

<http://researchcommons.waikato.ac.nz/>

Research Commons at the University of Waikato

Copyright Statement:

The digital copy of this thesis is protected by the Copyright Act 1994 (New Zealand).

The thesis may be consulted by you, provided you comply with the provisions of the Act and the following conditions of use:

- Any use you make of these documents or images must be for research or private study purposes only, and you may not make them available to any other person.
- Authors control the copyright of their thesis. You will recognise the author's right to be identified as the author of the thesis, and due acknowledgement will be made to the author where appropriate.
- You will obtain the author's permission before publishing any material from the thesis.

The capacity of restored urban forests to support native birds: Ecological or social restoration?

A thesis

submitted in fulfilment

of the requirements for the degree

of

Doctor of Philosophy in Ecology and Social Sciences

at

The University of Waikato

by

ELIZABETH ELLIOT NOE



THE UNIVERSITY OF
WAIKATO
Te Whare Wānanga o Waikato

2020



IN LOVING MEMORY OF
CAMERON KENWORTHY ELLIOT-HOGG

For You have made us for Yourself, and our hearts are restless until they rest in You.
- Saint Augustine of Hippo

Abstract

Urbanisation, which drastically and often irreversibly alters original habitat, is recognized as one of the leading causes of species extinctions. Biological invasions are viewed as the second most important cause of global biodiversity loss, and invasive species have been a major cause of island fauna extinctions. New Zealand cities are characterised by low levels of remnant indigenous vegetation and a unique, understudied assemblage of introduced mammals, resulting in a depauperate native forest bird community. The loss of native flora and fauna from urban areas is thought to lead to the ‘extinction of experience’, or loss of opportunities for city residents to interact with nature on a daily basis, which would allow them to develop a personal, emotional bond with the natural environment. This situation is believed to result in a vicious cycle of ever decreasing levels of native biodiversity and the continued disintegration of the human relationship with the natural world.

Habitat restoration in cities could disrupt this vicious cycle by recreating native ecosystems where the majority of the world’s population now lives. To date, urban restoration research both internationally and in New Zealand has focused on plants. It is often assumed that restoring native vegetation will result in the recovery of native fauna and the ecosystem services they provide, such as seed dispersal and pollination. This assumption, however, has rarely been tested empirically. Restoration success should be measured on the scale of human experience, connection to nature and wellbeing, for it is on this level that urban restoration efforts have the most to contribute to global conservation progress. People create cities and are the source of both the challenges and solutions presented by urban ecosystems. Yet, little is known about the relationship between city residents and urban biodiversity.

Recognizing that cities represent complex, interconnected networks of human actions and their effect on ecosystems and organisms, this thesis integrates the social and natural sciences in a multidisciplinary framework, with the overarching aim of investigating the capacity of urban forest restoration to break the vicious cycle of extinction of experience by providing habitat for native forest birds and reconnecting urban residents with nature in New Zealand cities.

I used a phenomenological methodology to explore the experiences of people living near urban greenspaces. Using qualitative interviews, I explored Hamilton City residents' perceptions and experiences of urban nature in frequently used parks and gardens. This research identifies a gap between what residents claim to value and their behaviour and actions affecting native biodiversity. Given this value-action gap, my interviews suggest we cannot currently rely on private gardens to provide adequate resources for native birds. This research highlights, however, the unique contribution that native bush parks make to human wellbeing and sense of place. Ecological restoration in cities therefore has great potential to enrich human lives.

To evaluate restoration success by determining how species richness, abundance, diversity and community composition of native forest birds vary along a restoration chronosequence, I monitored birds and mammals at 43 sites in two cities in New Zealand's North Island – Hamilton and New Plymouth. Sites represented three types of urban forest: unrestored ($n = 6$), restored ($n = 25$) remnant ($n = 6$), and rural forest remnants nearest to each city ($n = 6$). Restored sites formed an age gradient of 1 to 72 years since initial planting. We used NMDS for a visual representation, and PERMANOVA to test for differences in bird community composition across restored and reference sites. Structural equation modelling was used to investigate the relative importance of vegetation characteristics, landscape pattern and introduced mammals to native forest bird communities. Results demonstrate that urban forest restoration increases native bird species richness and diversity over time, and transforms the bird community from being dominated by introduced granivores and generalists, to one dominated by native forest birds. This research indicates, moreover, that lack of habitat is an important factor limiting native forest birds in New Zealand's North Island cities. Current restoration efforts were found to benefit widespread, common native forest birds, but greater investments in predator control or eradication will be needed to re-create forests capable of supporting bird species of conservation concern.

These same sites were used to investigate how introduced mammal communities change over time since initial restoration planting and what consequences this has for restoration success. I again used structural equation modelling to determine the relative importance of vegetation structure, landscape composition and interspecific interactions on introduced mammal community structure, as well as the impact of

mammals on native forest regeneration. This research demonstrated that urban forest restoration provides good habitat for rats, possums and mice within 9 years of initial restoration planting. Rats were also found to decrease native sapling species richness and density, thereby changing the trajectory of forest recovery.

This thesis advances theoretical and practical knowledge of the capacity of urban forest restoration to address the interconnected problem of native biodiversity loss in cities and the disintegration of human relationships with the natural world. Furthermore, it highlights the necessity of adopting a multidisciplinary approach in order to find ways of reconciling the needs of both people and native species in cities. Given that the health of people and the environment are inextricably linked, crossing the disciplinary boundary between the social and natural sciences provides one avenue for restoring human connections to the natural world.

Acknowledgements

Thesis survival was made possible by my family and friends who supported, encouraged and put up with me throughout my studies. I'm grateful to my husband and Tristan for being the family I want to come home to after a day of suffering (i.e., thesis writing). Thanks to my dad, who has read more versions of my thesis than he cares to remember and who wished to be acknowledged for his 'vast contribution to the cause of science and environmental nirvana'. My husband, dad, mother, brother and sister braved steep slopes, dense supplejack and muddy stream banks to assist with pest monitoring. I am also grateful for the time my sister put into improving the visual effect of the posters and presentations I've given over the years.

I am grateful to Bruce Clarkson for taking me on in the first place, despite my interest in birds and people (as opposed to plants...) and for his support over the years. I also thank John Innes for the many hours he was willing to volunteer to talk about rats, cats and birds (among other ecological topics). His wealth of knowledge of New Zealand's wildlife and his grounded approach to research were invaluable in shaping this thesis. I also benefitted from the many stacks of books, articles, theses and reports he managed to extract from his office. Thanks to Otilie Stolte for her courage in taking on an ecology student to do social science, and for her brilliant ability to step back and see the bigger picture, helping me incorporate it into my thesis. I am also grateful for her kindness and words of encouragement over the past 3½ years. Thanks to Chaitanya Joshi for guiding me through the perilous fog that is statistical analysis.

Toni Cornes provided invaluable technical support, plant identification and beautiful map making skills. Thanks to University of Waikato librarian Cheryl Ward for her endless patience in formatting this document. I'm grateful to Andrew Barnes and Kiri Wallace for their assistance with structural equation modelling and willingness to share R code. Many volunteers assisted with fieldwork over the years, including Stevie Noe, Annette Elliot, Cameron Elliot-Hogg, Mary Elliot-Hogg, Malcolm Elliot-Hogg, Toni Cornes, Lara Reynolds, Steven Newcombe, Melissa Collins, Lynda McCann, Kiri Wallace, Kévin Maurin, Maggie Shepherd, Jonás Agulló Brotons, Renee Denby, Sarah Busbridge, Genevieve Palmer, Catherine Zhang, Kaity Handley, Daniel Reason and Mitchell Rawlings.

Thanks to friends for lunches, teas, support and listening to grumblings – Stevie Noe, Bridgette Farnworth, Kiri Wallace, Catherine Kirby, Toni Cornes, Rachel Nepia, Hanna Spaak, Rebecca Bylsma, Kévin Maurin, Lynda McCann, Sarah Busbridge, Monique Hall and Andrew Barnes.

I am grateful to the Hamilton City Council, New Plymouth District Council and the Department of Conservation for granting permission to conduct research in Hamilton, New Plymouth, the Hakarimata and Kaitake Ranges. Thanks to P. Morris and A. Edgar for allowing access to their properties to collect data.

This research was funded by a Waikato University Doctoral Scholarship, Birds New Zealand Research Funds, a David Medway Scholarship, Forest and Bird Valder Conservation Grants, a Puke Ariki George Mason Charitable Trust Scholarship, a Trust Waikato Student Community Grant, a University of Waikato George Mason Charitable Trust Scholarship, a Faculty of Science and Engineering Student Trust fund and a Ministry of Business, Innovation and Employment grant (UOWX1601). Attendance at conferences to present my research was further funded by a Capstone Editing Travel Grant and a Claude McCarthy Fellowship.

Table of Contents

Abstract	i
Acknowledgements	v
Table of Contents	vii
List of Figures	ix
List of Tables.....	xiii
Chapter 1: General Introduction.....	1
1.1 Background: Unique opportunities and challenges for biodiversity conservation and restoration in New Zealand cities.....	1
1.2 Research aims, questions and thesis outline.....	15
Chapter 2: A qualitative exploration of personal experiences of urban nature in a New Zealand city	19
2.1 Introduction	19
2.2 Methods	31
2.2.1 Participants.....	31
2.2.2 Data collection and analysis.....	31
2.3 Results and Discussion	34
2.4 Summary and Conclusions	51
Chapter 3: The relative importance of vegetation structure, landscape pattern and invasive mammals to bird communities along an urban forest restoration chronosequence	57
3.1 Introduction	57
3.2 Methods	62
3.2.1 Study site.....	62
3.2.2 Bird surveys	65
3.2.3 Vegetation measurement.....	66
3.2.4 Predator monitoring	67
3.2.5 Landscape measures.....	69
3.2.6 Statistical analysis	69
3.3 Results	74
3.4 Discussion	96
3.5 Appendices	103
Chapter 4: Changes in introduced mammal communities along an urban forest restoration chronosequence result in unforeseen outcomes for restoration	111

4.1	Introduction	111
4.2	Methods	113
4.2.1	Study site.....	113
4.2.2	Predator monitoring	116
4.2.3	Predator control.....	117
4.2.4	Vegetation measurement.....	119
4.2.5	Landscape measures.....	119
4.2.6	Statistical analysis	120
4.3	Results	124
4.4	Discussion	144
4.5	Appendices	150
Chapter 5: General Discussion.....		157
5.1	Key findings	157
5.2	Recommendations for management	158
5.3	Recommendations for further research	161
References		165

List of Figures

Figure 1.1: A diagrammatic representation of the thesis structure and how the various components are interrelated.	16
Figure 3.1: Location of the 25 study sites in Hamilton and the Hakarimata Range, which included 16 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in Hamilton, and 3 in the Hakarimata Range.	62
Figure 3.2: Location of the 18 study sites in New Plymouth and the Kaitake Range, which included 9 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in New Plymouth, and 3 in the Kaitake Range.	63
Figure 3.3: Diagram of transect sampling method used, adapted from Wild about New Zealand. Birds detected within the 20 m belt were included in measures of abundance, while birds detected both within the belt and beyond were included in the calculation of species richness.	65
Figure 3.4: Hypothesized a priori structural equation model (SEM) path diagram.	73
Figure 3.5: Mean native bird (a) species richness, (b) abundance (individuals per 0.4 ha) and (c) taxonomic diversity (Shannon index) for spring counts in restored forests over time (middle section of each bivariate plot, n=25), compared with unrestored, urban and rural remnants (boxplots)	75
Figure 3.6: Mean native bird (a) species richness, (b) abundance (individuals per 0.4 ha) and (c) taxonomic diversity (Shannon index) for winter counts in restored forests over time (middle section of each bivariate plot, n=25), compared with unrestored, urban and rural remnants (boxplots).	76
Figure 3.7: Mean abundance of (a) fantails, (b) tūī, (c) grey warblers and (d) silvereyes for spring counts in restored forests over time (middle section of each bivariate plot, n=25), compared with unrestored, urban and rural remnants (boxplots).	78
Figure 3.8: Mean abundance of (a) fantails, (b) tūī, (c) grey warblers and (d) silvereyes for winter counts in restored forests over time (middle section of each bivariate plot, n=25), compared with unrestored, urban and rural remnants (boxplots).	79
Figure 3.9: Average number of native birds observed per site type for spring counts. Restored sites were divided into young (1-10, n = 8), medium (13-27, n = 10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.	81
Figure 3.10: Average number of native birds observed per site type for winter counts. Restored sites were divided into young (1-10, n = 8), medium (13-27, n=10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.	82

Figure 3.11: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean spring bird abundance data (stress = 0.16).	84
Figure 3.12: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean spring bird abundance data (stress = 0.16).	84
Figure 3.13: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean winter bird abundance data (stress = 0.18).	85
Figure 3.14: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean winter bird abundance data (stress = 0.18).	85
Figure 3.15: Mean spring native bird species richness and plant community attributes in all study sites.	88
Figure 3.16: Mean spring native bird abundance (no. of individuals per 0.4 ha) and plant community attributes in all study sites.	89
Figure 3.17: Mean spring native bird species richness and landscape variables for all urban sites (n = 37).	90
Figure 3.18: Mean spring native bird abundance (no. of individuals per 0.4 ha) and landscape variables for all urban sites (n = 37).	90
Figure 3.19: Structural equation model (SEM) path diagram, illustrating drivers of native bird species richness.	92
Figure 3.20: Structural equation model (SEM) path diagram, illustrating drivers of native bird abundance.	94
Figure 4.1: Location of the 25 study sites in Hamilton and the Hakarimata Range, which included 16 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in Hamilton, and 3 in the Hakarimata Range.	114
Figure 4.2: Location of the 18 study sites in New Plymouth and the Kaitake Range, which included 9 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in New Plymouth, and 3 in the Kaitake Range.	115
Figure 4.3. Hypothesized a priori structural equation model (SEM) path diagram.	123
Figure 4.4. Number of sites (n = 43) at which introduced mammals were detected using chew cards and camera traps over three seasons. The data for winter 2018 is not displayed due to camera malfunction.	125
Figure 4.5. Introduced mammals detected only once throughout the study: (A) stoat, (B) ferret, (C) hare and (D) weasel.	126
Figure 4.6. Mean abundance of (a) rats, (b) possums, (c) mice, (d) hedgehogs, (e) rabbits and (f) cats for spring data in restored forests over time (middle section of each bivariate plot, n=25), compared with unrestored, urban and rural remnants (boxplots).	127

Figure 4.7. Average number of introduced mammals detected using camera traps per site type for spring data. Restored sites were divided into young (1-10, n = 8), medium (13-27, n = 10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.....	129
Figure 4.8. Number of introduced mammals detected using camera traps per site type for winter 2017 data. Restored sites were divided into young (1-10, n = 8), medium (13-27, n = 10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.....	130
Figure 4.9. The four introduced mammals with the most camera triggers in urban areas: (A) cat, (B) dog, (c) brushtail possums and (D) hedgehogs. ...	131
Figure 4.10. The four introduced mammals with the most camera triggers in rural areas: (A) ship rat, (B) brushtail possum, (C) house mouse and (D) hedgehog	132
Figure 4.11. Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of 42 study sites (one was removed due to zeros) and spring mammal camera trigger data (stress = 0.19).....	133
Figure 4.12. Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of 40 study sites (three were removed due to zeros) and winter mammal camera trigger data (stress = 0.13).	134
Figure 4.13. Spring means of rat indices of relative abundance and plant community attributes in all study sites (n = 43).....	136
Figure 4.14. Spring means of possum indices of relative abundance and plant community attributes in all study sites (n = 43).....	137
Figure 4.15. Spring means of mouse indices of relative abundance and plant community attributes in all study sites (n = 43).....	138
Figure 4.16. Relationships between spring means of mouse, rat, rabbit and cat indices of relative abundance in all study sites (n = 43).	139
Figure 4.17. Relationships between winter means of mouse, rat, rabbit and cat indices of relative abundance in all study sites (n = 43).	139
Figure 4.18. Structural equation model (SEM) path diagram, illustrating drivers of native sapling species richness.	140
Figure 4.19. Structural equation model (SEM) path diagram, illustrating drivers of native sapling density.....	142

List of Tables

Table 3.1: Abbreviations used for bird species in NMDS ordination and corresponding full species names.....	85
Table 3.2: Summary of results from the structural equation model on native bird species richness. Asterisks denote statistically significant ($p < 0.05$) effect paths.	93
Table 3.3: Summary of results from the structural equation model on native bird abundance. Asterisks denote statistically significant ($p < 0.05$) effect paths.	95
Table 4.1: Summary of vegetation and landscape variables fitted to the spring data NMDS ordination with fitting statistics (squared correlation coefficient and corresponding p-value).....	133
Table 4.2: Summary of vegetation and landscape variables fitted to the winter data NMDS ordination with fitting statistics (squared correlation coefficient and corresponding p-value).....	135
Table 4.3. Summary of results from the structural equation model on native sapling species richness. Asterisks denote statistically significant ($p < 0.05$) effect paths.	141
Table 4.4. Summary of results from the structural equation model on native sapling density. Asterisks denote statistically significant ($p < 0.05$) effect paths.	142

Chapter 1: General Introduction

1.1 Background: Unique opportunities and challenges for biodiversity conservation and restoration in New Zealand cities

Cities: Where the people are

Today, over half the world's population lives in urban areas (United Nations, 2018) and the proportion of urbanites increases in certain developed countries such as New Zealand, where it exceeds 86% (The World Bank, 2018). As the human population shifts to cities, so do potential sources of conservation action in the form of votes, finances, volunteers and future conservation leaders (Dunn *et al.*, 2006). Scholars have argued that direct, personal experiences of the natural world are necessary to foster a willingness to care for and about nature (Chawla, 1998; Pyle, 2003; Miller, 2005), and these experiences further promote human health and wellbeing (Curtin, 2009; Keniger *et al.*, 2013). As McDonald (2016, p. 1) eloquently argued, 'if the health of nature depends upon cities, we are increasingly realizing that the health of our cities depends upon nature'. As the majority of people move to cities, it follows that most human experience of nature will take place in an urban setting.

The potential of cities to conserve biodiversity

While cities are designed and built to meet the needs and preferences of people, they can provide coincidental habitat for a variety of plants and animals (Melles, 2005). There is a growing recognition of the potential contribution cities can make to the conservation and restoration of native biodiversity (Savard *et al.*, 2000; Clarkson *et al.*, 2007a; Palmer *et al.*, 2008). Urban areas are often built in productive lowlands and riparian systems (Melles *et al.*, 2003), which prior to development would have supported a rich diversity of native species (Aronson *et al.*, 2014). Australian cities have been found to contain disproportionately more threatened species than non-urban areas (Ives *et al.*, 2016; Threlfall *et al.*, 2016), and small public gardens in Paris

supported a significant diversity of common species (Shwartz *et al.*, 2013). The Australian study emphasized, however, that this trend was more pronounced for plants than for animals (Ives *et al.*, 2016). Schwartz and colleagues (2002) similarly found that many rare and endangered plant species persisted in US cities. Shwartz and colleagues (2014) conducted a systematic, multi-disciplinary review of the value of urban biodiversity to city residents and global conservation. They concluded that the contribution of cities to global biodiversity conservation efforts remains limited, with very few studies finding that urban areas can support viable populations of unique, rare or endangered species. The studies that did support this claim were mainly investigating plants and invertebrates. Consequently, we still lack sufficient evidence that urban greenspaces can support viable populations of anything other than locally common animal species (Shwartz *et al.*, 2014a), and whether cities can contribute to global conservation efforts.

The unique challenges of urbanisation

Urbanisation is considered one of the leading causes of species extinctions, as it drastically and often irreversibly alters original habitat, stripping large areas of land of native vegetation (McKinney, 2006). Cities present unique challenges to native plants and animals, as urban greenspaces tend to be small, isolated, with higher levels of introduced and invasive species and decreased vegetative complexity (Marzluff & Ewing, 2001; Kang *et al.*, 2015; Belder *et al.*, 2018). Furthermore, urbanization often results in biotic homogenization, or replacement of unique endemics with globally widespread species, which are capable of surviving and even thriving in human-altered landscapes (McKinney & Lockwood, 1999; McKinney, 2006). These urban adapters and exploiters include a suite of mammalian omnivores and predators, such as the ship rat (*Rattus rattus*), domestic cat (*Felis catus*) or house mouse (*Mus musculus*), which can outcompete or prey on native species.

Urban bird communities are characterized by low species richness, with high densities of only a few generalist, behaviourally-flexible and often exotic birds (Chace & Walsh, 2006; Daniels & Kirkpatrick, 2006). For the avifauna of New Zealand cities, this has meant replacing the endemic tūī (*Prothemadera novaeseelandiae*), bellbird (*Anthornis*

melanura) or kererū (*Hemiphaga novaeseelandiae*) with the exotic and globally widespread house sparrow (*Passer domesticus*), rock pigeon (*Columba livia*) or European starling (*Sturnus vulgaris*). While Aronson and colleagues (2014) concluded that urban biotas continue to reflect their regional biogeographic species pool, with cities containing a median of 3% exotic birds, New Zealand is an exception to this trend with a study in Dunedin finding that 44% of all bird species counted were exotic (van Heezik *et al.*, 2008b). Human settlement of New Zealand, and resulting habitat degradation and introduction of mammalian predators, has posed the greatest threat to deep endemic forest birds¹, which disappeared from settled and deforested landscapes by the 1970s (Walker & Monks, 2017). Thus, most New Zealand cities today have a highly depauperate native forest bird community (Miskelly *et al.*, 2005; van Heezik *et al.*, 2008b).

The vicious cycle of biodiversity loss and the extinction of experience

Researchers have suggested that the loss of native biodiversity in cities leads to what Robert Pyle termed the extinction of experience, or the loss of direct, personal, daily contact between people and native nature (Pyle, 1993; Miller, 2005, 2006). Diminished opportunities to interact with nature result in people who care little about the natural world, and are therefore unmotivated to protect, conserve and enhance native landscapes and species either within the city or beyond. Thus, the vicious cycle continues, with each new generation growing up in cities with ever decreasing levels of native biodiversity (for a more in-depth discussion, see Chapter 2). One strategy for disrupting this cycle is habitat restoration in cities, which has the potential to reverse biotic homogenization and reconnect people with nature.

The potential benefits of restoration to birds and people

Restoration ecology arose out of the conviction that protected remnants alone will not be enough to reverse the impact of habitat loss and conserve biodiversity into the future

¹ Deep endemics are taxa that are endemic to New Zealand at the level of subfamily, family or order (Walker *et al.*, 2017)

(Hobbs & Norton, 1996; Rosenzweig, 2003). Little native forest cover remains in New Zealand cities (0-8.9%) (Clarkson *et al.*, 2007b) necessitating the restoration of these habitats for the benefit of native flora, fauna and urban residents. According to the Society for Ecological Restoration (2002, p. 3), ‘ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed.’ This ‘assistance’ often takes the form of replanting indigenous vegetation that would have been present before human disturbance. The focus in restoration is therefore primarily on plants (Young, 2000), with plant species richness being the most common measure of diversity recovery (Ruiz-Jaen & Mitchell Aide, 2005). It is generally assumed that the establishment and development of native vegetation will result in the recovery of an area’s fauna and the ecosystem services they provide, such as seed dispersal and pollination. This assumption, however, has rarely been the focus of research (Stagoll *et al.*, 2010; Bouma *et al.*, 2013), highlighting the importance of considering the entire ecosystem when planning a restoration project, and setting measurable goals which do not focus solely on plants.

While the response of birds to the process of restoration is highly species-specific, studies in non-urban areas have demonstrated that restoration can increase bird community diversity, promote community evenness, improve nesting success and result in recolonization of locally extirpated bird species (Ortega-Álvarez & Lindig-Cisneros, 2012). Restoration plantings in cities, however, may present a particular challenge to avifaunal recolonization given their generally small size, isolation from other restored or remnant patches or larger protected areas, and the often inhospitable nature of the surrounding urban matrix. Furthermore, while predation by introduced mammals is generally accepted to be the main cause of the ongoing decline of native forest birds (Innes *et al.*, 2010a), little is known about how urbanisation affects the presence, abundance and interaction of mammals in New Zealand (Russell & Stanley, 2018).

While restoration ecology has the potential to benefit native birds, birds in turn are crucial to the development and long-term sustainability of restoration plantings, performing key ecological services such as pollination and seed dispersal (Reay & Norton, 1999; Selwood *et al.*, 2009). Poor seed dispersal is a major limiting factor for forest recovery (Atkinson, 2001). In their review of bird pollination and fruit-dispersal

in New Zealand, Kelly and colleagues (2010) concluded that there is increasing evidence that the widespread loss of bird-plant mutualisms is threatening the reproduction of native plants. Introduced birds and mammals are not providing effective pollination or fruit dispersal of native plants, and these services are now almost entirely dependent on four native birds – tūī, silvereye (*Zosterops lateralis*), bellbird and kererū (Kelly *et al.*, 2006; Kelly *et al.*, 2010). Introduced mammals may further disrupt native forest regeneration through seed and seedling predation (Holl & Quiros-Nietzen, 1999; Overdyck *et al.*, 2013; Shiels *et al.*, 2017).

However, birds and mammals are not the only species affecting and affected by urban revegetation efforts. The Society for Ecological Restoration views restoration not only as a means of conserving biodiversity, but also of improving human wellbeing (McDonald *et al.*, 2016). Given that over half the world's population now lives in urban areas, restoring native ecosystems in cities has great potential to benefit people. Atkinson (2001) argued that restoration can be used as an outdoor classroom, capturing the imagination and motivating people who otherwise are unlikely to contribute their time and money to protecting native biodiversity. Recreating native habitats in the areas where most people spend the majority of their lives may further serve to reverse the vicious cycle of the extinction of experience.

The city as an ecological and social phenomenon

The capacity of cities to support native species will ultimately be determined by the attitudes, behaviour, consumption patterns and political support of city residents. Mascia and colleagues (2003) argue that social aspects are often the determining factor of a conservation intervention's success or failure. People are what make cities and they are the source of both the challenges and solutions presented by urban ecosystems (Groffman *et al.*, 2017), yet little is known about the relationship between city residents and urban biodiversity (Shwartz *et al.*, 2014a).

Urbanites generally have quite positive attitudes towards wildlife in general and birds in particular (Miller *et al.*, 2001; Baker & Harris, 2007). The general public's affinity for birds is reflected in the high percentage of people engaging in bird feeding and birdwatching activities (Jones & James Reynolds, 2008), with 46.6% of New

Zealanders feeding birds (Galbraith *et al.*, 2014). New Zealanders view native birds as positive symbols of their country, and so as part of their national identity (Hughey *et al.*, 2013). The charismatic and iconic nature of New Zealand's avifauna could be harnessed to promote public support for conservation and restoration. Positive attitudes towards birds, however, will not necessarily lead to responsible behaviour supporting native bird communities in cities or distant conservation lands. The relationship between residents' experience of wildlife in cities and their connection to nature needs further investigation.

The city represents a complex network of interdependent connections between the ecosystem within which the city was built, the built environment itself, the people whose values, needs and desires create and shape the city, and the organisms that survived urbanisation or even thrive in this highly altered ecosystem (Tanner *et al.*, 2014). This complex network of human actions and their effects on ecosystems and organisms highlights the need to integrate social and natural sciences in the study of urban ecosystems (McIntyre *et al.*, 2000). Cities cannot be understood and managed using the natural sciences alone.

The strengths of ecology and social science for studying the city

Forman (2014) defines ecology as the study of the interactions of organisms with their physical environment. He argues that when studying urban ecosystems, it is wise to maintain this core strength of ecology with its focus on plants, animals and microbes. To an ecologist, cities represent a special type of environment where humans are the keystone species controlling ecosystem structure and function (Walbridge, 1997). While ecologists are adept at 'thinking like a mountain'² (Leopold, 1949), they are much less comfortable thinking like a human (Tanner *et al.*, 2014). Researchers in the natural sciences prefer to view people as complex social animals (Parlange, 1998) and approach human behaviour as the outcome of a linear, rational process of individual

² A term used by environmentalist Aldo Leopold in his book 'A Sand County Almanac' to describe thinking in terms of an interconnected ecosystem. The specific illustration Leopold uses is of a mountain viewing wolves as beneficial (whereas people are focused on killing the wolves), because as an apex predator the wolf controls deer populations which, when unchecked, over-browse and thereby destroy the mountain.

decision-making (Hargreaves, 2012). As anthropologist Charles Redman explained, ‘I’m very sympathetic to the idea that people can be studied, and their institutions, their organizations, their attitudes can actually be understood enough to make predictions’ (Parlange, 1998, p. 582). Scientists seek to construct predictive models that will reduce the complexity of human action to fundamental, universal laws and rules, as they are used to doing in their own field. However, these reductionist approaches have had little to no success at predicting social phenomena (Christens, 2016) or changing people’s behaviour (Hargreaves, 2012). Christens (2016, p. 4) quips, ‘there is therefore at least one thing that we can predict with relative confidence: human communities will consistently evade precise and accurate predictions’. Flyvbjerg (2001, p. 166) argues that social science should not attempt to replicate the methods and theories that govern the natural sciences:

First, we must drop the fruitless effort to emulate natural science’s success in producing cumulative and predictive theory; this approach simply does not work in social science [...] We may transform social science to an activity done in public for the public, sometimes to clarify, sometimes intervene, sometimes to generate new perspectives, and always to serve as eyes and ears in our ongoing efforts at understanding the present and deliberating about the future.

While people are an integral component of urban ecosystems, I argue that human values, motivations and behaviour cannot be thoroughly investigated or adequately understood using only methods and frameworks from the natural sciences. In order to study the vicious cycle of the extinction of experience, with its interconnected problem of loss of native biodiversity and the disintegration of our relationship with the natural world, I concluded that I would need to address both the social and ecological aspects of the issue. I chose to do so, however, using a multidisciplinary approach. Accordingly, I have drawn on social science to study people, and ecology to study non-human organisms and their interactions with their environment.

Comparing the quantitative and qualitative research traditions

The epistemological assumptions (how we generate knowledge and what it is possible to know), underlying philosophy (the system of values an individual adheres to) and research methods of qualitative research differ fundamentally from those of quantitative research (Evely *et al.*, 2008; Yilmaz, 2013). The natural sciences are based on an objectivist epistemology and a positivist philosophy and rely on quantitative research methods to collect and analyse data. The positivist worldview holds that knowledge can only be generated through observation experienced through the physical senses and interpreted through reason and logic (Moon & Blackman, 2014). Objectivist epistemology claims there is an objective, static reality that exists independent of human thought, perceptions, emotions and imagination. Quantitative research tests theories and attempts to explain phenomena using numerical data analysed by means of statistics (Yilmaz, 2013). Quantitative research seeks to develop or discover explanatory universal laws by measuring and analysing causal relationships between variables, with the aim of making predictions and statistics-based generalisations. For this reason, natural scientists attempt to ensure their samples are representative of the population studied and their data meet the assumptions of the statistical methods they use (Moon & Blackman, 2014).

While the natural science disciplines are largely limited to positivism, objectivism and quantitative methods, the social science disciplines can draw on a wide variety of philosophical perspectives, epistemologies and research methods. Qualitative research methods arise from the epistemological conviction that people, their behaviour and resulting social phenomena are complex, interconnected, potentially contradictory and context-dependent and therefore cannot be reduced to a set of variables and universal, predictable, logical rules (Yilmaz, 2013). Qualitative research results from a subjectivist (we impose meaning and values on our world) or constructionist (reality is constructed through our engagement with the world – ‘what we call the world is a product of some mind’ (Bruner, 2009)) epistemology. It does not seek to discover a single, objective reality, but assumes that people will interpret and respond to a given phenomenon in different ways. The way in which individuals make sense of and navigate their world is influenced by their cultural background, social norms, personal experiences and emotions (Moon & Blackman, 2014). Of key interest to qualitative

researchers are people's interpretations of the world, because these are what shape human behaviour and experiences of the world and engagements with it. The aim of qualitative inquiry is an in-depth, detailed, descriptive understanding of how people perceive a given phenomenon, and the approach emphasises the importance of context, individual variation and meaning-making.

Qualitative research methods include interpretative phenomenological analysis, grounded theory, hermeneutic-interpretive research, ethnography and discourse analysis (Elliott & Timulak, 2005). These methods focus on linguistic rather than numerical data, pose open-ended, exploratory research questions and allow themes and descriptions to arise from the participants, as opposed to creating pre-determined options and rating scales (Elliott & Timulak, 2005). Research findings are then analysed within the context of previous studies and existing theoretical constructs (Hodgetts & Stolte, 2012).

The positivist approach of the natural sciences tends to dominate the field of restoration ecology and conservation biology, even when people and their behaviour are the focus of research (Evelly *et al.*, 2008). In this thesis, I embraced the epistemological assumption that people will perceive urban nature in different ways, and their interactions with the natural world in their city may be contradictory, complex, ambiguous and unpredictable. I therefore used qualitative research methods to facilitate a deeper understanding of the human relationship with urban nature in a New Zealand city (Chapter 2).

A tale of two cities: contextual background to the study sites

The study sites referred to in chapters 3 and 4 were originally chosen by MacKay (2006) and further developed by Overdyck (2014) and Wallace (2017). The research presented in these two chapters was conducted in two cities on New Zealand's North Island – Hamilton in the Waikato region and New Plymouth in the Taranaki region. The research presented in chapter 2 was undertaken solely in Hamilton.

The earliest recorded settlers of the Hamilton area, or Kirikiriroa ('long stretch of gravel') as it was first called, were Māori from the Tainui waka (canoe) who arrived

approximately 700-800 years ago. New Plymouth, originally Ngāmotu ('the islands'), was settled by Māori from three waka (Tokomaru, Kurahaupō and Aotea) around 1250 AD (Clarkson, 2011). These early settlers began to alter New Zealand's flora and fauna through hunting and gathering food, as well as through the introduction of predatory mammals – kurī (Polynesian dog) and kiore (Polynesian rat). The first permanent European settlers arrived in Hamilton in 1864 and New Plymouth in 1841, bringing with them over time a further 55 mammal species, 32 of which have become established (Courchamp *et al.*, 2003). From the 1860s, acclimatization societies began introducing a range of non-native birds, including starlings, yellowhammers, skylarks, chaffinches, blackbirds and thrushes (Walrond, 2008).

The temperate rainforests and wetlands of the Hamilton area were largely destroyed during urban development and this widespread habitat destruction continues to this day, although at a slower pace, since the establishment of the Resource Management Act of 1991 (Coleman & Clarkson, 2010). Currently only 2.1% of Hamilton's approximately 11,000 ha is covered in indigenous vegetation (Clarkson *et al.*, 2007b), the lowest percentage for any New Zealand city, largely a consequence of the gentle terrain and ease of clearance of original vegetation by fire and axe.

In contrast, New Plymouth has the highest percentage of native vegetation cover within the built-up matrix – 8.5% of its area (Clarkson *et al.*, 2007b). The main forest types remaining are hillslope temperate rain forest dominated by tawa (*Beilschmiedia tawa*), and gully forests with pukatea (*Laurelia novae-zelandiae*) and kahikatea (*Dacrycarpus dacrydioides*) and occasional waiwaka or swamp maire (*Syzygium maire*). The rolling terrain with well-defined hills and gullies was conducive to the survival of forest despite development pressure.

An early appreciation by European settlers of the scenic and aesthetic values of bush and landscape, however, played a significant role in the retention of indigenous vegetation. While New Plymouth's 1841 city plan (New Plymouth was one of the planned New Zealand Company settlements devised by Edward Gibbon Wakefield) provided for a town belt (green belt), unfortunately the land was allocated to settlers for farms (Pollock, 2010). However, the Taranaki Scenery Preservation Society, established in 1891 and led by the surveyor W. H. Skinner, was active through to the

mid-1900s and was responsible for the protection of many of the most significant bush remnants at the Ratapihipihi, Huatoki and Barrett domains. As the city grew these important reserves were incorporated into the developed city matrix (Clarkson *et al.*, 2007b).

A further factor was the focus on native plants fostered by horticulturalists Victor Davies and James Duncan. Established in 1910, Duncan & Davies Ltd. became a world-renowned plant supplier and their promotion of New Zealand flora led to the extensive incorporation of native plants into New Plymouth city parks and gardens, including the most well-known, Pukekura Park. Duncan & Davies Ltd. were the inaugural winners of the Loder Cup in 1929 and Victor Davies also received the cup in 1968 for his work in protecting and cultivating New Zealand native flora (Fryer, 1998).

Hamilton possesses an extensive network of branching gullies, which cover approximately 770 ha, or 7% of the city area (Cornes *et al.*, 2012). Since they represent the major remaining undeveloped greenspace in an otherwise highly built up environment, Hamilton gullies today are a central focus for the restoration of the city's native vegetation (Clarkson & McQueen, 2004). Gully clean up and beautification was the goal of pioneering restoration projects, which began in Hamilton City 40-50 years ago, originally initiated by private landowners (Coleman & Clarkson, 2010). The Gully Restoration Programme (Hamilton City Council) was formulated by Hamilton City Council in 2000 to encourage gully restoration on private and public land, in conjunction with the promulgation of a gully reserves management plan in 2001 (Hamilton City Council, 2007). A series of annual workshops, newsletters, a plant give-away scheme and awards for the best gully all helped to advance the cause of gully restoration. Reconstruction of indigenous habitat in Hamilton was further advanced in 2004 when 60 ha of public land were dedicated to the establishment of Waiwhakareke Natural Heritage Park (Clarkson *et al.*, 2012).

Hamilton is the second fastest growing population centre after Auckland, characterized by a large student population, with approximately 40,000 students attending the city's tertiary institutions. According to the 2013 census figures, Hamilton's population is 69.5% Pākehā/European, 21.3% Māori, 13.8% Asian, 5.1% Pacific Peoples and 2.0% Middle Eastern, Latin American and African (Statistics New Zealand, 2013).

New Plymouth's economy relies on dairy farming and the petroleum industry, the latter supported by the growing engineering, professional and hospitality sectors. According to the 2013 census figures, New Plymouth's population is 86.7% Pākehā/European, 15.7% Māori, 4.0% Asian, 1.8% Pacific Peoples and 0.5% Middle Eastern, Latin American and African (Statistics New Zealand, 2013).

The two study cities therefore provide strongly contrasting environmental histories and current social and demographic profiles. Comparative studies of this type can be useful in understanding the drivers and reasons for differences in the things being compared (Heidenheimer *et al.*, 1990). In this case, the focus is on bringing nature back into each city, and specifically on the benefits of restoration planting to urban residents and native forest birds.

The chronosequence method

The research presented in chapters 3 and 4 follows from work done by MacKay (2006), Overdyck (2014) and Wallace (2017). Hamilton study sites were originally identified by MacKay (2006), who used a chronosequence approach to explore change in vegetation over time since restoration planting. Site age was determined from planting plans and confirmed during vegetation surveys by tree coring (MacKay, 2006). Wallace (2017) expanded the study sites to include plots in New Plymouth for her research on change in urban forest structure and function over time since initial restoration planting.

The chronosequence approach, also termed space-for-time substitution, involves the extrapolation of a temporal trend from a series of sites formed from the same parent material or substrate that differ in time since formation (Pickett, 1989). Since few studies span more than several decades, and long-term monitoring is rarely done, the chronosequence approach allows investigation of change over time which cannot be achieved in any other way (Walker *et al.*, 2010). This approach has been used to investigate change in bee communities in restored US tallgrass prairies (Tonietto *et al.*, 2017), variation in dung beetle species diversity, community composition, functional diversity and ecological functions in restored rain forests in Australia (Derhé *et al.*,

2016) and changes in breeding bird abundance and species composition in regenerating New England forests (Duguid *et al.*, 2016).

As vegetation in restored urban forests in New Zealand matures over time and the forest habitat becomes more structurally complex with an increase in tree height, diameter at breast height (DBH), canopy cover and vegetation layers, it can be hypothesized that the avifaunal composition of the site will change as well, with a greater number of native forest species able to meet their foraging, nesting and breeding requirements within these sites. The relationship between time since initial restoration planting and native forest bird species richness, abundance and community composition will not be straightforward, however, and will be further influenced by a complex interaction of vegetation variables, landscape characteristics and predation pressure.

Analysing diverse sites together in a chronosequence assumes that they are following a similar restoration trajectory and that the same bird species are present in the surrounding landscape and able to use each site. Few native New Zealand forest birds persist in urban areas (Miskelly *et al.*, 2005; Stevens, 2006; van Heezik *et al.*, 2008b), with the same diurnal species detected in bird counts in both Hamilton and New Plymouth – grey warbler, fantail, silvereye, tūī, bellbird, kererū, kākā, kingfisher and shining cuckoo (Fitzgerald & Innes, 2015; Lambert & Hartley, 2016). Therefore, there is evidence to support the assumption that the same bird species are present in the surrounding landscape and able to use each study site. However, the assumption that every study site is following exactly the same restoration trajectory is highly improbable, though previous work within this study system supports the assumption that certain vegetation variables that change over time since restoration, such as canopy openness and DBH, are following similar, predictable trajectories (Wallace *et al.*, 2017). Thus, the use of the chronosequence method in this research is based on the assumption that structural vegetation variables at all study sites are following similar restoration trajectories, that the same native bird species are able to use these sites, and this use is affected by change in structural vegetation variables over time.

Brief review of main statistical methods used

In chapters 3 and 4, I use piecewise structural equation modelling (SEM) to evaluate direct and indirect effects and the relative importance of explanatory variables on response variables. According to Lefcheck (2016), structural equation modelling is a statistical framework that unites multiple predictor and response variables to model multivariate relationships in a single causal network. SEM assumes that relationships among variables are causal and linear. The causal relationships studied are presented using box-and-arrow diagrams which simultaneously represent scientific ideas and the series of regression equations used to test those ideas (Byrne, 2001; Grace, 2008; Laughlin & Grace, 2019). According to interviews conducted by Laughlin and Grace (2019), SEM is highly valued by ecologists for its capacity to test and quantify indirect, offsetting and suppressed effects. Currently the greatest barrier to using SEM in ecology is the lack of training opportunities (Laughlin & Grace, 2019).

In chapters 3 and 4, I use non-metric multidimensional scaling (NMDS) to graphically represent differences in bird and introduced mammal communities in two-dimensional space. In NMDS ordination, the distance between points reflects how similar or dissimilar each sample is to the other. Unlike other ordination techniques such as principal components analysis (PCA), NMDS only uses rank ordering of distances between samples, thus making few assumptions about the distribution of the data, and can therefore handle the non-normal distributions and large proportion of zeroes commonly found in ecological data sets (Gotelli & Ellison, 2004; Paliy & Shankar, 2016; Ramette, 2007). NMDS is computationally intensive, so can be slow for large data sets. This was not a problem for my own medium-sized matrices, however.

To test for statistical differences in bird and mammal community composition between site types, I used permutational multivariate analysis of variance, or PERMANOVA (Anderson, 2001b). PERMANOVA was chosen because it makes no assumptions regarding the distributions of original variables or dissimilarities (Anderson, 2014) – data can be severely non-normally distributed and zero-inflated.

1.2 Research aims, questions and thesis outline

This research draws on a multidisciplinary framework, using ecology and social science, with the overarching aim of investigating the capacity of urban forest restoration to break the vicious cycle of the extinction of experience by providing habitat for native birds and reconnecting urban residents with nature in New Zealand cities.

In much ecological restoration research and practice, the focus, quite understandably, is on plants and revegetation (Young, 2000) (Wallace and Clarkson 2019). Planting and weeding are often necessarily the first step in a terrestrial restoration project. However, the goals of restoration are not simply to recreate native vegetation communities, but entire ecosystems and the services they provide. Therefore, reflecting a multidisciplinary framework, this thesis considered the habitat value of native vegetation restoration to native birds, the impact of mammalian predators on native birds and seedling regeneration, as well as the complex interactions between plants, birds and mammals. Further, this interconnected ecosystem approach was extended to take in the human dimensions of ecological restoration. Figure 1.1 is an illustration of the interconnections occurring in and around urban restored sites between plants, birds, mammals and humans. The interconnections shown in Figure 1.1 are dynamic and may vary from site to site, while the interactions between all factors have a bearing on the success or otherwise of urban ecological restoration.

Adopting a multidisciplinary framework that combines ecological science with social science is increasingly important in the context of the Anthropocene, where human actions are transforming the earth and causing mass extinctions (Vitousek *et al.*, 1997). Given the extent of human impact on the environment, it is untenable for ecologists to focus solely on plants and animals. However, as argued above, studying human experiences and actions cannot be done through scientific methods alone. Accordingly, alongside the scientific study of plants, birds and mammals, this research has incorporated a phenomenological methodology to explore the experiences of people living near urban greenspaces. Reaching beyond the disciplinary boundary of ecological science has provided a human dimension to urban ecology. In particular, the multidisciplinary framework underlying this research provides a foundation for the

claim that improving the health of the natural environment in and around cities has win-win potential since it offers benefits to both native biodiversity and urban residents.

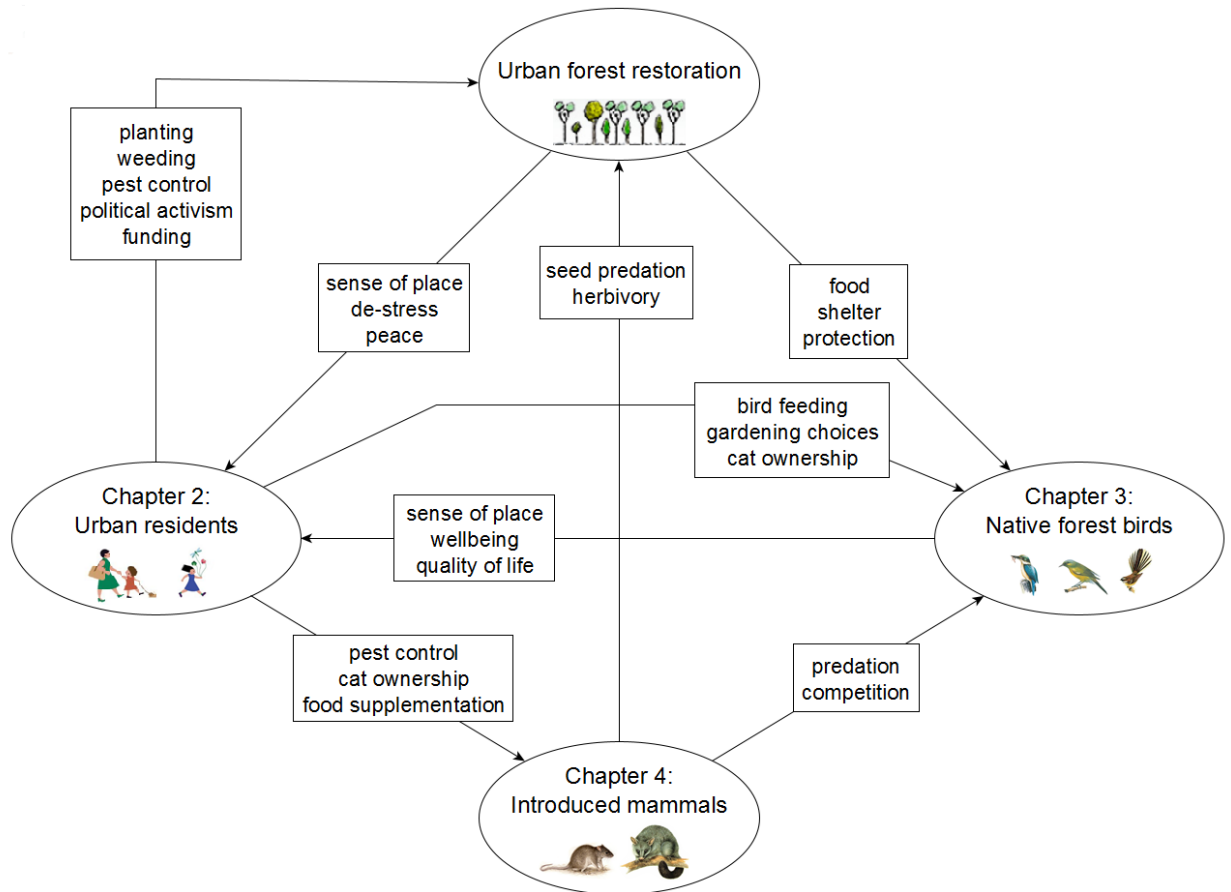


Figure 1.1: A diagrammatic representation of the thesis structure and how the various components are interrelated.

Thesis outline and research questions

This thesis consists of three empirical research chapters (Chapter 2-4). The final chapter (Chapter 5) synthesizes the results of the entire thesis and provides recommendations for management and future research.

In **Chapter 2**, I use methods, knowledge and theories developed in the social sciences to conduct a qualitative and exploratory study of New Zealanders' connection to nature in their city. The chapter begins with an exploration of how the concepts of sense of place, place attachment and dwelling can be employed to shed light on the current

relationships between people and nature in cities. Semi-structured, ‘go-along’ interviews allowed me to investigate Hamilton residents’ personal experiences of nature in their neighbourhood parks and home gardens. I used an interpretative phenomenological approach to analyse the data. The purpose of this qualitative component of the thesis was to improve understanding of what nature urban residents notice and appreciate, how they value gardens and parks, and what benefits they believe they derive from urban nature. I further attempted to differentiate the benefits participants derive from bush parks dominated by native vegetation and those from lawn parks dominated by introduced vegetation.

In order to investigate the current status of ‘connection to nature’ in urban New Zealand residents, I posed the following key research questions:

- 1) What is the relationship between people and nature in New Zealand cities?
- 2) How and where are urban residents experiencing nature?
- 3) Does the type of frequently visited park influence people’s perceptions of nature?

Chapters 3 and 4 use the methods and theories developed in ecology to investigate the potential of urban forest restoration to benefit native birds, and the related question of how the mammal community changes over time since restoration and the impact this has on both birds and native tree regeneration.

Chapter 3 investigates the relative importance of vegetation structure, landscape pattern and invasive mammals on bird communities along an urban forest restoration chronosequence, representing 71 years of forest development in two New Zealand cities.

To investigate the potential of urban forest restoration to provide habitat for native birds, I posed the following research questions:

- 1) How does the bird community change over seven decades of urban forest restoration?

- 2) What is the relative importance of vegetation structure, landscape pattern and invasive mammals to bird communities in restored urban forests?

Chapter 4 investigates the impact of vegetation structure, landscape composition and interspecific interactions on the presence and relative abundance of introduced mammals in the same chronosequence of restored urban forests. I further investigate the impact of introduced mammals on native forest regeneration in restoration plantings in two New Zealand cities.

To do so, I posed the following research questions:

- 1) Does the mammal community change over time since initial restoration planting?
- 2) What is the relative importance of vegetation structure, landscape pattern and interspecific interactions to mammal communities in restored urban forests?
- 3) What impact are introduced mammals having on the regeneration of restored urban forest?

Chapter 5 presents an overall synthesis and discussion of results from all chapters. Based on the findings of my studies, I then provide recommendations for management and further research.

Chapter 2: A qualitative exploration of personal experiences of urban nature in a New Zealand city

2.1 Introduction

This chapter focuses on the human dimensions of ecological restoration, which requires a different understanding of place – one that does not focus simply on the material and observable phenomena in a particular environment, but considers the subjective and emotional connections between people and places. Such dynamic human processes are hard to isolate and analyse using quantitative methods but are nevertheless crucial considerations for understanding the possibilities for ecological restoration in cities today and the barriers that stand in the way.

The vicious cycle of biological poverty

The majority of the world's urban population lives in what Turner *et al.* (2004) have termed 'biological poverty'. Urbanization often results in a reduction of biodiversity through habitat destruction and degradation as well as biotic homogenization, or replacement of unique endemics with globally widespread species not native to the region (McKinney & Lockwood, 1999; McKinney, 2006; Goddard *et al.*, 2010). The diminished levels of native biodiversity in cities, coupled with the loss of opportunities for people to experience, play in, and explore natural areas, have led to what entomologist Robert Pyle (1993) termed the 'extinction of experience'. This is a theoretical vicious cycle of increasing separation from the natural world, as each new generation grows up in cities with ever decreasing levels of native biodiversity (Belaire *et al.*, 2015; Cox & Gaston, 2015).

Urbanization and the resulting extinction of experience are believed to diminish our knowledge of the natural world. The urban public claims to appreciate greater species richness and believes increases in biodiversity lead to increases in their well-being.

Researchers find, however, that urbanites systematically underestimate species richness across all taxa and their estimates typically have no relationship with actual species richness in the parks or neighbourhoods studied (Dallimer *et al.*, 2012; Schwartz *et al.*, 2014b; Belaire *et al.*, 2015). Wildlife identification skills and knowledge of animal ecology and behaviour are poor among urban residents (Kellert & Berry, 1980; Dallimer *et al.*, 2012; Elliot *et al.*, 2016). Cosquer *et al.* (2012) go so far as to suggest that city dwellers have largely lost the practice and habit of observing the natural world.

The central assumption behind the extinction of experience theory is that people need direct, personal experiences of things, especially during childhood, if they are to care deeply about them. Environmental Studies professor David Orr notes that, ‘we tend to bond with what we know well’ (Orr in Kellert and Wilson, 1995, p. 422). Because urban residents are given fewer opportunities to interact and bond with nature, the extinction of experience is believed to result in apathy and indifference towards the natural world. Such observations have existed for some time. American palaeontologist Stephen Jay Gould argued, ‘we cannot win this battle to save species and environments without forging an emotional bond between ourselves and nature as well – for we will not fight to save what we do not love’ (Gould, 1993).

The concept of the extinction of experience stresses the importance of emotional connections to places, and the nature and people that make up these places. Cognitive and rational models of human behaviour focus on the knowledge gap between values and actions and therefore advocate for changing people’s behaviour through education. What people do in their daily lives, however, is never simply a linear process of deductive reasoning. According to Blake (1999), ‘people do not have a fixed, rational and ready-made set of values [...] rather, people’s values are negotiated, transitory and sometimes contradictory.’ Along similar lines, Hodgetts and Stolte (2012, p. 382) highlight the inconsistencies of human behaviour, claiming that ‘human existence is contradictory, complex and full of ambiguities’. Human behaviour and action are instead considered to be the result of an unpredictable and dynamic emotional process. Intrinsic motivation is more likely to stem from structures of feeling rather than knowledge of facts. Grob (1991) hypothesises that the stronger a person’s emotional reaction to environmental degradation, the more likely that person is to engage in pro-environmental behaviour.

People's feelings of connection and belonging in places are crucial to their willingness to invest in and care for them. Creating and strengthening emotional attachment to areas of native nature could have a strong and long-lasting impact on people's pro-environmental behaviour. Emotional connections and personal experiences may be the key to improving our appreciation of native nature and our relationship with it, resulting in greater commitment to its protection and enhancement. Given the great potential of emotional attachment to place to influence our actions, we review the concepts of sense of place, place attachment and dwelling. We explore how these concepts can be employed to shed light on the current situation in cities, and how they might contribute to reversing the vicious cycle of biological poverty.

Sense of place and place attachment

Relph (2009, p. 1) defines place as 'a territory of significance, distinguished from larger or smaller areas by its name, by its particular environmental qualities, by the stories and shared memories connected to it, and by the intensity of meanings people give to or derive from it'. Being in place is not simply about one's location in, or physiological responses to, a particular physical environment. Instead, place is also fundamentally an emotional, subjective experience (Tuan, 1986). Moreover, since our lives are always conducted *somewhere*, they are always interwoven with particular material, social, cultural and political contexts (Hodgetts *et al.*, 2020). This means place is simultaneously a universal and unique phenomenon, and something that is experienced both at individual and collective levels.

Places acquire meaning and become repositories of memories through people's lived experiences in them. At an individual level, a person's sense of place refers to that part of their personal and communal identity which develops through an association over time with the physical and symbolic features of a place and the social interactions which occur in that place (Bernardini & Irvine, 2007; Dallimer *et al.*, 2012). According to Twigger-Ross and Uzzell (1996), landscapes can be seen as memorials to the past. Over time, the experiences and memories associated with a place can strengthen people's attachment to these places.

Emotional attachment to a particular landscape can often result in a need to modify newly encountered areas in order to reproduce the characteristics of other important places, a concept in psychology termed place-congruent continuity (Bernardini & Irvine, 2007). This need to modify landscapes is well demonstrated in New Zealand, where the blackbirds, goldfinches and oaks of England were lovingly introduced by acclimatization societies and nostalgic settler farmers in the 1800s. Homogenization, or the loss of endemics and proliferation of globally common generalist species, can result in a place losing its cultural significance (Standish *et al.*, 2013). As a respondent to a qualitative survey on perceptions of native species in Scotland put it, ‘places lose their cultural identity if it’s all going to turn up looking much the same’ (Selge *et al.*, 2011). Thus, biotic homogenization can erode people’s emotional bonds with places.

Relph coined the term ‘placelessness’ to describe the replacement of diverse and distinctive places with anonymous and standardized landscapes. American strip malls would be one example of placelessness. They represent uniform, standardized, mass-produced cityscapes, since a mall in New York City would be indistinguishable from one in Los Angeles. Relph believes placelessness arises from disregarding the spirit of a place, an uncritical acceptance of mass values, and an overriding concern with efficiency as an end in itself (Seamon & Sowers, 2008). An ecologist might identify placelessness in the transformation of New Zealand’s unique landscapes of indigenous forest to the rolling green hills dotted with poplars and pin oaks characteristic of England.

The quality and intensity of the relationships developed with everyday urban environments depend on people’s familiarity with a place, which can be influenced by the frequency, intensity and type of use (Ryan, 2005). Different types of interaction with a place will result in different forms of attachment (Ryan, 2000). Hence, one can assume that a restoration volunteer actively involved in planting native trees may have a different attachment to a place than an urban resident who walks through the restored site on their way to work.

Feelings for places can be ambiguous, with attachment not necessarily predicated on the pleasantness of a place, as illustrated by the following quote from American novelist Henry James – ‘It is difficult to speak adequately or justly of London. It is not

a pleasant place; it is not agreeable, or cheerful, or easy, or exempt from reproach. It is only magnificent.’ Feelings for places are not always positive, and can range from discomfort or fear, to indifference and taken-for-granted unawareness, to a devotion so strong people are willing to defend and sacrifice their lives for their homes or countries (Seamon, 2014).

Ryan (2005) notes that the majority of studies on place attachment focus on people’s connection to residential settings, such as the home or neighbourhood, with fewer studies investigating attachment to natural settings. In his study of natural area users in Michigan, Ryan (2005) found that attachment to urban parks and natural areas is influenced by the physical characteristics of the landscape, respondents’ previous experiences within the area and their knowledge of nature in general. Frequent use often results in stronger attachment, suggesting that the emotional connection to the area is built over time through experiences in these landscapes. In their study of the wilderness experience of two groups of women, Fredrickson and Anderson (1999) found that the social interactions between the women were as important to the meanings they associated with particular natural areas as the visual appeal and biophysical features of the landscape. Such research demonstrates that the significance of place is not restricted to the features of a place, but also extends to what happens there – in other words, the human-centred processes of being in a place.

Dwelling

Dwelling is a concept developed by German philosopher Martin Heidegger. Dwelling describes the process by which the place(s) in which we exist can become a personally significant world and home (Seamon, 2000). Dwelling involves regularity, repetition and cyclicity (Seamon, 2000). Phenomenologist Bernd Jager describes dwelling as involving a ‘round world [...] the reoccurring [sic] times of seasons, of the cycles of birth and death, of planting and harvesting, of meeting and meeting again, of doing and doing over again’ (Jager, 1975, p. 26). Thus, dwelling describes a relationship with place that is built over time, as we move through our homes, gardens, neighbourhoods and workplaces, performing everyday activities and interacting with the people and objects around us. These habitual, taken-for-granted routines give meaning to specific

places – meanings which are created over time and are capable of constantly evolving and changing.

Dwelling recognizes that human lives and actions shape landscapes, and landscapes in turn affect our experiences. Ingold cites Merleau-Ponty's description of landscapes, which are 'not so much the object of our thoughts as the homeland[s] to our thoughts' (Ingold 1995, p. 76, citing Merleau-Ponty, 1962, p. 24). Dardel (cited in Relph 1985, p. 24) expresses a similar view – 'landscape is not, in its essence, made to be looked upon but, rather, is an insertion of man into the world, a site for life's struggle, the manifestation of his being and that of others.' According to Seamon, 'people and their worlds are integrally intertwined' (Seamon, 2014) and because they always exist together, they can only be understood as a holistic relationship (Seamon, 2000).

Dwelling involves the basic human need for stability, order, familiarity and continuity. Sibley (1999, p. 115) argues that 'familiarity and predictability are important to many people. There is a common desire to live in a place which is stable and orderly.' Feeling that one belongs to a place can give people a positive sense of security. Yet, some caution is needed, given that places do not produce universal homogenous experiences. For instance, a sense of belonging in a place can also be perceived more as an oppressive and restrictive responsibility (Sibley, 1985), and can result in some people feeling trapped or tied to a given place. Such ambiguities remind us that places are complex. While this complexity means places evade simple definitions or deductions, it also means human engagements with places are not necessarily locked into inevitable trajectories – this leaves the door open to possibilities for change and alternative futures.

According to Heidegger, dwelling involves cherishing, protecting, preserving and caring for the things, people and events which together make up the places where we choose to live (Seamon, 2000). This relationship with a place is characterised by active involvement with the physical environment (Seamon, 2000) and is full of concern and a sense of responsibility (Relph, 1985). Reflecting on contemporary urban contexts, (Seamon, 1985) warns of a breaking down of this relationship with place as people cease to be actively involved in the creation and care of the places where they live and work:

Our modern era cultivates journey, horizon and reach often at the expense of dwelling, centres and homes. One out of every three Americans changes place of residence every three years. Can we afford to continue this pattern? Do we want the interchangeable, artificial environments that Relph has called placelessness or should we consider the reestablishment of places, localities and communities grounded in landscapes and natural environments?

Seamon's warnings are of increasing relevance in contemporary cities, which are undergoing rapid change and urban intensification, with many communities facing increasing pressures from population increases, rising housing costs, transience, and social fragmentation. One could argue that it is now more necessary than ever to ensure that city dwellers can experience a connection to place and nature. Before going on to consider what the reestablishment of places grounded in landscapes and natural environments might look like, we first investigate our perceptions of and relationship with nature in cities today.

Nature in the urban environment

Urban landscapes are often characterized by plant communities which have been wholly determined by human choice (Hope *et al.*, 2003). In residential yards and gardens, as well as public green spaces, decisions on what to plant depend on aesthetic preferences, social conventions, the economic value seen for the land and the political system within which people operate (Nassauer, 1995a). Such human choices in cities are thus driven by a wide range of motives and trends, which do not necessarily align with sustaining native nature and biodiversity.

The majority of publications on landscape preferences have investigated rural or wilderness landscapes, and little research has been done on perceptions of native patches of vegetation within urban settings. The research that has been done in cities tends to focus on gardens and gardening practices. In their study of householders in Leicester, UK, Bernardini and Irvine (2007) show the differences in the way urban residents value public greenspaces and private gardens. Public parks were seen as community meeting places, valued for the opportunity they provide for meeting neighbours and developing a sense of belonging to a neighbourhood. Private gardens,

on the other hand, are valued for relaxation, peace, meditation and stress reduction. By allowing immediate and regular access to nature, gardens, which literally surround people's homes, mitigate respondents' sense of urban claustrophobia.

Gill *et al.* (2009) classified the values people hold for urban greenspaces into three categories: as important to community and individual identity, as restorative, and as places to engage with nature. In their study of visitors to a suburban bushland south of Sydney, these researchers found many participants view the bush as providing an escape from their everyday lives of home and work. This theme of escapism is reiterated in other studies of gardens, which describe them as a positive distraction or escape from the anxieties of everyday city life (Gross & Lane, 2007), a retreat from the bustle of the city (Longhurst, 2006) and a haven from the public world of work and politics (Bhatti & Church, 2000).

In her paper on gardening practices in New Zealand, Longhurst (2006) argues that domestic gardens in Hamilton and Auckland are paradoxical spaces, which challenge the dichotomy of nature and culture, private and public, individuality and sociality, leisure and work, and colonial and postcolonial. Longhurst identifies a shift from the UK-inspired trend of cottage gardening, which mixes flowers, fruits, vegetables and herbs, to a more ordered approach where plants are allocated separate spheres. Backyards are more private and are thus dedicated to vegetable gardens, fruit trees, clothes lines, child play areas and dog houses. Front gardens are more visible to the public and reserved for lawns, flowers and driveways. Longhurst further comments on the increased planting of native New Zealand species as well as subtropical plants associated with South Pacific nations, such as palms, cycads, taro, bird of paradise, hibiscus and frangipani. While this finding relating to the South Pacific plants may be more apparent in Auckland where the climate permits the growing of tropical plant species, Longhurst's conclusion remains relevant to other New Zealand cities. She interprets this shift from cottage to subtropical gardens as indicative of a change in New Zealand's identity from that of a British colony to an island nation aware of its indigeneity and autonomy from the motherland. People can feel patriotic about native landscapes, perceiving a symbolic link between native flora and fauna and their national identity.

Despite this increased interest in planting natives, studies suggest only a small percentage of plants in the majority of gardens are native species. In their study of Dunedin householders, van Heezik *et al.* (2012) found that while most participants report a preference for native plants, these only made up 17% of the total number of woody species, and the overwhelming majority of non-woody species in lawns and beds were exotic. Similarly, Doody *et al.* (2010) found that Christchurch residents report that they value native plants but would pull them out as weeds when they grew as self-dispersed seedlings on their lawns.

Feagan and Ripmeester (2001) describe lawns as a taken-for-granted cultural artefact and an icon of communal order, the importance of which is evidenced by the pervasiveness of lawns and the effort and resources invested in its maintenance. Homeowners are judged based on the appearance of their lawns and gardens. The owners of neat, tidy lawns are viewed as hard-working, conscientious stewards and good neighbours (Nassauer, 1995b). In their study comparing the perceptions of ‘traditional’ lawn owners and ecological gardeners in southern Ontario, Feagan and Ripmeester (2001) found that those who cultivated traditional lawns viewed alternative gardens with horror and discomfort. Such ‘unruly’ gardens were identified as being out of place, and their owners were seen as lazy, inconsiderate, irresponsible, poor neighbours, disgraceful, on welfare, and selfish.

According to Longhurst (2006, p. 585), gardens are often viewed as ‘improvements on nature, attempts to tame the wild, to order the disorderly; nature is ‘civilized’ in order to delight people.’ Gardens have further been described as the intersection of nature and culture (Kiesling & Manning, 2010) or the ‘balancing point’ between wild nature and human control (Francis *et al.*, 1990). In a study of green roofs in Chicago and Toronto, many participants disapproved of the messy aesthetics of the prairie-style green roof, describing it as unkempt and too ‘wild looking’, which the author concluded may be due to expectations of what kind of nature is acceptable and believed to ‘belong’ in the city (Loder, 2014). Thus, while prairies may be seen as beautiful and intriguing when they exist beyond the city limits, within urban areas residents may demand that nature conform to the modernist city ideals of order, control and tidiness.

Nassauer (1995) and Thayer (1989) suggest that the apparent disorder and chaos of natural habitats transgress Western cultural norms for the neat appearance of landscapes. These neat, tidy landscapes, characterized by large expanses of mown lawns and straight borders of trees and colourful flowers, are described by Marzluff and Ewing (2001, p. 8) as ‘ecological disasters that reduce biodiversity, contribute to global warming, stress water supplies, use global fossil fuels, and encourage the use of pesticides and herbicides’. Such conventional yards and gardens lack the deep leaf litter, woody debris, dense shrubs and scattered saplings that can provide habitat for a host of invertebrates, birds, small mammals and reptiles. Thus, there appears to be a stark contrast between what urban residents value in yards and gardens and what is beneficial to native plants and animals.

In their review of the literature on how people perceive and value urban biodiversity, Botzat *et al.* (2016) found that while people favour urban landscapes that show signs of careful design and management over more wild, naturalistic spaces, people are willing to accept ‘wild’ landscapes if signs of individual or collective care are present. Examples of such indications of care in western society, called ‘cues to care’ by Nassauer (1995b), include mowing along paths, the presence of fences and obvious trimming and pruning. Furthermore, studies have found that negative attitudes towards ‘messy’ landscapes can be changed or improved by providing people with information on the benefits ‘wilder’ landscape features provide to certain animals (Home *et al.*, 2009). Chicago residents who enjoyed observing birds in their backyards increased their wildlife gardening efforts, which involved the creation of thicker, denser, ‘messier’ habitat, in order to attract more birds (Belaire *et al.*, 2015). For Australian homeowners mixing native and non-native plants in their gardens, observing birds had become part of their daily routine and encouraged them to expand their native plantings (Head & Muir, 2006). Nassauer (1995) observed that people may value songbirds but are unlikely to value the woody habitats they require. It may be the case, however, that if urban residents value songbirds and are made aware of the habitats they require, they may change their appreciation of such landscapes and in turn, their gardening practices. Given the extent of residential yards and gardens in cities, wildlife gardening practices can provide a system of habitat patches and corridors throughout the urban matrix (Lerman & Warren, 2011).

Wild birds may form the main wildlife interaction experienced by urban residents on a daily basis (Cox & Gaston, 2015). The majority of these experiences appear to be positive, with negative aspects of urban birdlife not perceived as significant problems (Belaire et al., 2015). The importance of woody areas as a habitat for wildlife was found to be the most significant reason given by interviewees in Wales for their enjoyment of woody habitats and desire to protect them (Henwood & Pidgeon, 2001). In Hamilton, New Zealand, 34 per cent of gully owners thought it important to ‘bring back’ native birds (Jay & Stolte, 2011). Thus, native birds, and the habitat features they require, may act as powerful advocates of wildlife gardening and urban ecological restoration.

Ecological restoration: A potential solution to the vicious cycle of the extinction of experience

Urban restoration ecologists and conservation biologists have grappled with the problem of the vicious cycle of the extinction of experience as they seek a way to engage people in the protection of native flora and fauna. To a New Zealand ecologist, large expanses of mown lawns bordered by introduced tree species may represent the interchangeable, artificial environments characterized by Relph’s notion of placelessness. Urban restoration ecology recognizes the importance of dwelling as it seeks to create distinct places dominated by native vegetation and animals in the areas where the majority of the world’s population lives, works and plays (Rosenzweig, 2003). Creating or re-establishing these spaces in cities enables native nature to become a part of people’s daily routines and lives. In effect, ecological restoration has a dual purpose. It aims not simply to plant trees and restore native habitats, but also to cultivate people’s emotional bonds with a place and with native nature.

Gobster (2000) views the practice of restoration, which demands of practitioners both knowledge of ecological functioning and a willingness to get their hands dirty, as a way of re-establishing a relationship between people and nature. Like farmers or gardeners, restoration volunteers give their time and energy to tending the land. The act of physical engagement with the landscape coupled with a sense of responsibility and a duty of care are all components of dwelling identified by Seamon (1985) and Relph (1985). According to Jordan III (2000, p. 32), restoration is a way of generating an emotional

commitment to the landscape and can further provide opportunities for exploration, observation and understanding the natural world:

[*Restoration*] provides what seems to me a plausible basis for the creation of the large, deeply committed constituency that will be needed to ensure conservation of large natural – or wilderness – areas. It does this, moreover, with the material close at hand – the landscapes that most people actually inhabit, the urban parks, rights-of-way, stream corridors, and vacant lots of the urban and suburban landscape that have, or can be made to have, a powerful symbolic relationship with larger and more remote areas.

Finally, the community-based activity of ecological restoration allows people to participate in conservation of nature close to home. The satisfaction of participating in restoration projects and seeing nature thrive can counter the feelings of helplessness and apathy that often result from the enormity of the environmental crisis (Kaplan, 2000). It is further assumed that the knowledge of native nature and emotional connection to it, generated by restoration activities, will result in volunteers supporting wilderness conservation beyond the city limits (Jordan III, 2000).

Apart from providing volunteers and practitioners with hands-on involvement in revegetating an area, the practice of urban restoration can be presumed to benefit those city residents who regularly use these restored spaces, by providing them with everyday contact with native nature. Such daily, routine, intimate experience, especially during childhood, is seen as essential in forming emotional bonds between people and their environment (Hinds & Sparks, 2008). Thus, urban ecological restoration can be employed as a strategy to reverse the vicious cycle of biological poverty and the extinction of experience by tapping into people's emotional bonds with the places where they live. This approach may take considerable time to establish, but may offer cumulative, self-sustaining change over time. By re-establishing places in cities that are grounded in the landscapes and natural environments of Aotearoa, New Zealand, we may strengthen urban residents' attachments to these places and re-ignite nature wonder and appreciation. Accordingly, the benefits of urban ecological restoration are enduring, multifaceted and extend well beyond the confines of a restored site, in offering the potential for improving our quality of life and recovering biodiversity within and beyond urban contexts.

The aims of the following study were to investigate the current status of ‘connection to nature’ in residents of a New Zealand city, looking for evidence for the extinction of experience. A further aim was to explore the lived experiences, meanings and personal connections linking people to local greenspaces and the complex personal factors that shape individual inclination to engage with different kinds of urban greenspace. I further attempted to differentiate the benefits participants derive from bush parks dominated by native vegetation and those from lawn parks dominated by introduced vegetation. I posed the following guiding questions:

- 1) What elements of urban nature do people notice and appreciate?
- 2) Are people gardening to attract native wildlife? Do people see gardens as potential habitat for wildlife?
- 3) Does the type of frequently visited park influence perceptions of nature?
- 4) Which interactions and wellbeing benefits are supported by different kinds of greenspace?

2.2 Methods

2.2.1 Participants

The qualitative study was conducted from March to May, 2017, in Hamilton, New Zealand. Twenty-one Hamilton residents participated in the research – 7 people for each of the three categories: restoration volunteers, frequent users of parks dominated by native vegetation (from now on referred to as bush parks) and frequent users of parks dominated by non-native vegetation (referred to as lawn parks). Restoration volunteers were recruited by directly contacting leaders of community groups working in Hamilton, and park users by distributing flyers to houses adjacent to parks and by word of mouth. Sixteen participants were women and 5 were men. Ages ranged from 28 to 73. The majority identified as Pākehā, with 3 Māori participants, 1 Fijian and 1 Indian.

2.2.2 Data collection and analysis

Semi-structured ‘go-along’ interviews (Carpiano, 2009) were used to explore each participant’s personal experience of nature in their neighbourhood park and home garden. Interviewees were asked to give the primary researcher a tour of the park they

visit regularly, followed by a tour of their garden. During the tours, participants were asked about the plants and animals they encounter in their parks and gardens, and about the importance of these spaces and the ways in which they engage with them. Interviews lasted between 30 minutes and 2 hours, were audio-recorded with permission, and were later transcribed verbatim. All the participants were given pseudonyms to protect confidentiality.

An interpretative phenomenological approach was used to analyse the data (Smith *et al.*, 2009; Pietkiewicz & Smith, 2014). Phenomenology seeks to understand a situation or phenomenon as experienced by an individual in order to discover its deeper meaning in the context of the whole of human experience (C. Baker, Wuest, & Stern, 1992; Van Manen, 1984). According to Moustakas (1994, p. 13), ‘the aim is to determine what an experience means for the persons who have had the experience and are able to provide a comprehensive description of it’. Thus, in phenomenology, the source of data are people who have lived the experience being investigated and the focus is on a subjectively lived world – a world experienced by individuals through practical engagement with it (Eatough, 2012). The researcher attempts to see the world and the phenomenon under investigation from their participants’ point of view. The key objective is seeking to understand how participants make sense of their lifeworlds (Pietkiewicz & Smith 2014), while also recognizing that the researcher brings their own past experiences and preconceptions to the analysis. The ‘interpretative’ aspect of the method acknowledges that both the participant and the researcher are involved in the analysis – the interviewees through their verbal accounts and the researcher through their interpretation of these accounts. The study is idiographic, in that the main goal is an in-depth exploration of each individual’s lived experience – the process is highly time-consuming and therefore the sample size is relatively small. This method was chosen because the research is exploratory and the goal was a comprehensive, in-depth exploration of individuals’ experiences of urban nature. Consequently, no specific hypothesis or theory was being proposed or tested.

Following transcription, each individual interview was examined in detail, themes were identified, and a summary of each case study was written up. The next stage of the analysis involved identifying shared themes across the accounts. Five main themes

and multiple sub-themes emerged from the interviews. An overview is presented in Table 2.1, and the themes are described and discussed in detail in section 2.3 below.

Table 2.1: Overview of the main themes and sub-themes that emerged from the interviews

Themes	Sub-themes
Tensions between native and non-native in parks and gardens	Creation of separate spheres for native and non-native Competing values held for gardens – productivity (fruits and vegetables), colour (typically introduced flowers and leaves), unique New Zealand natives, compensation for global habitat destruction
Tension between tidy and messy and how much of each is acceptable	Lawns as unquestioned necessity Trees as source of conflicting emotions and values
Bird encounters in the city	Enjoyment of birds does not result in concrete action (e.g., protection from cats, gardening for birds) but untidiness in gardens is tolerated if seen to benefit birds Source of enjoyment, part of daily life, at times taken for granted Desire to see more native birds but also enjoy non-natives, need for spaces in cities for native birds
Values held for different urban greenspaces	Gardens: relaxing, observing nature, learning, enjoyable work, relief from stress, reconnect with surroundings Native bush parks: special, privilege, observing nature, escape from city, experience with all senses, “messy” is acceptable and desirable, place for rest and recovery, walking Lawn parks: physical activity, child play areas, sport, walking, social engagement, interaction with neighbours, meditation, escape from work and home chores Nearby greenspaces: source of pride, sense of ownership, duty of care which led to concrete action (involvement in restoration, pulling weeds, picking up rubbish) Tensions over park use: taking more than your share, conflicting views of what is a desirable plant, allowing children to play vs protecting plants, perceived safety

2.3 Results and Discussion

'Having it all': the tension between native and non-native in the garden and park

While quite a few interviewees voice a preference for native plants, this does not necessarily result in the preferential planting of natives, and most gardens tend to be dominated by non-natives [*personal observation*]. Similar findings were described by van Heezik et al. (2012) in their study of Dunedin home owners, where residents' gardens were primarily dominated by exotic species despite the participants' voicing a preference for native plants, thus highlighting the gap between stated values and practical choices (Kollmuss & Agyeman, 2002). Similarly, Doody et al. (2010) found that Christchurch residents claimed to value native plants, but would remove these same species as weeds if they grew as self-dispersed seedlings in their gardens. Quite a few of my participants have areas in their gardens specifically designated for native plants – Siobhan had decided to plant only natives in her front yard, Ngaire had a section along her house for shade-tolerant bush species, Artair sees his gully as an area that should be native, and Breena digs up all the native seedlings that appear on her lawn and moves them to one corner of the garden. This allocation of separate spheres for native and non-native plants seems to be one way residents resolve their desire for the colourful flowers and autumn leaves of introduced plants, as well as a productive vegetable garden and fruit trees, with their conviction that planting natives is important. This desire to 'have it all,' as two respondents described it, was well illustrated by Balfour,

It's a bit of a jumble but it works ok [...] I do have a preference for natives because it's nowhere else. If we don't look after it and care for it, no one else is going to [...] But I do enjoy deciduous trees as well, yeah, non-natives. It's quite nice looking through here, you get a mix of natives and non-natives, and it's quite nice in an urban setting. But I don't like going into a reserve or something and seeing other, you know, seeing maples or something growing [...] I've got a garden that's a combination of deciduous and native, and that's lovely too, I like it a lot really.

Balfour identifies the uniqueness of New Zealand plants as the reason for his appreciation for them. Restoration, or planting of native species, is often advocated as a way to re-establish regional and national identity (Feagan & Ripmeester, 2001). In

New Zealand, native plants started becoming fashionable in the second half of the 20th century, a trend that Longhurst (2006) and Jay and Stolte (2010) attribute to Pākehā New Zealanders' changing perceptions of their country from that of an English colony to that of an independent Australasian-Pacific nation proud of its unique biological heritage. Balfour further describes a sense of personal responsibility and duty of care for the preservation of these unique species. Participants in studies of urban green roofs (Loder, 2014) and sustainable gardening (Kiesling & Manning, 2010) voiced similar concerns over human destruction of the environment, and saw these spaces and gardening choices as compensation for this destruction.

Balfour's quote raises another interesting point – that the mix of native and introduced plants is seen as acceptable in an urban setting, but undesirable in bush remnants. This same mentality is played out on the finer scale of the city with certain areas, such as gullies and patches of remnant forest, seen as spaces that should be dominated by natives, while in gardens and parks respondents believe it is acceptable to have a mix. Similar views were found in a study by Selge et al. (2011), where certain non-native species were acceptable when found 'in the right place', but undesirable when found in areas viewed as pristine.

The colourfulness of introduced flowers and leaves was brought up by many of the participants as adding variety to a garden. Particular native plants, such as kōwhai, hebes and kākā beak, are specifically chosen for the garden because of their pretty flowers. Balfour described disagreements he has with his wife over what to plant, with him preferring natives but his wife asserting that they have no colour and instead advocating for introduced plants with pretty flowers. Caitrin was torn between her enjoyment of autumn colours and knowledge that the trees are not from New Zealand.

I get that all of these are European trees, or I don't know, American [...] We go through fads in trees. But I guess that's the problem, isn't it? People mix indigenous stuff, here in New Zealand it's kind of imported stuff there [*in the park*], and I don't know what to think. So I just carry on walking and think, that's a nice tree! I try not to get political about it [...] There's a tension there, like we were saying, there's a real tension! But, you know, I think you've just got to compromise, you know, a tree is a tree, and we've got to be really grateful that trees are here and sort out the politics later on.

Large trees in parks are highly valued, no matter their country of origin, and cutting them down is invariably seen as a shame, even if they are being cut down to make way for natives. Seonaid is particularly passionate about the importance of large old trees, describing their destruction as ‘a huge shock, a massive thing to do, tragic and distressing,’ which leaves her feeling ‘bereft and stunned.’ She believes their loss detracts from the character of the neighbourhood and further stressed their importance as habitat for birds.

Culturing the wild and woolly look or wild and woolly without the culture?

While reflecting on the limited amount of native plants in their gardens, many participants conclude that they do not have enough room for more planting. What they often mean by this, however, is that there is no more room in the thin strip of trees and shrubs bordering their lawn. Lawns are generally accepted as an obvious and unquestioned necessity, preferred for their perceived ease of maintenance and letting in of light. According to Feagan and Repmeester (2001), ‘the lawn has become deeply internalized as the most appropriate form of organizing private green space’. The large trees so greatly valued in urban parks become highly problematic in the garden and are cut down if they are seen to block the sunlight or threaten to fall on the house. One participant’s garden had originally been all bush,

When we first moved into this house it was almost as if it was in the bush [...] That whole front yard was just totally covered in bush right up to the windows. So most of what’s there we’ve actually pushed back a bit [...] because it was way too dark for us, so we took a lot of it out but kept some of the stuff around the edges [...] it was just far too dark for us and also there was no grass, so that’s why we did what we did.

This cultural norm of lawn bordered by trees and shrubs was questioned by a few interviewees who were interested in the concept of permaculture or organic gardening, which they often spoke of as gardening to provide food for insects as well as people, and more rarely wildlife in general, throughout the year. Kendall describes herself as not a fan of lawns, which she sees as masses of mown monoculture, preferring to garden with nature and looking at the landscape in terms of food for people and

animals. Similarly, Kaelin believes it is important not to be too tidy, declaring nature should not be constrained to straight lines and clean borders. She rarely mows what little grass is left in her yard, declaring long grass to be important for such wildlife as skinks, worms and wētā. She describes her gardening style as ‘muddly’ and designed to attain ecosystem balance with complex invertebrate communities that will in turn attract birds. She sees the ‘lawn garden’ with perimeter trees as barren and lacking in pollinators and predatory insects.

There appears to be a strong tension around the concept of what is tidy or messy and how much of each is acceptable. A respondent would speak of the importance of allowing things to look a bit wild in the garden one minute, and apologize for its untidiness the next. Due to injury and family circumstances, Cian is unable to spend as much time in her garden as she would like and her comment on the resulting ‘messier’ garden reveals this tension – ‘usually we culture the wild woolly look, now we’ve got the wild woolly look without any culture.’ Freeman *et al.* (2012) note how the ‘informality’ of nature appears to be at odds with garden ‘culture.’ According to Longhurst (2006), gardens are ‘improvements’ on nature – they are places where nature is controlled, ordered and altered to fit widely accepted social conventions of what a garden should look like. Studies indicate that people value areas they perceive to be neat and safe, and owners of ‘untidy’ or ‘messy’ gardens can be accused of being lazy and irresponsible (Nassauer, 1995a; Feagan & Ripmeester, 2001). Nassauer (1995:161) comments that ‘ecological quality tends to look messy [...]. What is good may not look good, and what looks good may not be good.’ Lindemann-Matthies and Marty (2013) found that although gardens described as wild and chaotic tended to be species-rich, they were not considered to be attractive. Thus, city residents potentially interested in creating ‘messier’ gardens that would benefit native species will come up against the social norms of large expanses of mown lawns and neat, tidy, colourful borders of exotic plants.

Trees were a constant source of conflicting emotions in the present study, described by one participant as a blessing and a curse, dropping branches and leaves, blocking gutters and shading houses or vegetable gardens. Balfour referred to the constant fight between having lovely things and beautiful trees but keeping them from taking over.

Similarly, Head and Muir (2005) found that while native trees were often highly valued, they were seen as ‘out-of-place’ when too close to the home.

The interviews revealed a conflicted relationship between what people claimed to value in parks and gardens and their actual gardening choices. Respondents greatly appreciate indigenous plants in bush parks, but in their own gardens their conviction of the need to promote New Zealand’s unique biodiversity conflicts with their desire for colour, beauty, seasonal changes, variety and productivity. Interviewees’ perceptions of what a garden could be is further constrained by the cultural convention of manicured lawns with tidy borders of colourful exotic flowers. This cultural convention that equates neat, tidy gardens with responsible, hard-working residents makes it difficult for people to reconcile ‘messy’ nature with ‘tidy’ gardens. People have a strong sense of where various plants belong – a native plant might be highly valued in a bush park or the appropriate shady spot in the garden but become a weed when it self-sows on the lawn. Large trees are highly valued in parks but become sources of constant frustration in gardens. Thus, the value of a plant does not depend solely on its country of origin, but also on the type of space it occupies, and the priorities or uses people see as acceptable for that space.

Bird encounters in the city

In a number of instances, features that were described as untidy, scruffy, messy or weedy are tolerated if they are seen to benefit birds, a finding which challenges Nassauer’s (1995) suggestion that residents may value songbirds in suburban environments, but are unlikely to value the woody habitats these birds require. Deoiridh leaves ‘scruffy-looking’ dying sunflowers, zinnias, asters and marigolds because they attract finches. She described this as her concession to bird feeding — ‘the fact that I leave them here is for the birds, I couldn’t say that I actually grew them for the birds.’

Similarly, Keira was going to cut off dead sunflower heads until she realized the goldfinches were enjoying them. The flowers had originally been put in because of their pretty colour; she had not realized the birds would be attracted to them. Aindrea leaves fallen branches and leaves, putting them in the garden or mowing them into the

lawn – ‘all deposited there to feed the birds’ — and Siobhan decided to leave in flaxes that she described as messy-looking because they were attracting tūī.

While a few studies have suggested that people who enjoy bird observations in their yards will increase their wildlife gardening efforts to attract even more birds (Belaire *et al.*, 2015), none of this study’s participants designed their gardens specifically to benefit native birds. The attractiveness of certain features to birds, both native and non-native, was often mainly a happy coincidence. Occasionally, a kōwhai or flax might be planted with the hope of attracting tūī. Those participants interested in permaculture and gardening organically tended to focus more on insect diversity and the presence of pollinators, although this in turn is seen as being beneficial to birds.

Interviewees’ perceptions of neighbourhood birds are overwhelmingly positive, irrespective of whether the bird was native to New Zealand. The few exceptions viewed as undesirable were magpies, mynas and starlings, which are described as aggressive ‘bullies’ of other birds, as well as sparrows and occasionally blackbirds, which are seen as overabundant pests. Seeing, hearing and observing birds was mentioned by the vast majority of interview participants, for whom birds are a welcome, though at times taken for granted, aspect of their daily lives, supporting previous claims that wild birds may be the most commonly encountered wild animal in the city (Cox & Gaston, 2015). Many participants commented on the return, or the increasing numbers, of tūī in Hamilton, with animated stories of the first tūī sighting in the city, an occasion often warranting a blurry photograph. Participants also voiced a desire to attract tūī into their own gardens and some fantasized about being woken by tūī song in the morning. When reflecting on birdsong, tūī and morepork calls were mentioned most frequently, possibly due to the very distinctive nature of their calls, which were described as recognizable, familiar, wonderful and lovely. One participant spoke of hearing a kiwi calling at night at her ‘bach’ [holiday house] on the Coromandel Peninsula as ‘a moment’s jubilation.’

A number of respondents mentioned enjoying watching birds and their behaviour – Taan and her partner put out sugar water and bird seed every few days, ‘and boy,’ she concludes, ‘it’s fabulous, so many birds, we could just sit there and watch them all day.’ Similarly, Cian comments that if she and her spouse are staying in, they will fill

up the bird feeder so they can just sit and watch the birds. Ngaio listed one of the benefits of building on a porch that she and her partner can sit outside at night watching the birds come in to roost. Cian was particularly observant when it came to bird use of her neighbourhood; during the tour of the park, she pointed out hollow trees where she had found kingfisher nests, described her experience seeing two grey warblers feeding a shining cuckoo chick as ‘one of those great thrills,’ and recounted coming across two moreporks sitting on a branch with their chicks as a stunning, wonderful encounter. Moreporks appear to be a highly charismatic bird that interviewees are excited to encounter and photograph.

Two respondents were unable to name the birds they saw, with Caitrin asserting, ‘I can see the birds but I don’t know their names’. When asked what birds he sees in the park, Inoke responded, ‘those white things [*spur-winged plovers*], obviously you’ve got your normal, what are they called, those little brown things [*house sparrows*].’ When asked the same question in his garden, Inoke mused, ‘in one sense, I suppose, you kind of just take birds for granted. It wouldn’t be until they were gone that you’d know that they weren’t there.’ Despite the taken-for-granted nature of these daily bird encounters, Inoke spoke of trying to attract fantails to their garden and described swallows nesting above their door, swooping and clicking above their heads, as an experience he and his partner enjoy.

The birdlife encountered in residents’ gardens and parks was viewed as ‘quite a diverse range’ or alternatively as ‘relatively little,’ and consequently ‘a bit boring’ and ‘quite disappointing.’ Those who described the birdlife as leaving much to be desired tended to focus on native bush birds, of which they had seen only tūī, fantails and silvereyes. Even those who think the birdlife is ‘wonderful for a city’ comment that the main birds they see are blackbirds, sparrows, starlings and mynas, and express the desire to see more native birds, especially kererū and bellbirds, with one respondent going so far as to wish she could go for a walk in Hamilton and hear kokako, kākāpō or kākārīki. Two participants contrasted the amount of birdsong they heard in their gardens at home to the perceived silence of the bush. Describing tramping in Pureora Forest many years back, Kendall lamented, ‘you wouldn’t hear a bird, and that just breaks my heart.’ Similarly, Laren spoke of frequent tramps around the South Island where the bush was

‘dead quiet, literally dead – dead of birds,’ which she contrasts with the ‘huge amount of bird sound at home, but not in the bush where they should be as well.’

While in general participants seemed to appreciate non-native finches and song thrushes as much as they did the native bush birds, a few respondents referred to the importance of creating spaces in cities where native birds could flourish. In their survey of Hamilton gully owners, Jay and Stolte (2010) found that 34% of respondents thought it was important to ‘bring back’ native birds. There is a tension, however, between wanting to ‘give native birds a chance’ and still enjoying pretty non-native species. Artair, while weighing up the benefits of bush parks and open lawn parks, deliberates:

We definitely want to encourage these [*native birds*], we’ve hammered these species really hard the last 100 years, whatever, and really what we’re trying to do is give them a chance [...] I wouldn’t go as far as to say that I want all parks to look like this [*bush remnant*], because I like these parks too [*lawn park*], so basically you want to have it all, don’t you? I mean, because all these other birds [*non-natives*], they’re actually quite beautiful too.

Reflecting on the same photographs of parks, Balfour concludes, ‘we’ve got to provide lots of homes for natives and these birds [*non-natives*] can live anywhere.’ Both respondents agree that they would only feel negatively towards non-native birds if they were shown to displace or compete with natives. Siobhan declares that it is very important to her to see native birds, like tūī and fantails, and encouraging them was part of the reason she wants native bush in the city – ‘for all the special birds.’ Kaelin and Kendall both stressed the importance of having food sources for wildlife in urban parks and gullies, as otherwise these areas may appear attractive to people but are incapable of supporting wild animals. The idea of sharing city spaces with wildlife was described as ‘comforting’ by Kaelin, as she talked of the bats that move through her gully. Inoke, when speaking of the birds he saw in the park comments,

Because you see, for us too, the birds and that sort of stuff, not that I could ever suggest to you that I’m into ornithology or know what birds are what, but you know that type of thing, the wildlife, is part of this place, and it makes it what it is for us, you know?

Thus, while Inoke admits his knowledge of birds is limited, he assumes not only that wildlife is present in the city, but that it has a place there. He further asserts that the wildlife of his neighbourhood park and garden is integral to the meanings he holds for these places, and therefore for his own sense of place and identity.

Respondents fell into roughly two camps on the question of cat ownership and the impact of cats on native birds. The first group was, as one interviewee put it, ‘philosophically against cats’ because they saw them to be incompatible with native wildlife and believed a complete mindset change was needed on cat ownership. The second group comprised the cat lovers who were torn between the companionship and enjoyment they got from their pet cat or cats, and their desire to see more birds in their garden. Participants dealt with these conflicting priorities in various ways. One interviewee built a large outdoor enclosure that connects to her house, guarantees the safety and happiness of her cats, and ensures the neighbourhood skink and wētā³ community do not end up on her pillow. Other respondents reasoned that their cat was too old, was not a good hunter, had never before caught a bird, or would never catch a native bird. A few participants put bells on their cats but do not believe them to be particularly effective.

Our respondents generally showed a strong affinity for birds, ranging from active feeding and bird-watching to the taken-for-granted enjoyment of seeing and hearing birds in their parks and gardens. Bird encounters are often a great source of joy, excitement and delight. This enjoyment of birds, however, does not necessarily result in interviewees taking concrete action in their own gardens to protect birds from pet cats or to attract and provide habitat and foraging opportunities for birds, and the attractiveness of certain features of gardens to birds was usually simply a happy coincidence. This finding once again highlights the gap between what we value and what we do. Valuing birds did not lead people to actively work to accommodate, attract and protect birds in their garden.

³ Wētā is the common name for a group of about 70 insect species in the families Anostomatidae and Rhaphidophoridae, endemic to New Zealand. Wētā are related to grasshoppers, locusts, crickets and katydids, all members of the order Orthoptera.

The value of gardens and parks in urban areas

For passionate gardeners, the garden is a private space for relaxing and enjoyable work, as well as a place to wander and observe the fruit of one's labours. Longhurst (2006) describes gardening as a labour of love – an activity where leisure and work merge. In a study in Leicester, UK, Bernardini and Irvine (2007) found that gardeners most highly valued the relaxation, peace and opportunity for meditation they obtained from gardening, followed by a sense of privacy and personal contact with nature.

Siobhan walks around her garden at least once a day, 'just to have a wee look and admire things, see how things are ripening up.' Ngaio spends 4 to 5 hours a week gardening, but emphasizes that she does not see it as a chore, as she enjoys being outside and learning from observing the changes in her garden. 'I'm a great believer, just from a gardener's perspective, that if you can walk around in your garden every day, you just learn so much, just by observing.'

For Cian, gardening is a way of relieving stress and reconnecting with her surroundings. 'I think it keeps you grounded; avoids depression, all those things that people talk about. It's like going for a walk, if you go for a walk, you can't be sad.' Gross and Lane (2007) claim that gardening provides a positive distraction and escape from the stress and anxiety of daily life in the city. Less active gardeners in this study still derive enjoyment from observing their garden through the window or sitting outside while studying.

Restored gullies, patches of remnant forest, and older native plantings along the river were described as special places that are a privilege to have, and provide a relaxing and restorative escape from city life. These greenspaces, dominated by native vegetation, are the ones respondents most frequently identify as places to sit peacefully and observe nature. Kendall enjoys watching what goes on in the forested section of her park, from decomposing logs to morning dew on spider webs. She is attentive not only to the sights but to smells and sounds as well, speaking of how lovely the park smells at night, and describing it as an aspect of the bush that most people miss out on, due to safety concerns. Her small patch of forest is appreciated as an escape from the city. 'I love the fact that when you come down here, apart from the sound, you wouldn't know you were in the city.' As was found in previous studies (Loder 2014, McDonald & Schreyer

1991, Fredrickson & Anderson 1999), these greenspaces and the nature they represent are valued as something to immerse oneself in and experience with all the senses – touch, smell and sound, as well as sight.

Ngaio expresses very similar feelings, describing the remnant forest near her house as a special place providing a beautiful vista of the river, where she enjoys watching the morning mist rise off the water and the changes to the park over the seasons. One of the qualities of the bush that she identifies as providing respite from city life was the unmanicured nature of the forest. ‘I like the idea that we can actually be living in the city and go and visit a piece of bush, actual bush, rather than something manicured.’ Unlike Loder’s (2014) study of perceptions of green roofs in Chicago and Toronto, my respondents did not speak of bush fragments as messy, unkempt or ‘wild looking.’ Loder (2014) proposed that urban residents disliked the messy aesthetics of the prairie-style green roof in part because of their expectations of what kind of ‘nature’ was acceptable in cities. Hamilton residents, however, appeared to appreciate the unmanicured, ‘wild’ nature of bush fragments in their city, although this view did not hold for their own gardens. Location was once again important – bush parks were places where nature was allowed to be ‘messy,’ but gardens must conform to a more ‘tidy’ social ideal.

Balfour views his park’s remnant forest as special because of the age of its trees and the scarcity of these patches in Hamilton, making them worth preserving and enhancing. He too was in the habit of observing nature. ‘I am interested in the bush and I look at it and look hard to see what I can see, any changes or any new birds or insects.’ Breena spoke of walking through her restored gully park to feel good and recharge. ‘I just love this, this is a special bit because it’s got the canopy and it’s always so peaceful in here, this is one of my favourite parts,’ she commented, walking through a stand of 60-year old kahikatea⁴. Kaelin brought up these same themes of the restorative value of observing nature in the replanted gully behind her house.

You can be down here in the right time of the year and you think, where am I? It’s not the city, it’s just ringing with birdsong [...] I can come down here, you know, wander along with my cup of tea in the morning when it’s just me and the stuff going on here [...] Often I’d just come

⁴ *Dacrycarpus dacrydioides* or kahikatea is a member of the podocarp family and endemic to New Zealand.

down, bring a cup of tea and just sit, just see what's happening down here. It's all good. And the air is different here, it smells different, you know, there's lovely wind and breeze coming through and there's no houses and it's just marvellous.

The findings from the qualitative study for this PhD support Kaplan's (1995) Attention Restoration Theory, which proposes that the natural environment facilitates rest and recovery from mental fatigue. It is further worth emphasizing that these themes of relaxation, close observation of the natural world, and escape from the rush of city life were brought up principally in reference to parks dominated by native vegetation. Nonetheless, previous studies have found that gardens also provide opportunities for observation of nature and reduce the sense of 'urban claustrophobia' (Bernardini & Irvine 2007). Shwartz et al. (2014) conclude that urbanites may claim to appreciate and benefit from greater levels of biodiversity, while simultaneously showing a poor ability to recognize and experience this diversity. This qualitative research, however, indicates that urban residents are responding to the greater complexity and species diversity of restored and remnant vegetation, by seeking out these areas for relaxing and observing the natural world. This assessment supports Loder's (2014) findings that 'wilder' green roofs elicit responses such as increased fascination with watching what is going on in the garden and reflection, which encourages creative thinking.

In their study of a citizen science butterfly monitoring project, Cosquer *et al.* (2012) suggest that city dwellers have lost the habit and the practice of observing nature. For my study participants, this was not necessarily the case, and the presence of gullies and remnant vegetation in close proximity to their homes is all the encouragement they need to get out, wander through their park and enjoy their daily encounter with the natural world. With results similar to these, Gill and colleagues (2009) found that many interviewees who are regular visitors to the Croom bushland south of Sydney are closely observing the bush area and noting the presence and absence of plants and animals, as well as daily and seasonal changes.

While bush parks are appreciated as peaceful, restorative refuges where nature can be observed and enjoyed, lawn parks are more frequently seen as spaces that allow for physical activity by providing children's play areas, as well as fields for organized

sports, such as soccer, cricket and ultimate frisbee. Whereas only one participant uses her park for a sport, the rest commented on the importance of these spaces for people who might want to engage in physical activity, but concluded that they did not use them for that purpose themselves.

Both types of park are used extensively for walking, and dog-walking in particular, an activity that appeared to facilitate social interactions which provide respondents with a strengthened sense of belonging in their neighbourhood. The role of dogs as social ice-breakers has been noted in previous studies (Power 2013). Dog-walking was identified as the main reason many interviewees use their parks regularly, often at least twice a day, and this was invariably seen as a positive aspect of dog ownership. Inoke spoke of how walking his dog forces him to get out, which he describes as a nice de-stressing activity. Dog-walking further provides an opportunity to meet other dog owners and often results in pairs or groups of neighbours meeting up regularly to walk their dogs. According to Power (2013), dogs shape our relationship with our neighbourhood by encouraging people to get outside on a regular basis and providing a neutral topic of conversation or a point of connection for other dog owners.

The benefits of regular walking in nearby parks brought up by interviewees included exercise, social engagement and interaction with neighbours, an opportunity to reconnect with family and partners, and the de-stressing, relaxing effect of walking in green areas. Caitrin appreciates the benefits of solitary walks. 'I like it, it's meditative, sometimes I take myself for a walk, it clears my head, so I find that very, very useful'.

Walking further allows respondents to get away from the routines and chores of home and work. Inoke, who walks with his wife and dog daily, comments: 'My wife and I, this has been our ritual thing that we've always done, walked around the park, you know? [...] It's just a good time for my wife and I to be able to talk, walk and talk and catch up, and I suppose it gets us away from our work at home'.

Previous studies have similarly emphasized the social benefits of access to greenspaces, with parks seen as providing a place for the community to come together and create a sense of neighbourliness and cohesion through the 'repeated-ness of meeting the same faces' (Bernardini & Irvine, 2007; Home *et al.*, 2010). Parks have also been found to provide an opportunity for what Herzog terms 'being away' – 'the

psychological feeling of being away from the usual routines and demands' (Herzog *et al.*, 2003; Gill *et al.*, 2009).

Many respondents enjoyed seeing people use their parks. Families having picnics, the Asian community gathering ginkgo fruit, children jousting with bamboo sticks or diving into the river, Muslims from the mosque playing soccer after service – all these and many more were listed as positive experiences enhancing residents' appreciation of their parks. Laren summarizes: 'what we love about this park is that it's so well used, so kids over there, cyclists coming through [...] sporting events in here and all sorts of people come and go, which is great, I just love that. It makes living in the city so wonderful, to have such beautiful greenspace.'

Cian takes particular pride in her engagement with her neighbourhood community, commenting that she knows all of the people in the area and has met most of them through spending time in the park. Objects in parks further serve as memorials to the past – trees have been planted in memory of local residents who have died and benches are inscribed with their names.

Interviewees often expressed great pride in their neighbourhood parks and felt a sense of ownership and responsibility for their upkeep. This duty of care manifests itself in small acts, such as collecting rubbish and pulling out weeds, or in greater time commitments, such as restoration, providing an example where values lead to concrete action. The majority of restoration volunteers interviewed are involved in activities such as planting, weeding and pest control in their neighbourhood park. A few respondents engage in 'guerrilla plantings' – when native seedlings come up in their gardens they take them down to the park. The garden versus park dichotomy became blurred for people living adjacent to parks, as they tended to view nearby greenspaces as an extension of their own backyard. When asked if he was interested in conservation, Inoke replied that although he was not the sort of person who tends to go on protest marches,

If somebody came over here and started messing with our park, then you would probably see something different, you know what I mean, I suppose in a way, you'll notice I just used the words 'our park,' so in a way my wife and I have taken ownership of the park because we see it as our, almost like our backyard. And I don't seem to think that's

necessarily a bad thing if people who live around it take some form of ownership because then they care about it.

Breena, whose house backs onto the restored gully where she both walks frequently and is involved with the local restoration group, voiced a similar opinion,

I do have to stop myself thinking that I own it. If you know what I mean, not that, it's just I get concerned about it, you know, sort of take ownership, I suppose, when you live near or you work in it.

Proximity to the park is an important factor in developing this sense of ownership, as well as frequent use in the form of regular walks or involvement in restoration activities, stressing the need to provide greenspaces within walking distance of residents' homes, schools and workplaces. In a study of natural area users in Michigan, Ryan (2000, p. 218) found a strong relationship between use of the site and attachment to it, and he concluded that 'building these connections between people and the natural environment is essential for creating advocates for natural area restoration and preservation'. This relationship between attachment to a place and intensity of use has been reported in other studies, with the caveat that familiarity with a place will not necessarily result in attachment to it if the space does not meet one's needs (Ryan, 2005; Bernardini & Irvine, 2007).

Tensions over park use

While the majority of participants appreciate their parks as a space to engage with their community, tensions can arise when residents have conflicting views about the use of an area. One participant was affronted when a Chinese group came through and cleared out patches of mushrooms – she saw this as an end to the New Zealand tradition of only taking enough for one's own family and leaving some for the rest of the neighbourhood. Another respondent described the clearing of a section of bush park that had once been 'infested' with hydrangeas as a positive act of restoration, while that same event was seen as disappointing by another participant who described the

hydrangeas as absolutely magnificent, attractive to butterflies, as well as a memorial to the friend who planted them and had since passed away.

Some commented on the tension between the importance of allowing children to play in bush parks and the need to protect native plants. Allowing children to explore and play in natural areas has been advocated as vital for combating the extinction of experience (Louv, 2008), and Taylor *et al.* (1998) found that children are more likely to engage in creative play in outdoor spaces with a greater amount of vegetation. Such tensions highlight the difficulties urban planners and managers may experience in facilitating public use of urban greenspaces, while simultaneously protecting native plants and animals (Ryan, 2005).

Issues of perceived safety were brought up by some of the participants as a deterrent to park use. Women spoke of feeling slightly uneasy or unsafe walking through certain bush parks, and husbands worried on account of their wives and children. When deciding on a home to rent, Awhina looked into surrounding parks not in terms of their attractiveness, but of the number of ‘dodgy’ people hanging around them. Jubilee Bush, the forested section of Claudelands Park, was seen by many as unsafe, and women in particular would not go through the bush when alone or after dark. Thus, the vegetational complexity of bush remnants and restoration plantings, which benefit native plants and animals, can act as a deterrent to park use if it is seen to provide cover for potential crime. Ryan (2000) found that fear of wilder areas was associated with a lack of ecological expertise and tended to be more prevalent among women, while respondents to a survey in Sheffield identified vandalism and neglect as much as dense vegetation as reasons for a park to be seen as unsafe (Fuller & Irvine, 2010).

In my study, the potentially threatening people were identified as layabouts, the ‘homeless element’ and groups of teenagers, though one interviewee concluded, ‘What does a person who might do you harm look like? It’s anybody’s guess, really.’ Many spoke of vandalism, such as setting fires, burning tree ferns, tagging trees (i.e., marking them with graffiti), stealing or destroying wētā boxes⁵ and pulling up and breaking young plants. A few respondents believed that putting in more lights would make the bush parks safer, pointing to a source of possible tension between human and wildlife

⁵ Wooden shelters built to provide habitat and protect wētā from predation by introduced mammals

needs, as light pollution in bush parks could negatively affect native nocturnal species such as wētā and morepork. One woman was convinced that the more people that use the park, the safer it would be. ‘It may be a silly attitude I’ve got but I reckon everybody’s got to take a stand and the more people that come, the safer it is. And so I always try and encourage people to come.’

The participants in this study were unanimous in their love of bush remnants, restored gullies, open lawn parks and gardens. My research demonstrates that urban residents appreciate a variety of greenspaces in cities which serve an array of functions, ranging from sport and recreation, to community engagement and cohesion, to a place to relax, observe nature and escape the stress of city life. Creating habitat in cities for wildlife, especially native birds, is therefore only one of the multiple purposes of greenspaces that respondents believe are important.

Interviewees did, however, stress the need to have spaces in cities where unique New Zealand plants and animals can thrive. Participants reflected on human destruction of natural environments and their own sense of responsibility to protect and preserve native species. Parks dominated by native vegetation, such as bush remnants and restored gullies, tend to be the places where respondents found the greatest opportunity for ‘escaping’ the stress of a highly urbanised existence through the peaceful observation and appreciation of nature.

In their qualitative study of the wilderness experience of two groups of women in Minnesota and Arizona, Fredrickson and Anderson (1999) found that many participants comment that they rarely had the time or opportunity to experience the natural world in their everyday lives, as they are too distracted by other responsibilities. The present qualitative study highlights the importance of maintaining and growing these spaces within cities, as my participants regularly seek them out and benefit from daily contact with native nature.

The tensions over park use, seen, for example, in the need to allow children to play in natural areas while simultaneously protecting native plants, or the effect of vandalism and perceived danger on residents’ use of densely vegetated parks, point to the unique challenges faced by city planners and park managers in designing greenspaces that will benefit both city residents and native wildlife. The strong sense of ownership and pride

that people living adjacent to parks feel for these greenspaces could be a potential source of conflict for park managers when priorities do not align, but could also be a powerful force for promoting restoration plantings and providing city residents the opportunity to experience nature on a daily basis.

2.4 Summary and Conclusions

The themes brought up by interviewees in our study appeared to organize themselves naturally into dichotomies – urban and ‘wild’ nature, native and introduced species, gardens and parks, tidy and messy, values and practicality. Dichotomies are defined as two mutually exclusive and contradictory groups or entities (Merriam-Webster). Examples of frequently encountered dichotomies in the social sciences include nature and culture, domestic and public, male and female, reason and emotions, quantitative and qualitative. The use of dichotomies, or binaries, in sociological analysis has been criticised for creating ‘either/or’ categories, into which phenomena are forcefully divided, with no middle ground. Thus, studies include such phrases in their titles as ‘overcoming dualisms’, ‘beyond the binaries’ and ‘a critique of dichotomy’. As suggested by Sprague and Zimmerman (1993), however, we present dichotomies not as exclusive, contradictory categories between which we must choose, but as evidence of the tensions we must struggle to integrate.

The overarching tension in our study was that between what people claim to value and their specific actions in relation to urban biodiversity. Social psychologist Michael Billig (1996, p. 209) reflected on this well recognized gap between attitudes and actions:

Attitude theorists have often been faced with evidence that there is frequently an inconsistency between general attitudes and specific beliefs. Often people claim to hold a general attitude, but some of their specific sentiments and actions are hard to square with that attitude [...] Above all, social psychologists have been discomforted [sic] to find that general attitudes are poor predictors of people’s actions [...] However, from a rhetorical perspective, there may be more to be gained by forgetting prediction and by concentrating upon that gap between the general attitude and the specific action.

In our study, the gap between values and actions appeared to stem from our respondents' conflicting or competing desires and priorities. Many participants claimed to value native plants, but their gardens were dominated by introduced species. Interviewees recognized this 'desire to have it all,' with their voiced preference for natives, often stemming from the uniqueness of New Zealand's biological heritage, competing with their desire for the variety and colour of introduced plants and the productivity of vegetable gardens, the social convention of lawns and the impracticality of large trees.

Mixing native and exotic plants was acceptable in gardens and city parks, but not in nature reserves beyond the city limits, highlighting the dichotomy between what 'nature' is seen as acceptable in an urban environment as opposed to 'pristine' forests. In contrast to previous studies, however, our respondents were not worried by 'messy,' 'wild looking' bush parks. The tension between messy and tidy, and how much of each was acceptable, only arose in the context of the participants' own gardens, where the form nature was allowed to take was much more constrained by widely accepted social conventions of what a garden should look like. Even interviewees influenced by permaculture, who tended to be more at ease with a 'messy' garden, were affected by the manicured garden norm and would apologize for perceived 'untidiness.'

Birds in the city were highly valued by the majority of respondents, who actively engaged in bird feeding and bird watching, could recognize a number of bird species and their calls, and were aware of where the birds nested or fed. A few participants could not name any birds and admitted to taking them for granted, but still appreciated birds as contributing to the meaning and attachments they held for their parks and gardens. These generally positive attitudes were little affected by the bird's country of origin and did not result in people actively designing gardens for birds or wildlife, or refraining from owning pet cats, once again highlighting the gap between values and specific actions and choices.

This research suggests urban residents have a strong affinity for nature and wildlife in cities. While valuing native plants and animals, however, respondents' gardening choices were influenced by competing priorities, which led to a disconnect between voiced preferences and actual garden maintenance. Some participants were strongly

influenced by permaculture because they had attended workshops at Hamilton City's environment centre. This finding suggests that more community education on the benefits of gardening for native wildlife has the potential to influence gardening styles and choices. In the meantime, private gardens cannot be relied on to contribute substantially to ecological restoration and habitat provision for our native fauna in urban contexts. At best, private gardens may currently provide sporadic, patchy sources of food and cover for some native species. These findings highlight the need for local authorities to invest more time and resources in ecological restoration of public greenspaces as well as in promoting wildlife gardening.

The tensions and value-action gaps in our study are in alignment with Blake's (1999) claim that environmental concern may be outweighed by other conflicting priorities. Our actions and decisions are not the result of a simple, linear process of logical reasoning, but rather are shaped by many competing values, attitudes, emotions and experiences. Johnston (2008) argues that 'the immediacy of everyday life means that a great deal of human daily behaviour is necessarily intuitive and spontaneous: thought interrupts action, breaks it up into fragments; but by no means does it constantly direct action. The fact that we can think things out in advance does not imply that we always do' (Johnston 2008, p. 642, citing Ingold, 1988c, p. 95). Our motivations for certain actions or inactions are not always clear to ourselves, and it is characteristic of human beings to be inconsistent and unpredictable. Billig (1996) contended that 'there is no reason for supposing that we can conveniently slot all the messy particulars of the world into our general attitudinal categories. We might assume that no one is perfectly consistent'. To better understand human behaviour, we need to explore the specific examples and nuances of what people do in particular places (Hodgetts & Stolte, 2012). Qualitative research methods allow us to examine in depth the value-action gap in individual cases in all their detailed singularity.

Our research underscores the importance of greenspaces in cities. Participants were unanimous in their appreciation of their parks, which they valued as places for walking, exercising, relaxing, escaping the bustle of city life, de-stressing, providing social engagement and cohesion, and reconnecting with nature or with family and friends. Providing habitat for wildlife in the city was therefore only one function of urban green space, with respondents wanting a variety of parks which would meet the range of their

community's needs. Bush parks, however, were identified as areas that best provide opportunities for peaceful, restorative observation of nature, counteracting the stresses of city life. In contrast to Cosquer and her colleagues' (2012) suggestion that city residents are losing the ability to observe nature, most of our respondents were highly attentive to the plants and animals present in their parks, which were a constant source of wonder and enjoyment. A few respondents commented on the need to create spaces in cities for native wildlife and many enjoyed sharing their parks and gardens with birds, bats, frogs and insects, reflecting on how these animals contributed to the meaning of the place.

Participants contrasted bush parks with the city – bush parks were appreciated because they were NOT the city. The city was a busy place of work, rush, deadlines, chores and responsibilities. As places of calm and tranquillity, bush parks offered escape from these things. Attention Restoration Theory (Kaplan, 1995) suggests that bush parks allow the mind to be completely immersed in the sights, sounds and smells of nature – such as the glistening of morning dew on a spider web, the trill of a grey warbler, or the heavy scent of native flowers at night mentioned by our participants.

Both bush and lawn parks contribute to our respondents' sense of community, neighbourliness and social cohesion within their neighbourhood. This sense of connectedness with others who use the park is built over time, as a consequence of frequent park use. Walking, and especially dog walking, is a daily routine or ritual and the frequency and cyclicity of walking means respondents are constantly meeting the same faces, as well as noticing the changes occurring in nature throughout the year.

The dichotomy between park and garden is blurred for participants who live near a park that they use frequently. These residents spoke of a sense of ownership, responsibility and pride in their park, which they view as an extension of their own backyard. This is a collective sense of ownership, similar to the European notion of 'the commons' or Māori notions of *kaitiakitanga*, contrasting with modern Western views of private property ownership. Due to this collective sense of ownership, respondents appreciate the variety of people able to enjoy their park, though some tensions arose over what was seen as appropriate use. This personal feeling of concern and care for one's neighbourhood park could be harnessed by council workers, encouraging residents

living around native bush parks to become involved in restoration efforts, as well as weed and pest control. However, this sense of ownership also highlights potential conflicts when residents' priorities for their park do not align with those of the city council. Further tensions that will need to be addressed include that between perceived safety and dense native vegetation, and balancing use of a park and protection of vulnerable plants and animals.

The multiple health and social benefits provided by bush parks suggest that urban planning laws should ensure the availability of native green spaces within walking distance of residents' homes and workplaces. As the number of people living in cities continues to rise, our research offers renewed evidence for the importance of reserving a space for nature in cities. Far from being a dispensable luxury, native bush parks are crucial for the health and wellbeing of both people and our native flora and fauna.

Finding ways to foster positive personal experiences of place, and the plants, animals, people and stories that give it meaning, is one way to increase city dwellers' emotional involvement with the urban environment. Such subjective bonds can drive the motivation required for people's everyday actions to nurture and protect what they love.

Residents' experiences of urban nature and the bonds they create with it have the potential to simultaneously enhance ecological restoration and public health. We need to consider how we can harness the power of subjective experiences and emotions relating to place to interrupt the vicious cycle of biological poverty and the extinction of experience. Drawing on the concepts of sense of place, dwelling and place attachment, I contend that ecological restoration in urban areas has great potential for restoring and enriching our own human lives and that of native nature in cities today.

Chapter 3: The relative importance of vegetation structure, landscape pattern and invasive mammals to bird communities along an urban forest restoration chronosequence

3.1 Introduction

At least 40 % of the world's bird species have declining populations as a result of human activities such as agricultural expansion, logging, introduction of invasive predators and urbanisation (BirdLife International, 2018). New Zealand has lost 49 per cent of its land bird species since the arrival of humans (Anderson *et al.*, 2011), leading to Jared Diamond's famous assertion that 'in New Zealand today we are not studying an avifauna but the wreckage of an avifauna' (Diamond, 1984). Many of the country's remaining land birds are unique, representing orders, families and genera found only in New Zealand (Gill *et al.*, 2010; Walker & Monks, 2017).

New Zealand's long history of isolation which gave rise to such a unique avifauna also left the endemic bird species particularly vulnerable to predation and habitat loss. Originally, indigenous forest covered 80% of New Zealand, of which 23.8% remains today (Walker *et al.*, 2017). Colonists introduced predatory mammals, such as rats, mustelids and cats, to an ecosystem that previously had none (King, 2005). The decline of vulnerable, endemic forest birds which began with human colonization continues today (Innes *et al.*, 2010a; Walker *et al.*, 2017).

There is a growing conviction that urban green spaces may provide habitat for a diversity of species, including those of conservation concern (Chace & Walsh, 2006; Palmer *et al.*, 2008; Shwartz *et al.*, 2014b; Threlfall *et al.*, 2016). Patches of remnant forest within cities have been shown to support more bird species and higher abundances than surrounding residential areas (Palmer *et al.*, 2008). In a New Zealand city, twice as many native bird species were found in green areas such as parks and gullies than in residential areas (Fitzgerald & Innes, 2015; Innes & Fitzgerald, 2018).

Urban forest remnants, however, are overwhelmingly comprised of small, isolated patches of vegetation with high edge-to-area ratios (Overdyck & Clarkson, 2012; Belder *et al.*, 2018). Habitat restoration could increase the quantity and quality of habitat available to native birds in New Zealand cities. While the goal of restoration has been described as assisting the recovery of degraded ecosystems so that their structure, species composition and ecosystem function resemble that of the historic ecosystem (Standish *et al.*, 2013), restoration activities in New Zealand cities have primarily focused on vegetation – planting of natives and removal of exotic weeds. The expectation that these plantings will provide adequate resources for native fauna has not been tested. This is a failing that characterises restoration efforts beyond New Zealand, with the most common measures internationally of restoration success being plant species richness (Ruiz-Jaen & Mitchell Aide, 2005). To date, there has been limited research supporting the assumption that establishment of native vegetation will result in the recovery of native fauna and ecological processes (Ruiz-Jaen & Mitchell Aide, 2005; Selwood *et al.*, 2009; Derhé *et al.*, 2016).

The widespread failure to plan simultaneously for the restoration of flora and fauna highlights the importance of setting measurable goals at the outset of a restoration project and clearly defining the outcomes that will indicate a restoration ‘success’. Studies have shown that restoration can increase bird community diversity (Ortega-Álvarez & Lindig-Cisneros, 2012), and birds in turn benefit restoration efforts through seed dispersal, pollination, nitrogen input and insect control (Norton, 2009; Anderson *et al.*, 2011; Barbaro *et al.*, 2012; Ortega-Álvarez & Lindig-Cisneros, 2012). The ability of native birds to utilize restoration plantings will be influenced by a complex interplay of landscape variables (e.g. patch size, distance to other areas of native forest), vegetation structure and composition, and competition and predation from introduced mammals.

Mammalian predators are generally believed to pose the greatest and most immediate threat to New Zealand’s native forest birds (Innes *et al.*, 2010a; Walker *et al.*, 2014; Ruffell & Didham, 2017). Ship rats (*Rattus rattus*) and brushtail possums (*Trichosurus vulpecula*), which prey on adult birds on nests, as well as eggs and chicks, and compete with native birds for food, are believed to be the major threat to native birds in large tracts of indigenous forest (Innes *et al.*, 2010a; Innes *et al.*, 2015). Little is known,

however, about the impact of introduced mammals on native birds in urban environments (Russell & Stanley, 2018). Predation processes are likely to be altered by the extreme deforestation, high human population densities and high pet densities characteristic of the urban landscape. One study of mammalian predators in Hamilton, New Zealand, found that rodents and possums were scarce in residential areas and amenity parks and largely restricted to gullies, whereas cats were ubiquitous (Morgan *et al.*, 2009).

While predation by introduced mammals is considered the primary limiting factor for forest birds in New Zealand, one study found that when forest cover in the landscape dropped below 5-10%, native bird species richness declined rapidly (Ruffell & Didham, 2017). Walker and colleagues (2014) similarly concluded that in a region characterized by extreme deforestation, habitat modification and high levels of mammalian predators, the lack of tall, woody vegetation was the primary factor limiting native bird diversity.

Vegetation structure and composition will determine whether a site can provide adequate nesting material, protection from predators and foraging resources for a given bird species. Vegetation variables found to increase native bird species richness, diversity or abundance include the amount of native vegetation (Day, 1995; Marzluff & Ewing, 2001; Chace & Walsh, 2006; van Heezik *et al.*, 2008b; Threlfall *et al.*, 2016), vegetation complexity (Marzluff & Ewing, 2001; Stagoll *et al.*, 2010; Belaire *et al.*, 2014; Threlfall *et al.*, 2016), tree height (Vale & Vale, 1976; Jokimäki, 1999) and canopy cover (Vesk *et al.*, 2008; Belder *et al.*, 2018).

While vegetation variables measure the vegetation composition and structure of each site, landscape variables consider the location of sites within the landscape and the configuration of the site itself (Belder *et al.*, 2018). The majority of studies found vegetation variables to be more important than landscape characteristics in determining bird presence, richness and abundance (van Heezik *et al.*, 2008b; Evans *et al.*, 2009; Selwood *et al.*, 2009; Schwartz *et al.*, 2013; Belaire *et al.*, 2014; Beninde *et al.*, 2015). Birds, however, are highly mobile and will respond to habitat characteristics at multiple spatial scales (Selwood *et al.*, 2009).

The theories of island biogeography and metapopulation dynamics assume that patches of suitable habitat are isolated from each other by an inhospitable matrix (MacArthur & Wilson, 1967). In a built-up urban environment, the island biogeography approach would view patches of native vegetation as islands containing metapopulations of species isolated from each other by surrounding roads and heavily built-up areas (Andren, 1994). While the urban landscape may be highly altered and fragmented, areas viewed by island biogeography theory as hostile or inhospitable may still provide foraging or cover for some species (Melles *et al.*, 2003) and highly mobile fauna such as birds will be able to move between suitable patches. The ‘continuum model’ assumes that landscapes are composed of gradually changing land cover types, with gradients of habitat suitability between these different land covers for different species (Fischer & Lindenmayer, 2006). Due to differences in dispersal abilities, isolation of a habitat patch will depend on the species being considered – a landscape that is highly fragmented for one species may be adequately connected for another (Wiens, 1995).

Landscape characteristics that have been shown to influence bird use of a site include the shape, size and isolation of an area. Patch size has consistently been found to positively affect species richness and diversity (Donnelly & Marzluff, 2004; Drinnan, 2005; Palmer *et al.*, 2008; Beninde *et al.*, 2015). A small number of studies have found the effect of corridors or connecting patches of vegetation to be equally important (Beninde *et al.*, 2015). A study of bird diversity across an urban gradient in the New Zealand city of Dunedin found proximity to bush fragments to be important in predicting the distribution of native bush birds (van Heezik *et al.*, 2008b). A study of native bird populations on Banks Peninsula concluded that native forest within 200-400 m had a strong positive effect on birds observed within a fragment (Deconchat *et al.*, 2009).

The purpose of the following study was to investigate factors influencing native forest bird site occupancy and use in urban restored forests and the time it takes for conditions to become suitable. We evaluated restoration success by investigating how species richness, abundance, diversity and community composition of native forest birds vary along a restoration chronosequence, representing 72 years of forest development. We compared restored areas with two target reference systems – lowland, temperate native forest remnants within the city (urban remnants) and beyond the city limits (rural

remnants), as well as with a control system – unrestored urban forest. Finally, we used structural equation modelling (Lefcheck, 2016) to investigate the relative importance of vegetation diversity and structure, landscape pattern and composition and pest relative abundance to native forest bird species richness and abundance.

We hypothesized that species richness, abundance and diversity of native forest birds would increase over time since restoration, as the vegetation becomes taller and more structurally complex.

We also hypothesized that the bird community would change over time since restoration. We expected that young restoration plantings would be dominated by granivores and feeding generalists, which are introduced species in the urban New Zealand context. As restoration plantings age and become more similar to older urban remnants in their vegetation structure and complexity, we hypothesized that the numbers of introduced granivores and generalists would decrease, while the numbers of native forest birds would increase.

Unrestored sites were expected to support higher numbers of introduced species compared to urban and rural remnants. Rural remnants were expected to provide habitat for endemic New Zealand forest birds that may be absent from urban remnants.

3.2 Methods

3.2.1 Study site

We conducted our research in two cities in New Zealand's North Island, and in the two native forest remnants closest to each city.

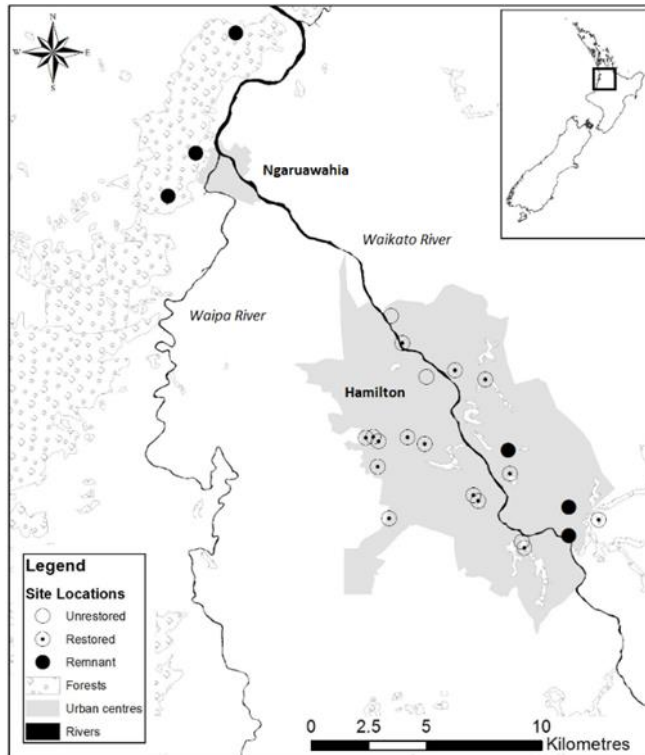


Figure 3.1: Location of the 25 study sites in Hamilton and the Hakarimata Range, which included 16 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in Hamilton, and 3 in the Hakarimata Range.

Hamilton City (37°46'59.99"S, 175°16'59.99"E) (Fig. 3.1) has an estimated population size of 165,400 (Statistics New Zealand, 2017). The climate is temperate oceanic, with mean maximum daily temperatures ranging from 21-26°C in summer, and 10-14°C in winter. The mean annual temperature is 13.7°C and mean annual rainfall is 1190 mm (NIWA, 2016). Hamilton City is bisected by the Waikato River and spans a network of steep sided gullies totalling approximately 770 ha (Cortes *et al.*, 2012). This gully system makes up a large proportion of the city's green space (Fitzgerald *et al.*, 2017). The term 'greenspace' is used in this research to describe any vegetated area within the city limits. Despite the gully network, less than 2% of Hamilton is covered by indigenous forest (Clarkson & McQueen, 2004; Clarkson *et al.*, 2007b).

The nearest rural forest remnant is the Hakarimata Range Scenic Reserve (37°39'45.2"S 175°07'28.1"E) (Fig. 3.1), which lies about 10 km north-west of Hamilton. The range forms the western boundary of the Waikato Basin with its highest point, Hakarimata, at 374 m. The Reserve encompasses 1,850 ha of native lowland forest dominated by broadleaved and podocarp species, such as tawa (*Beilschmiedia tawa*), kohekohe (*Dysoxylum spectabile*), hinau (*Elaeocarpus dentatus*), rewarewa (*Knightia excelsa*), mangeao (*Litsea calicaris*) and pukatea (*Laurelia novae-zelandiae*) (Department of Conservation).

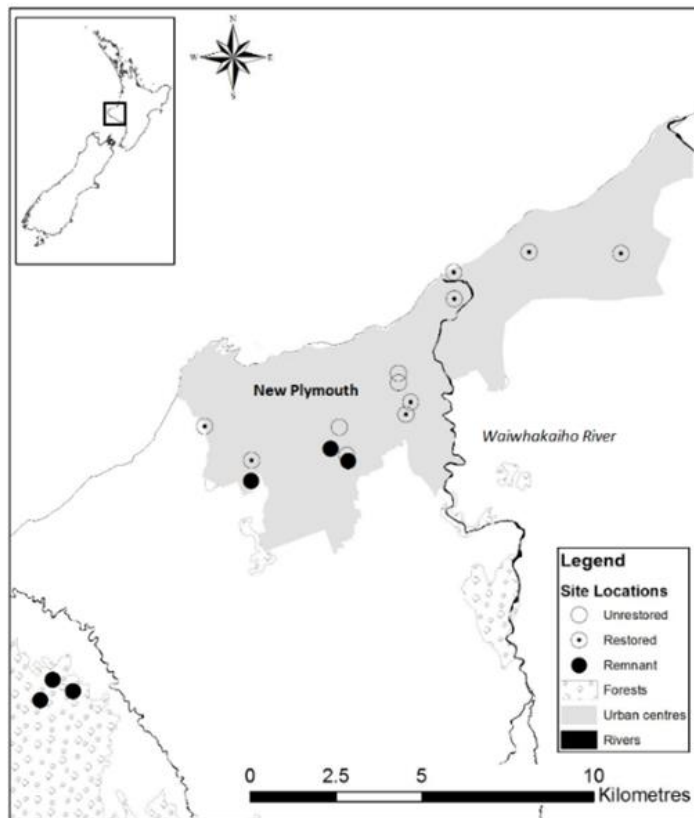


Figure 3.2: Location of the 18 study sites in New Plymouth and the Kaitake Range, which included 9 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in New Plymouth, and 3 in the Kaitake Range.

New Plymouth (39°4'0.01"S, 174°4'59.99"E) (Fig. 3.2) has an estimated population size of 58,300 (Statistics New Zealand, 2017). As in Hamilton, the climate is temperate oceanic with mean maximum summer temperatures between 19-24°C, mean maximum winter temperatures between 10-14°C, a mean annual temperature of 13.7°C and mean

annual rainfall of 1432 mm (NIWA, 2016). New Plymouth retains approximately 8.5% of its indigenous forest cover (Clarkson *et al.*, 2007b).

The nearest rural forest remnant to New Plymouth is the Kaitake Range (39°10'0"S, 173°58'1"E), a range of hills encompassing about 2,400 ha of forest, which lie approximately 15 km south-west of the city. The highest point is Patuha at 684 m. The range is covered in lowland and semi-coastal forest, dominated by kohekohe, tawa, and kamahi (*Weinmannia racemosa*) (Clarkson, 1985).

We conducted our research in urban restored, unrestored and remnant forests in Hamilton and New Plymouth, as well as in the rural forest remnants nearest to these two cities. Restored urban forests are green areas within the urban matrix, dominated by native vegetation that has been intentionally planted. Restored forests (25 from both cities) were chosen to represent an age gradient from 1-72 years since initial planting, as of 2017, and were compared to three types of reference forest:

- 1) Unrestored urban forest (n=6) was defined as a forested area affected by severe anthropogenic disturbance, such as clear-felling or sand mining, that was never subsequently restored, but was left to regenerate naturally. These forests tended to be dominated by exotic plant species.
- 2) Remnant urban forest (n=6) was defined as a patch of native, mature forest within the urban matrix that had never been clear-felled or mined;
- 3) Rural forest remnants (n=6) were large (>1,000 ha) areas of native lowland forest closest to the two cities.

Three sites were randomly located within each rural remnant (the Hakarimata and Kaitake Ranges) by overlaying the ranges with a grid of 10×10 m squares, assigning a number to each square, and randomly choosing three numbers per forest remnant. Three parallel 200 m transects were laid out for each site (the first transect starting at the middle of the randomly selected square), with each transect a minimum of 100 m apart. Thus a 'site' in the rural remnants is defined as a grid of three parallel 200 m transects. Sites were a minimum of 100 m apart and were only set up below 300 m above sea level, as the goal was to sample lower altitude lowland forest.

The names, years since planting (as of 2017), forest category and size (ha) are shown in Appendix 3.1.

3.2.2 Bird surveys

Transect lines (60-200 m length depending on the size of the site) were laid out at least 100 m apart in each of the urban research sites, using a hip chain and a Garmin handheld GPS. Start and end points were marked with flagging tape and saved to the GPS for relocation. Due to the small size (mean 2.06 ha, range 0.13-18.8 ha) of the forests, only three urban remnants were large enough to fit two transects. In the rural forest remnants, three 200 m transects were laid out per site, with each transect a minimum of 100 m apart.

All terrestrial birds within a 20 m belt along each transect were counted by the main author during the breeding season (November-January) and again during the winter (May-July), over a period of 2 years (spring 2016 – winter 2018). Each transect was surveyed twice per study season, over four seasons. We changed the order in which sites were surveyed for the second visit (e.g., if a site was surveyed in the morning on the first visit, it was surveyed in the afternoon on the second) to account for diurnal patterns of bird behaviour.

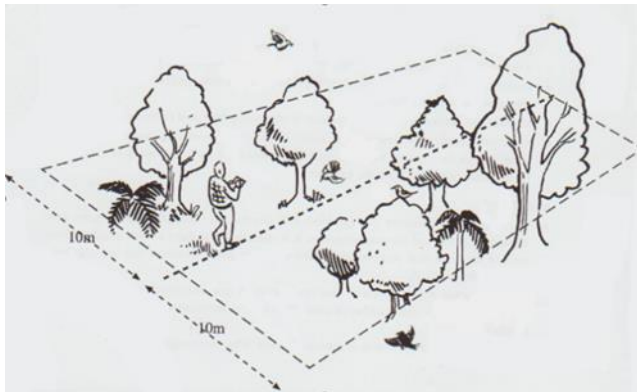


Figure 3.3: Diagram of transect sampling method used, adapted from Wild about New Zealand. Birds detected within the 20 m belt were included in measures of abundance, while birds detected both within the belt and beyond were included in the calculation of species richness.

Birds were recorded by sight and sound, and were assigned to one of two distance belts – within 10 m either side of the transect, and beyond 10 m but still within the research site (Bibby *et al.*, 2000; Sutherland *et al.*, 2004; Fitzgerald & Innes, 2015) (Fig. 3.3). The 10 m distance either side of the transect was measured at the beginning of each bird count

with a tape measure for calibration, and then measured throughout the survey by eye. Individual birds were knowingly recorded only once (e.g., if a fantail, *Rhipidura fuliginosa*, followed the observer, that same individual bird was recorded as one observation). Birds flying overhead were recorded if they appeared to be making use

of the site – e.g., welcome swallows (*Hirundo neoxena*) searching for insects on the wing or tūī (*Prosthemadera novaeseelandiae*) flying from one tree to another. Birds flying overhead more than 20 m above the canopy were not included in the analyses. Given the difficulty of accurately estimating distances from calling birds to the observer, especially in densely forested areas, estimates of distance were not recorded (Van Vianen *et al.*, 2018).

Bird counts were conducted from an hour after sunrise to an hour before sunset to avoid the period of peak conspicuousness at dawn and dusk that would undermine comparisons between sites (MacLeod *et al.*, 2015; Van Vianen *et al.*, 2018). Counts did not take place in strong winds or heavy rain to minimize possible bias caused by weather. We calculated species richness as the total number of species detected at a site over the two visits. We then took the mean number from the two spring and winter bird counts. When calculating species richness, we took into account birds detected both within the 20 m belt and beyond, but still within the site. We determined native forest bird diversity by calculating the Shannon Diversity Index for each site (Hill, 1973; Morris *et al.*, 2014). A relative index of abundance was calculated using only the birds detected within the 20 m belt, standardized as the mean number of individuals counted per 0.4 ha (i.e., a 20 × 200 m belt) across all surveys at a site.

3.2.3 Vegetation measurement

To quantify potential foraging, roosting and nesting resources, we measured vegetation structure and composition within three randomly located 10×10 m² plots within each site. These plots had been set up by Wallace *et al.* (2017) at all sites, apart from the rural forest remnants, one urban remnant (Hillcrest Park), and one young restored forest (Waiwhakareke, planted in 2016). Plots were a minimum of 1 m away from each other and from the forest edge. Due to the small size of the urban forests, edge effects cannot be avoided and will likely prevail throughout the majority of sites (van Heezik *et al.*, 2008b).

The structural complexity of the vegetation was measured using a complexity index developed by Landcare Research for the Department of Conservation (Hurst & Allen, 2007). The complexity index divides the vegetative cover into 6 vertical strata: <0.3,

0.3-2, 2-5, 5-12, 12-25, >25m above ground. Percent vegetation cover in each stratum is visually estimated and coded as: 1 (less than 1%), 2 (1-5%), 3 (6-25%), 4 (26-50%), 5 (51-75%), 6 (76-100%). The values for the three plots were averaged for each stratum and the average values for each site summed up to give the complexity index, ranging from 0-36.

Ground cover was visually estimated and recorded (to the nearest 5%) as percent cover comprised of vegetation, non-vascular plants, litter, bare ground and rock below 1.35 m within each plot (for category definitions, see Hurst & Allen 2007). The values for the three plots were averaged to give a single value per site. Tree heights were measured using an inclinometer – 7 canopy trees were measured per plot, for a total of 21 trees per site and the average tree height per site was calculated (Bibby *et al.*, 2000). Canopy openness was measured by taking hemispherical photos of the canopy from 1 m above the ground in each plot. Images were analysed using Gap Light Analyzer v. 2.0 (Institute of Ecosystem Studies, Millbrook, New York, USA) and the values from the three plots were averaged to give one value for each site. All live trees ≥ 2.5 cm diameter at breast height (DBH = 1.4 m) rooted within the three plots were identified to species and the DBH of each stem was recorded. DBH data were used to compute total basal area per hectare for each site. Sapling counts were conducted to provide data on the regeneration of tree and shrub species. Saplings are defined as woody plants < 2.5 cm DBH and were tallied by species in one randomly chosen quarter of each plot.

The data on canopy openness, basal area per hectare and sapling species richness and density were collected by Wallace and colleagues from 2015-2016. The same methods were used in the present study for the added sites – the rural forest remnants (6), one urban forest remnant (1) and the 2016 restoration planting (1). Data for these sites was collected in 2017. Data on tree height, structural complexity and ground cover for all sites were collected by the primary author in 2017.

3.2.4 Predator monitoring

Indices of relative abundance and presence/absence data on potential predators within each site were collected using chew cards for rats and possums and camera traps for cats (*Felis catus*), mustelids (weasels, *Mustela nivalis vulgaris*; ferrets, *Mustela furo*;

stoats, *Mustela erminea*) and hedgehogs (*Erinaceus europaeus*). The chew cards were 10×20 cm sheets of corrugated plastic with the edges filled with a mix of peanut butter and aniseed. Ten chew cards per transect were spaced at 20 m intervals along the transects used for bird counts and were deployed for two nights over the same time period as the bird counts (Sweetapple & Nugent, 2011; Ruffell *et al.*, 2015a; Ruffell *et al.*, 2015b). The chew cards were nailed to the nearest tree at 20-metre intervals, approximately 30 cm off the ground. In younger sites where the trees were too small, chew cards were mounted on metal rods. A relative index of rat and possum abundance was calculated as the proportion of chew cards bitten by rats and possums, respectively.

Three baited camera traps at small sites (< 5 ha) and 5 at larger ones (≥ 5 ha) were set up around the periphery of each site a minimum of 1 m from the edge, at roughly even intervals, facing inwards, to maximize the likelihood of animals entering or leaving the site detecting and investigating the camera trap lures (Lincoln, 2016). Camera traps were left out over a period of two nights. We used 2016 Browning Strike Force Elite cameras, which have a picture trigger and recovery speed of 0.65 seconds and 1.3 seconds, respectively, and a detection range of 15 m.

Camera trapping protocol largely followed that used by Glen *et al.* (2014). The lure consisted of cotton wool soaked in fish oil, placed in a perforated plastic vial and pegged to the ground 1.5 m in front of the cameras. Cameras were mounted on trees so that the lens was 10 cm above the ground. Cameras were programmed to take five still photographs when motion triggered. Photographs of the same species less than one minute apart were counted as a single detection.

In order to estimate the relative abundance of target species, a distance of 1.5 m in front of the camera was measured and marked with the lure to standardise the area monitored by each camera. Animals photographed beyond the 1.5 m zone were not included in measures of relative abundance but were recorded as species present at the study site. A relative index of cat abundance was calculated as the number of individually distinguishable cats detected over the 48-hour period within each site.

3.2.5 Landscape measures

Landscape variables measured for each site were patch area, perimeter, landscape composition within five buffer zones (see Appendix 3.2) and distance to nearest road and water source. Initial data analysis revealed that the 200 m buffer zone was most strongly correlated with all response variables. Therefore, further analyses only used the 200 m buffer zone.

Landscape composition within the 200 m buffer zone was classified on the basis of satellite imagery and the New Zealand Land Cover Database v. 4.1 (LCDB). The LCDB categories were combined into the following nine land cover categories: native forest, exotic forest, built environment, exotic crop and grassland, water, native scrub, exotic scrub, sand or gravel and native wetland (Appendix 3.3). The area of each land cover type within the 200 m buffer zone was calculated using ArcGIS.

3.2.6 Statistical analysis

All statistical analyses were performed in R version 3.5.2 (R Development Core Team 2018).

Change in bird communities over time

We investigated bivariate plots and fitted linear regression models to explore the relationships between time since initial restoration planting and native forest bird species richness, abundance and diversity for spring and winter bird counts.

We tested for differences in native bird species richness, diversity and abundance between the three reference categories (urban remnant, rural remnant and unrestored) using one-way ANOVA. We first confirmed equality of variances across groups with the Levene test for homoscedasticity, as it is quite robust to departures from normality (Levene, 1960).

We investigated bivariate plots and fitted linear regression models to explore the relationships between time since initial restoration planting and the abundance of the four most frequently detected native birds – fantail, tūī, grey warbler (*Gerygone igata*)

and silvereye (*Zosterops lateralis*). We tested for differences in abundance between the three reference categories using one-way ANOVA.

To visualise the effect of site type and time since initial restoration planting on individual bird species abundance for all native forest birds detected, we inspected a bar chart of average numbers of birds observed for spring and winter counts. We further divided the restored sites into young (1-10 years since initial restoration planting), medium (13-27 years) and old (34-72 years).

For a visual representation of the bird community composition in unrestored, restored, and remnant forests, we used a nonmetric multidimensional scaling (NMDS) ordination in the vegan package in R (Oksanen, 2011) with the Bray-Curtis dissimilarity index. Species abundance data were square-root transformed and standardized using a Wisconsin double relativization, which divides species by their maxima (Oksanen, 2011). The mean number of individuals for the two springs was used in the first ordination and the mean number of individuals for the two winters was used in the second. The data matrix consists of 24 species and 43 samples.

We positioned bird species names on the NMDS ordinations at the weighted average of site scores, where weights are determined by the abundances of each species.

To test for differences in bird community composition (relative abundances of birds of different species) between site types, we performed permutational multivariate analysis of variance (PERMANOVA, Anderson 2001), using the vegan package in R on the spring and winter bird count data. PERMANOVA is a non-parametric method for performing a multivariate analysis of variance based on permutation tests. Analyses were based on Bray-Curtis dissimilarity with 999 permutations. We evaluated which site types differed significantly from each other using the pairwise adonis function in R (Martinez Arbizu, 2019; Toft *et al.*, 2019).

We then evaluated which species contributed most to the significant overall differences in species composition between site types using similarity percentage (SIMPER) analysis with the vegan package in R.

Differences in vegetation attributes with time since restoration and site type

In order to visualise the effect of time since initial restoration planting and of site type on vegetation variables, and to simplify the complex vegetation data into a set of primary factors that capture the variation across the original variables, we conducted a principal components analysis (PCA) on the vegetation variables using the `prcomp` function in R.

PCA is a data reduction technique that generates a smaller number of uncorrelated variables, each of which is a composite of many of the original variables, which show common trends of variation (Peres-Neto *et al.*, 2003). We scaled and centred vegetation data prior to conducting the PCA, and square root or log transformed vegetation variables to meet assumptions of normality. To aid in interpretation of the first two PC axes, we assessed the strength of the correlation of the axes with each original vegetation variable.

While PCA was used to visually explore the vegetation data, this technique cannot disentangle direct and indirect effects of vegetation variables on bird community response variables. We therefore used structural equation modelling to evaluate direct and indirect effects of vegetation, landscape, pest and bird variables in a single causal network.

Relative effects of vegetation, landscape and pest variables on native bird species richness and abundance

We constructed bivariate plots and fitted linear regression models to investigate the relationship between vegetation attributes (canopy openness, basal area, tree height, complexity index, and native species richness) and native bird species richness and abundance.

We constructed bivariate plots and fitted linear regression models to investigate the relationship between landscape variables (site area, perimeter, and amount of native forest, built environment and exotic crop and grassland in a 200 m buffer area) and native bird species richness and abundance.

We used structural equation modelling with the piecewiseSEM package in R (Lefcheck, 2016) to evaluate direct and indirect effects and the relative importance of time since initial restoration planting, vegetation composition and structure, landscape variables and pest abundances on native bird species richness and abundance. PiecewiseSEM, also referred to as confirmatory path analysis, is a multivariate technique that analyses hypothesized relationships between multiple predictor and response variables in a single causal network. PiecewiseSEM can be used to investigate whether *a priori* hypothesized causal relationships are supported by the observed data (Whytock *et al.*, 2018).

We used theory, previous studies and expert knowledge to develop an *a priori* model of hypothesized direct and indirect relationships between our measured variables (Fig. 3.4). We hypothesized that age would have a direct, positive effect on tree basal area (Wallace *et al.*, 2017), vegetation complexity (Munro *et al.*, 2009) and native tree species richness (Ruiz *et al.*, 2005; Letcher & Chazdon, 2009). Basal area would in turn have a direct, negative effect on canopy openness (Wallace *et al.*, 2017). Canopy openness (Blair, 2004; Vesik *et al.*, 2008) and vegetation complexity (Robertson & Hackwell, 1995; Evans *et al.*, 2009; Wilson *et al.*, 2014; Belder *et al.*, 2018) were predicted to directly affect native bird species richness and abundance. Canopy openness was further predicted to influence native birds indirectly through its effect on native sapling species richness.

We hypothesized that native forest cover in the surrounding landscape (Deconchat *et al.*, 2009) and site area (Fernandez-Juricic & Jokimäki, 2001; Melles *et al.*, 2003; Drinnan, 2005; Daniels & Kirkpatrick, 2006) would have a direct, positive effect on native birds. We further hypothesized that the amount of built-up area within the surrounding landscape would influence cat numbers, as cats are expected to be more abundant near residential areas (Kikillus *et al.*, 2017), which we expected would in turn have a harmful effect on native birds (Belaire *et al.*, 2014).

We hypothesized that rat and possum indices of relative abundance would also respond to both vegetation and landscape variables. Rats and possums were expected to be influenced by basal area, vegetation complexity (Harper *et al.*, 2005), the amount of native forest in the surrounding landscape and canopy openness (King, 2005). Possums

were further hypothesized to respond to site area, whereas rats were expected to be hunted by cats (Gordon *et al.*, 2010). Rats (Innes *et al.*, 2010a; Innes *et al.*, 2015) and possums (Byrom *et al.*, 2016) were predicted to have a negative effect on native bird species richness and abundance.

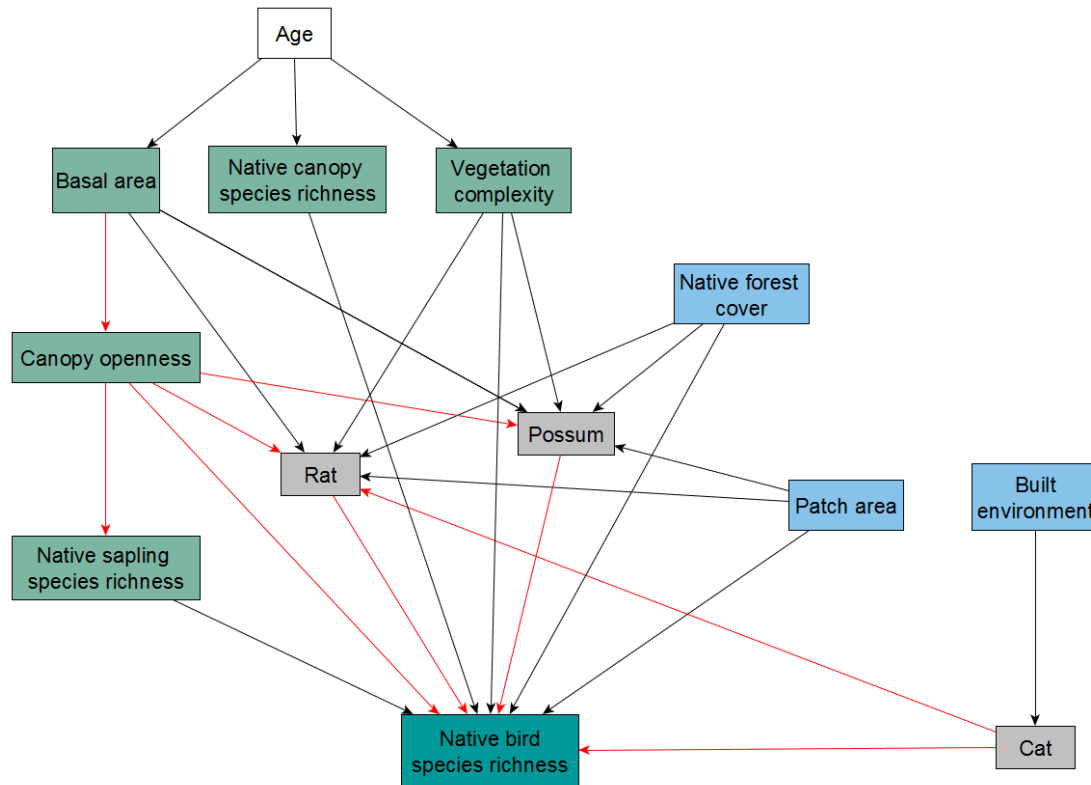


Figure 3.4: Hypothesized a priori structural equation model (SEM) path diagram.

Where necessary, variables were log or square root transformed to meet assumptions of normality, and then scaled to unit variance. We used generalized linear models for possum and rat indices of relative abundance which were binomially distributed (we used the binomial data on whether a chew card detected a possum or rat over the 48-hour period as the response variable), as well as for native canopy species richness, native sapling species richness, and native bird species richness and abundance which were Poisson distributed. We used the same *a priori* model to predict native bird species richness and abundance. Model fit was evaluated using Shipley's test of directed separation (Fisher's C) where p-values > 0.05 indicate that the data do not differ significantly from the specified model. Thus, models with p-values > 0.05 represent the data well. After constructing the linear and generalized linear models that

made up our structural equation model, we specified correlated errors, i.e., pairs of variables that we predicted may appear correlated but which we did not believe to have a causal relationship. Correlated errors were specified using the ‘% $\sim$ %’ operator in the `psem` function in the `piecewiseSEM` package in R (Lefcheck, 2018). We then examined the ‘tests of directed separation’ output in R in order to identify any significant ($p < 0.05$) causal pathways between our variables that we may have overlooked. Once these were added to the model, we removed insignificant ($p > 0.05$) pathways one by one and examined the effect of their removal on the Akaike Information Criterion (AIC). We present the model with the lowest AIC score. We report the standardised coefficients for each path where possible (for linear models and binomially distributed data), and unstandardised coefficients for Poisson distributed data.

3.3 Results

We observed 2,110 individuals comprising 28 species of terrestrial birds over the course of the study (Appendix 3.4). Bird species richness at sites ranged from 0 to 13 species. Of the 28 terrestrial bird species, 17 (61%) were introduced and 11 (39%) were native to New Zealand, including the shining cuckoo, which is a summer migrant.

The four most abundant birds across all sites ($n = 43$) for spring counts were silvereyes (mean = 163.5, total counted on all transects), house sparrows (mean = 81.5), fantails (mean = 72.5) and blackbirds (mean = 59.5). The four most abundant birds across all sites for winter counts were silvereyes (mean = 191.5), blackbirds (mean = 74), fantails (mean = 74) and tūī (mean = 50). The four most widespread birds during the spring counts (i.e., detected at the most sites, where the maximum number of sites is 43) were silvereye (detected at 41 sites), blackbird (38 sites), fantail (35 sites) and chaffinch (28 sites). The four most widespread birds during the winter counts were silvereye (42 sites), blackbird (39 sites), fantail (39 sites) and grey warbler (28 sites).

Three endemic New Zealand birds, tomtits, whiteheads and bellbirds, were only detected in rural remnants. The endemic kererū (New Zealand pigeon) was only detected in urban and rural remnants.

Change in bird communities over time

Bivariate plots and linear models revealed that time since restoration had a significant positive relationship with native bird species richness (spring: $R^2 = 0.56$, $p < 0.001$; winter: $R^2 = 0.50$, $p < 0.001$) and diversity (spring: $R^2 = 0.26$, $p = 0.009$; winter: $R^2 = 0.17$, $p = 0.04$), with these relationships being stronger for the spring count data (Fig. 3.5 and 3.6). The relationship between native bird abundance and time since restoration was not significant for either season.

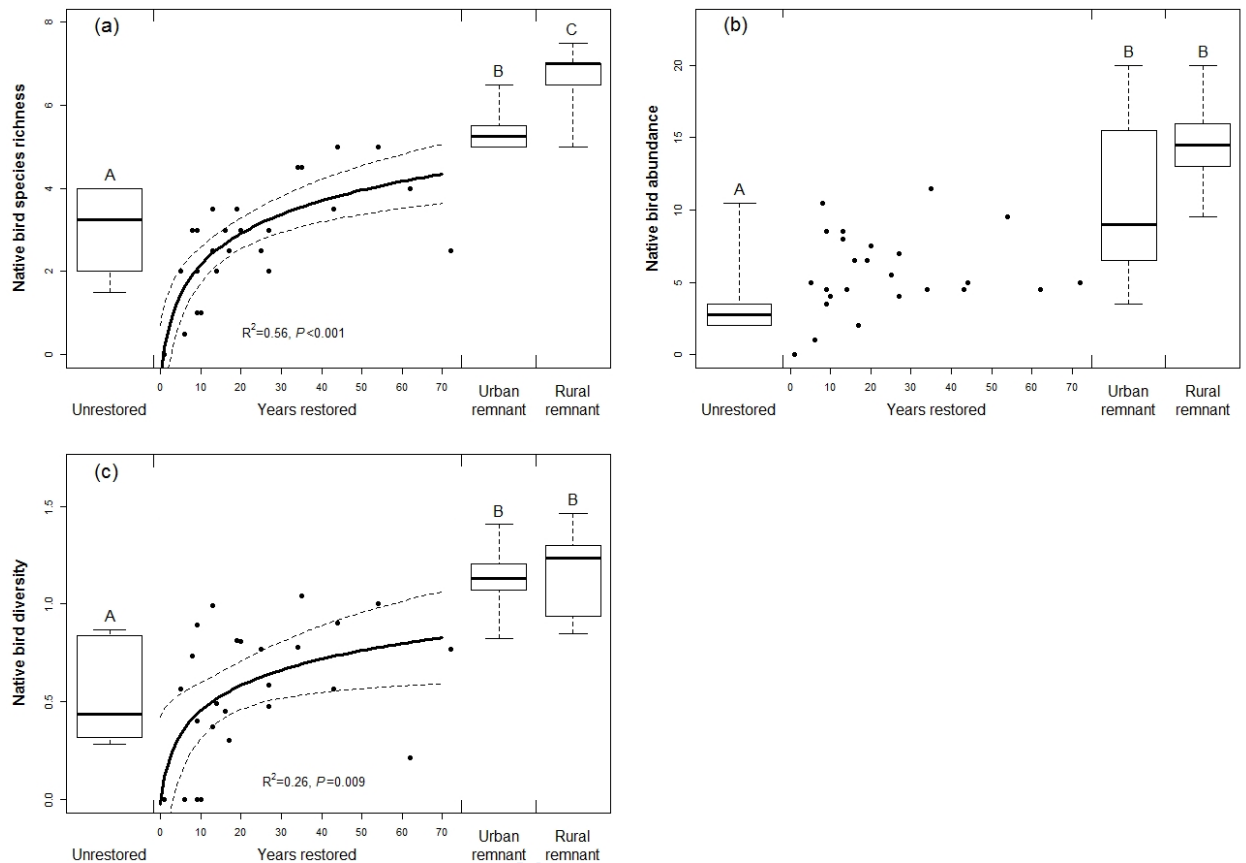


Figure 3.5: Mean native bird (a) species richness, (b) abundance (individuals per 0.4 ha) and (c) taxonomic diversity (Shannon index) for spring counts in restored forests over time (middle section of each bivariate plot, $n=25$),

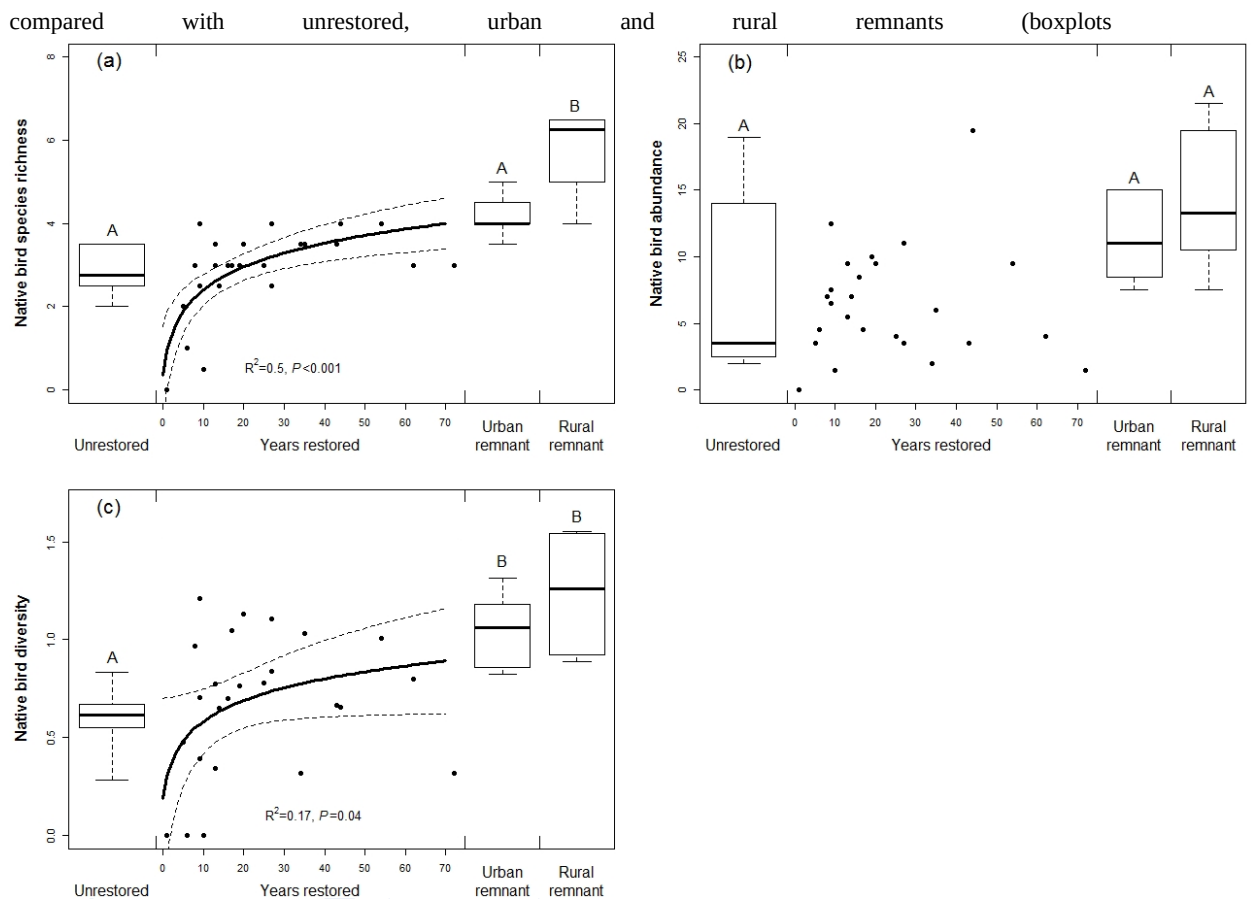


Figure 3.6: Mean native bird (a) species richness, (b) abundance (individuals per 0.4 ha) and (c) taxonomic diversity (Shannon index) for winter counts in restored forests over time (middle section of each bivariate plot, $n=25$), compared with unrestored, urban and rural remnants (boxplots).

We found a statistically significant difference between groups in native bird species richness as determined by one-way ANOVA ($F_{2,15} = 24.88$, $p < 0.001$) for the spring bird count data (Fig. 3.5). A Tukey post hoc test revealed that all three reference forest types differed significantly from each other ($p < 0.05$). There was a statistically significant difference between groups in native bird diversity as determined by one-way ANOVA ($F_{2,15} = 16.45$, $p < 0.001$) for the spring bird count data. A Tukey post hoc test revealed that unrestored forests differed significantly from both rural and urban remnants ($p < 0.001$). There was no significant difference between urban and rural remnants. There was a statistically significant difference between groups in native bird abundance as determined by one-way ANOVA ($F_{2,15} = 8.75$, $p = 0.003$) for the spring bird count data. A Tukey post hoc test revealed that unrestored forests differed significantly from rural remnants ($p = 0.002$). The difference between urban remnants and unrestored forests was just significant ($p = 0.05$) and the difference between urban and rural remnants was not significant.

We found a statistically significant difference between groups in native bird species richness as determined by one-way ANOVA ($F_{2,15} = 28.88$, $p < 0.001$) for the winter bird count data (Fig. 3.6). A Tukey post hoc test revealed that unrestored forests differed significantly from rural remnants ($p < 0.001$) and rural remnants differed significantly from urban remnants ($p < 0.001$). Urban remnants, however, did not differ significantly from unrestored forests. There was a statistically significant difference between groups in native bird diversity as determined by one-way ANOVA ($F_{2,15} = 12.61$, $p < 0.001$) for the winter bird count data. A Tukey post hoc test revealed that unrestored forests differed significantly from both rural and urban remnants ($p < 0.001$). There was no significant difference between urban and rural remnants. There was no statistically significant difference between groups in native bird abundance for the winter bird count data.

Bivariate plots revealed that time since restoration had a significant positive relationship with fantail ($R^2 = 0.23$, $p < 0.001$) and tūī ($R^2 = 0.17$, $p < 0.001$) abundance for the spring count data (Fig. 3.7), but no significant relationship for winter data (Fig. 3.8). The relationship between grey warbler and silvereye abundance and time since restoration was not significant for either season (Fig. 3.7 and 3.8).

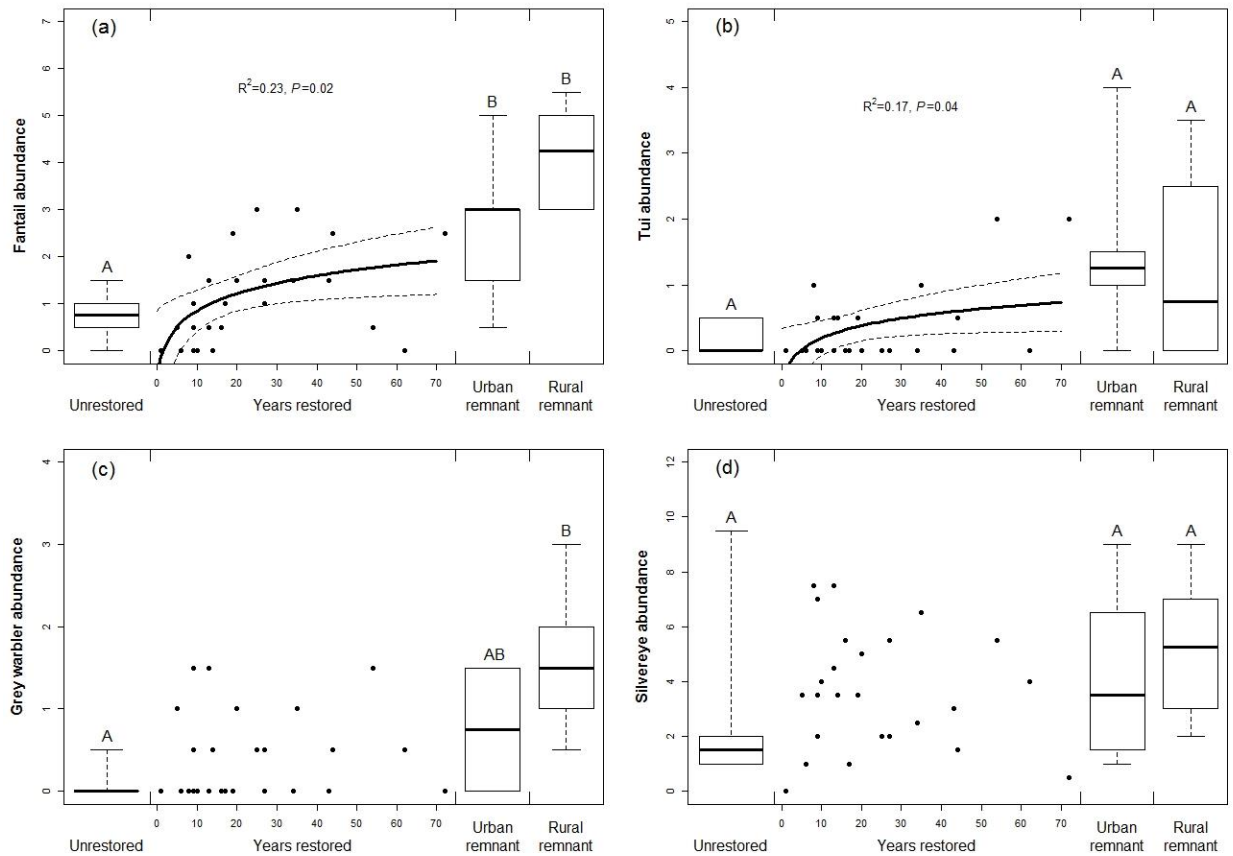


Figure 3.7: Mean abundance of (a) fantails, (b) tūī, (c) grey warblers and (d) silvereyes for spring counts in restored forests over time (middle section of each bivariate plot, $n=25$), compared with unrestored, urban and rural remnants (boxplots).

We found a statistically significant difference between groups in fantail abundance as determined by one-way ANOVA ($F_{2,15} = 14.24$, $p < 0.001$) for the spring bird count data (Fig. 3.7). A Tukey post hoc test revealed that unrestored forest differed significantly from both urban remnants ($p = 0.02$) and rural remnants ($p < 0.001$). There was no significant difference between urban and rural remnants. There was no statistically significant difference between groups in tūī abundance for the spring bird count data. We found a statistically significant difference between groups in grey warbler abundance as determined by one-way ANOVA ($F_{2,15} = 8.08$, $p = 0.004$) for the spring bird count data. A Tukey post hoc test revealed that unrestored forest differed significantly from rural remnants ($p = 0.003$). There was no significant difference between unrestored and urban remnants or urban and rural remnants. There was no statistically significant difference between groups in silvereye abundance for the spring bird count data.

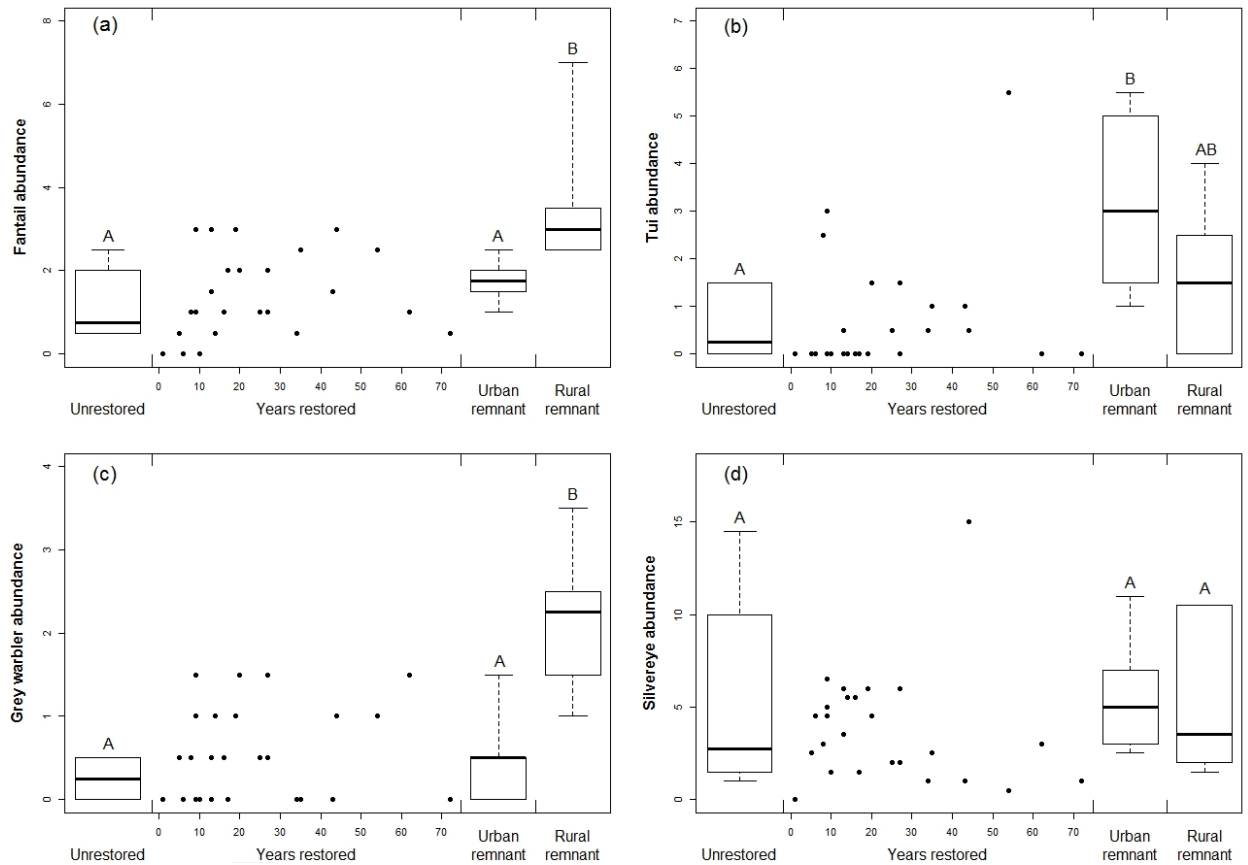


Figure 3.8: Mean abundance of (a) fantails, (b) tūī, (c) grey warblers and (d) silvereyes for winter counts in restored forests over time (middle section of each bivariate plot, $n=25$), compared with unrestored, urban and rural remnants (boxplots).

We found a statistically significant difference between groups in fantail abundance as determined by one-way ANOVA ($F_{2,15} = 7.01$, $p = 0.007$) for the winter bird count data (Fig. 3.8). A Tukey post hoc test revealed that unrestored forest differed significantly from rural remnants ($p = 0.007$) but not from urban remnants. There was also a significant difference between urban and rural remnants ($p = 0.04$). We found a statistically significant difference between groups in tūī abundance as determined by one-way ANOVA ($F_{2,15} = 4.89$, $p = 0.02$) for the winter bird count data. A Tukey post hoc test revealed that unrestored forest differed significantly from urban remnants ($p = 0.02$) but not from rural remnants. There was no significant difference between urban and rural remnants. We found a statistically significant difference between groups in grey warbler abundance as determined by one-way ANOVA ($F_{2,15} = 17.12$, $p < 0.001$) for the winter bird count data. A Tukey post hoc test revealed that unrestored forest differed significantly from rural remnants ($p < 0.001$) and urban remnants differed

significantly from rural remnants ($p < 0.001$). There was no significant difference between unrestored and urban remnants. There was no statistically significant difference between groups in silvereye abundance for the winter bird count data.

Inspection of bar graphs revealed that tomtits were only observed in rural remnants for the spring counts (Fig. 3.9), and whiteheads, tomtits and bellbirds were only recorded in rural remnants for the winter counts (Fig. 3.10). Kererū were only detected in rural and urban remnants. Native bird numbers show an increasing trend with age of restored site, with numbers tending to be highest in rural remnants. Tūi numbers were higher in urban remnants when compared with rural remnants, especially in the winter counts (Fig. 3.10).

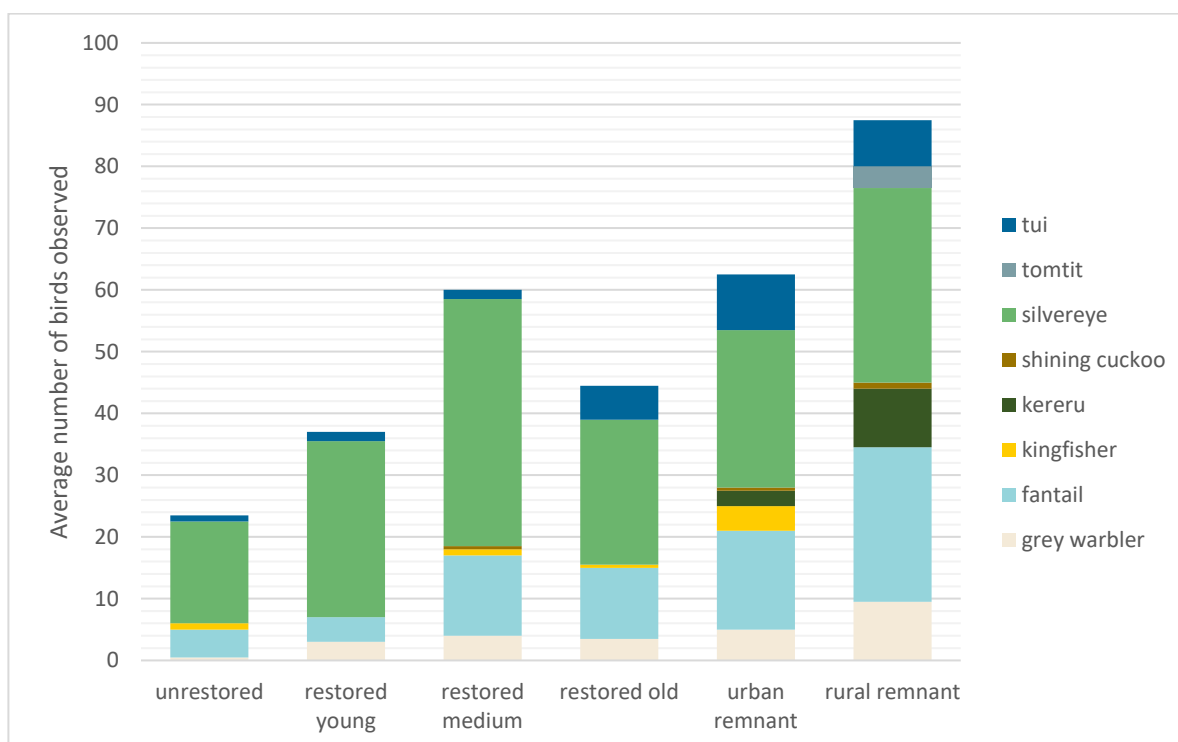
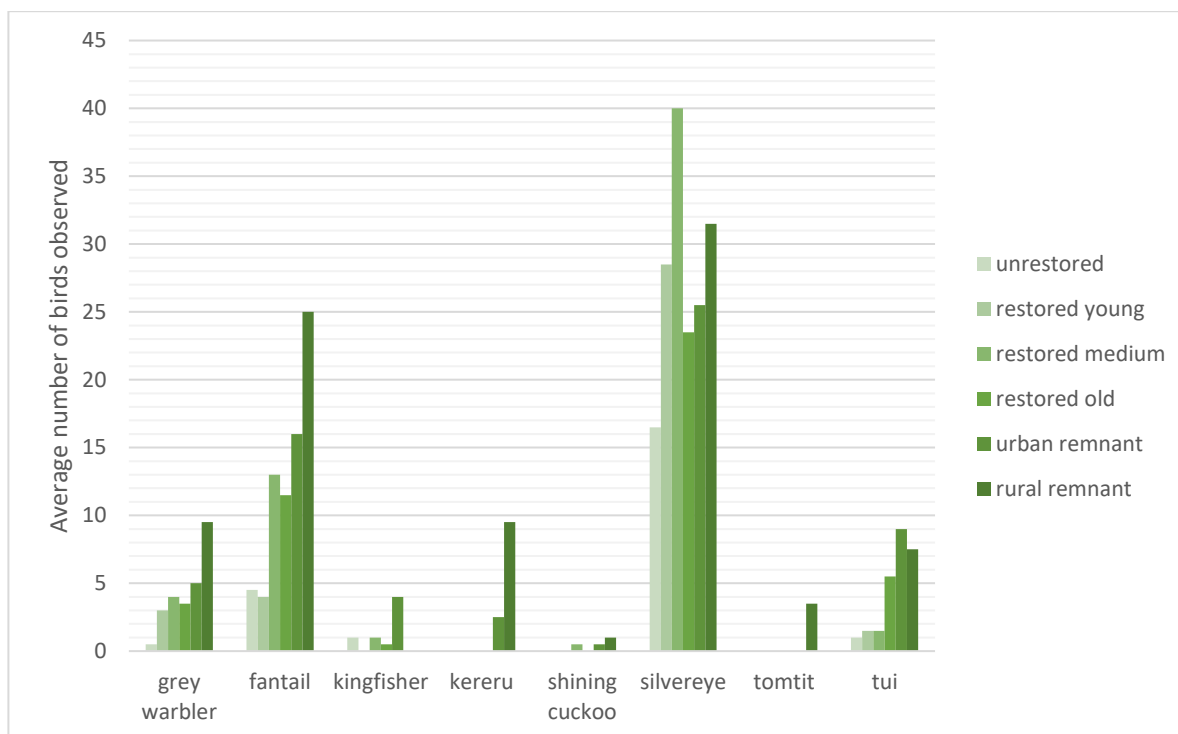


Figure 3.9: Average number of native birds observed per site type for spring counts. Restored sites were divided into young (1-10, n = 8), medium (13-27, n = 10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.

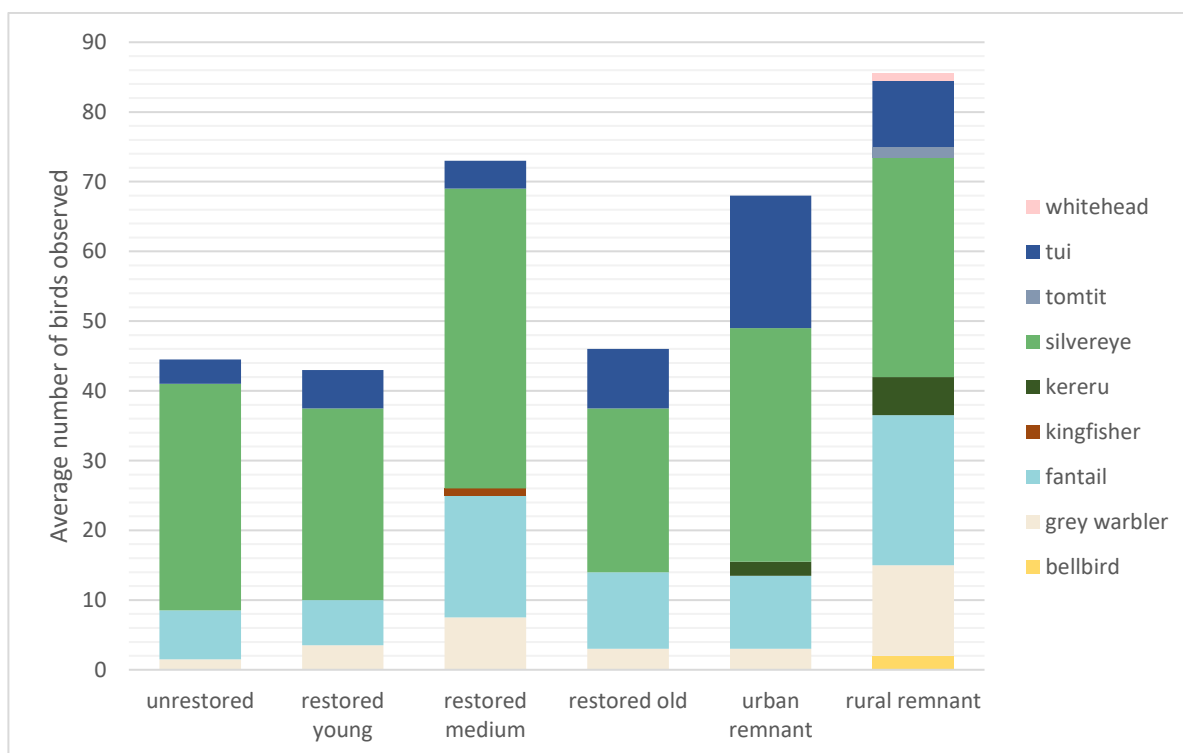
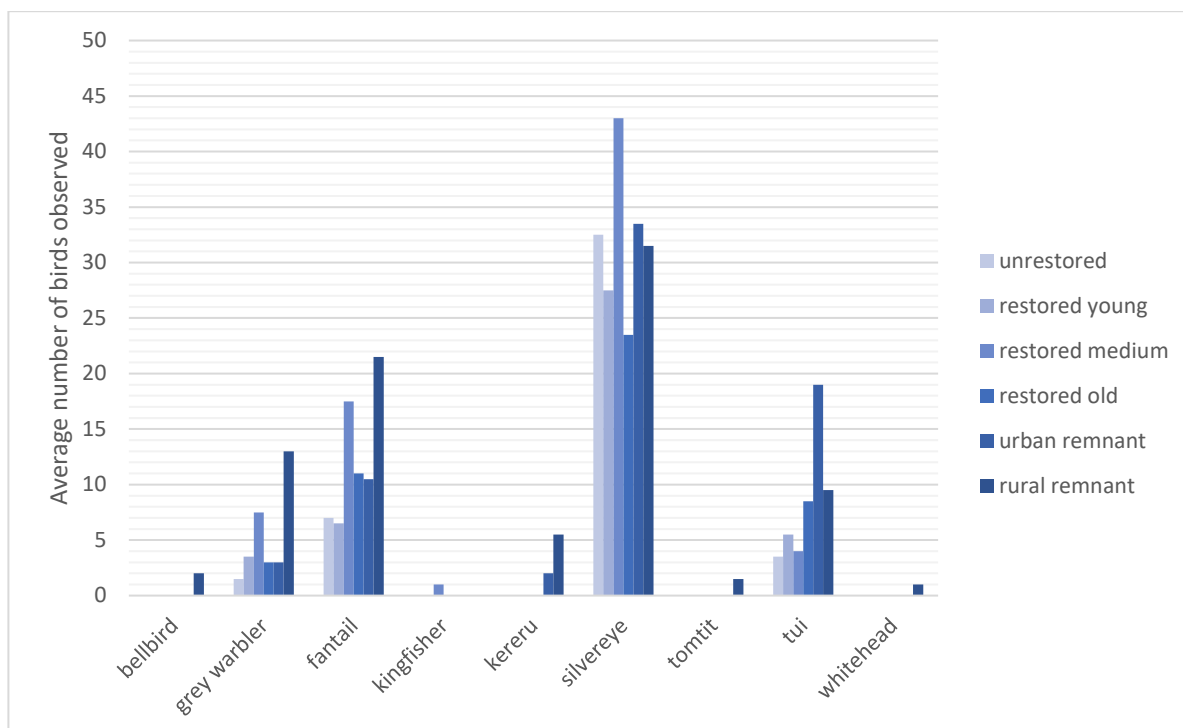


Figure 3.10: Average number of native birds observed per site type for winter counts. Restored sites were divided into young (1-10, n = 8), medium (13-27, n=10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.

The two dimensional NMDS ordination for the spring data reflects a change in bird community composition with site age and type (Fig. 3.11). There appears to be a gradient in the graph from the bottom right, with a cluster of young restored sites, to the top left with a cluster of remnant forests, both urban and rural. The middle section of the graph shows overlap between the young, medium and old restored sites as well as the unrestored sites. Therefore, while young restored sites and remnants appear quite distinct in bird community composition, the transition between the two is less clear cut. These patterns are not as clear for the winter bird count data.

There were two distinct outliers for the spring NMDS ordination – the urban remnant on the bottom left was identified as Claudelands Bush. Comparing its bird community composition to that of other urban remnants revealed that it had much higher numbers of blackbirds, as well as detections of goldfinches and song thrush not detected at the other urban remnants. The second outlier was Totara Park, which did not cluster with the other unrestored sites. Totara Park had much higher numbers of house sparrows and silvereyes compared to other unrestored sites, as well as detections of goldfinches and spotted doves, not detected at the other unrestored sites.

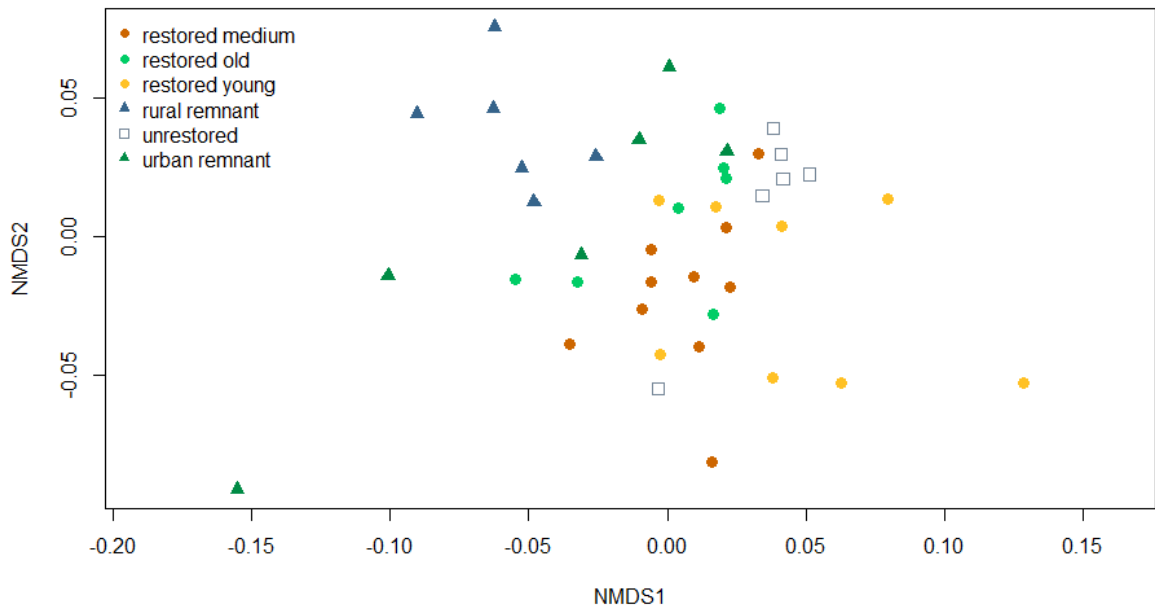


Figure 3.11: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean spring bird abundance data (stress = 0.16).

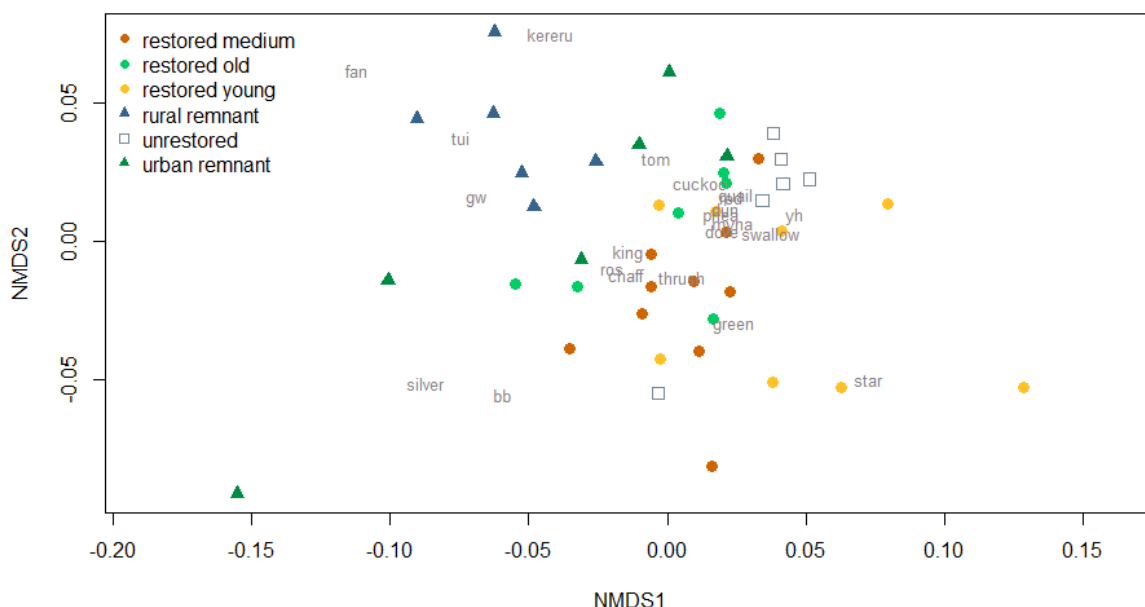


Figure 3.12: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean spring bird abundance data (stress = 0.16).

For the spring NMDS ordination, native forest birds, such as kererū, tūī, tomtit, fantail and grey warbler, clustered at the top left of the plot with rural and urban remnants (Fig. 3.12). Introduced grassland birds, such as starlings and greenfinches, were associated with younger restored sites.

The winter NMDS ordination showed similar patterns with forest birds, such as kererū, grey warbler, fantail, bellbirds, tomtits and whiteheads, being associated with rural remnants. Introduced birds, such as goldfinch, starling, song thrush and house sparrow were associated with young and medium restored sites (Fig. 3.13 and 3.14).

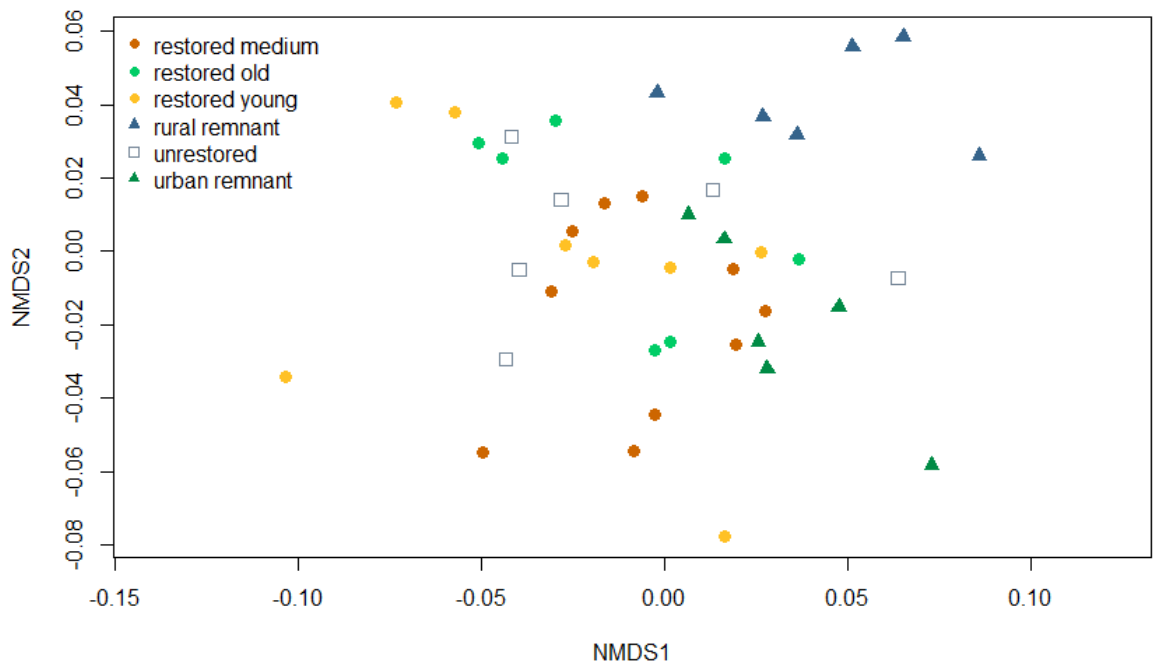


Figure 3.13: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean winter bird abundance data (stress = 0.18).

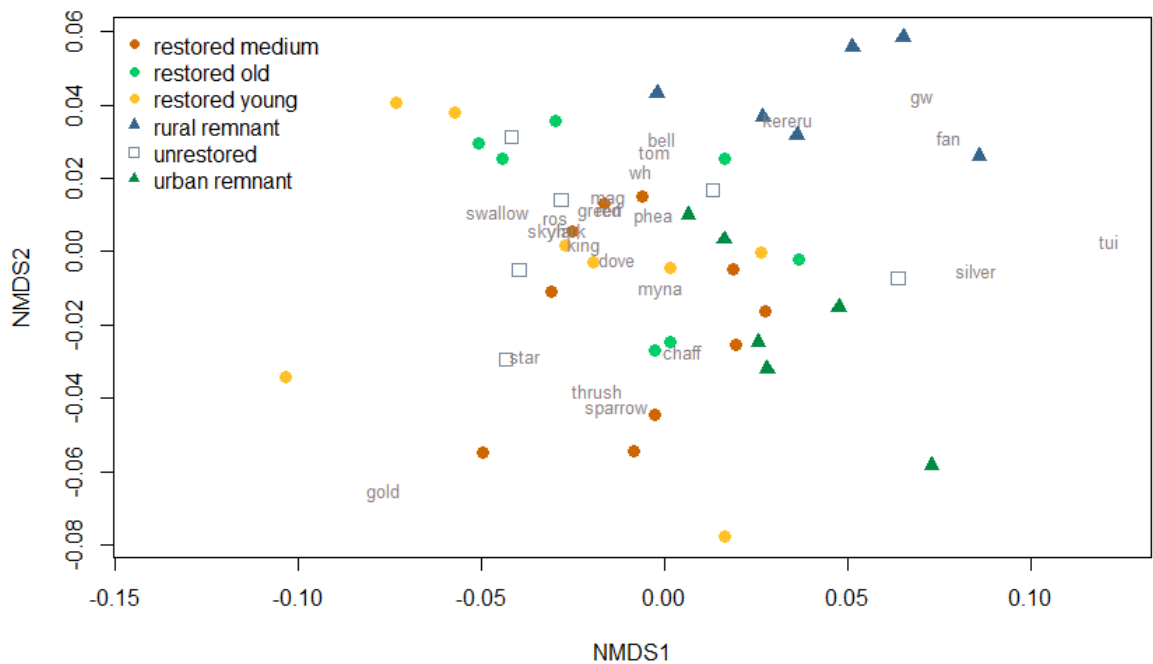


Figure 3.14: Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of the 43 study sites and mean winter bird abundance data (stress = 0.18). **Table 3.1:** Abbreviations used for bird species in NMDS ordination and corresponding full species names.

Abbreviations for species names in NMDS ordination	Full species name
bb	Blackbird

bell	Bellbird
chaff	Chaffinch
cuckoo	Shining cuckoo
dove	Spotted dove
dun	Dunnock
fan	New Zealand fantail
gold	Goldfinch
green	Greenfinch
gw	Grey warbler
kererū	New Zealand pigeon
king	New Zealand kingfisher
mag	Australian magpie
myna	Common myna
phea	Common pheasant
quail	California quail
red	Common redpoll
ros	Eastern rosella
silver	Silvereye
skylark	Eurasian skylark
sparrow	House sparrow
star	Common starling
swallow	Welcome swallow
thrush	Song thrush
tom	Tomtit
tūī	Tūī
wh	Whitehead
yh	Yellowhammer

PERMANOVA revealed significant differences between site types for the spring ($F_{5,37} = 3.37$, $p < 0.001$) and winter ($F_{5,37} = 2.97$, $p = 0.001$) bird count data. For the spring bird count data, significant differences were found between rural remnants and the five other site types ($p < 0.05$), as well as between urban remnants and medium restored sites ($p = 0.045$). The difference between urban remnants and young restored sites was not statistically significant ($p = 0.075$).

For winter bird count data, the only significant differences were between medium restored sites and urban remnants ($p = 0.045$), and medium restored sites and rural remnants ($p = 0.015$).

For the spring count data, seven species contributed just over 50% of the dissimilarity between young restored and urban remnant sites. More abundant in urban remnants

compared to young restored sites were fantails, blackbirds, tūi and grey warblers. More abundant in young restored sites compared to urban remnants were house sparrows, starlings and goldfinches. Six species contributed just over 50% of the dissimilarity between rural remnants and urban remnants. More abundant in rural remnants compared to urban were kererū, fantails, tomtits and grey warblers. More abundant in urban remnants were blackbirds and tūi. Six species contributed just over 50% of the dissimilarity between rural remnants and unrestored sites. More abundant in rural remnants compared to unrestored sites were fantails, grey warblers, kererū, tūi and tomtits. More abundant in unrestored sites were house sparrows.

For the winter count data, six species contributed just over 50% of the dissimilarity between medium restored sites and rural remnants. More abundant in medium restored sites compared to rural remnants were blackbirds and house sparrows. More abundant in rural remnants compared to medium restored sites were grey warblers, tūi, kererū and fantails. Six species contributed just over 50% of the dissimilarity between medium restored sites and urban remnants. More abundant in medium restored sites compared to urban remnants were house sparrows, song thrush and grey warblers. More abundant in urban remnants compared to medium restored sites were tūi, blackbirds and chaffinches.

Differences in vegetation attributes with time since restoration and site type

The first PC axis, accounting for 39% of the total variation (Appendix 3.5), is negatively correlated with basal area ($r = -0.83$), tree height ($r = -0.77$), percent litter cover ($r = -0.75$), native sapling species richness ($r = -0.74$), native sapling density ($r = -0.68$), native canopy species richness ($r = -0.56$), leaf litter depth ($r = -0.48$) and vegetation complexity ($r = -0.39$). The first PC axis is positively correlated with canopy openness ($r = 0.85$) and percent vegetation cover ($r = 0.73$). The first PC axis thus corresponds to the maturity of the site (Appendix 3.5) with younger restored sites spread from the bottom left to the centre of the plot and older restoration plantings and remnant sites clustering on the right. The second PC axis accounts for 13.7% of the total variation (Appendix 3.5) and is positively correlated with vegetation complexity ($r = 0.66$), non-native sapling density ($r = 0.50$), percent vegetation cover ($r = 0.45$),

leaf litter depth ($r = 0.45$) and non-native sapling species richness ($r = 0.40$). Thus, the second PC axis represents increasing vertical heterogeneity at various height tiers and percent vegetation cover, from the bottom to the top of the graph.

Relative effects of vegetation, landscape and pest variables on native bird species richness and abundance

Native bird species richness had a significant, negative relationship with canopy openness ($R^2 = 0.50$, $p < 0.001$) and was significantly, positively related to basal area ($R^2 = 0.42$, $p < 0.001$), tree height ($R^2 = 0.47$, $p < 0.001$), vegetation complexity index ($R^2 = 0.23$, $p < 0.05$), native sapling species richness ($R^2 = 0.35$, $p < 0.001$) and native canopy species richness ($R^2 = 0.23$, $p < 0.001$) (Fig. 3.15).

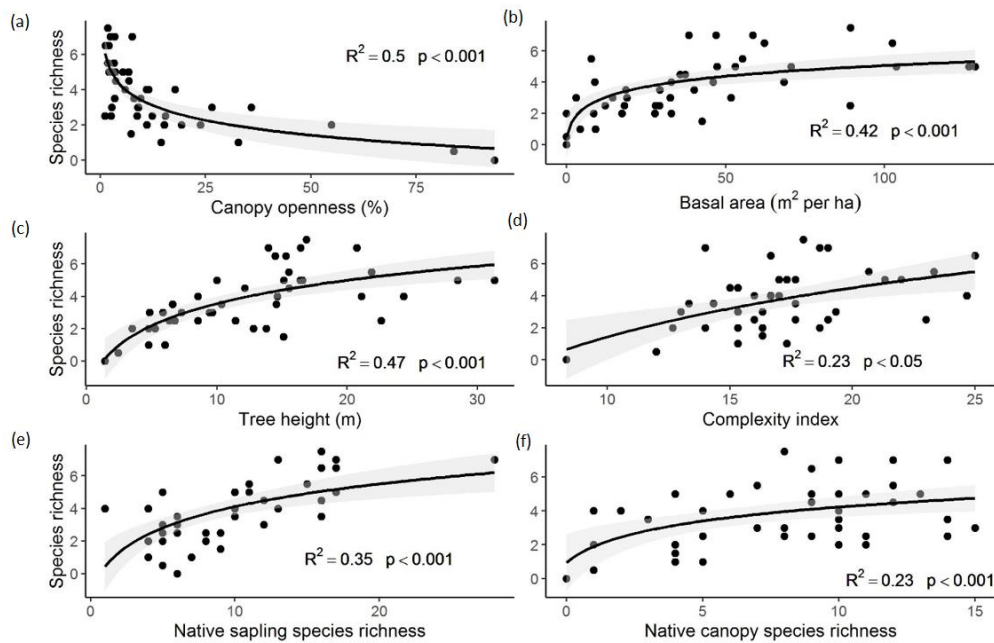


Figure 3.15: Mean spring native bird species richness and plant community attributes in all study sites. Native bird abundance had a significant, negative relationship with canopy openness ($R^2 = 0.28$, $p < 0.001$) and was significantly, positively related to basal area ($R^2 = 0.19$, $p = 0.004$), tree height ($R^2 = 0.18$, $p = 0.005$), native sapling species richness ($R^2 = 0.32$, $p < 0.001$) and native canopy species richness ($R^2 = 0.30$, $p < 0.001$) (Fig. 3.16).

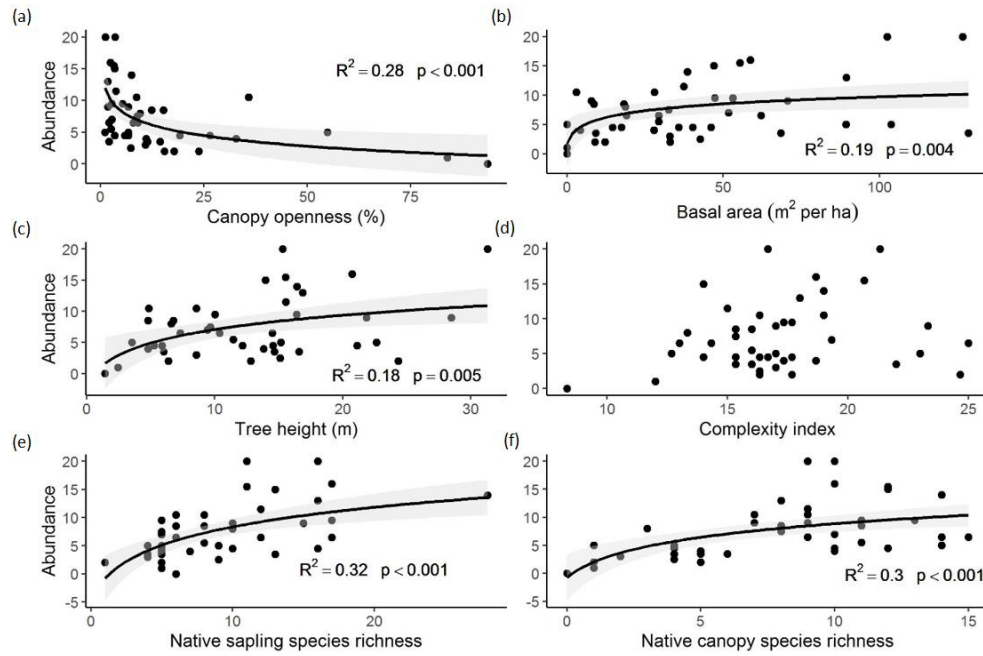


Figure 3.16: Mean spring native bird abundance (no. of individuals per 0.4 ha) and plant community attributes in all study sites.

Native bird species richness had a significant, positive relationship with site area ($R^2 = 0.30$, $p < 0.001$), perimeter ($R^2 = 0.20$, $p = 0.006$) and amount of native forest cover in a 200 m buffer area ($R^2 = 0.21$, $p = 0.005$). Native bird species richness had no significant relationship with the amount of built environment and exotic grass cover in the surrounding 200 m buffer (Fig. 3.17).

Native bird abundance had a significant, positive relationship with site area ($R^2 = 0.13$, $p = 0.03$), and no significant relationship with perimeter, the amount of native forest cover, built environment and exotic grass cover in the surrounding 200 m buffer (Fig. 3.18).

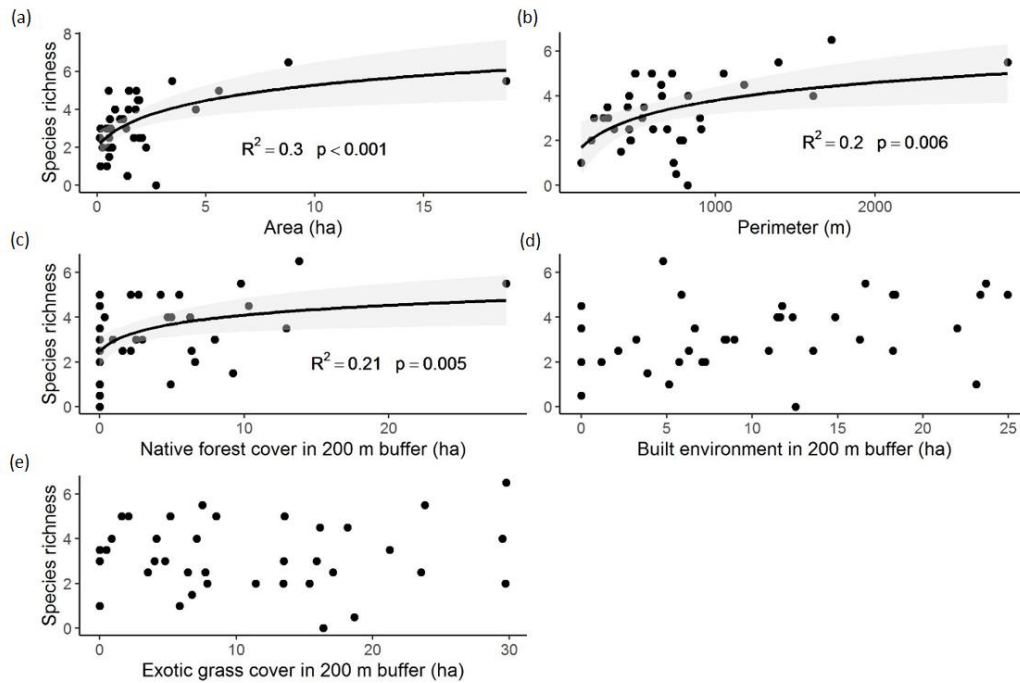


Figure 3.17: Mean spring native bird species richness and landscape variables for all urban sites ($n = 37$).

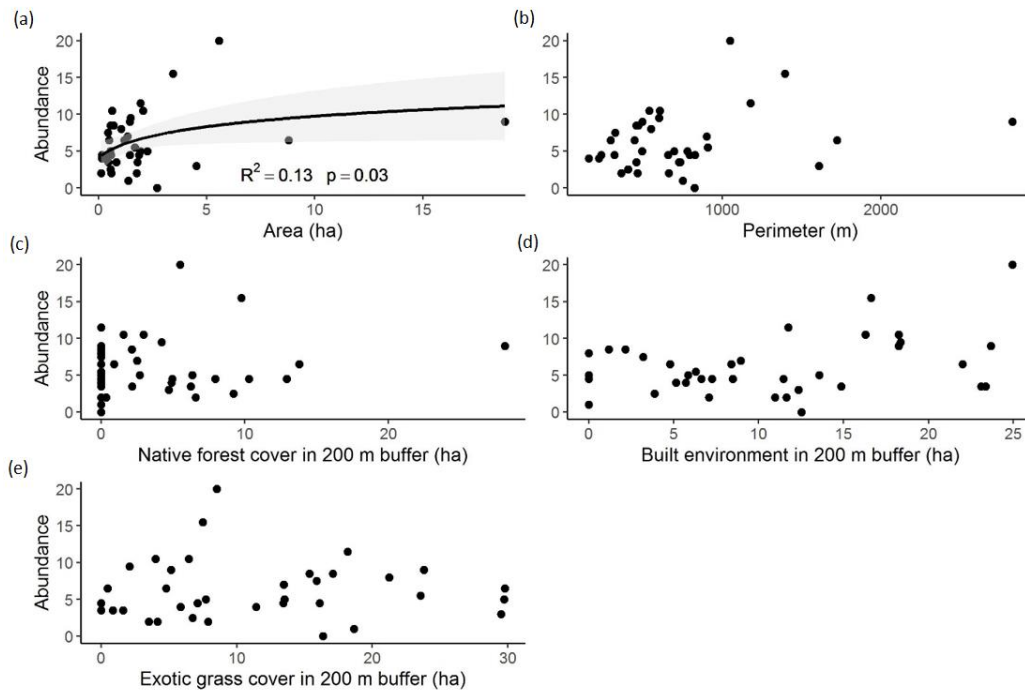


Figure 3.18: Mean spring native bird abundance (no. of individuals per 0.4 ha) and landscape variables for all urban sites ($n = 37$).

Native bird species richness and abundance had no significant relationship with rat, possum and cat indices of relative abundance (Appendix 3.6).

The final native bird species richness structural equation model fits the data well (AIC = 104.14, Fisher's C = 42.14, p-value = 0.83, df = 52), explaining 59% of the variation in native bird species richness (Fig. 3.19 and Table 3.2). The model also explains significant variation in the relative indices of rat (93%) and possum (94%) abundance.

Contrary to expectations, the final native bird species richness model reveals a strong effect of age (i.e., years since initial restoration planting) on native bird species richness that is not explained by the effects of age on measured vegetation variables. No other variables had significant effects on native bird species richness. Two variables, however, had non-significant, negative effects on richness – vegetation complexity and possum relative abundance. Native forest cover within a 200 m buffer zone had a non-significant positive effect on native bird species richness.

As predicted, basal area was greater in older restoration plantings, which in turn decreased canopy openness. Greater canopy openness had a negative effect on native sapling species richness. The model explained 67% of the variation in native sapling species richness. Rat indices of relative abundance and canopy openness had a negative effect on native sapling species richness, while amount of native forest cover in a surrounding 200 m buffer had a positive effect. Increasing age since restoration further increased native tree species richness and vegetation complexity. Rat indices of abundance were higher with higher levels of vegetation complexity, canopy openness and basal area. Rats were negatively correlated with native tree species richness. Possum indices of abundance were negatively affected by canopy openness, amount of native forest cover within a 200 m buffer zone, patch area and amount of built environment within a 200 m buffer zone.

Table 3.2: Summary of results from the structural equation model on native bird species richness. Asterisks denote statistically significant ($p < 0.05$) effect paths.

Response	Predictor	Estimate	Standardised estimate	Standard error	Degrees of freedom	p-value
Native bird species richness	Native forest cover (200 m)	0.06	NA	0.16	20	0.719
Native bird species richness	Possum abundance	-0.10	NA	0.51	20	0.852
Native bird species richness	Complexity index	-0.23	NA	0.21	20	0.277
Native bird species richness	Site age	0.55	NA	0.21	20	0.008*
Native sapling species richness	Native forest cover (200 m)	0.18	NA	0.08	21	0.019*
Native sapling species richness	Canopy openness	-0.17	NA	0.08	21	0.039*
Native sapling species richness	Rat abundance	-0.94	NA	0.39	21	0.017*
Rat abundance	Native canopy species richness	-0.25	-0.43	0.09	20	0.004*
Rat abundance	Complexity index	1.84	0.72	0.38	20	0.000*
Rat abundance	Canopy openness	1.38	0.54	0.57	20	0.016*
Rat abundance	Basal area	1.56	0.61	0.55	20	0.005*
Possum abundance	Complexity index	0.60	0.10	0.48	19	0.217
Possum abundance	Native forest cover (200 m)	-2.04	-0.34	0.45	19	0.000*
Possum abundance	Canopy openness	-5.82	-0.96	1.45	19	0.000*
Possum abundance	Area	-2.57	-0.42	0.79	19	0.001*
Possum abundance	Built environment (200 m)	-1.48	-0.24	0.50	19	0.003*
Canopy openness	Basal area	-0.92	-0.92	0.08	23	0.000*
Complexity index	Site age	0.77	0.77	0.13	23	0.000*
Native canopy species richness	Site age	0.42	NA	0.08	23	0.000*
Basal area	Site age	0.90	0.90	0.09	23	0.000*

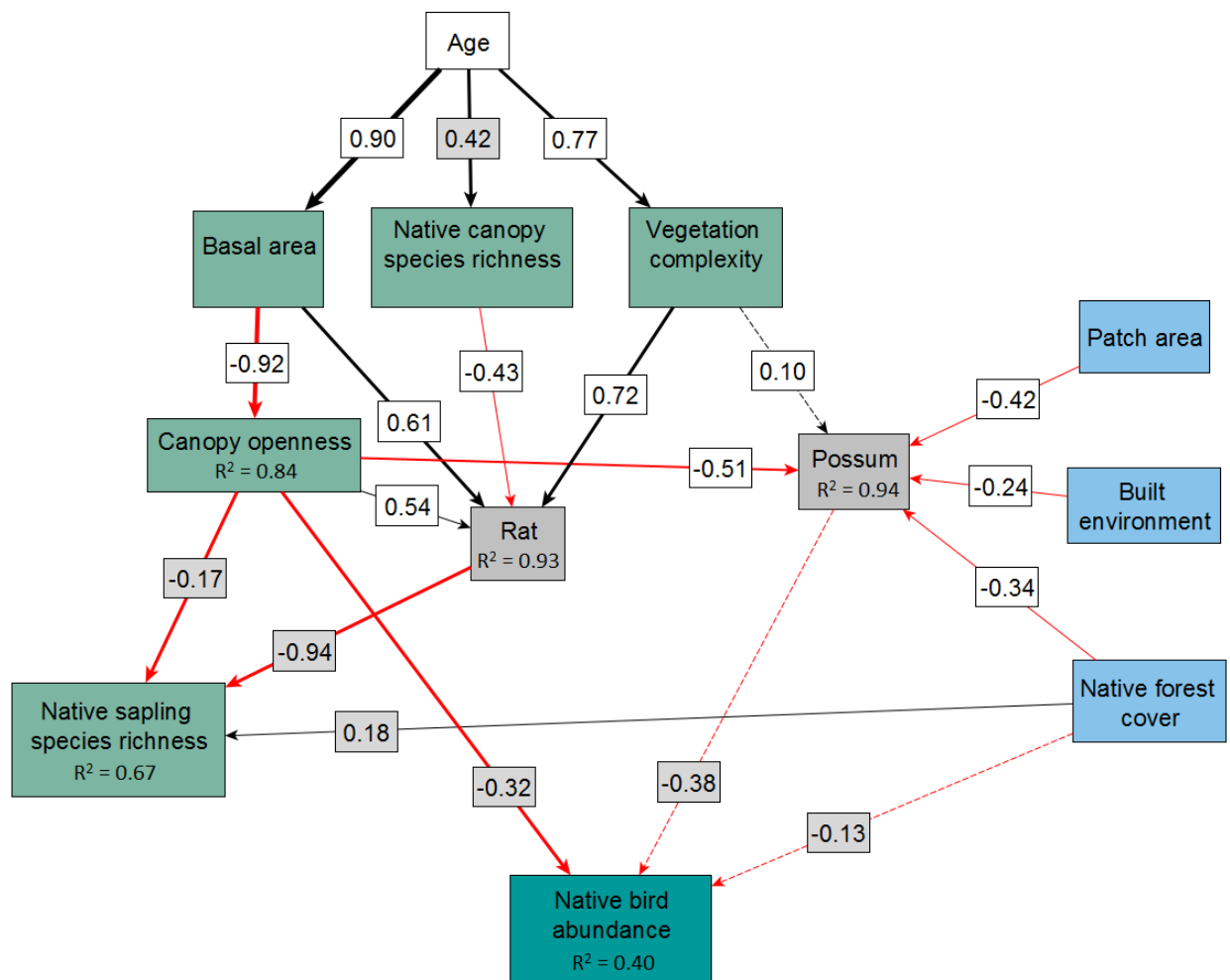


Figure 3.20: Structural equation model (SEM) path diagram, illustrating drivers of native bird abundance. The final native bird abundance model fits the data well (AIC = 100.63, Fisher's C = 40.63, p-value = 0.91, df = 54), explaining 40% of the variation in native bird abundance (Fig. 3.20 and Table 3.3). The model reveals a strong, negative effect of canopy openness on native bird abundance. Possum abundance and native forest cover within a 200 m buffer zone had non-significant, negative effects on native bird abundance.

Table 3.3: Summary of results from the structural equation model on native bird abundance. Asterisks denote statistically significant ($p < 0.05$) effect paths.

Response	Predictor	Estimate	Standardised estimate	Standard error	Degrees of freedom	p-value
Native bird abundance	Possum abundance	-0.38	NA	0.34	21	0.259
Native bird abundance	Native forest cover (200 m)	-0.13	NA	0.10	21	0.185
Native bird abundance	Canopy openness	-0.32	NA	0.11	21	0.002*
Native sapling species richness	Native forest cover (200 m)	0.18	NA	0.08	21	0.019*
Native sapling species richness	Canopy openness	-0.17	NA	0.08	21	0.039*
Native sapling species richness	Rat abundance	-0.94	NA	0.39	21	0.017*
Rat abundance	Native canopy species richness	-0.25	-0.43	0.09	20	0.004*
Rat abundance	Complexity index	1.84	0.72	0.38	20	0.000*
Rat abundance	Canopy openness	1.38	0.54	0.57	20	0.016*
Rat abundance	Basal area	1.56	0.61	0.55	20	0.005*
Possum abundance	Complexity index	0.60	0.10	0.48	19	0.217
Possum abundance	Native forest cover (200 m)	-2.04	-0.34	0.45	19	0.000*
Possum abundance	Canopy openness	-5.82	-0.96	1.45	19	0.000*
Possum abundance	Area	-2.57	-0.42	0.79	19	0.001*
Possum abundance	Built environment (200 m)	-1.48	-0.24	0.50	19	0.003*
Canopy openness	Basal area	-0.92	-0.92	0.08	23	0.000*
Complexity index	Site age	0.77	0.77	0.13	23	0.000*
Native canopy species richness	Site age	0.42	NA	0.08	23	0.000*
Basal area	Site age	0.90	0.90	0.09	23	0.000*

3.4 Discussion

Our study is one of the first in New Zealand to explore the responses of native birds to urban forest restoration. As hypothesized, we found that after initial restoration planting, native bird species richness and diversity increased over time. Similarly, Wilson *et al.* (2014) found more native bird species in shrubland than grassland in their study of the effects of secondary woody succession on the bird community in New Zealand drylands. This increase in native species richness with site age suggests that older sites provide a greater variety of resources, meeting the needs of more species over time. While detection of these species at restored sites indicates that they are using resources within the sites, further studies are needed to see if these restored areas can support the long-term survival and successful breeding of native forest birds.

Contrary to our predictions, we found no significant effect of time since restoration on native bird abundance. This suggests that as they begin to utilize a restored area, certain native species are reducing the numbers of others – i.e., there is no change in bird numbers, only a change in what bird species make up those numbers. It may further suggest that restoration plantings have reached carrying capacity in terms of the numbers of native birds they are capable of supporting.

Differences in native bird species richness and abundance between spring and winter counts reflect changes in the way the birds utilize restoration plantings between seasons. Tūī, for example, were more abundant in urban sites in the winter, a trend found in previous studies (Stevens, 2006). Tūī will travel large distances (> 20 km) over the non-breeding season in search of food and may congregate at sources of abundant fruit or nectar (Stevens, 2006; Fitzgerald & Innes, 2015; Burge *et al.*, 2017). Our observations were consistent with this finding, with tūī found in groups of ten or more individuals feeding on kahikatea (*Dacrycarpus dacrydioides*) fruit in urban forest patches (*pers. obs.*). In the breeding season (September – January), mated pairs defend smaller territories around nest sites and their foraging range decreases to approximately 0.5 km (Bergquist, 1985; Stevens, 2006; Fitzgerald & Innes, 2015). Thus, our detection of tūī in urban forests during the spring counts suggests they are using these areas to breed. Further research is needed on the success of these breeding attempts and on the

survival of the young. Silvereyes were also more abundant in winter, when the species forms small flocks (Heather & Robertson, 2015).

Our results align with the findings of previous studies (Belder *et al.*, 2018), revealing that the bird community in urban restored areas changes over time since initial restoration planting. Newly restored sites, characterised by high canopy openness and exotic grass cover, provide a habitat for introduced granivores and feeding generalists, such as starlings, greenfinches, goldfinches and house sparrows. Over time, as tree height and DBH increase and the canopy closes, exotic grass cover decreases and litter cover increases, the bird community changes as well, increasing in native forest bird species richness.

These patterns are not linear and clear cut along the restoration chronosequence, however, and we suspect they are greatly influenced by the small size of most of the restored areas and urban remnants. The vast majority (32 out of 37) of our urban study sites were < 3 ha and were thus likely to be strongly influenced by edge effects (Drinnan, 2005; Palmer *et al.*, 2008). The small area of our study sites and limited amount of native forest cover in surrounding buffer zones may have restricted our ability to detect the effects of patch size and landscape composition on native forest bird communities. Previous overseas studies identified a forest size threshold of approximately 3.5-5 ha, below which species richness of forest birds rapidly declines (Drinnan, 2005; Kang *et al.*, 2015). While our data support these findings, we stress that only five sites in our study were over 3 ha, four of which were urban remnants. Limited by cost, time and the availability of land, the overwhelming majority of restoration plantings in urban areas will be small habitat patches (Belder *et al.*, 2018). Thus, small patch size, the pressure of edge effects and the limited availability of large areas of native forest in the surrounding landscape must be taken into account when considering the goals of urban restoration projects. Further research is needed to better understand the minimum forest area requirements of native forest birds in New Zealand cities. This will not only include patch size, but shape, spatial configuration and connectivity as well.

Our structural equation model demonstrated that time since initial restoration planting was the single measured variable with a strong significant effect on native bird species

richness. This effect of age was not mediated by tree basal area, native canopy species richness or vegetation complexity, or by the impact of these vegetation variables on rats and possums. This may suggest that as time since restoration increases, more species have the opportunity to colonize or discover the area (Fernandez-Juricic & Jokimäki, 2001; Derhé *et al.*, 2016).

Alternatively, time since restoration could affect variables not directly measured in this study, such as the provision of nectar, fruit and invertebrates by these restored areas throughout the year. A study in Australia found that stands younger than 50 years did not provide substantial eucalypt floral resources for native birds (Vesk *et al.*, 2008). We recommend future studies investigate the seasonal provision of fruit and nectar in restored urban areas, and how this changes over time since restoration. Invertebrates are also an important food source both during the breeding season, with 96% of birds feeding insects to their young (Bouma *et al.*, 2013), and during winter, when fruit, nectar and foliage resources are scarce (Gray & van Heezik, 2016). Several studies have shown that native insects prefer native plants (Bouma *et al.*, 2013; Gray & van Heezik, 2016; Threlfall *et al.*, 2016) and further research is needed into the relationship between native plants, native insects and native birds in New Zealand. Furthermore, little is known about the response of invertebrates to restoration in New Zealand cities (but see Toft *et al.* (2019)) and how this in turn may be affecting native birds.

Our structural equation model demonstrated that native bird abundance was significantly and negatively influenced by canopy openness, which in turn decreased with time since restoration. Thus, while we did not detect a direct effect of age on abundance, our structural equation model revealed an indirect effect of age, mediated by canopy openness. Previous studies have found that the development of dense canopies is important for providing shelter and food to small birds (Vesk *et al.*, 2008), with an increase in canopy cover promoting bird species richness and abundance (Blair, 2004; Belder *et al.*, 2018). Dense canopy developed in these study sites approximately 18 years after restoration planting (Wallace *et al.*, 2017). Wallace and colleagues (2017) recommend that to achieve successful native tree regeneration, management activities focus on promoting rapid canopy closure. Our results suggest that this may further maximise the number of native birds benefiting from restoration planting.

Contrary to our predictions, our structural equation models found no relationship between the relative abundance of rats and cats and native bird species richness or abundance. We did, however, find non-significant negative effects of the relative abundance of possums on both native bird species richness and abundance. Further studies are needed to see if this relationship becomes significant with a larger sample size.

Rats and cats do prey on native birds and, in the case of rats, on their eggs (van Heezik *et al.*, 2008a; Morgan *et al.*, 2011; Innes *et al.*, 2015). However, we did not directly measure nesting success in this study. We suggest that native bird species richness and abundance were not correlated with the abundance of cats and rats in each study site because both the birds and the predators are operating on a larger spatial scale. Hence, island biogeography theory does not apply at the scale of our study sites, as birds and mammals are not restricted to patches of native vegetation but instead are using and traveling through the matrix as well. Thus, while nesting success, which we did not measure, may be correlated with the predators present at a site, counts of adult birds would not be.

Furthermore, studies suggest three of our widely detected native birds (grey warbler, fantail and silvereye) are capable of coping with a certain level of predation. Ruffell and Didham (2017) found that the abundance of four of the native bird species that they modelled (grey warbler, fantail, silvereye and tomtit) did not differ between landscapes with and without pest control. These findings support those of Innes *et al.* (2004), who observed that the relative abundances of these four species did not increase significantly following pest control.

Although predation by introduced mammals in large portions of native forest may be the most important limiting factor for native forest birds in New Zealand (Innes *et al.*, 2010a; Walker *et al.*, 2017), our results suggest that in urban areas, lack of habitat is an important limiting factor for native forest birds. These results support findings by Walker and colleagues (2017) that in addition to predation, habitat loss may be an important, and at times underestimated, limitation on native forest birds. Similarly, Innes and colleagues (2010) concluded that in areas where very little native forest

remained, habitat restoration would be a necessary precursor to the re-establishment of native birds.

Over the course of this study, we detected a maximum of five native forest bird species at any single urban restored site. A study of the effects of forest fragmentation on native birds in Auckland similarly found only five native species regularly occurring year-round at study sites (Stevens, 2006). This is a depressingly low number compared to similar studies from other countries and reflects New Zealand's unique history of isolation and colonisation. The arrival first of Māori, then European settlers introduced mammalian predators to islands that previously had none, resulting in the extinction of 49% of land bird species (Anderson *et al.*, 2011). A further 16 are extinct on the mainland, surviving only in predator free off-shore islands or fenced sanctuaries (Walker & Monks, 2017).

None of the native birds detected in urban restored areas were of conservation concern⁶ but instead are relatively widespread and common on the New Zealand mainland. These species are either able to move large distances and therefore use a number of forest patches to meet their needs (e.g., tūi and kererū), or are small omnivorous or insectivorous birds that are capable of using non-forest habitat, such as gardens and hedgerows (e.g., silvereyes, grey warblers and fantails) (Innes & Fitzgerald, 2018). The native birds detected in urban areas in this study are also less vulnerable to predation (Innes *et al.*, 2004; Burge *et al.*, 2017).

By far the most abundant, and ubiquitous, native bird across all 42 study sites was the silvereye. A relatively recent addition to New Zealand's avifauna, the silvereye colonised the islands from Australia in the 1850s. Studies have shown that silvereyes are the most numerous native garden bird in New Zealand (Spurr, 2012; Gray & van Heezik, 2016) and our results suggest it may be the most abundant, ubiquitous native bird in urban greenspaces. Heather and Robertson (2015) report that the biomass of silvereyes in some forests exceeds that of any other bird species. They caution that as seed dispersers and competitors for fruit, nectar and invertebrates, they are likely to have a significant impact on native forest habitats. Kelly and colleagues (2006)

⁶ I use the term 'birds of conservation concern' to describe native birds classified by the New Zealand Threat Classification System (NZTCS) as 'threatened' or 'at risk' (Robertson *et al.*, 2017)

concluded that the colonisation of New Zealand by humans shifted bird-plant mutualisms from currently threatened and declining endemics, such as stitchbirds (*Notiomystis cincta*), saddlebacks (*Philesturnus rufusater* and *Philesturnus carunculatus*) and bellbirds, to the ubiquitous silvereye. According to Kelly and colleagues (2010), silvereyes are now important for pollination of small flowers and dispersal of small fruits.

While it appears that native bird species richness and diversity in restored areas are progressing towards what is found in urban forest remnants, there is still a gap between the oldest restored areas and urban remnants. Thus, it may take more than 71 years since initial restoration planting for a forest to provide the same quality of habitat as remnant forest in New Zealand's North Island cities. Similar studies looking at changes in bird communities over time since restoration concluded that it may take many decades before the benefits for native wildlife of restoration plantings become evident (Mac Nally *et al.*, 2010; Stagoll *et al.*, 2010).

Bird community composition of rural remnants differed significantly from all other site types. Rural remnants supported higher numbers of grey warblers, fantails and kererū, and were the only sites where we detected whiteheads, tomtits and bellbirds. Seven decades of restoration can increase the number of native bird species able to use resources provided by an urban forest, but there may be no replacing the habitat value of large tracts of native forest to New Zealand's birds (Walker *et al.*, 2014).

In urban areas of New Zealand, characterised by extreme deforestation and habitat modification, increasing the area of native forest through restoration plantings is necessarily the first step in re-establishing our native forest bird communities. Restoration has always been meant to supplement conservation, and our results support the conclusion that we cannot afford to lose the few remaining forest remnants within our cities, as well as those beyond the city limits. Current restoration efforts can benefit widespread, common native birds, such as silvereyes, fantails, grey warblers and tūī, shifting the bird communities in restored sites over time from introduced granivores and generalists to adaptable natives, thereby advancing the Department of Conservation's goal of indigenous dominance within these areas (Lee *et al.*, 2005; Innes & Fitzgerald, 2018). To date, however, restoration in New Zealand cities

involving only replanting of native vegetation, without predator proof fences and reintroductions of locally extinct endemics, is not benefiting birds of conservation concern.

3.5 Appendices

Appendix 3.1

APPENDIX 3.1. Table 1. List of research sites with city, site type, years restored (as of 2017) and area in hectares.

City	Site	Site type	Years restored	Area (ha)
Hamilton	Sandford Unrestored	unrestored	0	1.77
Hamilton	Totara Park	unrestored	0	2.07
Hamilton	Swampy Creek	unrestored	0	4.54
New Plymouth	Te Henui 3 Cemetery	unrestored	0	0.56
New Plymouth	Te Henui 5 Adams Point	unrestored	0	0.6
New Plymouth	Camden	unrestored	0	0.83
Hamilton	Waiwhakareke 2016	restored	1	2.72
Hamilton	Waiwhakareke Young	restored	5	2.26
New Plymouth	Living Legends	restored	6	1.39
New Plymouth	Te Henui 1	restored	8	0.63
Hamilton	Hukanui Primary School	restored	9	0.17
Hamilton	Avalon Drive	restored	9	0.46
New Plymouth	Peringa Park	restored	9	0.7
New Plymouth	Te Henui 2	restored	10	0.17
Hamilton	Waiwhakareke Old	restored	13	1.06
New Plymouth	Herekawe Walkway	restored	13	0.56
Hamilton	Featherstone Park	restored	14	0.52
Hamilton	Hamilton Lake Young	restored	16	0.5
Hamilton	Sandford Restored	restored	17	0.13
Hamilton	Brymer Park	restored	19	1.2
Hamilton	Till's Lookout	restored	20	0.44
New Plymouth	Airport Turnoff	restored	25	1.69
Hamilton	Tauhara Park	restored	27	0.25
New Plymouth	Waipu Lagoon	restored	27	1.35
Hamilton	Morris Gully	restored	34	1.88
Hamilton	Minogue Park	restored	35	1.94
New Plymouth	Huatoki Restored	restored	43	0.59
Hamilton	Edgar Gully	restored	44	0.54
Hamilton	Seeley's Gully	restored	54	1.49
Hamilton	Hamilton Lake Old	restored	62	1.45
New Plymouth	Fred Cowling Reserve	restored	72	1.95
Hamilton	Hillcrest Park	urban remnant	n/a	1.46
Hamilton	Claudlands Bush	urban remnant	n/a	5.59
New Plymouth	Barrett Lagoon	urban remnant	n/a	8.79
Hamilton	Hammond Park	urban remnant	n/a	3.45
New Plymouth	Sheppards Bush	urban remnant	n/a	1.8
New Plymouth	Huatoki Mature	urban remnant	n/a	18.8
Hakarimata Range	3 sites	rural remnant	n/a	1,800
Kaitake Range	3 sites	rural remnant	n/a	2,400

Appendix 3.2

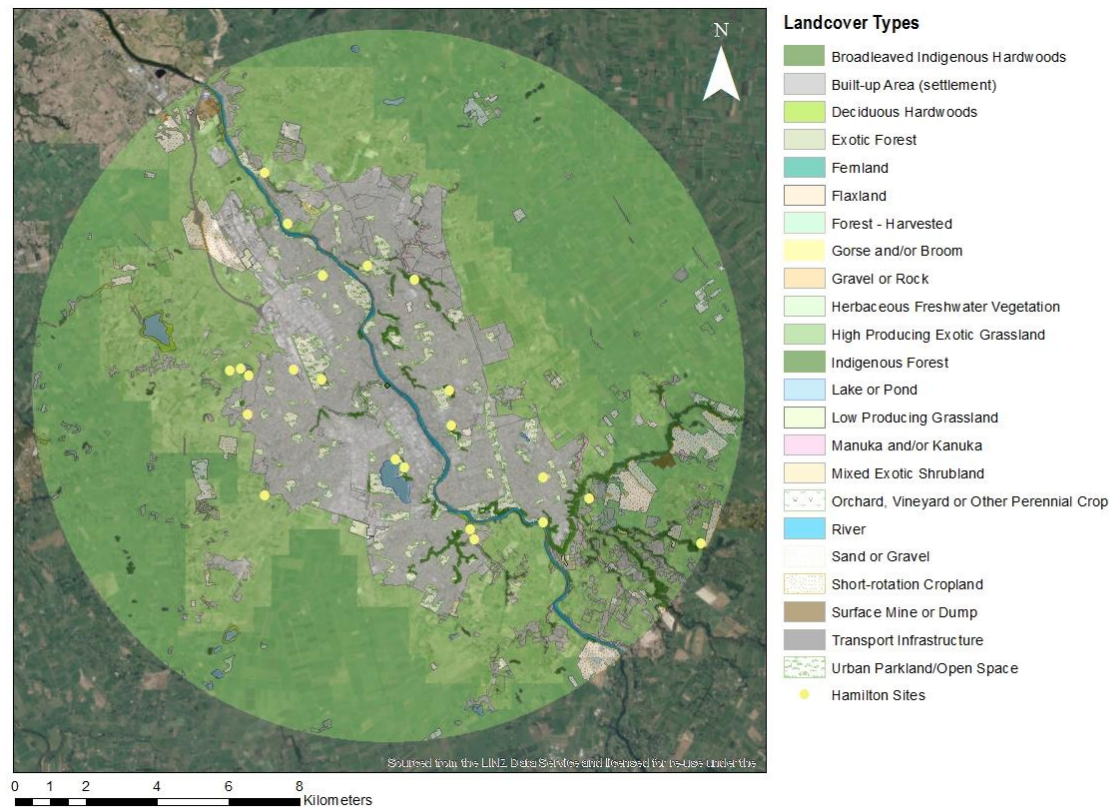
Because different bird species operate on different spatial scales, landscape composition was considered according to five scales:

- 1) That of the individual site and a 200 m and 400 m radius buffer zone around the periphery of the site (Fig. 1). Patches of native vegetation within this range were found by Deconchat *et al.* (2009) to have the strongest positive effect on bird populations on Banks Peninsula, New Zealand.



APPENDIX 3.2. Figure 1. Example research site (Hammond Park, a native forest remnant in Hamilton City) with 200 and 400 m buffer zones shown.

- 2) A 3 km buffer zone around the periphery of the patch. The 3 km buffer zone was based on the maximum natal dispersal capabilities of the grey warbler, a small native bush bird known to be found in urban habitats, that has the shortest known maximum dispersal distance of New Zealand forest birds detected in this study (Gill, 1982; Anderson, 2013).
- 3) A 10 km buffer zone centred approximately in the middle of Hamilton and New Plymouth (Fig.2). This zone was chosen to encompass the built-up areas of the two cities.



APPENDIX 3.2. Figure 2. Map of Hamilton City showing the 10 km buffer zone centred in the middle of the city area. Landcover types are categorized according to the New Zealand Land Cover Database v. 4.1 (LCDB).

- 4) A 100 km buffer zone centred approximately in the middle of each city. This distance represents the maximum seasonal dispersal distance of tūi (Robertson, 2013) and encompasses large areas of native forest around Hamilton and New Plymouth, such as Egmont National Park and the Hakarimata Range Scenic Reserve. Of all New Zealand forest birds known to occur in urban habitats, the kererū has the greatest known dispersal distance, with a satellite-tagged pigeon in Southland travelling over 480 km (Powlesland *et al.*, 2011). However, tagged kererū in Taranaki travelled up to 60 km to food sources (Powlesland, 2013), in Pelorus Bridge Scenic Reserve up to 20 km to other areas of native forest (Clout *et al.*, 1991) and in Whirinaki Forest Park up to 24 km between forest types (Hill, 2003). Therefore, a buffer zone of 100 km was deemed large enough to encompass movements of kererū.

Appendix 3.3

APPENDIX 3.3. Table 1. Aggregated categories and their corresponding original LCDB categories.

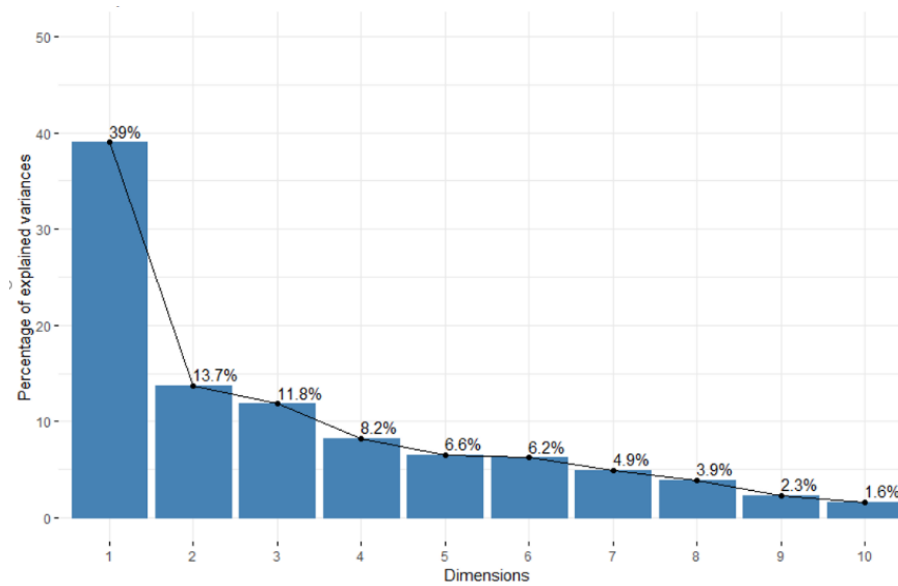
Aggregated category	Original LCDB categories
Native forest	Mangrove Broadleaved indigenous hardwoods Indigenous forest
Built environment	Surface mine or dump Built-up area (settlement) Transport infrastructure
Exotic crop and grassland	High producing exotic grassland Low producing grassland Urban parkland/open space Short-rotation cropland
Exotic forest	Exotic forest, Forest – harvested Orchard, vineyard or other perennial crop Deciduous hardwoods
Water	Lake or pond River Estuarine open water
Native scrub	Manuka and/or kanuka Alpine grass/herbfield Fernland Flaxland Matagouri or grey scrub Sub alpine shrubland Tall tussock grassland
Exotic scrub	Gorse and/or broom Mixed exotic shrubland
Gravel	Sand or gravel Gravel or rock Landslide
Native wetland	Herbaceous freshwater vegetation Herbaceous saline vegetation

Appendix 3.4

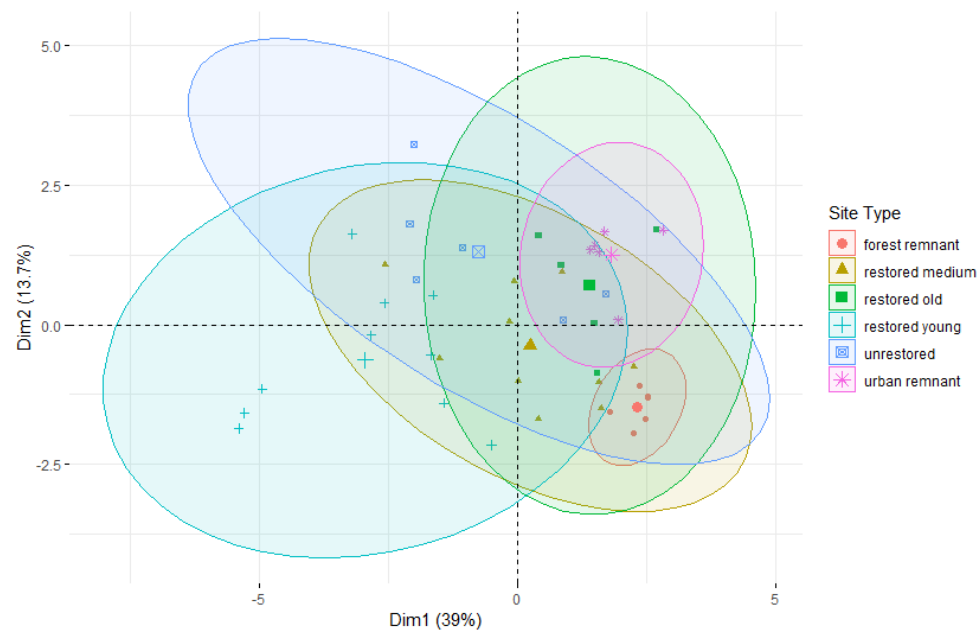
APPENDIX 3.4. Table 1. Names, taxonomic order and New Zealand status of birds recorded over the course of the study.

Common name	Scientific name	Order	Status
Australian magpie	<i>Gymnorhina tibicen</i>	Passeriformes	Introduced
Bellbird	<i>Anthornis melanura</i>	Passeriformes	Endemic
Blackbird	<i>Turdus merula</i>	Passeriformes	Introduced
California quail	<i>Callipepla californica</i>	Galliformes	Introduced
Chaffinch	<i>Fringilla coelebs</i>	Passeriformes	Introduced
Common myna	<i>Acridotheres tristis</i>	Passeriformes	Introduced
Common pheasant	<i>Phasianus colchicus</i>	Galliformes	Introduced
Common redpoll	<i>Carduelis flammea</i>	Passeriformes	Introduced
Common starling	<i>Sturnus vulgaris</i>	Passeriformes	Introduced
Dunnock	<i>Prunella modularis</i>	Passeriformes	Introduced
Eastern rosella	<i>Platycercus eximius</i>	Psittaciformes	Introduced
Eurasian skylark	<i>Alauda arvensis</i>	Passeriformes	Introduced
Goldfinch	<i>Carduelis carduelis</i>	Passeriformes	Introduced
Greenfinch	<i>Carduelis chloris</i>	Passeriformes	Introduced
Grey warbler	<i>Gerygone igata</i>	Passeriformes	Endemic
House sparrow	<i>Passer domesticus</i>	Passeriformes	Introduced
New Zealand fantail	<i>Rhipidura fuliginosa</i>	Passeriformes	Endemic
New Zealand kingfisher	<i>Todiramphus sanctus</i>	Coraciiformes	Native
New Zealand pigeon	<i>Hemiphaga novaeseelandiae</i>	Columbiformes	Endemic
Shining cuckoo	<i>Chrysococcyx lucidus</i>	Cuculiformes	Native
Silvereye	<i>Zosterops lateralis</i>	Passeriformes	Native
Song thrush	<i>Turdus philomelos</i>	Passeriformes	Introduced
Spotted dove	<i>Streptopelia chinensis</i>	Columbiformes	Introduced
Tomtit	<i>Petroica macrocephala</i>	Passeriformes	Endemic
Tūī	<i>Prothemadera novaeseelandiae</i>	Passeriformes	Endemic
Welcome swallow	<i>Hirundo neoxena</i>	Passeriformes	Native
Whitehead	<i>Mohoua albicilla</i>	Passeriformes	Endemic
Yellowhammer	<i>Emberiza citrinella</i>	Passeriformes	Introduced

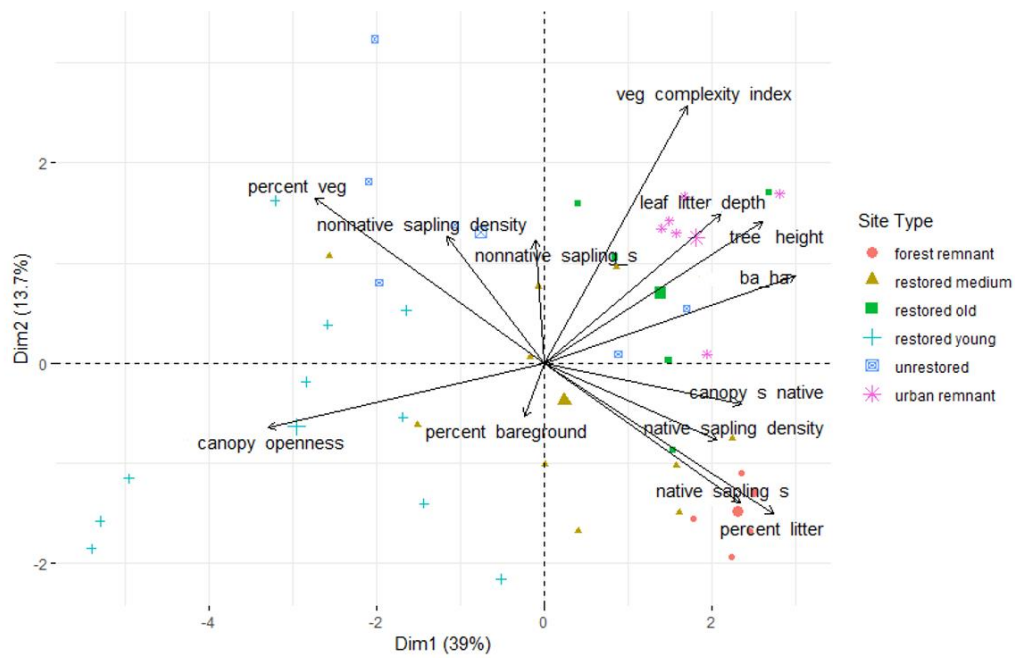
Appendix 3.5



APPENDIX 3.5. Figure 1: Scree plot for principal component analysis of the vegetation data. The percent of variance (the eigenvalue) explained by each component is shown in decreasing order.

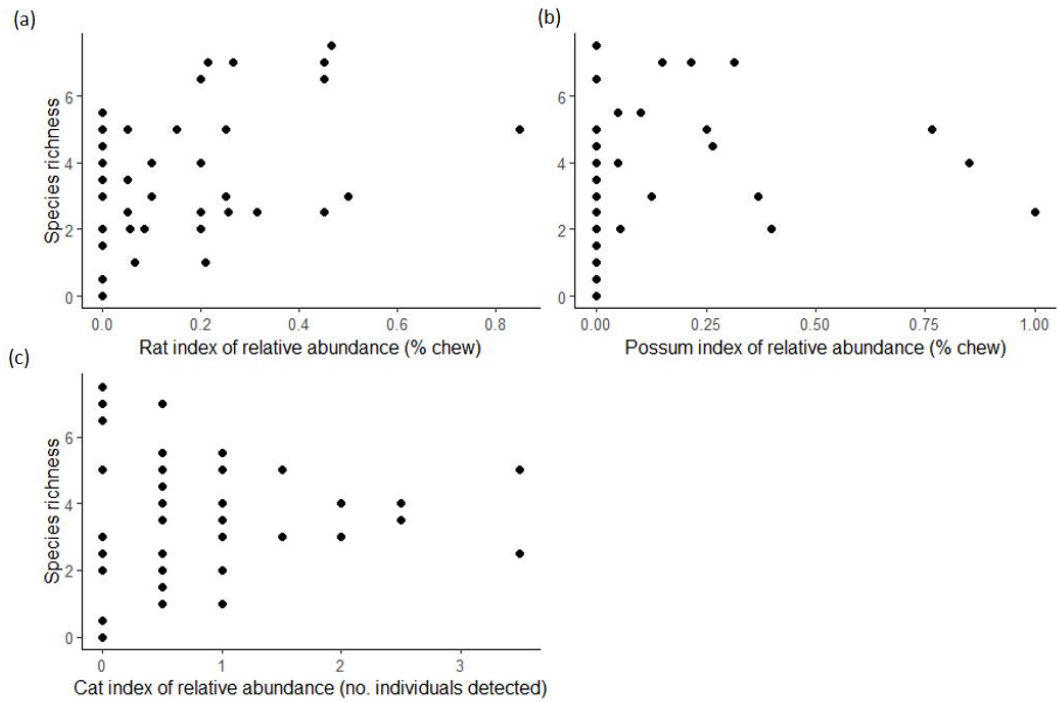


APPENDIX 3.5. Figure 2: Principal component analysis (PCA) biplot of vegetation variables with concentration ellipses demarcating the six site types. Each point represents a site and colours and shapes represent different site types. Restored sites were divided into young (1-13 years since initial restoration planting), medium (14-35 years) and old (41-72 years).

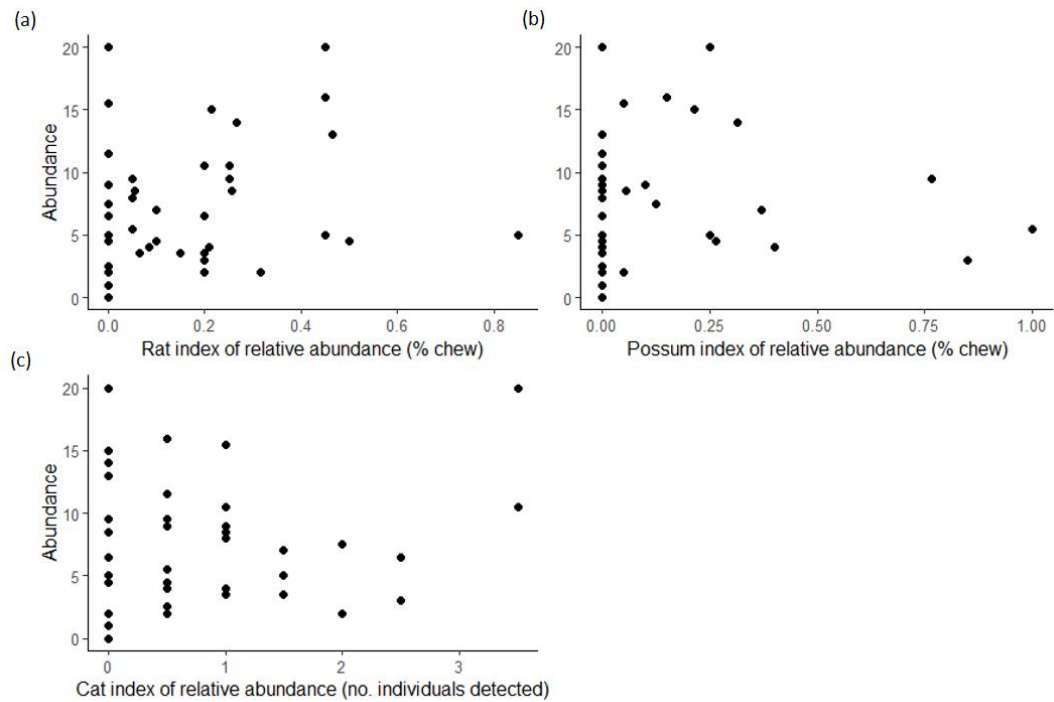


APPENDIX 3.5. Figure 3: Principal component analysis (PCA) biplot of vegetation variables. Each point represents a site and colours and shapes represent different site types. Restored sites were divided into young (1-13 years since initial restoration planting), medium (14-35 years) and old (41-72 years). Vectors denote strength of influence of individual variables on the first two PCA axes.

Appendix 3.6



APPENDIX 3.6. Figure 1. Mean spring native bird species richness and pest variables for all sites ($n = 43$). Scatterplot points represent individual sites.



APPENDIX 3.6. Figure 2. Mean spring native bird abundance (no. of individuals per 0.4 ha) and pest variables for all sites ($n = 43$). Scatterplot points represent individual sites.

Chapter 4: Changes in introduced mammal communities along an urban forest restoration chronosequence result in unforeseen outcomes for restoration

4.1 Introduction

Biological invasions are viewed as the second most important cause of global biodiversity loss, following habitat destruction and fragmentation (Vitousek *et al.*, 1997; Courchamp *et al.*, 2003). Island ecosystems are particularly vulnerable to invasive species, which have been the major cause of island vertebrate extinctions (Caut *et al.*, 2009). New Zealand's unique flora and fauna evolved for millennia in the complete absence of terrestrial mammalian predators (King, 2005). Since human colonization, 55 mammal species were introduced to the island archipelago, 32 of which have become established (Courchamp *et al.*, 2003). New Zealand cities are now home to top mammalian predators such as cats (*Felis catus*) and mustelids (weasels, *Mustela nivalis vulgaris*; ferrets, *Mustela furo* and stoats, *Mustela erminea*), commensal mesopredators such as rats (ship rats, *Rattus rattus* and Norway rats, *Rattus norvegicus*) and house mice (*Mus musculus*), an insectivore (the European hedgehog, *Erinaceus europaeus*), an omnivore (Australian brushtail possum, *Trichosurus vulpecula*) and a herbivore (European rabbit, *Oryctolagus cuniculus*) (Morgan *et al.*, 2009). Thus, New Zealand now has a unique assemblage of introduced mammals interacting with each other and with native species to carve out habitats for themselves within the highly altered and understudied environment of New Zealand cities.

Urbanization dramatically alters assemblages of vertebrate wildlife species, resulting in animal communities that differ greatly from adjacent, more natural environments (Fischer *et al.*, 2012). Through extreme habitat destruction and fragmentation, the introduction of invasive species and the provision of anthropogenic food sources, urbanization has the potential to drastically alter interspecific interactions and other

mechanisms that shape vertebrate communities (Shochat *et al.*, 2006). Previous overseas studies have shown that urban environments support high densities of generalist, opportunistic predators, such as corvids and raccoons (Rodewald *et al.*, 2011; Fischer *et al.*, 2012). The one published study of mammals in a New Zealand city found that cats and hedgehogs were ubiquitous but detections of rats and possums were largely confined to gully habitats (Morgan *et al.*, 2009).

Research further suggests urbanization decouples predator-prey relationships by increasing the carrying capacity of cities through the provision of anthropogenic food sources (Rodewald *et al.*, 2011). This has been termed the predator paradox; predators are found at higher densities in cities, but predation pressure on vertebrate prey is lower than in surrounding non-urban environments (Fischer *et al.*, 2012). It is hypothesized that year-round, easy access to human-provided food can support larger numbers of predators and omnivores, leading to diet shifts away from vertebrate prey (Fischer *et al.*, 2012).

Studies in non-urban environments have highlighted the potential for unexpected and unintended outcomes of conservation and restoration activities caused by complex interactions among invasive and native species (Caut *et al.*, 2009; Ruscoe *et al.*, 2011; Norbury *et al.*, 2013). The challenge to predicting restoration outcomes will be even greater in urban environments of New Zealand, where we have an entirely non-native suite of terrestrial mammals, responding to both urbanisation and the restoration process. Nothing is known about how mammals introduced to New Zealand respond to forest restoration in an urban environment.

The purpose of the following study was to investigate the impact of vegetation structure, landscape composition and interspecific interactions on the composition of introduced mammal communities in a chronosequence of urban forest restoration, representing an age gradient of 1-72 years since initial planting. We also investigated the impact of introduced mammals on forest regeneration, as measured by native sapling species richness and density.

We predicted that the mammal community would change over time since restoration, with young sites supporting higher numbers of mice and hedgehogs (King *et al.*, 1996; Nottingham, 2018), and older sites providing habitat for more rats and possums

(Cowan, 2005; Shiels *et al.*, 2017). We predicted that cats would not be influenced by landscape or vegetation variables, but instead would be abundant throughout the urban areas (Kikillus *et al.*, 2017). We hypothesised that higher densities of cats would decrease the abundance of rats, mice and rabbits within our study sites. Finally, we predicted that rabbits, mice, rats and possums would reduce native forest regeneration through browsing and seed or seedling consumption (Campbell, 1990; Holl & Quiros-Nietzen, 1999; Shiels *et al.*, 2017).

4.2 Methods

4.2.1 Study site

We conducted our research in two cities in New Zealand's North Island, and in the two native forest remnants closest to each city.

Hamilton City (37°46'59.99"S, 175°16'59.99"E) (Fig. 4.1) has an estimated population size of 165,400 (Statistics New Zealand, 2017). The climate is temperate oceanic, with maximum daily temperatures ranging from 21-26°C in summer, and 10-14°C in winter. The mean annual temperature is 13.7°C and mean annual rainfall is 1190 mm (NIWA, 2016). Hamilton City is bisected by the Waikato River and spans a network of steep sided gullies totalling approximately 770 ha (Cornes *et al.*, 2012). This gully system makes up a large proportion of the city's green space (Fitzgerald *et al.*, 2017). The term 'greenspace' is used in this research to describe any vegetated area within the city limits. Despite the gully network, less than 2% of Hamilton is covered by indigenous forest (Clarkson & McQueen, 2004; Clarkson *et al.*, 2007b).

The nearest rural forest remnant is the Hakarimata Range Scenic Reserve (37°39'45.2"S 175°07'28.1"E) (Fig), which lies about 10 km north-west of Hamilton. The range forms the western boundary of the Waikato Basin with its highest point, Hakarimata, at 374 m. The Reserve encompasses 1,850 ha of native lowland forest dominated by broadleaf-podocarp species, such as tawa (*Beilschmiedia tawa*), kohekohe (*Dysoxylum spectabile*), hinau (*Elaeocarpus dentatus*), rewarewa (*Knightia excelsa*), mangeao (*Litsea calicaris*) and pukatea (*Laurelia novae-zelandiae*) (Department of Conservation).

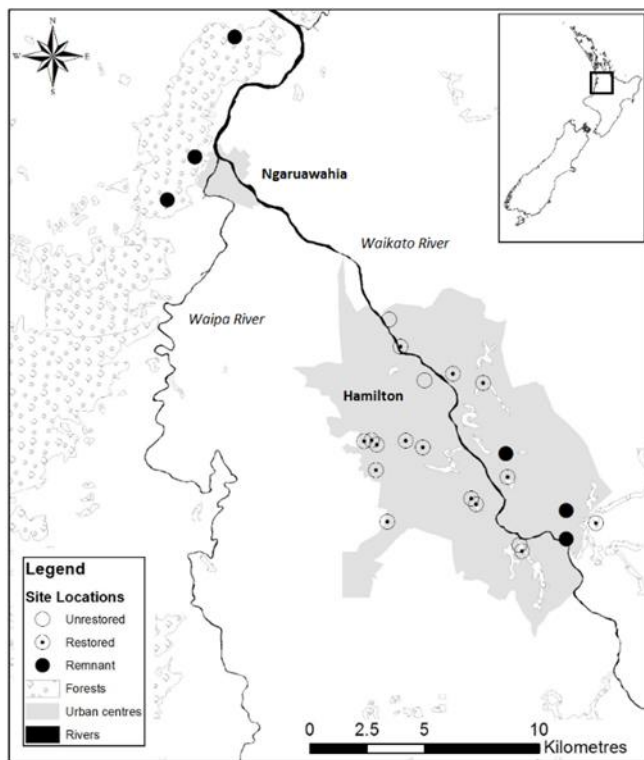


Figure 4.1: Location of the 25 study sites in Hamilton and the Hakarimata Range, which included 16 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in Hamilton, and 3 in the Hakarimata Range.

New Plymouth ($39^{\circ}4'0.01''\text{S}$, $174^{\circ}4'59.99''\text{E}$) (Fig. 4.2) has an estimated population size of 58,300 (Statistics New Zealand, 2017). As in Hamilton, the climate is temperate oceanic with maximum summer temperatures between $19\text{--}24^{\circ}\text{C}$, maximum winter temperatures between $10\text{--}14^{\circ}\text{C}$, a mean annual temperature of 13.7°C and mean annual rainfall of 1432 mm (NIWA, 2016). New Plymouth retains approximately 8.5% of its indigenous forest cover (Clarkson *et al.*, 2007b).

The nearest rural forest remnant to New Plymouth is the Kaitake Range ($39^{\circ}10'0''\text{S}$, $173^{\circ}58'1''\text{E}$), a range of hills encompassing about 2,400 ha of forest, which lie approximately 15 km south-west of the city. The highest point is Patuha at 684 m. The range is covered in lowland and semi-coastal forest, dominated by kohekohe, tawa, and kamahi (*Weinmannia racemosa*) (Clarkson, 1985).

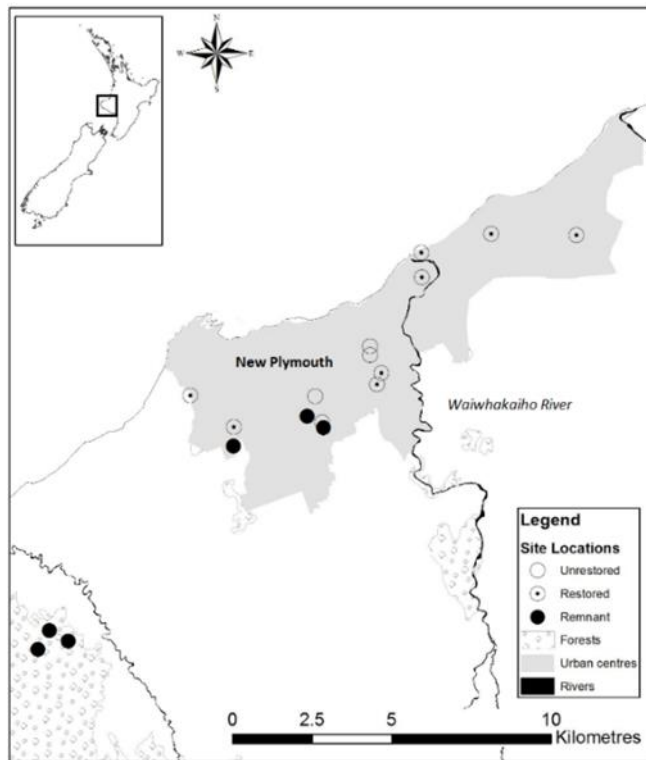


Figure 4.2: Location of the 18 study sites in New Plymouth and the Kaitake Range, which included 9 restored forests (dotted circles), 3 unrestored (empty circles), and 6 remnant forests (filled circles) – 3 in New Plymouth, and 3 in the Kaitake Range.

We conducted our research in urban restored, unrestored and remnant forests in Hamilton and New Plymouth, as well as in the rural forest remnants nearest to these two cities. Restored urban forests are green areas within the urban matrix, dominated by native vegetation that has been intentionally planted. Restored forests (25 from both cities) were chosen to represent an age gradient from 1-72 years since initial planting, as of 2017, and were compared to three types of reference forest:

- 1) Unrestored urban forest (n=6) was defined as a forested area affected by severe anthropogenic disturbance, such as clear-felling or sand mining, that was never subsequently restored, but was left to regenerate naturally. These forests tended to be dominated by exotic plant species.
- 2) Remnant urban forest (n=6) was defined as a patch of native, mature forest within the urban matrix that had never been clear-felled or mined;
- 3) Rural forest remnants (n=6) were large (>1,000 ha) areas of native lowland forest closest to the two cities.

Three sites were randomly located within each rural remnant (the Hakarimata and Kaitake Ranges) by overlaying the ranges with a grid of 10×10 m squares, assigning a number to each square, and randomly choosing three numbers per forest remnant. Three parallel 200 m transects were laid out for each site (the first transect starting at the middle of the randomly selected square), with each transect a minimum of 100 m apart. Thus a ‘site’ in the rural remnants is defined as a grid of three parallel 200 m transects. Sites were a minimum of 100 m apart and were only set up below 300 m above sea level, as the goal was to sample lower altitude lowland forest.

The names, years since planting (as of 2017), forest category and size (ha) are shown in Appendix 3.1.

4.2.2 Predator monitoring

Indices of relative abundance and presence/absence data on introduced mammals within each site were collected using chew cards for rats (ship rats, Norway rats), possums and mice and camera traps for cats, mustelids (stoats, ferrets, weasels), hedgehogs and rabbits. Hereafter, ‘rats’ refers to both species unless otherwise specified, since their bite marks cannot be distinguished on chew cards. The chew cards were 10×20 cm sheets of corrugated plastic with the edges filled with a mix of peanut butter and aniseed. Ten chew cards were spaced at 20-metre intervals along 60-200 m transects (depending on the size of the site) and were deployed for two nights (Sweetapple & Nugent, 2011; Ruffell *et al.*, 2015a; Ruffell *et al.*, 2015b). The chew cards were nailed to the nearest tree at 20-metre intervals, approximately 30 cm off the ground. In younger sites where the trees were too small, chew cards were mounted on metal rods. A relative index of rat, possum and mouse abundance was calculated as the proportion of chew cards bitten by rats, possums and mice, respectively.

Three baited camera traps at small sites (< 5 ha) and 5 at larger ones (≥ 5 ha) were set up around the periphery of each site a minimum of 1 m from the edge, at roughly even intervals (typically 40-100m apart), facing inwards, to maximize the likelihood of animals entering or leaving the site detecting and investigating the camera trap lures (Lincoln, 2016). Camera traps were left out for two nights. We used 2016 Browning

Strike Force Elite cameras, which have a picture trigger and recovery speed of 0.65 seconds and 1.3 seconds, respectively, and a detection range of 15 m.

Camera trapping protocol largely followed that used by Glen *et al.* (2014). The lure consisted of cotton wool soaked in fish oil, placed in a perforated plastic vial and pegged to the ground 1.5 m in front of the cameras. Cameras were mounted on trees so that the lens was 10 cm above the ground. Cameras were programmed to take five still photographs when motion triggered. Photographs of the same species less than one minute apart were counted as a single detection.

In order to estimate the relative abundance of target species, a distance of 1.5 m in front of the camera was measured and marked with the lure to standardise the area monitored by each camera. Animals photographed beyond the 1.5 m zone were not included in measures of relative abundance but were recorded as species present at the study site. A relative index of cat abundance was calculated as the number of individually distinguishable cats detected over each 48-hour period. For other species, the index used was the number of triggers over 48 hours.

4.2.3 Predator control

Rats and possums were controlled at the majority of study sites (37 out of 43), and rabbits at 3 sites out of 43. Control methods varied widely between sites, however.

For the past 5 years, Hamilton City Council has been using Timms traps and pindone bait stations to control rats and possums, and magtoxin and shooting (at one site) to control rabbits. Bait stations and traps are moved throughout the year from one group of Hamilton parks to another, where they are left for approximately 3 months. Community groups conducted pest control at two Hamilton sites – at Hammond Park, volunteers are using Goodnature® traps for rats and mice. Data could not be obtained for the second group.

The New Plymouth District Council has been using pindone bait stations and DOC 200 traps to control possums, rats, and mustelids at the study sites for the past 5 years. At one site, they are also using Goodnature® traps for rats and mice. Pest control is conducted every 2 months.

For the past 5 years, a contractor to Fulton Hogan (a large infrastructure construction, roadworks and aggregate supplier company) has been conducting rat and possum control in 106 ha in the south-eastern section of the Hakarimata Ranges. The contractor uses pindone, diphacinone and cholecalciferol bait stations and leg-hold traps every year from September to March, which encompasses the breeding season of most native forest birds. Moreover, fur recovery operators carry out sporadic possum control using leg-hold traps throughout the reserve.

The Department of Conservation aerially distributed baits containing sodium fluoroacetate (compound 1080) in the Kaitake Ranges in December 2016 and January 2017 to control possums, rats and mustelids. A local community group has also been controlling rats and mustelids with a network of DOC 200 traps.

Due to insufficient information on the timing of control and spacing of devices, we were unable to assign a reliable index of level of predator control to each study site. We therefore caution that the effects of vegetation, landscape and interspecific competition or predation will interact with predator control methods to determine mammalian presence and abundance at a site.

4.2.4 Vegetation measurement

To quantify potential resources for foraging and shelter, we measured vegetation structure and composition within three randomly located 10×10 m² plots within each site. These plots had been set up by Wallace *et al.* (2017) at all sites, apart from the rural forest remnants, one urban remnant (Hillcrest Park), and one young restored forest (Waiwhakareke, planted in 2016). Plots were a minimum of 1 m away from each other and from the forest edge. Due to the small size of the urban forests, edge effects cannot be avoided and will likely prevail throughout the majority of sites (van Heezik *et al.*, 2008b).

The structural complexity of the vegetation was measured using a complexity index developed by Landcare Research for the Department of Conservation (Hurst & Allen, 2007). The complexity index divides the vegetative cover into 6 vertical strata: <0.3, 0.3-2, 2-5, 5-12, 12-25, >25m above ground. Percent vegetation cover in each stratum

is visually estimated and coded as: 1 (less than 1%), 2 (1-5%), 3 (6-25%), 4 (26-50%), 5 (51-75%), 6 (76-100%). The values for the three plots were averaged for each stratum and the average values for each site summed up to give the complexity index, ranging from 0-36.

Ground cover was visually estimated and recorded (to the nearest 5%) as percent cover comprised of vegetation, non-vascular plants, litter, bare ground and rock below 1.35 m within each plot (for category definitions, see Hurst & Allen 2007). The values for the three plots were averaged to give a single value per site. Tree heights were measured using an inclinometer – 7 canopy trees were measured per plot, for a total of 21 trees per site and the average tree height per site was calculated (Bibby *et al.*, 2000). Canopy openness was measured by taking hemispherical photos of the canopy from 1 m above the ground in each plot. Images were analysed using Gap Light Analyzer v. 2.0 (Institute of Ecosystem Studies, Millbrook, New York, USA) and the values from the three plots were averaged to give one value for each site. All live trees ≥ 2.5 cm diameter at breast height (DBH = 1.4 m) rooted within the three plots were identified to species and the DBH of each stem was recorded. DBH data were used to compute total basal area per hectare for each site. Sapling counts were conducted to provide data on the regeneration of tree and shrub species. Saplings are defined as woody plants < 2.5 cm DBH and were tallied by species in one randomly chosen quarter of each plot.

The data on canopy openness, basal area per hectare and sapling species richness and density were collected by Wallace and colleagues from 2015-2016. The same methods were used in the present study for the added sites – the rural forest remnants (6), one urban forest remnant (1) and the 2016 restoration planting (1). Data for these sites was collected in 2017. Data on tree height, structural complexity and ground cover for all sites were collected by the primary author in 2017.

4.2.5 Landscape measures

Landscape variables measured for each site were patch area, perimeter, landscape composition within five buffer zones (see Appendix 3.2) and distance to nearest road and water source. Exploratory data analysis revealed that the 200 m buffer zone was

most strongly correlated with all response variables. Therefore, further analyses only used the 200 m buffer zone.

Landscape composition within the 200 m buffer zones was classified on the basis of satellite imagery and the New Zealand Land Cover Database v. 4.1 (LCDB). The LCDB categories were combined into the following nine land cover categories: native forest, exotic forest, built environment, exotic crop and grassland, water, native scrub, exotic scrub, sand or gravel and native wetland (Appendix 3.3). The area of each land cover type within the 200 m buffer zone was calculated using ArcGIS.

4.2.6 Statistical analysis

All statistical analyses were performed in R version 3.5.2 (R Development Core Team 2018).

Change in mammal communities over time since restoration

We investigated bivariate plots to explore the relationship between time since restoration and relative abundance of the six most commonly detected introduced mammals – rats, possums, mice, hedgehogs, rabbits and cats. We excluded dogs from the analyses, as their presence and numbers will primarily be determined by when and where people choose to walk them.

We tested for differences in indices of relative abundance between the three reference categories (urban remnant, rural remnant and unrestored) using Welch's ANOVA for unequal variances, followed by a Games-Howell post hoc test to determine which groups differed significantly from each other.

To visualise the effect of site type and time since initial restoration planting on relative abundance of introduced mammals, we inspected a bar chart of average numbers of mammals detected using camera traps for spring and winter data. We further divided

the restored sites into young (1-10 years since initial restoration planting), medium (13-27 years) and old (34-72 years) and plotted the data by both species and site type.

For a visual representation of the mammal community composition in unrestored, restored, and remnant forests, we used a nonmetric multidimensional scaling (NMDS) ordination in the *vegan* package in R (Oksanen, 2011), with the Bray-Curtis dissimilarity index. Species relative abundance data from camera triggers were square-root transformed and standardized using a Wisconsin double relativization (Oksanen, 2011). The mean number of triggers for the two spring seasons was used in the first ordination and the number of triggers for winter 2017 was used in the second. Data from the 2018 winter camera traps were excluded from all analyses due to camera malfunction. The data matrix consists of 7 species and 42 samples for spring (one site had to be removed because no predators were detected) and 40 samples for the winter (three sites removed due to lack of predator detections).

We positioned mammal species' names on the NMDS ordinations at the weighted average of site scores, where weights are determined by the relative abundances of each species.

In order to examine which vegetation and landscape variables were associated with the mammal community composition NMDS ordination, we fitted vectors from these variables onto the ordination using the *envfit* function in the *vegan* package in R (Oksanen, 2015). Arrows indicate the direction of the most rapid change in the variable and the length of the arrow corresponds to the relative strength of the correlation between the ordination and plotted variable. We limited plotting to variables with a p -value ≤ 0.05 .

To test for differences in mammal community composition (relative abundance of mammals of different species) between site types, we performed permutational multivariate analysis of variance (PERMANOVA, Anderson 2001), using the *vegan* package in R on the spring and winter data. Analyses were based on Bray-Curtis dissimilarity with 999 permutations. We then determined which site types differed

significantly from each other using the pairwise Adonis function in R (Martinez Arbizu, 2019).0

We evaluated which species contributed most to the significant overall differences in species composition between site types, using similarity percentage (SIMPER) analysis with the vegan package in R.

Relative effects of vegetation, landscape and interspecific interactions on indices of mammal abundance and impacts of mammals on restoration

We constructed bivariate plots to investigate the relationship between vegetation attributes (canopy openness, basal area, tree height, vegetation complexity index, and native tree species richness) and relative indices of mammal abundance for the five most frequently detected mammals – rats, possums, mice, hedgehogs and cats.

We constructed bivariate plots to investigate the relationship between landscape variables (site area, perimeter, distance to water, distance to road and amount of native forest, built environment, exotic crop and grassland, and water in a 200 m buffer area) and relative indices of mammal abundance.

We used structural equation modelling with the piecewiseSEM package in R (Lefcheck, 2016) to evaluate direct and indirect effects and the relative importance of time since initial restoration planting, vegetation composition and structure, landscape variables and mammal abundances on native tree species richness and density.

We used theory, previous studies and expert knowledge to develop an *a priori* model of hypothesized relationships between our measured variables (Fig 4.3). We hypothesized that age would have a positive effect on tree basal area (Wallace *et al.*, 2017) and vegetation complexity (Munro *et al.*, 2009). Basal area would in turn have a negative effect on canopy openness (Wallace *et al.*, 2017). Canopy openness was predicted to have a negative effect on native sapling species richness and density (Wallace *et al.*, 2017).

Possums were expected to be influenced by vegetation complexity (Harper *et al.*, 2005), the amount of native forest in the surrounding landscape, canopy openness and patch area (King, 2005). Rats and mice were expected to be negatively affected by cats

(Gordon *et al.*, 2010), and positively affected by the amount of native forest cover in the surrounding landscape and vegetation complexity (King, 2005; Shiels *et al.*, 2017). Rabbits were predicted to be positively affected by canopy openness and negatively by cats (Gillies, 2001; King, 2005).

Rats, possums, rabbits and mice were predicted to have a negative impact on native sapling species richness and density (Holl & Quiros-Nietzen, 1999; Shiels *et al.*, 2017), which we further predicted would be positively affected by amount of native forest cover in the surrounding landscape (Overdyck & Clarkson, 2012).

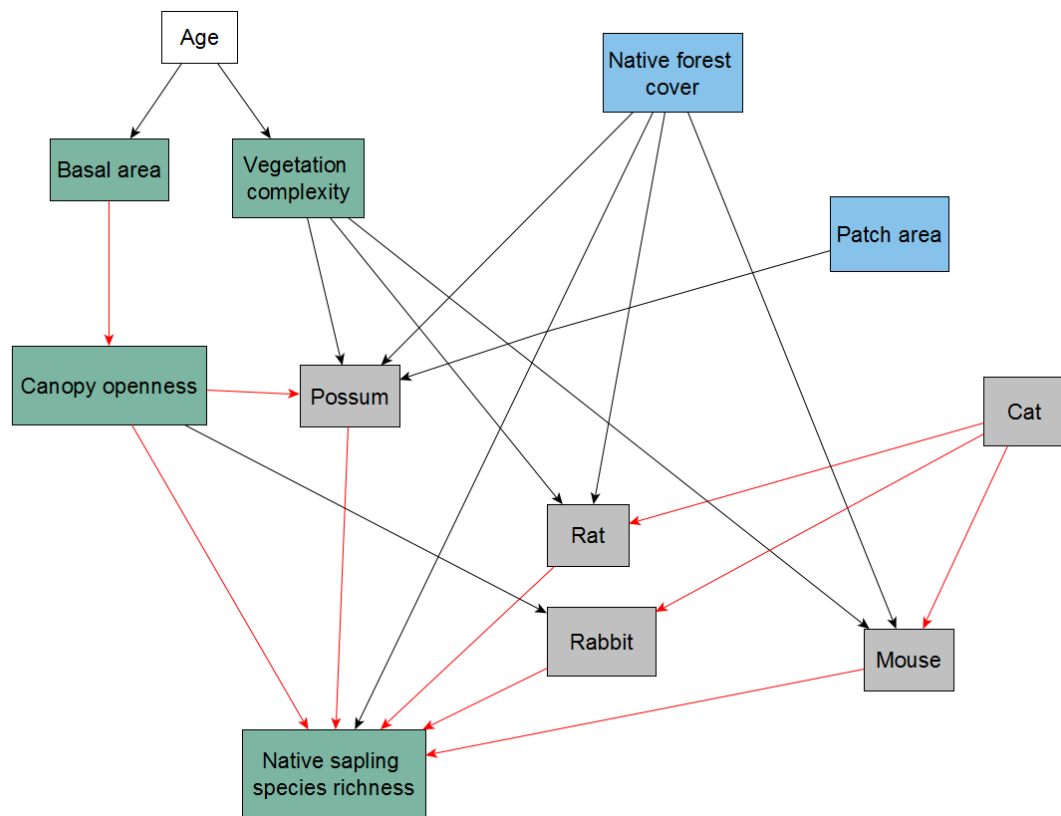


Figure 4.3. Hypothesized a priori structural equation model (SEM) path diagram.

Where necessary, variables were log or square root transformed to meet assumptions of normality, and then scaled to unit variance. We used generalized linear models for possum, rat and mouse indices of relative abundance which were binomially distributed (we used the binomial data on whether a chew card detected a possum, rat or mouse over the 48-hour period as the response variable), and for native canopy species richness, native sapling species richness, and rabbit and cat indices of relative

abundance which were Poisson distributed. We used the same *a priori* model to predict native sapling species richness and density.

Model fit was evaluated using Shipley's test of directed separation (Fisher's C) where p-values > 0.05 indicate that the data do not differ significantly from the specified model. Thus, models with p-values > 0.05 represent the data well.

After constructing the linear and generalized linear models that made up our structural equation model, we specified correlated errors, i.e., pairs of variables that we predicted may appear to be correlated but we did not believe to have a causal relationship. Correlated errors were specified using the “%~%” operator in the `psem` function in the `piecewiseSEM` package in R (Lefcheck, 2018).

We then examined the “tests of directed separation” output in R in order to identify any significant ($p < 0.05$) causal pathways between our variables that we may have overlooked. Once these were added to the model, we removed insignificant ($p > 0.05$) pathways one by one and examined the effect of their removal on the Akaike Information Criterion (AIC). We present the model with the lowest AIC score.

We report the standardised coefficients for each path where possible (for linear models and binomially distributed data), and unstandardised coefficients for Poisson distributed data.

4.3 Results

We detected 12 introduced mammals over the course of our study – ship rat, Norway rat, brushtail possum, house mouse, European hedgehog, European rabbit, brown hare (*Lepus europaeus*), stoat, weasel, feral ferret, domestic cat and dog (*Canis familiaris*, species names follow King, 2005). The stoat, weasel, ferret and hare were each detected only once – the stoat in Hamilton, the weasel in New Plymouth, the ferret in the Hakarimata Range and the hare in the Kaitake Range (Fig 4.5).

Bar charts revealed that rats, mice and cats were the three most widespread mammals across all 43 of our study sites over the three seasons, when considering data from both camera traps and chew cards (Fig 4.4).

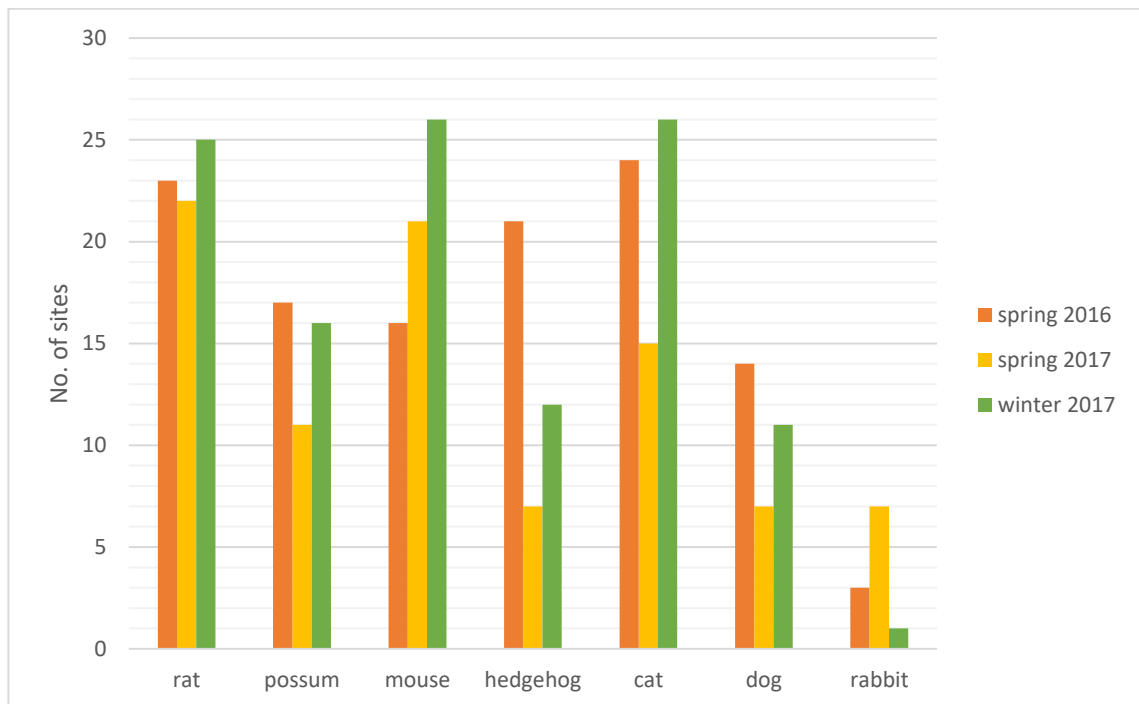


Figure 4.4. Number of sites ($n = 43$) at which introduced mammals were detected using chew cards and camera traps over three seasons. The data for winter 2018 is not displayed due to camera malfunction.



Figure 4.5. Introduced mammals detected only once throughout the study: (A) stoat, (B) ferret, (C) hare and (D) weasel.

Change in mammal communities over time since restoration

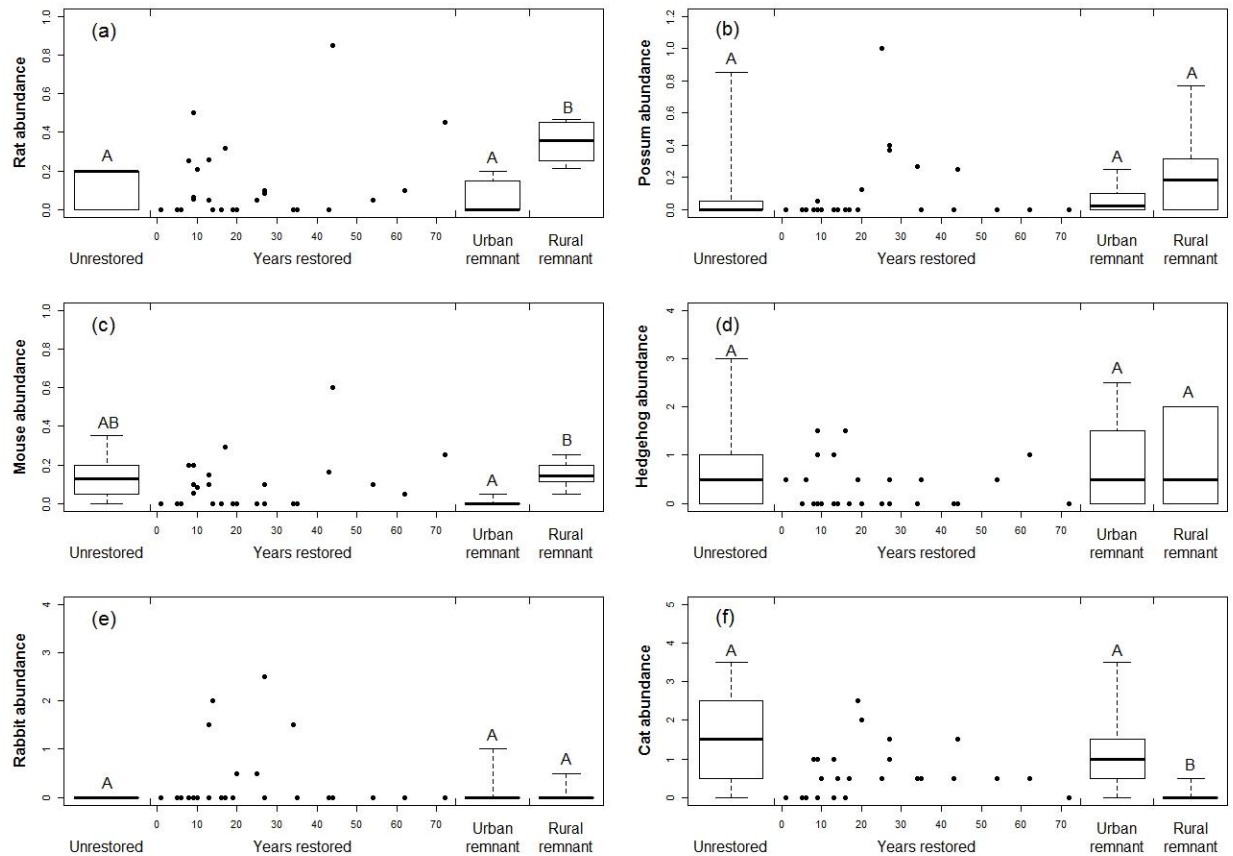


Figure 4.6. Mean abundance of (a) rats, (b) possums, (c) mice, (d) hedgehogs, (e) rabbits and (f) cats for spring data in restored forests over time (middle section of each bivariate plot, $n=25$), compared with unrestored, urban and rural remnants (boxplots).

Bivariate plots found no significant relationships between mammal indices of abundance and time since initial restoration planting (Fig. 4.6).

We found a statistically significant difference between the three reference categories (urban remnant, rural remnant and unrestored) in rat abundance as determined by Welch's unequal variances ANOVA ($F_{2,15} = 12.51$, $p = 0.001$) for the spring data (Fig. 4.6). A Games-Howell post hoc test revealed that unrestored forest differed significantly from rural remnants ($p = 0.018$) but not from urban remnants. There was also a significant difference between urban and rural remnants ($p = 0.002$). There was no statistically significant difference between groups in possum abundance. We found a statistically significant difference between groups in mouse abundance, as determined by Welch's unequal variances ANOVA ($F_{2,15} = 13.26$, $p = 0.004$) for the spring data. A Games-Howell post hoc test revealed that unrestored forest did not differ

significantly from either rural or urban remnants. There was, however, a significant difference between urban and rural remnants ($p = 0.008$). There was no statistically significant difference between groups in hedgehog abundance. There was no statistically significant difference between groups in rabbit abundance. We found a statistically significant difference between groups in cat abundance as determined by Welch's unequal variances ANOVA ($F_{2,15} = 5.77$, $p = 0.033$) for the spring data. A Games-Howell post hoc test, however, found no significant difference between the three reference sites. The difference between unrestored and rural remnants was not statistically significant ($p = 0.083$).

Bar graphs based on mean spring camera trap data revealed that cats were the most frequently detected mammals in unrestored, medium restored and urban remnant sites (Fig. 4.7). Rats and possums were the most frequently detected mammals in rural remnants. Cats and hedgehogs were ubiquitous in the spring, detected at all site types.

As with the spring data, bar graphs based on camera trap data from winter 2017 revealed that cats were the most frequently detected mammal in medium restored sites, and rats and possums in rural remnants (Fig. 4.8). Rat detections nearly quadrupled in rural remnants in the winter compared to spring. Cats, rats and possums were ubiquitous in the winter and were detected at all site types.

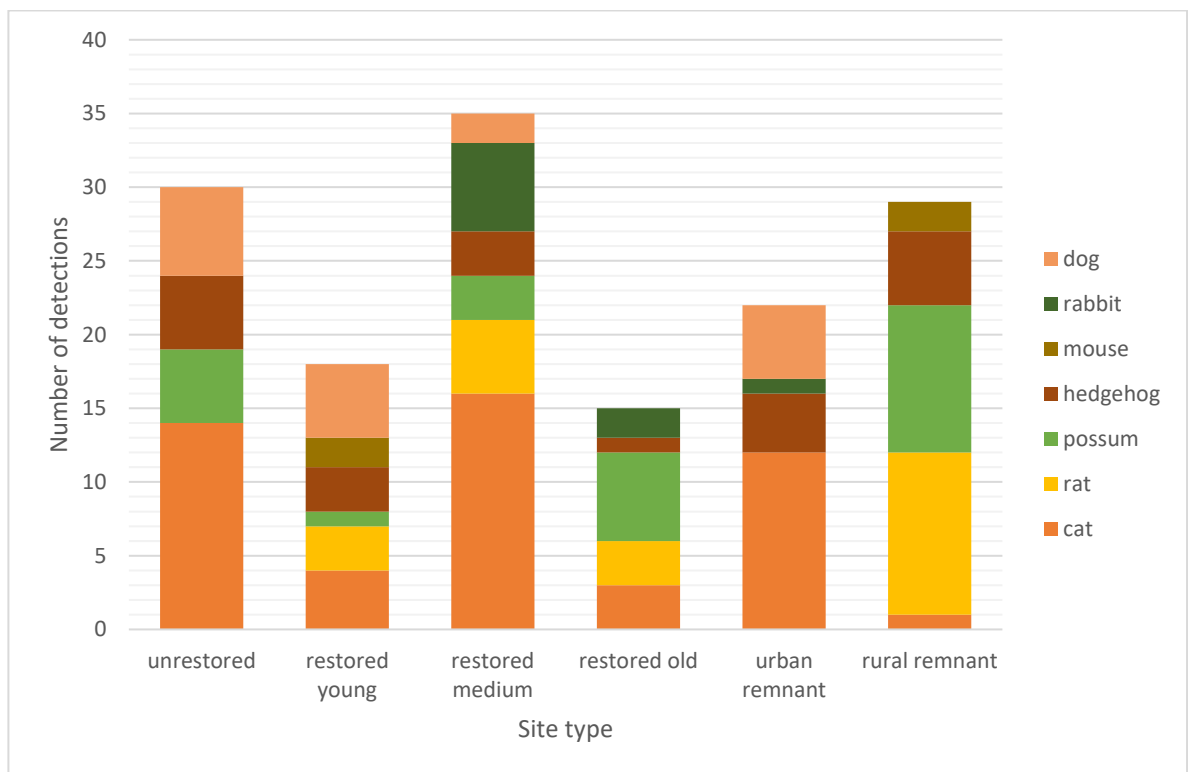
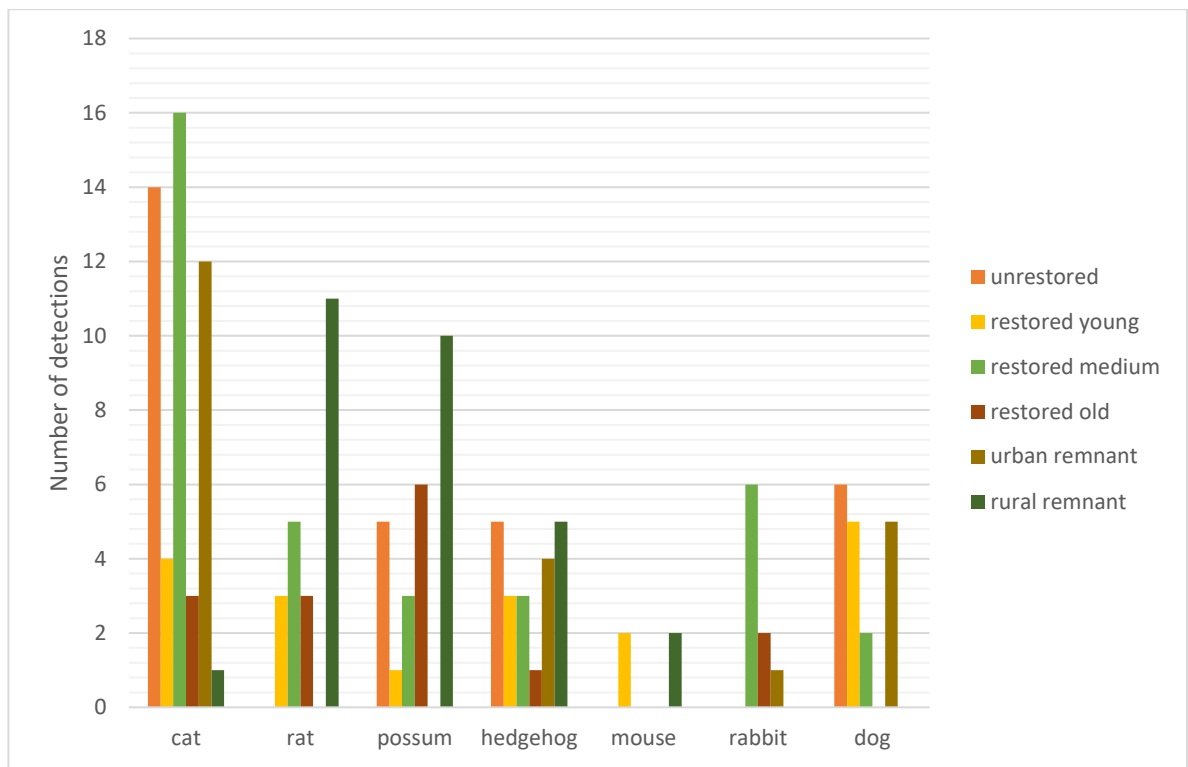


Figure 4.7. Average number of introduced mammals detected using camera traps per site type for spring data. Restored sites were divided into young (1-10, n = 8), medium (13-27, n = 10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.

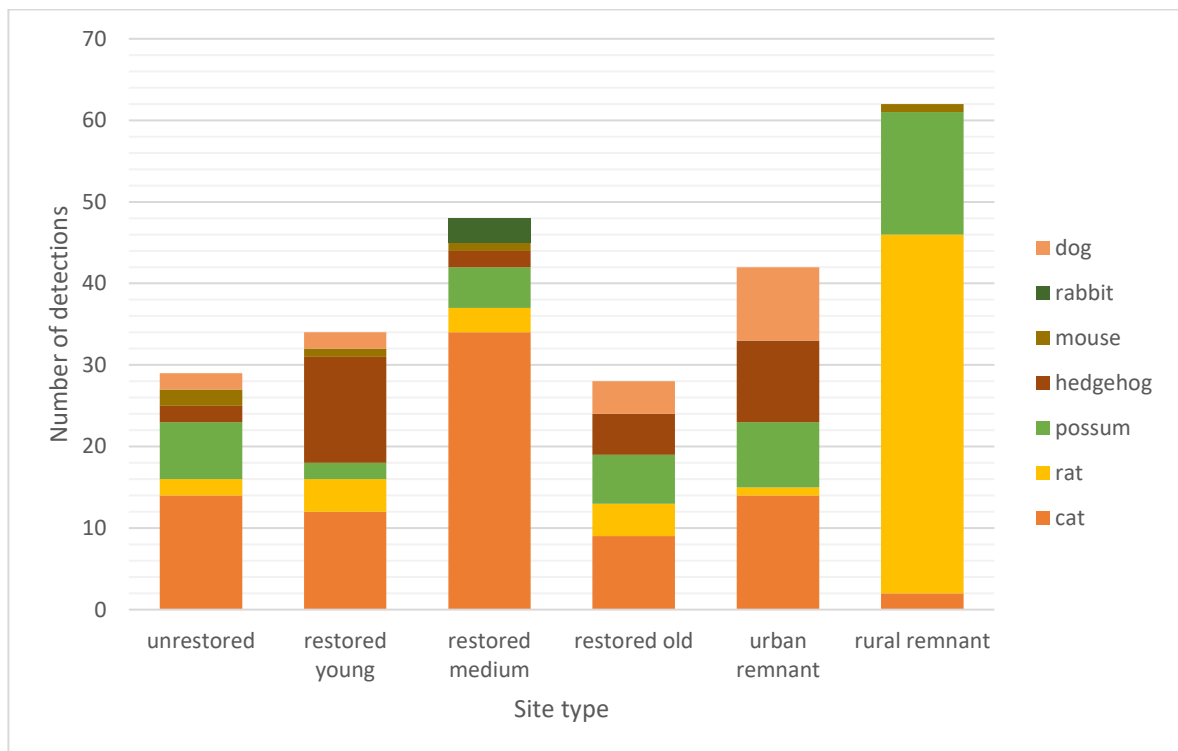
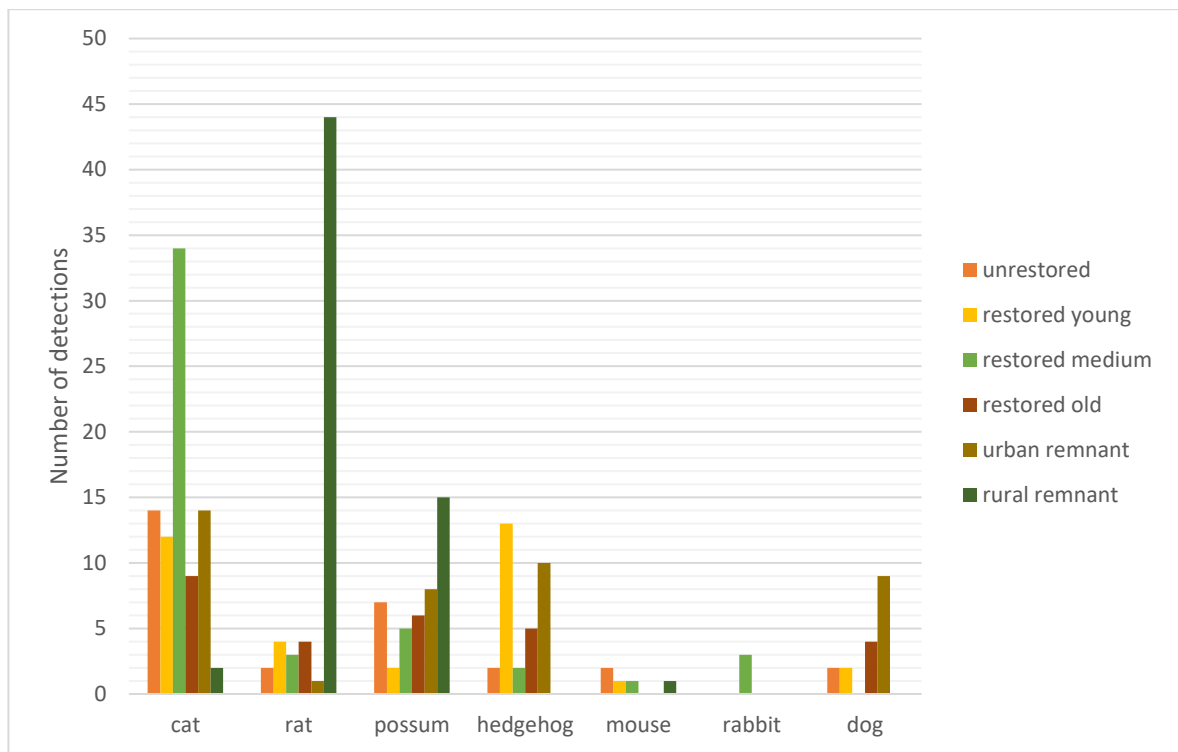


Figure 4.8. Number of introduced mammals detected using camera traps per site type for winter 2017 data. Restored sites were divided into young (1-10, n = 8), medium (13-27, n = 10) and old (34-72, n = 7). The top graph organizes data by species, the bottom one by site type.



Figure 4.9. The four introduced mammals with the most camera triggers in urban areas: (A) cat, (B) dog, (c) brushtail possums and (D) hedgehogs.



Figure 4.10. The four introduced mammals with the most camera triggers in rural areas: (A) ship rat, (B) brushtail possum, (C) house mouse and (D) hedgehog

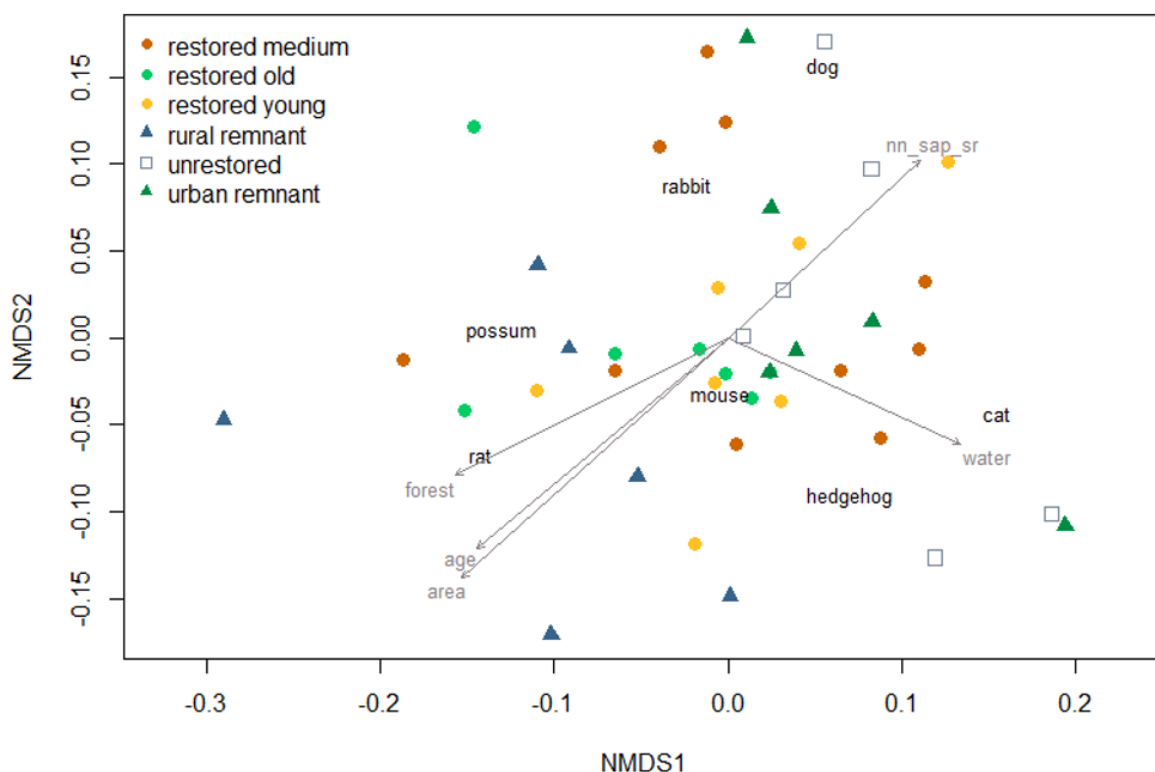


Figure 4.11. Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of 42 study sites (one was removed due to zeros) and spring mammal camera trigger data (stress = 0.19).

Table 4.1: Summary of vegetation and landscape variables fitted to the spring data NMDS ordination with fitting statistics (squared correlation coefficient and corresponding p-value).

Abbreviation for variable used in NMDS ordination	Original variable name	R ²	p-value
Forest	Native forest cover within 200 m buffer (ha)	0.23	0.006
Area	Size of study site (ha)	0.31	0.002
Water	Shortest distance to water source (m)	0.16	0.039
Age	Estimated age of oldest trees (years)	0.26	0.002
Nn_sap_sr	Non-native sapling species richness (no. species/300m ²)	0.17	0.025

The spring two-dimensional NMDS ordination did not reveal any clear clustering in the data, apart from all the rural remnants clustering to the bottom left of the graph (Fig. 4.11). Rural remnants were associated with higher relative abundances of rats and possums. Vector-fitting of vegetation and landscape variables to the pest community composition NDMS ordination showed two vegetation and three landscape variables that were significantly correlated with the pest community structure. The strongest correlations were for site area (31%), forest age (26%) and native forest cover within a

200 m buffer (23%). Increasing distance from a water source appeared to be positively correlated with increasing detections of cats and hedgehogs.

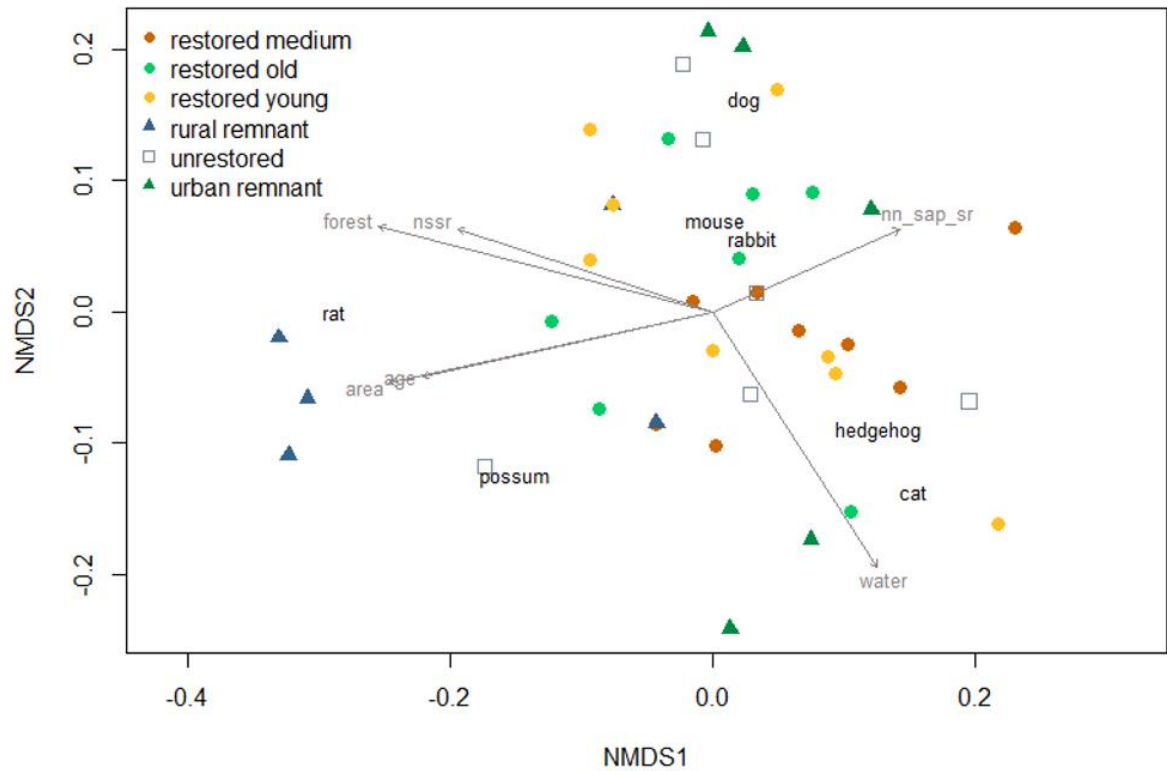


Figure 4.12. Plot of the first two dimensions of the nonmetric multidimensional scaling (NMDS) ordination of 40 study sites (three were removed due to zeros) and winter mammal camera trigger data (stress = 0.13).

Table 4.2: Summary of vegetation and landscape variables fitted to the winter data NMDS ordination with fitting statistics (squared correlation coefficient and corresponding p-value).

Abbreviation for variable used in NMDS ordination	Original variable name	R ²	p-value
Forest	Native forest cover within 200 m buffer (ha)	0.45	0.001
Area	Size of study site (ha)	0.41	0.001
Water	Shortest distance to water source (m)	0.35	0.001
Age	Estimated age of oldest trees (years)	0.33	0.002
Nssr	Native sapling species richness (no. species/300m ²)	0.27	0.006

The winter two-dimensional NMDS ordination revealed the same general patterns as the spring ordination, with rural remnants clustering to the bottom left of the graph (Fig. 4.12). Rural remnants were associated with higher relative abundances of rats and possums. Vector-fitting of vegetation and landscape variables to the pest community composition NDMS ordination showed two vegetation and three landscape variables that were significantly correlated with the pest community structure. The strongest correlations were for native forest cover within a 200 m buffer (45%), site area (41%) and shortest distance to a water source (35%). Increasing distance from a water source appeared to be positively correlated with increasing detections of cats and hedgehogs.

PERMANOVA revealed significant differences between site types for spring ($F_{5,36} = 1.88$, $p = 0.01$) and winter ($F_{5,36} = 2.86$, $p = 0.001$) data. Having found significant differences by PERMANOVA, we evaluated which site types differed significantly from each other using the pairwise Adonis function in R (Martinez Arbizu, 2019). For the spring data, we only found significant differences between rural and urban remnants ($p = 0.045$), and for the winter data, between rural remnants and medium restored sites ($p = 0.03$).

For the spring data, three species contributed 56% of the dissimilarity between urban and rural remnants. Rats and possums were more abundant in rural remnants compared to urban ones. Cats were more abundant in urban remnants.

For the winter data, two species contributed 56% of the dissimilarity between rural remnants and medium restored sites. Cats were more abundant in medium restored sites and rats were more abundant in rural remnants.

Relative effects of vegetation, landscape and interspecific interactions on indices of mammal abundance

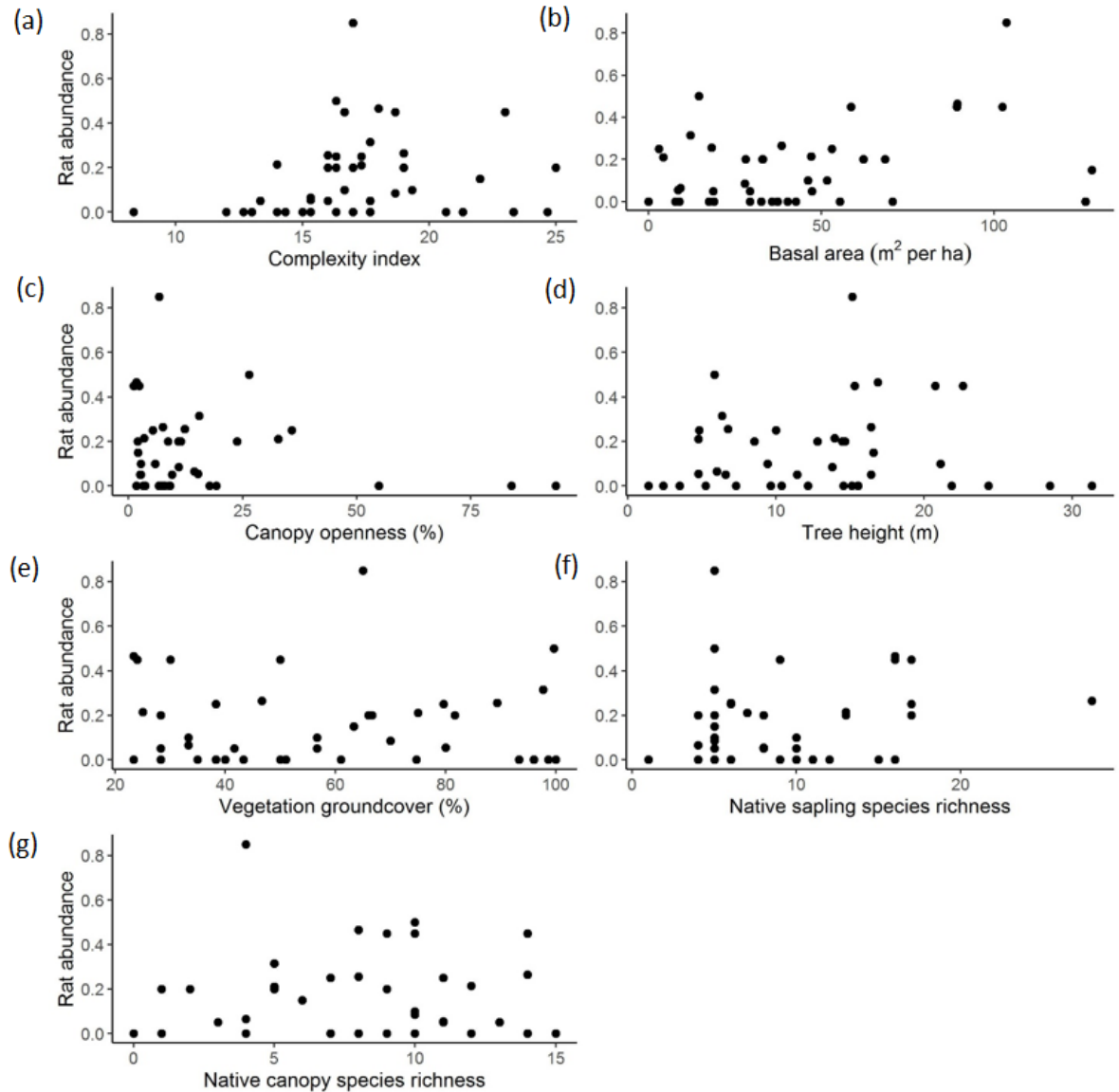


Figure 4.13. Spring means of rat indices of relative abundance and plant community attributes in all study sites (n = 43).

While bivariate plots revealed that rat abundance was not significantly associated with any vegetation variables measured, the plots do indicate cut-off points for vegetation complexity (Fig. 4.13a), canopy openness (Fig. 4.13c) and tree height (Fig. 4.13d). No rats were detected at sites with a vegetation complexity index less than 13, canopy openness greater than 35%, or tree height under 5 m.

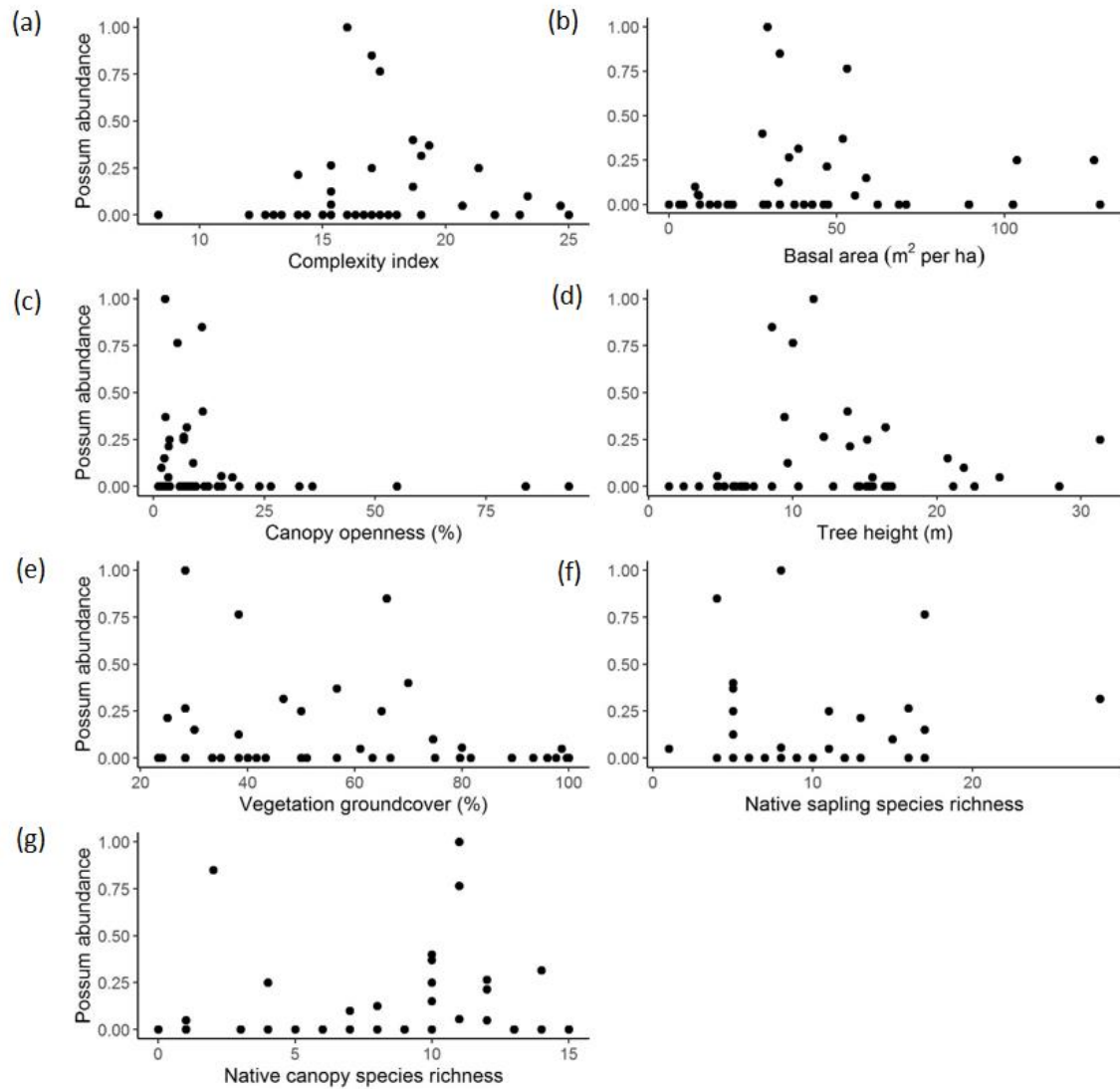


Figure 4.14. Spring means of possum indices of relative abundance and plant community attributes in all study sites (n = 43).

As for rats, bivariate plots did not reveal any significant, linear associations of possum relative abundance with any vegetation variables measured. The plots did, however, indicate cut-off points for vegetation complexity (Fig. 4.14a), canopy openness (Fig. 4.14c) and tree height (Fig. 4.14d). No possums were detected at sites with a vegetation complexity index less than 14, canopy openness greater than 13%, or tree height under 9 m.

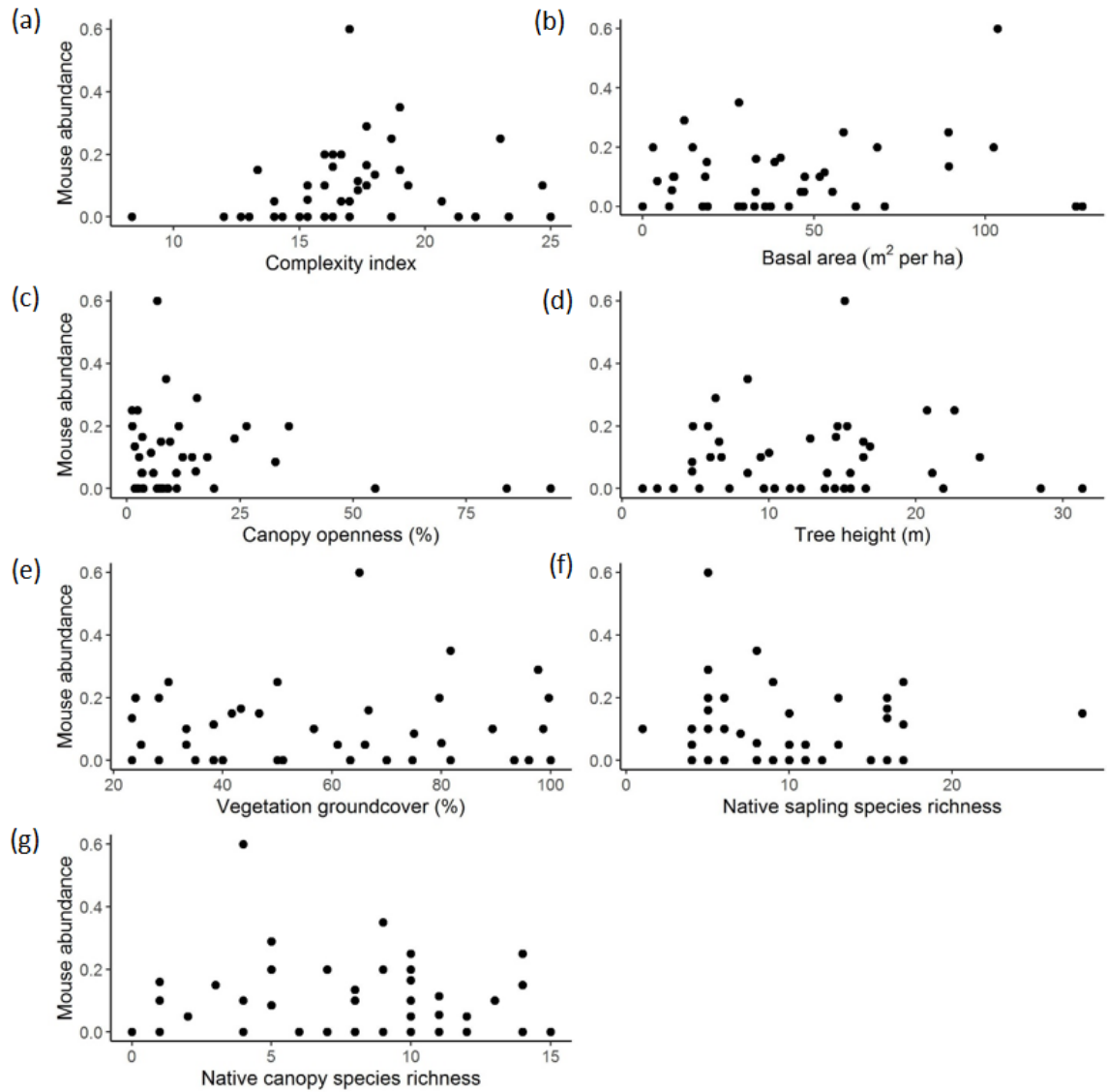


Figure 4.15. Spring means of mouse indices of relative abundance and plant community attributes in all study sites ($n = 43$).

As for rats and possums, bivariate plots did not reveal any significant, linear associations of mouse relative abundance with any vegetation variables measured. The plots did, however, indicate cut-off points for vegetation complexity (Fig. 4.15a), canopy openness (Fig. 4.15c) and tree height (Fig. 4.15d). Mice were not detected at sites with a vegetation complexity index less than 13, canopy openness greater than 36%, or tree height under 5 m.

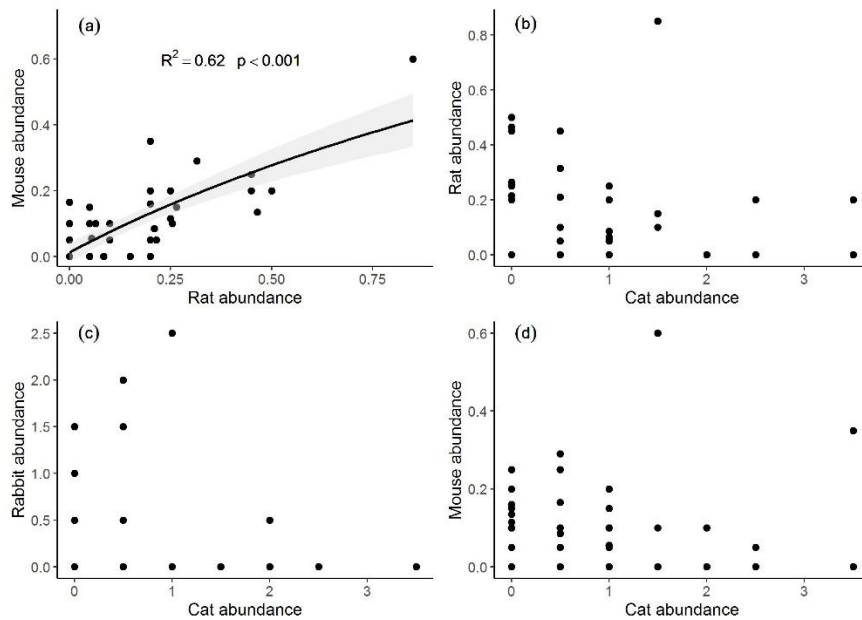


Figure 4.16. Relationships between spring means of mouse, rat, rabbit and cat indices of relative abundance in all study sites ($n = 43$).

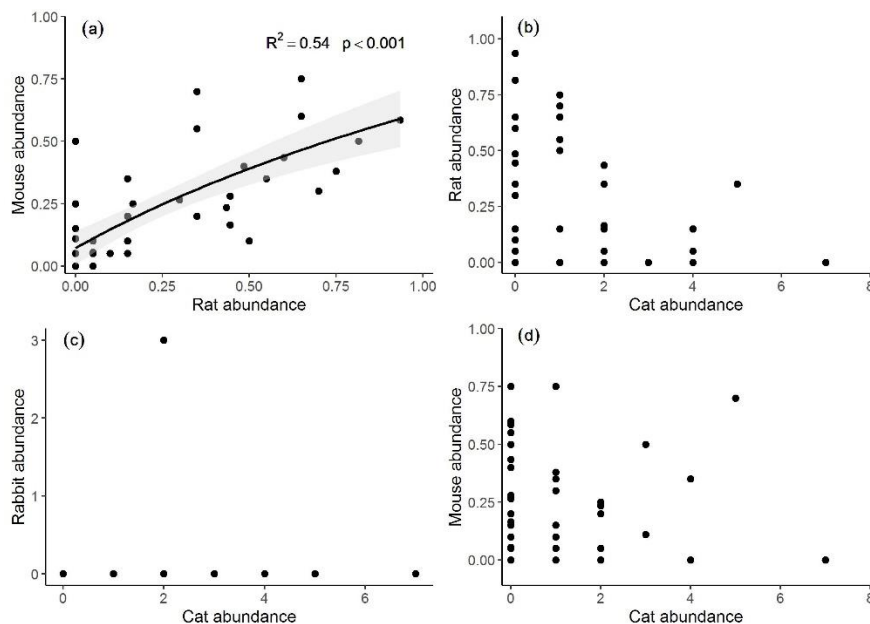


Figure 4.17. Relationships between winter means of mouse, rat, rabbit and cat indices of relative abundance in all study sites ($n = 43$). Bivariate plots did not reveal any significant associations of hedgehog or cat relative abundance with any vegetation variables measured (Appendix 4.1). Bivariate plots did not reveal any significant associations of rat, possum, mouse, hedgehog or cat relative abundance with any landscape variables measured (Appendix 4.1). Bivariate plots and linear models revealed that rat and mouse abundance were significantly and positively correlated for both spring ($R^2 = 0.62$, $p <$

0.001) (Fig. 4.16) and winter ($R^2 = 0.54$, $p < 0.001$) (Fig. 4.17) data. Cat abundance had no significant effects on rat, rabbit or mouse abundance.

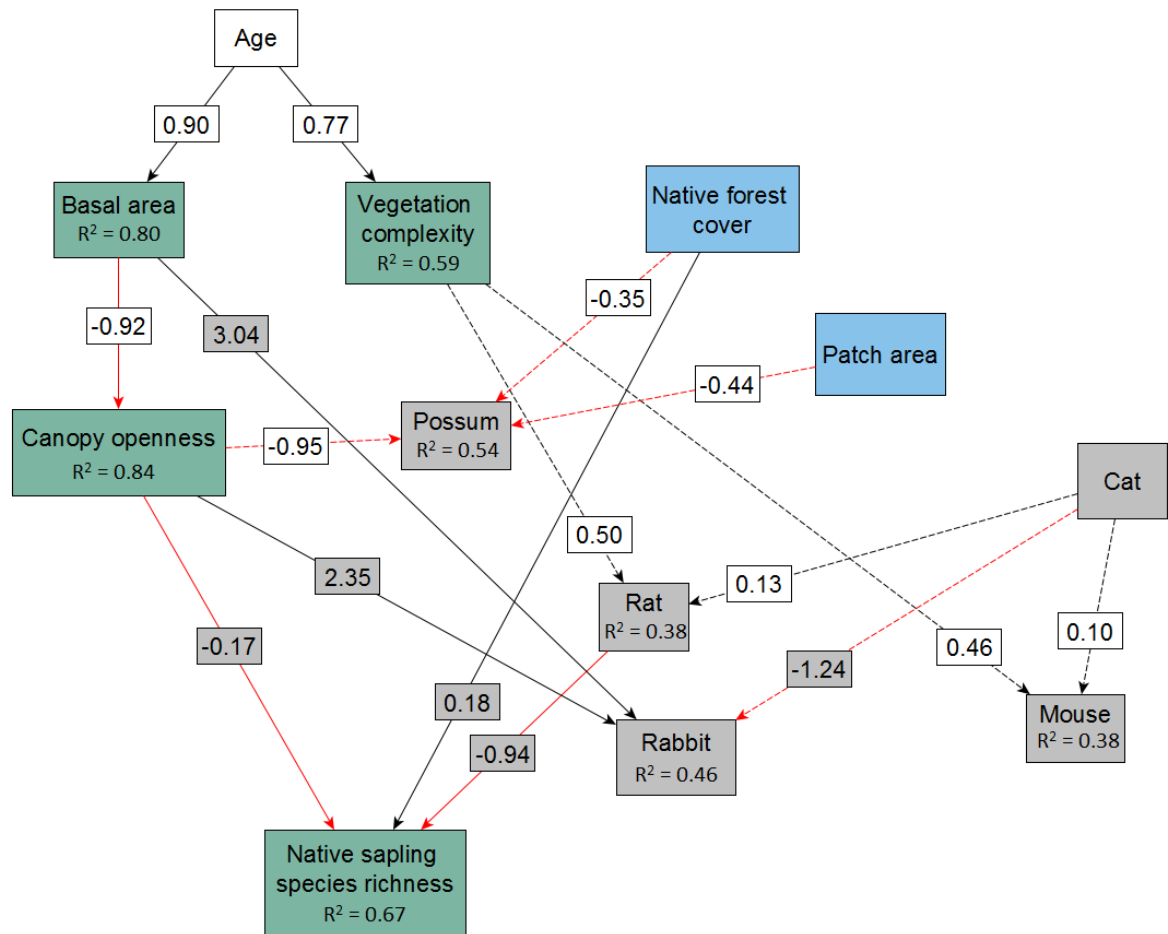


Figure 4.18. Structural equation model (SEM) path diagram, illustrating drivers of native sapling species richness.

The final native sapling species richness structural equation model fits the data well (AIC = 116.63, Fisher's C = 62.63, p -value = 0.92, $df = 80$), explaining 67% of the variation in native sapling species richness (Fig. 4.18 and Table 4.3). The model explained 54% of the variation in possum indices of relative abundance, 46% for rabbits, and 38% for both rats and mice. The final model revealed that canopy openness and rat abundance had significant, negative effects on native sapling species richness, whereas native forest cover in a 200 m buffer had a significant, positive effect. Rabbit abundance increased significantly with increasing canopy openness and tree basal area. The model revealed a non-significant, negative effect of cat abundance on rabbits. Contrary to our expectations, the effect of cats on rats and mice was not significant and

furthermore was positive. As predicted, over time basal area increased, decreasing canopy openness.

Table 4.3. Summary of results from the structural equation model on native sapling species richness. Asterisks denote statistically significant ($p < 0.05$) effect paths.

Response	Predictor	Estimate	Standardised estimate	Standard error	Degrees of freedom	p-value
Native sapling species richness	Canopy openness	-0.17	NA	0.08	21	0.039*
Native sapling species richness	Native forest cover (200 m)	0.18	NA	0.08	21	0.019*
Native sapling species richness	Rat abundance	-0.94	NA	0.39	21	0.017*
Mouse abundance	Complexity index	0.96	0.46	0.73	22	0.187
Mouse abundance	Cat abundance	0.26	0.10	0.72	22	0.721
Rat abundance	Cat abundance	0.35	0.13	0.69	22	0.610
Rat abundance	Complexity index	1.06	0.50	0.73	22	0.146
Possum abundance	Canopy openness	-5.16	-0.95	3.54	21	0.145
Possum abundance	Native forest cover (200 m)	-1.88	-0.35	1.26	21	0.135
Possum abundance	Area	-2.40	-0.44	1.95	21	0.219
Rabbit abundance	Cat abundance	-1.24	NA	0.69	21	0.074
Rabbit abundance	Canopy openness	2.35	NA	0.93	21	0.012*
Rabbit abundance	Basal area	3.04	NA	1.26	21	0.016*
Canopy openness	Basal area	-0.92	-0.92	0.08	23	0.000*
Complexity index	Site age	0.77	0.77	0.13	23	0.000*
Basal area	Site age	0.90	0.90	0.09	23	0.000*

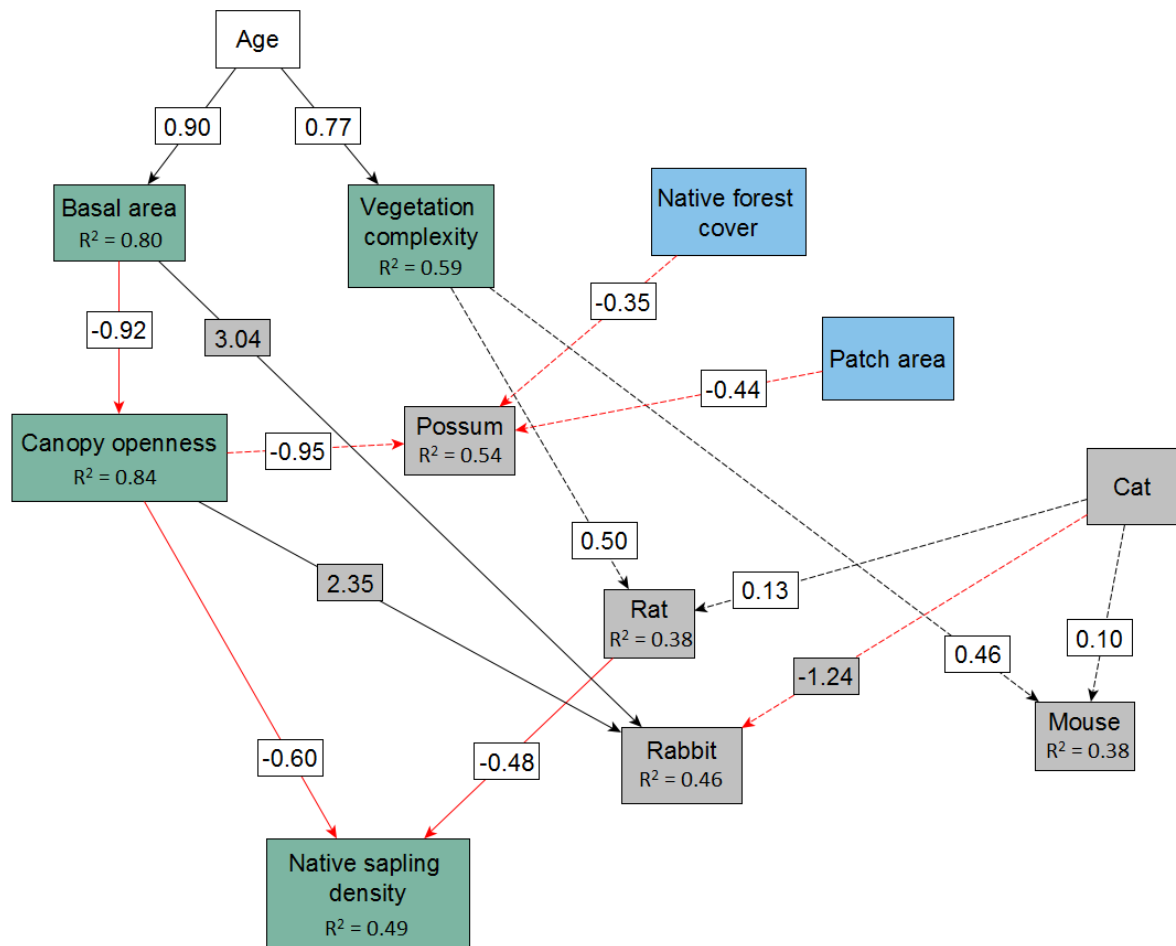


Figure 4.19. Structural equation model (SEM) path diagram, illustrating drivers of native sapling density.

The final native sapling density structural equation model fits the data well (AIC = 107.95, Fisher's C = 53.95.63, p-value = 0.97, df = 76), explaining 49% of the variation in native sapling density (Fig. 4.19 and Table 4.4). Rat relative abundance and canopy openness were the only variables significantly affecting native sapling density, with both having a negative effect.

Table 4.4. Summary of results from the structural equation model on native sapling density. Asterisks denote statistically significant ($p < 0.05$) effect paths.

Response	Predictor	Estimate	Standardised estimate	Standard error	Degrees of freedom	p-value
Native sapling density	Canopy openness	-0.60	-0.60	0.16	22	0.001*
Native sapling density	Rat abundance	-2.14	-0.48	0.68	22	0.005*

Mouse abundance	Complexity index	0.96	0.46	0.73	22	0.187
Mouse abundance	Cat abundance	0.26	0.10	0.72	22	0.721
Rat abundance	Cat abundance	0.35	0.13	0.69	22	0.610
Rat abundance	Complexity index	1.06	0.50	0.73	22	0.146
Possum abundance	Native forest cover (200 m)	-1.88	-0.34	1.26	21	0.135
Possum abundance	Canopy openness	-5.16	-0.95	3.54	21	0.145
Possum abundance	Area	-2.40	-0.44	1.95	21	0.219
Rabbit abundance	Basal area	3.04	NA	1.26	21	0.016*
Rabbit abundance	Canopy openness	2.35	NA	0.94	21	0.012*
Rabbit abundance	Cat abundance	-1.24	NA	0.69	21	0.074
Canopy openness	Basal area	-0.92	-0.92	0.08	23	0.000*
Complexity index	Site age	0.77	0.77	0.13	23	0.000*
Basal area	Site age	0.90	0.90	0.09	23	0.000*

4.4 Discussion

Change in mammal communities over time since restoration

Our results highlight the ubiquity of rats, mice, cats and hedgehogs in restoration plantings in New Zealand cities. This is unsurprising, given that ship rats and house mice are two of the world's most widespread rodents, capable of adapting to a wide range of environments, and with a long evolutionary history of association with humans (Innes, 2005; Banks & Smith, 2015). Furthermore, the highest densities of domestic cats are found in urban areas (Fischer *et al.*, 2012) and New Zealand has the highest recorded rate of cat ownership in the developed world (Kikillus *et al.*, 2017). Hedgehogs are also known to be urban adapters in the UK, and reach high densities in New Zealand's North Island cities and suburbs (Jones and Sanders, 2005).

Throughout this discussion, we stress that we lacked adequate information to quantify the levels of pest control conducted within each individual study site. Thus, the relationships we measured between mammals, vegetation and landscape variables are not fully unmanaged expressions of the ecological community. However, varying intensity and methods of pest control will be the reality in all New Zealand cities. Moreover, the home ranges of the majority of mammals studied will be greater than the individual study sites (King, 2005), so the mammal community will be affected by pest control and resources in the surrounding landscape.

With these caveats in mind, we found that the greatest differences in the introduced mammal communities were those between urban reference sites (both unrestored and remnants) and rural remnants, with urban areas supporting more cats, and rural remnants supporting more rats and possums. Similarly, Ruffell and colleagues (2015) found suggestive evidence that urbanisation decreased indices of rat abundance. While ship rats are found in diverse habitats in New Zealand, from buildings and urban parks to exotic forest, they are most abundant in mature, structurally complex, lowland podocarp-broadleaved forests (Innes, 2005; Efford *et al.*, 2006). A study of ship rat populations in urban areas of Sydney likewise found that rats were more abundant in naturally vegetated areas than in more built-up environments (Cox *et al.*, 2001). Our findings highlight a point made by Banks and Smith (2015) that we do not know the relative importance of urban versus non-urban areas for many commensal species.

Camera trap data revealed that cats were the most frequently detected mammal in urban areas, and the only mammal that was ubiquitous during both spring and winter fieldwork. This is consistent with Morgan and colleagues' (2009) conclusion that while rats and possums were scarce in residential areas and amenity parks and largely restricted to gullies, cats were ubiquitous in Hamilton. Our results further confirm Morgan and colleagues' finding that mustelids are uncommon in New Zealand cities. This difference in the mammal community composition between urban and non-urban areas may allow New Zealand cities to act as refugia for certain endemic plants and animals that are particularly vulnerable to predation by rats and possums, if suitable habitat is available. However, these mammals are still relatively widespread in the urban greenspaces we studied, and it is unclear whether the lower indices of abundance in comparison with rural areas can benefit endemic flora or fauna of conservation concern. Furthermore, while rats and possums may be rarer in cities compared to rural remnants, cats were abundant and ubiquitous. More research is needed to investigate what effect cats are having on endemic wildlife in New Zealand cities.

We did not find linear responses in the mammal communities to urban forest restoration. This was not surprising, given the ecological and behavioural versatility of rats and mice (King *et al.*, 1996). We also caution once again that the patterns in rat, possum and mouse abundance indices will have been influenced by varying levels of predator control. Older restoration plantings and urban remnants are often highly valued, and both local authorities and community groups will put more effort into predator control in these areas compared with younger restoration plantings. However, we did find evidence for threshold responses in rats, possums and mice, which were never detected at the three youngest sites. Our results suggest that rodents and possums prefer a certain level of vegetation complexity, canopy cover and tree height to be able to use a restoration planting. We found evidence that possums need higher levels of vegetation complexity, greater canopy closure and taller trees, compared to rats and mice. Once these habitat requirements are met after approximately 9 years, rats, possums and mice are relatively widespread in restoration plantings, underscoring the importance of conducting predator control once plantings have reached a certain age. We recommend that future studies on the changes in the pest community in restoration

plantings use a larger sample size of sites younger than 9 years, as this appears to be the period in which rats, possums and mice begin to benefit from the planted vegetation.

Thus, our results indicate that the changes in vegetation structure and complexity over time since restoration, which benefit native bird species richness and abundance (see Chapter 3), also provide habitat for introduced predators, such as rats, possums and mice. A dryland forest restoration project at Auwahi, Hawaii, similarly recorded an increase in ship rat numbers following restoration, which the authors hypothesized was driven by increases in structural complexity and the abundance and diversity of food resources (Shiels *et al.*, 2017). Increasing structural complexity with time since restoration was also hypothesized to decrease predation pressure on rats. Studies in both Australia and New Zealand support the finding that ship rats are less common in open, early-successional grassland compared to structurally complex forests (Innes *et al.*, 2010b). These results highlight the need to carefully plan and monitor restoration projects to avoid unintended outcomes caused by complex interactions among native and introduced species in the habitats we are recreating.

Interactions between mammal species

While the study at Auwahi documented a shift in the rodent community with restoration age, from house mice to ship rats (Shiels *et al.*, 2017), we found a strong positive correlation between rats and mice in our study, along with evidence of similar responses of the two species to vegetation complexity, canopy cover and tree height. Our results are more in keeping with those of Norbury and colleagues (2013), whose study in dry grasslands and shrubland of New Zealand's South Island found that mice were more abundant in shrubland, which they hypothesized was due to more abundant fruit and seed, as well as cover from predation provided by more structurally complex vegetation.

The majority of New Zealand studies conclude that rats can limit mouse populations, either through competition or predation, or a combination of the two (King *et al.*, 1996; Innes, 2005; Tompkins & Veltman, 2006; Ruscoe *et al.*, 2011). However, a study by Bridgman and colleagues (2018) in Pureora Forest found a positive correlation between house mice and ship rats in spring, which they concluded was unexpected and difficult

to explain. The positive correlation between rats and mice in our study was found for both spring and winter data. We suggest this may be evidence of how urbanisation can alter trophic links. If rodents feed on resources provided by people (such as compost heaps), it is possible that this supplementation releases rats and mice in cities from competition for food, and mice from predation by rats. The positive correlation between the two species may then simply reflect their similar preferences for dense, structurally complex vegetation. The authors are unaware of any studies comparing urban rat and mouse diets to those of rodents living in indigenous New Zealand forests and we suggest future research could investigate how rodent ecology and behaviour is affected by urbanisation.

It has been suggested that cats control rats and mice, thereby benefiting native birds, lizards and invertebrates (Matias & Catry, 2008; van Heezik *et al.*, 2010; Wood *et al.*, 2016). This was the case on Little Barrier Island, where removing cats without controlling Pacific rats (*Rattus exulans*) led to a decrease in Cook's petrel breeding success (Le Corre, 2008). Removing cats, the top predator in the system, resulted in the mesopredator release (Crooks & Soulé, 1999) of rats, which had a stronger overall detrimental impact on the petrel community than when both cats and rats were present. For our study system, however, our structural equation model did not support this claim. The effect of cats on rats and mice in our model was not significant, was relatively small, and positive.

While feral cats may regulate rodent populations in native New Zealand forest (Efford *et al.*, 2006), there are no studies I am aware of that confirm cats control populations of urban rats and mice. A study of domestic cat predation on Norway rats in Baltimore concluded that cat predation had no impact on the rat population (Childs, 1986). Similarly, Parsons and colleagues (2018) found that Norway rats in an industrial waste recycling facility in New York City could persist at high densities despite the presence of cats, simply by altering their movements to avoid direct encounters. Norway rats, however, are larger than ship rats and studies have shown that cats avoid killing Norway rats that are over 200 g (Childs, 1986). Ship rats generally weigh less than 200 g (Innes, 2005) and thus may be more susceptible to cat predation. New Zealand is unusual in having urban ship rats.

Further studies are needed to investigate whether the greater abundance of cats in urban areas contribute to lower rat numbers compared to rural remnants, or whether rats are instead limited by the amount of suitable habitat in cities. Given that rats and mice have coexisted with domestic cats for centuries, whereas New Zealand's native fauna has not, we urge caution in suggesting cats benefit native wildlife through their predation of rats and mice. More conclusive studies are needed before this argument can be used to inform wildlife management in New Zealand cities.

Impacts of mammals on urban forest restoration

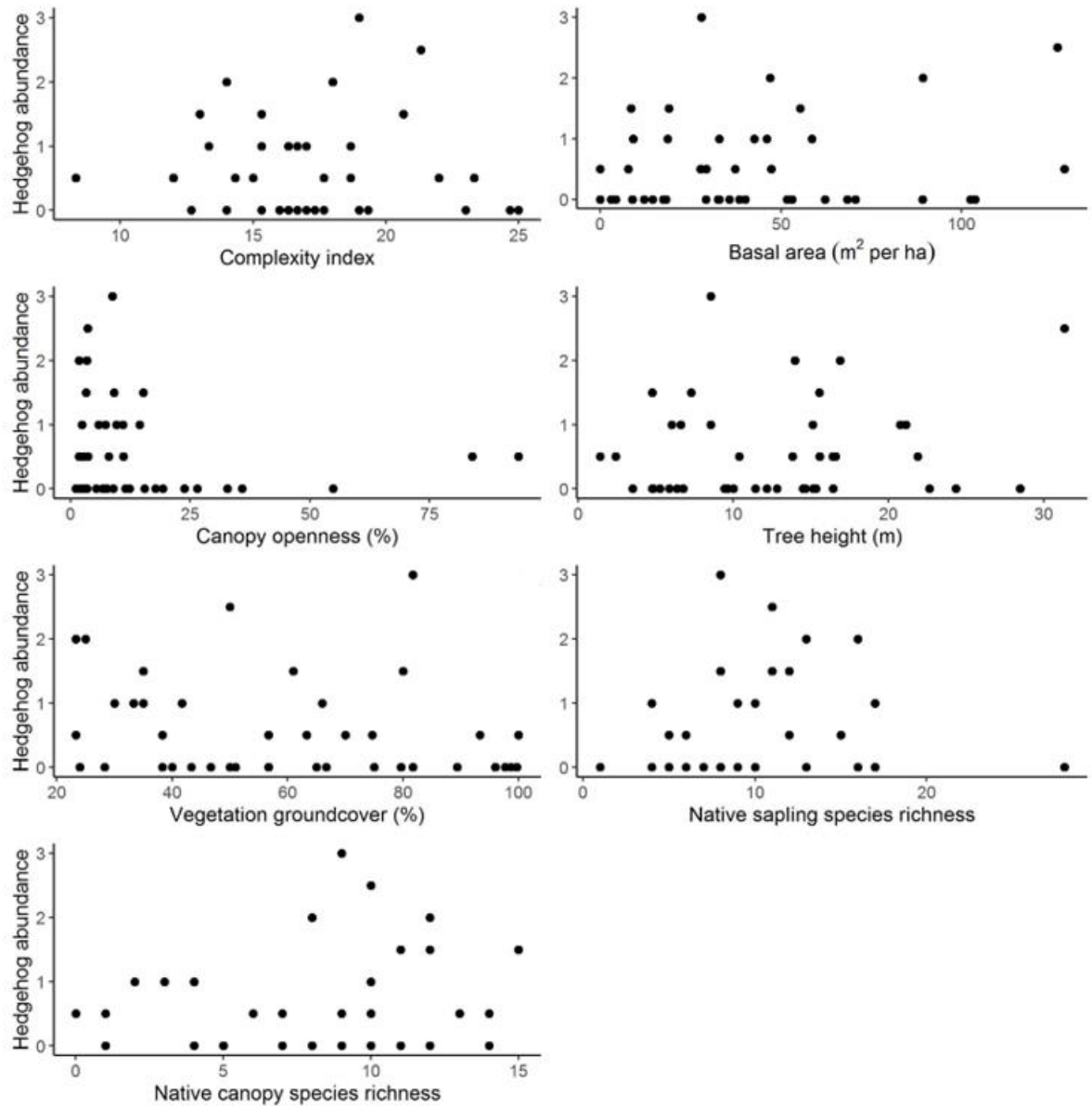
Our research is the first in New Zealand to investigate the effects of introduced mammals on the regeneration of native forest in urban areas. Our structural equation model suggested that rats were the only mammal significantly decreasing native sapling species richness and density. While the impact of possums on the structure and composition of native New Zealand forests has been relatively well studied (Bellingham *et al.*, 1999; Nugent *et al.*, 2001; King, 2005), less is known about the impact of rats (Campbell & Atkinson, 2002). In their study of the effects of deer and rodent herbivory on seedling establishment in Waitutu Forest in New Zealand's South Island, Wilson and colleagues (2006) concluded that the effect of rodents on the survival of planted seedlings was relatively small. Rats, however, are known to consume fallen fruit, as well as flowers and fruit in the tree crown, and possibly seedlings (Campbell & Atkinson, 2002). Mulder and colleagues (2009) detected a difference in seedling community structure between islands with and without rats. Pender and colleagues (2013) found that ship rats were significant and underappreciated seed predators of the endangered Hawaiian flowering plant, *Cyanea superba* subsp. *superba*. Further research is needed on the impact of rats on urban restoration plantings.

Our research is the first step in understanding how the introduced mammal community changes over time in urban restored forests in New Zealand. Our results indicate that positive restoration outcomes for native vegetation and birds may have undesirable consequences, such as improving habitat for introduced mammals. Increased numbers of rats, in turn, may be changing the trajectory of forest restoration through seed, and

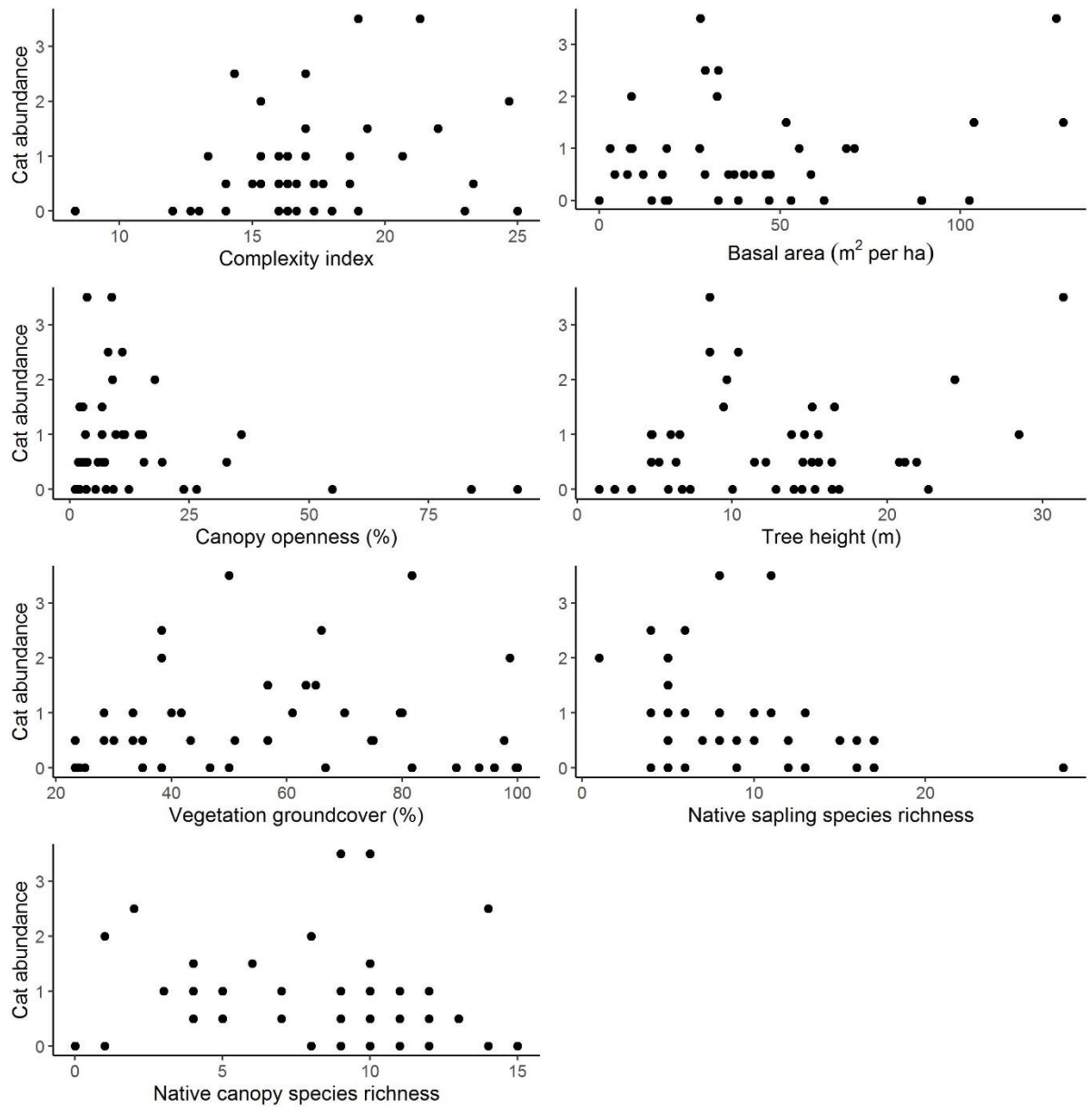
possibly seedling, predation. Our findings highlight the importance of considering a restored ecosystem in its entirety. We cannot simply focus on planting native species but must also consider how restoration will affect introduced mammalian predators in New Zealand cities. Introduced mammal control is vital to the success of a restoration project. Monitoring restoration planting over time will allow practitioners to detect unforeseen consequences of restoration and alter their practices accordingly.

4.5 Appendices

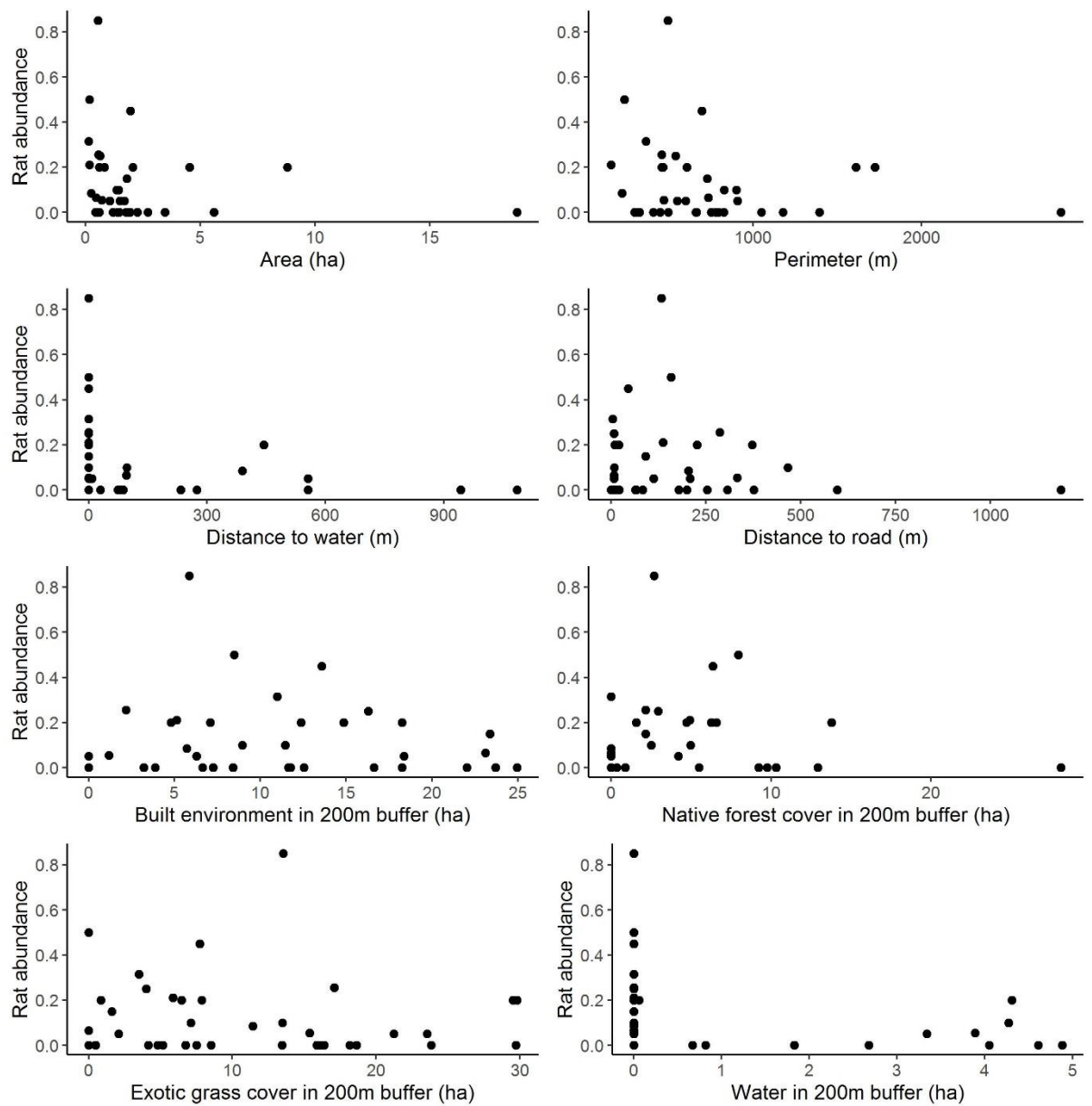
Appendix 4.1



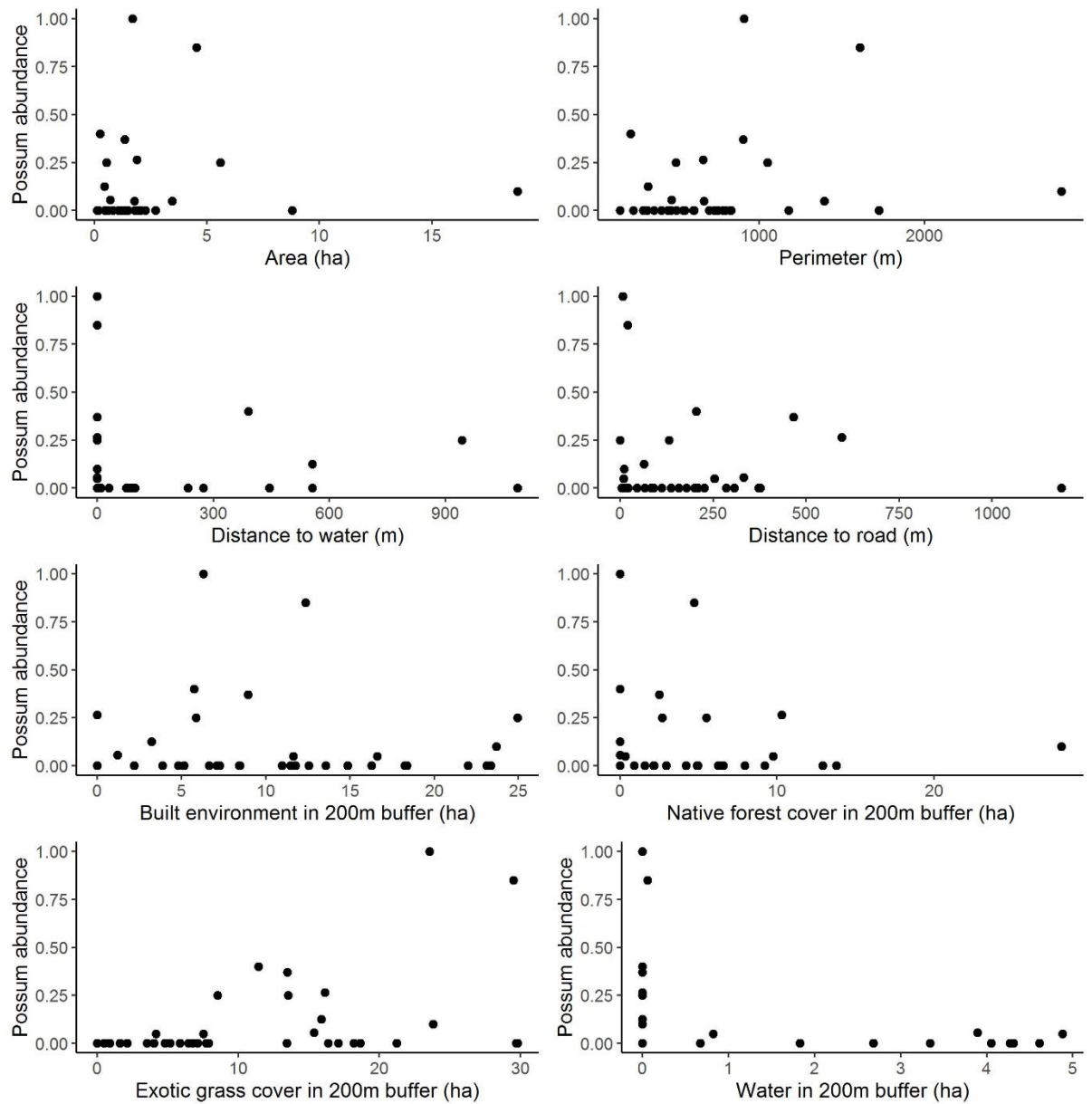
APPENDIX 4.1. Figure 1. Spring means of hedgehog indices of relative abundance and plant community attributes in all study sites ($n = 43$). Scatterplot points represent individual sites. Hedgehog indices of relative abundance are the number of camera triggers per site. Native sapling and canopy species richness is given as number of species per 300 m².



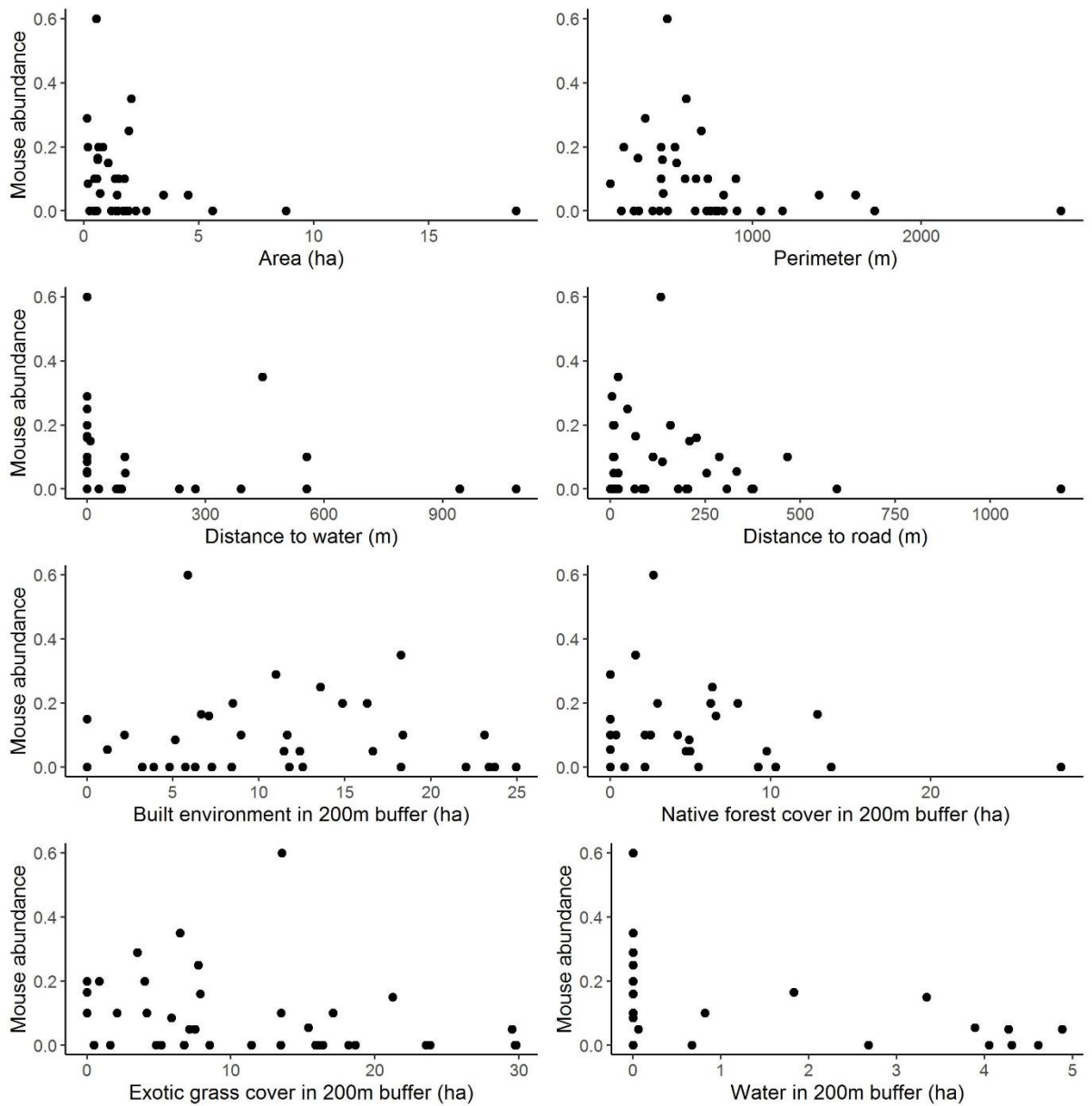
APPENDIX 4.1. Figure 2. Spring means of cat indices of relative abundance and plant community attributes in all study sites ($n = 43$). Scatterplot points represent individual sites. Cat indices of relative abundance are the numbers of individually recognizable cats photographed per site. Native sapling and canopy species richness is given as number of species per 300 m².



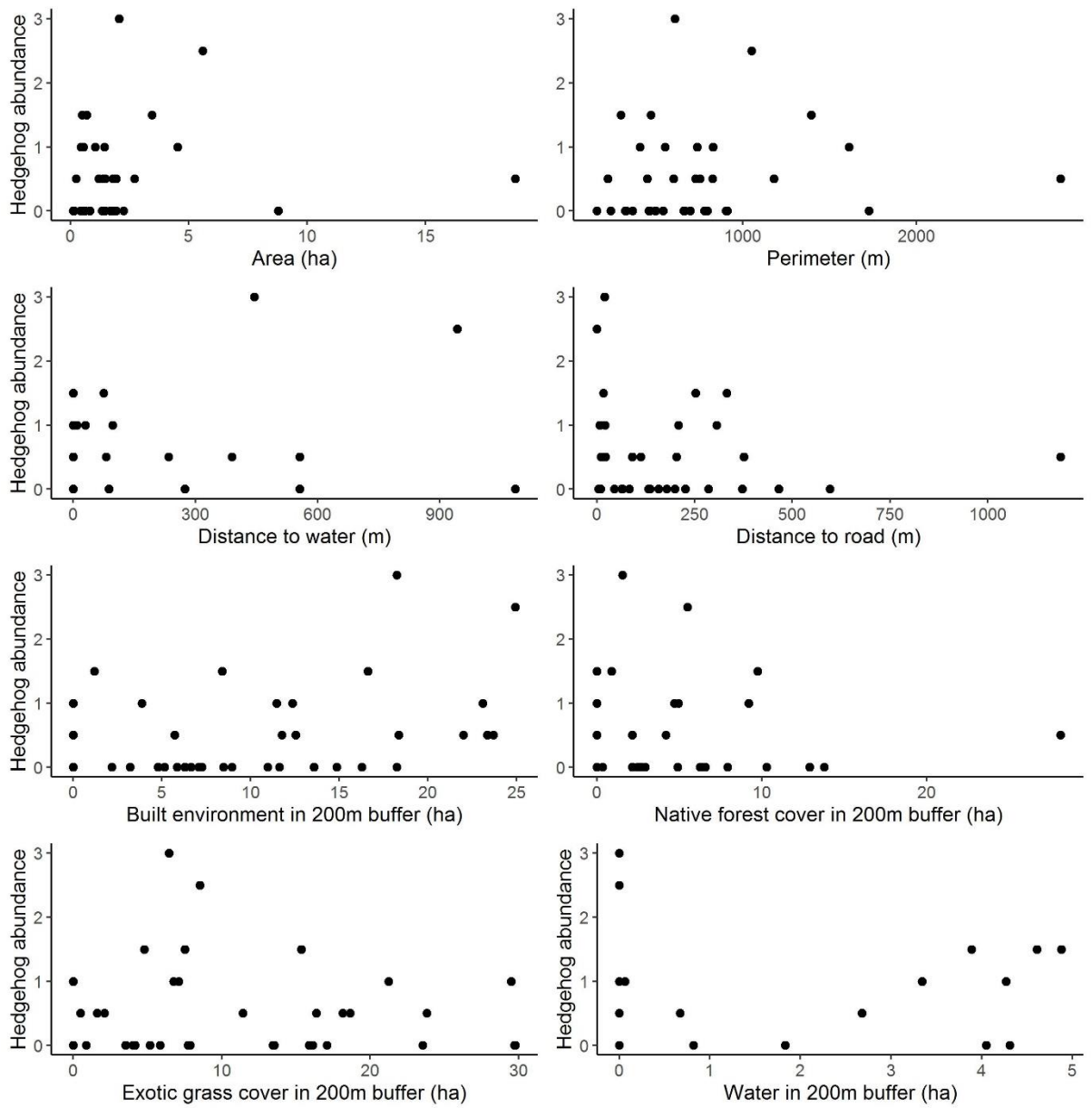
APPENDIX 4.1. Figure 3. Spring means of rat indices of relative abundance and landscape variables in all urban study sites ($n = 37$). Scatterplot points represent individual sites. Rat indices of relative abundance are the proportion of chew cards chewed per site.



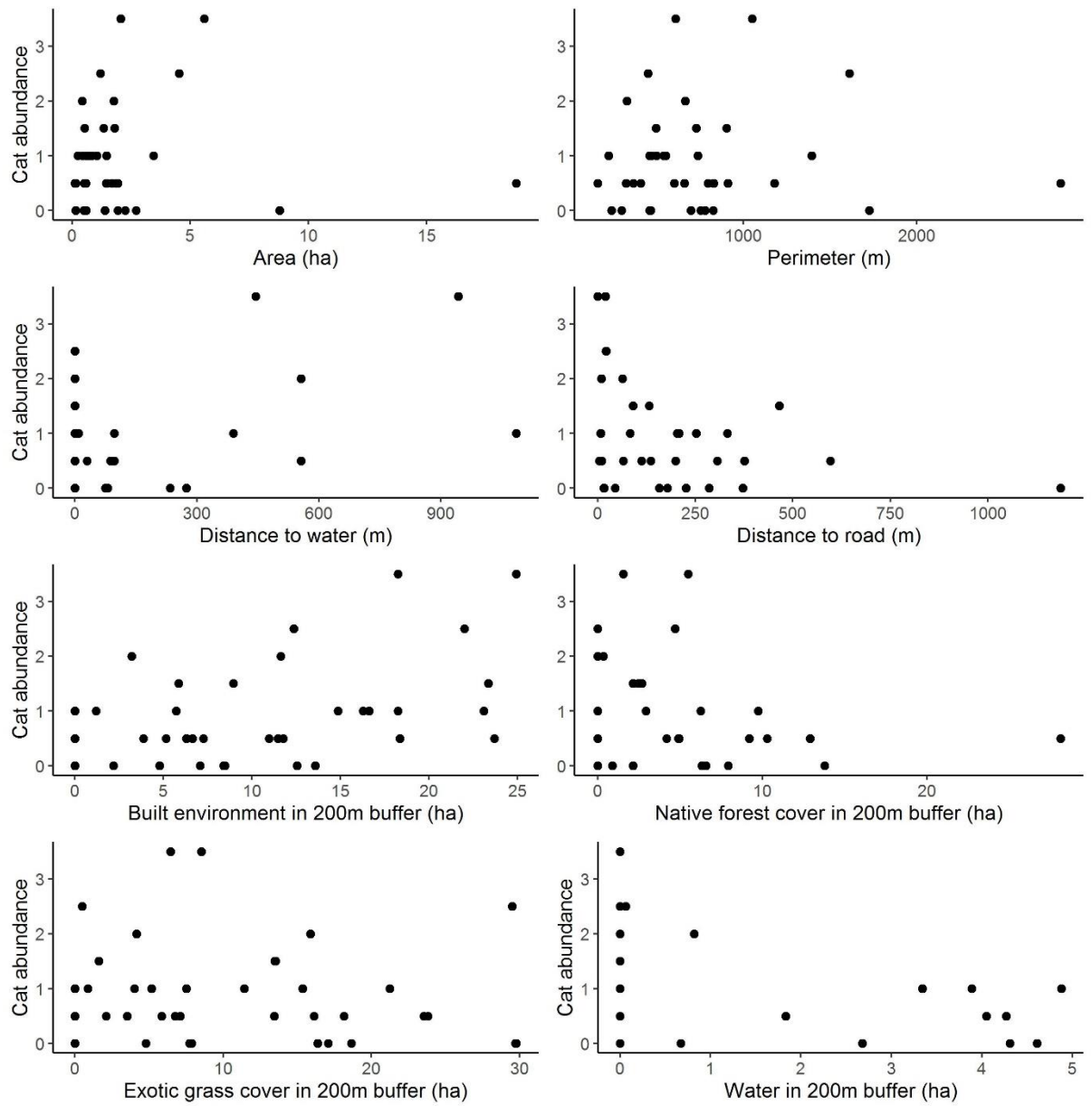
APPENDIX 4.1. Figure 4. Spring means of possum indices of relative abundance and landscape variables in all urban study sites ($n = 37$). Scatterplot points represent individual sites. Rat indices of relative abundance are the proportion of chew cards chewed per site.



APPENDIX 4.1. Figure 5. Spring means of mouse indices of relative abundance and landscape variables in all urban study sites ($n = 37$). Scatterplot points represent individual sites. Rat indices of relative abundance are the proportion of chew cards chewed per site.



APPENDIX 4.1. Figure 6. Spring means of hedgehog indices of relative abundance and landscape variables in all urban study sites (n = 37). Scatterplot points represent individual sites. Hedgehog indices of relative abundance are the number of camera triggers per site.



APPENDIX 4.1. Figure 7. Spring means of cat indices of relative abundance and landscape variables in all urban study sites ($n = 37$). Scatterplot points represent individual sites. Cat indices of relative abundance are the numbers of individually recognizable cats photographed per site.

Chapter 5: General Discussion

5.1 Key findings

This thesis advances knowledge of the capacity of urban forest restoration to address the interconnected problem of loss of native biodiversity in cities and the disintegration of human relationships with the natural world. My findings reveal that ecological restoration in cities has great potential for enriching the lives of city residents and providing habitat for native wildlife. However, this research also highlights that much more work needs to be done to fully realize this potential.

Chapter 2 focused on the human dimension of native biodiversity in cities. Using a phenomenological approach allowed me to discover a tension between what respondents claimed to value and their specific actions influencing urban biodiversity. For instance, participants voiced a preference for native plants, while I observed that their gardens were dominated by introduced species. Similarly, participants wanted to attract native birds, but most allowed their pet cats to roam free. This gap between values and actions stemmed from participants holding conflicting and competing desires and priorities. The value-action gap further highlights the difficulties inherent in promoting gardening practices that provide habitat for native species. Based on my interviews, I conclude that we cannot currently rely on private gardens in Hamilton to provide habitat for native birds. However, this research offers renewed evidence of the benefits of urban greenspaces to people and highlights the unique contributions of native bush parks to human wellbeing. Ecological restoration in cities has great potential to enrich human lives, increasing urban residents' emotional involvement with the natural world, thereby disrupting the vicious cycle of the extinction of experience.

In order to break the cycle by allowing city residents daily contact with native nature, cities will need to include areas that support native flora and fauna. Given that we cannot currently rely on private gardens to provide adequate habitat for native birds (Chapter 2), the alternative is for local authorities to promote the ecological restoration of public greenspaces. Chapter 3 is the first study in New Zealand to investigate

whether restoring urban forest benefits native forest bird communities. My findings reveal that native bird species richness and diversity increase over time since initial restoration planting. The bird community changes over time, with younger restored sites supporting introduced granivores and generalists, and older sites providing better habitat for native forest birds. I conclude that in New Zealand cities, which are characterized by extreme deforestation and habitat degradation, lack of habitat is an important factor limiting native forest birds. Forest restoration in cities, therefore, is necessarily the first step towards re-establishing our native forest bird communities. My research further demonstrated, however, that seven decades of restoration cannot match the habitat value of older forest remnants. Protection of existing forest remnants should therefore be a priority in a broader strategy, which also seeks to create new habitat and links between them. Furthermore, while current restoration efforts are benefiting widespread, common native forest birds, revegetation alone will not create forests capable of supporting bird species of conservation concern⁷. To achieve this goal would require greater investment in predator control or eradication, and the reintroduction of locally extinct endemics.

Chapter 4 addresses the complete lack of research on the effects of urban forest restoration on introduced mammal communities and the impact of mammals in turn on forest regeneration. My results highlight the ubiquity of rats, mice, cats and hedgehogs in urban restored forests. The study revealed that urban forest restoration creates habitat for rats, mice and possums within 9 years of initial restoration planting. The research further demonstrated that rats were significantly decreasing native sapling species richness and density in restored forests. Urban forest restoration, in the absence of intensive pest control, is therefore creating habitat for rats, which in turn may be changing the trajectory of forest recovery through seed or seedling predation.

5.2 Recommendations for management

The findings of this study emphasize the importance of considering ecosystems in their entirety when planning, conducting and evaluating restoration projects. I took the

⁷ I use the term ‘birds of conservation concern’ to describe native birds classified by the New Zealand Threat Classification System (NZTCS) as ‘threatened’ or ‘at risk’ (Robertson *et al.*, 2017)

analysis of restoration success beyond a focus on plants, investigating the complex interactions between vegetation, native forest birds and introduced mammals. This thesis recognizes the pivotal role people play in urban ecosystems, reaching beyond the disciplinary boundary of ecology to incorporate human values and behaviour in an evaluation of the benefits of urban forest restoration.

Adopting a multidisciplinary framework allowed this study to support the claim that the health of people and the natural environment are inextricably linked. My findings illustrate once again the importance of urban greenspaces to human wellbeing and demonstrate that native forest parks are highly valued for the sense of escape and for reconnecting people with nature. Urban planning laws should therefore ensure the availability of native forest parks within walking distance of city residents' homes, workplaces and schools. In the UK, the government agency Natural England recommends that every resident have access to a minimum of two hectares of natural greenspace within 300 m of their homes (Soga & Gaston, 2016). We recommend the same for New Zealand cities. People can develop strong attachments to the places where they live and are therefore most likely to contribute their time and money to the care of local greenspaces and biodiversity (Shanahan *et al.*, 2018). This emotional attachment to place can be harnessed by urban greenspace managers by focusing their efforts on attracting volunteers from the local community.

Given that our health, happiness, comfort and security are inextricably intertwined with the health of the natural world, urban planning and design should focus on finding ways of reconciling the needs of both people and native species (Miller, 2006; McDonald *et al.*, 2008; Tanner *et al.*, 2014). This will require a revolutionary approach to urban design (McKinney, 2006) and a willingness on the part of urban residents to accept the less comfortable aspects of the natural world (Elliot *et al.*, 2016). The results of this thesis reveal that urban greenspace managers will need to grapple with tensions related to the perceived lack of safety of densely vegetated areas, as well as balancing people's park use with protection of vulnerable native flora and fauna. Finally, given that private gardens in Hamilton do not currently provide adequate resources for native birds, local authorities need to invest more time and money in the ecological restoration of public greenspaces and in promoting wildlife gardening.

This thesis demonstrated that increasing the cover of native vegetation in New Zealand cities is a vital first step to the reestablishment of native forest bird communities, while simultaneously emphasizing the irreplaceable value of forest remnants both within and beyond the city limits. This emphasis accords with the stance of the Society for Ecological Restoration (McDonald *et al.*, 2016), who caution that ‘ecological restoration should never be considered a substitute for sustainably protecting and managing existing native ecosystems.’ Managers must therefore prioritize the protection of existing remnants and promote the reconstruction of native forest, where possible, in order to increase habitat available to native birds.

My findings highlight the need to conduct effective pest control in restored urban forests. The research has demonstrated that rats are decreasing sapling species richness and density, possibly through seed and seedling predation, thereby altering the trajectory of forest recovery. Consequently, managers must focus not only on planting trees, but on controlling populations of introduced mammals if urban forest restoration is to be successful. My data indicate that forest restoration creates habitat for rats, mice and possums within 9 years. Therefore, I recommend restoration projects budget for intensive pest control to commence within the first 9 years of planting.

This thesis demonstrates the importance of monitoring restoration projects in order to evaluate their success, assess whether goals are being met, and allow for early detection of unintended consequences which require adaptive management to correct. In their review of how success has been evaluated in restoration projects, Ruiz-Jaen and Mitchell Aide (2005) analysed 468 articles published in *Restoration Ecology* and found that only 68 (14.5%) evaluated restoration success after seeding or planting. However, when measuring the effectiveness of restoration, especially in urban environments, it is vital to look beyond direct benefits to native flora and fauna. As Miller (2006, p. 360) argued, ‘if this is our only yardstick, nature reserves or conservation action in urban environments often pale in comparison to their rural counterparts’. Instead, restoration success should also be measured on the scale of human experience. It is on this scale of human wellbeing and connection to nature that urban restoration projects may have the most to contribute (van Heezik & Seddon, 2018). Thus, ecological restoration in cities should always be done for and with people (Shwartz *et al.*, 2014a) and measurements of success should include the benefits provided to urban residents.

My research highlights the need to embrace a multidisciplinary management approach to urban restoration. Designing and managing cities that reconcile the needs of people and native biodiversity will require new collaboration with ecologists, social scientists, city planners, landscape architects, educators, politicians and media experts. Nevertheless, as Lélé and Norgaard (2005, p. 975) point out, ‘doing collective interdisciplinary research, especially projects stressing the feedbacks between social and environmental systems, is difficult at the best of times’. Entrenched academics trained in a single discipline may be unwilling to accept that their area of expertise cannot provide all the solutions to a given problem. However, as Daily and Ehrlich (1999) argued, few significant human problems, whether it be biodiversity conservation or climate change, lie within the boundaries of current disciplines. We need academics and professionals who are comfortable crossing the ‘informational, ideological and political chasms that separate disciplines and are obstacles in the way of working together for a more sustainable lifestyle’ (Aronson *et al.*, 2006). I argue that this does not mean, for example, that ecologists must become social scientists as well, but that management teams should be comprised of ecologists and social scientists who are willing to work together, respecting the knowledge and expertise of other disciplines.

5.3 Recommendations for further research

Given the limited availability of public land that can be converted to urban forest, future research should focus on gardening for wildlife in New Zealand cities. Research should investigate whether wildlife gardening practices are capable of providing resources to New Zealand’s endemic fauna, thereby contributing to a city-wide provision of habitat for native lizards, invertebrates, bats and birds. As with management practices, I recommend a multidisciplinary approach, and ideally a multidisciplinary research team to investigate the benefits of wildlife gardening for both native wildlife and urban residents. The insights of social science will be key, as this thesis and previous research reveal that values are not closely coupled with actions and people struggle with competing priorities (Billig, 1996; Kollmuss & Agyeman, 2002; van Heezik *et al.*, 2012). Finding practical solutions that allow gardening practices to meet the needs of

both people and wildlife will require an understanding of the ecosystem of the garden and the city, as well as the motivations driving human choices.

While this research has demonstrated that urban forest restoration can increase native bird species richness and diversity over time, we need further studies to investigate whether these forests support the successful breeding and long-term survival of native forest birds, with a focus on deep endemics that are more vulnerable to predation. We further need data on the minimum forest area required by individual native bird species to sustain viable populations within New Zealand cities. While currently the amount of native habitat in New Zealand's urban areas may be insufficient to support viable populations of endemic birds of conservation concern, it may be adequate for other native fauna, such as invertebrates and lizards. Similar studies to this one could be conducted with a focus on these taxa.

Since I found a strong positive correlation between forest age and native bird species richness that was not mediated by the vegetation variables measured, I recommend that future research focus on the seasonal provision of fruit, nectar and invertebrates in restored forests and how this changes over time since initial restoration planting. In view of the importance of invertebrates to birds during the breeding season and winter, it is likely that invertebrates play a key role in supporting native bird communities and the link between native plants, invertebrates and birds needs further investigation.

While my data suggest rats are negatively impacting native sapling recruitment, studies are needed to explore the mechanisms behind this relationship, as it could be the result of seed or seedling predation, as well as browsing. Furthermore, while my model did not detect a direct effect of any other introduced mammal on native forest regeneration, it is likely that possums have at least a deleterious effect on forest regeneration and health, and I recommend more research into the impact of introduced mammals on urban forest restoration.

While this thesis offers fascinating insights into the interspecific interactions of introduced mammals in New Zealand cities, much more work is needed to disentangle the complex interactions of cats, rats, mice, rabbits, birds, lizards, invertebrates, hedgehogs and mustelids, to name a few. It is vital we further investigate the effect of cats on native wildlife in cities, though I acknowledge the difficulty of designing and

conducting research that can provide conclusive answers to the question of whether cats have a primarily positive or negative impact on native New Zealand fauna in urban areas. This question of cat ownership and its consequences for native fauna will also require a multidisciplinary approach, as people's love of cats is unlikely to be swayed by scientific facts alone. Finally, I recommend future studies investigate how urbanisation affects rodent ecology and behaviour, for example diet, habitat preferences, interspecific interactions and home ranges. This thesis suggests urbanization alters trophic interactions between top predators in the system (cats and mustelids) and rodents. We need more studies in urban areas in order to understand the assembly and function of novel communities of introduced and native species within the highly altered ecosystem and environmental conditions of the modern city (Groffman *et al.*, 2017). Understanding how urbanisation changes rodent behaviour and ecology could also contribute to achieving the goals of Predator Free NZ (Predator Free NZ, 2019).

As the notion of the Anthropocene suggests, humans have altered every ecosystem on the planet. The most serious consequences are seen in cities, where the loss of biodiversity threatens to further alienate humans from the natural world. Nevertheless, restoring nature is not simply a site-specific attempt to recapture an unadulterated biological past. The formidable global environmental issues of the 21st century have emerged from the disconnection of people and nature. In response, building a bridge between the social and ecological restoration sciences provides one avenue for fostering greater reconnection and more functional human relationships with the natural world.

References

- Anderson, D. R. (2001a). The need to get the basics right in wildlife field studies. *Wildlife Society Bulletin (1973-2006)*, 29(4), 1294-1297.
- Anderson, M. G. (2013). *Grey warbler*. New Zealand Birds Online. Retrieved 23.08, 2016, from <http://nzbirdsonline.org.nz/species/grey-warbler>.
- Anderson, M. J. (2001b). A new method for non-parametric multivariate analysis of variance. *Austral ecology*, 26(1), 32-46.
- Anderson, M. J. (2014). Permutational multivariate analysis of variance (PERMANOVA). *Wiley StatsRef: Statistics Reference Online*, 1-15.
- Anderson, S. H., Kelly, D., Ladley, J. J., Molloy, S., & Terry, J. (2011). Cascading effects of bird functional extinction reduce pollination and plant density. *Science*, 331(6020), 1068-1071.
- Andren, H. (1994). Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: A review. *Oikos*, 71(3), 355-366.
- Aronson, J., Milton, S. J., Blignaut, J. N., & Clewell, A. F. (2006). Nature conservation as if people mattered. *Journal for Nature Conservation*, 14(3-4), 260-263.
- Aronson, M. F., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S., Cilliers, S., & Clarkson, B. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society of London B: Biological Sciences*, 281(1780), 1-8.
- Atkinson, I. A. (2001). Introduced mammals and models for restoration. *Biological Conservation*, 99(1), 81-96.
- Baker, P. J., & Harris, S. (2007). Urban mammals: What does the future hold? An analysis of the factors affecting patterns of use of residential gardens in Great Britain. *Mammal Review*, 37(4), 297-315.
- Banks, P. B., & Smith, H. M. (2015). The ecological impacts of commensal species: Black rats, *Rattus rattus*, at the urban-bushland interface. *Wildlife Research*, 42(2), 86-97.
- Barbaro, L., Brockerhoff, E. G., Giffard, B., & van Halder, I. (2012). Edge and area effects on avian assemblages and insectivory in fragmented native forests. *Landscape Ecology*, 27(10), 1451-1463.
- Belaire, J. A., Westphal, L. M., Whelan, C. J., & Minor, E. S. (2015). Urban residents' perceptions of birds in the neighborhood: Biodiversity, cultural ecosystem services, and disservices. *The Condor*, 117(2), 192-202.

- Belaire, J. A., Whelan, C. J., & Minor, E. S. (2014). Having our yards and sharing them too: The collective effects of yards on native bird species in an urban landscape. *Ecological Applications*, 24(8), 2132-2143.
- Belder, D. J., Pierson, J. C., Ikin, K., & Lindenmayer, D. B. (2018). Beyond pattern to process: Current themes and future directions for the conservation of woodland birds through restoration plantings. *Wildlife Research*, 45(6), 473-489.
- Bellingham, P., Wiser, S., Hall, G., Alley, J., Allen, R. B., & Suisted, P. (1999). Impacts of possum browsing on the long-term maintenance of forest biodiversity. *Science for Conservation*, 103, 1-60.
- Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters*, 18(6), 581-592.
- Bergquist, C. A. (1985). Movements of groups of tui (*Prothemadera novaeseelandiae*) in winter and settlement of juvenile tui in summer. *New Zealand Journal of Zoology*, 12(4), 569-571.
- Bernardini, C., & Irvine, K. (2007). The 'nature' of urban sustainability: Private or public greenspaces? *WIT Transactions on Ecology and the Environment*, 102, 661-674.
- Bhatti, M., & Church, A. (2000). 'I never promised you a rose garden': Gender, leisure and home-making. *Leisure Studies*, 19(3), 183-197.
- Bibby, C. J., Burgess, N. D., Hill, D. A., & Mustoe, S. (2000). *Bird census techniques*. (2nd ed.). London: Academic Press.
- Billig, M. (1996). *Arguing and thinking: A rhetorical approach to social psychology*. Cambridge, UK: Cambridge University Press.
- BirdLife International. (2018). *State of the world's birds: Taking the pulse of the planet*. BirdLife International, Cambridge, UK. Retrieved 14.03.2019, from https://www.birdlife.org/sites/default/files/attachments/BL_ReportENG_V11_spreads.pdf.
- Blair, R. (2004). The effects of urban sprawl on birds at multiple levels of biological organization. *Ecology and Society*, 9(5), 2.
- Blake, J. (1999). Overcoming the 'value-action gap' in environmental policy: Tensions between national policy and local experience. *Local Environment*, 4(3), 257-278.
- Botzat, A., Fischer, L. K., & Kowarik, I. (2016). Unexploited opportunities in understanding liveable and biodiverse cities. A review on urban biodiversity perception and valuation. *Global Environmental Change*, 39, 220-233.

- Bouma, C., Huizenga, E., & Warners, D. (2013). Assessing a reconciliation ecology approach to suburban landscaping: Biodiversity on a college campus. *The Michigan Botanist*, 52, 93-104.
- Bridgman, L., Innes, J., Gillies, C., Fitzgerald, N., Rohan, M., & King, C. (2018). Interactions between ship rats and house mice at Pureora Forest Park. *New Zealand Journal of Zoology*, 45(3), 238-256.
- Bruner, J. S. (2009). *Actual minds, possible worlds*. Cambridge, MA: Harvard University Press.
- Burge, O. R., Innes, J., Fitzgerald, N., & Richardson, S. J. (2017). *Habitat availability for native New Zealand bird species within the Cape-to-City footprint: A preliminary assessment*. Landcare Research, Landcare Research.
- Byrne, B. M. (2001). *Structural equation modeling with AMOS: Basic concepts, applications, and programming*. Mahwah, NJ: Laurence Erlbaum Associates, Inc., Publishers.
- Byrom, A. E., Innes, J., & Binny, R. N. (2016). A review of biodiversity outcomes from possum-focused pest control in New Zealand. *Wildlife Research*, 43(3), 228-253.
- Campbell, D. (1990). Changes in the structure and composition of a New Zealand lowland forest inhabited by brushtail possums.
- Campbell, D., & Atkinson, I. A. (2002). Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. *Biological conservation*, 107(1), 19-35.
- Caut, S., Angulo, E., & Courchamp, F. (2009). Avoiding surprise effects on Surprise Island: Alien species control in a multitrophic level perspective. *Biological Invasions*, 11(7), 1689-1703.
- Chace, J. F., & Walsh, J. J. (2006). Urban effects on native avifauna: A review. *Landscape and urban planning*, 74(1), 46-69.
- Chawla, L. (1998). Significant life experiences revisited: A review of research on sources of environmental sensitivity. *The Journal of Environmental Education*, 29(3), 11-21.
- Childs, J. E. (1986). Size-dependent predation on rats (*Rattus norvegicus*) by house cats (*Felis catus*) in an urban setting. *Journal of Mammalogy*, 67(1), 196-199.
- Christens, B. D. (2016). Pragmatism, praxis, and predictive theory. *Global Journal of Community Psychology Practice*, 7(2), 1-6.
- Clarkson, B., Bryan, C., & Clarkson, F. (2012). Reconstructing Hamilton's Indigenous Ecosystems: The Waiwhakareke Natural Heritage Park. *City Green*, 4, 60-67.

- Clarkson, B. D. (1985). The vegetation of the Kaitake Range Egmont National Park, New Zealand. *New Zealand Journal of Botany*, 23(1), 15-31.
- Clarkson, B. D. (2011). Ecological heritage in the Taranaki Region. In J. Ruru, J. Stephenson & M. Abbot (Eds.), *Making our place: Exploring land use tensions in Aotearoa New Zealand* (pp. 115-128). Dunedin, New Zealand: Otago University Press.
- Clarkson, B. D., & McQueen, J. C. (2004). *Ecological restoration in Hamilton City, North Island, New Zealand*. Presented at the 16th International Conference of the Society for Ecological Restoration, Victoria, Canada, retrieved from http://www.gullyguide.co.nz/files/019_Ecological_restoration_in_Hamilton_City.pdf.
- Clarkson, B. D., Wehi, P. M., & Brabyn, L. K. (2007a). A spatial analysis of indigenous cover patterns and implications for ecological restoration in urban centres, New Zealand. *Urban Ecosystems*, 10(4), 441-457.
- Clarkson, B. D., Wehi, P. M., & Brabyn, L. K. (2007b). *Bringing back nature into cities: Urban land environments, indigenous cover and urban restoration*. Research Report 52. Centre for Biodiversity and Ecology Research, University of Waikato, Hamilton, New Zealand. Retrieved 26.09.2019, from https://www.waikato.ac.nz/__data/assets/pdf_file/0010/471997/CBER_52.pdf.
- Clout, M., Karl, B., & Gaze, P. (1991). Seasonal movements of New Zealand pigeons from a lowland forest reserve. *Notornis*, 38(1), 37-47.
- Coleman, E. J., & Clarkson, B. D. (2010). Cities biodiversity index for Hamilton City, New Zealand. *The University of Waikato, Hamilton*.
- Cornes, T. S., Thomson, R. E., & Clarkson, B. D. (2012). *Key ecological sites of Hamilton City*. Volume I, CBER Contract Report. Centre for Biodiversity and Ecology Research, University of Waikato, Hamilton, New Zealand. Retrieved 01.02.2018, from https://researchcommons.waikato.ac.nz/bitstream/handle/10289/6565/CBER_121.pdf?sequence=1.
- Cosquer, A., Raymond, R., & Prevot-Julliard, A.-C. (2012). Observations of everyday biodiversity: A new perspective for conservation? *Ecology and Society*, 17(4), 2.
- Courchamp, F., Chapuis, J.-L., & Pascal, M. (2003). Mammal invaders on islands: impact, control and control impact. *Biological Reviews*, 78(3), 347-383.
- Cowan, P. E. (2005). Brushtail possum. In C. King (Ed.), *The Handbook of New Zealand Mammals*. Sydney, Australia: Oxford University Press.
- Cox, D. T., & Gaston, K. J. (2015). Likeability of garden birds: Importance of species knowledge and richness in connecting people to nature. *PloS one*, 10(11).

- Cox, M. P., Dickman, C. R., & Cox, W. G. (2001). Use of habitat by the black rat (*Rattus rattus*) at North Head, New South Wales: An observational and experimental study. *Austral Ecology*, 25(4), 375-385.
- Crooks, K. R., & Soulé, M. E. (1999). Mesopredator release and avifaunal extinctions in a fragmented system. *Nature*, 400(6744), 563.
- Curtin, S. (2009). Wildlife tourism: The intangible, psychological benefits of human-wildlife encounters. *Current Issues in Tourism*, 12(5-6), 451-474.
- Daily, G. C., & Ehrlich, P. R. (1999). Managing earth's ecosystems: An interdisciplinary challenge. *Ecosystems*, 2(4), 277-280.
- Dallimer, M., Irvine, K. N., Skinner, A. M., Davies, Z. G., Rouquette, J. R., Maltby, L. L., Warren, P. H., Armsworth, P. R., & Gaston, K. J. (2012). Biodiversity and the feel-good factor: Understanding associations between self-reported human well-being and species richness. *BioScience*, 62(1), 47-55.
- Daniels, G., & Kirkpatrick, J. (2006). Does variation in garden characteristics influence the conservation of birds in suburbia? *Biological Conservation*, 133(3), 326-335.
- Day, T. D. (1995). Bird species composition and abundance in relation to native plants in urban gardens, Hamilton, New Zealand. *Notornis*, 42(3), 172-186.
- Deconchat, M., Brockerhoff, E., & Barbaro, L. (2009). Effects of surrounding landscape composition on the conservation value of native and exotic habitats for native forest birds. *Forest Ecology and Management*, 258, S196-S204.
- Department of Conservation. (n.d.). *Hakarimata Walkway*. Retrieved 06.02., 2019, from <https://www.doc.govt.nz/parks-and-recreation/places-to-go/waikato/places/hakarimata-scenic-reserve/tracks/hakarimata-walkway>.
- Derhé, M. A., Murphy, H., Monteith, G., & Menéndez, R. (2016). Measuring the success of reforestation for restoring biodiversity and ecosystem functioning. *Journal of Applied Ecology*, 53(6), 1714-1724.
- Diamond, J. M. (1984). Distributions of New Zealand birds on real and virtual islands. *New Zealand Journal of Ecology*, 7, 37-55.
- Donnelly, R., & Marzluff, J. M. (2004). Importance of reserve size and landscape context to urban bird conservation. *Conservation Biology*, 18(3), 733-745.
- Doody, B. J., Sullivan, J. J., Meurk, C. D., Stewart, G. H., & Perkins, H. C. (2010). Urban realities: The contribution of residential gardens to the conservation of urban forest remnants. *Biodiversity and Conservation*, 19(5), 1385-1400.
- Drinnan, I. N. (2005). The search for fragmentation thresholds in a southern Sydney suburb. *Biological Conservation*, 124(3), 339-349.

- Duguid, M. C., Morrell, E. H., Goodale, E., & Ashton, M. S. (2016). Changes in breeding bird abundance and species composition over a 20 year chronosequence following shelterwood harvests in oak-hardwood forests. *Forest Ecology and Management*, 376, 221-230.
- Dunn, R. R., Gavin, M. C., Sanchez, M. C., & Solomon, J. N. (2006). The pigeon paradox: Dependence of global conservation on urban nature. *Conservation Biology*, 20(6), 1814-1816.
- Efford, M., Fitzgerald, B., Karl, B., & Berben, P. (2006). Population dynamics of the ship rat *Rattus rattus* L. in the Orongorongo Valley, New Zealand. *New Zealand Journal of Zoology*, 33(4), 273-297.
- Elliot, E. E., Vallance, S., & Molles, L. E. (2016). Coexisting with coyotes (*Canis latrans*) in an urban environment. *Urban Ecosystems*, 19(3), 1335-1350.
- Elliott, R., & Timulak, L. (2005). Descriptive and interpretive approaches to qualitative research. In J. Miles & P. Gilbert (Eds.), *A handbook of research methods for clinical and health psychology* (Chapter 147-159). Oxford, UK: Oxford University Press.
- Evans, K. L., Newson, S. E., & Gaston, K. J. (2009). Habitat influences on urban avian assemblages. *Ibis*, 151(1), 19-39.
- Evely, A. C., Fazey, I. R. A., Pinard, M., & Lambin, X. (2008). The influence of philosophical perspectives in integrative research: a conservation case study in the Cairngorms National Park. *Ecology and Society*, 13(52).
- Feagan, R., & Ripmeester, M. (2001). Reading private green space: Competing geographic identities at the level of the lawn. *Philosophy & Geography*, 4(1), 79-95.
- Feng, A. Y., & Himsforth, C. G. (2014). The secret life of the city rat: A review of the ecology of urban Norway and black rats (*Rattus norvegicus* and *Rattus rattus*). *Urban Ecosystems*, 17(1), 149-162.
- Fernandez-Juricic, E., & Jokimäki, J. (2001). A habitat island approach to conserving birds in urban landscapes: Case studies from southern and northern Europe. *Biodiversity & Conservation*, 10(12), 2023-2043.
- Fischer, J., & Lindenmayer, D. (2006). Beyond fragmentation: The continuum model for fauna research and conservation in human-modified landscapes. *Oikos*, 112(2), 473-480.
- Fischer, J. D., Cleeton, S. H., Lyons, T. P., & Miller, J. R. (2012). Urbanization and the predation paradox: the role of trophic dynamics in structuring vertebrate communities. *Bioscience*, 62(9), 809-818.
- Fitzgerald, N., & Innes, J. (2015). *Hamilton City biennial bird counts: 2004-2014*. Landcare Research. Retrieved 30.10.2018, from

<https://www.hamilton.govt.nz/our-services/environment-and-health/biodiversity/Documents/Landcare%20Hamilton%20bird%20survey.pdf>.

- Fitzgerald, N., Innes, J., & Sandoval, N. (2017). *Rat and possum movement in a Hamilton gully*. Report Number LC2816. Landcare Research, Landcare Research, Hamilton. 1-11p.
- Flyvbjerg, B. (2001). *Making social science matter: Why social inquiry fails and how it can succeed again*. Cambridge, UK: Cambridge University Press.
- Forman, R. T. (2014). *Urban ecology: Science of cities*. Cambridge, UK: Cambridge University Press.
- Francis, M., Hester Jr, R. T., & Hester, R. T. (1990). *The meaning of gardens: Idea, place, and action*. Cambridge, MA: MIT Press.
- Fredrickson, L. M., & Anderson, D. H. (1999). A qualitative exploration of the wilderness experience as a source of spiritual inspiration. *Journal of Environmental Psychology*, 19(1), 21-39.
- Freeman, C., Dickinson, K. J., Porter, S., & van Heezik, Y. (2012). “My garden is an expression of me”: Exploring householders' relationships with their gardens. *Journal of Environmental Psychology*, 32(2), 135-143.
- Fryer, A. (1998). *Davies, Victor Caddy*. Te Ara - The Encyclopedia of New Zealand. Retrieved 26.10., 2019, from <https://teara.govt.nz/en/biographies/4d5/davies-victor-caddy>.
- Fuller, R. A., & Irvine, K. N. (2010). Interactions between people and nature in urban environments. *Urban Ecology*, 134.
- Galbraith, J. A., Beggs, J. R., Jones, D. N., McNaughton, E. J., Krull, C. R., & Stanley, M. C. (2014). Risks and drivers of wild bird feeding in urban areas of New Zealand. *Biological Conservation*, 180, 64-74.
- Gill, B. J. (1982). Breeding of the grey warbler *Gerygone igata* at Kaikoura, New Zealand. *Ibis*, 124(2), 123-147.
- Gill, B. J., Bell, B. D., Chambers, G. K., Medway, D. G., Palma, R. L., Scofield, R. P., Tennyson, A. J. D., & Worthy, T. H. (2010). *Checklist of the birds of New Zealand, Norfolk and Macquarie Islands, and the Ross Dependency, Antarctica*. (4th ed.). Wellington, New Zealand: Te Papa Press.
- Gill, N., Waitt, G., & Head, L. (2009). Local engagements with urban bushland: moving beyond bounded practice for urban biodiversity management. *Landscape and Urban Planning*, 93(3), 184-193.
- Gillies, C. (2001). Advances in New Zealand mammalogy 1990–2000: House cat. *Journal of the Royal Society of New Zealand*, 31(1), 205-218.

- Glen, A., Warburton, B., Cruz, J., & Coleman, M. (2014). Comparison of camera traps and kill traps for detecting mammalian predators: A field trial. *New Zealand Journal of Zoology*, 41(3), 155-160.
- Gobster, P. H. (2000). Restoring nature: Human actions, interactions, and reactions. In P. H. Gobster & R. B. Hull (Eds.), *Restoring nature: Perspectives from the social sciences and humanities* (pp. 1-19). Washington, DC: Island Press.
- Goddard, M. A., Dougill, A. J., & Benton, T. G. (2010). Scaling up from gardens: Biodiversity conservation in urban environments. *Trends in Ecology & Evolution*, 25(2), 90-98.
- Gordon, J., Matthaei, C., & Van Heezik, Y. (2010). Belled collars reduce catch of domestic cats in New Zealand by half. *Wildlife Research*, 37(5), 372-378.
- Gotelli, N. J., & Ellison, A. M. (2004). *Primer of ecological statistics*. Sunderland, MA: Sinauer Associates Publishers.
- Gould, J. (1993). *Eight little piggies: Reflections in natural history*. New York, NY: WW Norton and Company.
- Grace, J. B. (2008). Structural equation modeling for observational studies. *The Journal of Wildlife Management*, 72(1), 14-22.
- Gray, E. R., & van Heezik, Y. (2016). Exotic trees can sustain native birds in urban woodlands. *Urban Ecosystems*, 19(1), 315-329.
- Grob, A. (1991). *Meinung, verhalten, umwelt*. Bern: Peter Lang Verlag.
- Groffman, P. M., Cadenasso, M. L., Cavender-Bares, J., Childers, D. L., Grimm, N. B., Grove, J. M., Hobbie, S. E., Hutyrá, L. R., Jenerette, G. D., & McPhearson, T. (2017). Moving towards a new urban systems science. *Ecosystems*, 20(1), 38-43.
- Gross, H., & Lane, N. (2007). Landscapes of the lifespan: Exploring accounts of own gardens and gardening. *Journal of Environmental Psychology*, 27(3), 225-241.
- Hamilton City Council. (2007). *Gully Reserves Management Plan*. December 2007. Hamilton, New Zealand. Retrieved 15.10.2019, from <https://www.hamilton.govt.nz/our-city/parks/parksandgardens/Documents/Gully%20Reserves%202007.pdf>.
- Hamilton City Council. (n.d.). *Gully Restoration Programme*. Retrieved 30.10., 2019, from <http://www.gullyguide.co.nz/index.asp>.
- Hargreaves, T. (2012). Questioning the virtues of pro-environmental behaviour research: Towards a phronetic approach. *Geoforum*, 43(2), 315-324.

- Harper, G. A., Dickinson, K. J., & Seddon, P. J. (2005). Habitat use by three rat species (*Rattus* spp.) on Stewart Island/Rakiura, New Zealand. *New Zealand Journal of Ecology*, 29, 251-260.
- Head, L., & Muir, P. (2006). Suburban life and the boundaries of nature: resilience and rupture in Australian backyard gardens. *Transactions of the Institute of British Geographers*, 31(4), 505-524.
- Head, L. M., & Muir, P. (2005). Living with trees: Perspectives from the suburbs. In M. Calver, H. Bigler-Cole, G. Bolton, A. Gaynor, P. Horwitz, J. Mills & G. Wardell-Johnson (Eds.), *Sixth National Conference of the Australian Forest History Society Inc.* (pp. 85-94). Augusta, Western Australia: Millpress Science Publishers.
- Heather, B. D., & Robertson, H. A. (2015). *The field guide to the birds of New Zealand*. Penguin Random House New Zealand.
- Heidenheimer, A. J., Heclo, H., & Adams, C. T. (1990). *Comparative public policy: The politics of social choice in America, Europe, and Japan*. New York, NY: St. Martin's Press.
- Henwood, K., & Pidgeon, N. (2001). Talk about woods and trees: Threat of urbanization, stability, and biodiversity. *Journal of Environmental Psychology*, 21(2), 125-147.
- Herzog, T. R., Maguire, P., & Nebel, M. B. (2003). Assessing the restorative components of environments. *Journal of Environmental Psychology*, 23(2), 159-170.
- Hill, M. O. (1973). Diversity and evenness: a unifying notation and its consequences. *Ecology*, 54(2), 427-432.
- Hill, M. T. (2003). *Diet, dispersal and distribution of kereru (Hemiphaga novaeseelandiae novaeseelandia) in a lowland podocarp-hardwood forest*. thesis, Massey University, Palmerston North, New Zealand.
- Hinds, J., & Sparks, P. (2008). Engaging with the natural environment: The role of affective connection and identity. *Journal of Environmental Psychology*, 28(2), 109-120.
- Hobbs, R. J., & Norton, D. A. (1996). Towards a conceptual framework for restoration ecology. *Restoration Ecology*, 4(2), 93-110.
- Hodgetts, D., & Stolte, O. (2012). Case-based research in community and social psychology: Introduction to the special issue. *Journal of Community & Applied Social Psychology*, 22(5), 379-389.
- Hodgetts, D., Stolte, O., Sonn, C., Drew, N., Nikora, L., & Carr, S. (2020). *Social psychology and everyday life*. (2nd ed.). Houndmills, Basingstoke: Red Globe Press/Macmillan Education.

- Holl, K. D., & Quiros-Nietzen, E. (1999). The effect of rabbit herbivory on reforestation of abandoned pasture in southern Costa Rica. *Biological Conservation*, 87(3), 391-395.
- Home, R., Bauer, N., & Hunziker, M. (2010). Cultural and biological determinants in the evaluation of urban green spaces. *Environment and Behavior*, 42(4), 494-523.
- Home, R., Keller, C., Nagel, P., Bauer, N., & Hunziker, M. (2009). Selection criteria for flagship species by conservation organizations. *Environmental Conservation*, 36(02), 139-148.
- Hope, D., Gries, C., Zhu, W., Fagan, W. F., Redman, C. L., Grimm, N. B., Nelson, A. L., Martin, C., & Kinzig, A. (2003). Socioeconomics drive urban plant diversity. *Proceedings of the national academy of sciences*, 100(15), 8788-8792.
- Hughey, K. F., Kerr, G. N., & Cullen, R. (2013). *Public perceptions of New Zealand's environment: 2013*. EOS Ecology. Christchurch, New Zealand. Retrieved 10.04.2016, from <https://researcharchive.lincoln.ac.nz/handle/10182/5744>.
- Hurst, J., & Allen, R. (2007). *The Recce method for describing New Zealand vegetation: field protocols*. Lincoln, New Zealand: Landcare Research.
- Ingold, T. (1995). Building, dwelling, living: how animals and people make themselves at home in the world. In M. Strathern (Ed.), *Shifting contexts: Transformations in anthropological knowledge* (pp. 57-80). London, UK: Routledge.
- Innes, J. (2005). Ship rat. In C. King (Ed.), *The Handbook of New Zealand Mammals*. Australia: Oxford University Press.
- Innes, J., & Fitzgerald, N. (2018). *Hamilton Halo Strategic Review: Future options for Halo goals and implementation*. Landcare Research, Hamilton.
- Innes, J., Kelly, D., Overton, J. M., & Gillies, C. (2010a). Predation and other factors currently limiting New Zealand forest birds. *New Zealand Journal of Ecology*, 34(1), 86.
- Innes, J., King, C., Bartlam, S., Forrester, G., & Howitt, R. (2015). Predator control improves nesting success in Waikato forest fragments. *New Zealand Journal of Ecology*, 39(2), 245.
- Innes, J., King, C. M., Bridgman, L., Fitzgerald, N., Arnold, G., & Cox, N. (2010b). Effect of grazing on ship rat density in forest fragments of lowland Waikato, New Zealand. *New Zealand Journal of Ecology*, 34, 227-232.
- Innes, J., Nugent, G., Prime, K., & Spurr, E. B. (2004). Responses of kukupa (*Hemiphaga novaeseelandiae*) and other birds to mammal pest control at Motatau, Northland. *New Zealand Journal of Ecology*, 28, 73-81.

- Ives, C. D., Lentini, P. E., Threlfall, C. G., Ikin, K., Shanahan, D. F., Garrard, G. E., Bekessy, S. A., Fuller, R. A., Mumaw, L., & Rayner, L. (2016). Cities are hotspots for threatened species. *Global Ecology and Biogeography*, 25(1), 117-126.
- Jager, B. (1975). Theorizing, journeying, dwelling. In A. Giorgi, C. Fischer & E. Murray (Eds.), *Duquesne studies in phenomenological psychology* (pp. 235-260). Pittsburgh, PA: Duquesne University Press.
- Jay, M., & Stolte, O. (2011). A human ecology of urban ravine restoration: A New Zealand example. *Urban Habitats*, 6, 1541-7115.
- Johnston, C. (2008). Beyond the clearing: towards a dwelt animal geography. *Progress in Human Geography*, 32(5), 633-649.
- Jokimäki, J. (1999). Occurrence of breeding bird species in urban parks: Effects of park structure and broad-scale variables. *Urban Ecosystems*, 3(1), 21-34.
- Jones, D. N., & James Reynolds, S. (2008). Feeding birds in our towns and cities: A global research opportunity. *Journal of Avian Biology*, 39(3), 265-271.
- Jordan III, W. R. (2000). Restoration, community, and wilderness. In P. H. Gobster & R. B. Hull (Eds.), *Restoring nature: Perspectives from the social sciences and humanities* (pp. 23-36). Washington, D.C.: Island Press.
- Kang, W., Minor, E. S., Park, C.-R., & Lee, D. (2015). Effects of habitat structure, human disturbance, and habitat connectivity on urban forest bird communities. *Urban Ecosystems*, 18(3), 857-870.
- Kaplan, S. (1995). The restorative benefits of nature: Toward an integrative framework. *Journal of Environmental Psychology*, 15(3), 169-182.
- Kaplan, S. (2000). New ways to promote proenvironmental behavior: Human nature and environmentally responsible behavior. *Journal of Social Issues*, 56(3), 491-508.
- Kellert, S. R., & Berry, J. K. (1980). *Knowledge, Affection and Basic Attitudes Toward Animals in American Society: Phase III*. Washington, DC: US Fish and Wildlife Service.
- Kelly, D., Ladley, J. J., Robertson, A. W., Anderson, S. H., Wotton, D. M., & Wiser, S. K. (2010). Mutualisms with the wreckage of an avifauna: The status of bird pollination and fruit-dispersal in New Zealand. *New Zealand Journal of Ecology*, 34(1), 66.
- Kelly, D., Robertson, A., Ladley, J., Anderson, S., & McKenzie, R. (2006). The relative (un) importance of introduced animals as pollinators and dispersers of native plants. In R. Allen & W. Lee (Eds.), *Biological Invasions in New Zealand* (pp. 227-245). Berlin: Springer.

- Keniger, L. E., Gaston, K. J., Irvine, K. N., & Fuller, R. A. (2013). What are the benefits of interacting with nature? *International Journal of Environmental Research and Public Health*, 10(3), 913-935.
- Kiesling, F. M., & Manning, C. M. (2010). How green is your thumb? Environmental gardening identity and ecological gardening practices. *Journal of Environmental Psychology*, 30(3), 315-327.
- Kikillus, K. H., Chambers, G. K., Farnworth, M. J., & Hare, K. M. (2017). Research challenges and conservation implications for urban cat management in New Zealand. *Pacific Conservation Biology*, 23(1), 15-24.
- King, C. M. (2005). *The handbook of New Zealand mammals*. (2nd ed.). Oxford, UK: Oxford University Press.
- King, C. M., Innes, J., Flux, M., Kimberley, M., Leathwick, J., & Williams, D. (1996). Distribution and abundance of small mammals in relation to habitat in Pureora Forest Park. *New Zealand Journal of Ecology*, 20, 215-240.
- Kollmuss, A., & Agyeman, J. (2002). Mind the gap: Why do people act environmentally and what are the barriers to pro-environmental behavior? *Environmental Education Research*, 8(3), 239-260.
- Lambert, R., & Hartley, B. (2016). Ornithological Society of New Zealand records.
- Laughlin, D., & Grace, J. (2019). Discoveries and novel insights in ecology using structural equation modeling. *Ideas in Ecology and Evolution*, 12, 28-34.
- Le Corre, M. (2008). Conservation biology: Cats, rats and seabirds. *Nature*, 451(7175), 134.
- Lee, W. G., McGlone, M., & Wright, E. (2005). *Biodiversity inventory and monitoring: A review of national and international systems and a proposed framework for future biodiversity monitoring by the Department of Conservation*. Landcare Research. Retrieved 28.06.2019, from https://www.landcareresearch.co.nz/publications/researchpubs/biodiv_inventory_system_review_framework.pdf.
- Lefcheck, J. S. (2016). piecewiseSEM: Piecewise structural equation modelling in R for ecology, evolution, and systematics. *Methods in Ecology and Evolution*, 7(5), 573-579.
- Lefcheck, J. S. (2018). *piecewiseSEM: Piecewise Structural Equation Modeling in R*. Retrieved 18.04., 2019, from <https://cran.r-project.org/web/packages/piecewiseSEM/vignettes/piecewiseSEM.html#correlated-errors>.
- Lélé, S., & Norgaard, R. B. (2005). Practicing interdisciplinarity. *BioScience*, 55(11), 967-975.

- Leopold, A. (1949). *A Sand County Almanac and Sketches Here and There*. New York: Oxford University Press.
- Lerman, S. B., & Warren, P. S. (2011). The conservation value of residential yards: Linking birds and people. *Ecological Applications*, 21(4), 1327-1339.
- Letcher, S. G., & Chazdon, R. L. (2009). Rapid recovery of biomass, species richness, and species composition in a forest chronosequence in northeastern Costa Rica. *Biotropica*, 41(5), 608-617.
- Levene, H. (1960). Robust tests for equality of variances. In I. Olkin (Ed.), *Contributions to Probability and Statistics: Essays in Honor of Harold Hotelling* (pp. 279-292). Palo Alto, CA: Stanford University Press.
- Lincoln, S. (2016). *Indirect impacts of mammalian pest control: Behavioural responses of cats (Felis catus) to rodent control in urban forest fragments*. thesis, The University of Auckland, Auckland, New Zealand.
- Lindemann-Mathies, P., & Marty, T. (2013). Does ecological gardening increase species richness and aesthetic quality of a garden? *Biological Conservation*, 159, 37-44.
- Loder, A. (2014). 'There's a meadow outside my workplace': A phenomenological exploration of aesthetics and green roofs in Chicago and Toronto. *Landscape and Urban Planning*, 126, 94-106.
- Longhurst, R. (2006). Plots, plants and paradoxes: Contemporary domestic gardens in Aotearoa/New Zealand. *Social & Cultural Geography*, 7(4), 581-593.
- Louv, R. (2008). *Last child in the woods: Saving our children from nature-deficit disorder*. Chapel Hill, NC: Algonquin Books.
- Mac Nally, R., De Vries, L., & Thomson, J. R. (2010). Are replanted floodplain forests in southeastern Australia providing bird biodiversity benefits? *Restoration Ecology*, 18(1), 85-94.
- MacArthur, R. H., & Wilson, E. O. (1967). *The theory of island biogeography*. Princeton, NJ: Princeton University Press.
- MacKay, D. B. (2006). *Ecology of restored gully forest patches in Hamilton Ecological District*. thesis, University of Waikato, Hamilton, New Zealand.
- MacLeod, L. J., Dickson, R., Leckie, C., Stephenson, B. M., & Glen, A. S. (2015). Possum control and bird recovery in an urban landscape, New Zealand. *Conserv. Evidence*, 12, 44-47.
- Martinez Arbizu, P. (2019). *pairwiseAdonis: Pairwise multilevel comparison using adonis. R package version 0.3*. Retrieved 29.03, 2019, from <https://github.com/pmartinezarbizu/pairwiseAdonis>.

- Marzluff, J. M., & Ewing, K. (2001). Restoration of fragmented landscapes for the conservation of birds: a general framework and specific recommendations for urbanizing landscapes. *Restoration Ecology*, 9(3), 280-292.
- Mascia, M. B., Brosius, J. P., Dobson, T. A., Forbes, B. C., Horowitz, L., McKean, M. A., & Turner, N. J. (2003). Conservation and the social sciences. *Conservation Biology*, 17(3), 649-650.
- Matias, R., & Catry, P. (2008). The diet of feral cats at New Island, Falkland Islands, and impact on breeding seabirds. *Polar Biology*, 31(5), 609-616.
- McDonald, R. I. (2016). Urban ecology for the urban century. *Ecosystem Health and Sustainability*, 2(7), e01221.
- McDonald, R. I., Kareiva, P., & Forman, R. T. (2008). The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, 141(6), 1695-1703.
- McDonald, T., Gann, G., Jonson, J., & Dixon, K. (2016). *International standards for the practice of ecological restoration—including principles and key concepts*. Washington, DC: Society for Ecological Restoration.
- McIntyre, N. E., Knowles-Yáñez, K., & Hope, D. (2000). Urban ecology as an interdisciplinary field: Differences in the use of “urban” between the social and natural sciences. *Urban Ecosystems*, 4, 5-24.
- McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, 127(3), 247-260.
- McKinney, M. L., & Lockwood, J. L. (1999). Biotic homogenization: A few winners replacing many losers in the next mass extinction. *Trends in Ecology & Evolution*, 14(11), 450-453.
- Melles, S., Glenn, S., & Martin, K. (2003). Urban bird diversity and landscape complexity: Species-environment associations along a multiscale habitat gradient. *Conservation Ecology*, 7(1), 5.
- Melles, S. J. (2005). Urban bird diversity as an indicator of human social diversity and economic inequality in Vancouver, British Columbia. *Urban Habitats*, 3(1), 25-48.
- Merleau-Ponty, M. (1962). *Phenomenology of Perception [Phénoménologie de la Perception]*. London: Routledge & Kegan Paul Ltd.
- Miller, J. R. (2005). Biodiversity conservation and the extinction of experience. *Trends in Ecology & Evolution*, 20(8), 430-434.
- Miller, J. R. (2006). Restoration, reconciliation, and reconnecting with nature nearby. *Biological Conservation*, 127(3), 356-361.

- Miller, J. R., Fraterrigo, J. M., Hobbs, N. T., Theobald, D. M., & Wiens, J. A. (2001). Urbanization, avian communities, and landscape ecology. In J. M. Marzluff, R. Bowman & R. Donnelly (Eds.), *Avian ecology and conservation in an urbanizing world* (pp. 117-137). New York, NY: Kluwer Academic Publishers.
- Miskelly, C., Empson, R., & Wright, K. (2005). Forest birds recolonising Wellington. *Notornis*, 52(1), 21-26.
- Moon, K., & Blackman, D. (2014). A guide to understanding social science research for natural scientists. *Conservation Biology*, 28(5), 1167-1177.
- Morgan, D., Waas, J., & Innes, J. (2009). An inventory of mammalian pests in a New Zealand city. *New Zealand Journal of Zoology*, 36(1), 23-33.
- Morgan, D., Waas, J., Innes, J., & Fitzgerald, N. (2011). Identification of nest predators using continuous time-lapse recording in a New Zealand city. *New Zealand Journal of Zoology*, 38(4), 343-347.
- Morris, E. K., Caruso, T., Buscot, F., Fischer, M., Hancock, C., Maier, T. S., Meiners, T., Müller, C., Obermaier, E., & Prati, D. (2014). Choosing and using diversity indices: insights for ecological applications from the German Biodiversity Exploratories. *Ecology and Evolution*, 4(18), 3514-3524.
- Mulder, C. P., Grant-Hoffman, M. N., Towns, D. R., Bellingham, P. J., Wardle, D. A., Durrett, M. S., Fukami, T., & Bonner, K. I. (2009). Direct and indirect effects of rats: Does rat eradication restore ecosystem functioning of New Zealand seabird islands? *Biological Invasions*, 11(7), 1671-1688.
- Munro, N. T., Fischer, J., Wood, J., & Lindenmayer, D. B. (2009). Revegetation in agricultural areas: The development of structural complexity and floristic diversity. *Ecological Applications*, 19(5), 1197-1210.
- Nassauer, J. I. (1995a). Culture and changing landscape structure. *Landscape Ecology*, 10(4), 229-237.
- Nassauer, J. I. (1995b). Messy ecosystems, orderly frames. *Landscape Journal*, 14(2), 161-170.
- NIWA. (2016). *Climate summaries*. Retrieved 06.02., 2019, from <https://www.niwa.co.nz/education-and-training/schools/resources/climate/summary>.
- Norbury, G., Byrom, A., Pech, R., Smith, J., Clarke, D., Anderson, D., & Forrester, G. (2013). Invasive mammals and habitat modification interact to generate unforeseen outcomes for indigenous fauna. *Ecological Applications*, 23(7), 1707-1721.
- Norton, D. A. (2009). Species Invasions and the Limits to Restoration: Learning from the New Zealand Experience. *Science*, 325, 569-571.

- Nottingham, C. M. (2018). *The impact of hedgehogs in urban forest fragments*. Unpublished master's thesis, University of Auckland, Auckland, New Zealand.
- Nugent, G., Fraser, W., & Sweetapple, P. (2001). Top down or bottom up? Comparing the impacts of introduced arboreal possums and 'terrestrial' ruminants on native forests in New Zealand. *Biological Conservation*, 99(1), 65-79.
- Oksanen, J. (2011). Multivariate analysis of ecological communities in R: vegan tutorial. *R package version*, 1(7), 11-12.
- Oksanen, J. (2015). Multivariate analysis of ecological communities in R: vegan tutorial. *R Doc*, 43, 11-12.
- Orr, D. (1995). Love it or lose it: The coming biophilia. In S. R. Kellert & E. O. Wilson (Eds.), *The biophilia hypothesis*. Washington DC: Island Press.
- Ortega-Álvarez, R., & Lindig-Cisneros, R. (2012). Feathering the scene: The effects of ecological restoration on birds and the role birds play in evaluating restoration outcomes. *Ecological Restoration*, 30(2), 116-127.
- Overdyck, E. (2014). *Thresholds for sustainable regeneration in urban restoration plantings in Hamilton City, New Zealand*. Unpublished doctoral thesis, University of Waikato, Hamilton, New Zealand.
- Overdyck, E., & Clarkson, B. D. (2012). Seed rain and soil seed banks limit native regeneration within urban forest restoration plantings in Hamilton City, New Zealand. *New Zealand Journal of Ecology*, 36(2), 177-190.
- Overdyck, E., Clarkson, B. D., Laughlin, D. C., & Gemmill, C. E. (2013). Testing broadcast seeding methods to restore urban forests in the presence of seed predators. *Restoration Ecology*, 21(6), 763-769.
- Paliy, O., & Shankar, V. (2016). Application of multivariate statistical techniques in microbial ecology. *Molecular Ecology*, 25(5), 1032-1057.
- Palmer, G. C., Fitzsimons, J. A., Antos, M. J., & White, J. G. (2008). Determinants of native avian richness in suburban remnant vegetation: Implications for conservation planning. *Biological Conservation*, 141(9), 2329-2341.
- Parlange, M. (1998). The city as ecosystem. *Bioscience*, 48(8), 581-585.
- Parsons, D., Michael, H., Banks, P. B., Deutsch, M. A., & Munshi-South, J. (2018). Temporal and space-use changes by rats in response to predation by feral cats in an urban ecosystem. *Frontiers in Ecology and Evolution*, 6, 146.
- Pender, R. J., Shiels, A. B., Bialic-Murphy, L., & Mosher, S. M. (2013). Large-scale rodent control reduces pre-and post-dispersal seed predation of the endangered Hawaiian lobeliad, *Cyanea superba* subsp. *superba* (Campanulaceae). *Biological Invasions*, 15(1), 213-223.

- Peres-Neto, P. R., Jackson, D. A., & Somers, K. M. (2003). Giving meaningful interpretation to ordination axes: Assessing loading significance in principal component analysis. *Ecology*, 84(9), 2347-2363.
- Pickett, S. T. (1989). Space-for-time substitution as an alternative to long-term studies. In G. Likens (Ed.), *Long-term studies in ecology* (pp. 110-135). New York, NY: Springer-Verlag.
- Pietkiewicz, I., & Smith, J. A. (2014). A practical guide to using interpretative phenomenological analysis in qualitative research psychology. *Psychological Journal*, 20(1), 7-14.
- Pollock, K. (2010). *City parks and green spaces*. Te Ara - the Encyclopedia of New Zealand. Retrieved 26.10., 2019, from <http://www.TeAra.govt.nz/en/city-parks-and-green-spaces>.
- Powlesland, R. G. (2013). *New Zealand pigeon*. New Zealand Birds Online. Retrieved 23.08, 2016, from <http://www.nzbirdsonline.org.nz/species/new-zealand-pigeon>.
- Powlesland, R. G., Moran, L. R., & Wotton, D. M. (2011). Satellite tracking of kereru (*Hemiphaga novaeseelandiae*) in Southland, New Zealand: Impacts, movements and home range. *New Zealand Journal of Ecology*, 35(3), 229-235.
- Predator Free NZ. (2019). *Predator Free NZ*. Retrieved 06.06., 2019, from <https://predatorfreenz.org/>.
- Pyle, R. M. (1993). *The thunder tree: Lessons from an urban wildland*. Boston, MA: Houghton Mifflin Boston.
- Pyle, R. M. (2003). Nature matrix: Reconnecting people and nature. *Oryx*, 37(02), 206-214.
- Ramette, A. (2007). Multivariate analyses in microbial ecology. *FEMS Microbiology Ecology*, 62(2), 142-160.
- Reay, S. D., & Norton, D. A. (1999). Assessing the success of restoration plantings in a temperate New Zealand forest. *Restoration Ecology*, 7(3), 298-308.
- Relph, E. (1985). Geographical experiences and being-in-the-world: The phenomenological origins of geography. In D. Seamon & R. Mugerauer (Eds.), *Dwelling, place and environment: Towards a phenomenology of person and world* (pp. 15-31). Dordrecht: Martinus Nijhoff Publishers.
- Relph, E. (2009). A pragmatic sense of place. *Environmental and Architectural Phenomenology*, 20(3), 24-31.
- Robertson, H. A. (2013). *Tui*. New Zealand Birds Online. Retrieved 23.07., 2016, from <http://nzbirdsonline.org.nz/species/tui>.

- Robertson, H. A., Baird, K., Dowding, J. E., Elliott, G. P., Hitchmough, R. A., Miskelly, C., McArthur, N., O'Donnell, C. F. J., Sagar, P. M., Scofield, R. P., & Taylor, G. A. (2017). *Conservation status of New Zealand birds, 2016*. Wellington, New Zealand: Department of Conservation.
- Robertson, H. A., & Hackwell, K. R. (1995). Habitat preferences of birds in seral kahikatea *Dacrycarpus dacrydioides* (Podocarpaceae) forest of South Westland, New Zealand. *Biological Conservation*, 71(3), 275-280.
- Rodewald, A. D., Kearns, L. J., & Shustack, D. P. (2011). Anthropogenic resource subsidies decouple predator–prey relationships. *Ecological Applications*, 21(3), 936-943.
- Rosenzweig, M. L. (2003). Reconciliation ecology and the future of species diversity. *Oryx*, 37(02), 194-205.
- Ruffell, J., & Didham, R. K. (2017). Conserving biodiversity in New Zealand's lowland landscapes: does forest cover or pest control have a greater effect on native birds? *New Zealand Journal of Ecology*, 41(1), 23-33.
- Ruffell, J., Innes, J., Bishop, C., Landers, T., Khin, J., & Didham, R. K. (2015a). Using pest monitoring data to inform the location and intensity of invasive-species control in New Zealand. *Biological Conservation*, 191, 640-649.
- Ruffell, J., Innes, J., & Didham, R. K. (2015b). Efficacy of chew-track-card indices of rat and possum abundance across widely varying pest densities. *New Zealand Journal of Ecology*, 39(1), 87.
- Ruiz-Jaen, M. C., & Mitchell Aide, T. (2005). Restoration success: How is it being measured? *Restoration Ecology*, 13(3), 569-577.
- Ruiz, J., Fandiño, M. C., & Chazdon, R. L. (2005). Vegetation Structure, Composition, and Species Richness Across a 56-year Chronosequence of Dry Tropical Forest on Providencia Island, Colombia 1. *Biotropica: The Journal of Biology and Conservation*, 37(4), 520-530.
- Ruscoe, W. A., Ramsey, D. S., Pech, R. P., Sweetapple, P. J., Yockney, I., Barron, M. C., Perry, M., Nugent, G., Carran, R., & Warne, R. (2011). Unexpected consequences of control: Competitive vs. predator release in a four-species assemblage of invasive mammals. *Ecology Letters*, 14(10), 1035-1042.
- Russell, J. C., & Stanley, M. C. (2018). An overview of introduced predator management in inhabited landscapes. *Pacific Conservation Biology*, 24(4), 371-378.
- Ryan, R. L. (2000). A people-centered approach to designing and managing restoration projects: Insights from understanding attachment to urban natural areas. In P. H. Gobster & R. B. Hull (Eds.), *Restoring nature: Perspectives from the social sciences and humanities* (pp. 209-228). Washington, D.C.: Island Press.

- Ryan, R. L. (2005). Exploring the effects of environmental experience on attachment to urban natural areas. *Environment and Behavior*, 37(1), 3-42.
- Savard, J.-P. L., Clergeau, P., & Mennechez, G. (2000). Biodiversity concepts and urban ecosystems. *Landscape and urban planning*, 48(3), 131-142.
- Schwartz, M. W., Jurjavcic, N. L., & O'brien, J. M. (2002). Conservation's disenfranchised urban poor. *BioScience*, 52(7), 601-606.
- Seamon, D. (1985). Reconciling old and new worlds: The dwelling-journey relationship as portrayed in Vilhelm Moberg's 'Emigrant' novels. In D. Seamon & R. Mugerauer (Eds.), *Dwelling, place and environment: Towards a phenomenology of person and world* (pp. 227-245). Dordrecht, Netherlands: Martinus Nijhoff Publishers.
- Seamon, D. (2000). A way of seeing people and place: Phenomenology in environment-behaviour research. In S. Wapner, J. Demick, T. Yamamoto & H. Minami (Eds.), *Theoretical perspectives in environment-behavior research: Underlying assumptions, research problems, and methodologies* (pp. 157-178). Dordrecht, Netherlands: Kluwer Academic Publishers.
- Seamon, D. (2014). Place attachment and phenomenology. In L. C. Manzo & P. Devine-Wright (Eds.), *Place attachment: Advances in theory, methods and applications* (pp. 12-22). Abingdon, Oxfordshire: Routledge.
- Seamon, D., & Mugerauer, R. (1985). *Dwelling, place and environment: Towards a phenomenology of person and world*. Dordrecht, Netherlands: Martinus Nijhoff Publishers.
- Seamon, D., & Sowers, J. (2008). Place and placelessness (1976): Edward Relph. In P. Hubbard, R. Kitchin & G. Valentine (Eds.), *Key texts in human geography* (pp. 43-51). London: Sage.
- Selge, S., Fischer, A., & van der Wal, R. (2011). Public and professional views on invasive non-native species—a qualitative social scientific investigation. *Biological Conservation*, 144(12), 3089-3097.
- Selwood, K., Mac Nally, R., & Thomson, J. R. (2009). Native bird breeding in a chronosequence of revegetated sites. *Oecologia*, 159(2), 435-446.
- Shanahan, D., Ledington, J., & Maseyk, F. (2018). Motivations for conservation action in peopled landscapes. *Pacific Conservation Biology*, 24(4), 341-348.
- Shiels, A. B., Medeiros, A. C., & Allmen, E. I. (2017). Shifts in an invasive rodent community favoring Black rats (*Rattus rattus*) following restoration of native forest. *Restoration Ecology*, 25(5), 759-767.
- Shochat, E., Warren, P. S., Faeth, S. H., McIntyre, N. E., & Hope, D. (2006). From patterns to emerging processes in mechanistic urban ecology. *Trends in Ecology & Evolution*, 21(4), 186-191.

- Shwartz, A., Muratet, A., Simon, L., & Julliard, R. (2013). Local and management variables outweigh landscape effects in enhancing the diversity of different taxa in a big metropolis. *Biological Conservation*, 157, 285-292.
- Shwartz, A., Turbé, A., Julliard, R., Simon, L., & Prévot, A.-C. (2014a). Outstanding challenges for urban conservation research and action. *Global Environmental Change*, 28, 39-49.
- Shwartz, A., Turbé, A., Simon, L., & Julliard, R. (2014b). Enhancing urban biodiversity and its influence on city-dwellers: An experiment. *Biological Conservation*, 171, 82-90.
- Sibley, D. (1999). Creating geographies of difference. In D. B. Massey, J. Allen & P. Sarre (Eds.), *Human geography today* (pp. 115-128). Cambridge: Polity Press.
- Smith, J. A., Flowers, P., & Larkin, M. (2009). *Interpretative phenomenological analysis: Theory, method and research*. London: Sage Publications Ltd.
- Society for Ecological Restoration International Science & Policy Working Group. (2004). *The SER International Primer on Ecological Restoration*. Society for Ecological Restoration International, Tucson, AZ. https://www.ctahr.hawaii.edu/littonc/PDFs/682_SERPrimer.pdf.
- Soga, M., & Gaston, K. J. (2016). Extinction of experience: The loss of human–nature interactions. *Frontiers in Ecology and the Environment*, 14(2), 94-101.
- Spurr, E. B. (2012). New Zealand garden bird survey: Analysis of the first four years. *New Zealand Journal of Ecology*, 36(3), 1.
- Stagoll, K., Manning, A. D., Knight, E., Fischer, J., & Lindenmayer, D. B. (2010). Using bird–habitat relationships to inform urban planning. *Landscape and Urban Planning*, 98(1), 13-25.
- Standish, R. J., Hobbs, R. J., & Miller, J. R. (2013). Improving city life: Options for ecological restoration in urban landscapes and how these might influence interactions between people and nature. *Landscape Ecology*, 28(6), 1213-1221.
- Statistics New Zealand. (2013). *Ethnic group (total responses), for the census usually resident population count, 2001, 2006, and 2013 Censuses (RC, TA, AU)*. <http://nzdotstat.stats.govt.nz/wbos/Index.aspx?DataSetCode=TABLECODE8020>.
- Statistics New Zealand. (2017). *Subnational population estimates: At 30 June 2017 (provisional)*. Retrieved 06.02., 2019, from <https://www.stats.govt.nz/information-releases/subnational-population-estimates-at-30-june-2017-provisional>.
- Stevens, H. (2006). *Native birds in forest remnants*. Unpublished doctoral thesis thesis, University of Auckland, Auckland, New Zealand.

- Sutherland, W. J., Newton, I., & Green, R. (2004). *Bird ecology and conservation: A handbook of techniques*. (Vol. 1). London: Oxford University Press.
- Sweetapple, P., & Nugent, G. (2011). Chew-track-cards: A multiple-species small mammal detection device. *New Zealand Journal of Ecology*, 35(2), 153-162.
- Tanner, C. J., Adler, F. R., Grimm, N. B., Groffman, P. M., Levin, S. A., Munshi-South, J., Pataki, D. E., Pavao-Zuckerman, M., & Wilson, W. G. (2014). Urban ecology: Advancing science and society. *Frontiers in Ecology and the Environment*, 12(10), 574-581.
- Taylor, A. F., Wiley, A., Kuo, F. E., & Sullivan, W. C. (1998). Growing up in the inner city: Green spaces as places to grow. *Environment and Behavior*, 30(1), 3-27.
- Thayer, R. L. (1989). The experience of sustainable landscapes. *Landscape Journal*, 8(2), 101-110.
- The World Bank. (2018). *Urban population (% of total population)*. Retrieved 03.05., 2019, from <https://data.worldbank.org/indicator/sp.urb.totl.in.zs>.
- Threlfall, C. G., Williams, N. S., Hahs, A. K., & Livesley, S. J. (2016). Approaches to urban vegetation management and the impacts on urban bird and bat assemblages. *Landscape and Urban Planning*, 153, 28-39.
- Toft, R. J., Ford, D. E., Sullivan, J. J., & Stewart, G. H. (2019). Invertebrates of an urban old growth forest are different from forest restoration and garden communities. *New Zealand Journal of Ecology*, 43(1), 0-0.
- Tompkins, D. M., & Veltman, C. J. (2006). Unexpected consequences of vertebrate pest control: Predictions from a four-species community model. *Ecological Applications*, 16(3), 1050-1061.
- Tonietto, R. K., Ascher, J. S., & Larkin, D. J. (2017). Bee communities along a prairie restoration chronosequence: Similar abundance and diversity, distinct composition. *Ecological Applications*, 27(3), 705-717.
- Tuan, Y.-F. (1986). *The Good Life*. Madison, WI: University of Wisconsin Press.
- Turner, W. R., Nakamura, T., & Dinetti, M. (2004). Global urbanization and the separation of humans from nature. *Bioscience*, 54(6), 585-590.
- Twigger-Ross, C. L., & Uzzell, D. L. (1996). Place and identity processes. *Journal of Environmental Psychology*, 16(3), 205-220.
- United Nations. (2018). *World Urbanization Prospects: The 2018 Revision, Online Edition*. <https://esa.un.org/unpd/wup/Publications>.
- Vale, T. R., & Vale, G. R. (1976). Suburban bird populations in west-central California. *Journal of Biogeography*, 3, 157-165.

- van Heezik, Y., Dickinson, K. J., & Freeman, C. (2012). Closing the gap: Communicating to change gardening practices in support of native biodiversity in urban private gardens. *Ecology and Society*, 17(1), 34.
- van Heezik, Y., Ludwig, K., Whitwell, S., & McLean, I. G. (2008a). Nest survival of birds in an urban environment in New Zealand. *New Zealand Journal of Ecology*, 32(2), 155-165.
- van Heezik, Y., & Seddon, P. J. (2018). Animal reintroductions in peopled landscapes: Moving towards urban-based species restorations in New Zealand. *Pacific Conservation Biology*, 24(4), 349-359.
- van Heezik, Y., Smyth, A., Adams, A., & Gordon, J. (2010). Do domestic cats impose an unsustainable harvest on urban bird populations? *Biological Conservation*, 143(1), 121-130.
- van Heezik, Y., Smyth, A., & Mathieu, R. (2008b). Diversity of native and exotic birds across an urban gradient in a New Zealand city. *Landscape and Urban Planning*, 87(3), 223-232.
- Van Vianen, J., Burge, O. R., MacFarlane, A. T., & Kelly, D. (2018). The effects of single aerial 1080 possum-control operations on common forest birds in the South Island, New Zealand. *New Zealand Journal of Ecology*, 42(2), 0-0.
- Vesk, P. A., Nolan, R., Thomson, J. R., Dorrough, J. W., & Mac Nally, R. (2008). Time lags in provision of habitat resources through revegetation. *Biological Conservation*, 141(1), 174-186.
- Vitousek, P. M., Mooney, H. A., Lubchenco, J., & Melillo, J. M. (1997). Human domination of Earth's ecosystems. *Science*, 277(5325), 494-499.
- Walbridge, M. (1997). Urban ecosystems. *Urban ecosystems*, 1(1), 1-2.
- Walker, L. R., Wardle, D. A., Bardgett, R. D., & Clarkson, B. D. (2010). The use of chronosequences in studies of ecological succession and soil development. *Journal of Ecology*, 98(4), 725-736.
- Walker, S., & Monks, A. (2017). *New Zealand's native land birds: Status and change on the mainland from estimates of occupancy for 1969-1979 and 1999-2004*. Landcare Research. <https://www.pce.parliament.nz/media/1706/new-zealand-s-native-land-birds-status-and-change-on-the-mainland-from-estimates-of-occupancy-for-1969-1979-and-1999-2004.pdf>.
- Walker, S., Monks, A., & Innes, J. (2017). *Status and change in native forest birds on New Zealand's mainland, 1969-1979 to 1999-2004*. Landcare Research. <https://catalogue.data.govt.nz/dataset/new-zealand-native-land-birds/resource/237da3f2-6b01-44f8-9571-74a95b64cb67>.

- Walker, S., Wilson, D. J., Norbury, G., Monks, A., & Tanentzap, A. J. (2014). Effects of secondary shrublands on bird, lizard and invertebrate faunas in a dryland landscape. *New Zealand Journal of Ecology*, 38(2), 242-256.
- Wallace, K. J. (2017). *Urban restoration ecology: Investigating environmental change, ecological function, and enrichment planting*. Unpublished doctoral thesis, University of Waikato, Hamilton, New Zealand.
- Wallace, K. J., Laughlin, D. C., & Clarkson, B. D. (2017). Exotic weeds and fluctuating microclimate can constrain native plant regeneration in urban forest restoration. *Ecological Applications*, 27(4), 1268-1279.
- Walrond, C. (2008). *Acclimatisation - Early acclimatisation societies*. Te Ara - the Encyclopedia of New Zealand. Retrieved 08.10., 2019, from <https://teara.govt.nz/en/acclimatisation/page-2>.
- Whytock, R. C., Fuentes-Montemayor, E., Watts, K., Barbosa De Andrade, P., Whytock, R. T., French, P., Macgregor, N. A., & Park, K. J. (2018). Bird-community responses to habitat creation in a long-term, large-scale natural experiment. *Conservation Biology*, 32(2), 345-354.
- Wiens, J. A. (1995). Habitat fragmentation: Island v landscape perspectives on bird conservation. *Ibis*, 137(s1), S97-S104.
- Wilson, D. J., Norbury, G., & Walker, S. (2014). How does woody succession affect population densities of passerine birds in New Zealand drylands? *New Zealand Journal of Ecology*, 38(2), 257-267.
- Wilson, D. J., Ruscoe, W. A., Burrows, L. E., McElrea, L. M., & Choquenot, D. (2006). An experimental study of the impacts of understorey forest vegetation and herbivory by red deer and rodents on seedling establishment and species composition in Waitutu Forest, New Zealand. *New Zealand Journal of Ecology*, 30(2), 191-207.
- Wood, V., Seddon, P. J., Beaven, B., & van Heezik, Y. (2016). Movement and diet of domestic cats on Stewart Island/Rakiura, New Zealand. *New Zealand Journal of Ecology*, 40(1), 186-190.
- Yilmaz, K. (2013). Comparison of quantitative and qualitative research traditions: Epistemological, theoretical, and methodological differences. *European Journal of Education*, 48(2), 311-325.
- Young, T. P. (2000). Restoration ecology and conservation biology. *Biological Conservation*, 92(1), 73-83.