Biodiversity outcomes of an urban revegetation programme in Wellington, New Zealand: The role of patch size, isolation, age and the urban matrix

By

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ABSTRACT

Urban revegetation programmes are an important contribution to the biodiversity of our environment. Wellington City has, for nearly two decades, promoted revegetation of the city, with a focus in latter years on eco-sourced native species. This is a substantial urban greening project planting around 100,000 native plants per annum. This thesis analyses the native revegetation programme and posits recommendations for continuance and enhancement for consideration.

Data were collected from a sample set of twenty revegetated sites and four reference sites in Wellington City. Two transects per site were set up with invertebrate pitfall traps, lizard pitfall traps, lizard tree covers, bird count stations and vegetation surveys. The data were collected over a twelve-month period. Ordination was used to examine the community composition of revegetation sites in relation to each other and the four reference sites. Multiple regression was used to examine the influence of patch age, patch size, isolation and residential land cover upon a range of biodiversity variables.

The key findings of this study are that revegetated sites lacked the level of ground cover by native seedlings that were characteristic of the mature reference sites. Also found was that revegetation sites in Wellington City require around ten years of growth before natural regeneration of native seedlings began to become apparent.

Large native birds were more likely to be found in close proximity to mature primary bush, confirming that mature primary bush is an essential element of the landscape. Weta were found in increasing numbers as distance from mature primary bush increased. Northern grass skinks were the only species of lizard found, in association with a higher proportion of residential area in the matrix.

The key management recommendations of this study are: the development of a collaborative connectivity strategy; further enrichment planting designed to maximize structural diversity over time; the investment in well planned robust monitoring programmes.

This research contributes to the understanding of biodiversity outcomes of an urban native revegetation programme, providing baseline data for future monitoring purposes.

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I'd like to thank my fieldwork and lab colleagues, Erin, Asher, and Olivia; for your generosity of time and interest in my study.

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Chapter 1

INTRODUCTION

Human-induced disturbances have radically altered the biota of New Zealand (Atkinson, 1989). While dense forest originally covered approximately 80% of the land, over the past 800 years vast areas of native forest have been cleared for agricultural use and urban settlement, resulting in large-scale fragmentation and habitat loss (Department of Conservation [DoC], 1996; Wilmhurst, 2012). Native forest cover today is only 25% of the land. In addition to this, the introduction of pests, particularly mammalian pests and exotic plants, has resulted in substantial losses of New Zealand species, with many more under threat of extinction (Saunders & Norton, 2001). de Lange, Heenan, Norton, Rolfe, & Sawyer (2010) report that of the 2370 native vascular plants in New Zealand, "6 species are known to have gone extinct and 184 are regarded as threatened with extinction" (2010, p.33). Added to these losses are 64 animal species that are also extinct (Wilmhurst, 2012), the majority of which are birds (DoC, 2015a).

> *"With the exception of the hills facing the Strait and the high land around Evans Bay, the hills around Port Nicholson are covered with the richest verdure to their summits"* (Henry Petre cited in Boffa Miskel, 1998, p.3).

Petre's first view of a heavily forested landscape reflects the scene that greeted most European settlers arriving in Wellington (Port Nicholson) in 1840 (Boffa Miskel, 1998; Museum of Wellington City & Sea, 2001). The lowland forests of Wellington were broadleaf/podocarp dominated by canopy species such as rimu (*Dacrydium cupressinum*), matai (*Prumnopitys taxifolia*), totara (*Podocarpus totara*), kahikatea (*Dacrycarpus dacrydioides*) and northern rata (*Metrosideros robusta*). Below the canopy, layers of trees, shrubs, climbers, epiphytes, ferns and grasses completed the structural diversity (Gabites, 1993).

A rich birdlife was supported by this verdant vegetation and was commented upon in many settlers' accounts (Boffa Miskel, 1998). However, the wide-ranging clearance of native forest cover (DoC, 1996; Park, 1999) and the introduction of mammalian predators resulted in the local extinction of many bird species (Miskelly, Empson, & Wright, 2005).

The native forest cover of Wellington was reduced from 90% to less than 5% in just a few decades, between 1840 and 1870 (Marjot, 1992; Wellington City Council [WCC], 2007). The surviving forest patches are important seed sources (Park, 1999) and habitat for local fauna. Secondary forest has gradually developed in gullies and on urban fringes (DoC, 1996; Park 1999) creating the vegetation linkages that are required for flora and fauna populations to survive in fragmented landscapes (Saunders, Hobbs, & Margules, 1991; Marzluff & Ewing, 2001).

Habitat destruction along with the introduction of pest mammals have also impacted on the lizard fauna of Wellington, with various species now extinct and others only surviving on pest-free islands (Wellington Regional Lizard Network [WRLN], n.d.). Nevertheless, the herpetofauna in Wellington remains highly diverse (DoC, 2009; WRLN, n.d.) and action plans and strategies are in place to conserve and restore lizard communities.

Urban ecology is a combination of the science of understanding the "distribution and abundance of organisms in and around cities" (Picket et al., 2011, p.333), and the planning perspective that seeks to reduce environmental impact of urban function and amenity provision (Picket et al., 2011). Any remaining fragments of vegetation in and around cities are critical habitat for native flora and fauna (Ruiz-Jane & Aide, 2006); in fact ecologists "have been surprised by the presence and vitality of organisms in and near cities" (Picket et al., 2011, p.333).

Urban ecological restoration projects aim to conserve and enhance the biodiversity in and around urban areas (Atkinson, 1994) and are generally led by territorial authorities and non-governmental organisations. There has been a significant rise in community involvement in urban restoration in recent times that provides additional benefits to the health and well-being of the community (Beatley, 2010; Buchan, 2007).

Wellington City Council [WCC] initiated several revegetation projects during the 1990's aimed at restoring, conserving and promoting biodiversity. The results of community engagement in native revegetation on public land are commendable in Wellington, with a total of 110 community groups planting around 35,000 plants per annum (A. Benbrook, personal communication, April 1, 2015). Community groups have a Memorandum of Understanding with the Council and liaise with the Park Rangers. This exists alongside other community greening initiatives such as the Road Reserve Scheme in which the Council provides a total of 10,000 native plants annually to residents, via an application process, for enhancement of public road reserve land. A further significant revegetation initiative administered by WCC is the Council staff-planting project of 45,000 plants per annum. As the Manager of the WCC nursery that propagates and distributes all of these plants, I am keenly interested in the biodiversity outcomes a revegetation strategy such as this offers the urban environment of Wellington. It is the staff-planting initiative that is the focus of this research, referred to hereafter as 'the WCC native revegetation programme'.

The following section introduces the key contexts of this research in that there is a discussion on the four main areas of enquiry and analysis: *Patch Size, Isolation, Site Age,* and *Landscape Matrix.*

PATCH SIZE

A patch, in ecological terms, is a continuous area of habitat that is functioning somewhat independently and is separated from other patches by incompatible habitat (Hanksi & Gilpin, 1997). The size of a patch influences ecosystem dynamics within it. A larger patch has a greater internal area and is, therefore, less exposed to 'edge effects'. The patch edge, "or zone of transition" (Harris, 1988, p.1), experiences different levels of solar radiation, water and wind to that of the core. The microclimate of the patch is influenced by the amount of edge versus the internal area (Saunders, Hobbs, & Margules, 1991). The edge is also where non-resident flora and fauna may enter the patch, altering the species interactions of the patch (Janzen, 1983). There are varying opinions on the positive and negative consequences of size and edge effects of a patch (Murcia, 1995).

For this research project, the term patch is also used to define areas of habitat that are not self-sufficient but are intended to provide habitat for small biological populations. By this I mean, the revegetated sites in the study sample. These small patches are subject to various levels of permeability within the landscape matrix as a result of the urban environment, presenting different dispersal challenges.

ISOLATION

Distance between patches is a measure of isolation that can affect species colonization and community survival (Saunders, Hobbs, & Margules, 1991). An organism's mode of dispersal and the distance to travel will determine whether or not it is likely to colonize another patch. Population survival is influenced by distance when the patch size is such that the organisms are required to traverse the landscape to forage for food. Connectivity between patches is achieved when corridors or stepping-stones are spatially interspersed at intervals sufficient for species to migrate to, or travel between (Hanski & Gilpin, 1997; Yu, Xun, Shi, Shao, & Liu, 2012; Zipperer, Foresman, Walker, & Daniel, 2012). Stepping-stones are effectively "a series of small patches connecting otherwise isolated patches" (Baum, Haynes, Dillemuth, & Cronin, 2004, p. 2671), and could be as simple as small groups of trees that provide perching and foraging prospects for birds in the landscape matrix (Fisher & Lindmeyer, 2002).

SITE AGE

The age of a revegetated site will influence a variety of factors for consideration and comparison. These include the fact that there is growth of plants with age; structural complexity increases with age, and habitats for animals and invertebrates develop to provide increased foraging opportunities. With age, and as the vegetation grows, the microclimate within the area alters. Site age is, therefore, an element that contributes to the development of community composition (Reay & Norton, 1999).

However, age alone is not the sole element of the ecological development of a site. Site age is the sum of a number of site influences that contribute to habitat development. The rate of development is dependent on several variables, meaning that change does not necessarily correspond to site age in a strict sense of time progression. Factors affecting the rate of development in a revegetation planting include site aspect, soil conditions, climate, plant selection, plant quality, site preparation, follow-up maintenance, and pest control. The term 'site age' in this study is defined as the time since first revegetation planting on the site.

LANDSCAPE MATRIX

The matrix is the term used for the landscape, the background landscape that patches are surrounded by and often considered unsuitable for target species The matrix may be agricultural pasturelands, surrounding patches of remnant forest, or wheat fields surrounding wetlands. For the purposes of this research the matrix is the urban environment surrounding the patches of native habitat planted by WCC. The urban matrix is therefore a complex mosaic of buildings, roads, parks and reserves, and urban 'wildlands'.

SIGNIFICANCE

This research is significant for the development of knowledge and usefulness of the Council's native revegetation programme in Wellington City. Wyant, Meganck & Ham (1995) consider that a "planning and decision-making framework for ecological restoration... [contributes to a] broad and objective perspective of ecological and socioeconomic knowledge" (p.789). They further consider that this framework is important as a foundation for the "scientific approach to... restoration ecology" (p.789). With this in mind the twenty revegetated sites and four reference sites (48 transects in total) that were sampled for this research on the abundance of native seedling recruitment, birds, lizards, and invertebrates constitute a substantial foundation for future research. Indeed, this research could be the beginning of a continued monitoring programme of the biological trajectory within these revegetated urban sites.

THESIS STRUCTURE

The following chapters of this thesis have been written in distinctively different styles. Chapter Two has been constructed as a self-contained journal article focusing on the method of research, the fieldwork, the analysis and findings, written in a style for both a New Zealand and international readership. There is inevitably some repetition between the thesis introduction (Chapter One) and the more concise introduction at the start of Chapter Two. Chapter Three has been written as a set of Management Recommendations for the continued advancement of the WCC native revegetation programme. Chapter Three develops further the discussion of Chapter Two with greater reference to published and 'grey' literature and personal observations from the field, including my experience as a WCC employee participating in the revegetation programme over the past decade. These two chapters are purposefully designed in this manner so that Chapter Two can be presented for publication without extensive alteration and Chapter Three can be presented independently from this thesis to WCC to utilize as a Management Report. The thesis can be considered as a piece of 'practitioner' research. Chapter Four brings the thesis together with final conclusions.

Chapter 2

Biodiversity outcomes of an urban revegetation programme in Wellington, New Zealand: the role of patch size, isolation, age and the urban matrix.

INTRODUCTION

New Zealand has experienced significant deforestation as well as fragmentation of native forests since the arrival of humans, and in particular since European colonization in the mid 1800's (Park, 1995).

Habitat loss and fragmentation are, for the most part, a result of agricultural intensification and urbanization. Native biodiversity has subsequently become threatened as species dependent on forest habitats struggle to exist within modified environments (Battles, Whittle, Stehle, & Johnson, 2013; Marzluff & Ewing, 2001; Standish, Hobbs, & Miller, 2012; Yu, Xun, Shi, Shoa, & Liu, 2012; Zipperer, Foresman, & Walker, 2012). Remaining natural habitats may not be capable of supporting the biota that was left within it. The result may be a new equilibrium between the remaining communities in small fragments and continued loss of species that cannot survive in the 'patch' (Saunders, Hobbs, & Margules, 1991).

Reduction in patch size also results in a relative increase in edge and exposure to different climatic conditions within the fragment (Hanski & Gilpin, 1997; Saunders et al., 1991). Near-edge species are vulnerable to predators (pest animals), and edge habitats experience increased invasion of weedy non-native plants (i.e. pest plants), as well as increased human activity, pollution and erosion (Murcia, 1995).

The theory of island biogeography (MacArthur & Wilson, 1967) refers to species-area relationships, area-diversity patterns, and stepping-stones and biotic exchange. Although the theory was originally based on observations from off-shore islands, the concept has subsequently been transferred to mainland situations (Saunders & Norton, 2001). For example, geographical boundaries and natural physical barriers can create natural 'mainland islands' of functioning, self-sustaining ecosystems. These mainland islands can be "habitat islands involving isolated forest remnants in essentially modified landscapes dominated by farmland, or habitat complexes featuring core management areas within a larger complex of similar habitat" (Saunders & Norton, 2001, p.112). New Zealand's mainland islands are also created in the form of pest-free sanctuaries to protect and enhance remnant vegetation patches and reintroduce threatened species (Watts, Thornburrow, Cave & Innes, 2014). Remaining habitat patches including newly established mainland islands are, however, likely too small to sustain viable populations of some species in the long term. Rather, such populations may depend on continuous immigration (e.g., metapopulation dynamics) (Hanski & Gilpin, 1997).

Ecological restoration emerged in urban landscapes in response to fragmentation and biodiversity decline in the places where people live (Standish, Hobbs, & Miller, 2012). Conserving and restoring nature at the urban fringes and restoring remnant patches of urban nature are two fundamental areas of urban restoration (Standish et al, 2012). In recent times regional and local authorities in New Zealand have been increasingly proactive in restoring native vegetation to urban areas. Urban community groups dedicated to protecting, enhancing and re-establishing native vegetation have gained momentum year upon year (Sullivan, Meurk, Whaley, & Simock, 2009). The benefits of community involvement in urban restoration projects are widely recognized as holistically valuable (health & well-being) to those directly involved, the wider community, and environmental biodiversity (Beatley, T, 2010; Clarkson, Wehi, & Brabyn, 2007).

Wellington, the capital city of New Zealand, is the third largest urban centre in the country, with a population of just under 200,000 people (Statistics New

Zealand, 2013). Less than 5 percent of the original lowland forest of Wellington remains today (Gabites, 1993). However, this 5 percent of remnant forest (appendix 1) is a valuable seed source for natural plant recruitment, and habitat for native wildlife. Natural regeneration has gradually reclaimed areas of abandoned hillsides and gullies adjacent to surviving remnants, effectively increasing the forest cover across the urban landscape (DoC, 1996; Park, 1999; Gabites, 1993). This natural process has provided habitat for native fauna, and a seed source for further recruitment. Further to that, Wellington City Council [WCC] began native revegetation plantings throughout the city in the mid 1990's. The original aim was to reclaim exotic tree removal sites and hazardous mowing sites, but the focus quickly shifted to a more dedicated and planned approach of conserving and restoring nature. In 2007 WCC published a Biodiversity Action Plan as a means to "coordinate biodiversity activities, identify local priorities and actions, and restore biodiversity" (WCC, 2007. p.1). Objective 4.2 of the plan was a commitment to monitor biodiversity indicators and set up monitoring systems designed to assess the council's restoration programmes (WCC, 2007). Various surveys and audits have been conducted providing valuable data that contributes to planning and management of the city's biodiversity. However, the implementation of on-going monitoring systems to repeat sampling regularly, and build up a more robust set of data to work from, has not yet been fully realized.

In this chapter, I assess the biodiversity outcomes of an urban revegetation programme, to develop a clearer understanding of the successional process in council-led eco-restoration projects. The questions I addressed are:

1. How does the age of the revegetation patch influence vegetation structure and other biodiversity variables?
2. Does distance from remnant primary forest patches have an impact on the biodiversity of revegetated sites?
3. Does patch size affect biodiversity?
4. Does the surrounding landscape matrix affect the biodiversity of the revegetation sites?

METHODS

Study area

The study was conducted in Wellington, New Zealand. The city of Wellington is located in south-western North Island, New Zealand, latitude 41°, 17"S, 174°, 27"E. Wellington covers an area of 290km^2 and has a population of 190,956 people (Statistics NZ, 2013). The topography is varied from gently rolling to steep hills. Elevations range from sea level to 445 m. Wellington has a temperate climate, with a mean annual temperature of 12.8 °C, an average of 2065 sunshine hours and 1249 mm of rain per year. The prevailing wind is north-west, averaging 22 km per hour. Gale days (>63 km per hour) are recorded on an average of 22 days per year (National Institute of Water and Atmospheric Research, 2012).

A total of 24 sites were selected for this study; 20 areas revegetated by Wellington City Council and four areas of mature forest (appendix 2). The 20 sample sites were selectively chosen from across the city. Data files on native revegetation were obtained from Wellington City Council. The vegetation type was restricted to lowland forest, omitting all coastal and riparian plantings, for relative consistency of sample plots. I did not consider sample sites planted after 2008 or less than 0.2 ha. Sites were also eliminated if access was prohibitive, due to private property or density of vegetation. This process resulted in the final set of 20 revegetation sites with an age range of 7 to 18 years post-planting and a size range of 0.23 to 1.27 ha.

Sites were surrounded by a variety of habitats including mown grass reserves, mature exotic trees and regenerating native forest, as well as roads and houses. Many site surrounds were a combination of two or more of these environments, and almost all sites had a track either running through or alongside. All revegetated sites were exposed to a variety of predators, including cats, dogs, rats, mice and stoats.

Four reference sites were selected to represent advanced stages of lowland forest ecosystems within the urban environment. Two areas of remnant native forest were chosen, one area of mature exotic woody forest, and an area of secondary native forest. The four reference sites were considerably older and larger than the sample sites, allowing for the comparison to be made between patch size and age.

Field methods

At each site 2 x 30 m transects were fitted, a minimum of 6 m apart and not less than 2 m from a site edge. Transects were purposefully placed so that they were predominantly within the planted areas of the sites, avoiding open areas of grassland and walkways. Four taxonomic groups of organisms were sampled at each site: vascular plants, birds, lizards, and terrestrial invertebrates. An Animal Ethics Approval was granted by Victoria University of Wellington (appendix 11) and a Wildlife Act Permit was granted by the Department of Conservation to catch and handle lizards, as required by the Wildlife Act 1953 (appendix 12).

Field work was carried out over a 12 month period. Site plots (appendix 4) were set up between September and November 2013. Vascular plant surveys were completed in February and March 2014. Five-minute bird counts (Dawson & Bull, 1975) were carried out September to October 2013, and again in September 2014. The lizard and invertebrate traps were active during February and March 2014. (for exact dates see appendix 6, 6a, 6b)

Vascular plants

Vascular plants were sampled by conducting an abundance-based survey along each transect, encompassing 2 m each side of each 30 m transect. The surveys were carried out in autumn 2014, focusing on structural elements. Percentage cover of the native tree saplings 1-3 m high, native trees 3-6 m high, native trees 6+ m, ground ferns, native seedlings less than 1 m high, exotic trees 20+ m, leaf litter cover, and long grass cover was recorded. Percentage cover was determined by visual estimation. Native seedlings less than 1m high were determined as naturally occurring seedlings through horticultural and field knowledge of plant physiology and local growth rates.

Birds

Bird presence was recorded using the standard five-minute bird count method (Dawson & Bull, 1975). This method of bird count was used for consistency with other bird surveys being undertaken in Wellington, and across New Zealand. Two bird counts per site were conducted, once during spring 2013 and again in spring 2014, between the hours of 6 am and 9 am. The counts were completed within a 3 week period each year, by an experienced observer (a different observer each year). All birds within a 100 m radius of the count station were counted, including those flying overhead, but paying close attention to ensure that the same bird was not counted twice within a five-minute interval. Bird counts were only undertaken on calm mornings with no precipitation.

Lizards

Lizards were sampled using two different methods, during the summer months. Terrestrial lizards were sampled by installing pitfall traps and arboreal lizards with closed cell tree covers (ACOs).

Pitfall traps:

Two pitfall traps were installed per transect, one at each end of each transect, effectively 30 m apart. Pitfall traps consisted of 4 litre round plastic buckets, opened for three consecutive days in late summer 2014. The traps were dug into the ground ensuring the rims were level with the soil surface. Covers were fixed 100-200 mm above ground level, allowing entry for the lizards as well as protection from rain, wind and predators. Each trap was set with a handful of leaf litter in the bottom for refuge, a wet sponge for moisture, and a piece of tinned pear for bait. Traps were checked daily and items replenished as necessary. Any lizards found in the traps were captured and identified to species level, on site, before being released back into the area. Traps were opened on day one, checked on day two, then checked and removed on day three.

Closed cell tree covers:

Two closed cell tree covers per transect were placed on tree trunks at a height of 1.5-2 m. Where no suitable tree trunks were available, tree stumps substituted to fix the tree covers to. Tree covers were fixed to the nearest suitable tree at the beginning and end of each transect. The covers were sheets of black polyethylene foam, 3-4 mm thick, cut to 700 x 300 mm each (Bell, 2009). They were nailed to the trees at least 3 months prior to being checked for lizards sheltering underneath them. Tree covers were checked once in late summer 2014, recordings made and the covers then removed.

Terrestrial invertebrates

Terrestrial invertebrates were sampled by installing lethal pitfall traps. Twelve pitfall traps were positioned per site – six per transect, in two groups of three in a 30-50 cm triangle configuration, at approximately 10 m and 20 m along each transect. The traps were 880 ml round plastic containers, measuring 200 mm deep and 105 mm in diameter, and dug into the ground ensuring that the rims were flush with the soil surface. Each trap was set up with 40 ml of saline solution (25% saturated solution of table salt, NaC1) and a small ball of chicken wire that sat above the level of the solution. The purpose of the chicken wire was to ensure any lizards that fell in could rest on top of the wire until the next daily check was done. Covers were fixed 100 mm above ground level, allowing invertebrate entry while providing protection from rain, wind and predators. Traps were opened for three consecutive days (two nights) during late summer 2014 – opened on day one, checked (for lizards) on day two, and removed on day three. The short time period of three days of sampling was decided on due to constraints related with public safety and lizard sampling. In order to lengthen the sampling period it would have been necessary to use lethal solution for preservation of the invertebrates. As all of the sites were in public parks and reserves and reasonably close to tracks the risk of children or dogs coming in to contact with the solution was taken in to account. The sampling time period was also limited by the need to monitor the pitfall traps on a daily basis for lizards, in order to minimize the risk of lizard mortality and uphold obligations agreed to in the Wildlife Act Authority approval.

In the laboratory the invertebrates were taken out of the saline solution and transferred to 75% ethyl alcohol solution. Identification to order was recorded for each specimen, and further to genus for Orthoptera to determine abundance of weta.

Landscape matrix and isolation data

Each site sampled in this study was identified and outlined on a geographic information system (GIS) before measuring a 100 m buffer zone around each patch. Desktop photointerpretation was undertaken to classify land cover and land use within the buffer zone. Landcover of the matrix measured within the buffer zones was classified as: primary bush, secondary bush, mown grassland, unmown grassland and low scrub, roadways, or residential properties (appendix 5). These six matrix variables were measured in units of square metres. Data used included WCC asset data, with further classification determined by visually assessing aerial imagery (scale 1:500, 0.2 m accuracy). All measurements were made in ARCGIS Desktop v10.2, January 2013.

As an index of isolation the distance to areas of primary bush from each sample site was measured (appendix 8). The measurement was taken between the two closest points of the primary bush site and the sample site. Fragments of primary bush less than 1 ha in area were not included when measuring isolation of the revegetation site.

Statistical analysis

Two broad types of data analysis were undertaken. Firstly, three ordinations of the revegetation and reference sites were undertaken based on the vegetation, bird and invertebrate communities, respectively. The Bray-Curtis measure of community dissimilarity (Bray & Curtis, 1957) was calculated, using the square-root number of individuals for bird species and invertebrate orders, and the percent cover data for plant groups. Non-metric multidimensional scaling (NMDS) was then used to display the relationships between sites in 2 dimensions using the default settings of the metaMDS function of the 'vegan' package (Oksanen et al, 2015), implemented in the 'R' statistical and computing environment v.3.1 (R Core Team, 2013).

Linear regressions were then used to test the effect of four landscape variables on biodiversity responses of interest. The explanatory landscape variables included in the linear models were patch age, patch size, isolation from primary bush and the percentage cover of residential land use in the 100 m buffer. Preliminary checks confirmed that there were no strong correlations between these four variables (all Pearson's $r < 0.3$, all $P > 0.2$). The response variables were the two axes of the NMDS plots, bird richness, number of weta, and square-root number of skinks. Residential land-use was used as a single variable to characterize the nature of the matrix as preliminary analyses had shown that it was strongly correlated ($r=0.99$) with the first axis of a principal coordinates analysis of the proportions of the six land-cover matrix variables (i.e., it captured the major amount of variation between sites in respect of their surrounding land-use). Furthermore, the percentage of residential land-use showed a negative correlation with percentage cover of bush in the matrix ($r= -0.67$).

RESULTS

Native vascular plants

A significant result is the effect of isolation, as reflected in axis 1 (Figure 1). This is largely driven by AS and BGC (site name abbreviations Appendix 3) with higher levels of grass cover and lack of trees in the 3-6 m range, being two of the most isolated sites, and IP at the opposite end being the least isolated and having a large number of trees 3-6 m.

The natural succession of native vascular plants increased with site age, although not significantly. In revegetated sites aged 7 to 10 years the presence of naturally occurring native seedlings typically ranged from 0 – 7.5% (mean value of sites across two transects). Native seedling recruitment increased gradually with the ages of the sites. Sites aged 11 to 18 years recorded 5 – 50% native seedlings. The reference sites, all classified as 64+ years of age, presented with 42.5 – 90% native seedling recruitment.

Establishment of ground ferns was also associated with older sites, with all reference sites supporting a ground cover of ferns, and 4 revegetated sites aged between 9 and 18 years recording the presence of ground ferns (appendix 9).

Ordination of vegetation structure showed reference sites clustered together (Figure 1). The first axis of the NMDS was associated with isolation, while the second axis was not associated with any of the four landscape variables (Tables 1 & 2).

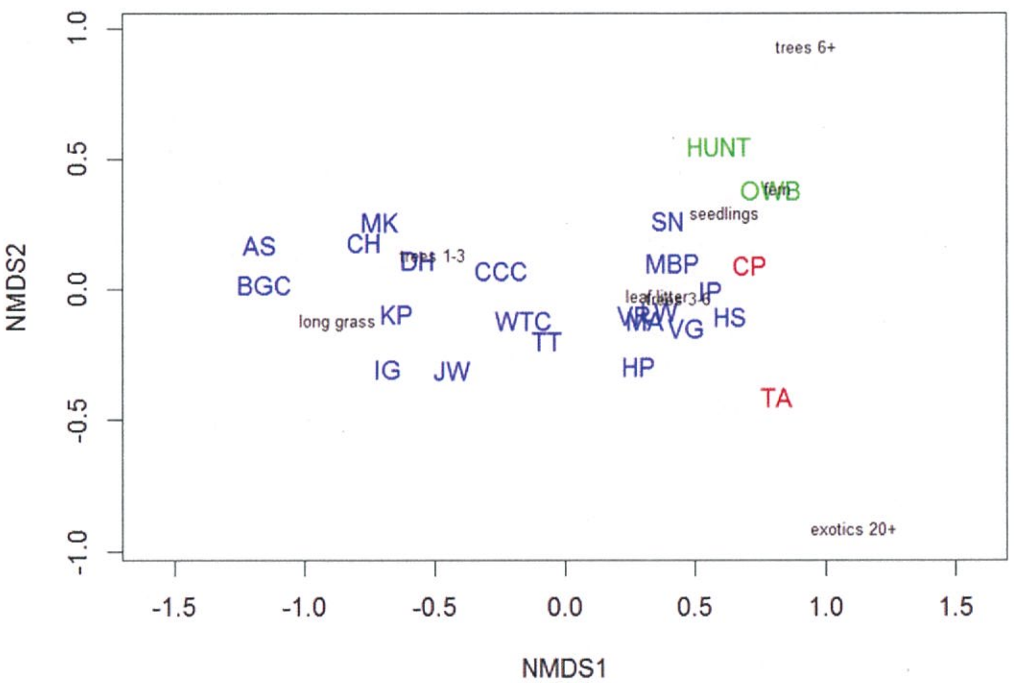


Figure 1. Ordination of plants across revegetation and reference forest sites (non-metric multidimensional scaling), stress = 0.103. Transects from mature reference sites tend to associate with low values of axis 1. Site abbreviations in capitals follow; blue = revegetation sites, green = reference primary native sites, red = reference secondary forest sites. Vectors for site names in lower case black font (see appendix 3 for site abbreviations).

Table 1. Results of a multiple regression testing the effect of four landscape variables on NMDS.axis1.veg Model R-squared = 3.153, $F_{4,15} = 0.062$, p = 0.045.

Variable	Slope	S.E	t	P-value
age	-0.0365	0.0277	-1.315	0.208
size	-1.0679	0.3503	-3.048	0.008**
isolation	0.0064	0.0526	0.123	0.904
%residential	0.0059	0.0043	1.374	0.189

Table 2. Results of a multiple regression testing the effect of four landscape variables on NMDS.axis2.veg Model R-squared = -0.132, $F_{4,15} = 0.4463$, p = 0.7735.

Variable	Slope	S.E	t	P-value
age	-0.0187	0.0169	-1.107	0.286
size	0.0829	0.2139	0.388	0.704
isolation	0.0064	0.0321	0.202	0.843
%residential	-0.0002	0.0026	-0.078	0.939

Birds

A total of 1274 individuals, from 29 species, were recorded in the five-minute bird counts. Forty one percent of species were native, with silvereye (*Zosterops lateralis*), tui (*Prosthemadera novaeseelandiae*) and grey warbler (*Gerygone igata*) the three most abundant native species. The most commonly recorded introduced species were blackbird (*Turdus merula*), starling (*Sturnus vulgaris*) and chaffinch (*Fringilla coelebs*).

Tui and silvereye were present in 23 of the 24 sites, and grey warbler in 22 of the 24.

There was no relationship between bird richness and site age, size, or isolation, neither was bird richness affected by the proportion of residential land-use in the surrounding matrix (Table 3).

Table 3. Results of a multiple regression testing the effect of four landscape variables on bird richness per site. Model R-squared = 0.308, $F_{4,15}$ = 1.672, p = 0.209.

Variable	Slope	S.E	t	P-value
age	-0.148	0.102	-1.447	0.168
size	1.5	1.29	1.166	0.262
isolation	0.166	0.194	-0.858	0.404
% residential	-0.028	0.016	-1.773	0.097

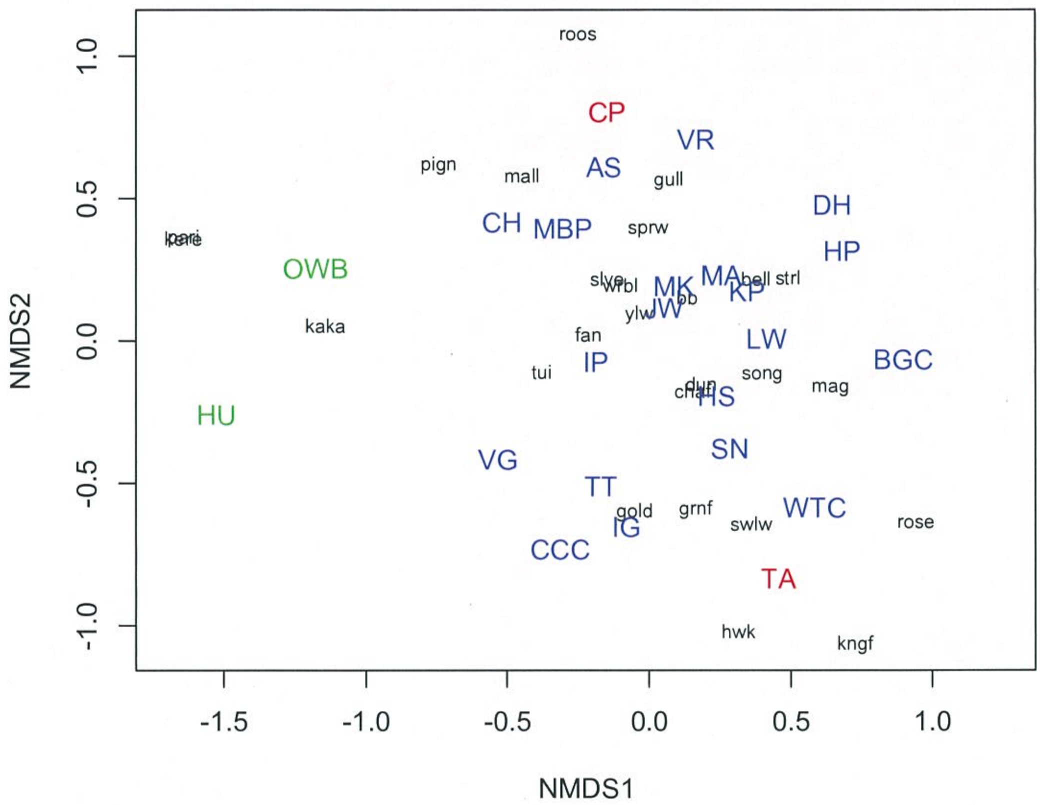


Figure 2. Ordination of the bird species community across revegetation and reference forest sites (non-metric multidimensional scaling), stress = 0.24. Site abbreviations (appendix 3) in capitals follow; blue = revegetation sites, green = reference primary native sites, red = reference secondary forest sites. Vectors for abundant bird species in lower case black font (see appendix 7 for species abbreviations).

The first axis of the NMDS ordination (Fig 2) shows that large native birds, kereru and kaka, are associated with mature native forest, OWB and HU. Introduced birds such as eastern rosella and magpie are mainly seen in open landscapes and near edges of tree stands, BGC and WTC. The second axis appears to reflect proximity to the built urban environment, supported by rock pigeons and roosters. The position of the kingfisher, in relation to both axes is representative of elevation and openness for hunting prey, along with nesting opportunities in tall mature trees. There was no significant association between the four landscape variables and either of the NMDS axes (Tables 4 & 5).

Table 4. Results of a multiple regression testing the relationship between four landscape variables and NMDS.axis1.birds Model R-squared = 0.104, $F_{4,15}$ = 0.434, p = 0.782.

Variable	Slope	S.E	t	P-value
age	-0.0047	0.0173	-0.274	0.788
size	-0.0668	0.2184	-0.306	0.764
isolation	-0.0215	0.0328	-0.657	0.521
% residential	0.0026	0.0027	0.965	0.350

Table 5. Results of a multiple regression testing the effect of four landscape variables on NMDS.axis2.birds Model R-squared = 0.104, $F_{4,15}$ = 0.434, p = 0.782.

Variable	Slope	S.E	t	P-value
age	-0.0047	0.0173	-0.274	0.788
size	-0.0668	0.2184	-0.306	0.764
isolation	-0.0215	0.0328	-0.657	0.521
% residential	0.0026	0.0027	0.965	0.350

Lizards

The one lizard species found in this study was the northern grass skink (*Oligosoma polychroma*). A total of 38 northern grass skinks were captured in the pitfall traps, from six revegetated sites (appendix 10). Seventeen skinks were captured in the lizard traps, and 21 skinks in invertebrate traps. Although all 24 sites were sampled for lizards, captures were only made at six of the revegetated sites. Long grass was present in all six sites, along one or other of the two transects, and the skinks were located in the grass habitat. Thirty two of the 38 skinks were found in one site alone (DH).

Table 6. Results of a multiple regression testing the effect of four landscape variables on the square root of lizards per site. Model R-squared = 0.417, $F_{4,15}$ = 2.683, p = 0.072.

Variable	Slope	S.E	t	P-value
age	0.137	0.392	0.350	0.731
size	1.937	4.948	0.371	0.716
isolation	0.451	0.743	0.607	0.553
% residential	0.191	0.061	3.151	0.007 **

Lizards were located in sites with a higher proportion of residential area in the matrix, but were not influenced by the age, size or distance from primary bush (Table 6).

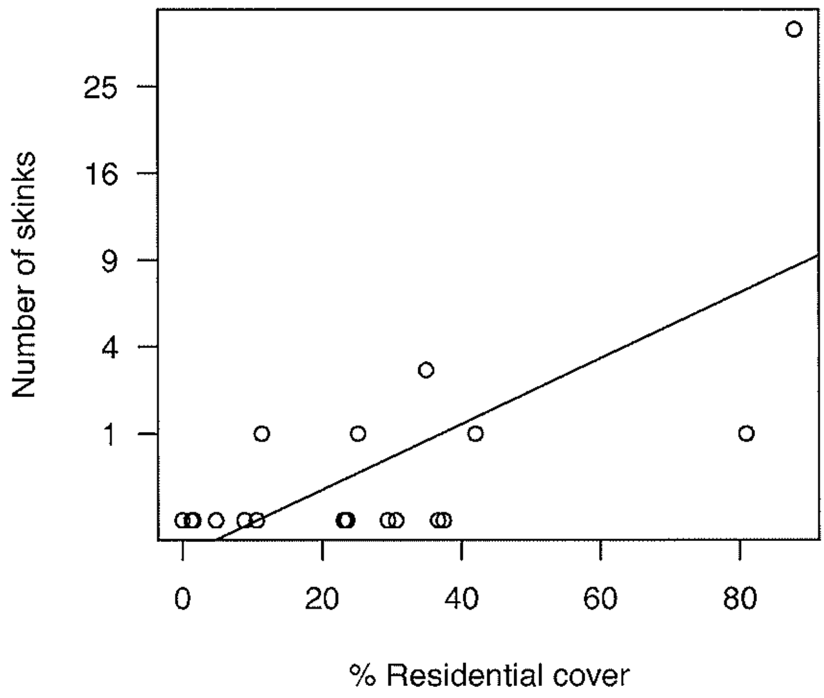


Figure 3. Number of skinks per site in relation to percentage of residential cover of landscape matrix. Model R-squared = 0.494, $F_{1,18}$ = 17.57, p = 0.0005. Number of skinks counted from four lizard pitfall traps and 12 invertebrate pitfall traps per site.

One skink was found in a site with less than 20% residential landuse in the surrounding 100 m, four skinks in sites with 21-40% residential landuse, and 34 skinks in sites with 41-100% residential landuse.

Terrestrial invertebrates

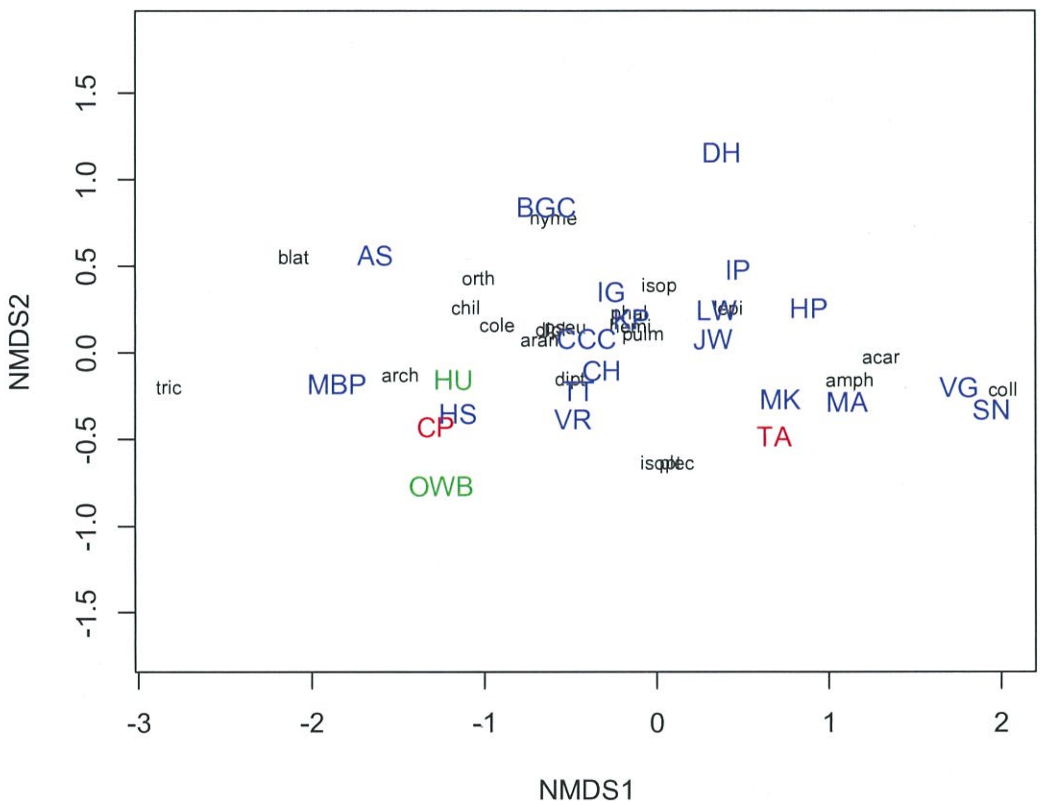


Figure 4. Ordination of the invertebrate community across revegetation and reference forest sites (non-metric multidimensional scaling), stress = 0.09. Site abbreviations (appendix 3) in capitals follow; blue = revegetation sites, green = reference primary native sites, red = reference secondary forest sites. Vectors for invertebrates in lower case black font (see appendix 7 for genus abbreviations).

The second axis of the NMDS ordination (Fig 4) represents age of the sites, with DH & BGC less than 10 years old, and OWB, CP and TA all reference sites over 60 years. HS, VR and SN range in age between 15 and 20 years. Isoptera, being decomposers of dead wood, are positively associated with the older sites. There was no significant association between the four landscape variables and either of the NMDS axes (Tables 7 & 8).

Table 7. Results of a multiple regression testing the effect of four landscape variables on NMDS.axis1.inverts Model R-squared = 0.2484, $F_{4,15}$ = 1.239, p = 0.3362.

Variable	Slope	S.E	t	P-value
age	0.0175	0.0470	0.372	0.7151
size	-0.3590	0.5943	-0.604	0.5548
isolation	-0.1833	0.0892	-2.055	0.0578.
% residential	0.0024	0.0073	0.325	0.7496

Table 8. Results of a multiple regression testing the effect of four landscape variables on NMDS.axis2.inverts Model R-squared = 0.2484, $F_{4,15}$ = 1.239, p = 0.3362.

Variable	Slope	S.E	t	P-value
age	0.017500	0.047049	0.372	0.7151
size	-0.359048	0.594298	-0.604	0.5548
isolation	-0.183283	0.089203	-2.055	0.0578.
% residential	0.002372	0.007296	0.325	0.7496

Terrestrial invertebrates – weta

A total of 171 weta individuals were recorded across the 24 sites. One hundred and fifty one tree weta (*Henideina crassidens*) were found under the closed cell tree covers (ACOs) in 18 of the 24 sample sites. Fourteen cave weta were found across 8 sites, and 6 ground weta from 3 sites. The cave weta and ground weta were caught in invertebrate pitfall traps.

The number of weta (ground, cave and tree weta combined) per site increased with increasing distance from primary bush (Table 9, Fig. 5), but was not influenced by the age or size of the revegetation site, or the proportion of residential area in the surrounding matrix (Table 9).

Table 9. Results of a multiple regression testing the effect of four landscape variables on the number of weta per site. Model R-squared = 0.314, $F_{4,15}$ = 1.713, p = 0.200.

Variable	Slope	S.E	t	P-value
age	0.706	0.461	1.531	0.147
size	0.0003	0.0006	0.536	0.600
isolation	0.0020	0.0009	2.309	0.036 *
% residential	0.0178	0.0715	0.249	0.806

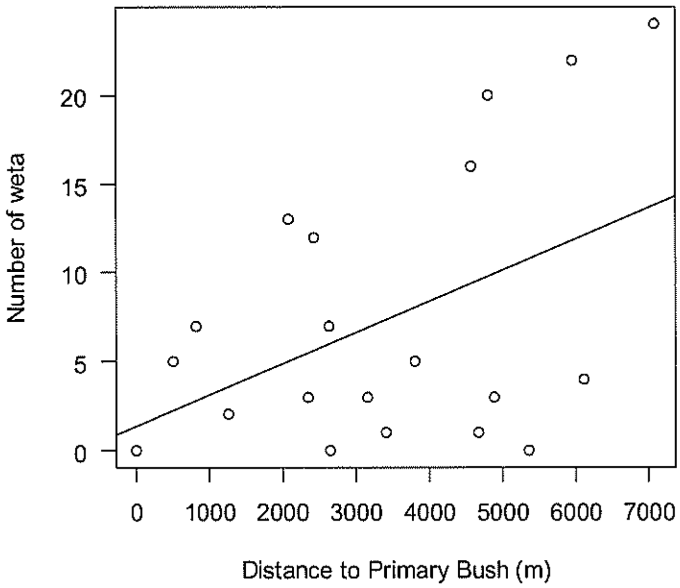


Figure 5. Number of weta per site in response to distance to nearest primary bush. R^2 = 0.159, $F_{1,18}$ = 4.59, p = 0.046. Number of weta counted from 4 ACOs and 12 pitfall traps per site.

DISCUSSION

The objectives of this study were to identify relationships between biodiversity and the age, size or isolation of revegetated sites within an urban environment. I also tested for effects of the landscape matrix on biodiversity.

Vascular plants

Structural complexity and natural regeneration of native vascular plants increased with the age of the site. Age was weakly correlated with the second axis of the multi-dimensional plot; sites that recorded long grass within them were less likely to have young seedlings and ferns present. Seedlings and ferns were more closely aligned with tall and mid-range trees.

Isolation from primary bush was an important variable significantly correlated with first axis of the NMDS of vegetation structure. Isolation appeared to be positively associated with the natural succession of native plants, as revegetation sites in closer proximity to mature forest sites had a higher proportion of native seedlings present.

These results are consistent with the outcomes of Reay and Norton's (1999) assessment of restoration plantings in the South Island of New Zealand. They found that vascular plant species diversity and species richness increased with increasing age of the restoration sites. The youngest site, with 12 year old plantings, still had approximately 50% grass cover and had the lowest values for species diversity and species richness. As the site ages increased so too did the species diversity and richness. A grassland site was included in their study to represent pre-restoration conditions. Regenerating plants were absent from the grassland site (Reay & Norton, 1999). Competition with grass cover and a lack of vegetative structure and fruiting resource were reported to be major limitations in natural forest regeneration in a recent Puerto Rican study of urban forest restoration (Ruiz-Jaen, 2006).

A recent assessment of Hamilton (New Zealand) city's urban gully restoration plantings (MacKay, Wehi & Clarkson, 2011) found that natural regeneration was not as successful in the planted plots as it was in the control plots (of same age) or the mature forest. The ecological function of the planted plots was found to improve with age, but was not as advanced as the control plots or mature forest. Competition from exotic species and distance to native seed sources were suggested as the main barriers to more successful functioning (MacKay, Wehi & Clarkson, 2011).

The results of my study suggest that age is positively associated with the natural succession of the flora. The establishment of native seedlings increased with the age of the sites, as did the presence of ground ferns. As trees and shrubs mature the structure of the habitat develops, increasing the benefits provided to fauna.

Dispersal of seed requires seed availability and the presence of appropriate dispersal agents. The success of animal dispersal agents, such as birds, is dependent on site attributes attracting the birds in the first instance. Wotton & McAlpine (2013) list perch availability, structural complexity and food sources as key attractants. Consequently, as the age of the vegetation increases, the site becomes more appealing to avian fauna. Perches and nesting sites develop as trees and shrubs grow, and nectar and fruits increase in abundance as the plants mature. It follows that as birds are more frequently attracted to a site, the seed dispersal rates increase. McDonnell & Stiles (1983) conducted field research investigating the importance of structural complexity of vegetation for the recruitment of bird-dispersed plant species. The results showed that seed dispersal was greater at sites with higher structural complexity and fruit availability. The younger the site, the lower the seed dispersal rate. A single layer of vegetation was described as the impeding factor of the youngest site – equating to a lack of perching structure and little or no fruit. Artificial perches were installed at one young site and this factor alone increased bird visitation, and consequently seed dispersal (McDonnell & Stiles, 1983). The results of my study concur with McDonald and Stiles' observations, in that the younger sample sites, with single layer vegetation and therefore little structural complexity, were not supporting much, if any, plant recruitment. This suggests that seed dispersal, either by parent plant or dispersal agents, is not yet happening within these immature sites.

Birds

Bird richness did not appear to be influenced by age or size of the site, isolation, or the proportion of residential land in the surrounding matrix. However, the results of a community ordination showed that large native birds, kereru and kaka, were associated with mature native forest. Although kereru and kaka were uncommon in the two bird counts, kereru were only sighted at primary forest sites, and kaka at or close to primary forest. Both species are known to range in search of food, with populations based around native forest patches (NZ Birds Online, 2015; Lindsay & Morris, 2000). Kereru can travel up to 60 km a day to feed on seasonal fruit (NZ Birds Online, 2015) or up to 1500 m in a single flight (Wotton, 2007). Kaka living on northern off-shore islands have been radio-tracked flying up to 25 km between islands as they move around the forest trees feeding on fruit (NZ Birds Online, 2015). Regular observations of kereru and kaka across Wellington city would suggest that these birds are travelling around the urban landscape to feed on the trees within the wider mosaic, with the populations based in primary native forest patches.

The three most common native birds recorded in this study were tui, silvereye and grey warbler. Tui and silvereye were present at 23 of the 24 sites, and grey warbler at 22 sites. These results are consistent with those of the Greater Wellington Regional Council [GWRC] data. GWRC initiated a new bird monitoring programme in 2011 (McArthur, Harvey & Flux, 2013). The programme has 100 bird count stations across Wellington city parks and reserves and is conducted annually in November and December. Tui, silvereye and grey warbler were the top 3 native birds recorded in 2011 and 2012 (McArthur, et al., 2013). Previous surveys conducted by Pacific Eco-Logic Ltd reported a significant increase in tui counted in Wellington city reserves between 2001 and 2009 (Froude, 2009). Miskelly, Empson, & Wright (2005) reported an increase of native bird species diversity, abundance and distribution in Wellington city parks and reserves. This increase was attributed to effective pest control in forest reserves, the completion of a predator-proofed wildlife sanctuary (Zealandia, 2015) in Wellington, and a pest-free island (Matiu-Somes) in Wellington's harbour.

There is strong evidence that bird species richness is positively associated with patch size, reduced isolation, and connectivity to large areas of forest (Ferenc, Sedlacek, & Fuchs, 2014). Ferenc et al. (2014) determined that the amount of tree cover, tree species richness, and structural complexity of trees were important factors in supporting enriched bird communities in the city of Prague, Czech Republic. They suggested that trees and shrubs of private gardens were a valuable part of the urban green cover, improving the connectivity between larger habitat patches. A complex structure of vegetation is beneficial in supporting bird species richness, indicating that shrub layers are important within the overall tree cover (Sandstrom, Angelstam, & Mikusinski, 2006; Evans, Newson, & Gaston, 2009).

The lack of positive correlation in this study, between bird richness and age, size or residential cover, may be a result of sufficient overall urban tree cover to sustain a range of bird communities. It is possible that the landscape matrix of Wellington has a network of trees and shrubs that are providing suitable connection and structural complexity to support widespread bird communities. The current urban tree cover network is inclusive of remnant forest patches, natural secondary growth patches, revegetation sites, parks, reserves, street trees and residential plantings.

Lizards

All six sites where lizards, specifically northern grass skinks, were caught were located in outer suburban residential areas. These areas were primarily developed within the last 60 years, as opposed to central city suburbs with older infrastructure and higher density housing. The northern grass skink is the most widespread lizard in New Zealand, a generalist species found in abundance in native habitats and urban environments (Towns & Elliott, 1996; Whittaker, 1987). Van Heezik & Ludwig (2012) found that the northern grass skink was uncommon in central Dunedin suburbs, being mainly associated with residential properties in outer suburbs. Although distribution was reported to be patchy in the Dunedin study, long exotic grass was the most positive vegetation type influencing skink presence (Van Heezik & Ludwig, 2012).

Long exotic grass was present in each of the six Wellington sample sites of my study where skinks were found. However, long grass was also present in sites where no skinks were detected. This observation suggests that skinks are more likely to be found in areas of long grass but long grass will not guarantee the presence of skinks. The presence of northern grass skinks in open long grass habitat is congruent with the type of habitat they occupy – "a very wide range of generally open habitats including... grasslands, shrublands, rocky sites... " (Melzer & Bell, 2014, p.8).

Weta

Weta were recorded at 21 of the 24 sites, indicating that weta are widespread in the Wellington urban landscape. The results showed no association between weta and site age or size, or the proportion of residential land in the surrounding matrix. There was, however, a positive relationship with distance to primary bush. Weta abundance increased with increasing distance from primary bush. Tree weta in particular are very well adapted to suburban habitats, and have developed avoidance behaviours to minimise capture by introduced mammalian predators (Gibbs, 1998).

It is possible that weta living nearer to areas of primary bush are more vulnerable to native predators, such such as *Ninox novaeseelandiae* (morepork owl) and *Nestor meriodionalis* (kaka). These 2 bird species are more likely to be living in or near mature bush (NZ Birds Online, 2015; Lindsey & Morris, 2000) and therefore primarily foraging for prey in the vicinity. This postulation would suggest that native avian predators, although less widespread, are more successful predators of weta than introduced mammalian predators such as the Norwary rat (*Rattus norvegicus*) and the ship rat (*Rattus rattus*). It is, however, widely recognised that alien mammals are currently the most extensive and significant predators of weta (Gibbs, 1998; Trewick & Morgan-Richards, 2000; Rufaut & Gibbs, 2003; Watts,Armstrong, Innes, & Thornburrow, 2011).

CONCLUSION

Revegetation sites in urban Wellington appear to require approximately 10 years of growth before effective facilitation of native seedling growth commences. Sites over 10 years also support a more complex structure with tree heights ranging up to 6 m. In contrast, the younger sites (under 10 years) did not generally have taller trees, native seedlings or ground ferns present.

Revegetation sites closer to mature forest sites had a comparatively high proportion of native seedlings present. This indicates that isolation was strongly associated with the structural vegetative complexity of these sites.

Site age, size or isolation did not have any effect on bird richness in this study. There was also no relationship between bird richness and the proportion of residential land-use in the surrounding matrix.

Large native birds (i.e. kereru and kaka) do appear to be associated with remnant forest patches, highlighting the importance of retaining and protecting such areas. It follows that successful complex revegetation plantings undertaken now will be valuable resources for large native birds in decades to come.

Northern grass skinks were most abundant in sites with grass cover and were positively associated with the proportion of residential land-use in the surrounding matrix, but not with site age, size or isolation.

Weta were widespread across the revegetated sites, and showed increasing numbers with increasing distance from mature primary bush. Further monitoring of weta to decipher the reasons for an increase in numbers as the distance from mature primary bush increases would be beneficial. It would also be beneficial to carry out pest monitoring near the weta monitoring sites to identify the impact that mammalian predators may or may not have on weta populations.

The emergent result was that age and isolation do appear to influence the structural vegetation composition of sites. In particular, the older the age of the site the more structurally complex they were. Higher trophic levels (birds, lizards and invertebrates) showed few direct relationships with site age, isolation, size and matrix, although it was generally true that revegetation sites had distinctively different faunal compositions from the mature reference sites, suggesting that further successional processes and colonisation should be expected (and perhaps encouraged) beyond 7-18 years of establishment.

Chapter 3

MANAGEMENT RECOMMENDATIONS

INTRODUCTION

> "Urban areas are highly modified and complex landscapes, within which green or open areas are seen as valuable for human well-being as well as wildlife" (Angold et al., 2005, p.1).

Native revegetation programmes in urban centres are important for maintaining and enhancing biological diversity and also, as Angold et al. (2005) acknowledge, of high value for the citizens and ratepayers of cities. The Wellington City Council [WCC] revegetation programme is now in its eighteenth year, which displays a substantial investment of time and money in developing the native greening and biodiversity of the city. The objectives of this urban native greening project have been modified over the last ten years to focus more directly on biodiversity, conservation and connectivity. The Council is dedicated to planting a total of 2 million trees by 2020, via a number of initiatives, one of which is the native revegetation programme. This particular programme currently provides for the planting of 45,000 eco-sourced plants across the city each year. This programme alone is a major restoration initiative and the biodiversity benefits include conservation of species and the provision of ecosystem services, such as; clean water, clean air, erosion control, and healthy soils, amongst others. The council investment into the native revegetation programme indicates that evaluation and study of the biodiversity outcomes are warranted on an ongoing basis. My study, which included an extensive fieldwork programme, was designed in such a way that it could be replicated in the years ahead.

The following is a summary of the questions and key findings from my research of the WCC native revegetation programme, followed by a discussion of potential recommendations for land managers and policy-makers.

1. **How does age of the revegetation patch influence vegetation structure and other biodiversity variables?**

Results from my research demonstrated that native plant regeneration increases with site age. Revegetation sites less than ten years old did not exceed 7.5% native tree seedling cover, whereas sites aged between 11 and 18 years recorded 5 – 50% native seedling cover. Remnant forest patches, mature secondary forest, and mature exotic forest with native understory were found to support 42.5 – 90% native seedling recruitment. Ground cover ferns were also recorded in all four mature sites (Otari-Wilton's Bush, Huntleigh Park, Te Ahuairangi, Cenntenial Park), and just five of the revegetated sites (Makara Bike Park, Izard Park, Wadestown Tennis Court, Lakewood Reserve, Seton Nositor) aged between 9 and 18 years. These results suggest that significant benefits from revegetation programmes may become apparent in as little as 15 years.

2. **Does distance from remnant forest patches have an impact on the biodiversity of revegetated sites?**

Isolation was strongly associated with the structural vegetative complexity of sites. Revegetation sites in closer proximity to mature forest sites had a relatively high proportion of native seedlings present (Figure 1, chapter 2).

The results of the research also suggest that isolation has an effect on the abundance of weta. The number of weta increased as distance from mature forest increased (Table 9, chapter 2). As discussed in Chapter 2 the reasons for this are not clear as they were not identifiable in the research design. It may be that native avian predators of the weta are more successful than introduced mammalian predators. Alternatively, it is possible that mammalian predators are present in higher numbers in proximity to mature forest.

Large native birds, specifically kereru and kaka, were only recorded within or in close proximity to mature forest (Figure 2, chapter 2). Kereru were sighted at Otari-Wilton's Bush and Ian Galloway Park. Kaka were recorded at Otari-Wilton's Bush, Ian Galloway Park, Izard Park, Huntleigh Park, Karori Park and Makara Bike Park.

3. Does patch size affect biodiversity?

The results of my research suggest that size of the revegetated sites did not influence biodiversity measures. However, as all of the sample sites were less than 1.3 ha, the lack of relationship to size may simply be due to the relatively small size of all sites in the research sample, and the fact that many of them abut onto secondary regrowth forest or a combination of exotic forest with a native understory.

4. Does the surrounding landscape matrix affect the biodiversity of the revegetation sites?

The percentage of residential land use in the matrix showed a positive relationship with the number of skinks found in this research (Figure 5, chapter 2). In Chapter 2 I identified that northern grass skinks were associated with open grassland habitats, which are more common in the outer suburbs of large New Zealand cities.

There was no relationship between landscape matrix (summarized by the percent residential land use) and the bird or invertebrate communities, or seedling recruitment of native plants.

MANAGEMENT RECOMMENDATIONS

Following a comprehensive field research project conducted over 2013/14 to review the biodiversity outcomes of the Wellington City Council [WCC] native revegetation programme, recommendations are proposed as adjuncts and continuation to the management and development of this revegetation programme. These recommendations are in support of a programme that is successfully contributing to the biological diversity of Wellington City. After each set of bullet-pointed recommendations I review the logic behind the recommendations, based on published literature and my own results. In some cases, specific sites are used as examples to illustrate particular points.

Recommendation One

Continue planting native species

- Continue to revegetate areas of low amenity and biodiversity value, replacing weedy sites and unused grassland sites with native plants.
- Continue to infill existing plantings with appropriate species to obtain canopy cover.
- Increase the size of planted sites wherever possible.

The continuation of planting native species is important, as native revegetation of the urban environment will help to restore ecosystems and maintain genetic biodiversity (Janssen, 2004).

In 2009 WCC contracted Peter Handford & Associates to undertake an audit of a sample of 55 existing revegetation sites throughout Wellington city. Handford (2009) reported that revegetation plantings were taking about 9 years to reach a canopy cover, 3 m high and over, on most sites. The results of my study suggest that revegetation sites are taking over 10 years to achieve recruitment of native seedlings above 5% (ground cover).

Furthermore my study found that native seedlings and ferns were associated with trees 3 m high and over.

A bush vitality assessment score sheet designed for New Zealand native bush patches (Janssen, 2004) lists three size categories, over 25 ha, 5-25 ha, and less than 5 ha. The under 5 ha category is defined as 'small habitat area' with varied levels of ability to support populations depending on the average width of the site. The lowest scoring size and shape, less than 5 ha and average width under 20 m, is charactersied as follows "Small habitat area, narrow strip, likely to be forest along a river margin, no forest interior, all edge forest. Sites that can support good populations of native regeneration and shrubs may sustain fragile original populations" (Janssen, 2004, p.175). The maximum 1.3 ha size of the revegetated sites in my research sample therefore resemble small strips of edge forest. Although sites of this small size will not support an 'interior' zone they are capable of supporting edge populations and supporting larger patches by providing linkages. They may also buffer or seal the edge of larger fragments where they have been planted alongside existing bush.

It is apparent from my study that revegetation sites with long grass cover are less likely to support woody plant seedling recruitment. Dispersal agents of woody plants require plants that offer perching opportunities and seasonal fruits to feed on. In general, WCC revegetation sites with long grass cover are younger sites with immature plants unable to offer birds perches and fruit. A recent study on Mana Island in the Wellington Region, reported considerably higher seed rain density at forested sites than in areas of grassland – 3742 seeds per m^2 forested and 7.8 seeds per m^2 grassland (Wotton & McAlpine, 2013). Further to that, the likelihood of woody plant seedlings emerging through the grass is not very high, as seedling growth and survival is impeded by grass sward (Anton, Hartley, & Wittmer, 2015; Gunaratne et al., 2009; Ruiz-Jaen & Aide, 2006).

Planting a species-rich mix of site appropriate woody trees, shrubs and herbaceous plants will further enhance the success of revegetation projects by increasing canopy complexity and species recruitment (Zedler, Callaway, & Sullivan, 2001).

Recommendation Two

Plan to increase connectivity

- Provide as many 'stepping stones' as possible, identifying and filling in gaps wherever possible
- Work with council teams to encourage suitable plant choices for birds, insects, lizards across the city
- Communicate biodiversity messages to all staff to align them with relevant plans

Increasing and enhancing connectivity "will improve food source availability, mating opportunities and allows pollen and seed interchange between bush remnants" (Janssen, 2004, p. 27).

Patch isolation may limit the ability for species dispersal (Saunders, Hobbs, & Margules, 1991; Zipperer et al., 2012). Species colonization is associated with modes of dispersal, and "dependent to some extent on the distance of the remnant from other areas of native vegetation, be they other remnants or nearby uncleared areas" (Saunders et al., 1991, p.23). Plant dispersal is achieved via wind, gravity, water, explosion, and animal vectors. Animal dispersal relies on the ability of the individual to traverse the landscape. The degree of connectivity between patches adds to the effects of isolation, or distance (Saunders et al., 1991). The principal dispersal methods between patches of primary, secondary and recently revegetated areas in Wellington are via wind and birds. Dispersal of seeds by lizards is limited by reduced populations, ranging ability and small range of plant frugivory (Jana Prado, 2012). Weta are also seed dispersers but again not effective over longer distances as they have small home-ranges (Jana Prado, 2012).

Connectivity can be enhanced by maintaining, or providing, "linkages among patches through a network of habitat stepping-stones" (Zipperer et al., 2012, p.543). Linking urban green areas is important for maintaining and enhancing urban biodiversity. As noted by Rudd, Vala & Schaefer (2002) in their analysis of urban green space connectivey, "a way of preserving the biological integrity of a landscape, corridors and habitat matrices must be in place to allow dispersal between green spaces" (p.374). Vegetation corridors or stepping-stones provide the link between patches (remnant, secondary, revegetated, and amenity), facilitating movements of fauna and flora. (Beier & Noss, 1998; Damschen, Haddad, Orrock, Tewksbury, & Levey, 2006; Lindenmayer, Margules, & Botkin, 2000; Savard, Clergeau, Mennechez, 2000).

The network of WCC revegetated sites sampled in this research is contributing to the linkage of remnant and secondary patches of native forest in Wellington, represented in the data showing that native plant recruitment is occurring within revegetation sites. It appears that recruitment was higher at closer proximity to mature native forest, and at an increasing rate as the site age increases. However, recruitment was not a response variable that was tested by itself (only as part of the multivariate description of vegetation structure) and so is an opportunity for further study.

Provision of connectivity across the urban landscape necessitates forward planning, taking into account the nature of the matrix as inter-patch movement is influenced by the heterogeneous landscape matrix.

> The movement of indigenous birds such as the kereru and tui between different zones in the city demonstrates the need for co-ordinated management of biodiversity around cities. Kereru nest in cities such as New Plymouth with high indigenous cover, but in other cities such as Invercargill and Hamilton with reduced indigenous land-cover in the central city, they are absent or visit seasonally from the edges (Clarkson, Wehi, & Brabyn, 2007, p. 451)

My study showed that 38% of bird species, and 33% of individuals (birds), across the Wellington urban landscape are native. While this is a fairly positive result, it could be improved with the further enhancement of native connectivity. Large native birds such as kereru and kaka were associated with mature native forest (Figure 2, chapter 2), and while kaka are commonly observed foraging further afield, kereru are not as commonly seen across the city of Wellington (personal observations). Results from the Great Kereru Count 2014 (Brumby, Hartley, & Salmon, 2015) show that 70% of kereru sightings were in suburbs along the western hills of Wellington, where the largest remnants of native forest are located. Research shows that while kereru will browse on introduced tree species they are selective and prefer native trees, in particular *Sophora, Parsonsia, Coprosma, Paratrophis, Melicytus, Hoheria,* and *Plagianthus* spp (Clout & Hay, 1989). Along with this are the native trees with fruits larger than 1cm diameter (Clout & Hay, 1989), such as tawa (*Beilschmedia tawa),* taraire (*B.taraire*), miro (*Prumnopitys* ferruginea), puriri (*Vitex lucens*), karaka (*Corynocarpus laevigatus*), tawapou (*Planchonella costata*), (Wotton, 2007; Emeny, Powlesland, Henderson, & Fordham, 2009). Kereru are important seed dispersers of native plants, being capable of consuming large fruit and depositing the seed intact (Wotton, 2007; Kelly, Ladley, Robertson, Anderson, Wotton, & Wiser, 2009). The above list of plants browsed by kereru is by no means exhaustive, and further to Clout & Hay's findings (1989) singling kereru out as the only native bird to feed on fruits >14mm diameter, kokako (*Callaeas cinerea*) and tui have since been reported feeding on tawa, taraire, puriri and karaka (Kelly et al., 2009).

The planned inclusion of native trees and shrubs throughout the urban landscape will enhance connectivity for birds and insects (Reale & Blair, 2005; Conniff, 2014). Reale & Blair (2005) suggested that the increased use of native plants in the urban landscape would potentially increase the number of urban-dwelling native bird species. They arrived at this conclusion after investigating nesting success of birds in urban areas. The results showed that the vegetation gradient from remnant patch to urban centre reduced from tall native trees to short specimen trees and low growing shrubs. Nesting success was greatest at higher nesting sites (i.e. taller trees), and with "mean nest height decreasing with urbanization" (Reale & Blair, 2005, p.9) the chances of successful nesting decrease as distance from remnant patches of tall native trees increase. Their research is in line with the results of a quantitative review of matrix effects (Prevedello & Vierira 2009) suggesting that structural similarity between the matrix and the remnant patches increases connectivity. Structural similarity is best gained by planting native species similarly growing in the remnant patch.

Emeny et al. (2009) concluded "even small patches of native species in exotic forest can benefit kereru" (p.122), which could feasibly be translated to include the inclusion of small patches of native species in the urban environment. In a city such as Wellington this is entirely possible, given the inner green belt, stream and gully systems that currently have a combination of exotic trees and regenerating natives. A concerted effort to plant more kereru-friendly plant species in these areas would enhance the foraging field opportunities for this species across the city.

A new initiative in Baltimore County, Maryland, requires that "canopy trees, rather than specimen or ornamental trees, must make up 80% of any planting on county land, and half of them need to be oaks" (Conniff, 2014, p.2). Oak trees, 22 species of them, are native to the area but had not been commonly planted by the state. Earlier research revealed that oak trees harbour large quantities of caterpillars, and native birds require large numbers of caterpillars to feed their young (Conniff, 2014). Entomologist, Douglas Tallamy, stated that "if you want the birds, you need the caterpillars, and to get the caterpillars you need the right trees" (Tallamy cited in Conniff, 2014, p.2).

Habitat and foraging opportunities for native birds will be augmented with the increase of native trees, which will potentially be followed by a rise in native seed dispersal and, as Nathan & Muller explain, this is important because "[s]eed dispersal determines the potential rates of recruitment, invasion, range expansion and gene flow in plant populations" (2000, p. 282). The provision of increased native seed sources within the urban environment will assist with natural regeneration throughout the landscape matrix.

WCC has an in-house team of Parks specialists (horticultural, arboricultural, urban ecology, landscape architecture, project management) that, with collaboration, could design a connectivity strategy for the linking of urban green areas. The strategy requires clear guidelines around species selection, and dedicated communication concerning upcoming works - in parks, reserves and streetscapes. Collaboration across disciplines involved in developing biodiversity policies is an important aspect of successful and ongoing ecological restorations programmes (Wyant, Meganck, & Ham, 1995).

Recommendation Three

Structural diversity

- Plant a variety of species, including emergent species, to achieve structural diversity – plan the enrichment plantings to occur over several years
- Leave large trees on site, even if they are exotic species
- Leave logs on site, and import them if feasible

Structural diversity is important in order to provide a variety of foraging opportunities for birds, lizards, insects, and invertebrates. Enrichment plantings of canopy and emergent tree species, vines and ferns are best planned to be undertaken in subsequent years following the initial plantings. Some of the canopy and emergent species such as tawa, matai, miro, and rimu will establish more readily when planted into existing plantings as they are then afforded some protection from exposure to the elements. Vines, supplejack (*Ripogonum scandens*), native jasmine (*Parsonsia* spp.), and native passionfruit (*Passiflora tetrandra*) require tree framework to climb up and so cannot be planted until earlier plantings have grown to a suitable supportive size. Ferns and ground covers can be planted after some tree cover is established to replicate the required lower light levels and protection from above that they naturally grow under.

The influence of site age on succession and colonization is attributed to the growth and development of the woody vegetation. As the vegetation structure develops the herbaceous ground cover decreases, allowing the emergence of tree seedlings. The maturation of woody vegetation also increases leaf litter and alters the microclimate under the canopy, which provides a more favourable environment for invertebrates, lizards, birds and further seedlings (Ruiz-Jaen & Aide, 2006).

Structural diversity of the vegetation facilitates wider species diversity and ecosystem processes. Plants and animals colonize more structurally diverse areas of vegetation, and nutrient cycling occurs as litter layers build up (Ruiz-

Jaen & Aide, 2005; Reay & Norton, 1999). The structural diversity of revegetated sites can be planned for in the planting design stage, selecting a range of species that will provide vertical and horizontal elements. A greater number of vegetation layers is positively related to bird community diversity as they provide more perching, nesting and feeding opportunities, along with hiding spots for protection from predators (Marzluff & Ewing, 2001). I visually estimated the density of three height layers of native trees across the sites of my study and used this information as a rudimentary indicator of structural diversity.

Floral diversity, plus logs and litter, are also necessary elements of successful plant cover to provide for a wide range of faunal species recolonization (Reay & Norton, 1999). As with structural diversity, floral diversity can be planned for in the planting design phase, considering pollinators and nectar feeders in the area.

Leaving trees to age, drop branches and host decay organisms also contributes to structural diversity. This part of the cycle is required to provide habitat diversity for decomposers, invertebrates and lizards – which in turn provides foraging opportunities for birds (Marzluff & Ewing, 2001; McAlpine & Drake, 2002). Fallen branches and logs provide refuge for lizards (Anderson, Bell, Chapman, & Corbett, 2012) and invertebrates (Minor & Robertson, 2006), and birds utilize fallen wood to forage for insects. The invertebrate decomposer community recycle the nutrients as they break down the wood.

Tree hollows are an important resource for cavity-nesting birds (Newton, 1994) and are primarily associated with tree age. With the exception of the NZ kingfisher, New Zealand's cavity-nesters do not excavate holes themselves, as northern hemisphere woodpeckers do, and therefore require natural processes to develop the holes for them to inhabit. Furthermore, although the kingfisher is an excavator it typically uses earth banks rather than trees. New Zealand has 23 species of cavity nesting landbirds, 10 of which are obligate cavity nesters (Rhodes, O'Donnell, & Jamieson, 2009).

In the absence of aged, damaged and decaying trees, the provision of artificial nest boxes is an option. However, this alternative measure requires careful consideration, planning and monitoring with regards to size, suitability, placement, and predators.

Recommendation Four

Continue with and establish new monitoring regimes, including repeating studies already undertaken, to construct a rich and practical data set to inform natural resource managers.

- Audit plantings using consistent methods
- Measure plant growth rate, mortality rates, species success
- Bird counts
- Lizard surveys
- Weta counts
- Invertebrate surveys
- Pest control – tracking tunnels, bait stations, traps

The establishment of baseline data and evaluation of changes to relevant ecosystems, populations or species contributes to making informed management decisions. It is important to design a monitoring plan well before beginning, with questions that are carefully considered and clearly defined. (Lindenmayer & Likens, 2009). As identified by Clarkson et al. "restoration projects require attention to environmental parameters which will ensure that the species or ecosystem under restoration flourishes" (2007, p. 27). Fundamentally, a solid understanding of the organisms contributing to the ecosystem, and the environmental factors that will influence the distribution, abundance and activity of them, is essential.

Various audits, surveys and research reports have been conducted in relation to biodiversity in Wellington, providing informative results. However, greater benefit could be gained from this material by designing long term monitoring plans to assess changes over time that will further inform of effective natural resource management.

The value of data collected could be strengthened with increased collaboration between interested parties, such as the regional council, the Department of Conservation, Forest & Bird, local community groups and universities. The majority of reporting on urban restoration projects in New Zealand is presented in case studies (Clarkson et al., 2007), or individual studies that are not deposited into a central database. To obtain a clearer understanding of the actual outcomes of small-scale projects it would be beneficial to collect specific data and analyse at a broader scale. A regional plan for data collection, and the use of consistent methods and collation of the data, would provide a deeper overall understanding of local ecosystem functioning and changes. Indeed, Thompson et al. (2001) suggest that there is a need to consider "new ways of working together, new tools, and greater access to an ever growing number of databases" (p. 23) which would be best achieved through collaboration and integration. Such systems for reporting on the 'state of the environment' are beginning to be more widely implemented for freshwater and forest ecosystems, but there is currently no standard approach for use in cities.

This study itself has provided a solid foundation for continued research into the WCC native species revegetation programme and replication of this study can easily be achieved. Mapping information has been created and stored by the use of GPS and GIS which is "a powerful tool for ecological restoration... [as it has] the ability to easily update or modify existing geospatial databases" (Michener, 1997, p. 333). It is recommended that the study be replicated every two years and this could provide rich, valuable and ongoing data into the biodiversity outcomes of this WCC investment.

My study, although comprehensive, does have limitations to acknowledge. Ideally more time and resource would have been available in order to carry out repeat samplings of pitfall traps, tree covers, and bird counts. The data would have been enhanced if they could have been collected at seasonal intervals over a twelve-month period, giving a more comprehensive picture of the invertebrates, lizards and birds in the site areas. I do suggest that the ability to repeat sample is factored in to future designs to maximize effort and outcomes.

Three existing studies were useful in providing base information from which to consider my findings. The Greater Wellington Regional Council bird count surveys (McArthur, Harvey, & Flux, 2013) that are carried out annually provide valuable information about the diversity, abundance and distribution of birds in Wellington City reserves. The Ecogecko lizard survey report (Melzer & Bell, 2014) provides a good overview of lizard distribution in Wellington City reserves and was a helpful resource. A further example of helpful data is the forest gecko research undertaken in Otari-Wilton's Bush, Wellington (Romijn, Nelson, & Monks, 2013).

TAXON-SPECIFIC RECOMMENDATIONS AND DISCUSSION

This section addresses two taxa that were included in the revegetation programme study, and whose findings warrant further exploration. This discussion complements the recommendations of the previous section of this report.

Weta

The results of my research suggest that isolation has an effect on the abundance of weta. The number of weta increased as distance from primary bush increased. Two possible reasons for this could be:

1. The success of native avian predators in close proximity to primary bush outweighs the success of mammalian predators in the matrix.
2. The abundance of mammalian predators is higher in the primary bush than it is in the matrix.

Several native birds associated with mature native forest consume weta, for example, kingfisher (*Todiramphus sanctus*), robin (*Petroica australis*), tomtit (*Petroica macrocephala*) and ruru or morepork (*Ninox novaeseelandiae*).

Tree weta are also a major component of the diet of rats, and as a result are under pressure in areas without pest control. Recent studies have documented increased weta presence following mammal eradication (Watts et al., 2011, Ruscoe et al., 2012). Thirty percent more adult weta were recorded four years after commencing the removal of mammalian pests in the Maungatautari Reserve, Waikato, New Zealand; dramatically altering the age structure of the population. However, the increase in numbers observed is also thought to be influenced by a change in behaviour as tree weta are reported to 'relax' and spend more time outside of their galleries in the absence of mammalian predators (Watts et al., 2011; Rufaut, 1995). The abundance of Auckland tree weta increased three-fold following rat baiting and control, maintaining rat populations at less than four per hectare over a three-year period (Rusoce, Sweetapple, Perry & Duncan, 2012).

Weta are an important component of native New Zealand ecosystems, assisting with pollination and seed dispersal of some plants, and nutrient recycling (Bowie & Sirvid, 2004). At the other end of the food chain, weta are an energy source for native birds, reptiles and bats (Department of Conservation, 2015a; Wilson, 1987).

Tree weta (*Hemideina* spp.) have recently been identified as a possible indicator species for monitoring the health of forest ecosystems and forest restoration programmes. The Department of Conservation (2013) published a report, *Selection of potential indicator species for measuring and reporting on trends in widespread native taxa in New Zealand*, that lists 106 taxa, selected to represent the range of habitat types, pressures, and taxonomic groups in New Zealand, suitable for monitoring programmes designed to measure biodiversity outcomes. Tree weta are listed under forest terrestrial invertebrates, along with giant landsnails (*Powelliphanta* spp.).

A pest control programme is in place in Wellington, targeting rats, mustelids and possums. Traps and bait stations are laid out through primary and secondary forested areas of the city, as well as revegetated sites. I suggest further investigation into presence-absence (and abundance) of tree weta within primary and secondary forest patches in Wellington, and the possible relationship between the results and the degree of pest control.

Lizards

The lizards found in my study were positively associated with the amount of residential landscape in the immediate vicinity. The only species found was the northern grass skink (*Oligosoma polychroma*), which is broadly distributed in the Wellington region and lives in a wide range of open areas.

The 2014 Ecogecko lizard survey (Melzer & Bell, 2014), carried out for WCC across council parks and reserves, found only four identified species of skink and gecko, as well as a number of unidentified skinks and geckos. The total number of lizards found by Melzer and Bell (2014) was 236. The four identified species found were raukawa gecko (*Woodworthia maculata*), ngahere gecko (*Mokopirirakau* aff. *granulatus* 'Southern North Island), minimac gecko (*Woodworthia* 'Marlborough mini'), and northern grass skink (*Oligosoma polychroma*). The most commonly found were the raukawa gecko and the northern grass skink. Melzer and Bell (2014) report that "generally, lizard abundance was higher in coastal parks" (p.33) and attribute this to the complex habitat of the coast that provides thermal radiation and refuge opportunities.

Of the 26 parks and reserves surveyed by Ecogecko in 2104, there were four locations in close proximity to those in my study. Within these four locations (Centennial Park, Mt Albert, Vice Regal, Seton Nossitor) a total of four lizards were found at Centennial Park only (Melzer & Bell, 2014).

The six sites in my study where lizards were found were Karori Park, Derry Hill, Lakewood Reserve, John Walker Park, Meekswood Reserve, and Helston Park. All lizards found at these parks in this study were northern grass skinks (*Oligosoma polychroma*), a total of 39 in all. None of these parks and reserves was on the list of those surveyed by Ecogecko.

The methods for lizard detection used by Ecogecko in the 2013-2014 survey included a combined pitfall trapping/onduline artifical cover objects approach, visual day searches, and visual night searches using spotlight-mounted binoculars and torches. A total of 263 person hours were invested in the day and night searches, and 154 pitfall traps installed. Pitfall traps were opened for two consecutive nights. (Melzer & Bell, 2014). Ecogecko also called for public sightings of lizards and while the response was positive most sightings reported were not within WCC parks and reserves.

I sampled for lizards by installing pitfall traps and closed-cell foam tree covers. The pitfall traps were open for two consecutive nights and checked daily. The tree covers were installed 3-6 months in advance to allow time for animals to get used to them. The covers were checked once only for sheltering lizards (and weta). No lizards were found under the tree covers, although many weta were. Common skinks were found in both the lizard and invertebrate pitfall traps, 17 in the lizard traps and 21 in the invertebrate traps. The lizard pitfalls had a piece of pear and dry leaf litter in them. The invertebrate traps had a small amount of saline solution in the bottom and a piece of scrunched up wire netting for any lizards to rest on out of the solution, until I checked the traps and released them. The invertebrate pitfall traps were checked each day alongside the lizard pitfalls for this reason.

The closed-cell foam tree covers are a novel technique tested and recommended by Bell (2009) in New Zealand. Bell found that the covers "were the most effective technique for sampling geckos where they are abundant" (2009, p.421). Further to that, sparsely populated arboreal geckos were also found using the covers and Bell concluded that this was an improvement on alternative sampling techniques.

As no arboreal geckos were found sheltering under the tree covers in my study it could be suggested that arboreal geckos are not abundant in the areas sampled. However, a recent survey conducted at Otari-Wilton's Bush (Romjin, et al., 2013) recorded 47 Southern North Island forest geckos (*Mokoprirakau* 'Southern North Island') in 30 hours of searching. Romijn et al., (2013) searched during the day and at night, and also attached radio transmitters to a small number of the geckos to determine habitat use and movement patterns. They observed that this arboreal, nocturnal gecko is just as active during the day as it is at night, and the average range of an individual per day is 10m. The majority of the geckos were found in areas with tall trees, and high up in the trees, on either trunks or branches. It was also noted that individuals displayed "strong site fidelity and may spend most of their time on a single tree" (Romijn et al., 2013, p.10). However, geckos that lived in the forest edges were more often located on the ground.

In March 2007 a BioBlitz was carried out at Otari-Wilton's Bush (Lewington & West, 2008). The idea was to locate and identify as many species of flora and fauna as possible, within a 24 hour period. The area selected for the Otari BioBlitz was the natural forest area, omitting the landscaped garden area. Four lizard species were found during the 24 hours – forest geckos (*Mokoprirakau granulatus*), northern grass skinks (*Oligosoma polychroma*), ornate skinks (*Oligosoma ornatum*), and glossy brown skinks (*Oligosoma zealandicum*) (Lewington & West, 2008).

The sample area of my study within Otari-Wilton's Bush was dominated by tall trees and was not a forest edge. As no arboreal geckos were found under the tree covers it may suggest that further sampling over a longer period of time, and including visual searches, would be useful to gather more reliable data to add to that of previous surveys.

Lizards are cryptic animals (Bell, 2009; Romijn et al., 2013) and therefore monitoring efforts require experienced searchers and repeated efforts for robust results.

CONCLUSION

This chapter has provided four recommendations into the management of the WCC native revegetation programme. These recommendations are ideally considered as a contribution to the future direction of the WCC biodiversity action plan. Planning is an important aspect of succesful revegetation projects, which continuously require attention to environmental parameters. These plans can support the ongoing development of revegetation programmes so that the species or ecosystem under restoration flourishes. This is especially pertinent in Wellington city where several revegetation programmes are in action.

The recommendations are congruent with plans already developed by WCC and the first of these recommendations is to continue native revegetation in order to increase the urban biodiversity. However, I also recommend increasing the size of planted areas as research concludes that "the smaller a... [patch] is, the greater the influence external factors are likely to have" (Saunders et al., 1991, p. 24).

The second recommendation is for the provision of 'stepping stone' patches to be considered as influential in the connectivity of the nativeness of the city. Equally encouragement and education of the strategic biodiversity messages of WCC should be an aspect of biodiversity planning for the council teams involved in the delivery of the plan.

The third recommendation considers the importance of structural diversity in the success of the native revegetation programme. Structural diversity will develop habitat for native species to flourish. This may be as pragmatic as leaving logs on sites that are being prepared for revegetation, but also includes the purposeful non-removal of exotic flora within the planting programme of native flora.

The fourth and final recommendation is to encourage the replication of research already undertaken to further examine the biodiverity of the revegetation programme, so that investment in the ecology of urban Wellington continues. The research and monitoring ideally would ideally focus upon key restoration indicators such planting audits, plant growth, pest control, and surveys of fauna.

These recommendations are supported by the findings of my study and existing literature that advocates that an aligned and co-ordinated plan for enhancing urban biodiversity is of foremost importance.

Chapter 4

CONCLUSION

Research into urban ecological restoration that collects robust monitoring information is an important activity in support of seemingly successful established restoration programmes. Scientific research that is methodologically reliable contributes to the sustained investment from territorial authorities into these programmes and may contribute to increases in investment. Researchers of ecological restoration projects become collaborators with natural resource managers and other stakeholders (Young, Peterson, & Clary, 2005). With this in mind, the specific objectives of my study were to identify any correlations between biodiversity and the age, size, or isolation of revegetated sites within the Wellington urban environment. Also tested were the effects of the landscape matrix on biodiversity within the revegetation patches.

Isolation and site age appear to have the greatest impact on the biological diversity of the Wellington City Council sites revegetated during the 18 years of the native revegetation programme.

This study has shown that the proportion of native seedlings was considerably higher in sites over 10 years of age. By this stage of maturity the vegetation had developed beyond a single layer of vegetation and many trees had reached over 3 m high. Ground ferns were only associated with sites aged over 10 years. Conversely, many of the younger sites (i.e. less than 10 years of age) had very little or no native seedlings growing in them. This is attributed to the low height of single layer of vegetation, and often the presence of long grass within the sites studied.

Isolation from mature primary bush was associated with differences in the revegetation structure, as revegetation sites located at closer proximity to primary bush supported a higher proportion of native seedlings and long grass. The closeness of mature bush to revegetaton programme sites likely improves pollination and seed transfer opportunities within these newly planted environments, which is a useful finding for future planning.

Large native birds such as kereru and kaka were only found in close proximity to mature primary bush. Native birds in general however, were widely dispersed across the urban environment. The age, size, isolation and surrounding landscape of revegetated sites did not appear to have an effect on bird richness over the urban environment studied. However, richness is only one index of biodiversity and in hindsight it may have been useful to analyse the proportion of native vs exotic birds in each site.

Northern grass skinks were associated with the proportion of residential land-use in the surrounding landscape, and in particular the outer suburb residential areas. This finding is consistent with the outcomes of a recent study on urban lizard distribution in another of New Zealand's main cities. Therefore, Northern grass skink populations are strong in young revegetated sites in outer suburbs but are likely to reduce as these sites mature and reduce the open grass habitat.

The Wellington City Council native revegetation programme provides immense benefit to the urban environment. After almost two decades of converting hillsides and small areas of grasslands into native patches, the urban landscape is now supporting widespread bird richness and weta numbers. The native plantings are developing, and for many taxa are likely to be providing small stepping-stone patches for greater connectivity across the city. The native plantings are slowly but surely greening the topography of the urban environment.

The success of the native programme is, first and foremost, due to the council-led support of, and investment into, revegetation projects. The continuation of planting native trees will in time decrease isolation as the 'stepping-stones' increase in age and diversity. The range of revegetation programmes governed by Wellington City Council, including the native revegetation programme, could be considered exemplars in terms of proactivity, however there are a few enhancements that could be considered to augment the revegetation plan.

Enhancement of the native restoration programme could include the development of a connectivity strategy in collaboration with Wellington City Council amenity operational teams. A connectivity strategy will ensure consistency is applied to biodiversity outcome guidelines across council.

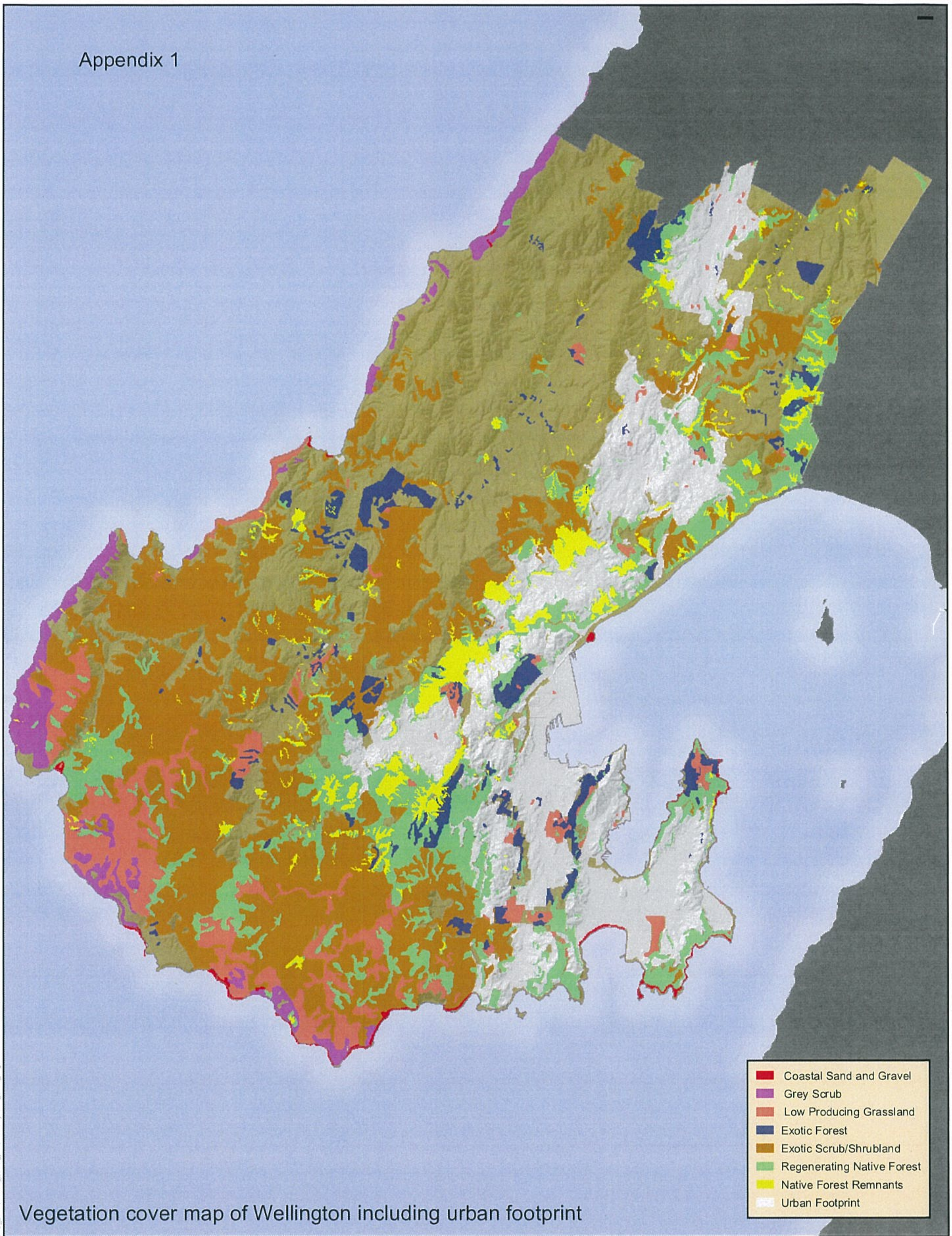
Another opportunity for advancement of the revegetation plan is the addition of structural diversity of the revegetated patches. Structural diversity in terms of enrichment planting of emergent species will increase the number of canopy layers. Structural diversity can also be achieved by leaving fallen trees in these sites. Structural diversity of plants and trees also enhances habitat by increasing foraging opportunities for native fauna, encouraging a full range of ecological functions such as pollination and seed dispersal. Over time, this will assist with supporting more diverse ecological communities.

In order to maximize biodiversity outcomes in the urban environment, it is important for Wellington City Council to invest in robust monitoring programmes. A review of current data sets and relevant research projects that have been carried out would enable the development of a long-term plan for data collection and monitoring. It is important that this process clearly identifies the areas of monitoring that would provide the most useful information for long-term adaptive management of the natural environment.

The method used my study to sample for arboreal lizards (closed cell tree covers) was not successful in this instance for lizard detection, but did prove to be an effective technique for weta-sampling. This method could be a novel and efficient way of monitoring for weta distribution and population within treed areas. Weta are considered to be a potential native forest indicator species of predator populations (Watts, Armstrong, Innes & Thornburrow, 2011; Trewick & Morgan-Richars, 2000; and therefore a simple monitoring technique such as this could provide affordable and valuable long-term information for evaluating the success of mammalian pest control programmes operating in the Wellington forests and revegetated urban landscape.

My study has constructed a foundation for future fieldwork that could easily replicate the research undertaken over 2013/14. It could be converted into a longitudinal study, which would provide rich and ongoing data over many years. Ecological restoration research that provides practical planning guidance as an outcome, is what natural resource managers and planners want from ecologists (Young, Peterson, & Clary, 2005) and is the reason I embarked upon this research.

Finally, there were emergent areas of enquiry that were not considered in detail during this study, and that would be complementary to this study and useful to investigate in the future. Of particular interest is further evaluation of the connectivity of the urban landscape matrix and a more detailed investigation into its contribution to indigenous biodiversity across the city of Wellington.



Vegetation

Map 2

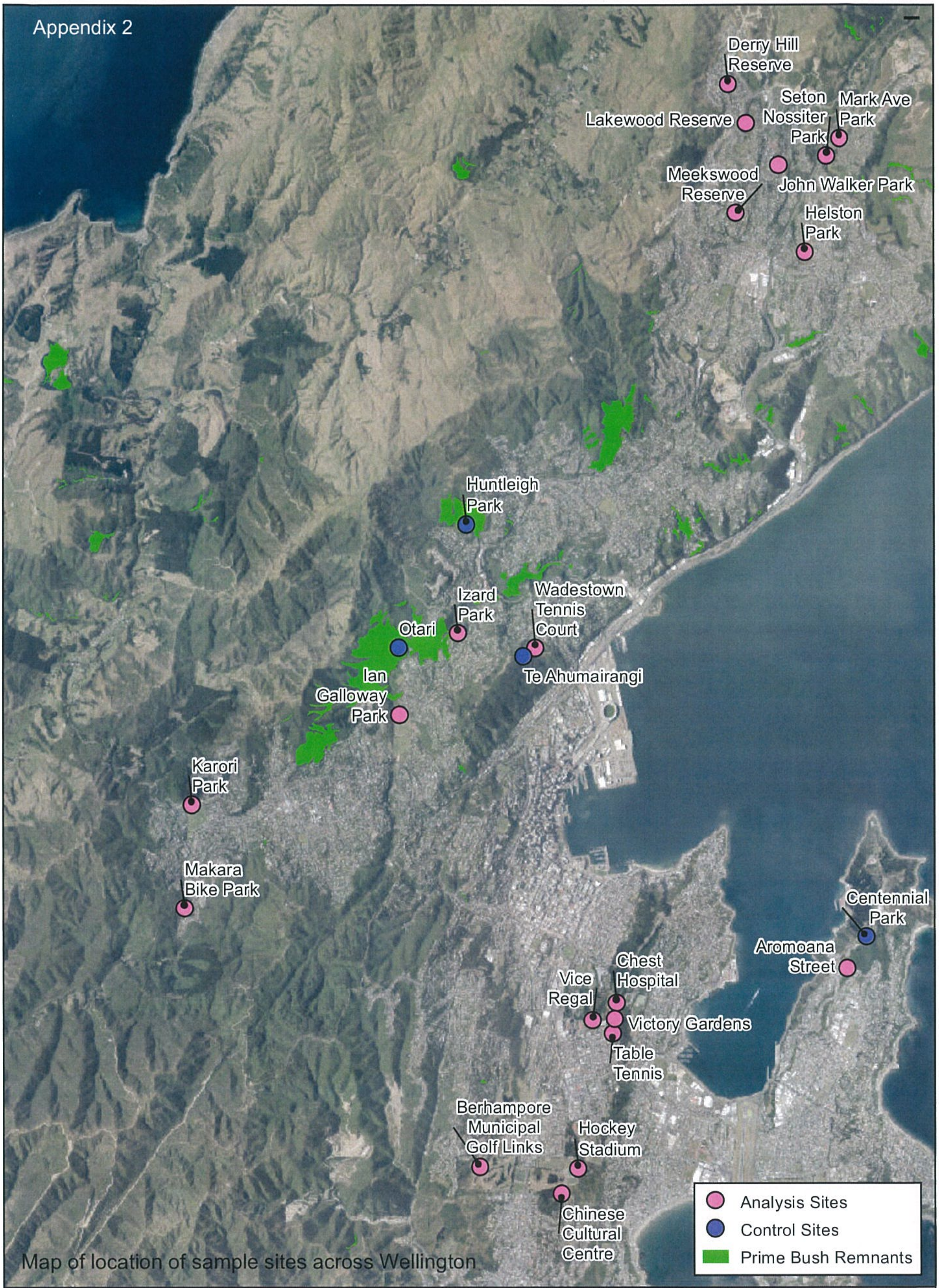
0 1,650 3,300 6,600
metres
scale 1:129,000

MAP PRODUCED BY:
Wellington City Council
101 Wakefield Street
WELLINGTON, NZ

ORIGINAL MAP SIZE: A4
AUTHOR: presto2j
DATE: 14/01/2015
REFERENCE:

Absolutely Positively
Wellington City Council
Me Heke Ki Pōneke

Filepath: Z:\Open_spaces_and_environment\Parks_Environment_Ecology\Biodiversity\BAP_2014_vegetation.mxd



Revegetation Analysis Sites

Appendix 5
T1
T2
Example of a sample site showing a revegetated site boundary, location of transects, 100m buffer zone around the site and classification of land cover uses within the buffer zone.
Analysis Site 100m Buffer
Mown
Unmown
Roadway
Residential
Bush
Derry Hill Reserve

Appendix 3

Site names, abbreviations, age & size

	Site Name	Suburb	Age	Size m2
CP	Centennial Park	Maupuia	>64	>1000000
OWB	Otari Wilton's Bush	Wilton	>64	>1000000
HUNT	Huntleigh Park	Ngaio	>64	>1000000
TA	Te Ahumairangi	Wadestown	>64	>1000000
AS	Aromoana Street	Maupuia	9	4546
CH	Chest Hospital	Mt Victoria	7	10354
VG	Victory Gardens	Mt Victoria	15	3500
TT	Table Tennis	Mt Victoria	14	4393
CCC	Chinese Cultural Centre	Mt Albert	7	3972
HS	Hockey Stadium	Mt Albert	16	3808
BGC	Berhampore Golf Course	Berhampore	7	6441
VR	Vice Regal	Mt Victoria	15	11753
MBP	Makara Bike Park	Karori	11	8019
KP	Karori Park	Karori	9	5905
IG	Ian Galloway Park	Wilton	10	6878
IP	Izard Park	Wilton	18	6432
WTC	Weld St Tennis Court	Wadestown	9	12715
DH	Derry Hill	Churton Park	9	8061
LW	Lakewood Reserve	Churton Park	11	2701
JW	John Walker Park	Churton Park	7	4118
MK	Meekswood Reserve	Johnsonville	7	7405
HP	Helston Park	Helston	13	2757
MA	Mark Ave Sportsfield	Grenada	7	3548
SN	Seton Nossitor	Grenada	18	2346

Appendix 4


T1
T2
Transect lines
Revegetation Area
Bird Count Station
Tree Cover
Invertebrate Pitfall
Lizard Pitfall
Transect Layout Example

Appendix 6

Vegetation survey & lizard tree covers timetable

Site	Vegetation survey	Lizard tree covers
JW	27/01/14	28/01/14
DH	27/01/14	28/01/14
LW	27/01/14	28/01/14
MK	27/01/14	28/01/14
BGC	3/02/14	4/02/14
CCC	3/02/14	4/02/14
HS	3/02/14	4/02/14
VR	3/02/14	4/02/14
HP	10/02/14	11/02/14
MA	10/02/14	11/02/14
SN	10/02/14	11/02/14
VG	10/02/14	11/02/14
TT	25/02/14	26/02/14
CP	25/02/14	26/02/14
AS	25/02/14	26/02/14
CH	25/02/14	26/02/14
IG	10/03/14	11/03/14
IP	10/03/14	11/03/14
MBP	10/03/14	11/03/14
KP	10/03/14	11/03/14
WTC	17/03/14	18/03/14
TA	17/03/14	18/03/14
HUNT	17/03/14	18/03/14
OWB	17/03/14	18/03/14

Appendix 6a

Lizard & invertebrate pitfall trapping timetable

Site	Opened	Checked	Checked	Final Check
JW	25/01/14	26/01/14	27/01/14	28/01/14
DH	25/01/14	26/01/14	27/01/14	28/01/14
LW	25/01/14	26/01/14	27/01/14	28/01/14
MK	25/01/14	26/01/14	27/01/14	28/01/14
BGC	1/02/14	2/02/14	3/02/14	4/02/14
CCC	1/02/14	2/02/14	3/02/14	4/02/14
HS	1/02/14	2/02/14	3/02/14	4/02/14
VR	1/02/14	2/02/14	3/02/14	4/02/14
HP	8/02/14	9/02/14	10/02/14	11/02/14
MA	8/02/14	9/02/14	10/02/14	11/02/14
SN	8/02/14	9/02/14	10/02/14	11/02/14
VG	8/02/14	9/02/14	10/02/14	11/02/14
TT	23/02/14	24/02/14	25/02/14	26/02/14
CP	23/02/14	24/02/14	25/02/14	26/02/14
AS	23/02/14	24/02/14	25/02/14	26/02/14
CH	23/02/14	24/02/14	25/02/14	26/02/14
IG	8/03/14	9/03/14	10/03/14	11/03/14
IP	8/03/14	9/03/14	10/03/14	11/03/14
MBP	8/03/14	9/03/14	10/03/14	11/03/14
KP	8/03/14	9/03/14	10/03/14	11/03/14
WTC	15/03/14	16/03/14	17/03/14	18/03/14
TA	15/03/14	16/03/14	17/03/14	18/03/14
HUNT	15/03/14	16/03/14	17/03/14	18/03/14
OWB	15/03/14	16/03/14	17/03/14	18/03/14

Appendix 6b

Bird count timetable

Site	Erin & Nicky	Site	Asher & Nicky
CP	24/09/13	VR	2/09/14
TT	24/09/13	AS	2/09/14
VG	24/09/13	CH	2/09/14
CH	24/09/13	VG	2/09/14
AS	24/09/13	TT	2/09/14
VR	24/09/13	HUNT	2/09/14
IP	2/10/13	CP	2/09/14
IG	2/10/13	DH	3/09/14
HUNT	2/10/13	LK	3/09/14
OWB	2/10/13	JW	3/09/14
TA	16/10/13	MK	3/09/14
KP	16/10/13	HP	3/09/14
WTC	16/10/13	MA	11/09/14
BGC	16/10/13	SN	11/09/14
HS	16/10/13	WTC	11/09/14
CCC	16/10/13	OWB	11/09/14
MBP	16/10/13	TA	11/09/14
SN	17/10/13	MBP	12/09/14
MA	17/10/13	KP	12/09/14
HP	17/10/13	IG	12/09/14
MK	17/10/13	IP	12/09/14
JW	17/10/13	CCC	18/09/14
LK	17/10/13	HS	18/09/14
DH	17/10/13	BGC	18/09/14

Appendix 7

Bird & invertebrate abbreviations

Birds abbreviation	Common name
bell	Bellbird
bb	Blackbird
chaf	Chaffinch
dun	Dunnock
rose	Eastern rosella
fan	Fantail
gold	Goldfinch
grnf	Greenfinch
wrbl	Grey warbler
hwk	Harrier hawk
kaka	Kaka
kere	Kereru
kngf	Kingfisher
mag	Magpie
mall	Mallard duck
pari	Paradise duck
pign	Pigeon
roos	Rooster
gull	Seagull
slve	Silvereye
song	Song thrush
sprw	Sparrow
strl	Starling
swlw	Swallow
tui	Tui
ylw	Yellowhammer

Inverts abbreviation	Genus
Amph	Amphipoda
Aran	Araneae
Acar	Acarina
Blat	Blattodea
Colem	Colembola
Coleo	Coleoptera
Chil	Chilopoda
Dipl	Diplopoda
Dipt	Diptera
Hyme	Hymenoptera
Hemi	Hemiptera
Isopo	Isopoda
Lepi	Lepidoptera
Orth	Orthoptera
Phal	Phalangida
Plec	Plectoptera
Pulm	Pulmonata
Pseu	Pseudoscorpian
Arch	Archaeopgnatha
Tric	Tricoptera
Isopt	Isoptera

Appendix 8

Age, size, isolation, matrix residential land cover m2
DMJRPRM = distance to primary bush measured in metres

SITE	AGE	SIZE m2	DMJRPRM	RESIDENTIAL M2 (MATRIX)
Centennial Park	>64	>1000000	6993	0
Otari-Wilton's Bush	>63	>1000000	0	0
Huntleigh Park	>64	>1000000	0	0
Te Ahumairangi	>64	>1000000	737	0
Aromoana St	9	4546	7077	23959.7
Chest Hospital	7	10354	4663	0
Victory Gardens	15	3500	4809	6280.49
Table Tennis	14	4393	4888	17704.27
Chinese Cultural	7	3972	6110	0
Hockey Stadium	16	3808	5943	1223.83
Golf Course	7	6441	5360	7371.59
Vice Regal	15	11753	4572	26614.93
Makara Bike Park	11	8019	2084	27206.13
Karori Park	9	5905	1260	7185.55
Ian Galloway	10	6878	823	942.75
Izard Park	18	6432	0	18857.93
Wadestown Tennis Court	9	12715	505	18997.79
Derry Hill	9	8061	3801	86306.9
Lakewood	11	2701	3410	20787.75
John Walker	7	4118	3159	24842.35
Meekswood	7	7405	2338	61280.87
Helston Park	13	2757	2657	17019.16
Mark Ave	7	3548	2424	23058.72
Seton Nossitor	18	2346	2637	2877.21

Appendix 9

Percent cover of native vegetation, 2 transects/site.
NT=native trees. NS=native seedlings. LL=leaf litter. LG=long grass.

SITE	T/SECT	NT 1-3m	NT 3-6m	NT 6m+	NS 0-1m	FERNS	LL	LG	Exotics 20m+
CP	1	0	85	0	70	40	100	0	0
CP	2	15	100	0	15	15	100	0	0
OWB	1	10	100	100	45	30	100	0	0
OWB	2	20	80	100	85	30	100	0	0
HUNT	1	50	100	100	90	30	100	0	0
HUNT	2	55	100	100	90	20	100	0	0
TA	1	15	65	0	45	20	100	0	100
TA	2	10	30	0	45	5	100	0	100
AS	1	65	0	0	5	0	0	85	0
AS	2	55	0	0	5	0	0	90	0
CH	1	35	0	0	0	0	0	100	0
CH	2	90	0	0	0	0	80	20	0
VG	1	0	100	0	5	0	100	0	0
VG	2	0	90	0	10	0	70	30	0
TT	1	0	90	0	5	0	50	50	0
TT	2	30	50	0	5	0	50	50	0
CCC	1	30	40	0	5	0	50	20	0
CCC	2	50	20	0	0	0	50	20	0
HS	1	0	95	0	5	0	95	5	0
HS	2	0	95	0	5	0	100	0	0
BGC	1	50	0	0	0	0	0	50	0
BGC	2	50	0	0	5	0	0	50	0
VR	1	0	100	0	5	0	90	10	0
VR	2	50	50	0	5	0	80	20	0
MBP	1	50	80	0	30	5	95	5	0
MBP	2	25	100	0	20	5	100	0	0
KP	1	55	30	0	10	0	50	5	0
KP	2	35	15	0	5	0	0	100	0
IG	1	40	40	0	10	0	25	75	0
IG	2	30	20	0	5	0	10	90	0
IP	1	15	95	0	20	5	100	5	0
IP	2	10	100	0	10	5	100	0	0
WTC	1	40	5	0	5	5	10	50	0
WTC	2	10	80	0	15	5	85	15	0
DH	1	50	20	0	5	0	15	85	0
DH	2	70	10	0	0	0	80	20	0
LW	1	10	80	0	10	5	70	30	0
LW	2	20	90	0	15	0	100	0	0
JW	1	25	50	0	5	0	50	50	0
JW	2	10	20	0	0	0	20	80	0
MK	1	75	0	0	5	0	70	30	0
MK	2	75	0	0	0	0	20	80	0
HP	1	0	60	0	5	0	75	25	0
HP	2	20	60	0	0	0	90	10	0
MA	1	20	80	0	5	0	90	10	0
MA	2	20	80	0	5	0	80	20	0
SN	1	80	100	0	80	25	100	5	0
SN	2	25	75	0	20	10	100	5	0

Appendix 10

Number of birds, lizards, weta & invertebrates per site (individuals)

SITE	Native Birds	Introduced Birds	Lizards	Tree Weta	Cave Weta	Ground Weta	Inverts Totals
Centennial Park	15	28	0	4	0	0	59
Otari-Wilton's Bush	32	6	0	1	1	0	60
Huntleigh Park	23	7	0	5	6	0	73
Te Ahumairangi	6	39	0	5	2	0	275
Aromoana St	30	43	0	24	0	0	46
Chest Hospital	20	35	0	0	1	0	110
Victory Gardens	32	29	0	20	0	0	712
Table Tennis	13	36	0	3	0	0	96
Chinese Cultural	20	34	0	3	0	0	127
Hockey Stadium	9	35	0	22	0	0	58
Golf Course	14	39	0	0	0	0	171
Vice Regal	35	52	0	16	0	0	100
Makara Bike Park	33	35	0	13	0	0	46
Karori Park	15	54	1	1	1	0	174
Ian Galloway	11	29	0	7	0	0	175
Izard Park	16	29	0	0	0	0	310
Wadestown Tennis Court	14	39	0	3	1	1	1657
Derry Hill	7	46	32	0	1	4	383
Lakewood	9	36	3	0	1	0	258
John Walker	17	46	1	2	0	1	258
Meekswood	17	31	1	3	0	0	313
Helston Park	9	51	1	0	0	0	463
Mark Ave	11	41	0	12	0	0	491
Seton Nossitor	8	37	0	7	0	0	921

Appendix 11

VICTORIA UNIVERSITY OF WELLINGTON

ANIMAL ETHICS COMMITTEE

December 18, 2013

Memorandum to: Dr Stephen Hartley
School of Biological Sciences

Cc Nicky OliverSmith

Re: Evaluation of AEC applications:

2013R19 An assessment of the biodiversity outcomes of the Wellington City Council native revegetation programme

Your application to use live animals in your research at Victoria University of Wellington was approved by the AEC at its meeting on Thursday December 12, 2013. Approval is given as requested for one-and-a half years until June 1, 2015.

We thank you for your cooperation in helping the Committee ensure appropriate animal welfare standards and procedures are in place at Victoria University.

John H Miller

John Miller
AEC Executive Officer
School of Biological Sciences

Appendix 12

Department of Conservation
Te Papa Atawbai

File Ref: NHS-12-01-03

18 July 2013

Wellington City Council
P.O. Box 2199
Wellington

For the attention of: Nicky Oliversmith

Re: WILDLIFE ACT AUTHORITY APPLICATION 36882-FAU APPROVAL

I am pleased to advise you that your application for a Wildlife Act Authority has been approved and I am now able to offer you an authority outlining the terms and conditions of this approval. Please find the authority enclosed.

This document contains all the terms and conditions of your authorisation to operate on non public conservation land and represents the formal approval from the Department for Wellington City Council to carry out the activity.

Please read the terms carefully so that you clearly understand your obligations. It is advised that you seek legal advice.

Payment of Processing Fees

Thank you, your payment of $110.00 + GST covered the cost of processing your application.

Rob Stone
Area Manager
Kapiti Wellington Area Office

Department of Conservation
Te Papa Atawhai

Wildlife Act Authority for wildlife not located on public conservation land

National Permit Number: 36882-FAU
File Number: NHS-12-01- 03

THIS AUTHORITY is made this 18th day of July 2013

PARTIES:

The Director General of Conservation (the Grantor)

WELLINGTON CITY COUNCIL (the Authority Holder)

BACKGROUND

A. The Director General of Conservation is empowered to issue authorisations under the Wildlife Act 1953.

B. The Authority Holder wishes to exercise the authorisation on the Land subject to the terms and conditions of this Authority.

OPERATIVE PARTS

In exercise of the Grantor's powers under the Conservation legislation the Grantor **AUTHORISES** the Authority Holder under Section 53 of the Wildlife Act 1953 subject to the terms and conditions contained in this Authority and its Schedules.

SIGNED on behalf of the Grantor by

Rob Stone
Area Manager
Kapiti Wellington Area Office

acting under delegated authority in the presence of:

Witness Signature: ____________

Witness Name: Daniel Palmer

Witness Occupation: Biodiversity Ranger

Witness Address: 10 Hume St, Lower Hutt

A copy of the Instrument of Delegation may be inspected at the Director-General's office at 18-22 Manners Street, Wellington

SCHEDULE 1

1.	***Authorised activity (including approved quantities of wildlife and collection methods).*** *(clause 2)*	**(1) Authorised Activity** – To handle, capture and release the following protected wildlife ("The Protected Wildlife") (a) *Naultinus punctatus* (b) *Mokopirirakau* (c) *Hoplodactylus maculates* (d) *Oligosoma polychrome* (e) *Oligosoma ornatum* (f) *Oligosoma aeneum* (g) *Oligosoma zealandicum* **(2) Quantity** – As many as can be located **(3) The Collection Methods** – By using onduline retreats, foam tree covers, pitfall traps
2.	***The Location*** *(clause 2)*	(a) Otari Wilton's Bush (b) Seton Nossiter Park (c) Trellisick Park
3.	***Authorised Personnel*** *(clause 3)*	Nicky Oliversmith
4.	***Term*** *(clause 4)*	Commencing on and including 1 November 2013 and ending on and including 30 April 2014
5.	***Authority Holder's address for notices*** *(clause 8)*	*The Authority Holders address in New Zealand is:* 101 Wakefield Street Wellington Phone: (04) 389-4289 Email: nicky.oliversmith@wcc.govt.nz

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