Prunus laurocerasus	6	Naturalised
Myrsine salicina	6	Non-local native
Hydrangea spp	5	Naturalised
Rhododendron spp	5	Naturalised
Olearia spp	5	Non-local native
Aristotelia serrata	5	Local native
Lophomyrtus obcordata	5	Local native
Viburnum spp	4	Naturalised
Mahonia bealei	4	Naturalised
Podocarpus spp	4	Non-local native
Pittosporum ralphii	4	Non-local native
Pittosporum spp	4	Local native
Myrsine australis	4	Local native
Alnus spp	3	Naturalised
Prunus ceraifera cv. Pissardii	3	Naturalised
Pseudopanax laetus	3	Non-local native
Phormium spp	3	Local native
Parsonsia heterophylla	3	Local native
Prumnopitys taxifolia	3	Local native
Pseudopanax colensoi	3	Local native
Veronica spp	3	Local native
Cotoneaster	2	Naturalised
Fatsia japonica	2	Naturalised
Berberis spp	2	Naturalised
Crataegus spp	2	Naturalised
Maytenus boaria	2	Naturalised
Solanum pseudocapsicum	2	Naturalised
Melicytus spp	2	Non-local native
Pseudopanax spp	2	Local native

Table A.2 continued

Species	Count	Simple_Biostatus
Alnus glutinosa	1	Naturalised
Cytisus scoparius	1	Naturalised
Elaeagnus reflexa	1	Naturalised
Celastraceae spp	1	Naturalised
Cotoneaster spp	1	Naturalised
Fraxinus spp	1	Naturalised
Sarcococca ruscifolia	1	Naturalised
Sorbus aucuparia	1	Naturalised
Agathis australis	1	Non-local native
Beilschmiedia tawa	1	Non-local native
Melicytus obovatus	1	Non-local native
Weinmannia racemosa	1	Non-local native
Dacrydium cupressinum	1	Local native
Kunzea robusta	1	Local native
Myoporum laetum	1	Local native
Parsonsia heterophylla	1	Local native
Pseudopanax colorata	1	Local native
Veronica salicifolia	1	Local native

Table A.3: Tree Species list of Chang's Survey Data

Species	Count	Simple_Biostatus
Pittosporum eugenioides	110	Local native
Pittosporum tenuifolium	80	Local native
Fuscospora solandri	77	Local native
Plagianthus regius	76	Local native
Pseudopanax arboreus	74	Local native
Podocarpus totara	65	Local native
Cordyline australis	42	Local native
Kunzea robusta	37	Local native
Coprosma robusta	36	Local native
Aristotelia serrata	31	Local native
Dacrycarpus dacrydioides	29	Local native
Griselinia littoralis	26	Local native
Dodonaea viscosa	20	Local native
Hoheria angustifolia	18	Local native
Acer pseudoplatanus	17	Naturalised
Melicytus ramiflorus	14	Local native
Alectryon excelsus	12	Local native
Alnus glutinosa	11	Naturalised
Fuscospora fusca	11	Local native
Pseudopanax crassifolius	11	Local native
Quercus robur	11	Naturalised
Sophora microphylla	11	Local native
Coprosma linariifolia	10	Local native
Ulmus spp	10	Exotic
Cyathea dealbata	9	Local native
Pittosporum crassifolium	8	Non-local native
Dicksonia fibrosa	6	Local native
Corynocarpus laevigatus	5	Non-local native
Hoheria populnea	5	Non-local native
Coprosma crassifolia	4	Local native
Hedycarya arborea	4	Local native
Phyllocladus trichomanoides	4	Local native
Dacrydium cupressinum	3	Local native
Nestegis cunninghamii	3	Non-local native
Olearia paniculata	3	Local native
Prumnopitys ferruginea	3	Local native
Prunus spp	3	Naturalised
Rhododendron spp	3	Exotic
Sambucus nigra	3	Naturalised
Taxus baccata	3	Naturalised
Agathis australis	2	Non-local native
Carpodetus serratus	2	Local native
Cyathea medullaris	2	Non-local native
Fraxinus excelsior	2	Naturalised

Table A.3 continued

Species	Count	Simple_Biostatus
Hoheria sexstylosa	2	Local native
Juglans regia	2	Naturalised
Myoporum laetum	2	Local native
Myrsine australis	2	Local native
Ackama rosifolia	1	Non-local native
Azara microphylla	1	Naturalised
Coprosma areolata	1	Local native
Coprosma australis	1	Non-local native
Coprosma grandifolia	1	Non-local native
Coriaria arborea	1	Local native
Corokia buddleioides	1	Non-local native
Crataegus monogyna	1	Naturalised
Cupressus macrocarpa	1	Naturalised
Elaeocarpus hookerianus	1	Local native
Entelea arborescens	1	Non-local native
Feijoa sellowiana	1	Exotic
Fuscospora menziesii	1	Non-local native
Hebe salicornioides	1	Non-local native
Ilex aquifolium	1	Naturalised
Olearia avicenniifolia	1	Local native
Paratrophis banksii	1	Non-local native
Prumnopitys taxifolia	1	Local native
Pseudopanax adiantifolius	1	Local native
Pseudopanax colensoi	1	Local native
Pseudopanax discolor	1	Local native
Pseudopanax hyrbid	1	Non-local native
Pseudotsuga menziesii	1	Naturalised
Salix fragilis	1	Naturalised
Sophora tetraptera	1	Non-local native
Streblus banksii	1	Non-local native
Viburnum	1	Exotic

Table A.4: Seedling Species list of Chang's Survey Data

Species	Count	Simple_Biostatus
Acer pseudoplatanus	1768	Naturalised
Prunus spp	1283	Naturalised
Pittosporum tenuifolium	1124	Local native
Quercus robur	677	Naturalised
Sophora microphylla	625	Local native
Plagianthus regius	531	Local native
Cordyline australis	468	Local native
Coprosma robusta	394	Local native
Sambucus nigra	312	Naturalised
Coprosma grandifolia	263	Non-local native
Coprosma propinqua	240	Local native
Pittosporum eugenioides	173	Local native
Coprosma linariifolia	168	Local native
Hedycarya arborea	158	Local native
Pseudopanax arboreus	119	Local native
Coprosma hyrbid	117	Non-local native
Coprosma virescens	114	Local native
Podocarpus totara	111	Local native
Dacrycarpus dacrydioides	89	Local native
Alectryon excelsus	86	Local native
Coprosma spp	80	Local native
Aristotelia serrata	75	Local native
Pittosporum crassifolium	74	Non-local native
Dodonaea viscosa	72	Local native
Coprosma crassifolia	71	Local native
Coprosma rotundifolia	68	Local native
Quercus cerris	67	Naturalised
Melicytus ramiflorus	66	Local native
Corynocarpus laevigatus	65	Non-local native
Piper excelsum	60	Local native
Griselinia littoralis	41	Local native
Pseudopanax crassifolius	40	Local native
Coprosma rhamnoides	30	Local native
Fraxinus spp	30	Exotic
Euonymus europaeus	29	Naturalised
Muehlenbeckia australis	25	Local native
Brachyglottis repanda	24	Non-local native
Hoheria angustifolia	24	Local native
Pennantia corymbosa	24	Local native
Dicksoni spp	14	Exotic
Coprosma areolata	13	Local native
Pseudopanax hyrbid	12	Non-local native
Alnus glutinosa	11	Naturalised
Myoporum laetum	11	Local native

Table A.4 continued

Species	Count	Simple_Biostatus
Fuscospora fusca	10	Local native
Hoheria sexstylosa	10	Local native
Parsonsia heterophylla	10	Local native
Hedera helix	9	Naturalised
Melicope ternata	8	Non-local native
Myrsine australis	7	Local native
Pseudopanax lessonii	7	Non-local native
Pseudopanax laetus	6	Non-local native
Veronica spp	6	Local native
Coprosma repens	5	Non-local native
Rhododendron spp	5	Exotic
Kunzea robusta	4	Local native
Pittosporum divaricatum	4	Non-local native
Prumnopitys ferruginea	4	Local native
Crataegus monogyna	3	Naturalised
Dacrydium cupressinum	3	Local native
Fraxinus excelsior	3	Naturalised
Fuscospora solandri	3	Local native
Taxus baccata	3	Naturalised
Coprosma australis	2	Non-local native
Cyathea medullaris	2	Non-local native
Daphne laureola	2	Exotic
Hoheria populnea	2	Non-local native
Ilex aquifolium	2	Naturalised
Lophomyrtus obcordata	2	Local native
Ripogonum scandens	2	Non-local native
Clematis spp	1	Naturalised
Coprosma tenuicaulis	1	Local native
Elaeocarpus hookerianus	1	Local native
Fuchsia magellanica	1	Exotic
Hebe salicornioides	1	Non-local native
Metrosideros umbellata	1	Non-local native
Myrsine salicina	1	Non-local native
Olearia paniculata	1	Local native
Prumnopitys taxifolia	1	Local native
Pseudopanax adiantifolius	1	Local native
Pseudopanax colensoi	1	Local native
Pseudowintera colorata	1	Local native
Salix fragilis	1	Naturalised